

**Maximising energy recovery from the brewery wastewater
treatment system: A case study evaluating the anaerobic
digestion wastewater treatment plant at SAB's Newlands
Brewery**

A dissertation submitted for the degree of
Master of Science in Chemical Engineering

By

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Abstract

As environmental sustainability and sustainable development continue to dominate socio-economic discourse, many organisations are working on strategies to mitigate environmental burden. Replacement of non-renewable or fossil energy sources and efficient resource use are central to sustainable development. In particular, the transformation of organic waste matter into value-added products can contribute to efficient use of virgin material. For example, SABMiller Breweries makes use of their on-site anaerobic wastewater treatment plant (SABWTP) to reduce the organic content of brewery wastewater. The SABWTP produces biogas, used to fuel a boiler for production of steam. The steam is used as a heating utility in the brewing process. This strategy has a dual-purpose as a bioremediation and non-fossil based energy production process.

This study has been encouraged by the successful recovery of useful energy from brewery wastewater using anaerobic digestion technology. It aims to evaluate the environmental benefits or burden of improving energy production by using organic brewery by-products as additional feedstock into the SABWTP. An environmental impact assessment on the SABWTP and its associated process was carried out using life cycle assessment (LCA) tools. Anaerobic digestibility of the two major organic brewery by-products, brewer's spent grain and brewer's spent yeast, was evaluated experimentally using laboratory bench scale reactors. The results were used to postulate the feasibility of adding these feedstocks into the SABWTP. Based on these findings, three viable processing scenarios were synthesised and assessed in terms of environmental impact analysis. In the environmental impact analysis, the three scenarios were compared using average process conditions and the main contributing factors to environmental burdens associated with each scenario were identified.

The results from anaerobic digestion of spent grain showed inadequate transformation of the organic matter into biogas. The resistance of the spent grain to digestion without pre-treatment was attributed to its high composition of lignocellulosic matter. In an attempt to improve digestibility, the spent grain was processed in a food blender and classified with respect to particle size. Comparison between digestion of small grain particles and larger grain particles showed that there were no differences in organic matter reduction and biogas production. It was thus concluded that spent grain was not a suitable additional feedstock to the SABWTP without additional pre-treatments such as those used for large-scale processing of lignocellulosic fractions. Anaerobic digestion of spent yeast was most promising with conversions above 88% and digestion rates comparable to those of brewery wastewater. The digestion profiles of the intact yeast cell followed first order kinetics while digestion of soluble organics derived from the spent yeast cells and spent yeast liquor followed Monod kinetics. However, a first order kinetic model could also be fitted to the soluble organic conversion process. A diauxic profile for conversion of the spent yeast solids was observed. This suggested that the spent yeast solids were

comprised of favourable and readily biodegradable matter characterised by the first portion of the digestion profile and slowly biodegradable matter characterised by slow degradation at the tail portion of the profile.

A plot of degradation rate against substrate concentration showed a higher increase in degradation rate of soluble organic matter with substrate concentration as compared to degradation rate of the yeast solids. This observation suggested degradation of soluble organic matter was faster than degradation of the solid organic matter for the most part of the tested conditions. Filtration of spent yeast cells from spent yeast liquor and its subsequent feeding into the SABWTP could provide the following benefits:

- Increased biogas productivity resulting from the increased organic loading into the SABWTP,
- Reduced transport cost resulting from the reduced volume of spent yeast to be transported off site.

The following three processing scenarios were modelled and compared with respect to their environmental profile:

- Option 1: Current situation with entire spent yeast stream transported off site
- Option 2: Entire spent yeast stream fed to the SABWTP
- Option 3: Spent yeast filtered, spent yeast liquor (filtrate) fed to SABWTP and concentrate of spent yeast cells transported off site.

Practical constraints of feeding the spent yeast with the wastewater to the SABWTP were considered. Analysis of historical data (dated between January 2010 and January 2013) showed that 2030 kL/day of 4230 kL/day total volumetric capacity and about 13 tonnes-COD of 18 tonnes-COD total organic capacity remained unused. The ratio of brewery wastewater and spent yeast suspension is small (about 0.7%) on volume basis. As such, the hydraulic function of the wastewater treatment plant would not be significantly affected by the addition of spent yeast. Further, mixed feed of influent wastewater and the spent yeast does not exceed the design capacity of the SABWTP as the spent yeast contributes at most 35% to the total COD of the mixed feed.

When modeling the performance of the SABWTP during feeding of spent yeast or spent yeast liquor, the following considerations were taken into account:

- The SABWTP is fitted with an upflow sludge blanket reactor (UASBR) in which anaerobic digestion takes place. The flow dynamics of the UASBR are such that minimal escape of solids occurs and that reactions take place mainly in the sludge bed. However, the effect of feeding the spent yeast on microbial granulation was not considered.
- The sludge bed is characterised by continuous stirred tank reactor (CSTR) flow patterns. Hence, all reactions are modelled as taking place in a CSTR.

The model results showed an increase in the COD reduction rate for Option 2 as compared to Option 1 and Option 3 due to increased COD concentration in the reactor, indicating increased methane productivity. However, the effluent concentration for Option 2 was higher than that of Option 1 and Option 3, suggesting reduced effluent treatment efficiency. Therefore, increasing influent concentrations or OLR, which results in increased reactor concentration of organic matter, increases the methane productivity and therefore the energy output of the wastewater treatment system, but reduces the quality of treated effluent, thereby reducing the wastewater treatment efficiency of the system.

An environmental impact assessment was carried out using the LCA framework on the three processing scenarios (Option 1, Option 2 and Option 3) considering ten impact categories (*abiotic depletion, acidification, eutrophication, global warming potential, ozone layer depletion, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity and photochemical oxidation*). The scenarios were analysed in terms of two units for comparison: **kg-COD** fed to SABWTP and **L-beer** produced by the brewery. The **kg-COD** unit provided insights into the relative efficiency in the SABWTP, whereas the **L-beer** unit demonstrated the absolute change in the environmental performance of the wastewater management system. Using Option 1 as a base case, the analysis of inventories showed about 10% increase in energy recovery efficiency for Option 2, and about 7% decrease in energy recovery efficiency for Option 3, whereas, the net energy produced increased by more than 45% and about 6% for Option 2 and Option 3 respectively.

Environmental impact assessment results showed environmental relief for all impact categories other than *eutrophication*. The relief was due to net output of energy replacing burdens associated with electricity production, whereas the *eutrophication* burden was due to disposal of nitrogen (N) and phosphorus (P). Option 1 and Option 3 were about 90% and 80% less environmentally efficient (relative scores per kg-COD), respectively, and both about 50% less relief (scores per L-beer) than Option 2 for all impact categories other than *eutrophication, global warming potential and photochemical oxidation*. Option 1 and Option 3 were about 80% and 60% less environmentally efficient than Option 2 for *global warming*, respectively, and 85% and 65% for *photochemical oxidation*. The relative environmental relief for Option 1 and Option 2 per L-beer were between 40% and 50% for both *global warming potential* and *photochemical oxidation* compared to Option 2. For *eutrophication*, the total burdens (per L-beer) were about 40% and 50% for Option 1 and Option 3 relative to Option 2. Therefore, increasing the OLR by introducing a stream rich in organic matter (COD), nitrogen (N) and phosphorus (P), such as spent yeast, into the wastewater treatment system improves environmental performance for all considered impact categories other than *eutrophication*.

Spent yeast is currently used as a protein or nitrogen source in animal feed. The effect of re-purposing the spent yeast for use in anaerobic digestion and replacing with another nitrogen source on the environmental impact results of the wastewater management system was evaluated. Highest

Abstract

environmental impact scores across common sources of nitrogen were used to represent overestimates. The amount of alternative nitrogen source required to deliver the same functionality also depended on the relative nitrogen retention efficiency associated with it (i.e. amount of nitrogen from the alternative source retained by the animal's body divided the nitrogen from spent yeast retained by the animal (ξ)). No effect was observed for the ξ value greater than 0.1. This suggested that the burden of replacing spent yeast for animal feed had negligible effect even when the replacer product is ten times less efficient at delivering nitrogen than spent yeast. As such, spent yeast provides greater environmental benefits when is used as additional feed to anaerobic digestion that when used as a nitrogen source for animals.

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Nomenclature

AD	Anaerobic digestion
AWTW	Athlone wastewater treatment works
BSG	Brewer's spent grain
BSY	Brewer's spent yeast
$C_{i,j}$	Concentration of component i in stream j
COD	Chemical oxygen demand
CP	Crude protein
CSTR	Continuous stirred tank reactor
DWAF	Department of Water Affairs and Forestry
E_i	Energy defined according to i
GLSS	Gas-liquid-solids separations
HRT	Hydraulic retention time
LCA	Life Cycle Analysis (or Assessment)
LCIA	Life Cycle Impact Analysis (or Assessment)
LCI	Life Cycle Inventory
LCFA	Long chain fatty acids
N	Nitrogen
$M_{i,j}$	Mass or mass flowrate of component i in stream j
OLR	Organic loading rate
P	Phosphorus
PFR	Plug-flow reactor
Q_j	Volumetric flowrate of stream j
SABWTP	SABMiller Wastewater Treatment Plant

SCOD	Soluble chemical oxygen demand
SRT	Solids retention time
SSD	Strategic sustainable development
TCOD	Total chemical oxygen demand
TDT	Total distance travelled
TP	Total phosphorus
TS	Total solids
UASBR	Upflow anaerobic sludge blanket reactor
VFA	Volatile fatty acids
Y_{ij}	Yield of component i from component j
ξ_i	Efficiency defined according to i

1 Introduction

1.1 Bioenergy and biomass for sustainability

Issues of climate change, fossil depletion and environmental responsiveness are becoming more and more a part of everyday conversations and political discourse. As a result, many of the companies and research institutions are working on ways to replace non-renewable energy sources and minimisation of waste and environmental burden. Renewable energy forms an integral part of sustainable development. The abundance of biomass, and our ability to produce it, makes biofuels and bioenergy important renewable energy sources of the future (Chang et al. 2010). Other major contributors to renewable energy that have reached application stage include hydropower, geothermal, wind, solar and marine energy (Buekens 2005).

Biomass as resource for bioenergy production can be sourced from cultivation of energy crops, from existing forest residues, and from organic wastes (Amin 2009). While, the bioenergy space is among the most active fields in alternative energy, its sustainability depends largely on the type of biological feedstock. Biomass that requires special production channels are less attractive than those that are more readily available. On the other hand, industries have been focused on reducing their waste disposal costs (Fillaudeau et al. 2006). Recovery of energy from organic wastes is thus regarded as one of the most sustainable strategies. This approach addresses the energy problem and waste management issues jointly, and is often associated with energy savings and reduction of environmental burden. The dual purposing of anaerobic treatment of organic waste for both remedial purposes and energy generation is one such example. As part of their sustainability goals and environmental responsiveness, many companies are implementing such sustainable strategies. To ensure that prioritisation of energy production does not compromise the primary goal of waste (water) treatment, there is a need model and understand the impact of prospective strategies on the environment. Life cycle assessment (LCA) is an effective and useful tool for quantifying the environmental impacts associated with products and services (United Nations Environment Programme 2011). LCA can be used to compare between products or processes in a holistic and fair manner, and as a tool to monitor or predict the effect of change or improvement of a process on the environment.

1.2 Renewable energy in the brewing context

The increasing environmental consciousness and stringent waste (water) disposal regulations have been a catalyst for the beer industry to implement strategies for improving the efficiency of the brewing process and minimise waste. In particular, effluent water discharge tariffs have encouraged brewing companies to install and run their own wastewater treatment facilities. Over the last decade, the South

African Breweries (SAB) has installed anaerobic wastewater treatment plants on several of their breweries across South Africa. One of the first of SAB anaerobic wastewater treatment plants (SABWTP) was commissioned in response to discharge tariff hikes in 2004 at their Newlands Brewery in Cape Town. The plant was designed to treat up to 18000 kg-COD day⁻¹. The effluent wastewater from this plant is pumped to the local municipal waste treatment facilities for further treatment. Initially, the biogas produced by the SABWTP was flared to the atmosphere. Cohen (2006) completed a life cycle assessment case study aimed at quantifying the environmental benefits of adding the on-site effluent treatment system. The results showed that the addition of the SABWTP was environmentally beneficial, and the integration of biogas as additional energy source for use in the brewery was recommended. The brewery has since materialised an opportunity to make use of the biogas from the SABWTP as fuel for production of steam. Currently, the steam generated from this strategy accounts for around 10% to 12% of the total steam required in the brewing process with the balance generated from electricity. The SABWTP is not operated at its full capacity, as shown by Figure 1.1, due to low content of organic matter in the brewery wastewater. This presented an opportunity to intensifying the biogas production, and therefore energy recovery, by increasing organic loading into the plant. However, the effect of this strategy on the environment should also be modelled to aid the decision making.

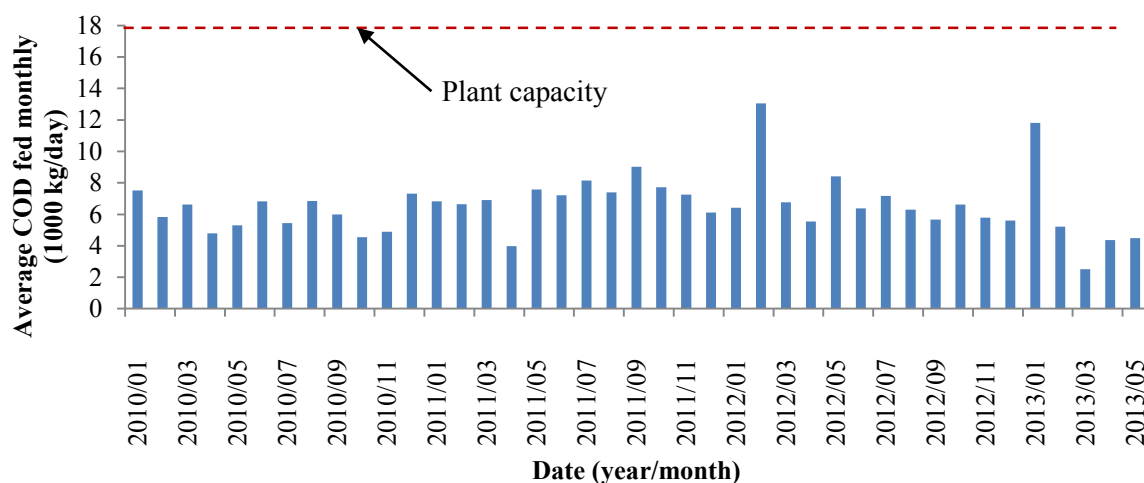


Figure 1.1: Average monthly COD fed to the SABWTP.

1.3 The purpose and scope of the project

This project has sought to evaluate the viability of intensifying the biogas output of the SABWTP by increasing the organic loading of the digester. This was done by evaluating the feasibility of using biomass brewer's spent grain (BSG) and brewer's spent yeast (BSY) as candidate additional feedstock to the SABWTP, and analysing the environmental benefits or burdens of instigating this change. The selection of BSG and BSY as candidate feedstock was encouraged by the ready availability these

resources. While the project has been positioned in the context of the SABWTP at Newlands Brewery, Cape Town, the project was formulated such that its findings have applicability to a broader range of applications of the anaerobic digestion of solid waste streams from the brewery sector, including application to other breweries and the use of these waste streams to provide resources into related processes.

1.4 Thesis structure

This thesis is separated in to several chapters. Chapter 1 presents an introduction to the thesis. The introduction includes the context and rationale of the study as well as the structure of the thesis. Chapter 2 presents a review of relevant literature used to understand the details and motivation of the work done. These include sustainable development and environmental sustainability as goals to which the systems investigated aim to contribute, tools and frameworks of monitoring and designing environmentally sustainable systems, and anaerobic digestion as a key technology investigated in the current study. Chapter 3 presents a detailed description of the Materials and Methods used during experimental studies on anaerobic digestion of BSG and BSY. In Chapter 4 results observed during the experimental studies are presented and a discussion of the reasons for the observed results given. The development of a mathematical model for describing the results based on the experiments and review of published literature is provided. Chapter 5 focuses on the implication of the experimental results on the operation of the SABWTP. The chapter presents a description the current operating conditions of the SABWTP followed by the prospective change in operation caused by the introduction of the additional feed, based on the mathematical model developed in Chapter 4. Chapter 6 presents all the work pertinent to analysing the environmental impact instigated by the prospective improvement strategies. The chapter includes description of the goal and scope of the environmental impact study, the boundary system, a description of all pertinent materials and energy flows for each subsystem included in the system boundary, analysis of materials and energy inventories of each subsystem and the assessment of impact for the prospective strategies. This is followed by a discussion on further improvement strategies. Chapter 7 presents conclusions drawn from this thesis and the recommendations for further action.

2 Literature Review

2.1 Introduction

The current project investigated the effect of intensifying energy production in combination with waste(water) treatment using anaerobic digestion through conducting a case study on the SABWTP through assessing the environmental performance of brewery wastewater management systems. While the economic gains of the proposed strategy are apparent, the environmental implications require further scrutiny. The concept of sustainable development and frameworks used in assessing environmental sustainability are thus important to understand. This chapter presents an organisation of literature relevant to understanding the basis and motivation for the study. The chapter starts with a section on sustainable development and industrial ecology (Section 2.2). This section presents a review of literature pertinent to understanding the concept and issues related to sustainable development as a goal towards which the study aims to contribute, and industrial ecology as a framework for achieving this goal. Section 2.3 describes life cycle analysis (LCA) with a particular focus on environmental impact analysis of the process or processes with regards to its relevance in the study. Some practical advantages and limitations of LCA are also presented. Section 2.5 presents a literature review on anaerobic digestion aimed at understanding the process and providing the basis for the formulated hypotheses. Section 2.6 draws together and integrates relevant knowledge based on the literature review presented to provide a foundation for the work done in this thesis. Section 2.7 presents the problem statement and objectives, as well as the statement of hypotheses and key questions used to guide the research.

2.2 Sustainable development and industrial ecology

2.2.1 Sustainable development

Sustainable development as a concept has become one of the most important elements of policy in guiding governments, companies and agencies (Mebratu 1998). The inception and acceptance of this concept sprung from concerns around depletion of resources and degradation of the environment caused by anthropogenic trends in resource consumption and waste generation. With the increased discourse around sustainable development, variations in the interpretation of the concept have emerged. However, many agree that sustainable development focuses on ecological integrity, inter-generational and inter-societal human health, balanced environmental and resource equity (Goodland & Daly 1996; Garner & Keoleian 1995; Korhonen 2004).

Many manufacturing companies, such as breweries, are concerned with environmental sustainability associated with their processes. Goodland & Daly (1996) prescribed the basic conditions for attaining

environmental sustainability in terms of the inputs and outputs of a project or process. These conditions were described as follows:

- **Inputs:** Harvesting rates of renewable resource should be kept within regenerative capacity of the environment, and rate of non-renewable resource depletion should not exceed the rate of development of their renewable substitutes.
- **Outputs:** Waste emissions should be kept within the assimilative capacity of the local environment without compromising its future capacity to absorb waste or provide other essential services.

Robèrt et al. (1997) highlighted the need for guidelines to achieve progress towards sustainability in response to the increasing complexity and effects of environmental problems. Since various tools for monitoring sustainable development have emerged, Robèrt (2000) highlighted contradictions between the application and interpretation of various methods. He described a strategic sustainable development (SSD) framework within which all tools should be applicable and contribute to the common goal of sustainability. The SSD framework consists of five levels or stages (Robèrt 2000; Robèrt et al. 2002, cited by Korhonen 2004).

- *Level 1:* Principles that define the system being studied are identified and described. The global ecosystem is the main system, with economic systems and social systems as subsystems.
- *Level 2:* The desired state of sustainability is defined i.e. the goal of the sustainable development.
- *Level 3:* The process to achieve a successful outcome is identified in line with some process principles. Back-casting as a principle allows for one to start at some future point of sustainability (defined in level 2) where the vision is defined, then walk back to the present while identifying or defining the process or path that will lead to that desired point. Flexible platforms are principles in which investments are made in the present with consideration for their potential to not only solve current problems but also form stepping stones toward the future vision.
- *Level 4:* Practical actions are carried out.
- *Level 5:* Tools and matrices that audit and monitor success of the actions carried out in level 4 are employed.

2.2.2 Industrial ecology

In the context of industrial environmental sustainability, utilisation of the industrial ecology concept as a complementary framework to the SSD model offers several benefits (Korhonen 2004). Industrial ecology is the study of material and energy flows within and between components of the industrial system with the aim of reducing the overall environmental burden exerted by economic activity on the natural ecosystem system (Garner & Keoleian 1995; Erkman 1997; Lifset 1997; Korhonen & Snakin 2005) by integrating these components.

Industrial ecology views the industrial system as a metaphorical ecosystem (industrial ecosystem) thereby recognising the possibility of mutualistic interrelations within the system. Within the industrial ecosystem, waste material from one process can be used as raw material for other processes (Frosch & Gallopoulos 1989). The holistic approach of the concept allows for development of networks which interlink processes to minimise industrial waste streams that are absorbed into natural sinks (Korhonen 2004; Frosch & Gallopoulos 1989; Erkman 1997), as well as extraction of natural resources into the industrial system. The industrial ecosystem model offers more options for waste minimisation than linear models. Waste reduction strategies can be implemented within the processes and by linking processes. Further, flexible platforms may be created where more than one process is available to take in a certain type of waste from another process. In some cases, the industrial ecology approach may suggest that production of a particular waste product be intensified to allow it to be marketable to other processes (Erkman 1997).

Some authors have used industrial ecology synonymously with industrial metabolism, which is the transportation or transformation of materials and energy within the industrial system (Seager & Theis 2002). Others take the view that industrial ecology goes beyond the analysis of the flows within the industrial system (Socolow 2004; Erkman 1997; Frosch & Gallopoulos 1989). Erkman (1997) distinguished between industrial metabolism and industrial ecology, by describing industrial metabolism as study of material and energy flows in and out of the industrial ecosystem aimed at understanding the interaction of the industrial ecosystem with the natural ecosystem, whereas industrial ecology aims to use knowledge about those interactions in order to alter the industrial system to make it compatible with the functions of the natural ecosystem.

Frosch & Gallopoulos (1989) described the ideal industrial system as one where all materials are consumed within the industrial ecosystem and do not crossover into the natural ecosystem, i.e. a closed industrial system. However, Erkman (1997)'s description of industrial ecology suggests that an ideal industrial system may allow crossover of material into the natural ecosystem in a way that is compatible with the functions of the natural system. The latter ideal provides a realistic and realisable goal as movement of effluents from the industrial systems to the natural ecosystem is sometimes unavoidable. Where minimisation of waste transfer from the industrial system to the natural ecosystem is set as the

most important criterion, unsustainable process combinations may be selected where shifting of burden might occur. For example, the process of cleaning wastewater to the point of potability might have a more adverse environmental effect over its partial treatment to some acceptable standard.

Industrial ecology can provide other benefits apart from environmental sustainability. Figure 2.1 depicts the benefits of applying the industrial ecosystem concept as presented by Korhonen (2004). Korhonen (2004) placed the benefits of the industrial ecology concept in the context of the SSD framework by considering its contributions to each level in the SSD model. The focus of industrial ecology on tracking flows of materials and energy highlights the importance of thermodynamics, physical, economic and ecological considerations when defining a system in *level 1* of the SSD model. Further, it shows the dependency of the industrial system on the natural ecosystem. The industrial ecosystem metaphor is key to *level 2* by prescribing that the industrial system emulate the natural ecosystem since the natural ecosystem does not exceed the regenerative capacities of its resources and assimilative capacities of its waste streams. Korhonen (2004) suggests learning from nature in developing strategic guidelines toward sustainability in *level 3*. The contribution of industrial ecology to *level 4* arises from the use of the systems approach, which offers an inter-organisational perspective in the implementation of the sustainable strategies. The systems approach also supports the use of regional environmental monitoring systems as opposed to those employed by individual firms. This contributes to *level 5* (Korhonen 2004).

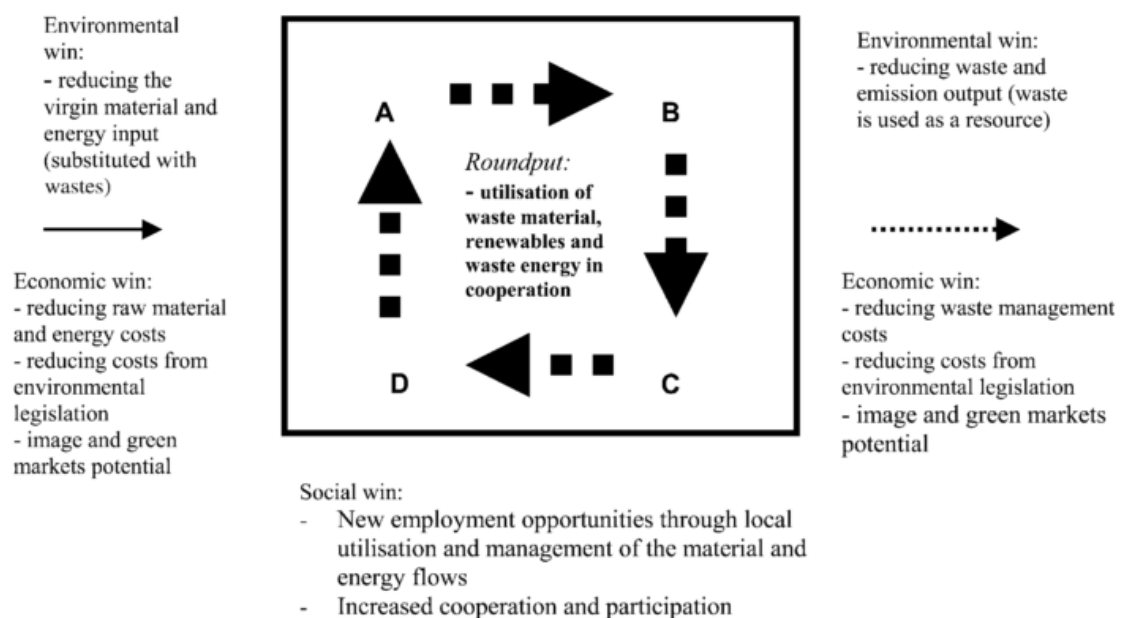


Figure 2.1: Environmental, economic and social ‘wins’ in the vision of a successful industrial ecosystem (Korhonen 2004).

2.3 Life cycle assessment

2.3.1 Introduction

Life cycle assessment (LCA) is a systematic framework used to assess the potential environmental impacts and resource footprint associated with a product's life cycle (Pehnt 2006; Finnveden et al. 2009; International Organisation for Standards 1997). This tool can be used to trace all the activity related to the product, from the extraction of the raw materials from the earth, their products through agriculture, manufacturing, packaging, use and disposal (the cradle-to-grave approach) (Cohen 2006; Pehnt 2006), and to quantify all the environmental burdens associated with the process by regarding the inputs and outputs of a system (Friedrich & Buckley 2002). These characteristics of the tool allow for a fair and holistic comparison of processing options. Further, the tool enables identification of major contributors to the environmental burdens within a system boundary, hence enabling targeting of specific areas for improvement. LCA is a useful analytical tool in industrial ecology due to its holistic approach as opposed to a minimalistic one (Seager & Theis 2002). Further, it provides useful matrices with which to measure sustainability in *level 5* of the strategic sustainable development model.

Authors distinguish between two types of LCAs, attributional LCA and consequential LCA. These are described with regards to the purpose of the LCA study (Brent et al. 2002; Pennington et al. 2004; Finnveden et al. 2009). In attributional LCA, the aim is to describe the environmental properties of a product system and its subsystems and to identify subsystems that contribute significantly to the impact of the product system on the environment. Consequential LCA aims to describe the environmental effects of changing conditions within a product system (Finnveden et al. 2009). While developments are made continuously, the International Standards Organization (ISO) provides the consensus for the LCA framework through the ISO 14000 series (Rebitzer et al. 2004). Life cycle assessments are carried out in four phases: goal and scope definition, inventory compilation, impact assessment and interpretation (International Organisation for Standards 1997).

2.3.2 Goal and scope definition

During this first stage, the goals of the life cycle assessment study are stated and the boundaries of the system are identified appropriately in line with the goals. The goal of the LCA must include an unambiguous statement of the intended application, the motivation for the study and the intended audience (International Organisation for Standards 1997). The definition of the scope includes descriptions of functions of the product stream and the functional unit, the product system boundaries, allocation procedures, impact categories and impact assessment tools, assumptions made, data requirements and data quality requirements (International Organisation for Standards 1997). From here on, the word 'product' shall refer to goods, services or processes.

Functional unit

Every product has a function for which it is primarily designed. A functional unit is defined in terms of the use of the product under study and it provides a reference upon which all activities in the system boundary will be defined (International Organisation for Standards 1997). The unit should enable fair comparisons between alternative goods or services (International Organisation for Standards 1997; Rebitzer et al. 2004) and should be defined in line with the goal of the LCA study. For example, a comparison between plastic bags and paper bags is based on carrying capacity (function which it performs), as opposed to the amount of material used (Rebitzer et al. 2004; Masters & Ela 2008).

System boundaries

The system boundary must include all activities associated with the product and its functions, and is defined in line with the goal of the study (International Organisation for Standards 1997). For example, if the goal is to analyze the impact of manufacturing a certain product, the boundary should encompass raw material extraction and acquisition up to the finished product excluding the use (cradle-to-gate). However, if the goal is to analyze the impact of using one product as compared to some alternative product, the boundary must include the usage of the product. The motivation of LCA study informs the type of required data (part of the scope), thus influences where the boundary of the system lies. The description of the targeted audience helps with understanding and describing the interests of the study more clearly. These examples demonstrate the influence of goal definition on the selection of the product system boundaries and the importance of clarity in defining and motivating the goal.

The influence of the type of LCA, attributional LCA and consequential LCA, on the system boundaries is also apparent in the aforementioned examples. The first example seeks to attribute environmental impacts of manufacturing a product, while the second example may seek to evaluate the consequence of using one product in place of another. In attributional LCA, energy and material flows are tracked systematically to upstream until the extraction of natural resources, and to downstream to the final disposal of waste, including processes that are deemed to be significant (Rebitzer et al. 2004). Consequential LCA includes processes that are expected to be affected by the changes proposed by the study (Rebitzer et al. 2004).

Life cycle inventory (LCI) compilation and analysis

Life cycle inventory (LCI) aims to quantify the material and energy inputs, and waste streams attributable to the product's life cycle (Rebitzer et al. 2004) with reference to the functional unit. At this stage, data collection and calculation procedures are described (International Organisation for Standards 1997) and the results are presented. Figure 2.2 depicts the product system boundary and life cycle inventory results. Analysis of the life cycle inventory seeks to interrogate the validity of the LCI and predict the impact of the various materials and energy flows presented in the LCI. By so doing, the quality of the data from which the inventories are derived may be improved and the mass and energy

models revised in line with some practical considerations. Sensitivity analyses are useful at this point to assess the potential effect of uncertainty in critical data on the observed outcome of LCA. Upon refining the inventory, the prediction of impacts caused by various material and energy flows may be discussed. This is not only useful to consider for impacts that may not be included or implicit in the selected impact categories, but also to consider the effect of change in operational behaviour on the LCA outcomes.

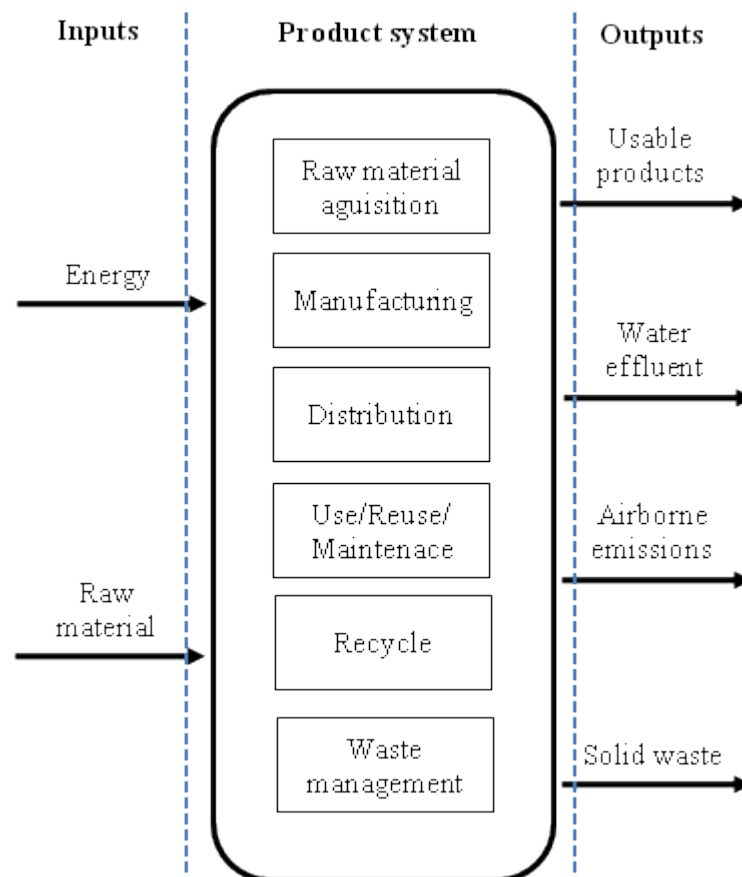


Figure 2.2: Depiction of product system boundary and life cycle inventory. Adapted from Masters & Ela (2008).

Life cycle impact analysis (LCIA)

The impact assessment phase of LCA associates or assigns the inventory data to specific environmental impact category to evaluate the significance of those potential impacts. The process includes the classification of inventory data to impact categories, modelling of inventory data within impact categories (characterisation) and weighing impact data into one environmental score where applicable (International Organisation for Standards 1997). The characterisation step identifies contributors to the impact from within the inventory data and quantifies the impact. Normalisation, weighting, ranking or grouping of the environmental impacts are optional (Pennington et al. 2004; Product Ecology Consultants 2010). Figure 2.3 depicts the flow of information in conducting a life cycle impact analysis.

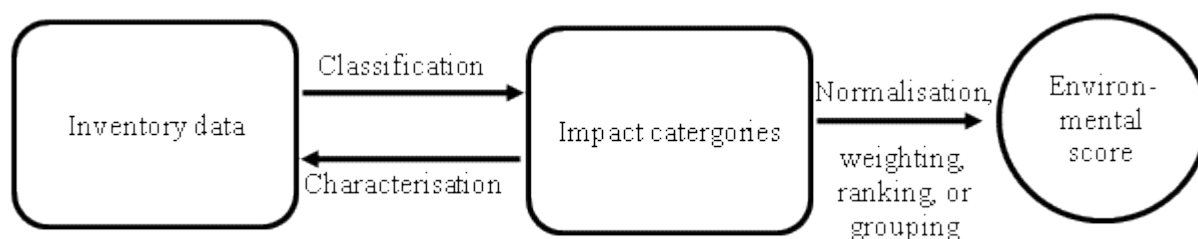


Figure 2.3: Life cycle impact analysis procedure.

Products incur environmental impacts due to consumption of resources and emission of substances into the natural ecology (Cohen, 2006). The choice and importance of impact categories and the level of detail depend on the goal of the study (International Organisation for Standards 1997). The environmental impacts recognised by ISO fall within the following categories: *ecotoxicity*, *eutrophication*, *global warming*, *acidification*, *abiotic depletion*, *stratospheric ozone depletion (ozone layer depletion)* and *tropospheric ozone creation or photochemical oxidation*.

Table 2.1: Description of impact categories, mechanisms by which they occur and references for standardisation of the impact. Compiled according to Cohen (2006) and referees.

Impact categories	Impact	Mechanisms
<i>Ecotoxicity:</i>	Toxicity to bio-organisms (Human and other terrestrial organisms, fresh water and marine organisms)	Inhalation, ingestion or absorption of toxic substances (airborne emissions or solid and liquid effluents) at harmful concentrations. Reference substance: 1,4 dichlorobenzene (1,4-DB) .
<i>Eutrophication:</i>	Loss of biodiversity on soils due to excessive growth or dominant species. Aquatic death caused by oxygen depletion which is associated with excessive algal growth.	Disposal of nutritious effluents on soil or waters. Reference substance: Phosphate (PO₄) .
<i>Global warming:</i>	Ice-cap melting causing sea levels to raise, regional climate changes, spreading of deserts, natural disasters.	Emissions and accumulation of greenhouse gases in the atmosphere. Carbon dioxide (CO₂) is the reference substance.
<i>Acidification:</i>	Soil acidity mobilises harmful ions which are subsequently absorbed by plants. Acidic	Oxidation of sulphur dioxide (SO ₂), nitrogen oxides (NO _x) and ammonia (NH ₃). Reference substance: Sulphur dioxide (SO₂) .

	run-offs harm aquatic ecosystems.	
Abiotic depletion:	Reduced resource base causes difficulty in acquiring those resources and equitable distribution ^a .	Reference substance: Lead (Pb) .
Consumption of non-renewable and renewable natural and non-living resources.		
Ozone layer depletion:	The ozone layer is responsible for preventing harmful wavelengths of ultraviolet light from passing through to the Earth's atmosphere	Reactions of ozone-depleting effluents with the finite amount of ozone-layer substances. Reference substance: Trichlorofluoromethane (CFC11) .
Reduction of the amount of ozone in the stratosphere.		
Tropospheric ozone creation or photochemical oxidation:	Respiratory problems and aggravation of existing heart and lung problems.	Winter smog: Ground temperatures are lower than upper atmospheric temperature, preventing air from rising, and thus, the dispersion of airborne pollutants. Higher humidity levels in winter enable oxidation of sulphur dioxide to sulphuric acid causing acidic smog. Summer smog: Interaction of sunlight with nitrogen oxides, volatile organic compounds and other pollutants results in formation of low level ozone. Reference substance: Ethylene (C₂H₄) .
Formation of smog near the Earth's surface.		

^aOpinion of the current author.

Life cycle interpretation

The interpretation phase of LCA combines findings from the inventory analysis and impact assessment contrasted with the goal and scope in order to draw conclusions and formulate recommendations (International Organisation for Standards 1997). According to ISO (1997) and supported by Rebitzer et al. (2004) and Pennington et al. (2004), interpretation should occur at every stage of the life cycle assessment. Rebitzer et al. (2004) conceded that inventory analysis phase may offer conclusive information when comparing alternative products.

2.4 LCA findings in South Africa

This section presents review of some findings of life cycle assessments with particular focus on those conducted in South Africa. The aim of this section was to identify typical causes of environmental impacts of relevance regionally which were used in the formulation of hypotheses with regards to the current study.

Due to large coal reserves, the electricity and petrochemical products rely mainly on coal (90% of the electricity and one-third of fuels) (Brent et al. 2002). Many life cycle assessment and environmental footprint studies done in South Africa have identified the coal-generated electricity usage as a major contributor to environmental impacts. This has been fuelled historically by “cheap” electricity, but is changing rapidly. Friedrich et al. (2009) traced most of the environmental burden created by water supply and sanitation systems to direct consumption of electricity and consumption of electricity during manufacturing of materials used in the systems. Cohen (2006) identified power used to pump wastewater from the Newlands brewery to the Athlone Municipal Waste Treatment works (AWTW) in Cape Town as a major contributor to the brewery wastewater management system. In Cohen’s analysis, the second most important contributor to environmental burdens was caustic consumption; however, the burdens were mainly incurred through electricity input during the caustic manufacturing process.

2.5 Anaerobic digestion

2.5.1 Anaerobic digestion background

The current study considers reduction of environmental burdens by dual appropriation of anaerobic digestion (AD) for wastewater treatment and energy generation in the brewery context. Anaerobic digestion is the process whereby organic matter is converted to biogas by microorganisms in the absence of oxygen (Mckendry 2002). Anaerobic digestion technology has been applied in wastewater treatment for reduction of organic wastes for over 30 years (Neira & Jeison 2010). It has been applied in the treatment of organic household, industrial and agricultural wastes as well as sewage sludge (Angelidaki & Ellegaard 2003; Gomez et al. 2006; Bolzonella et al. 2006). However, over the past several years, there has been an increasing interest in the simultaneous application of anaerobic digestion for energy generation (Lyberatos & Skiadas 1999).

This section presents review of literature pertinent to laying foundation for understanding the process of anaerobic digestion and formulation of hypotheses to guide this research. It provides the background to anaerobic digestion, starting with a description of the process including classification of the digester feed and a description of the biochemical processes. Operating conditions and process inhibitions are discussed. Potential of the proposed feedstock for methane generation is reviewed.

2.5.2 Feed classification

The effectiveness or efficiency of an anaerobic digester system depends on the composition of the feed material (Pfeffer 1968). Anaerobic digester feed components can be classified in terms of their physical and chemical properties, as demonstrated by Ekama & Wentzel (2008) in their classification of wastewater to be treated in any biological organic matter removal system. Figure 2.4 shows classification of feed components and their fate in anaerobic digestion systems. All biodegradable components of the feed undergo anaerobic digestion.

Feed constituents			Fate	
Organic matter	Particulate	Settleable	Biodegradable	Bioconversions to active cell biomass and biogas
			Unbiodegradable	Accumulates in sludge mass
		Suspended	Biodegradable	Bioconversions to active cell biomass and biogas
			Unbiodegradable	Accumulates in sludge/Leaves with effluent
	Soluble	Dissolved	Biodegradable	Bioconversions to active cell biomass and biogas
			Unbiodegradable	Leaves with effluent
Inorganic matter	Particulate	Settleable		Accumulate in sludge mass
		Suspended		
	Soluble	Dissolved	Biologically utilisable	Bioconversions to active cell biomass and biogas
			Precipitable	Precipitate to solids and accumulate in sludge mass or leave with affluent
			Non biologically utilisable and precipitable	Leaves with effluent

Figure 2.4: A depiction of feed constituents, their transformation and fate in anaerobic digestion systems. Adapted from Ekama & Wentzel (2008)'s classification and transformation of feed constituents in biological organic matter removal systems.

2.5.3 Anaerobic digestion: Bioconversions

Anaerobic digestion of complex organic molecules is completed in four stages: hydrolysis, acidogenesis, acetogenesis and methanogenesis (Gómez et al. 2006; Batstone et al. 2002; Ezeonu & Okaka 1996; Anderson & Yang 1992). Degradation of generic organic compounds (carbohydrates, proteins and lipids) is illustrated by Figure 2.5.

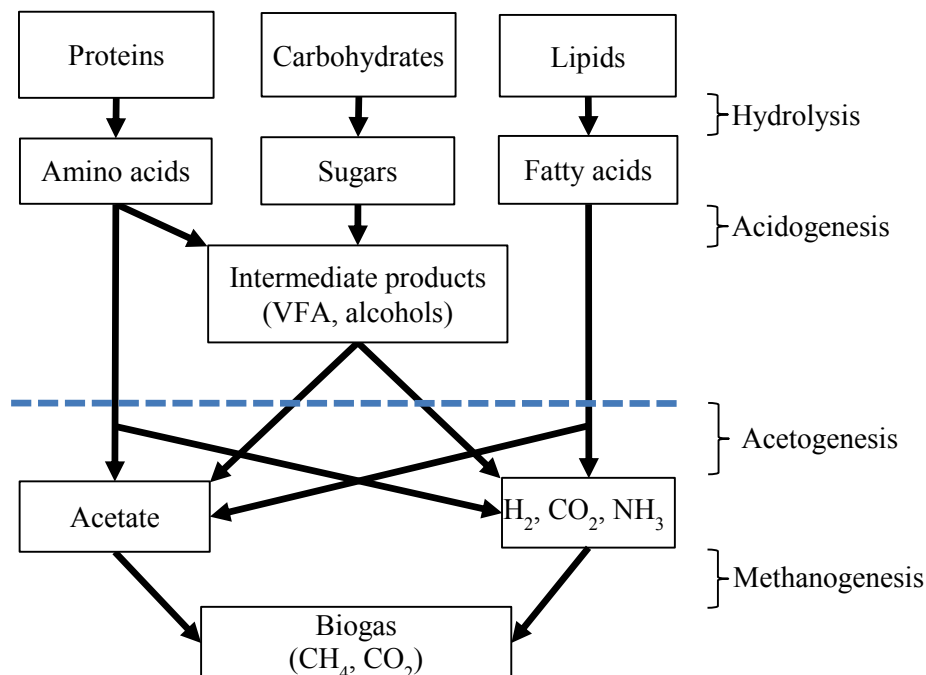


Figure 2.5: Schematic of anaerobic digestion pathways. Adapted from Pfeffer, Leiter & Worlund (1967); Parkin & Owen (1986); Batstone et al. (2002); Appels et al. (2008).

Hydrolysis, acidogenesis and acetogenesis

Biochemical hydrolysis occurs by acidogenic bacteria producing extracellular enzymes to breakdown biodegradable portion of complex organic substrates (van Lier et al. 2008; Shin & Song 1995) to make them accessible for utilisation. Carbohydrates are broken down to starch and soluble sugars. Lipids are broken down to long chain fatty acids (LCFAs) and volatile fatty acids (VFAs) by lipolytic enzymes. The LCFA are broken down further by the same mechanism. Proteins are broken down to hydrolysable amino acids. During anaerobic digestion of complex organic compounds, the hydrolysis is often identified as the rate-limiting step (Zheng et al. 2009; Gómez et al. 2006; Mosey & Fernandes 1989).

Fermentation of soluble organic matter and oxidation of LCFA by acidogenic bacteria result in production of acetic acid, propionic acid and valeric acid, hydrogen, carbon dioxide and water. The minimum doubling time for growth of acidogenic bacteria was reported to be as low as 30 minutes (Mosey, 1983 cited by Gerber & Span, 2008). van Lier, Mahmoud & Zeeman (2008) reported

conversion rate of 13 g-COD/g-VSS.day for the acidogenesis process with a maximum specific growth rate of 2.00 day⁻¹ for acid generating bacteria during the anaerobic digestion process.

Acetogenic groups derive their energy by oxidising volatile fatty acids other than acetic acid (VFAs including butyrate, valerate and propionate) generated by acidogenesis to acetate and hydrogen (Batstone et al. 2002; Gerber & Span 2008). Minimum doubling times for acetogenic bacteria between 1 to 5 days have been reported (Gerber & Span 2008; Lawrence & McCarty 1969).

Methanogenesis

Methane is produced mainly by catabolism of acetate by the acetoclastic methanogenic group (Gerber & Span 2008; Parkin & Owen 1986). This group is responsible for the removal of acidity from AD systems. The rest of the methane is produced from hydrogen and carbon dioxide by hydrogen-utilising methanogenic group. Doubling times of 2 to 3 days have been reported for the acetoclastic methanogens (Gerber & Span 2008) while hydrogen-utilising methanogens had a doubling time of only 6 hours. The COD conversion rate for methanogenesis is 3 g-COD/g-VSS.day.

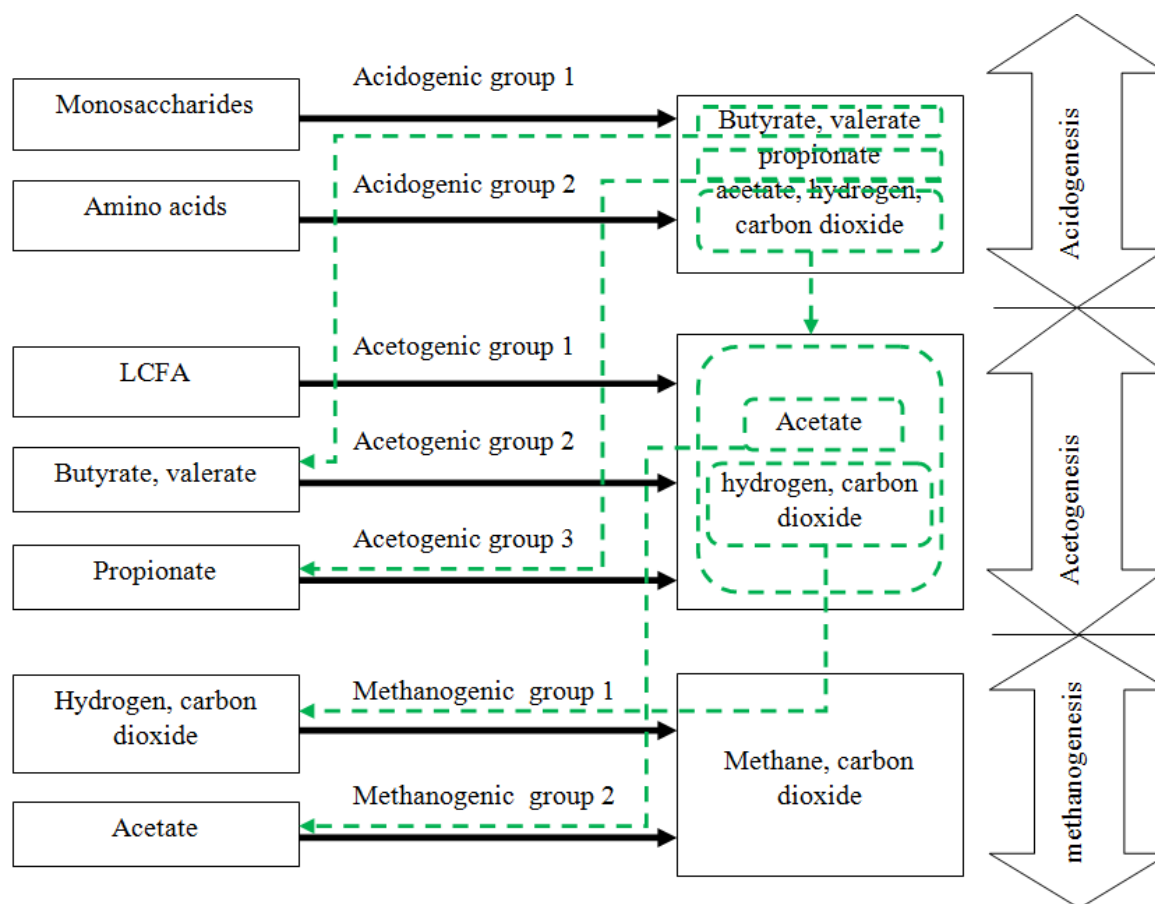
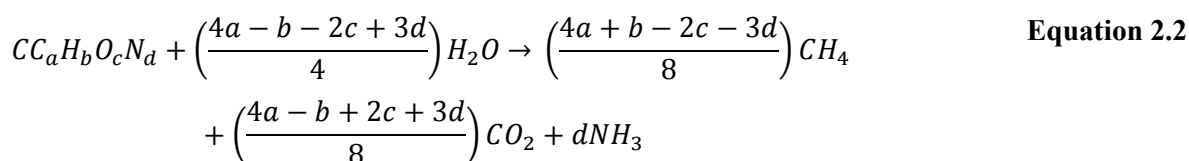
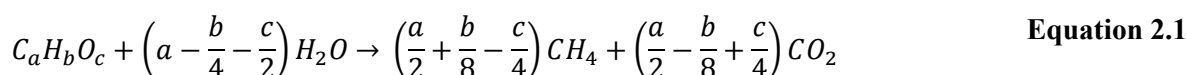


Figure 2.6: Biological conversions by specific microbial groups. The dashed lines show the flow or conversion of the various intermediate products into other products.

Biogas production by black box stoichiometry

According to Angelidaki and Sanders (2004), the theoretical methane yield from anaerobic digestion can be calculated provided the elemental composition of the substrate constituents are known. Equation 2.2 is an adaptation of the Buswell Equation (Equation 2.1) to include nitrogenous substrate. From Equation 2.1 and Equation 2.2, the specific methane yield in litres of methane per gram of total solids (L CH₄/g-TS) can be calculated by Equation 2.3. It was noted by Sialve, Bernet & Bernard (2009) that Equation 2.1 and Equation 2.2 do not account for substrate losses due to microbial cell anabolism. About 5% to 10% of the degraded substrate is utilised for bacterial growth (Angelidaki & Sanders 2004).

Table 2.2 shows theoretical characteristics of some typical substrates and methane content of biogas produced from them. Apart from the utilization of substrate for microbial cell anabolism, several other factors influence the disparity between the predicted theoretical methane yield and actual methane yield. These factors include efficiency of hydrolysis, accessibility of substrate, substrate retention time and operating conditions (Angelidaki & Sanders 2004).



$$Y_{CH_4} = \frac{4a + b - 2c - 3d}{8(12 + b + 16c + 14d)} \times V_m, \text{ where} \quad \text{Equation 2.3}$$

V_m is the molar volume of methane

Table 2.2: Theoretical characteristics of some typical substrates; adapted from Angelidaki & Sanders (2004).

Substrate	Molecular formula	COD/VS (g COD/g VS)	CH ₄ yield (L/g-VS)	CH ₄ yield (L/g-COD)	CH ₄ fraction in biogas (% vol.)
Carbohydrate	$C_6H_{10}O_5$	1.19	0.415	0.35	50
Protein*	$C_5H_7NO_2$	1.42	0.496	0.35	50
Lipids	$C_{57}H_{107}O_6$	2.90	1.014	0.35	70
Ethanol	C_2H_6O	2.09	0.730	0.35	75
Acetate	$C_2H_4O_2$	1.07	0.373	0.35	50
Propionate	$C_3H_6O_2$	1.51	0.530	0.35	58

*N is converted to NH₃

2.5.4 Anaerobic digestion dynamics and control

The key design requirements of anaerobic digestion reactors are to maximise methane production and organic loading rate, while minimising reactor volume (Ward et al. 2008). Several factors such as pH, temperature and concentrations of inhibitory agents influence the performance of anaerobic digestors. Good control of these factors is pivotal for successful operation and prevention of reactor failure. Mixing is also important as it informs the effective digester volume which reflects on COD removal rates and methane productivity per volume of reactor, as well as preventing regions of unsuitable conditions within the reactor.

Operating temperature

Anaerobic digestion reactors can be operated at mesophilic temperatures (25-40°C) or thermophilic temperatures (50-65°C). Thermophilic operations have higher reaction rates which allow for higher OLR (organic loading rate) and more complete and rapid destruction of pathogens than mesophilic operations (Hegde & Pullammanappallil 2007; Borja 2011; Ward et al. 2008; Buekens 2005). In addition, higher degradation efficiencies have been claimed (Buekens 2005). However, thermophilic AD microorganisms are more sensitive to toxins and changes in environmental conditions than mesophilic microorganisms. Hence, mesophilic anaerobic digestion operations are currently more popular due to their relative stability (Buekens 2005). Another advantage of mesophilic operations over thermophilic operations is the low energy required to maintain the operating temperatures. The temperature maintained in the absence of heating or cooling (desirable economically) is affected by reactor configuration, organic load and metabolic activity.

Operating pH

The pH of a digester is one of the most important parameters. The optimum pH for methanogens is around pH 7 (Gunaseelan 1995) whereas the optima for hydrolysis and acidogenesis are between 5.5 and 6.5 (Ward et al. 2008). According to Buekens (2005), most methanogens are only active in a pH range between 6.7 and 7.4. While the difference in optima suggest segregation of the different digestion stages into different reactors or reactor zones (multi-stage digestion), single-stage digestion offers some advantages. The main advantage of single-stage digestion is that compounds formed in one stage are immediately converted, preventing side reactions that may form unwanted or inhibiting compounds (Hendriks & Zeeman 2009). The pH in anaerobic digestion is controlled to favour methanogenic activity (Buekens 2005), as this group is most sensitive to pH changes. Methanogenic activity was greatly reduced when pH dropped below 6.2 (Mosey & Fernandes 1989; Borja 2011). Furthermore CO₂ production is favoured by fermentative microbes at acidic pH. Borja (2011) reported a preferred operating pH range biogas formation of between pH 6.5 and 7.8 for single stage digestors.

Activity of microbial communities and pH

A drop in pH in the digester is usually due to accumulation of VFAs, owing to increased rate of acid formation relative to acid consumption (Borja 2011). Acetoclastic methanogens are mainly responsible for the removal of VFAs and thus pH regulation. Acid generating bacteria have a minimum doubling time of around 30 minutes, while acetoclastic methanogens have a much longer doubling time (2 to 3 days) (Gerber & Span 2008). The disproportion in doubling time implies that the acid consuming methanogens inherently have lower specific activity than acidogenic bacteria. It is, thus, vital to achieve suitable microbial communities in the system (i.e. more methanogens than acidogens) such that methanogenic rates are enhanced, be they growth-associated or associated with cell maintenance.

Organic loading rate and pH

VFA accumulation can either reflect overloading with volatile organic matter or inhibition of the acid consuming methanogens. When the digester is overfed with volatile organic compounds, acidogenic bacteria thrive, producing excessive amounts of VFAs. Accumulation of VFAs may cause the pH to become inhibitory to methanogens (Buekens 2005). However, if the OLR is reduced before the acetoclastic methanogens are inhibited, the ratio of acid production to acid consumption becomes balanced, thus reducing VFA accumulation. Inoculum to feed ratio has also been shown to have an effect on pH (Gunaseelan 1995), thus, suggesting that higher OLR can be supported by a higher concentration of active microorganisms.

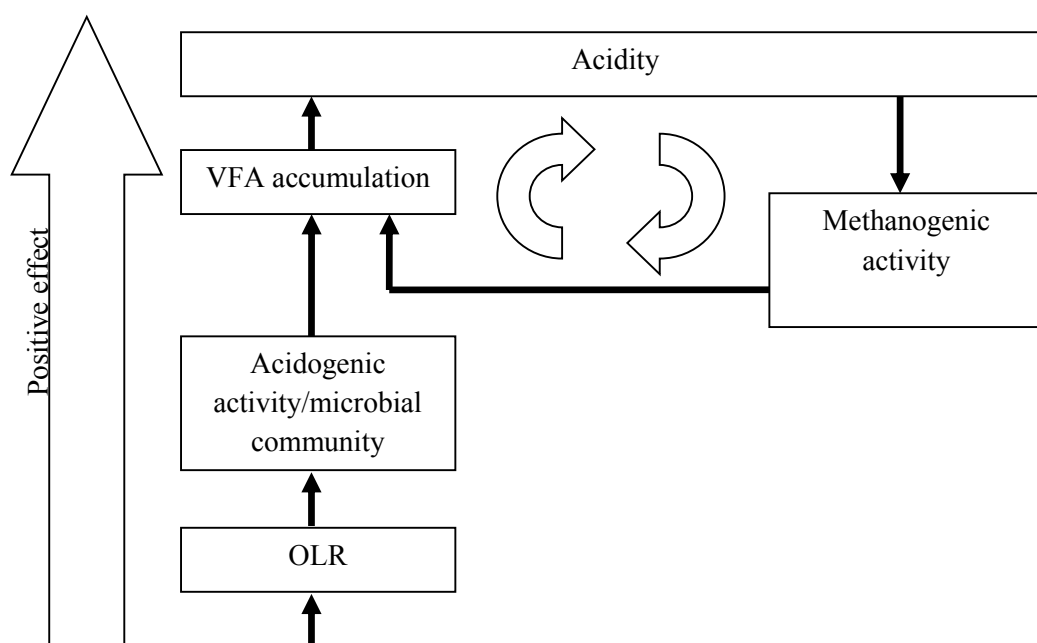


Figure 2.7: Events leading to anaerobic digestion reactor failure chain.

Buffering capacity and pH

Buffering capacity is the ability of an AD reactor to resist a rapid change in pH. It occurs as a result of acid-base equilibria. In AD systems, buffering capacity is determined by measurement of alkalinity and has been directly related to VFA accumulation (Borja 2011). When VFAs accumulate, the buffering capacity is reduced before the pH drops significantly, thus allowing an early warning system. Therefore measurements of alkalinity provide a quick route to anticipate digester imbalances (Ward et al. 2008; Guwy et al. 1997; Borja 2011). Digestors are operating favourably without risk of acid induced failure where the ratio of VFA to alkalinity is less than 0.4 equiv. acetic acid/equiv. CaCO_3 (Rincon et al. 2008). According to Batstone et al. (2002) the most important acid-base pairs in anaerobic digestion reactors are $\text{NH}_3/\text{NH}_4^+$, $\text{CO}_2/\text{HCO}_3^-$ and VFA/VFA^- (see Table 2.3). The ammonia reacts with carbon dioxide to form ammonium ions and bicarbonate ions (Equation 2.4). In the presence of excess carbon dioxide, the reaction is almost complete. As such, almost all nitrogen contained in the biodegradable substrate contributes to an increase in alkalinity and therefore, buffering capacity (Anderson & Yang 1993).

External pH control strategies include addition of a strong base to increase pH and reduce gaseous carbon dioxide from the headspace by converting it into bicarbonate (Borja 2011) according to Equation 2.5. A more rapid and accurate method for increasing the buffering capacity is direct addition of bicarbonate (Ward et al. 2008).

Table 2.3: Acid-base pairs in anaerobic digestion reactors.

Acid/base	Equilibrium reaction	Equation	Reference
$\frac{\text{NH}_3(aq)}{\text{NH}_4^+}$	$\text{NH}_3(aq) + \text{H}_2\text{O} + \text{CO}_2 \leftrightarrow \text{NH}_4^+ + \text{HCO}_3^-$	Equation 2.4	(Anderson & Yang 1992)
$\frac{\text{CO}_2(aq)}{\text{HCO}_3^-}$	$\text{CO}_2(aq) + \text{OH}^- \leftrightarrow \text{HCO}_3^-$	Equation 2.5	(Batstone et al. 2002)
$\frac{\text{VFA}}{\text{VFA}^-}$	$\text{VFA} \leftrightarrow \text{H}^+ + \text{VFA}^-$	Equation 2.6	(Batstone et al. 2002)

2.5.5 Inhibiting factors

Ammonia inhibition

Ammonia in anaerobic digestion reactors originates from the degradation of highly nitrogenous substrates and exists as free ammonia (NH_3) and ammonium (NH_4^+) in the liquid bulk. The associated form of ammonia has been postulated to be the main causes of ammonia inhibition of digestors as it diffuses passively through cell membrane where it causes proton imbalances and potassium deficiency (Strik et al. 2006; Chen et al. 2008; Kadam & Boone 1996; Hansen et al. 1998). In single stage anaerobic

digestion systems, methanogenic microorganisms are least tolerant to ammonia inhibition (Chen et al. 2008). Figure 2.8 shows the speciation of ammonia and other acid-base pairs with change in pH. High carbon to nitrogen ratio (C/N) in the reactor feed leads to accumulation of ammonia in the system which may lead to pH values above 8.5 (Buekens 2005). In these pH environments, ammonia associates to its free form according to Equation 2.4. Koster and Cramer (1987) observed the evidence of ammonia inhibition when they saw about 56% losses of methanogenic populations caused by increase in ammonia from 4051 mg-NH₃/L to 5734 mg-NH₃/L. In addition, some 50% reduction of methane production were observed when total ammonia nitrogen increased from 1700 mg/L to 14000 mg/L (Chen et al. 2008; Kroeker et al. 2013). C/N ratios between 20 and 30 in the feed were suggested for optimal operation of AD processes (Buekens 2005).

Inhibition by ammonia is reversible. Increase in methane production by up to four fold with decrease in pH from 7.5 to 7.0 was reported by Zeeman et al. (1985) as cited by Chen et al. (2008). In a different study by Braun, Huber & Meyrath (1981), cited by Chen et al. (2008), liquid piggery manure was digested at a pH of 8 and VFAs allowed accumulation up to 316 mg/L; thereafter the pH was adjusted to 7.4 where reutilisation of the VFAs to a final concentration of 20 mg/L occurred.

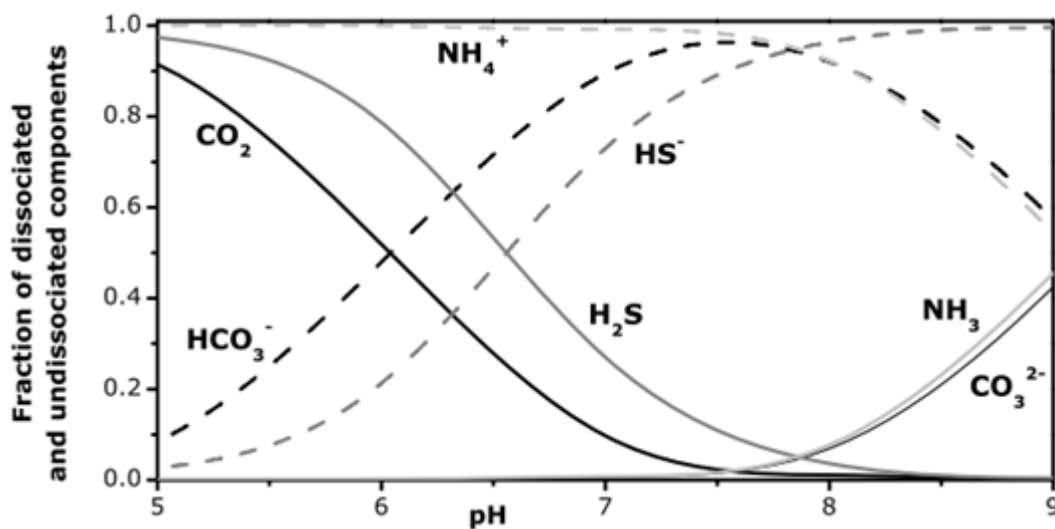


Figure 2.8: Speciation of acid-base pairs in anaerobic digestors (Gerber & Span 2008).

Sulphide and sulphur inhibition (competition)

Sulphate is common in many industrial wastewaters, including those from the brewery (O'Flaherty et al. 1998). During anaerobic digestion oxidised sulphur compounds are reduced by two groups of sulphate reducing bacteria (SRB). The first group are incomplete oxidisers which convert compounds such as lactate to acetate and carbon dioxide. The second group are acetoclastic SRBs (complete oxides) which convert acetate to carbon dioxide (Hilton & Oleszkiewicz 1988; O'Flaherty et al. 1998; Chen et al. 2008). Inhibition of methane production occurs by competition of acetoclastic SRBs and acetoclastic methanogens for the same carbon source (acetate) and as a result of toxicity of the hydrogen sulphide

on the methanogens (Hilton & Oleszkiewicz 1988; Chen et al. 2008). According to Figure 2.8, the hydrogen sulphide (H₂S) fraction of the total sulphur is higher at lower pH than at higher pH. Hilton and Oleszkiewicz (1988) found that carbon removal mainly occurred via the methanogenic pathways at higher pH values (lower H₂S fraction) indicating favourable conditions for methanogenic activity. Both acetoclastic methanogens and SRBs were inhibited in low pH (6.5) environments (higher H₂S fraction).

Metal ions

Light metal ions (such as Ca²⁺, Mg²⁺, Na⁺, K⁺) and heavy metal ions (Cr³⁺, Fe²⁺, Cu²⁺, Zn²⁺, Cd²⁺ and Ni²⁺) may have concentration dependant inhibitory effects in anaerobic digestion reactors (Chen et al. 2008). In particular, the calcium and magnesium ions affect the system by aiding the precipitation of useful nutrients for microbial nutrition as well as affecting the buffering capacity. The heavy metals impact on enzyme structure and function (Chen et al. 2008).

2.5.6 Potential of brewer's spent grain and yeast as feed to anaerobic digestion

Potential of brewer's spent grain

Brewer's spent grain (BSG) is a major by-product of the beer brewing process as it accounts for 30% to 60% of the BOD of the total brewery by-products (Aliyu & Bala 2011; Mussatto & Teixeira 2010). BSG is produced in the mashing process. The temperature in the mash tun is increased up to 78°C to promote hydrolysis of starch from the malt to fermentable sugars. The residual solid fraction is BSG. Table 2.4 shows the typical composition of BSG. The high lignocellulose (lignin, hemicellulose and cellulose) and arabinoxylan content of the BSG is of particular concern. The molecular structures of these constituents are complex and therefore difficult to hydrolyse.

Table 2.4: Composition of brewer's spent grain (BSG). Values sourced from Aliyu & Bala (2011) and Mussatto & Teixeira (2010).

Component	Dry weight %
Protein	15.3 – 27
Lipid	6.4 – 10.6
Arabinoxylan	21.8 – 28.4
Cellulose	16.8 – 25.4
Hemicellulose	30 – 35
Lignin	11.7 – 27
Ash	2.4 – 4.6

Cellulose is comprised of a collection of long polymer chains. The chains are usually arranged in a crystalline structure (Mussatto & Teixeira 2010; Taherzadeh & Karimi 2008; Palmqvist & Hahn-Hagerdal 2000). Cellulose crystallinity, degree of polymerisation, moisture content and lignin content are main limiting factors for lignocellulosic hydrolysis (Hendriks & Zeeman 2009). Highly crystalline

cellulose structures have less surface area for enzyme contact than amorphous structures (Fang et al. 2011; Zheng et al. 2009; Hendriks & Zeeman 2009). This is because the crystalline structures are tightly packed such that the material in the middle of the structure cannot be accessed by the enzymes. Long chains take time to break down to smaller chains, thus reducing the rate of hydrolysis. Lignin content affects hydrolysis by covering the surface of the (hemi)cellulose, preventing enzyme contact. Removal of hemicellulose has been shown to increase the mean pore size of lignocellulosic substrate thereby increasing the probability for cellulose to be hydrolysed (Hendriks & Zeeman 2009). Many pre-treatment methods to enhance hydrolysis of lignocellulosic material have been studied and can be categorised as follows: mechanical pre-treatment, chemical pre-treatment, biological pre-treatment and thermal pre-treatment. Other strategies make use of combinations of these methods (Fang et al. 2011).

Ezeonu & Okaka (1996) studied the process kinetics and digestion efficiency of BSG using batch anaerobic digestion with rumen liquor as inoculum. Degradation efficiencies of 59.5% and 38.9% for cellulose and lignin were achieved after 15 days of digestion. Cumulative biogas production profiles suggested that the digestion depended on two different carbon sources. Ezeonu & Okaka (1996) postulated that the sources may be acetate and intermediates (propionate and butyrate). They further suggested that the accumulation of propionate which was toxic to hydrogen-utilising microorganisms may have caused the first plateau in cumulative biogas production. In addition, they referred to a change of pH as another possible cause of methanogenic inhibition. However, supporting data were not shown for any of the claims. Further, examination of the literature suggests the plateau could not be caused by inhibition of hydrogen-utilising microbes as they are responsible for only 30% of the total methane produced. More likely, the first stage was digestion of readily digestible organic matter and the second digestion of hydrolysed lignocellulose. Although BSG has a considerable amount of lignocellulose as seen in Table 2.4, it can be expected that the crystalline structure of the lignocellulose may have been loosened during the mashing process, thereby making it easier to hydrolyse.

Potential of brewer's spent yeast

Brewer's spent yeast (BSY) is the second major by-product from brewing industry (Ferreira et al., 2010). Yeast converts sugars to ethanol and CO₂ during fermentation. Brewer's yeast has been used as a source of nutrients for human and fish nutrition, microbial growth, production and industrial use of yeast components (Ferreira et al., 2010). Yeast biomass is also a source of proteins and an excellent source of B-complex vitamins, nucleic acids, vitamins and minerals (Ferreira et al, 2010).

Yeast cells are enveloped by thick cell wall comprised of glucan, chitin, phosphomannans and protein (Walker 1998; Mallick et al. 2010). Complete digestion of brewer's spent yeast may be difficult to achieve due to low hydrolysis of the cell walls. Neira & Jeison (2010) studied the effect of various pre-treatment methods on BSY hydrolysis and co-digestion of the BSY with brewery wastewater. In their study Neira & Jeison (2010) evaluated the effect of mechanical, chemical and thermal pre-treatments

on hydrolysis. The measured COD and protein release at different treatment condition, shown in Table 2.5. The substrates treated at the level which had the highest effect on hydrolysis were used for subsequent methanogenic potential experiments.

Table 2.5: Effect of mechanical, chemical and thermal pre-treatments on hydrolysis of brewer's spent yeast (BSY); results of Neira & Jeison (2010).

Method	Conditions	Results
Mechanical: high pressure homogenisation	10 minute cycles at 6000, 18000 and 24000 rpm	No increase in SCOD and soluble protein concentrations (data not shown)
Chemical: Addition of NaOH	NaOH concentrations of 2, 5, 10 and 20 g/L	Higher NaOH concentrations and longer exposure times increased SCOD concentrations
Thermal: Thermostatic bath at 90°C	Exposure times of 3, 6, 12 and 24 hours	SCOD concentration dropped from 100 g/L to around 45 g/L in 12 hours

In their analysis of the results Neira & Jeison (2010) concluded mechanical pre-treatment did not improve hydrolysis. For this reason, substrate treated at the lowest level (6000 rpm) was used for the methanogenic potential experiments. For chemical pre-treatment, alkaline treatment at 20 g-NaOH/L resulted in highest solubilisation. However, this level of treatment would have high cost associated with the amount of NaOH needed. Results from the 5 g-NaOH/L and 10 g-NaOH/L treatments did not differ significantly and SCOD concentrations did not increase significantly after 3 hours of treatment. Hence the 5 g-NaOH/L treatment level was and for duration of 3 hours was chosen as the best for further experimentation. During thermal pre-treatment, a decrease in SCOD was observed and attributed to the evaporation of ethanol. As such, thermal pre-treatment to enhance energy recovery from BSY was deemed unsuitable as energy is lost due to evaporation.

From their methane potential experiment, Neira & Jeison (2010) concluded that the tested pre-treatment conditions did not offer significant improvement in total methane productivity and yield. However, it was noted that very little cell disruption by the selected mechanical pre-treatment conditions was achieved. Similar methane production rates and yield were observed when comparing digestion of spent yeast suspended in tap water to digestion of spent yeast suspended in brewery wastewater. The findings of Neira & Jeison (2010) suggested that pre-treatment of brewer's spent yeast by the aforementioned methods was not beneficial to the anaerobic digestion process. Further, the introduction of spent yeast into an anaerobic digester would not affect the methane productivity and yield.

2.6 Summary of literature review and gaps in the literature

Industrial ecology is a holistic approach of studying interaction within industrial systems and with the natural ecosystem. It upholds the goals of sustainable development in the industrial context by offering a wide range of solutions for resource utilisation, waste management and reduction of adverse impact on the environment. LCA has emerged as the most important tool for analysing the interaction between the industrial and natural ecosystems. LCA studies offer decision making indicators with regards to a process' environmental footprint and give insights into the impact of process development. Anaerobic digestion is a process where complex organic matter is converted by microorganisms into biogas through a network of biochemical reactions under oxygen-depleted environments. During the anaerobic digestion process, complex organic molecules are hydrolysed to their simpler and more soluble constituents. The soluble compounds are subsequently converted to hydrogen, carbon dioxide, ammonia, alcohols and organic acids by acidogenic bacteria. Acetogenic bacteria convert some of these compounds into acetate. Finally, methanogenic bacteria convert the acetate to methane and hydrogen (Angelidaki et al. 1999). The process is completed by various groups of microbial consortia, with each group responsible for specific bioconversions (Figure 2.6). The anaerobic digestion process can contribute to wastewater treatment by reducing organic load and converting organic wastes to methane, the energy carrier of biogas.

The two functionalities of anaerobic digestion, waste (water) treatment and bioenergy production, can offer reductions in environmental burden. However, the effect of prioritising one of the functions may have negative effects on the ability of the anaerobic digester to perform the other function. The framework and methodology to assess this potential conflict has not been presented in literature. From a general level, this study seeks to understand the effect of prioritising energy production on the environmental performance of the anaerobic digester and its associated processes using parts of the LCA framework as a tool and the SABWTP as a case study.

2.7 Project definition

2.7.1 Problem statement and objectives

Increasingly breweries treat their wastewater on-site with concomitant methane production for energy. A large component of the solid waste from the breweries is made up of spent grain and yeast from breweries which are often sold as livestock feed. These by-products have potential to be used as additional organic feed stock for the production of methane. The aim of this study was to evaluate the feasibility of using brewer's spent grain and yeast as additional feed to the anaerobic digester processing effluent wastewater to increase the methane production, thereby increasing capacity for local energy production and reducing dependency on coal-generated electricity. While the reduced dependency on coal-generated electricity would have some environmental benefits, as suggested by LCA studies

conducted in South Africa, the impact of this strategy on the operation of the on-site brewery wastewater treatment plants and the magnitude of consequent environmental impacts are unknown. This study set out to evaluate the digestibility of the spent grain and yeast through a series of anaerobic digestion experiments, to propose various processing options (in addition to the existing platforms) and compare the environmental efficiency of the processing options using life cycle assessment as an analytical tool. The study also set out to identify key contributors to environmental impacts, which are critical to consider for improved environmental sustainability. The SABMiller anaerobic wastewater water treatment plant (SABWTP) at Newlands Brewery was used as a case study.

Key parameters used to evaluate the digestion efficiency of the brewery by-products were: 1) the rate of removal of COD per useful volume of reactor (e.g. mg-COD/L_{reactor}.day), 2) the residual COD concentration in the effluent, and (3) the conversion of COD (COD reacted/COD fed). Biogas and methane production were measured during the anaerobic digestion experiment and used in the LCA study. Further, concentrations of organic acids, pH and alkalinity in the bulk liquid of the anaerobic reactors were measured to provide insight into reaction mechanisms and physicochemical conditions of the particular systems. Based on the results of the anaerobic digestion experiments together with literature considerations, mathematical models were formulated and used, in conjunction with plant historical data, to predict the performance (materials and energy flows) of the on-site wastewater treatment plant upon feeding the additional organic matter. The materials and energy flows were used to compile expected inventory data which, in turn, was used in the environmental assessment study.

2.7.2 Hypotheses

Anaerobic degradation of brewer's spent grain and spent yeast

- Methane productivity of the brewery wastewater system will be increased due to increased organic loading of spent brewers' yeast and grain and anaerobic digestion of the soluble and readily hydrolysable constituents.
- The addition of the brewer's spent grain or spent yeast to the brewery wastewater treatment system reduces the overall digestion efficiency due to the high content of slowly degradable lignocellulosic material and complex walls of the yeast cells, and thus contributes to methane productivity in a limited manner.

Environmental impacts of anaerobic digestion of brewery by-products

- Production of methane by anaerobic digestion and its subsequent use for steam generation reduces the requirement of the brewery for electricity-generated steam, thereby reducing the environmental burden associated with coal-based electricity generation. By further increasing the methane production by anaerobic digestion of organic by-products, including brewer's spent grain and spent yeast, the environmental performance of the brewery is further improved.

- Reduction of volumes of the brewery by-products to be transported from site to farmers will improve the environmental performance of the brewery waste management system by reducing transportation impacts incurred.

2.7.3 Key questions

Methane productivity and yield

- What are the methane yields and COD conversions achieved on anaerobic digestion of untreated brewer's spent grain and spent yeast? i.e. what are the specific methane productivity and COD removal rates?
- How much more methane can be produced by anaerobic digestion of the brewer's spent grain or brewer's spent yeast using surplus digester capacity at the Newlands brewery?

Environmental impact assessment

- By what magnitude are environmental burdens associated with the brewery wastewater treatment systems reduced by the anaerobic digestion of the additional organic matter at SABWTP and subsequent use of the produced methane for generation of steam?
- Which processes and substances contribute significantly to the environmental impacts?
- How much of the reduction in environmental impact results from reduction in transportation requirements for the brewery by-product(s) fed to the SABWTP?
- How does replacing the diverted brewery by-products previously used as animal feed on the farms with an alternative affect the outcome of the environmental impact assessment?

3 Materials and Methods

3.1 Introduction

This chapter presents descriptions of the materials and methods used during laboratory anaerobic digestion experiments and all associated experiments.

3.2 Materials

3.2.1 Mechanical pre-treatment of brewer's spent grain

Brewer's spent grains were sliced into smaller particles in a blender. The progression hydrolysis of organic components was measured by determining soluble chemical oxygen (SCOD) demand concentration in the blended samples after various time periods of blending.

3.2.2 Anaerobic digestion reactors

Anaerobic cultures used to inoculate all anaerobic digestion reactors were obtained from an anaerobic wastewater treatment reactor at Newlands Brewery.

Biomethane potential of spent grain

Biomethane potential tests were carried out in a controlled temperature environment at 37 ± 2 °C. Serum bottles (100 mL) capped with self-sealing rubber septa were used as reactors (see Figure 3.1). A needle pierced on the septum of each bottle served as a gas collection port for biogas volume measurement. Samples to quantify methane content were collected from the head space through the rubber stopper by a 100 μ L *Hamilton* gas syringe. The methane content was analysed by gas chromatography. Gas volume measurement is described in the next subsection. Each bottle, except for the control, was loaded with 1000 mg of spent grain on a dry weight basis. The progression of digestions was determined at various time intervals by sampling biogas for total volume produced and methane content. The pH and COD concentrations measurements were obtained by sacrificing bottles at the various time points.

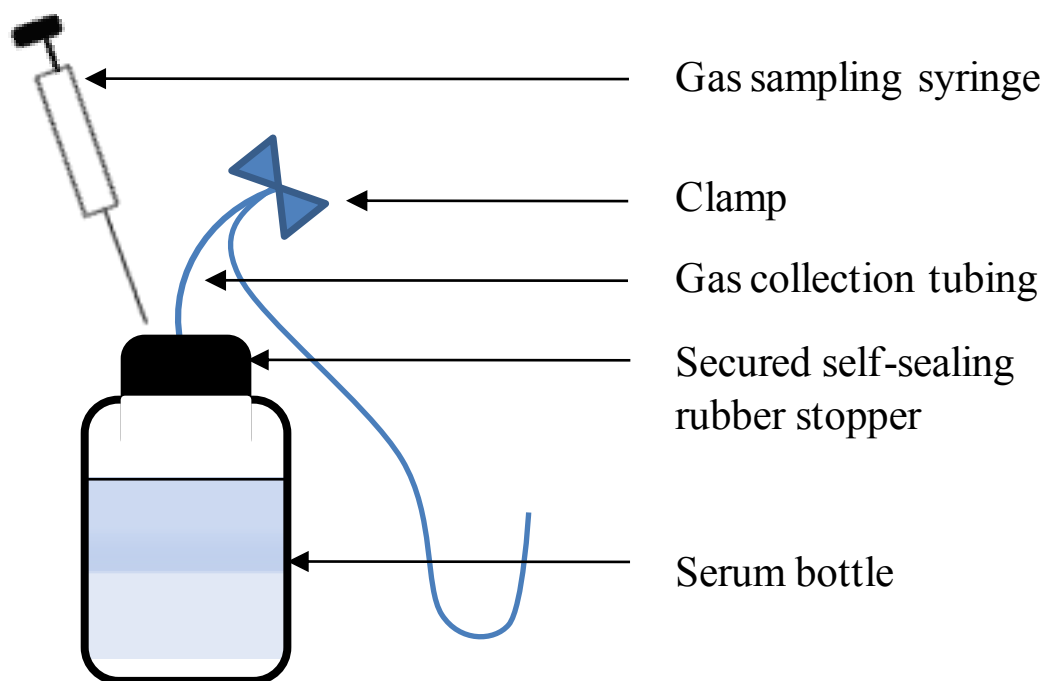


Figure 3.1: Apparatus for biomethane potential tests. Not drawn to scale.

Anaerobic digestion in 1 Litre and 2 Litre Duran bottles

The 1 L and 2 L anaerobic reactor setups were the same except for the volume differences. The experiments were carried out in a controlled temperature environment at 37 ± 2 °C. Each cap of the Duran bottles used was modified to allow for fitment of a metal piece which had three sampling ports. One of the sampling ports was fitted with a self-sealing septum which allowed for sampling of the gas in the headspace by a needle gas syringe. Gas samples obtained from this port were analysed for methane content using gas chromatography. Another port was fitted with tubing which was submerged into the reactor liquor thereby allowing for sampling of the reactor contents. The outside end of the port was connected to a syringe. The remaining port was connected to an inverted volumetric cylinder for biogas volume measurement. The cylinder was filled with a sodium chloride water solution before being inverted with the top submerged in a bucket of water. The NaCl was added in order to reduce the solubility of carbon dioxide. The rubber tubing connected to the reactor port released biogas into the inverted cylinder thereby displacing the water within the cylinder. The volume of the water displaced was taken as the measure of the volume of biogas produced.

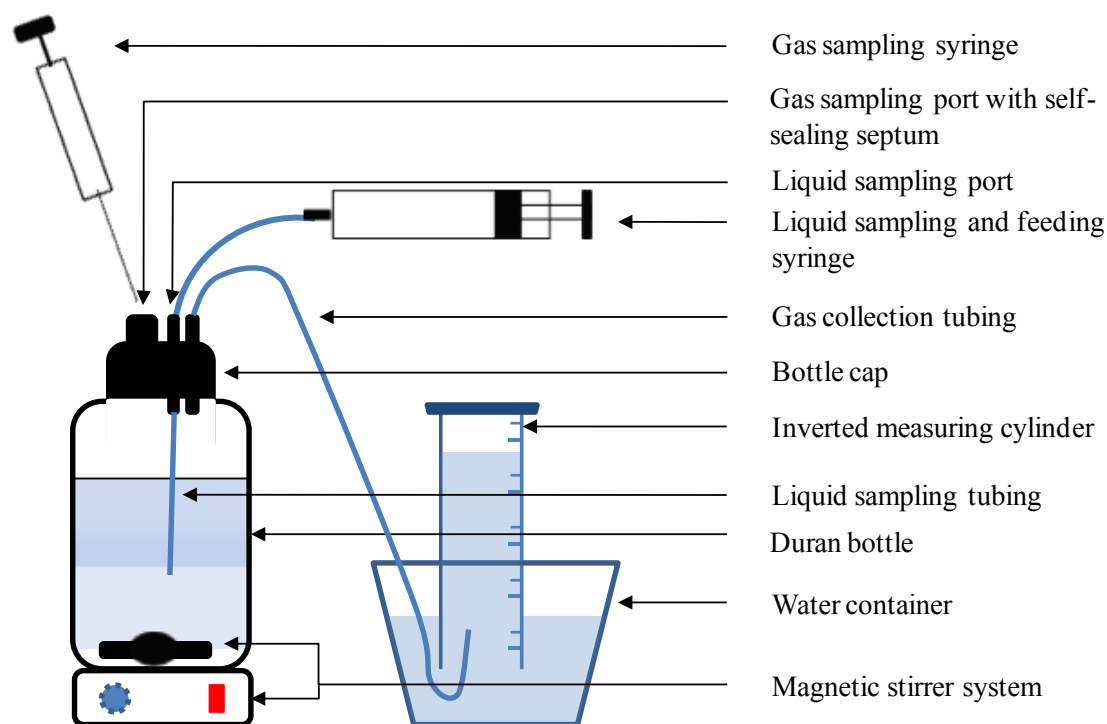


Figure 3.2: Apparatus for 1 L and 2 L anaerobic digestion laboratory reactors. Not drawn to scale.

3.3 Methods

3.3.1 Total solids and volatile solids

Total solids (TS) concentrations were measured gravimetrically by weighing dried samples from known initial volume. The samples of known volume were dried for 48 hours at 70 °C. The TS concentration was determined as the mass of dry samples over the initial volume of the sample. To determine the volatile solids (VS) sample of known volumes were dried for 48 hours at 70 °C in pre-weighed ceramic crucibles. The crucibles with dried samples were weighted to determine the TS concentration before being placed in a pre-heated furnace at 550 °C for 20 minutes. The crucible and remaining ash were then immediately placed in desiccators and allowed to cool to room temperature in a . The cool crucibles with ash were weighted. The VS concentration was determined as the difference between dry sample weight and the ash sample weight over the initial sample volume.

3.3.2 pH and alkalinity

The pH measurements on liquid samples from reactors were carried out using a pH meter. The pH meter was kept in a standard buffer solution at pH of 4.0 and calibrated against a buffer solution of pH of 7 before each use.

Alkalinity was measured according to APHA (1999). A 0.1 N hydrochloric acid (HCl) solution was used to titrate about 40 mL from a starting pH to a pH of 5. Alkalinity in mg-CaCO₃ was then calculated according to Equation 3.1.

$$\text{Alkalinity [mg - CaCO}_3\text{/L]} = \frac{A \times N \times 50\,000}{V}, \text{ where} \quad \text{Equation 3.1}$$

A [mL] is volume of standard acid(HCl)used,

N [-]is the normality of the standard acid,

V [mL] is the volume of sample.

3.3.3 Chemical oxygen demand

The chemical oxygen demand (COD) measurements were carried out using the *Merk* reagent method for high concentrations. All COD measurements, including standard sample for generation of the standard curve, were carried out as follows. The samples were diluted to COD concentrations within the range of the protocol (1500 mg-COD/L to 10000 mg-COD/L). Each measurement required a sample volume of 1 mL. A pipette was used to add 2.2 mL of *Merk COD solution A* to each sample. Solution A contained sulphuric acids which digests and dissolved organic matter in the sample. *Merk COD solution B* at a volume of 1.8 mL was added to each sample. The solution B contained potassium dichromate which was yellow in colour. The dichromate solution changed colour when it oxidised the organic matter in the sample thus changing colour. The extent of colour change was measured in a spectrophotometer at a wavelength of 610 nm. This extent of colour change indicated the amount of material oxidised and therefore the organic content of the sample assuming that all organic matter present was oxidised. Standard solutions of potassium hypochlorite were used to generate the standard curve.

3.3.4 Biogas methane content

Methane content of the biogas was determined using flame ionisation gas chromatography (FID GC) on the *Perkin Elmer Autosystem* gas chromatograph. A *Supelco* column was used. The detector was set at 280°C and the oven temperature of 50°C. Nitrogen was used as the carrier gas at a pressure of 234 kPa. A gas-tight Hamilton syringe of volume 50 µL was used extract gas samples from reactors and to inject the gas samples into the column. The column was calibrated with a standard gas containing 50% methane by volume.

3.3.5 Biomethane potential protocol

Substrate characterisation

Characterised (known total solids (TS), volatile solids (VS), chemical oxygen demand (COD) concentrations and particle size range) dried and ground brewer's grains were used as substrate.

Inoculum preparation

Fresh sludge from the Newlands Brewery UASBR was used as inoculum in volume proportions of 50% sludge and 50% tap water at ambient conditions. The sludge pH was between 6.8 and 7.2 before and after addition of the tap water respectively. The inoculum was incubated at 37°C for 3 days without feeding in order to deplete residual organic content. The inoculum was sampled for TS, VS, and total and soluble COD.

Biomethane potential tests

Equal volumes (100 mL) of the prepared inoculum were added into 100 mL serum bottles. Weighted mass of the dried spent grain substrate were added to the 100 mL serum bottles. The bottles were then sealed with self-sealing septa which were fitted with clamped rubber tubing. Gas was released once a day onto an inverted volumetric cylinder filled with water to determine biogas volume. Methane content was determined by sampling the head space of each bottle by a gas tight 100 μ L *Hamilton* syringe and analysing the sample using the *Perkin Elmer Autosystem* gas chromatography.

4 Anaerobic Digestion: Results, Analyses and Modelling

4.1 Introduction

Maximising resource productivity and minimising environmental impact are key foci of sustainable development. The Newlands Brewery's wastewater treatment plant (SABWTP) increases the productivity of organic inputs into the brewery and reduces environmental burden by recovering energy from the organic fraction of the brewery's effluent water through anaerobic digestion. The current study seeks to enhance energy production by investigating the feasibility of methane (or energy) recovery from BSG and BSY through anaerobic digestion. This chapter presents the experimental work done on anaerobic digestion of BSG and BSY at laboratory scale for recovery of energy from these substrates using laboratory scale reactors. The experimental program was designed to provide data of the rate and extent of anaerobic digestion. Simplistic empirical mathematical models to describe anaerobic digestion were formulated from the results of these laboratory experiments in conjunction with literature findings. The models and data were for modelling the effect of addition of these substrates on the performance of the SABWTP as presented in Chapter 5, as well as environmental impact of recovering methane for steam generation as presented in Chapter 6.

4.2 Anaerobic digestion of brewer's spent grain

Brewer's spent grains were chopped in a blender to increase ease of representative sampling for chemical oxygen demand (COD) tests during anaerobic digestion. However, the blending was expected to promote hydrolysis by flaking (or mechanically hydrolysing) the grains and releasing readily soluble organic residues, and by reducing the particle size and increasing surface area available for enzymatic hydrolysis during digestion. As such, blending would introduce a favourable bias in comparison with whole grain digestion. To account of these effects, samples were taken at different time points during the blending to determine the amount of COD released into the liquid phase. The release of organic matter into the liquid phase reached a plateau at approximately 35 mg-COD per g-dry spent grain as shown by Figure 4.1. The released COD was minimal in comparison to the total COD of dry spent grain of 1200 mg-COD/g, hence it was concluded that the blending did not enhance hydrolysis on its own.

To test the digestibility of spent grain, biomethane potential tests were carried out on blended grain samples of various sizes. The spent grain was chopped in a food processing blender at 2000 rpm for 3 minutes. Thereafter, dried and classified through sieves. The classification was carried out to evaluate the effect of particle size on digestibility, as well as possibly segregating the various components of the spent grain with respect to their detachment from the husks during the blending process. Chemical oxygen demand assays were carried out on the largest and smallest particle sizes. The particles retained by the 2 mm sieve contained approximately 1279 ± 18 mg-COD/g-chopped spent grain > 2 mm on a

dry basis, while particles passing through the 0.850 mm sieve contained 1183 ± 15 mg-COD/g-chopped spent grain <0850 mm on a dry basis. Figure 4.2 shows the mass distribution after classification. The larger particles (retained by 1.18 mm and 2.00 mm aperture) were visibly comprised mainly of unchopped lignocellulosic barley husks while the remainder passing through were fragments of the chopped husks.

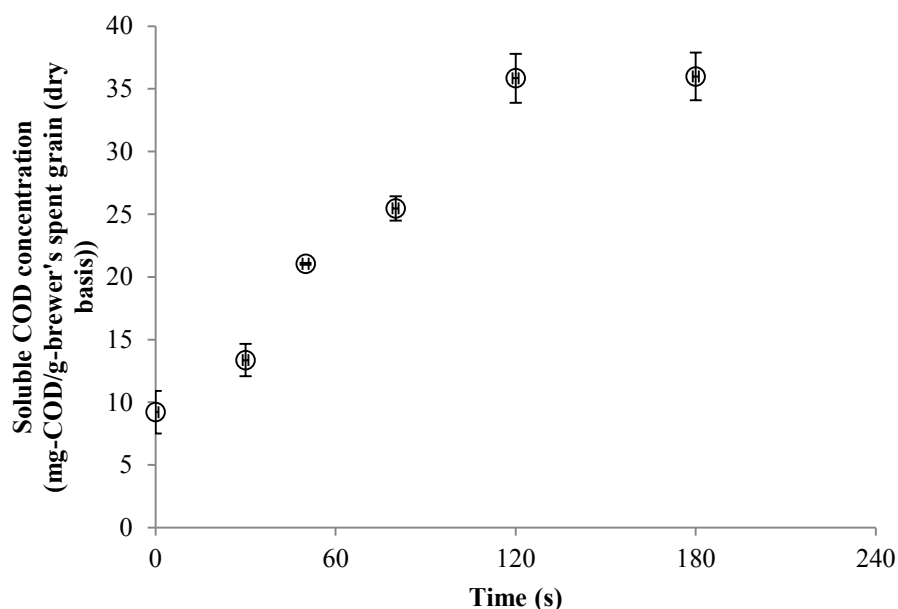


Figure 4.1: Soluble COD released as a function of time of blending.

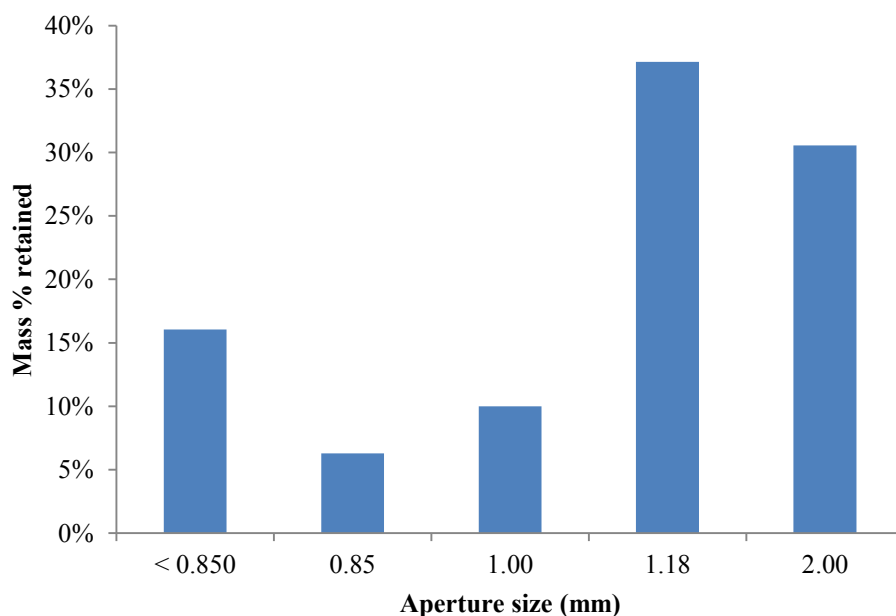


Figure 4.2: Particle size distribution on blended and dried spent grain.

4.2.1 COD utilisation during anaerobic digestion

Anaerobic digestion of chopped, dried and classified spent grain was carried out by the biomethane potential experiment in 100 mL serum bottles. The control bottle was inoculated with sludge but no solid feed was added. The tests were carried out in duplicate, except for the control, for 19 days. With each bottle, the difference in total chemical oxygen demand (TCOD) between the start and the end of the experiment were similar to those observed on the control bottle (See Figure 4.3). This indicated that much of the organic matter consumed during the experiment was the residual substrate in the inoculum. The difference between total COD and soluble COD represented the contribution of the organic solids. There were no significant differences between the solid COD concentrations between day 0 and day 19. This indicated that the change in TCOD during the digestion period was mainly due to consumption of soluble COD and that the solid spent grains were not significantly hydrolysed. It was, thus, concluded that the spent grain was not degraded within the 19 days of batch digestion. In another study by Ezeonu & Okaka (1996), only 2.4% overall conversion of volatile organic solids was obtained after 15 days of batch digestion. Of the 2.4% conversion, 59.5% was attributed to destruction of cellulose and 38.9% destruction of lignin. These results were similar to the current study despite the fact that a different source of inoculum was used. Ezeonu & Okaka (2006) used fresh cow rumen liquor. Cow rumen is expected to contain microbial communities with probability of being capable of producing lignocellulose degrading enzymes as fibres form the major component of cattle feed.

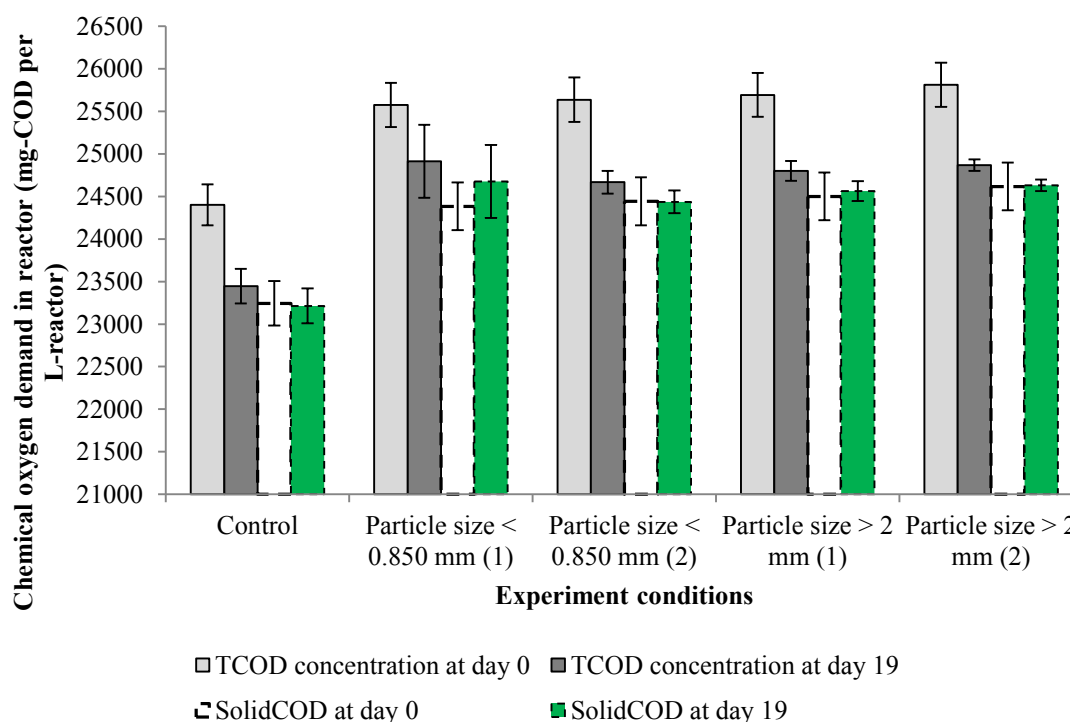


Figure 4.3: COD consumed during the biomethane potential of spent grain experiment.

4.2.2 Biomethane potential

Figure 4.4 shows the cumulative volume of methane collected during the biomethane potential tests. All reactors showed similar trends except for the control and one of the test bottles fed with substrate particles sizes <0.850 mm. It was observed that the integrity of the rubber stopper for the latter test bottle was compromised on day one of sampling. This resulted in gas leakage and the lower value of methane collected. An attempt to mend the test bottle was made by sealing the leakage site with silicone. Despite the above mentioned discrepancies, the results showed similar methane potential across the substrate particle sizes. This suggested that particle size did not affect the methane potential of the substrate and, therefore, pre-treatment by grinding was defective. Methane yields were 76-110 L/kg-COD consumed for substrate particle sizes <0.850 mm, and 77-82 L/kg-COD consumed for substrate particles >2 mm. These values are very low compared to the theoretical value of 350 L/kg-COD consumed at standard temperature and pressure.

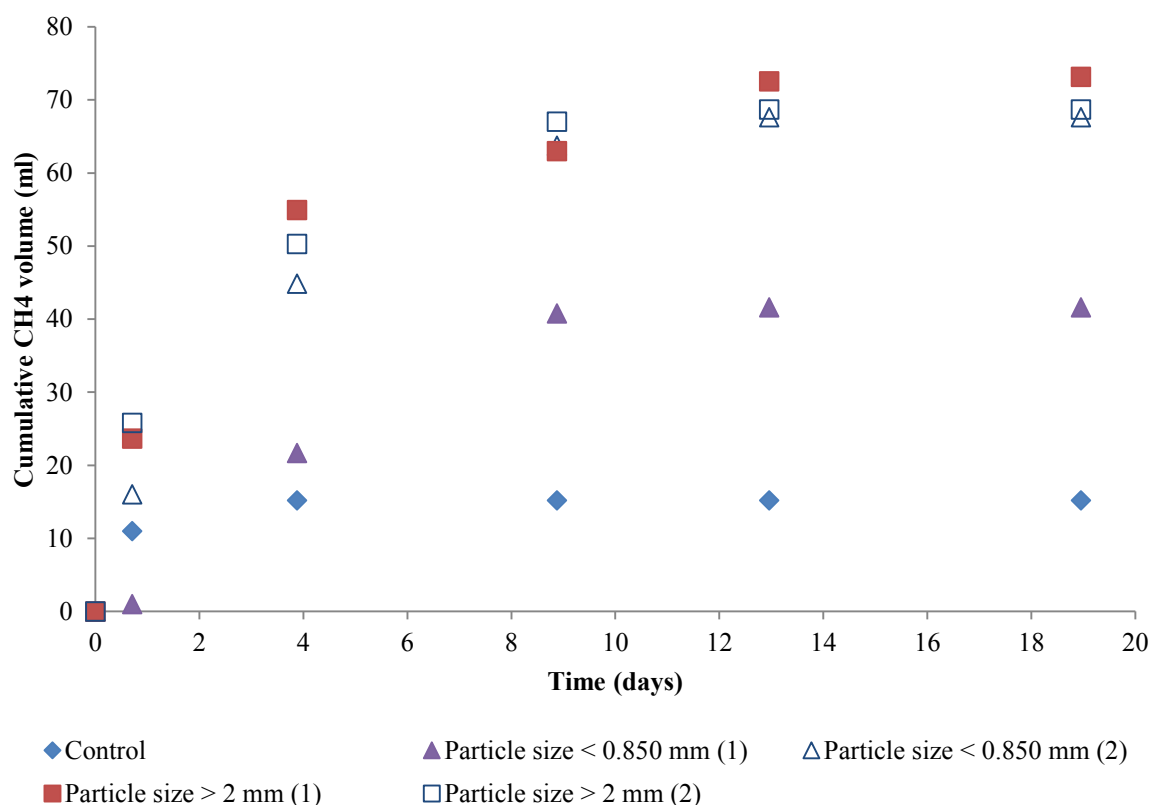


Figure 4.4: Cumulative methane collected during biomethane potential tests on brewer's spent grains chopped to various particle sizes in a blender. The experiments were conducted in duplicates, (1) and (2) for each particle size.

4.3 Anaerobic digestion of brewer's spent yeast

4.3.1 Characterisation of spent yeast

The brewer's spent yeast is a concentrated suspension of yeast cells in beer. The cell or solids concentration of the suspension differs according to the beer recovery process after fermentation. In this study, the total COD of spent yeast used ranged between 100 g-COD/L and 137 g-COD/L with an average (of 19 samples) of 110 ± 9.6 g-COD/L. Soluble COD ranged from 61 g-COD/L to 68 g-COD/L with an average of 65 ± 2.2 g-COD/L, contributing a minimum of 56% and a maximum of 64% with an average of $60\% \pm 2.7\%$ to the total COD. However, Neira & Jeison (2010) measured average TCOD concentration of 200 g-COD/L with soluble COD concentration of 90 g-COD/L. The disparities in the concentrations presented above are mainly a result of varying solids (cell) concentrations in the spent yeast suspensions, which is influenced by the beer recovery strategy of the specific brewery.

Protein is the most abundant constituent of yeast cells as shown in Table 4.1. As such, the degradation of protein is important in achieving high degradation efficiencies. Carbohydrates and phospholipids make up the majority of the COD not contributed by protein. Together, protein, carbohydrates and phospholipids make up about 80% of the cell mass.

The carbon to nitrogen ratio of the yeast cells is 4.6, which is much lower than the ideal ratio (between 20 and 30) as recommended by Buekens (2005) to avoid excess of ammonia. The implications of this on anaerobic digestion of spent yeast at the SABWTP are discussed in Section 4.3.7. Zupančič et al. (2012) reported spent yeast from Brewery Laško having average total solids concentration of 188 g-TS/L of which 95% were volatile solids. The average measured total COD concentration of the spent yeast was 277 g-COD/L (Zupančič et al. 2012), corresponding to an average COD content of 1.55 g-COD/g-dry cell (VS). This value is similar to the 1.36 g-COD/g-cell calculated according to the composition of yeast cells as presented in Table 4.1.

The spent yeast liquor is essentially unrecovered beer, which is made up of a variety of readily digestible volatile compounds. In addition to various forms of carbohydrates and alcohol, low amounts of other compounds such as lactic and pyruvic acids, adenosine, uridine, tyrosine have been found in beer (Almeida et al. 2006).

Table 4.1: Average elemental composition of yeast biomass and COD contribution of the constituents. Elemental composition and weight contribution data sourced from Villadsen et al. (2011). COD calculated according to theoretical oxygen requirement of a combustion reaction.

Constituent	Elemental composition	Weight contribution	Yield g-COD/g-yeast cell	COD contribution
Protein	$\text{CH}_{1.58}\text{O}_{0.31}\text{N}_{0.27}\text{S}_{0.004}$	57%	0.763	55.8%
RNA	$\text{CH}_{1.25}\text{O}_{0.75}\text{N}_{0.38}\text{P}_{0.11}$	16%	0.167	12.2%
DNA	$\text{CH}_{1.15}\text{O}_{0.62}\text{N}_{0.39}\text{P}_{0.10}$	3%	0.032	2.4%
Carbohydrates	$\text{CH}_{1.67}\text{O}_{0.83}$	10%	0.143	10.5%
Phospholipids	$\text{CH}_{1.91}\text{O}_{0.23}\text{N}_{0.02}\text{P}_{0.02}$	10.8%	0.200	14.6%
Neutral fat	$\text{CH}_{1.84}\text{O}_{0.12}$	2.5%	0.052	3.8%
Cellular metabolites	$\text{CH}_{1.8}\text{O}_{0.8}\text{N}_{0.2}\text{S}_{0.01}$	0.7%	0.011	0.8%
Yeast biomass	$\text{CH}_{1.596}\text{O}_{0.396}\text{N}_{0.216}\text{S}_{0.0024}\text{P}_{0.017}$	100%	1.367	100%

4.3.2 Comparing digestion of washed spent yeast cells and spent yeast supernatant

A comparison between digestion of spent yeast supernatant (beer) and washed spent yeast cells provided insight into the digestibility of the spent yeast suspension. The progression of digestion was monitored by measurement of total COD (TCOD) and soluble COD (SCOD) concentrations in the reactors. Solid COD was calculated by the difference. The results are represented by Figure 4.5, with (b) representing the repeat experiment for reactor fed with washed spent yeast cell (a), and (d) representing repeat experiment for reactors fed with spent yeast supernatant (c).

The final TCOD concentration for both experimental setups was slightly higher than the initial TCOD concentration before the substrates were added. This was accompanied by slightly higher solid COD concentrations in the case of results presented in Figure 4.5 (a). In fact, the difference between initial TCOD and final TCOD was due to increase solid COD. The solids concentrations during anaerobic digestion of the washed cells and spent yeast liquor increased by 351 ± 108 mg-COD/L and 725 ± 90 mg-COD/L respectively. The increase in solid COD concentration suggested that a portion of the substrate was not converted to biogas within the duration of the experiment. This portion formed part of the sludge either as slowly biodegradable solids or converted to microbial biomass. The washed spent yeast contained small amount of SCOD. During digestion of the washed cells, the steepest decrease in TCOD was observed between day 0 and day 1. However, the accompanying drop in SCOD was much lower indicating that the TCOD decrease was mainly due to degradation of the solids present. A small drop in TCOD between day 1 and day 2 was accompanied by an increase in SCOD. This suggested that solids were being hydrolysed at a much faster rate than the conversion of their soluble products to

methane and carbon dioxide. During this time interval, methanogenesis was likely to be the rate limiting step. Between day 2 and day 4, SCOD concentration dropped and was accompanied by a slight increase in solid COD concentration. This may have indicated an increase in methanogenic activities resulting from an increase in acetoclastic methanogenic communities, if the premise that an increase in solid COD indicated microbial growth. During anaerobic digestion of the spent yeast supernatant the highest degradation rate was observed between day 0 and day 1. However, the TCOD degradation rate was lower than that of the SCOD over this period. This suggested that the solubilised organic matter was used to produce biomass which resulted in lower removal of COD from the reactor.

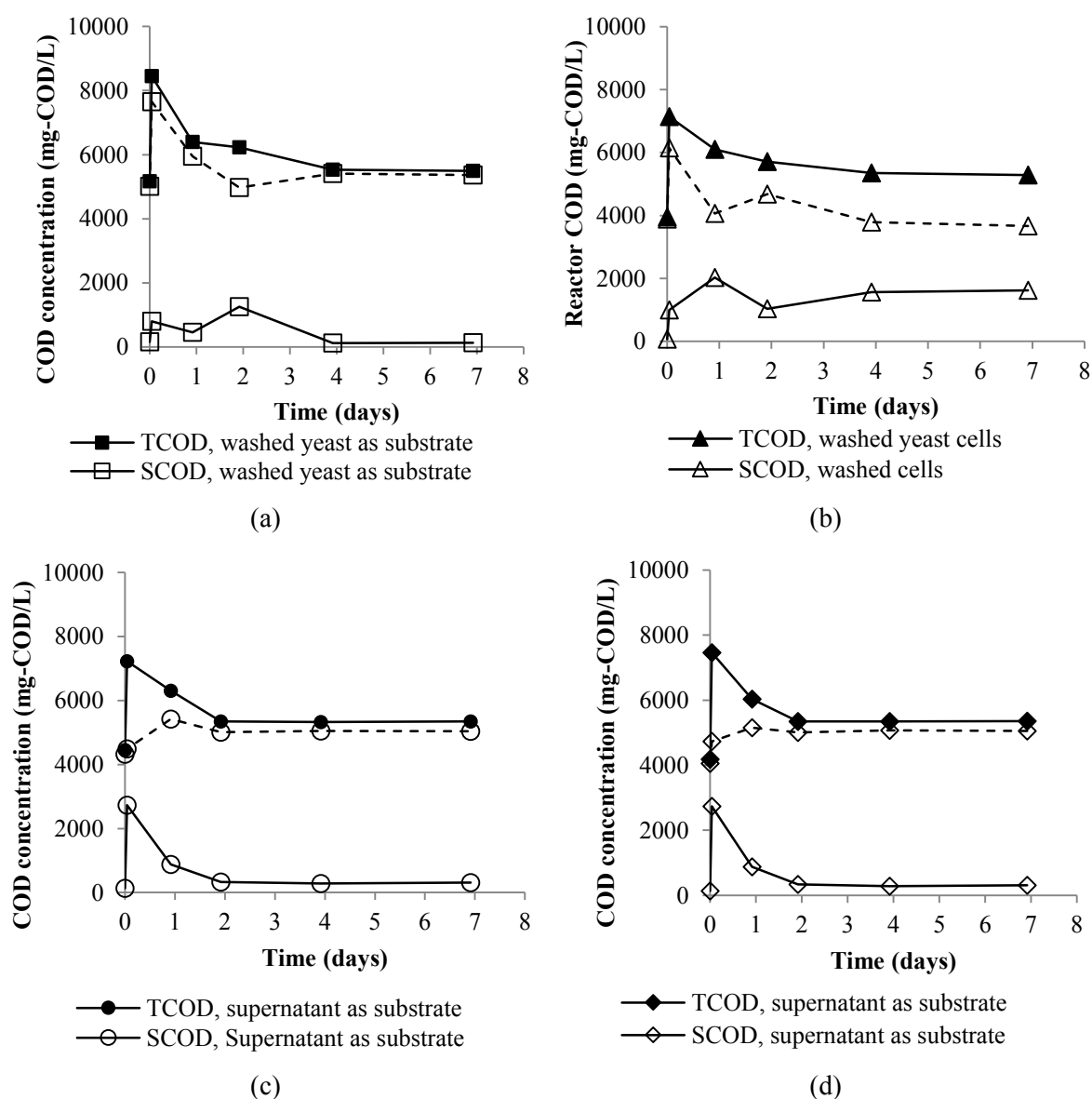


Figure 4.5: Total oxygen demand (TCOD) and soluble oxygen demand (SCOD) profiles for batch digestion of washed spent yeast and spent yeast supernatant. Reactors represented by (a) and (b) were fed with washed yeast cells and reactors represented by (b) and (c) were fed with spent yeast supernatant.

4.3.3 Gradual increase of organic feed and solids concentration

Anaerobic digestion of spent yeast liquor

A sequential batch reaction study was carried out on the spent yeast liquor. Spent yeast was centrifuged and the liquor decanted to separate from the cell biomass. The liquor was digested in order to evaluate anaerobic digestion of the soluble organics, while the yeast biomass were re-suspended in tap water and digested in a separate experiment. At the end of each batch cycle, the reactor samples were drawn such that the sum of all sample volumes drawn in that cycle would be equal to the feed volume of the subsequent cycle, in order to keep the initial reactor volume constant. The COD profiles shown by Figure 4.6 indicate that the soluble organic matter fed to the reactor was consumed almost completely. The unconverted soluble COD fraction after each feeding cycle was calculated as the difference between the initial soluble COD concentration before feeding and the initial soluble COD concentration of the next succeeding feeding cycle. While the positive effect of substrate concentration on removal rates within the tested range is echoed by the Monod model for specific growth rate, some of influence of adjustment of microbial community structures or microbial physiology to the various feed concentrations may have occurred. Parkin & Owen (1986) highlighted the importance of microbial community structures by as key consideration in design and operation of digestors.

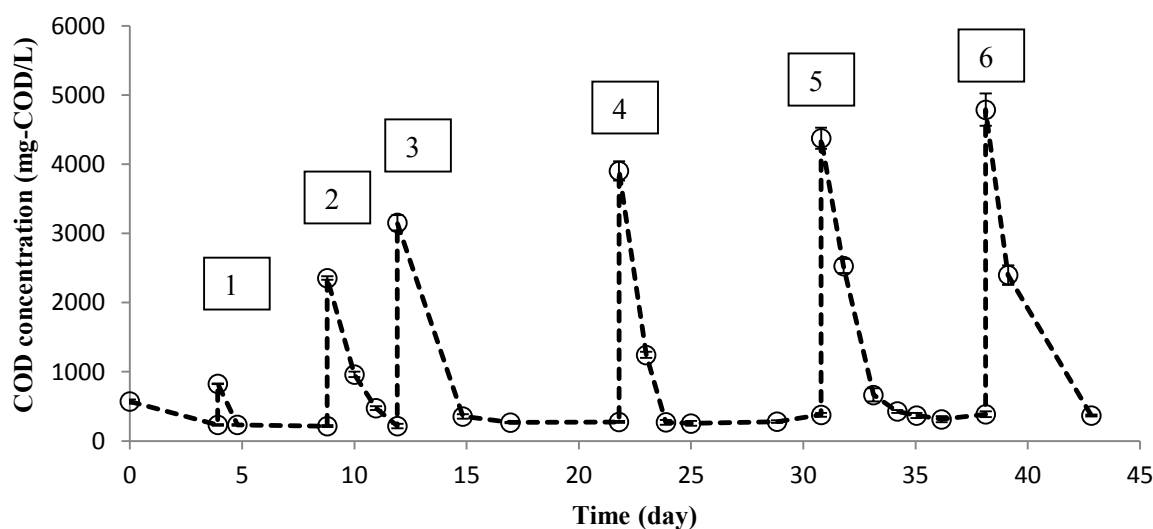


Figure 4.6: Soluble COD profiles of reactor fed with yeast supernatant in fed-batch mode. Each peak represents a response to feeding at the start of a batch cycle.

The pH profile shown in Figure 4.7 suggested a gradual decrease in the ratio of acid consuming methanogens to acid producing microorganisms. The increase in pH between batch cycles 1 and 3 indicated removal of acidity from the system, which may have resulted from lower acidogenic activity versus methanogenic activity. This increase in pH may also be attributed to production of alkalinity. The notable dips in pH from cycle 4 onwards may have indicated that the acid producing steps of the

digestion process where favoured by addition of feed and exceeded methanogenic rates. However, the acetoclastic methanogens began to thrive as their organic acids substrate concentration increased causing an increase in the pH. The prominent fluctuations in pH after batch cycle 3 suggested that the system was near the maximum loading capacity above which external methods of pH control would have to be administered in order to avoid reactor failure.

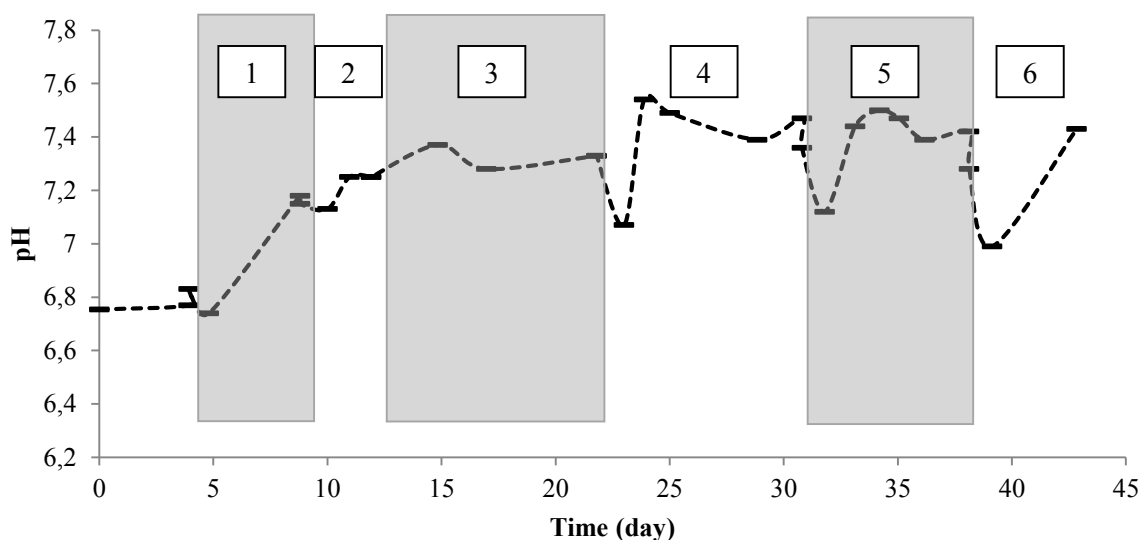


Figure 4.7: pH profiles of reactor fed with yeast supernatant in fed-batch mode.

Effect of solids concentration on digestion

After feeding the spent yeast liquor for approximately 30 days, washed spent yeast cells were fed into the reactor. The washed spent yeast feed was concentrated by centrifugation and re-suspended in tap water to increase the COD contribution by of yeast cells. The total COD fed at each cycle and the contribution of the solid are depicted in Figure 4.8. The resulting TCOD and SCOD profiles are shown in Figure 4.9.

Figure 4.9 shows that degradation was not completely inhibited by the lack of soluble organic material as solid COD concentration decreased even in the presence of SCOD. At the end of cycle 7, 27% of the total solid COD fed to the reactor was remaining. This occurred over a 12 day period. During the start of the cycle 8 of digestion, solid COD concentration initially decreased at a higher rate than SCOD. This indicated the result of a system that was more acclimated to the solid yeast cells as substrate. Further, the lower initial removal rate of SCOD was an indication of the hydrolysis of the solid part of the feed, thereby contributing to the SCOD concentration. However, SCOD concentration started decreasing more rapidly than solid COD concentration indicating improved methanogenic rates as compared to hydrolysis. This corresponded to an increase in methane content of the biogas (see Figure 4.14).

Initial degradation rates of solid COD in the subsequent batch fermentation of spent yeast suspension were higher as compared to SCOD initial degradation rates. These results could be attributed to improved acclimation of the sludge to the feed. Since hydrolysis is an enzymatic reaction, concentrations of hydrolytic enzymes may have been accumulating overtime leading to the higher degradation rates of the solid substrate. Further, the accumulation of hydrolytic enzymes may have been enhanced by the lack of soluble biodegradable matter during the final periods of the batch cycles. This would have forced the acidogens to source energy from the slowly degradable organic matter, thereby releasing the hydrolytic enzymes.

Solid COD profile calculated as the difference between the measured TCOD and SCOD are presented in Figure 4.10. During cycle 7 and cycle 8, the initial rate of solid COD removal was high, followed by a stationary section and subsequent lower removal rates. This diauxic property suggested that the yeast cells can be classified into at least two groups of constituents according to their degradation rates. It was apparent that there are some constituents of the yeast cells were more readily hydrolysable and their digestion which were characterised by the initial high solid COD removal rates. According to the solid COD profile of cycle 8 (Figure 4.10), the readily hydrolysable portion of the yeast cells constituted about 50% of the total solid COD added. Yeast cell walls contain β -glucan, which is comprised mainly of β -1,3-glucan chain and about 3% β -1,6-glucan web (Thammakiti et al. 2004). Since the hydrolysis of complex organic usually limits the rate of degradation in single-stage digestion (Zheng et al. 2009; Mosey & Fernandes 1989), the high degree of polymerisation of the β -1,3-glucan chain was expected to limit the rate of degradation spent yeast cells. As such, the β -glucan components can be attributed to the slowly biodegradable portion of the yeast cells.

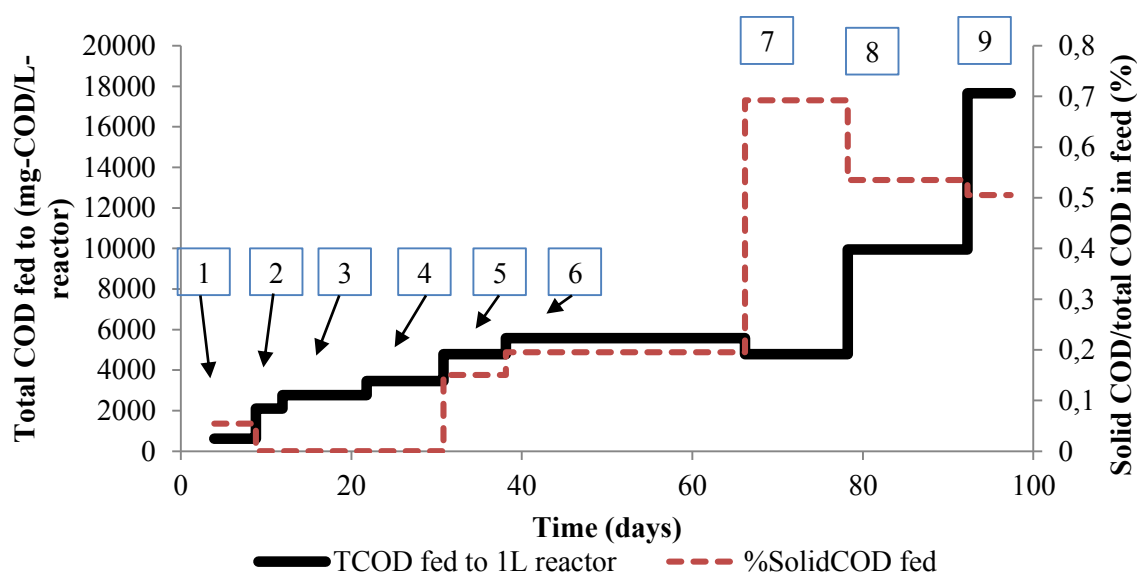


Figure 4.8: Total COD fed and solid COD contribution of the feed at each batch cycle. Each step represents the feed loading at the start of the batch cycle.

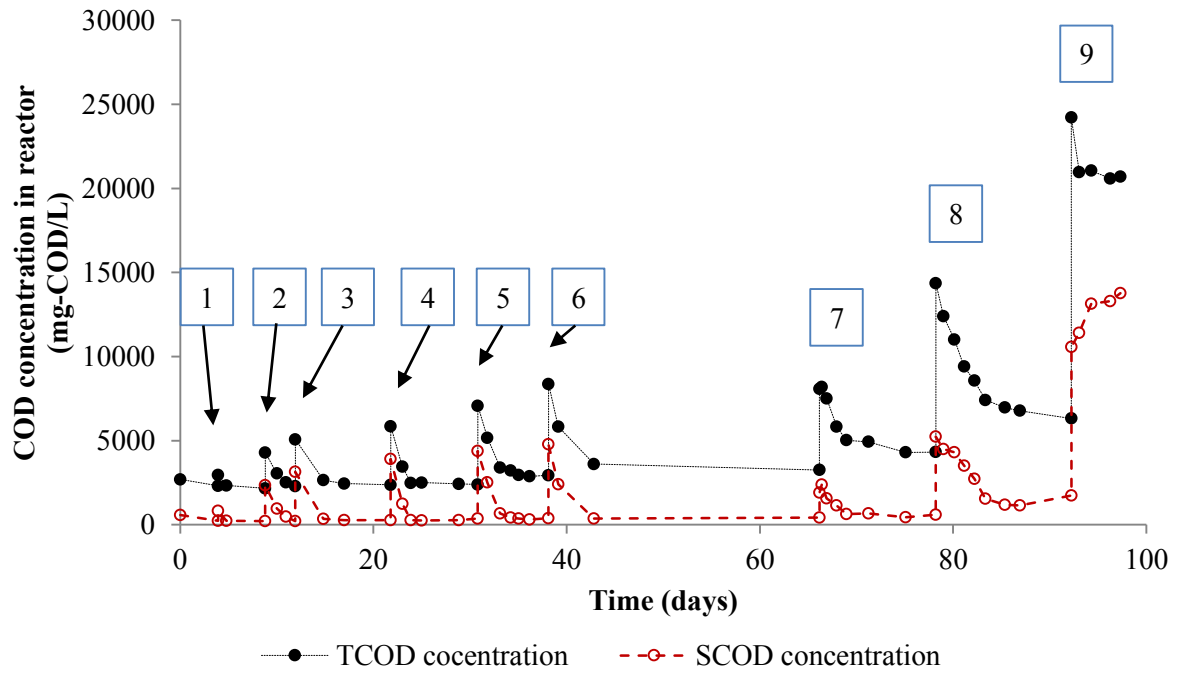


Figure 4.9: COD profiles of fed-batch anaerobic digestion with varying concentration of solids in the feed. Each peak represents a response to feeding at the start of a batch cycle.

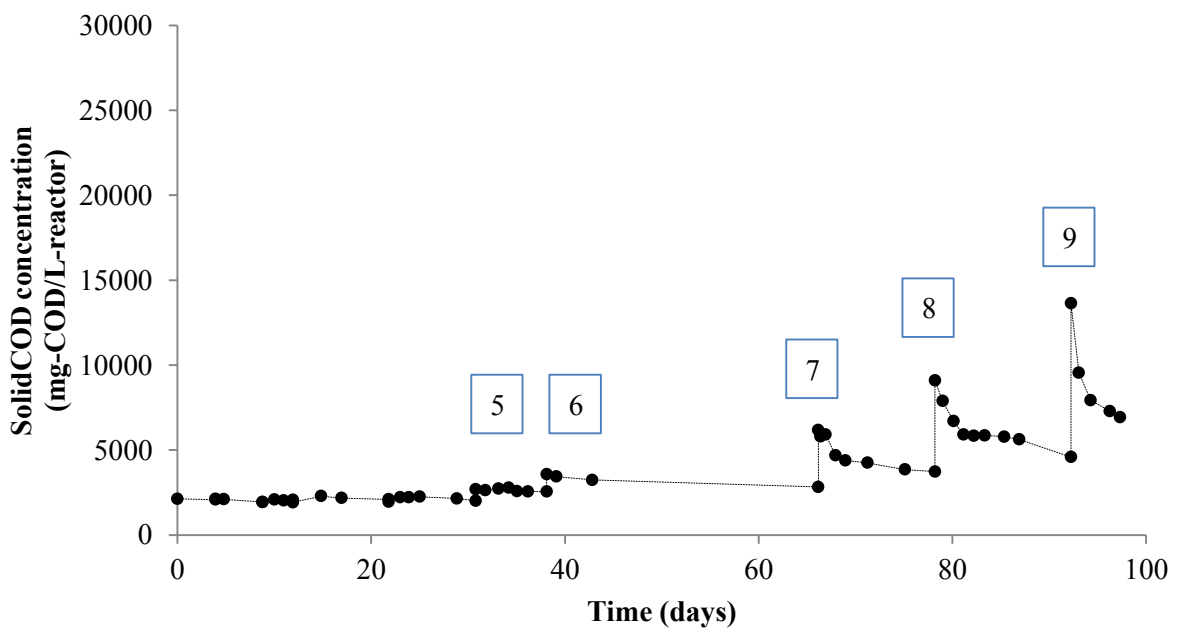


Figure 4.10: Solid COD profiles calculated as the difference between measured TCOD and SCOD (shown in Figure 4.9).

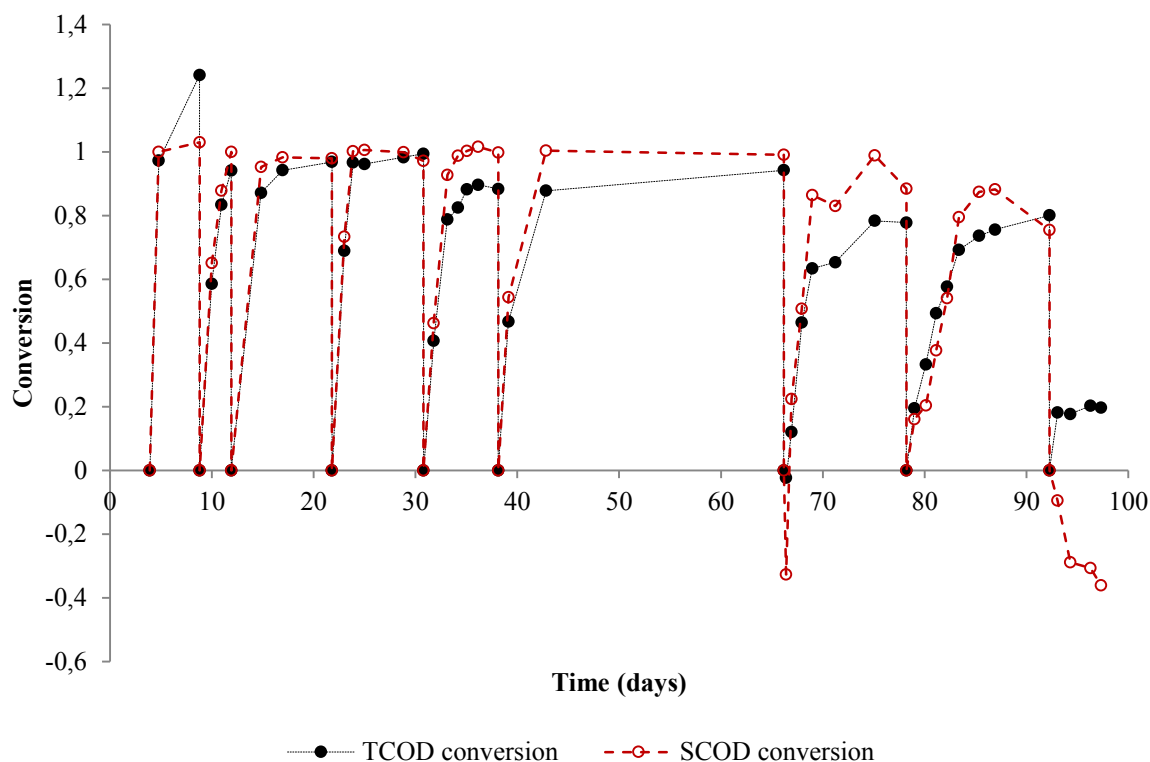


Figure 4.11: Conversion of TCOD and SCOD with time.

Cumulative conversion was calculated as the cumulative TCOD reacted divided by the cumulative TCOD fed. The results are shown in Figure 4.12. Cumulative conversion demonstrates the overall effect of feed addition with each cycle on digestion efficiency. The length of each step in Figure 4.12 adds an implicit quantitative reference point aimed at drawing the time dependency of conversion. A negative effect of solid yeast cell addition on digestion efficiency was observed. This presented two possible scenarios. One scenario was that a portion of the yeast cells was slowly biodegradable and not allowed enough time in a cycle to react completely as previously conceded. Another scenario was that the increase supply of nitrogen introduced by the addition of yeast cells encouraged the growth of the anaerobic microorganisms within the reactor, thus contributing to a rise in the COD baseline.

The COD removal rates observed increased with succession of the cycles, which corresponded increase in initial substrate as shown in Figure 4.13. The noise in this data shown reflected the effect caused by the variability in time intervals during the calculation of the rates. The increase in removal rates of soluble COD with increase in substrate concentration was higher than the increase in solid COD removal rates. The lower removal rates of the solid COD suggested that in the case of an increase in total COD and solid COD fed, the hydrolysis of the yeast cells will be the rate limiting step.

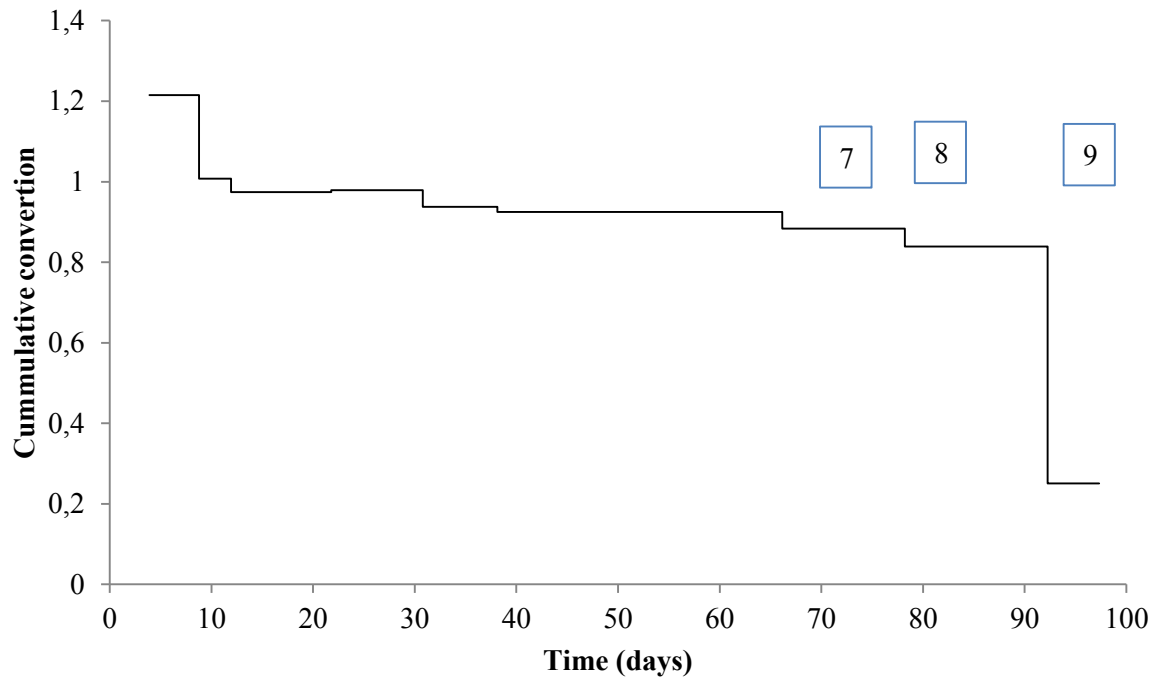


Figure 4.12: Cumulative conversion. Each step represents the cumulative conversion attained at the end of that cycle. Cumulative conversion was calculated as the cumulative TCOD removed by conversion divided by the cumulative TCOD fed.

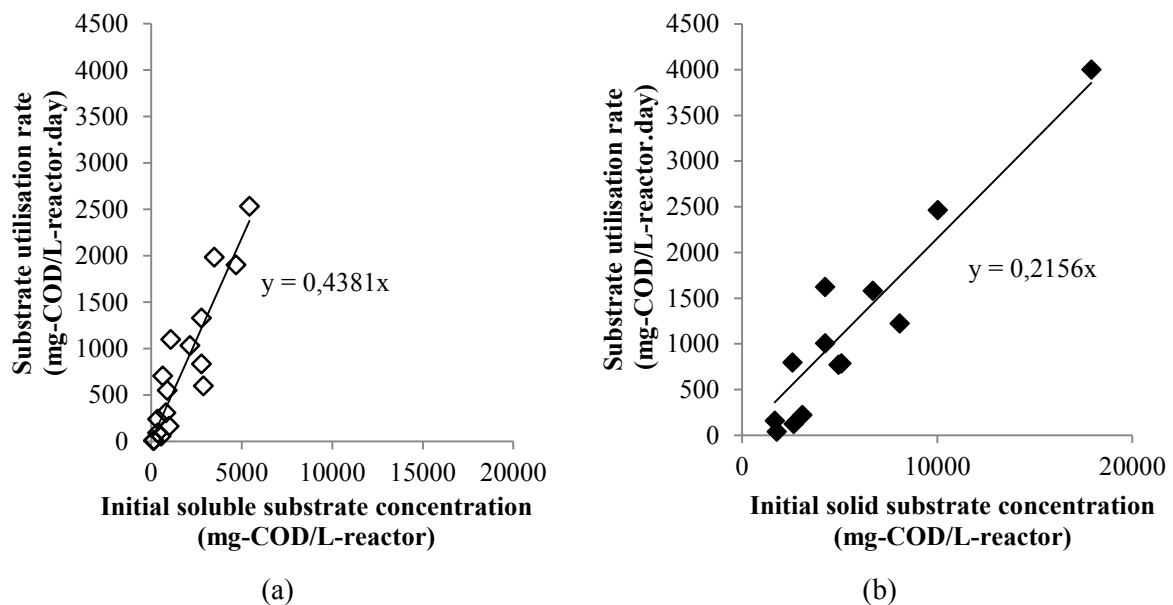


Figure 4.13: Comparison of degradation rates of spent yeast liquor and spent yeast cells (a) Soluble COD utilisation against soluble substrate concentration during batch anaerobic digestion of spent yeast liquor, (b) Solid COD utilisation rate against initial substrate concentration during batch anaerobic digestion of yeast suspension.

4.3.4 Biogas production and methane yield

Biogas production

Figure 4.14 and Figure 4.15 show cumulative gas volumes and cumulative COD removed from the reactor respectively during batch digestion cycle 8. Methane volume produced was calculated in two ways; (1) by using measured methane content directly and, (2) by using the average between the previous methane content measurements and the current methane content measurement. This was done in order to determine whether the change in methane content between the points had a significant effect on cumulative methane volume calculated.

The results in Figure 4.14 showed that there were no significant differences between the two calculations. The similarities between the cumulative gas profiles and the cumulative TCOD removed profiles show that the rate of both biogas and methane was dependent upon the rate of COD removal, which was, sequentially, a function of the concentration of TCOD in the reactor. The cumulative solid COD removed profile reached a plateau after 3 days while the TCOD removed continued to increase at a relatively steady rate till day 5. This showed that methane fermentation of the SCOD hydrolysed from the solid COD during the first 3 days was being consumed. Although methane production was taking place between day 0 and day 2, an increase in methane content of the biogas up to 80% (v/v) was only observed between days 2 and 4. This suggested that degradation of soluble organic matter from hydrolysis of the yeast (from day 2 onwards) had higher methane to carbon dioxide ratio than degradation of the spent liquor (between day 0 and day 2).

The decrease in methane content from day 5 and day 7 supported this premise as when digestion of solid COD had momentarily ceased during this period. After day 7, slight increase in solid COD removal was observed, and resulted in an increase of methane content of the biogas produced. After day 5 the rate of TCOD removal decreased, while the rate of biogas production remained relatively similar till day 7. This corresponded to lower methane content of the biogas, and consequently, methane productivity. Since carbon dioxide does not exert chemical oxygen demand, it was expected that methane production would resemble COD removal more as compared to the total biogas production as observed in Figure 4.14 and Figure 4.15.

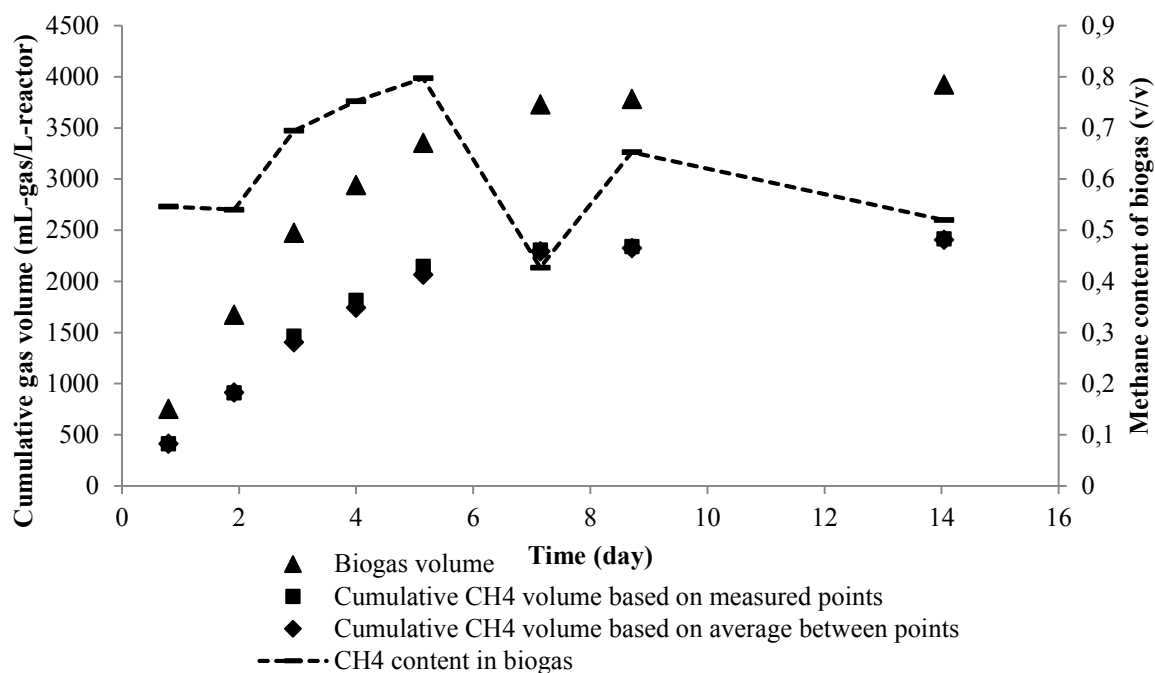


Figure 4.14: Cumulative biogas volume, cumulative methane volume and methane content of biogas collected at each data point during anaerobic digestion of spent yeast suspension (batch cycle 8 of Figure 4.9).

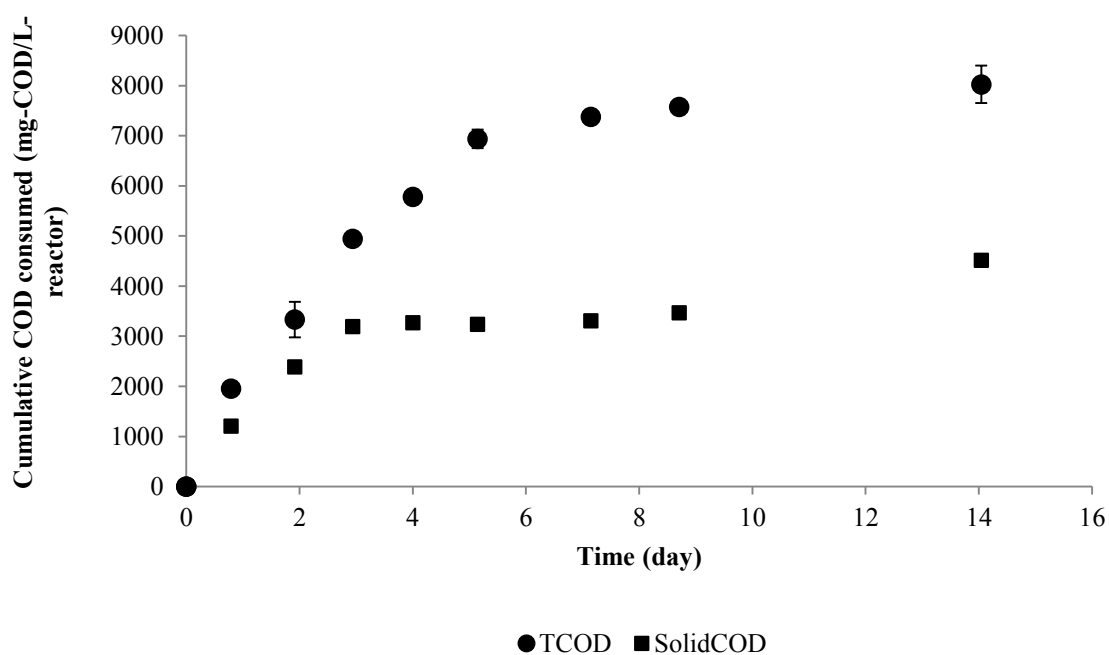


Figure 4.15: Cumulative total oxygen demand (TCOD) and cumulative solid chemical oxygen demand (solid COD) removed during anaerobic digestion of yeast suspension (batch cycle 8 of Figure 4.9).

Methane yield

Cumulative gas yields (Figure 4.16) were calculated as the ratio of cumulative gas volumes (Figure 4.14) to cumulative COD (Figure 4.15). The theoretical methane yield is 0.350 mL-CH₄/mg-COD reacted at standard temperature and pressure, when assuming that all COD reacted was converted to exit the reactor as methane. However, the methane yield observed at the end of the batch digestion was around 0.300 mL-CH₄ /mg-COD reacted. This yield was not only comparable to the theoretical yield, but also to the values observed by other authors. Neira & Jeison (2010) reported methane yield of 0.330 mL-CH₄/mg-COD removed during anaerobic co-digestion of brewer's spent yeast and brewery wastewater using a batch reactor. In another study by Zupančič et al. (2012), methane yields ranging between 0.219 and 0.408 mL-CH₄/mg-COD reacted were reported during anaerobic co-digestion of brewery spent yeast and brewery wastewater using a 12 L pilot upflow sludge blanket reactor (UASBR).

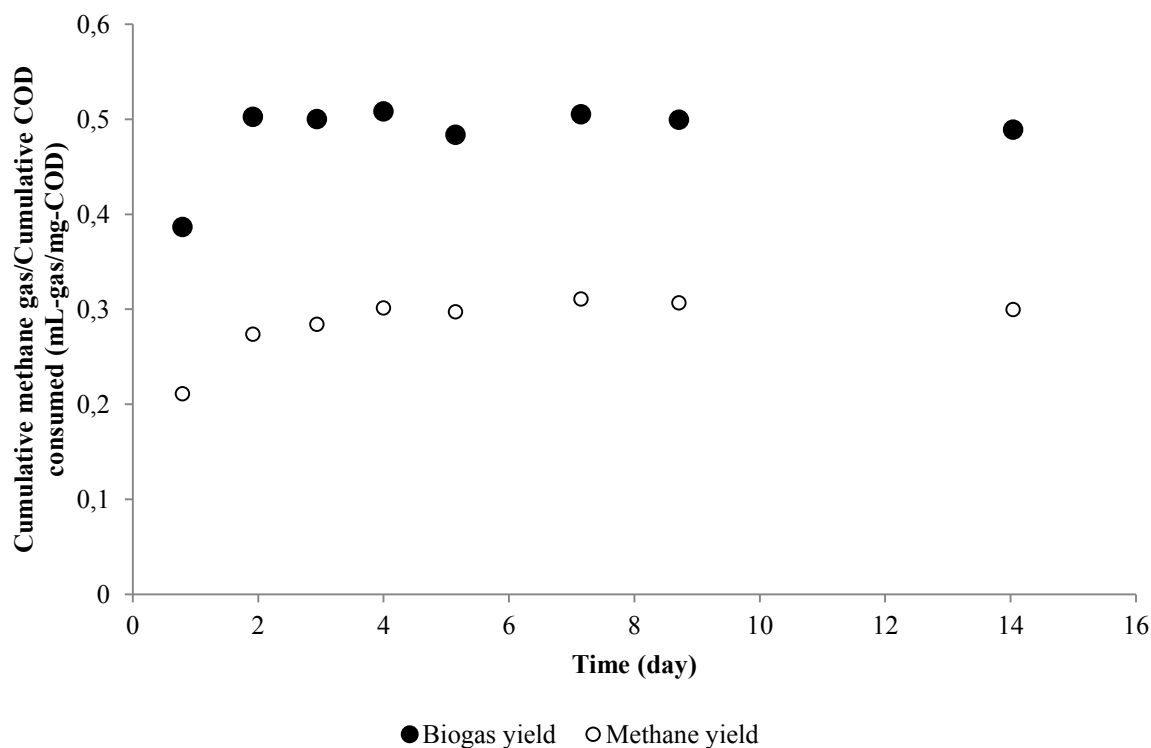


Figure 4.16: Biogas and methane yields based on cumulative gas produced per cumulative COD removed.

4.3.5 Overall COD balance

During anaerobic digestion chemical oxygen demand (COD) is not destroyed, but rearranged. As such, a COD balance can be obtained by considering COD in the feed, COD in the liquid effluent, COD escaping as biogas (methane) and the COD accumulating as sludge (van Lier et al. 2008; Eastman & Ferguson 1981). Digestion of spent yeast slurry during cycles 7 was considered for the COD balance. Once again, a cumulative balance was carried out. At the beginning of cycle 7, 8 and 9, the volumes of yeast slurry fed were 50 mL, 100 mL and 200 mL respectively. For all measurements of COD concentration in the reactor, a 5 mL sample was drawn and analysed. This represented the effluent in the COD balance. Since the reactor was well mixed, the COD concentrations in the reactor were considered the same as the effluent COD. The COD balance was carried out as follows:

$$COD_{feed,n} = C_{reactor,n+1}V_{reactor,n+1} - C_{reactor,n}V_{reactor,n}, \text{ where} \quad \text{Equation 4.1}$$

$$C_{reactor,n+1} > C_{reactor,n}, \text{ and}$$

n is a point of measurement

$$COD_{effluent,n} = C_{reactor,n}V_{effluent,n} \quad \text{Equation 4.2}$$

The COD attributed to the gas was calculated from the measure methane volumes using molar volume of an ideal gas at 1 atm and 25 °C, molar oxygen demand to combust methane to carbon dioxide and water, and the molecular weight of oxygen to determine mass chemical oxygen demand (see Equation 4.3). The COD balance resulted in Equation 4.4. The measured accumulation term was calculated as the total COD in the reactor bulk at any given time minus the total COD in the reactor bulk at the start of the experiment (i.e. at the start of cycle 7) (Equation 4.5).

$$COD_{gas} = V_{CH_4}[L] \times \frac{1 \text{ mol } - CH_4}{22.4 \text{ L } - CH_4} \times \frac{2(\text{mol } - O_2)}{\text{mol } - CH_4} \times \frac{32\text{g } - O_2}{\text{mol } - O_2} \quad \text{Equation 4.3}$$

$$COD_{accumulated,n} = COD_{feed,cumulative} - COD_{out,cumulative}, \quad \text{Equation 4.4}$$

where $COD_{out,cumulative}$ was the sum of $COD_{effluent,cumulative}$

and $COD_{gas,cumulative}$

$$COD_{accumulated,n} = C_{reactor,n}V_{reactor,n} - C_{reactor,0}V_{reactor,0}, \quad \text{Equation 4.5}$$

To close the COD balance, the difference between cumulative COD calculated by Equation 4.4 and cumulative COD calculated by Equation 4.5 would be zero. However, the difference was 14% relative to accumulated COD calculated by Equation 4.4 (i.e. only 86% of the COD which entered the reactor was accounted for). The loss may be attributed to losses due to gas leakages or some aerobic digestion due to the presence of oxygen micro-zones in the reactor.

4.3.6 Mathematical description of brewer's spent yeast degradation kinetics

Anaerobic digestion kinetic models

There are various different components constituting spent yeast as discussed in Section 4.3.1. The wide variety of components making up the substrate presents difficulties in applying mathematical models to describe kinetics of anaerobic digestion. The difficulty lies in the need to fully characterised the feed and measure concentrations of the various components and metabolites during digestion. Further, knowledge the role of various microbial communities and their growth kinetics would need to be specified. This would require microbial community analysis at each of the data points. The high data requirement of using these models limits the practicality of application (Gerber & Span 2008).

Degradation kinetics of solid organics (yeast cells)

The hydrolysis process in anaerobic digestion is partly biological and partly non-biological (Batstone et al. 2002). The non-biological disintegrating of biopolymers may occur by mechanical grinding and digestion by chemical species present such as pH control agents. As such, the rate of hydrolysis is also influenced by other factors such as mechanical shearing, pH and reactor temperature (Batstone et al. 2002; Eastman & Ferguson 1981). The biological hydrolysis are catalysed by extracellular enzymes (Shin & Song 1995; van Lier et al. 2008; Batstone et al. 2002). In a favourable environment (constant temperature and pH), enzymatic reactions are expressed by the Michelis-Menten kinetics (Levenspiel 1999). However, concerns about the need for additional information during mixed culture anaerobic digestion have been raised. Therefore, kinetic expressions based on empirical observations offer the easier way of charactering the rate of hydrolysis.

The general approach followed by many authors in reaction engineering is to test if the concentration of reactants or products changes linearly with time (i.e. zero order with reactant concentration). If this is not true, then a first order kinetic model is fitted. If it does not fit, then a second order model is fitted, and so on. The non-linear nature of the solid COD destruction (Figure 4.5 and Figure 4.10) was apparent. Cycle 8 was chosen to be used to determine evaluate the mathematical models describing the

degradation of solid COD as it was expected that some acclimation of the microbes to the solid substrate had occurred during cycle 7. First order kinetic model approximated the rate of hydrolysis sufficiently well. Eastman & Ferguson (1981) applied the first order kinetic model to describe solubilisation complex particulates from raw primary domestic sludge. This model for hydrolysis was also applied successfully by Veeken & Hamelers (1999) on digestion of whole-wheat bread, leaves, bark, straw, orange peelings, grass and filter paper, and by Angelidaki et al. (1999) in *A comprehensive model of anaerobic bioconversion of complex substrates to biogas*. Further, it was embraced by Batstone et al. (2002) in *The IWA (International Water Association) Anaerobic Digestion Model No 1 (ADM1)*.

Degradation kinetics of soluble organics

Soluble organic compounds in anaerobic digestion are converted through metabolic activities of the various different microorganisms as discussed in Section 2.5.3. The rate of degradation of the soluble compounds is related to the metabolic activity which results in growth of the microorganisms. As such, the microbial specific growth rates are used to describe the degradation kinetics of soluble organic matter (Batstone et al. 2002). Soluble organic compounds are removed from the anaerobic reactor mainly by conversion of acetate to methane and carbon dioxide by acetoclastic methanogens. Acidogenic and acetogenic processes are generally more stable and faster than acetoclastic Methanogenesis. Therefore, ultimately, the acetoclastic methanogenic activity is the main limiting step during anaerobic digestion of soluble compounds. As such, the rate of removal of soluble organic matter can be expressed by one equation which explains the activity of acetoclastic methanogens.

Combined kinetic models

Shin & Song (1995) argued that biochemical reaction kinetics during anaerobic digestion of complex organic matter could be described as two reaction steps: (1) *acidification* (which included hydrolysis) and (2) methanation. The argument was supported by the fact that digester stability depends on VFA concentration, which depends on the rate of VFA removal by methanogens, and the rate of VFA production by *acidification* which is limited by hydrolysis (Shin & Song 1995). During anaerobic digestion of spent yeast supernatant and spent yeast slurry in the current study, each feeding cycle was followed by a sharp drop in pH which indicated rapid accumulation in VFAs. This proved that the rate of soluble organic matter removal was not limited by acidogenic reaction step. This meant that soluble organic compounds were acidified faster than they were removed from the reactor as soon as they became available. In conclusion, the two potentially rate limiting steps were hydrolysis in the case of feed containing the yeast biomass and methanogenesis.

Applying the kinetic models

The mathematical expressions and application of the kinetic models as described above are presented in this subsection. For digestion of yeast biomass, two biodegradable constituents of the total solid substrate (SS) were classified with respect to their degradation kinetics into fast degrading solid

substrate (FDSS) and slowly degradable solid substrate (SDSS). The total solid substrate (SS) and the sludge are constituents of the total reactor solids (TRS). Equation 4.6 and Equation 4.7 explain relationships between the various solid constituents in the reactor. The rate of change in concentration of biodegradable solid matter in the reactor depended mainly on degradation of FDSS. A change of substrate to SDSS was apparent when FDSS ran out. During this period, the rate of change in solid concentration was due to the change in SDSS concentration. The sludge concentration was assumed to be constant as the rate change in biomass COD was insignificant during the period of digestion. The kinetic model for solids degradation was executed according to Equation 4.9 to Equation 4.14.

$$C_{TRS} = C_{SS} + C_{sludge} \quad \text{Equation 4.6}$$

$$C_{SS} = C_{FDSS} + C_{SDSS} \quad \text{Equation 4.7}$$

$$\frac{dC_{SS}}{dt} = \frac{dC_{FDSS}}{dt} + \frac{dC_{SDSS}}{dt} = R_{SS}, \text{ where} \quad \text{Equation 4.8}$$

for $C_{FDSS} > 10 \text{ mg} - \text{COD L}^{-1}$,

$$\frac{dC_{FDSS}}{dt} = -k_{FDSS}C_{FDSS}, \quad \text{Equation 4.9}$$

$$\frac{dC_{SDSS}}{dt} = 0, \quad \text{Equation 4.10}$$

for $C_{FDSS} \leq 10 \text{ mg} - \text{COD L}^{-1}$,

$$\frac{dC_{FDSS}}{dt} = 0 \quad \text{Equation 4.11}$$

$$\frac{dC_{SDSS}}{dt} = -k_{SDSS}C_{SDSS}, \quad \text{Equation 4.12}$$

where k_{FDSS} and k_{SDSS} are the degradation constants for FDSS and SDSS respectively.

for all C_{FDSS}

$$\frac{dC_{sludge}}{dt} = 0 \quad \text{Equation 4.13}$$

$$\frac{dC_{TRS}}{dt} = \frac{dC_{sludge}}{dt} + \frac{dC_{FDSS}}{dt} + \frac{dC_{SDSS}}{dt} \quad \text{Equation 4.14}$$

where k_{FDSS} and k_{SDSS} are the degradation rate constants for $FDSS$ and $SDSS$ respectively.

The kinetic model for soluble organic matter degradation was executed as described by Equation 4.15 to Equation 4.19. The lumped growth constant (k_{lump}) is inclusive of the maximum specific growth rate and the yield of substrate to microorganisms. There was always some residual soluble COD (SCOD) in the reactor.

$$C_{SSCOD} = C_{TRSCOD} - C_{ResSCOD} \quad \text{Equation 4.15}$$

$$\frac{dC_{ResSCOD}}{dt} = 0, \quad \text{Equation 4.16}$$

$$\frac{dC_{SSCOD}}{dt} = -k_{SSCOD}C_{SSCOD} - \frac{dC_{TRS}}{dt}, \quad \text{Equation 4.17}$$

$$\frac{dC_{TRSCOD}}{dt} = \frac{dC_{ResSCOD}}{dt} + \frac{dC_{SSCOD}}{dt}, \text{ where} \quad \text{Equation 4.18}$$

$$k_{SSCOD} = \frac{k_{lump}}{k_s + C_{SSCOD}}, \quad \text{Equation 4.19}$$

k_{lump} is the lumped growth constant, and k_s is Monod's substrate concentration at half the maximum growth rate.

Figure 4.17 and Figure 4.18 shows the result of the model applied to the data sets of cycle 5 and cycle 8 respectively. The kinetic parameters for SCOD degradation were obtained from cycle 5 data sets while the kinetic parameters for solid COD degradation were derived from cycle 8 data sets. The dependency of degradation rate on substrate concentration was very apparent and has been shown in Figure 4.13. However, as the degradation rates presented in Figure 4.13 were calculated from the slopes of straight lines between points, the values of the rates were influenced by the size of the time intervals between

data points. The model expression for rates of degradation rate allow for better interpolation of missing data as well as extrapolation of untested conditions.

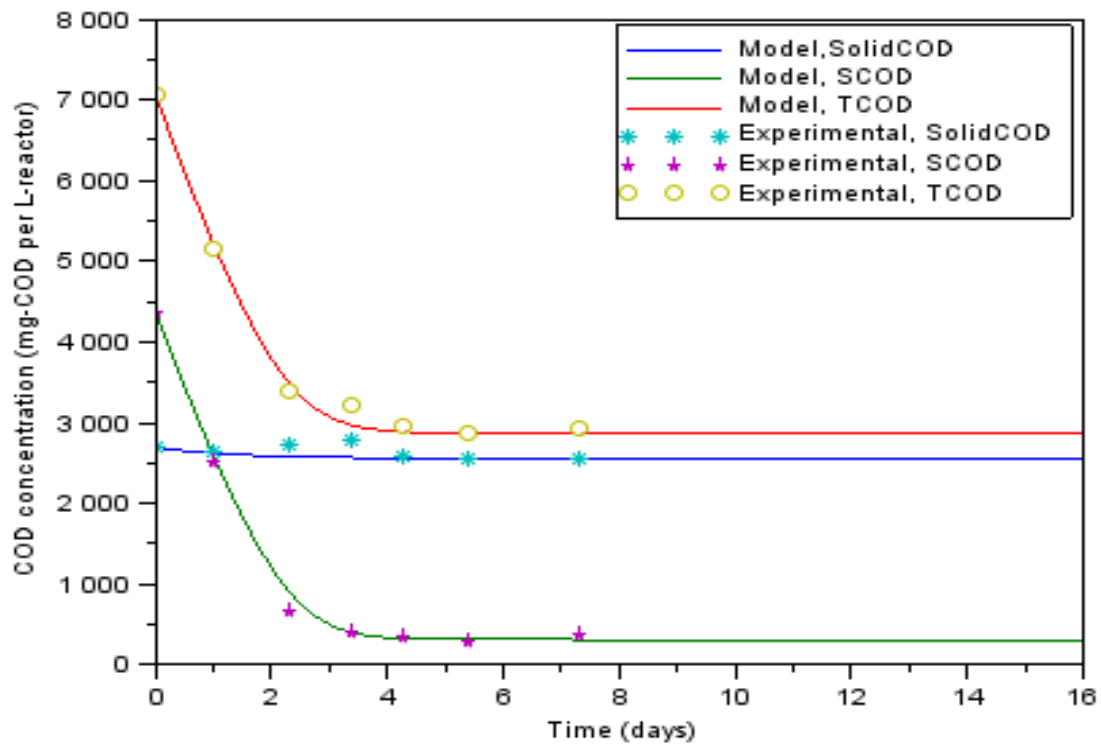


Figure 4.17: Model fit for anaerobic digestion of spent yeast using data set of batch cycle 5.

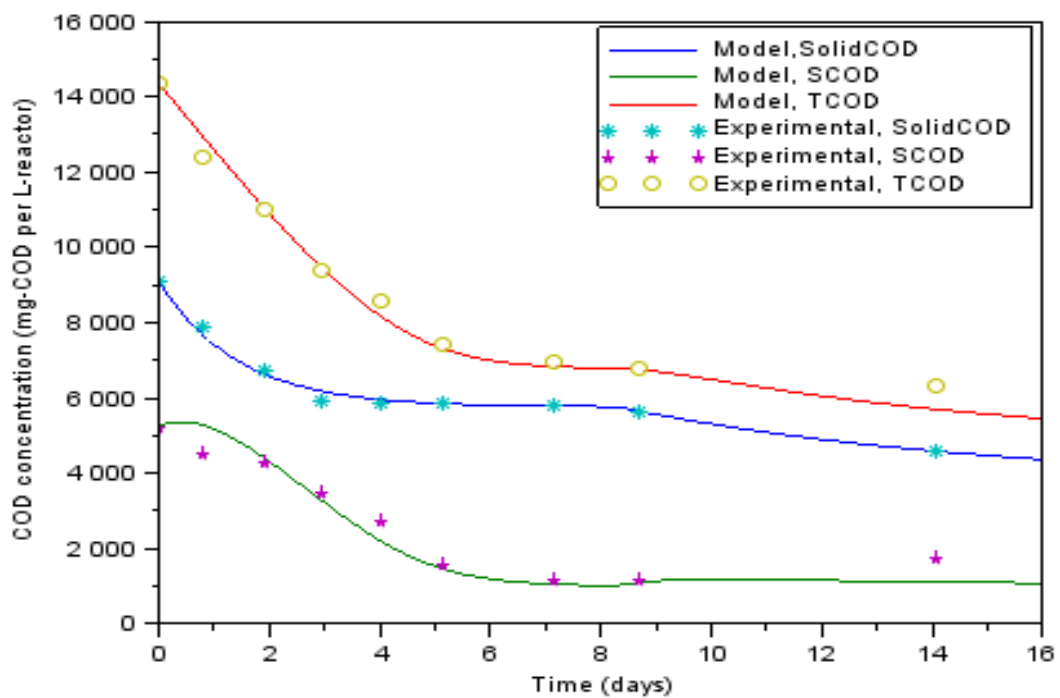


Figure 4.18: Model fit for anaerobic digestion of spent yeast using data set batch cycle 8.

4.3.7 Practical considerations for implementation at SABWTP

This section presents discussions on potential issues and benefits relating to the use of spent yeast as addition feed to the SABWTP.

Acclimation and feeding schedule on pH control

The need for acclimation of microbial systems to new operating conditions is very well studied. In anaerobic digestion, acclimation is mostly related to the acclimation of methanogenic microorganisms as they are most sensitive to pH changes and have low growth rates. In the current study, many failed experiments have shown that sudden increase in feed substrate concentration had negative impact on the system by causing the pH to drop rapidly. This resulted from imbalances of reaction rates in the acid generating steps and acid consuming acetoclastic methanogenesis as discussed in 2.5.4 of the Literature Review chapter. This gave rise to the approach of gradually increasing the organic loading as presented in section 4.3.3. It is thus recommended that the spent yeast suspension or liquor be added in small amounts across a time period in order to allow for methanogenic rates to balance with the increased acidogenic rates. The reduced imbalances between acid generating steps and methanogenesis can, thus, reduce the requirement for addition of pH control agents (caustic addition).

It was also conceded the spent yeast may be used as a pH control agent in itself to some extent, thus reducing dependency on caustic as a control agent. This may occur in several ways. Firstly, the ammonia derived from nitrogen in the spent yeast can be a source of alkalinity by reacting with dissolved carbon dioxide to form bicarbonate as described by Equation 2.1 in section 2.5.4. Secondly, the variability on wastewater concentration entering the anaerobic digestion reactor can be graduated by controlling the amount of yeast fed. This would reduce the imbalances between acidogenic rates and methanogenic rates, thus requiring less caustic addition. The spent yeast would be a good COD graduating agent as it has a much higher COD content than the wastewater, thus, not altering the influent flow significantly. The spent yeast can also be used to lower in the pH in cases where pH goes too high. In these high pH cases, the nitrogen in the spent yeast will not provide alkalinity, but will escape as ammonia according to Figure 2.8 in section 2.5.4.

Solids retention time on solids in the effluent and sludge accumulation

Solid spent yeast cells consist of both readily biodegradable and slowly degradable constituents. Increased solid COD content due to the slow degradation of the yeast cells and possible accumulation of the slowly biodegradable portion in the sludge were of concern. To reduce solid COD in the effluent, the solids retention times would be required. The SABWTP is equipped with an upflow sludge blanket reactor (UASBR) reactor which has properties that allow for retention of solids. As such, the UASBR will enhance the COD removal efficiency by retaining the slowly degrading constituents of the spent

yeast in the reactor for long. Further analyses and discussion on the imprecation of solids retention time and the properties of the UASBR are presented in Chapter 5.

4.4 Conclusions

4.4.1 Anaerobic digestion of brewer's spent grain

The COD reduced during 19 days of anaerobic digestion of spent yeast was the same as that of the control experiment which was fed with inoculum only. Following this poor digestion of spent grain, it was concluded that the inoculum used, from the anaerobic brewery wastewater treatment plant, did not have microbial community capable of producing lignocellulose degrading enzymes. While the use of rumen inoculum would improve digestion of spent grain as suggested by the results of Ezeonu & Okaka (1996), the low conversion after 15 days proved unattractive for the purposed of energy generation in a plant designed primarily for wastewater treatment. Addition of spent grain to an anaerobic brewery wastewater treatment reactor is expected to increase solid accumulation rate in the sludge bed and subsequent overflow of the solids as effluent if the reactor is not de-sludged regularly. The biomethane potential yield of the spent grain were low (76-110 L/k-COD) when compared to the theoretical methane yield of 350 L/kg-COD. Therefore, the additional energy produced from the spent grain is expected to be low, thereby defeating both the goal of reducing organic waste and of increasing energy recovery. Considering the findings of the current studies and the published literature, the brewer's spent grain has not shown much promise as candidate for being used as additional feed to the anaerobic wastewater treatment plant. For these reasons, brewer's spent grain was not considered for further analysis in the current study.

4.4.2 Anaerobic digestion of brewer's spent yeast

Anaerobic digestion of spent yeast in batch cycles provided concentration profiles of total and soluble COD concentrations with time. This allowed for calculation of solid COD concentrations, showing the progression of solids hydrolysis. These profiles indicated simultaneous digestion of both solid COD and SCOD, implying that degradation of the yeast cells (solids) was not inhibited completely by the availability of soluble organic matter. This implied that conversion of the yeast cells in to biogas could be achieved in a continuous system where constant supply of both solid COD and SCOD would be maintained.

In some cases, degradation rates of the spent yeast cells were lower than degradation rates of the yeast liquor. The lower degradation rates of the spent yeast suggest that higher residual COD concentrations may be observed in the SABWTP effluent. However, this is subject to residence time of the spent yeast slurry in the reactor. In particular, the retention time of the solid yeast cells is important to allow for maximum conversion of the solids. Potentially, the effect of the rate-limiting hydrolysis step can be

overcome by altering the solids retention time. In cases where observed rates of soluble COD removal were lower than the rates of solid COD degradation, the cause was either depleted soluble COD or simultaneous increase in soluble COD from hydrolysis.

The digestion profiles also gave insight into the kinetics of the spent yeast anaerobic digestion. Substrate-dependent growth, according to the Monod theory, was applied to the SCOD concentration profiles. First order kinetic theory was applied successfully to the solid COD. Degradation profiles of the solid COD showed a diauxic nature, which resulted in classification of the solid COD into a fast degrading portion and a slowly biodegradable portion during the model implementation.

5 Anaerobic digestion at SABWTP: Current situation and prospective scenarios modeling

5.1 Introduction

Chapter 4 presented the experimental results for anaerobic digestion of brewer's spent grain and spent yeast for the possibility of using as addition feed to the SABMiller wastewater treatment plant (SABWTP) to increase energy output in the form of methane. It was concluded that spent grain was not suitable for use as feed in the SABWTP, while spent yeast showed much promise for this purpose. In this chapter, the feeding of the spent yeast to the SABWTP is considered. Being able to predict the effect of such operational changes is key to running a well-functioning plant that deliver intended results. The effects of feeding the brewer's spent yeast or any operational other changes on performance of the SABWTP are, thus, important to predict and analyse.

In this chapter, the current performance of the SABWTP is presented with the purpose of understanding the operation and highlighting the opportunity for improved methane productivity. As part of understanding the functions of the SABWTP, a detailed description of the upflow sludge blanket reactor (UASBR) in general and in relation to the SABWTP is presented. The understanding of the UASBR served to highlight practical constraints and to inform the model used to predict effect of operational changes on performance.

Improvement scenarios are synthesised and presented with their possible advantages and disadvantages. The possible practical constraints and additional requirements of feeding the brewer's spent yeast and spent yeast liquor to the SABWTP are discussed. A model to predict the performance of the SABWTP in the case of spent yeast or spent liquor addition informed by the experimental results and analysis of anaerobic digestion of spent yeast slurry and spent yeast liquor presented in Section 4.3.6 of Chapter 4, in conjunction with historical SABWTP performance data is presented and implemented. The results of the model were used for creating materials inventory for the environmental impact assessment study presented in Chapter 6.

5.2 Current performance of the SABWTP

The SABWTP treats wastewater with variable concentrations of organic components and flowrates. The most important performance indicator of the plant is the organic effluent concentration, while N and P concentrations affect final effluent quality and need for further processing. While the feed-rate and organic content of the wastewater has been variable, the plant has been able to maintain the effluent concentration under 1000 mg-COD/L. This has been achieved through re-circulation of the wastewater

where one pass through the reactor has not been sufficient. As such, if the organic loading rate is increased, relatively low effluent concentrations can still be achieved, provided the influent volume is not so high as to provide no spare capacity for recycle.

Figure 5.1 shows monthly average influent and effluent concentrations of the wastewater entering and leaving the SABWTP at Newlands brewery as well as the flowrates for the period January 2010 to January 2013. To determine the overall performance of the SABWTP over this time period, cumulative COD fed and removed were considered. This approach reduced the influence of small variations and errors in logging and allowed for a more inclusive analysis of the plant's long term performance. Figure 5.2 and Figure 5.3 represent cumulative volume and cumulative total COD respectively as a function of time, for the influent to and effluent from the SABWTP.

The average influent and effluent concentrations, COD feed, removal and treatment rates, hydraulic retention time (HRT), as well as the design parameters for the SABWTP are presented in Table 5.1. When the rolling averages were considered, the unused capacity with respect to volumetric throughput and organic loading rate were approximately 2030 kL/day and 13 t-COD/day respectively. The practical implications of this unused capacity with regard to the feeding of spent yeast discussed in 5.4.2 below.

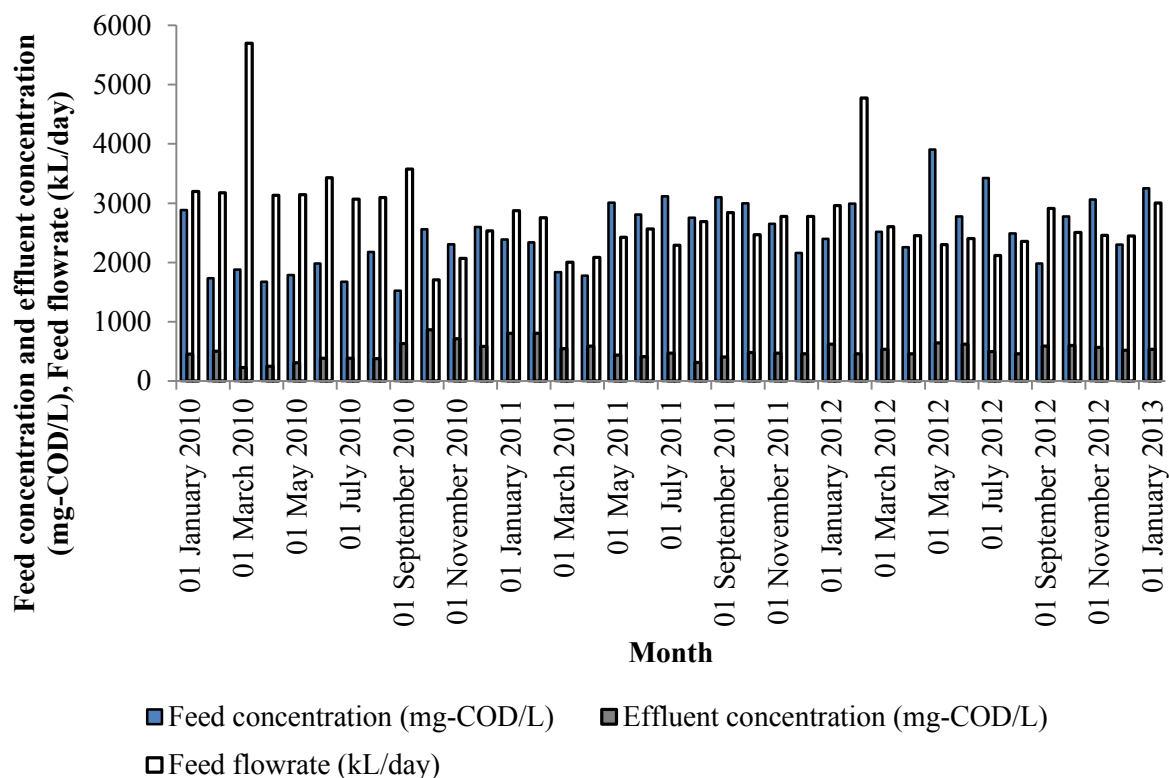


Figure 5.1: Average influent and effluent concentrations on a monthly basis at SABWTP from January 2010 to January 2013.

Table 5.1: Average parameters for SABWTP between January 2010 and January 2013.

Parameter (symbol)	Unit	Average	Rolling average	Design	Unused capacity
Average influent concentration (C_{in})	mg-COD/L	2425 ± 1256	2294	< 4255	~1960
Average influent flowrate (Q_{feed})	kL/day	2255 ^a	2196	< 4230	~2030
Average COD fed	t-COD/day	5 ^b	5	<18	~13
Average effluent concentration (C_{eff})	mg-COD/L	515 ± 284	513	< 1000	-
Average influent flowrate (Q_{feed})	kL/day	1892 ^a	1917	-	-
Average COD in effluent	t-COD/day	0.9 ^b	1	-	-
Hydraulic retention time (HRT)	days	-	1.82	-	-
Average COD reaction rate ($COD_{reacted}$)	mg-COD/L.day	1025	1014	-	-

^a Calculated as the gradients of the respective graphs shown in Figure 5.2.

^b Calculated as the gradients of the respective graphs shown in Figure 5.3.

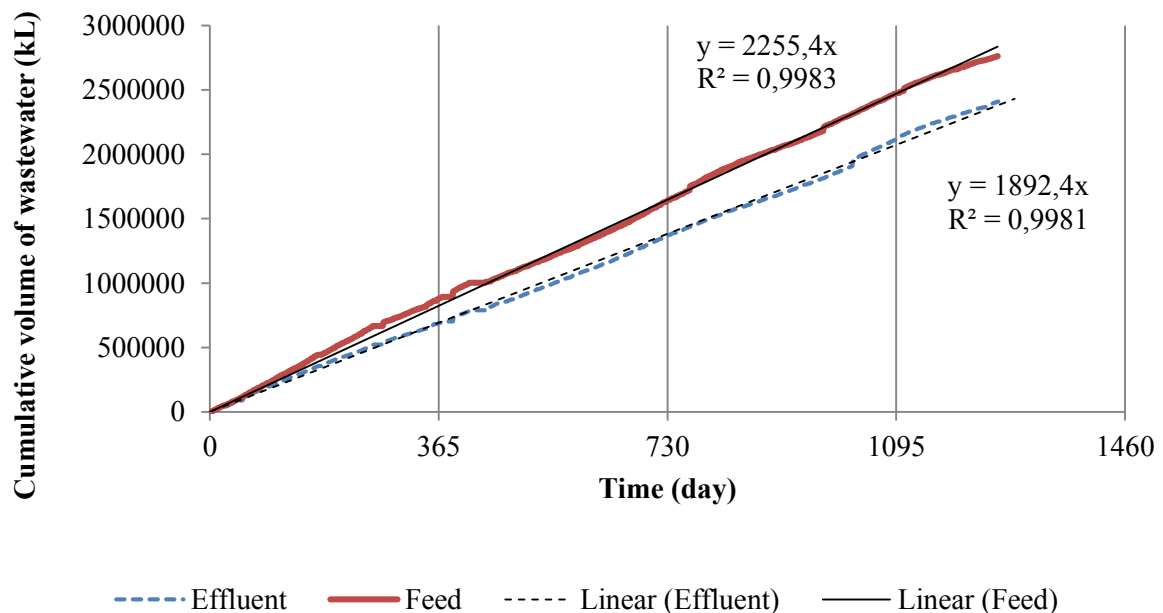


Figure 5.2: Cumulative volume of wastewater feed to and effluent from the SABWTP between January 2010 and January 2013.

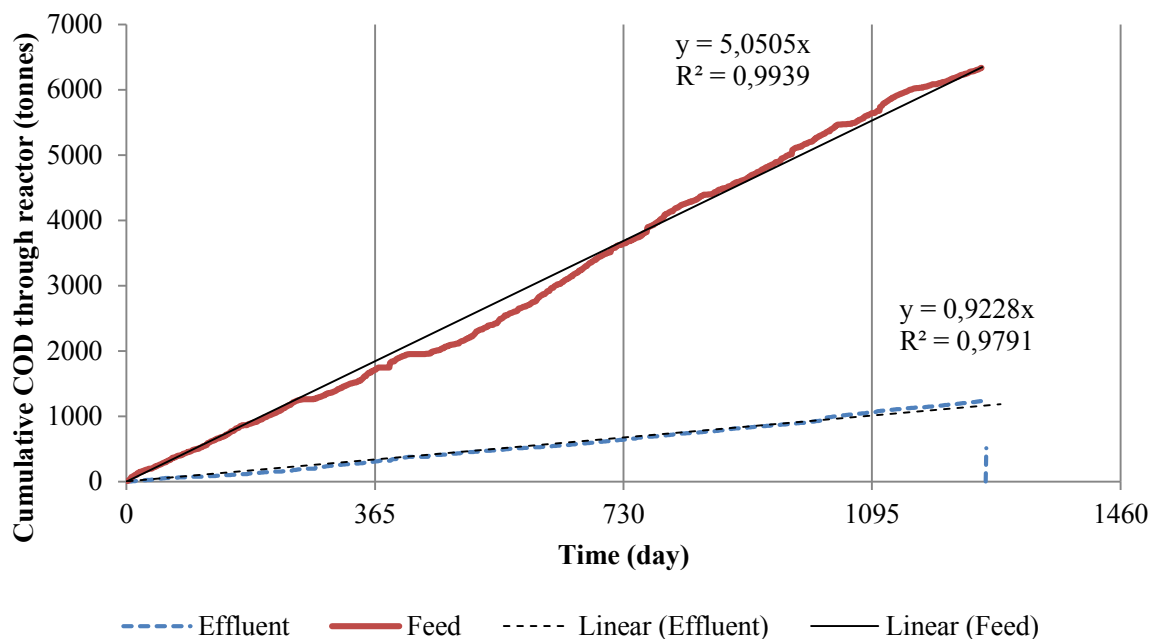


Figure 5.3: Cumulative COD feed to and effluent from the SABWTP between January 2010 and January 2013.

5.3 UASBR: The reactor of choice for long solid retention time (SRT), SABWTP and other large scale anaerobic digestion operations

5.3.1 Introduction to the UASBR and comparison with CSTR

Longer solids retention times (SRT) in anaerobic digestion are important for retaining high viable biomass concentrations and thus the catalytic capacity of a reactor system. Further, insoluble organic matter may need longer to digest due to limitations in the rate of hydrolysis. In a continuous stirred tank reactor (CSTR), SRT depends on the hydraulic volumetric flowrate through the reactor, and thus the hydraulic retention time (HRT) of the reactor. However, solids hold-up time can be isolated from HRT by providing a solid material on which the sludge particle would be trapped for biomass retention, introducing a solid reclaiming unit outside the reactor for biomass recycle, or by allowing the solid particles to settle in the reactor for solids retention (van Lier et al. 2008). The latter strategy is mostly applied in sludge bed reactors. The upflow sludge blanket reactor (UASBR) is one of the most widely used reactors for anaerobic digestion, accounting for almost 90% of anaerobic reactors for treatment of industrial wastewater installed between the years 2002 and 2007 (van Lier et al. 2008). This reactor type is also employed at the SABWTP. The ability of the UASBR to isolate SRT from HRT allows for higher hydraulic throughput (low HRT) with minimal danger of washout viable biomass. Table 5.2

shows a comparison of typical performances between UASBRs and continuous stirred tank reactors (CSTRs).

Table 5.2: Comparison between the upflow sludge blanket reactor (UASBR) and continuous and the stirred tank reactor (CSTR) (Rajeshwari et al. 2000).

Reactor type	Typical loading rates (mg-COD L ⁻¹ day ⁻¹)	HRT (days)
CSTR	25-3000	10-60
UASBR	10000-30000	0.5-7

5.3.2 UASBR process description

Physical structure and hydrodynamics

Figure 5.4 shows a schematic representation of the UASBR. The UASBR is fed from the bottom in the upward direction through a sludge bed. The sludge bed consists of settleable biomass flocs and occupies about half of the reactor volume. The influent is treated within the sludge bed by microbial conversion of biodegradable components into biogas and biomass (van Lier et al. 2008). Some solid components are trapped and slowed down by the sludge bed, thus, allowing for longer contact time. The untrapped solids flow out of the top of the reactor with the liquid while the gas produced bubbles into gas collection domes (van Lier et al. 2008).

The clarifying zone of the UASBR (above the sludge bed) is a gas-liquid-solids separations (GLSS) system that provides a liquid-gas separation surface and a gas collection dome. It also serves as a physical barrier which prevents sludge particles from exiting the reactor. Further it reduces turbulence in the upper compartment of the reactor which enhances settling of the trapped sludge particles and the particles in the sludge bed (van Lier et al. 2008). According to van Lier et al. (2008), two main factors are important to consider for successful retention of sludge in the UASBR: the formation of flocs which make up the sludge, and the design of the internal GLSS system. The formation of flocs enhances settling characteristics of the sludge (i.e. bigger flocs settle better) (van Lier et al. 2008; Richardson et al. 2002) and therefore improves solids retention.

Wu & Hickey (1997) studied the hydrodynamics of the UASBR using tracer studies. They described the UASBR by a compartment model in which the sludge bed was represented by a CSTR with dead volume and the clarification zone represented by a plug-flow reactor (PFR). In a different tracer study, Bayoumi (2007) determined that the CSTR model with dead zones and bypass described the overall hydrodynamics of the UASBR. Singhal et al. (1998) also found that results the results of a tracer study on a UASBR mimic a non-ideal CSTR. According to Wu & Hickey (1997), the concentration of viable and biomass in the clarification zone is very low compared to the sludge bed, such that reactions

occurring in this zone can be ignored. As such, the performance of the UASBR can be described reasonably well by considering only the sludge bed, which is represented by CSTR model. This model is, however, sufficient only when considering the influent and effluent organic concentrations and not the dynamic characteristics of the USBR system.

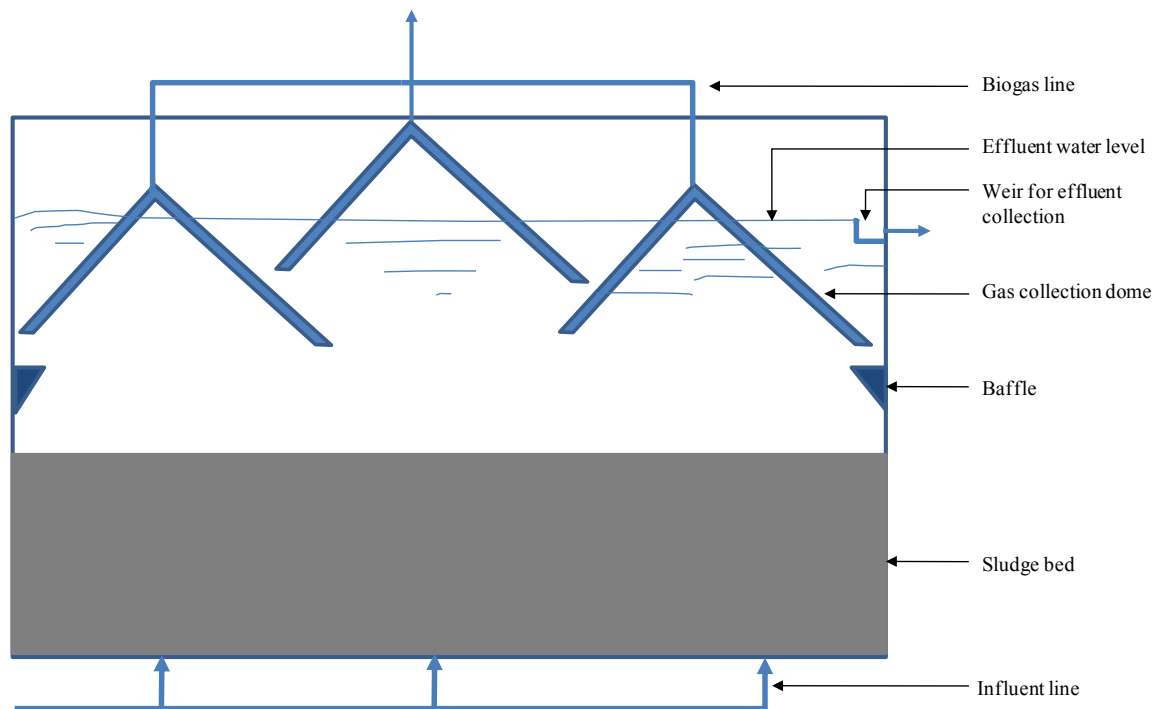


Figure 5.4: Schematic representation of the upflow sludge blanket reactor (UASBR).

Reactor operation

The organic loading rate (OLR) of the reactor is one of the most important parameters. Too high an OLR can cause digester failure due to acid accumulation. To prevent acid build-up, OLR has to be less than or equal to the rate of conversion of the organic acids to biogas by acetoclastic methanogenic (i.e. OLR is limited by the reactor kinetics). The OLR may be constrained further by the desirable organic effluent concentration (C_{eff}). If lower C_{eff} is desired for a given reactor volume and feed concentration (C_{feed}), then Q_{feed} has to be decreased (i.e. increase retention time), which will result in lower OLR. For a given organic concentration of wastewater (C_{feed}) and reactor volume ($V_{reactor}$), the volumetric feed rate (Q_{feed}) can be adjusted to control OLR according to Equation 5.1. Formation of aggregates or flocs is governed mainly by the upward velocity of the liquid (u_{upL}) in the reactor. The volumetric feed rate (Q_{feed}) is directly related to the upward velocity (u_{upL}) of the liquid in a specified reactor as shown by Equation 5.2.

$$OLR = \frac{C_{feed} \times Q_{feed}}{V_{reactor}}$$

Equation 5.1

$$u_{upL} = \frac{Q_{feed}}{A},$$

Equation 5.2

where A is the cross sectional area of the reactor

The upward flow velocity applicable has to be such that:

- the turbulence caused by fluid flow does not cause the flocs to disintegrate, thus reducing the settleability of the sludge, and
- the upward velocity on the flocs caused by fluid flow does not result in washout out of the sludge.

As such, there exists a maximum allowable upward velocity ($u_{upL,max}$), above which significant sludge disintegration or sludge washout occurs. The maximum allowable upward velocity corresponds to a minimum cross sectional area (A_{min}) which can be calculated by replacing A and u_{pL} by A_{min} and $u_{pL,max}$ respectively in Equation 5.2. The volume of the reactor ($V_{reactor}$) is determined by the desired hydraulic retention time (HRT). The required height (H) can be calculated by rearranging Equation 5.4.

$$V_{reactor} = HRT \times Q_{feed}$$

Equation 5.3

$$HRT = \frac{A_{min} \times H}{Q_{feed}}$$

Equation 5.4

5.4 Proposed improvement options and practical considerations

5.4.1 Processing scenarios, their advantages and disadvantages

Anaerobic digestion of the spent yeast was selected as the focal point for process synthesis considerations for the increased use of capacity of the anaerobic digester to facilitate increased energy generation for the plant. The degradation rates and effluent COD concentration were selected as key considerations for evaluating the performance of spent yeast anaerobic digestion. Three operational scenarios were considered, based on the prospective advantages listed below.

- Option 1: In this scenario, the current operation of the SABWTP is considered. This provides the base case scenario.
- Option 2: In this scenario, the effect of feeding spent yeast slurry into the wastewater treatment plant was considered. This option provided the advantage of a higher COD loading with potential for higher methane productivity. Potential negative consequences of this scenario include increased COD, N and P content of the SABWTP effluent and the need for a replacement source of protein for animal feed, which is currently provided by the spent yeast.
- Option 3: In this scenario, only the spent yeast supernatant is digested. This option provided the opportunity of avoiding possible retardation of the anaerobic digestion process caused by

lower rates of hydrolysis while providing some additional COD with minimal negative effect on the efficiency of the wastewater treatment plant. In addition, the possible advantage reducing the cost of transporting the larger volumes of the yeast slurry through its dewatering to a yeast paste was considered.

5.4.2 Effect of additional feed on the operation of the SABWTP

Residual COD in the effluent and methane generation are the most important performance parameters when considering the addition of spent yeast or spent yeast liquor with the aim to exploit the biogas generation capacity of the anaerobic digester. This is mainly influenced by the rate of degradation of the additional feed and the solid and hydraulic retention times. The retention time depends on the influent flowrate and recycle ratio, and is limited by the design flowrate as discussed in Section 5.3.2.

The SABWTP was designed for maximum influent wastewater concentration of 4255 mg-COD/L and influent rate of 4230 kL/day to be treated to effluent concentrations lower than 1000 mg-COD/L. Neira & Jeison (2010) and Zupančič et al. (2012) determined that the average volume ratio of spent yeast to wastewater produced in a brewery was the 0.7% (v/v). The COD contribution of spent yeast to the spent yeast and wastewater mixture at 0.7% (v/v) spent yeast was 35% (Neira & Jeison 2010; Zupančič et al. 2012). Therefore, while the volume contribution of spent yeast to the mixed feed of spent yeast and wastewater would be negligible (increased by 0.7%), and thus not affecting volumetric feed rate, the COD contribution of the spent yeast would increase the inlet concentrations significantly (by up to 35%). However, the spent yeast COD contribution is not high enough to exceed the design capacity of 4255 mg-COD/L in the combined influent. The negligible volume contribution by the spent yeast allows for the required recirculation of the wastewater in order to achieve the desired final effluent concentration. Based on the ability of the UASBR to retain solids as described previously (Section 5.3.2), it could be assumed that the solids concentration in the effluent from the reactor processing spent yeast would be negligible.

5.4.3 Spent yeast dewatering: method and equipment selection for Option 3

Solid-liquid separation methods

Separation of cell biomass from liquid growth medium is a common step of bioprocesses (Ladisch 2001; Grima et al. 2003; Foley 2006). Solid-liquid separations are primarily done by centrifugation, sedimentation or filtration processes (Ladisch 2001; Richardson et al. 2002). The choice of the type of separation process and equipment used is facilitated by the goal of the separation (Ladisch 2001), the nature of suspension (Atkinson & Mavituna, 1983; Ladisch, 2001; Pickering, 2005; Robinson & Harrison, 2001) as well as the time and energy expenditure allowable for a particular process. In large-scale operations a combination of these processes may be used (Pickering 2005; Ladisch 2001).

Centrifugation methods are not economical for low value, high volume products due to their energy requirement, while sedimentation processes results in dilute sludge. Further, sedimentation methods are too slow to be economical and often require addition of flocculants to improve settleability (Grima et al. 2003; Watt 1965). As such, these methods were deemed not suitable for the purpose of dewatering spent yeast for maximum recovery of the liquor with minimum energy expenditure and addition of other materials.

Choice of filtration equipment

Filter presses and rotary vacuum filters are generally used to dewater yeast slurries (Pickering 2005). Filter presses are batch operations whereas rotary vacuum filters allow for both continuous and batch processing of suspensions (Pickering 2005; Ladisch 2001). Higher throughput per surface area are achievable for continuous operations as compared to batch filtration (Ladisch 2001; Richardson et al. 2002), hence the rotary vacuum filter may be favoured over a filter press. Further, according Richardson et al. (2002), filtration equipment cost is closely related to the filtering area, which supports this choice. Therefore, the flexibility of operating a rotary vacuum filter is much desired in light of the intermittent supply of spent yeast slurry from the batch or semi-batch brewing process. Further support of this choice was that the rotary drum vacuum filter is reported to be the most widely used filtration equipment (Watt 1965; Sinnott 1999; Richardson et al. 2002; Pickering 2005).

Separation efficiency

Typical yeast biomass concentration during production of lager is 1.7 g/L to 2.3 g/L of final product. According to Ferreira et al. (2010), brewer's spent yeast typically has 10% to 14% dry weight solids and may retain about 1.5 to 2.5% of the total beer production as spent yeast liquor. Similar values (solids content of 10 dry wt% and beer losses of 1.5% to 3% of total volume of produced beer) were presented by Fillaudeau et al. (2006). Fillaudeau et al. (2006) and Ambrosi et al. (2014) reported that brewery spent yeast can be processed at about 17 L/h.m² and 20 L/h.m² to a concentration of up to 20 wt%. Therefore, the suspension was concentrated from 10% to 20% on a dry weight basis, resulting in half the total volume the spent yeast suspension being recovered of filtrate.

5.5 Modelling of the SABWTP performance for the current and prospective scenarios

5.5.1 Spent yeast liquor and wastewater

Spent yeast liquor can be handled similarly to wastewater as, in both cases, the COD existed dominantly as soluble COD. In Figure 5.6, the degradation rate of dilute spent yeast was predicted as a function of the organic loading rate using the kinetic model and rate constants determined in Section 4.3.6, and the degradation of wastewater based on the monthly average data. The effluent COD values for the spent yeast liquor were generated as random numbers between 200 mg-COD/L-reactor.day and 700 mg-COD/L-reactor.day in *Microsoft excel*, based on the assumption that the liquid can be circulated in the plant for sufficiently long to reach the desired final effluent concentration. The steady-state concentrations used to determine the reduction rates (data points) were determined by letting the accumulation term in the material balance equal zero (see Equation 5.8 below). The accumulation term was squared to stabilise the solution before setting to zero. Figure 5.5 shows a schematic description of the algorithm followed when calculating reaction rates as functions of OLR to produce the graph of spent yeast liquor digestion in Figure 5.6. Figure 5.6 shows that the degradation rates for spent yeast liquor and the wastewater were similar and strong functions of organic loading rate. In further modelling of the SABWTP, it was thus, assumed that the kinetic description of the degradation of soluble COD in the spent yeast slurry presented in Section 4.3.6 is applicable to the treatment of the wastewater.

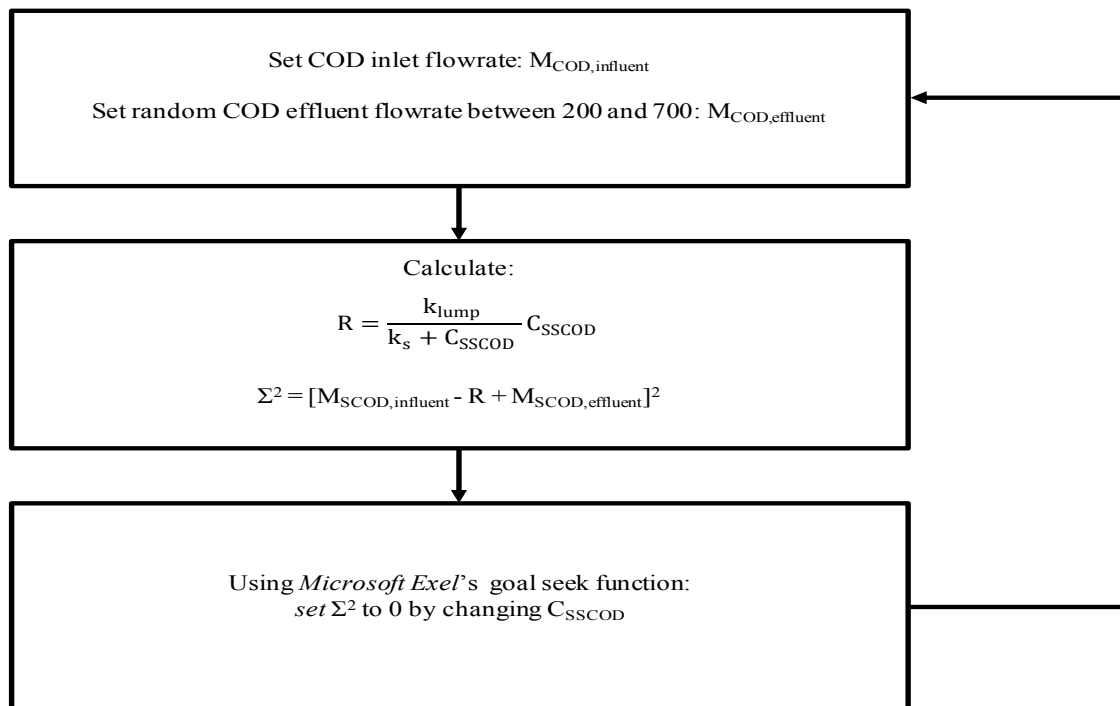


Figure 5.5: Algorithm for calculating reaction rates (R) at given organic loading rates (OLR or $M_{\text{COD,influent}}$). $M_{\text{SCOD,influent}}$ and $M_{\text{SCOD,effluent}}$ have units of mg-COD/L-reactor.day; C_{SSCOD} is the concentration in the reactor with units of mg-COD/L-reactor; Σ is the accumulation term of the material balance expression.

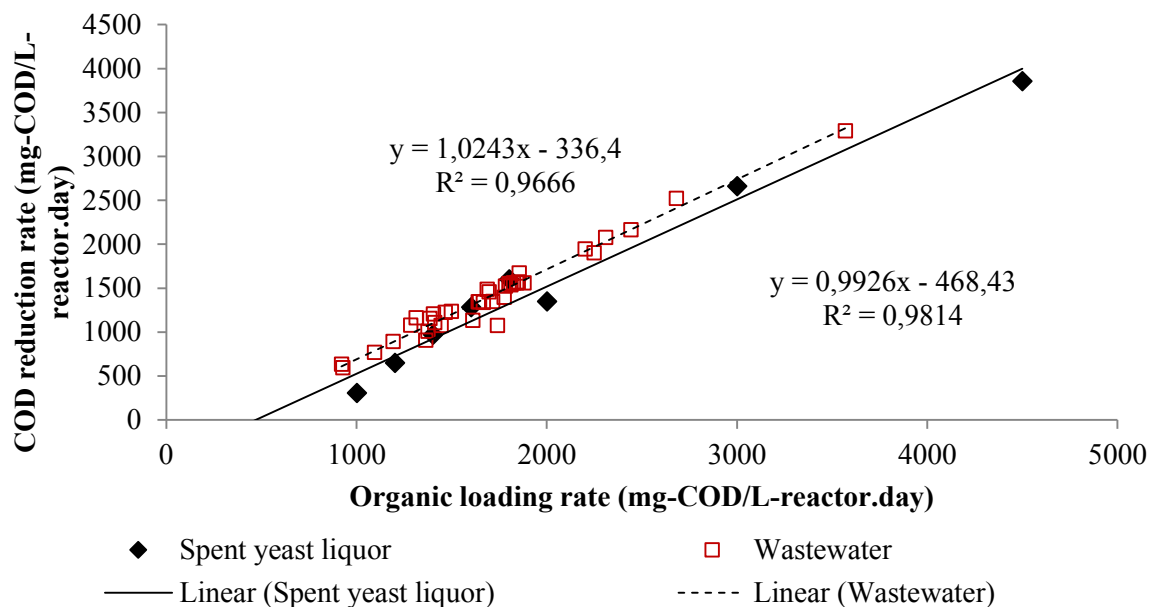


Figure 5.6: Relationship between COD reduction rate and organic loading rate for brewery wastewater and spent yeast liquor.

5.5.2 Model parameters and results

Based on the above discussion, the SABWTP UASBR was modelled as a CSTR with the useful volume equivalent to the sludge bed. The model was implemented according to Equation 5.5 through Equation 5.9. Equation 5.5 and Equation 5.7 describe the rate of hydrolysis of the biodegradable solids (R_{solids}) and degradation rate of soluble organic matter (R_{soluble}) according to the models discussed in Section 4.3.6. Since the degradation of soluble organic matter in the spent yeast liquor is similar to degradation of the brewery wastewater, as demonstrated by Figure 5.6, the major difference in SABWTP performance between addition of spent yeast suspension (Option 2) and spent yeast liquor (Option 3) depends on the degradation of the yeast cells. It has also been shown that once the yeast cells have been hydrolysed, the rate of COD consumption of the resulting soluble organic matter is the essentially the same as the rate of consumption of the soluble organic matter in the yeast liquor. As such, the degradation rate of all SCOD can be expressed with a single rate equation, represented by Equation 5.7. Equation 5.6 describes the material balance of the solids assuming negligible solids concentration in the effluent. Hence the exit flow term has been left out from the expression. Equation 5.8 describes the material balance of soluble organic matter, where the reactor concentration is the same as the effluent concentration (Equation 5.9) according to CSTR design.

The model results were reported at steady state conditions as shown in Table 5.1. The organic conversion rate for Option 2 (1732 mg-COD/L.day) was greater than that of Option 1 (990 mg-COD/L.day) and Option 3 (1097 mg-COD/L.day). This was expected since COD degradation rates

increase with increasing OLR within the un-inhibitory range as demonstrated by Figure 5.6. The effluent COD concentration of the Option 2 was greater than that of Option 1 and Option 3. This suggested that, while the increased OLR and influent concentration increases the reactor efficiency in terms of degradation rates, it reduced the wastewater treatment efficiency by increasing the effluent concentration.

Process		
Rate of solids hydrolysis (R_{solid})	$R_{solid} = k_{solid}C_{solid}$, where k_{solid} [day ⁻¹] is the hydrolysis rate constant, C_{solid} [mg – COD/L] is the solids concentration in reactor	Equation 5.5
Solid COD balance	$\frac{dC_{solid}}{dt} = -R_{solid} + \frac{1}{HRT}C_{solid,in}$, where $C_{solid,in}$ [mg – COD/L] is the influent solids concentration	Equation 5.6
Rate of soluble COD degradation ($R_{soluble}$)	$R_{soluble} = \frac{k_{lump}}{k_s + C_{soluble}}C_{soluble}$, where k_{lump} [day ⁻¹] is the hydrolysis rate constant, k_s [mg – COD/L] is the half velocity constant, $C_{soluble}$ [mg – COD/L] is the soluble concentration in reactor	Equation 5.7
Soluble COD balance	$\frac{dC_{soluble}}{dt} = -R_{soluble} + R_{solid} + \frac{1}{HRT}(C_{soluble,in} - C_{soluble,out})$, with $C_{soluble,out} = C_{soluble}$	Equation 5.8 Equation 5.9
Rate of CH ₄ production	$R_{CH_4} = Y_{CH_4/COD} \times R_{soluble}$	Equation 5.10

Table 5.3: Model inputs and results for SABWTP prospects based on average values, assuming that the amount of solids escaping in the effluent is negligible and that the hydrodynamics in the sludge bed mimic those of a stirred tank reactor.

	Parameter	Unit	Option 1	Option 2	Option 3
	HRT	day	1.8	1.8	1.8
Influent	SolidCOD	mg-COD/L	0	955	0
	SCOD	mg-COD/L	2295	2904	2600
	TCOD	mg-COD/L	2295	3859	2600
Effluent	SolidCOD	mg-COD/L	0	0	0
	SCOD	mg-COD/L	513	741	625
	TCOD	mg-COD/L	513	741	625
Total COD	mg-COD/L.day	990	1732	1097	
Conversion	COD fed/ COD reacted		0.78	0.81	0.76

5.6 Conclusions

Based on the discussions presented in this chapter, the following conclusions are made. The SABWTP operated at an average capacity below its design capacity. The minimum upflow velocity was maintained by using a recycle stream which can also serve to increase the digestion efficiency in cases where the maximum organic content (1000 mg/L) of the effluent is exceeded. The volumetric ratios of spent yeast suspension to wastewater produce from the brewery is small (0.7%). As such, the increase in volume fed to the SABWTP upon feeding the spent yeast or spent yeast liquor would not affect the volumetric capacity of the plant that allows for recirculation in order to meet the specified effluent concentration. The CSTR model can describe the hydrodynamics of the sludge bed zone, and therefore, can also describe the UASBR reasonably well considering negligible active biomass concentration in the clarification zone. However, the model does not consider the effect of changes in feed characteristics on the granulation of the sludge which may be important in UASBR operation. In addition, UASBRs, such as that at SABWTP, have the ability to retain solids. Thus, the addition of spent yeast in suspension is not expected to have severe effect on the plant's efficiency as demonstrated by the model results in

Table 5.3. In conclusion, the SABWTP has sufficient infrastructural and functional capacity to handle the addition of both spent yeast suspension and spent yeast liquor without significantly compromising its wastewater treatment capabilities. The model results showed increased degradation rates and effluent concentration with increased OLR and influent concentration (comparing Option 2 compared to Option 1 and Option 3). As such, increasing influent concentration and OLR has a positive effect on degradation rates and therefore methane productivity, but a negative effect on the treatment function of the anaerobic digester with respect to reducing organic content of the effluent.

6 Environmental impact assessment

6.1 Introduction

Increasingly, factories producing effluent water rich in organic matter are commissioning on-site anaerobic digestion wastewater treatment plants. These have the ability to both, reduce the contamination levels of effluent water before discharge and to generate energy in the form of methane. The balance between the two functions of anaerobic digestion in terms of environmental footprint is not straight forward. Prioritising one of these functions may have adverse effects on the plant's ability to perform the other function. Further, the effect of the operational changes at on-site plant may also have some effects on the efficiency of the further treatment units (usually municipal treatment works), thereby affecting the environmental efficiency of the wastewater management system as whole. The current study presents the development of methodology to model changes in environmental impacts of prioritising energy production, as well as the results of the model using SABWTP as a case study and LCA as a framework.

In Chapter 4 and Chapter 5, brewer's spent yeast was highlighted as a promising additional feedstock for the SABWTP to augment the production of biogas for steam generation, providing potential to reduce the overall energy demand of the brewery and improve its environmental footprint. Two scenarios for processing the spent yeast were considered: 1) the entire spent yeast stream (spent yeast biomass plus associated beer or liquor) fed to the SABWTP and, 2) the spent yeast biomass filtered and the filtrate or supernatant fed to the SABWTP. To analyse the effect of these proposed changes, these strategies were compared with the operation which feeds only wastewater to the SABWTP thereby producing three processing options for environmental impact analysis as described in Section 5.4 and Figure 6.1, Figure 6.2 and Figure 6.3 in this chapter. In this chapter, analyses are presented to determine the environmental benefit of these proposed options in comparison to the status quo, using the SABMiller's Newlands Brewery as a case study. Further, analyses to profile the major sources of environmental impact associated with brewery wastewater management systems, thereby highlighting target areas for improvement of environmental efficiency are also presented.

Aspects of the life cycle assessment (LCA) framework were followed to assess the environmental benefit. The chapter begins with the definition of the goal and scope of the study which include description of the three processing options, description of the functional unit (unit basis for comparison) and system boundaries with supporting reasons for the choices. Inventory data relating to materials and energy consumption associated with each scenario are presented and analysed comparatively. This is followed by environmental assessment and comparison of the proposed options done using life cycle assessment criteria. The computer software, SimaProTM is used.

6.2 Goal definition and scope of the study

6.2.1 Goal definition

The main goal of this LCA, in general, is to assess the effect of prioritising the energy generation function of on-site anaerobic digestion wastewater treatment plants on the environmental performance of the wastewater management system; and, in particular, to analyse the effect of adding spent yeast to the SABWTP on the environmental performance of the brewery wastewater management system. Therefore, the results of this assessment are intended to aid the decision-making process for selecting the best of the three options by comparing their environmental impacts. The audience of the work are decision makers at SABMiller Newlands Brewery. However, aspects relating to methodology and data provided in this study may be applicable to other studies and thus find audience in other interested parties. Other factors which may influence the decision-making, such as economic factors, are not included in this assessment. Further discussion on the scope and choice of the system boundary is presented in the next section (Section 6.2.2).

6.2.2 Scope of the study

The main system evaluated includes the SABWTP, boiler, transportation and the spent yeast filtration where applicable. These are the process units affected by the proposed changes within the wastewater management system. For the purposes of comparing environmental footprints of the proposed downstream process options it is enough to consider only what happens after the beer manufacturing process. The steam produced by the biogas boiler is treated as energy output of the system. The energy currency of the electric boiler is electricity; hence energy produced by the biogas boiler was considered as reduction in electricity-associated environmental footprint and addition of the environmental footprint of the biogas steam production.

System boundary

The system boundary for this study encompasses the SABWTP, the Athlone wastewater treatment works (AWTW), Boiler, and the Transportation of the spent yeast as illustrated by Figure 6.1, Figure 6.2 and Figure 6.3 representing Option 1, Option 2 and Option 3 respectively, where:

- Option 1 is the current operation of the SABWTP and associated and further treatment units,
- Option 2 involves feeding of the entire spent yeast slurry produced by the brewery into the SABWTP, and
- Option 3 includes filtration of the spent yeast and feeding of the filtrate into the SABWTP.

The main goal of this study was to improve the environmental performance of the wastewater management system using the SABWTP as a case study, with the proposed strategies (Option 2 and Option 3) as improvement cases. It aimed to understand the effect of the changes in the performance of

the on-site anaerobic digester on the performance of further treatment units and the overall environmental performance of the waste and wastewater management system. The study, therefore, considered, at first, only the units that are involved in waste disposal (sending spent yeast to farm) and wastewater treatment system and its by products (SABWTP, AWTW, Boiler and associate emissions). The system boundary was later expanded to consider the effect of replacing spent yeast at the farm (see Section 6.7), thereby considering the holistic effect of the proposed strategies.

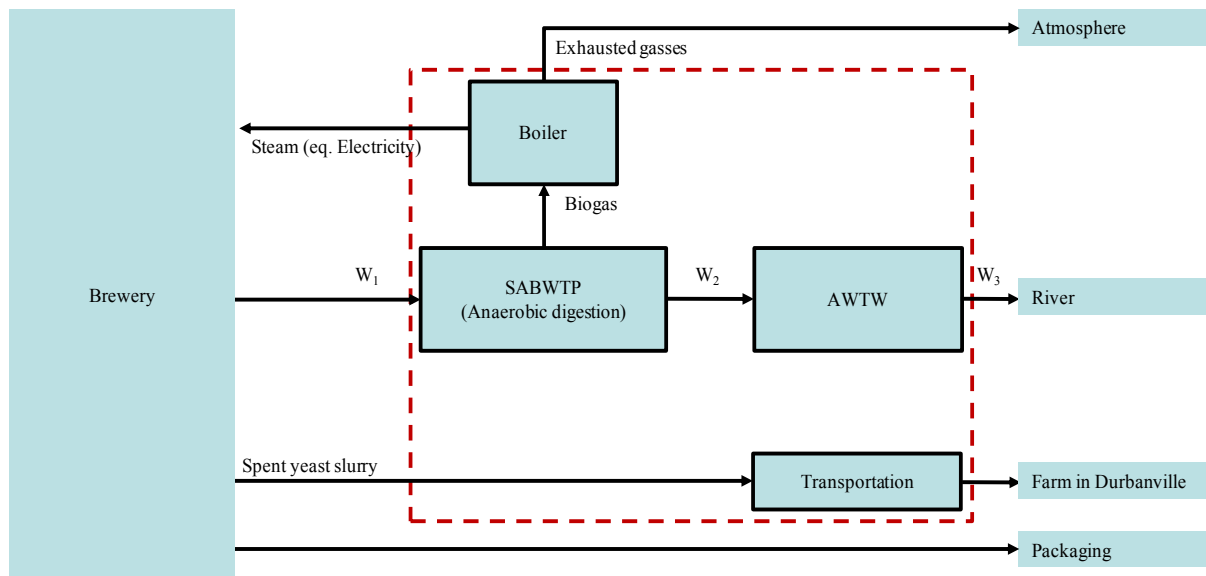


Figure 6.1: Illustration of the system boundary for Option 1 (base case scenario with spent yeast transported to a farm in Durbanville).

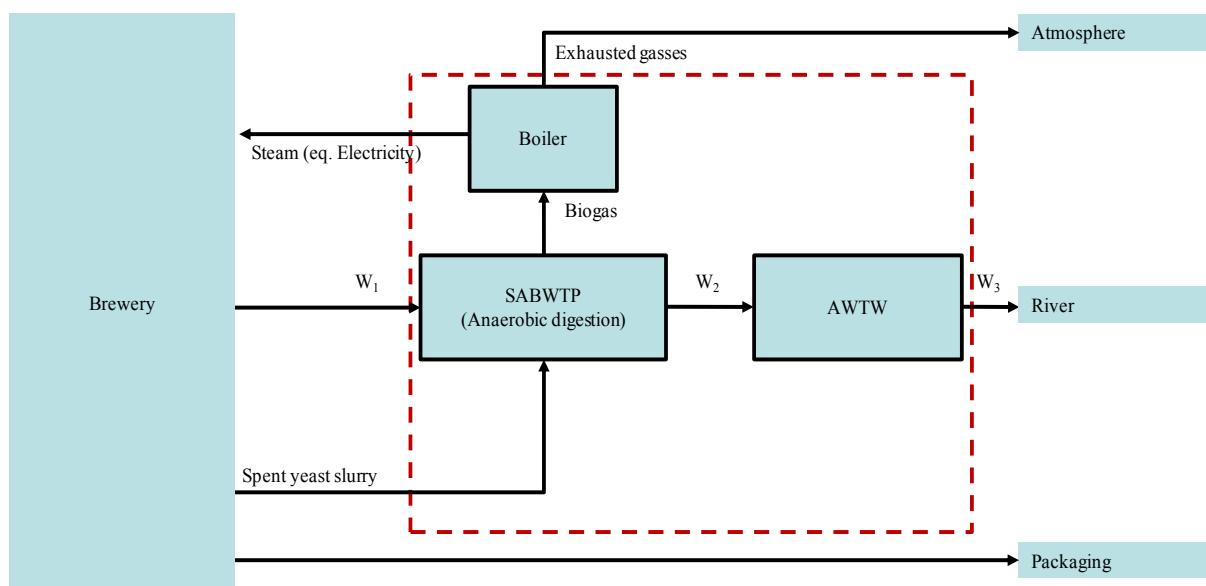


Figure 6.2: Illustration of the system boundary for Option 2 (the scenario with spent yeast fed to the SABWTP).

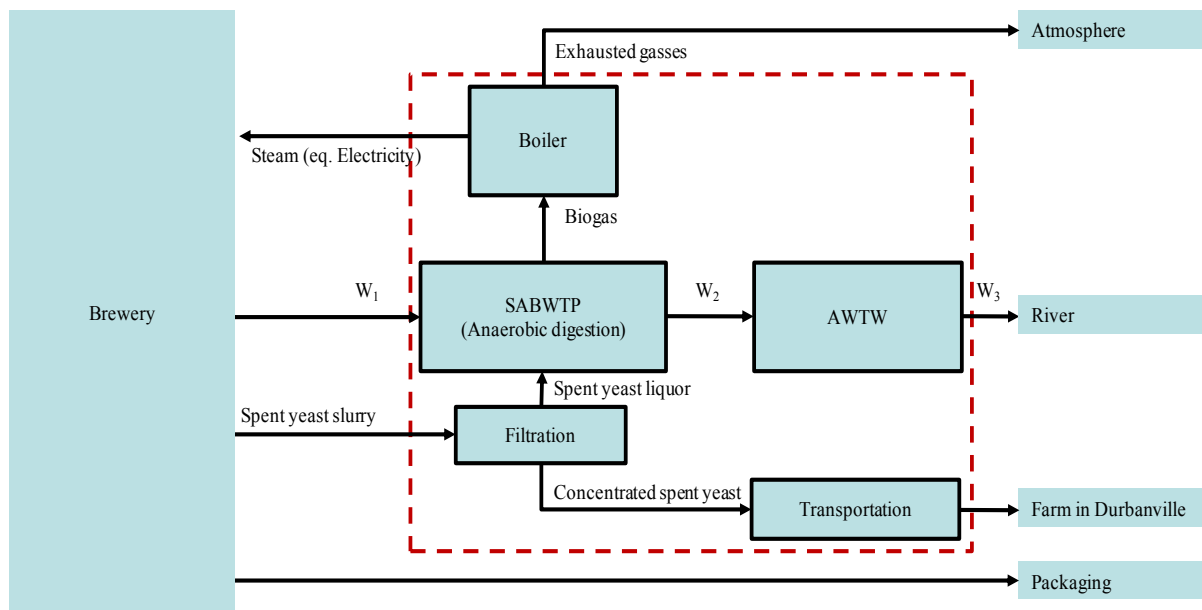


Figure 6.3: Illustration of the system boundary for Option 3 (the scenario with spent yeast filtration followed by feeding of spent yeast liquor to the SABWTP and transportation of the concentrate to a farm in Durbanville).

Functional unit

In the previous LCA study by Cohen (2006), the material and energy flows were expressed per kilogram of chemical oxygen demand (kg-COD) entering the SABWTP. This was due to the fact that the primary function of the SABWTP was to reduce the COD content of the wastewater. For the current study, a litre of beer produced was considered as an additional functional unit since the extra capacity of the SABWTP is used where spent yeast undergoes anaerobic digestion, hence the COD baseline is affected. For example, if the amount of methane produced from a kilogram of COD consumed turns out to be the same for Option 1 and Option 2, the fact that the amounts of methane produced and COD consumed vary across the scenarios would not be reflected when kg-COD is used as a reference unit. A litre of beer as a functional unit also allows for direct association between the primary product of the brewery (beer) and its environmental burdens. Further, COD from the spent yeast may not be considered as a waste to be treated, but rather an additional energy source for which the demand is determined in terms of beer volumes. Therefore, volume of beer provides a more standardised unit of comparison than kilogram of COD. The environmental impact analysis is presented with both units to allow comparison to previous work and the differences between them are addressed (see Section 6.5.2).

6.3 Process description and inventory methodology

6.3.1 Introduction

This section describes the material and energy balance methodology for the unit operations constituting the processes under evaluation with underlying assumptions given, and the resulting environmental impacts. The quality of the inventory data is interrogated and validated by cross checking against multiple literature findings as well as measured values from the SABWTP. Thus, the mass and energy flows for each scenario were generated based using findings of the current study, the literature, SABWTP data and the SimaPro™ data bases. Any other data required were calculated with underlying assumptions stated.

6.3.2 Material balances

Anaerobic wastewater treatment plant (SABWTP)

Chemical oxygen demand (COD) balances

To provide data for material balances across the SABWTP, rolling average data were generated using the plant data between January 2010 and January 2013. This was done in order to cancel the effect of noise and spikes in the data and to get representative values across the period. The cumulative COD profiles in Figure 5.3 were calculated by sequential summation of daily COD loading and effluent. The daily COD flows were calculated from the measured COD concentrations and volumetric flowrate of the SABWTP feed and effluent. Assuming negligible accumulation of COD in the plant over this period (pseudo steady-state), the total COD reacted was calculated as the difference between the COD fed and effluent COD. Similarly, the biogas produced was calculated over this period. A key assumption required to generate inventories for Option 2 and Option 3 was that the composition of COD in the spent yeast liquor is similar to that of the brewery wastewater. Hence the anaerobic digestion of degradation of the spent yeast liquor and the brewery wastewater is similar as validated by the plot of COD reduction rate as a function of organic loading rate (see Figure 5.6). Compositions of N, P and SO_4^{2-} relative to COD in the yeast liquor were also assumed to be the same in the brewery wastewater. In reality, the beer N, P and SO_4^{2-} to COD ratios are less than that of the wastewater due to the much higher COD content of the beer. Hence the assumed spent liquor N, P and SO_4^{2-} represented an overestimate.

Nitrogen (N), phosphorus (P), sulphate (SO_4^{2-}), calcium (Ca^{2+}) and hydrogen sulphide (H_2S) balances

The input nitrogen, phosphorus and sulphate values for brewery wastewater were taken from Cohen (2006). However, Cohen's output values were not assumed to hold as effluent concentrations are a function of the inlet concentrations and operation of the SABWTP, the latter being variable with time. Cohen's data were collected shortly after startup of the SABWTP (from year 2005 to 2006), whereas this study was conducted between 2010 and 2013 following significant plant optimisation, hence the N and P values in the effluent were expected to differ. Understanding the relationship between

SABWTP operation or performance and effluent concentration is preferable. In addition, Cohen's N and P values were based, erroneously, on the assumption that the removal efficiency of these nutrients was the same as the COD removal efficiency in the SABWTP; however, anaerobic digestion is focussed on the reduction of the carbon species with negligible N and P removal. For example, Akunna et al. (1994) reported influent and effluent TKN concentrations of 333 mg/L and 323 mg/L respectively (i.e. little difference, 3% removal), while the TCOD concentration were 5318 mg/L and 1213 mg/L respectively (77% removal). Zupančič et al. (2012) also found that the N and P concentrations influent to their anaerobic digestion reactors were roughly the same as the effluent concentrations. As such, N and P flowrates exiting the SABWTP were assumed to be the same as the influent flowrates. This assumption was also supported by the well-researched finding that in anaerobic digestion, almost all N in the substrate is converted to ammonia (see Equation 2.1) and that the ammonia exists in the liquid bulk as ammonium at pH below 8 (Figure 2.8). The hydrogen sulphide (H₂S) in the gas output stream was determined by a sulphur balance assuming no negligible accumulation of sulphur in the reactor over the period. The calcium exiting the reactor was assumed to be in the form of calcium carbonate and was equated to the amount of calcium carbonate entering the reactor.

The amounts N, P and S contributed by the spent yeast cells were calculated based on the elemental composition of *S. cerevisiae* presented by Villadsen, Nielsen & Lidén (2011) as shown Table 4.1. The N, P, S and COD contribution of the yeast cells in a suspension were calculated by Equation 6.1, where *i* represented each on the above components (See the derivation of this equation in Appendix 1.B.1). The caustic requirement has been assumed to be constant with COD loading. Thus, for every kilogram of COD fed, there was a set amount of caustic needed to neutralise the reactor. However, it should be noted that this is a very simplistic model based on historical data.

$$M_{i,suspension} = \frac{y \times \rho_L \times x_{cells} \times (1 - c_{cells})}{(1 - x_{cells})}, \text{ where} \quad \text{Equation 6.1}$$

$$M_{i,cells} \left[\frac{mg - i}{L - suspension} \right] \text{ is the mass } i \text{ from cells,}$$

$$y \left[\frac{mg - i}{mg - dry cell} \right] \text{ is the } i \text{ content of the yeast cells on a dry basis,}$$

$$\rho_L [mg L^{-1}] \text{ is the density of the liquid,}$$

$$x_{cells} \left[\frac{M_{dry cells}}{M_{suspension}} \right] \text{ is the mass fraction of the yeast cells in the suspension,}$$

$$c_{cells} \left[\frac{V_{dry cells}}{V_{suspension}} \right] \text{ is the volumetric concentration of cells in the suspension}$$

Biogas yield

The yield of CH₄ was 0.272 m³-CH₄/kg-COD_{reacted}. While this yield value was lower than the theoretical value of 0.350 m³-CH₄/kg-COD_{reacted} at STP, other authors have found comparable yield values ranging between 0.225 and 0.344 m³-CH₄/kg-COD_{reacted} during anaerobic digestion of brewery wastewater (Cohen 2006; Rao et al. 2007; Alvarado-Lassman et al. 2008; Neira & Jeison 2010; Zupančič et al. 2012). A sensitivity analysis demonstrating the effect of methane yield on energy output of the boiler is presented by Figure 6.9 in Section 6.4.5. The biogas was assumed to contain 70% methane (v/v) on average with the balance being mainly carbon dioxide. Higher fractions of methane in biogas have been reported for anaerobic digestion of brewery wastewater (Neira & Jeison 2010; Shao et al. 2008; Zupančič et al. 2012), a mixture of brewery wastewater and spent yeast (Neira & Jeison 2010; Zupančič et al. 2012) and for spent yeast as observed in the current study (see Figure 4.14). The higher methane contents reported by the various studies resulted in lower carbon dioxide contributions. Therefore, the relatively low average methane content assumed for this study represents an over estimate of the carbon dioxide released to the atmosphere.

Athlone wastewater treatment works (AWTW)

Nitrogen (N) and phosphorus (P)

The AWTW was primarily designed to remove nitrogen (N) and phosphorus (P) from wastewaters biologically using activated sludge systems. The resulting sludge is taken through anaerobic digestion process, thickened in sand drying lagoons and then applied to agricultural land. Nitrogen exits the AWTW in the final wastewater stream, as gaseous nitrogen oxides and as part of the sludge, whereas phosphorus only exits with the final wastewater stream and sludge.

According to Ekama & Wentzel (2008), the average mass fractions of N and P in the sludge resulting from activated sludge treatment of municipal wastewaters are relatively constant. They are 0.1 mg-N/mg-sludge mass and 0.025 mg-P/mg-sludge mass respectively. The N incorporated into the sludge is about 0.025 mg-N/mg-COD_{influent}, whereas TKN/COD_{influent} ratios in typical municipal influent wastewaters range between 0.07 and 0.13 (Ekama & Wentzel 2008). This corresponds to at most 36% recovery of total nitrogen to sludge. This indicated that a large portion of the N in the influent undergoes the nitrification to nitrate (Ekama & Wentzel 2008). Typically, about 20% to 25% of the total P is removed with the biological sludge as limited by microbial accumulation of P of about 0.016 mg-P/mg-COD_{Biomass}. Further biological removal of P can be carried out when reactor conditions favour phosphate accumulating organisms. These microorganisms have a much higher content of P incorporated as polyphosphates (Ekama & Wentzel 2008).

Data pertaining to the municipal wastewater treatment was obtained from the facility's log books by Cohen (2006) between February 2005 and July 2005. The data included input and output volumetric flowrates, concentrations of suspended solids, volatile suspended solids, settleable solid, COD, TKN,

nitrogen as ammonia, total phosphorus and total electricity consumption by the facility. The treatment efficiency (total contaminants removed per contaminants entering the plant) of the AWTW depends mainly on the influent characteristics as the final effluent characteristics are predetermined to comply with the Department of Water Affairs and Forestry (DWAF) of South Africa effluent guidelines i.e. the facility must operate such that the effluent has concentrations lower than the allowable maximum concentrations regardless of the influent characteristics. In this way, the material flows from the SABWTP can be tracked through to the final effluent of the AWTW when the change in volumetric flowrate across the treatment facility is known as demonstrated in Figure 6.5.

The AWTW set the maximum P concentration at in the final effluent at 8.7 mg-P/L. In reality the removal of P is a characteristic of the biological processes that occur at the treatment plant. For this reason, a conservative estimate of 20% removal of P from the wastewater into the sludge fraction was assumed, in line with the observations of Ekama & Wentzel (2008), to accommodate possible cases of treatment to concentrations lower than the set criterion. Figure 6.4 depicts the general algorithm followed for calculating P in the sludge for the three scenarios.

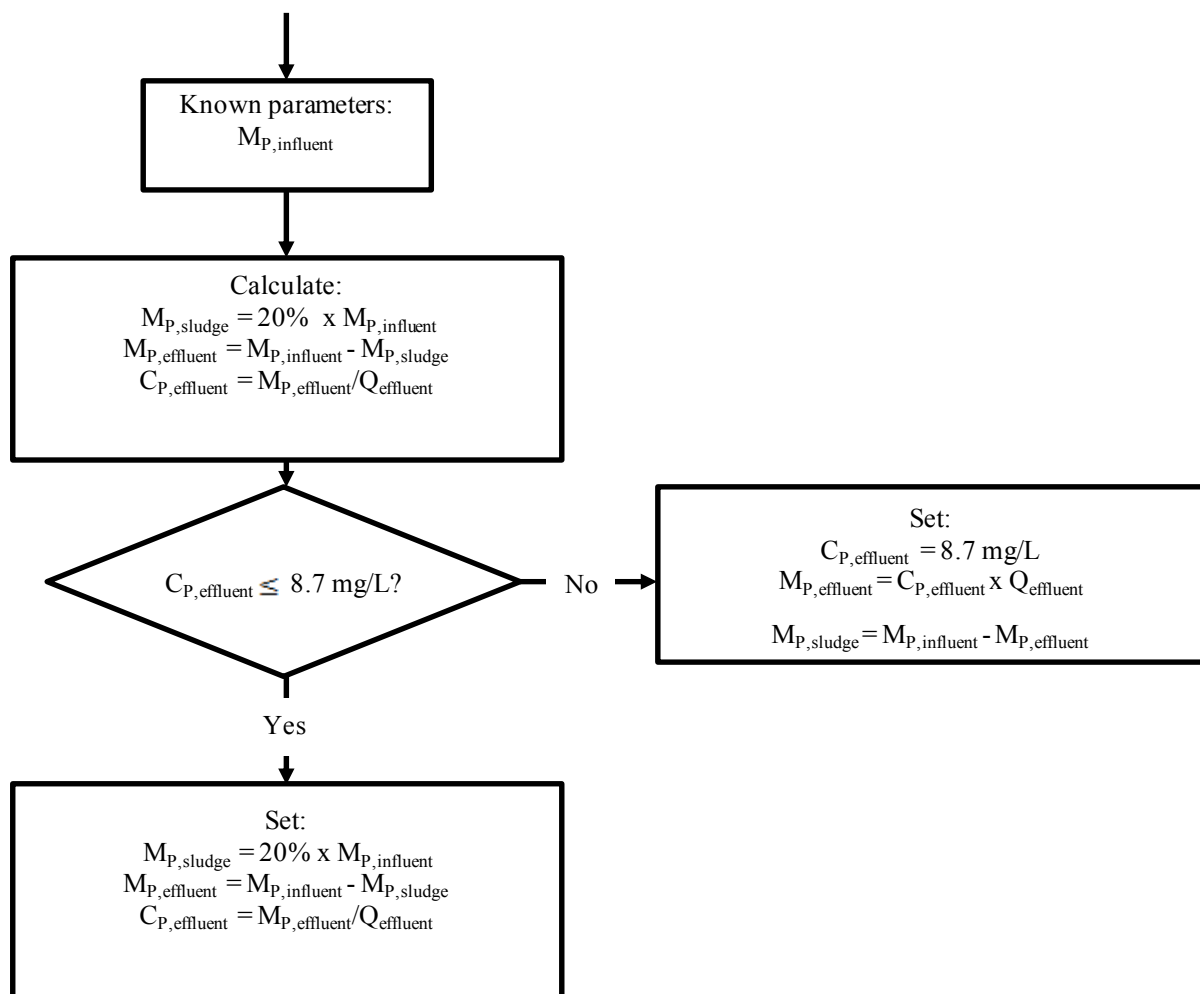


Figure 6.4: Representation of the algorithm for calculating phosphorus exiting with sludge at the Athlone Wastewater Treatment Works (AWTW). $M_{P, \text{sludge}}$ is the flowrate of phosphorus (P) exiting in the sludge; $M_{P, \text{influent}}$ and $M_{P, \text{effluent}}$ are flowrates of P in the influent and effluent water respectively; Q_{effluent} is the effluent water volumetric flowrate; $C_{P, \text{effluent}}$ is concentration of P in the effluent water.

From SABWTP to AWTW: Burden allocation strategy

The effluent from SABWTP (stream 1 in Figure 6.5) mixes with wastewater from other sources stream 2 in Figure 6.5) before entering the AWTW. The burdens incurred due to operation of the AWTW to be attributed to the SABWTP effluent had to be decoupled from the treatment of wastewaters from the other sources. There important factors or assumptions were considered for this, as follows:

- **Power consumption is mainly a function of wastewater water volumes processed.** Therefore, electricity consumption at AWTW attributed to treatment of SABWTP effluent was calculated based on the SABWTP effluent volume.
- **Change in AWTW feed due to addition of spent yeast in the SABWTP does not change the efficiency of the AWTW.** While increased organic content in the AWTW feed would improve the efficiency of the treatment process, the change in the feed concentration due to effluent from co-digestion of the spent yeast and brewery wastewater would be insignificant, since the SABWTP is only a fraction of the AWTW feed volume.
- **The magnitude of burdens associated with disposal of substances (N, P, SO_4^{2-} etc) are a function of the amounts (mass) of those substance emitted.** For this reason, substances released at the AWTW are attributed to the SABWTP based on the effluent mass flowrates of these substances.

Following the above approach, inventories of the AWTW process attributable to the treatment of the SABWTP can be determined without knowledge of the characteristics of wastewater coming from the other sources (stream 2 in Figure 6.5) as follows:

$$M_{i,6} = C_{i,6} \times Q_3 \quad \text{Equation 6.2}$$

$$M_{i,5} = x_{i,5} \times M_{i,3} \quad \text{Equation 6.3}$$

$$M_{i,4} = M_{i,3} - M_{i,5} - M_{i,6} , \quad \text{Equation 6.4}$$

where

$M_{i,j}$, $C_{i,j}$ and $x_{i,j}$ are the mass flowrate, concentration and recovery of component i in stream j respectively and Q_j is the volumetric flowrate of stream j

The inventories for AWTW were expressed with respect to the volume of water entering this treatment facility (presented in Table 6.5 of Section 6.4.4). This approach allowed for analysis of the effect of inlet wastewater (to AWTW) characteristics on the functioning of the facility. Hence, the effect of changes in the operation of the SABWTP on the efficiency AWTW could be assessed. Further, the approach allows for environmental performance of the AWTW as a standalone process to be assessed under various inlet wastewater characteristics.

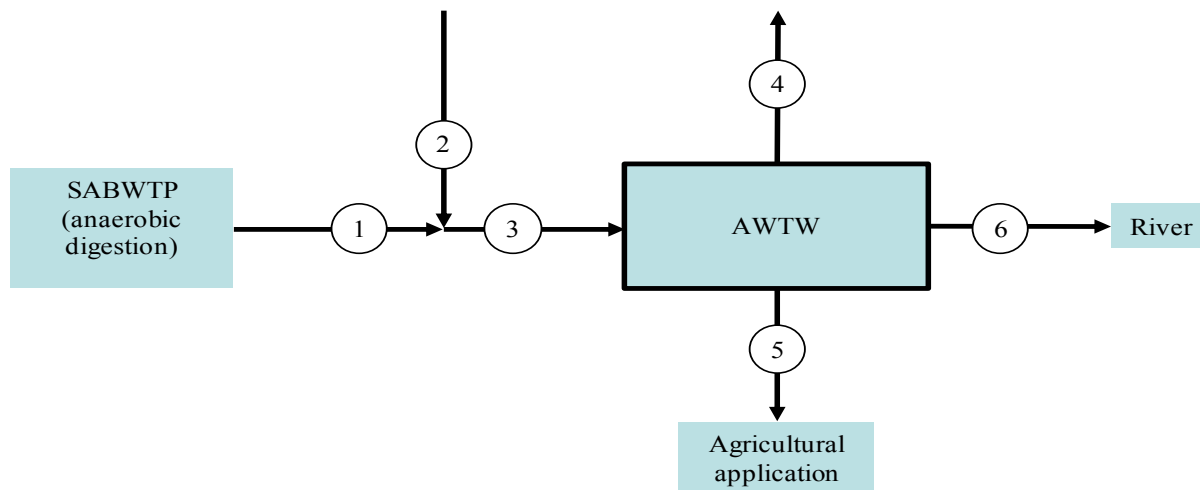


Figure 6.5: Wastewater allocation strategy from SABWTP to AWTW.

Spent yeast transportation to farm

Transportation emissions and fuel consumption were reported per distance travelled. Based on the volume per load of one tanker, the number of trips required per volume of spent yeast produced was calculated. This, together with the fuel consumption, distance transported and emissions per consumption were used to determine the emissions and fuel consumption per volume of yeast transported. Volume of yeast produced was related to COD of wastewater by the yeast to wastewater ratio of 0.7% (v/v), determined by Zupančič, Škrjanec & Logar, (2012), and confirmed by Neira & Jeison, 2010). The spent yeast was transported in a 20 ton truck (truck capacity) to Durbanville, some 32 km (distance per trip, $D_{\text{per trip}}$) away from Newlands. The total distance was calculated according to Equation 6.5.

$$D_{\text{total}} = \frac{M_{\text{SY}}}{M_{\text{truck capacity}}} D_{\text{per trip}},$$

Equation 6.5

where

M_{SY} [kg] is the mass of spent yeast transported,

$M_{truck\ capacity}$ [kg] is the maximum mass load carried by the truck,

$D_{per\ trip}$ [km] is the distance traveled for one trip

6.3.3 Power consumption and energy recovery

SABWTP and AWTW

The ratio of power consumed to COD metabolised cannot be assumed to be unchanged from that presented by Cohen (2006) due to the possible change in wastewater concentration and reactor loading, influencing the amount of water pumped per unit COD. The power consumption is mainly a function of the amount of wastewater processed. Cohen (2006) determined that the power required by the SABWTP per cubic meter of wastewater fed was 0.0245 kW (i.e. 0.0245 kW/m³-feed) as shown in

Table 6.1: Power requirements of the SABWTP and the AWTW as presented by Cohen (2006)

. This value was calculated based on the total installed power (i.e. power requirements from all installed equipment) and maximum inlet volumetric flowrate allowable. This vastly over estimates the power requirement of the SABWTP as seen by comparison with the AWTW power input. Since no other information on the power consumption of the SABWTP was available, this value was used to calculate the SABWTP power requirement in the current study, using the volume of wastewater treated. The monthly power requirement of the AWTW was obtained from the facility power bill. The power consumption of the AWTW is based on the maximum volumetric capacity of the plant and the average

Power requirements	Units	SABWTP	AWTW
Effluent plant feeder	kW	35	-
Reactor pump	kW	11	-
Recirculation pump	kW	7.5	-
2 stirrers (total power)	kW	18	-
Waste gas fan 1	kW	3	-
Waste gas fan 2	kW	2.2	-
Waste gas fan 3	kW	7.5	-
Miscellaneous pumps	kW	11	-
Biofilter	kW	15	-
Total installed power	kW	110.2 ^a	-
Total capacity	kWh/month	79344 ^b	65615
Maximum capacity (influent flowrate)	kL/day	4500	150000
Power delivered	kW/kL	0.0245	0.0146 ^c

power consumption by the Municipal facility between February 2005 and July 2005. This value includes the power consumption of the pumping station that feeds the AWTW.

Table 6.1: Power requirements of the SABWTP and the AWTW as presented by Cohen (2006)

^a The sum of the power requirement from all installed equipment.

^b Based on a 30 day month and the total installed power.

^c Based on a 30 day month and the monthly power consumption. Not originally reported by Cohen (2006).

Energy requirement of the vacuum filter for Option 3

In filtration, the basic goal is to obtain fluid flow through the filter medium (which may include the cake) by applying pressure drop across the medium. The energy consumption of filtering is directly related to the volume of filtrate and the pressure required to overcome the resistance of the filter medium (i.e. the pressure required to suck or push that volume of filtrate through the filter medium) as represented by Equation 6.6. The fundamentals of filtration are independent of how the pressure drop is achieved (Svarovsky 1977). As such, the required pressure drop across the medium is only a function of the properties of the filter medium. The main assumption in this calculation was that spent yeast is filtered at constant pressure with no cake build-up.

$$E_{filt} = \Delta P_{filt} \times V_{filt} \quad \text{Equation 6.6}$$

Boiler

Daily production rates of CH₄ and steam were obtained from SAB log book. The ratio of production of steam to CH₄ provided the conversion of CH₄ to steam (kg-steam/kg-CH₄). The amount of CH₄ utilised was calculated assuming 100% conversion to CO₂ and H₂O. The steam produced was thus calculated by the multiplication of the steam/CH₄ ratio and the CH₄ produced by the SABWTP. Energy produced as steam was calculated by assuming that all the heat comes from the latent heat of vaporisation of water (2.23 MJ/kg-H₂O or 0.628 kWh/kg-H₂O) according to Equation 6.7. Based on an efficiency (88% kWh-steam/kWh-electricity consumed) of the electric boiler an equivalent amount of electricity (eElectricity) replaced by the biogas boiler was calculated according to Equation 6.8.

$$E_{steam\ produced} = H_{vap,H_2O} \times Y_{steam/CH_4} \times Y_{CH_4/COD} \times M_{CH_4}, \quad \text{Equation 6.7}$$

where

$H_{vap,H_2O} \left[\frac{kWh}{kg - steam} \right]$ is the heat of vaporisation of water,

$Y_{steam/CH_4} \left[\frac{kg - steam}{kg - CH_4} \right]$ is the ratio of steam produced per CH₄ burned,

$Y_{CH_4/COD} \left[\frac{kg - steam}{kg - CH_4} \right]$ is the methane yield from COD reacted,

$M_{CH_4} [kg]$ mass of CH_4 burned

$$e_{Electricity\ produced} = \frac{E_{steam\ produced}}{\xi_{e.boiler}}, \quad \text{Equation 6.8}$$

where

$\xi_{e.boiler} \left[\frac{kWh - steam}{kWh - electricity} \right]$ is the electric boiler efficiency (energy of

steam produced per electricity consumed by the electric boiler).

6.4 Results and Analysis: Inventories

6.4.1 Introduction

This section presents the inventory results determined according to the discussions presented in Section 6.3, and discussion on the inventory data for the various processes involved. This is followed by sensitivity analyses on the power consumption and energy output as electricity was most likely to contribute significantly to the environmental impact scores as highlighted in Section 2.4 of the Literature Review chapter.

6.4.2 SABWTP

Inputs

Table 6.2 shows the inventory results of the SABWTP and its associated processes. The input values to the SABWTP process for Option 1 and Option 3 are the same with respect to P, SO_4 and N (Table 6.2). This resulted from the assumption that the ratios of these components to total COD in the wastewater are the same as in the spent yeast liquor. The P and N inputs values for Option 2 are higher due to higher P/COD (0.0108 mg-P/mg-COD) and N/COD (0.0788 mg-N/mg-COD) as compared to the 0.0023 mg-P/mg-COD and 0.0557 mg-N/mg-COD respectively for Option 1 and Option 3. The sulphate input was lower for Option 2 due to lower SO_4 /COD ratio of the spent yeast cells as compared to the wastewater and spent yeast liquor. The $CaCO_3$ input is the same across the three scenarios. This was due to the assumption that the pH control agent ($CaCO_3$) was required according to the amount of COD fed to the SABWTP, hence the $CaCO_3$ /COD remains constant. However, the requirement of $CaCO_3$ is based on many other operational factors influenced by the plant control philosophy and the microbial responses,

and would; thus, require intensive data collection and mathematical complex modeling. The values for CaCO_3 as presented in Table 6.2 are indicative. Electricity input per mass of COD fed to the SABWTP was lower for Option 2 as compared to Option 1 and Option 3. This was expected since power consumption mainly a function of the volume wastewater treated, and the COD content (COD/wastewater volume) of the influent wastewater is much higher for Option 2 as compared to the other scenarios.

Outputs

The amount of CH_4 produced per kilograms of COD fed to the SABWTP was highest for Option 2. The higher COD concentration in the reactor for Option 2 resulted in higher COD reduction rates per unit volume (as presented in

Table 5.3), which is proportional to the amount of CH_4 produced, due to the positive dependency of rates of substrate concentration as discussed in Section 4.3.6, Section 5.4 and Section 5.5. Option 3 had slightly lower CH_4 produced per COD fed than Option 1 which was counter-intuitive according to COD degradation rates presented in

Table 5.3 and associated discussions. This indicated that while there is a benefit of gain in the rate of COD reduction for Option 3, the amount of COD fed is relatively higher. However, the difference between the values for Option 1 and Option 3 is small enough to be considered constant across the two scenarios. The volume of treated wastewater per COD fed was lowest for Option 2, followed by Option 3. This was expected since both these scenarios have high COD fed compared to Option 1 while the volumetric flowrates are relatively similar across the three scenarios.

Table 6.2: Inventories for the SABWTP with associated processes based on a kilogram of COD fed (kg-COD).

		Option 1	Option 2	Option 3	
SABWTP	Inputs				
	TCOD	kg/kg-COD	1.0000	1.0000	1.0000
	P	kg/kg-COD	0.0023	0.0058	0.0023
	SO_4	kg/kg-COD	0.1904	0.1150	0.1904
	N	kg/kg-COD	0.0557	0.0651	0.0557
	Inputs from technosphere				
	CaCO_3	kg/kg-COD	0.0295	0.0295	0.0295
	Electricity	kWh/kg-COD	0.2563	0.1524	0.2261
	Outputs to technosphere				
	Boiler	kg- CH_4 /kg-COD	0.1391	0.1448	0.1361

	AWTW-Option 1	L/kg-COD	380	-	-
	AWTW-Option 2	-	-	200	-
	AWTW-Option 3	-	-	-	330
Associated processes	Transport	km-TDT/kg-COD ^a	0.0049	-	0.0022
	Yeast filtration	kWh/kg-COD	-	-	0.0931

^a Refer to Section 6.3.2 in 'Spent yeast transportation to farm' for explanation of this unit.

6.4.3 Boiler and transport

Boiler

Inventories for the boiler operation were based on mass of CH₄ input as presented

Table 6.3. The values presented in

Table 6.3 are applicable to all investigated processing scenarios since the flows of CO₂ and amount of equivalent power produced are only variable with CH₄ fed. This is true when applying the assumption that the conversions CH₄ to equivalent power and CO₂ as well as the biogas composition are constants. The calculations relating to these inventory values are presented in the Boiler subsection of Section 6.3.3 represented by Equation 6.7 and Equation 6.8.

Table 6.3: Inventories for the Boiler based on a kilogram of methane burned (kg-CH₄).

Boiler		
Inputs		
CH ₄	kg/kg-CH ₄	1.000
CO ₂	kg/kg-CH ₄	1.223
Avoided products		
eElectricity	kWh/kg-CH ₄	17.20
Emissions to air		
CO ₂	kg/kg-CH ₄	3.575
H ₂ O	kg/kg-CH ₄	1.125

Yeast transport

The inventories for the yeast transport were presented per total distance travelled as shown in Table 6.4. These are applicable to all three scenarios as with the boiler inventories. The inventories were determined according to Equation 6.5 in Section 6.3.2.

Table 6.4: Inventories for the Yeast transport process based on kilometres of total distance travelled (km-TDT).

Yeast transport		
Inputs from technosphere		
Diesel	kg/km-TDT	0.2496
Emissions to air		
VOC	kg/km-TDT	2.78E-04
CO	kg/km-TDT	1.44E-03
NO _x	kg/km-TDT	5.35E-03
Diesel soot	kg/km-TDT	2.62E-04

6.4.4 AWTW

Inputs

Inventories for AWTW are presented per litre of influent (effluent from SABWTP) as shown in Table 6.5. This approach allowed for analysis of the AWTW performance as a standalone processes, as discussed in Section 6.3.2. The TCOD input values across the three scenarios are slightly higher than the SABWTP effluent concentrations. This was due to the fact the SABWTP consumes water, thereby reducing the volumetric flowrate entering the AWTW. As expected, the addition of yeast to the SABWTP causes an increase in loading of contaminants into the AWTW as shown by the input values of Option 2 in comparison to Option 1 (Table 6.5). The difference between the input values of Option 1 and Option 3 reflect the low volumetric contribution of the spent yeast liquor as well as the assumed similarities between the brewery wastewater and the spent yeast liquor.

Outputs

The amount of CH₄ produced by the AWTW for Option 2 was greater than CH₄ produce for Option 1 and Option 3. This is was expected as the amount of COD per litre of wastewater entering the AWTW was highest for Option 2 and the allowable maximum concentration in the final stream was specified. Hence the TCOD emissions to water per litre of influent were the same across the three Options. Similarly, the amount of CO₂ produced for Option 2 was the highest. The NO₃ emitted to air was produced from the nitrogen in the influent wastewater, and was also highest for Option 2 due to limitation in the capacity of the sludge to incorporate the nitrogen. The P emitted to water was greatest for Option 2 due to the relatively higher content in the influent wastewater. Option 1 and Option 3, benefited from the capacity of the sludge to incorporate P such that the final wastewater concentrations were much lower than the legislated values. The removal of CaCO₃ from the wastewater was assumed to be negligible.

Table 6.5: Inventories for the Athlone Wastewater Treatment Works (AWTW) based on a litre of wastewater processed (L of influent to the AWTW). All emissions to soil are carried out via sludge disposal.

		AWTW - Option 1	AWTW - Option 2	AWTW - Option 3
Inputs				
TCOD	kg/L-influent	5.88E-04	9.58E-04	7.29E-04
P	kg/L-influent	6.06E-06	2.89E-05	6.99E-06
SO ₄	kg/L-influent	5.01E-04	5.74E-04	5.14E-04
TKN	kg/L-influent	1.46E-04	3.25E-04	1.69E-04
CaCO ₃	kg/L-influent	7.75E-05	1.47E-04	8.94E-05
Inputs from technosphere				
Electricity	kWh/kg-influent	1.46E-05	1.46E-05	1.46E-05
Emissions to air				
CH ₄	kg/L-influent	8.29E-05	1.49E-04	1.08E-04
CO ₂	kg/L-influent	6.84E-05	1.23E-04	8.93E-05
NO ₃	kg/L-influent	3.40E-04	8.46E-04	4.03E-04
Emissions to water				
TCOD	kg/L-influent	1.25E-04	1.25E-04	1.25E-04
P	kg/L-influent	4.85E-06	2.31E-05	5.59E-06
SO ₄	kg/L-influent	5.01E-04	5.74E-04	5.14E-04
TKN	kg/L-influent	1.70E-05	1.70E-05	1.70E-05
CaCO ₃	kg/L-influent	7.75E-05	1.47E-04	8.94E-05
Emissions to soil, Agriculture				
P	kg/L-influent	1.21E-06	5.77E-06	1.40E-06
TKN	kg/L-influent	5.27E-05	1.17E-04	6.08E-05

6.4.5 Sensitivity analyses

The current study seeks to understand the effect of improving energy production on the remediation function and the environmental performance of the wastewater treatment system. It has been shown previously that energy consumption and energy output of the wastewater treatment system has the most profound effect on the outcome of the LCA (Section 2.4 in Chapter 2). It is therefore, important to understand the impact of input parameters related to the wastewater operations on power consumption and energy recovery, as well as the sensitivity of power consumption and energy recovery to variations in the input parameters. The analysis is presented in terms of kg-COD to analyse operational efficiency of the wastewater management system and associated processes, and L-beer produced for a non-changing comparative reference point that allows for representation of the brewery's environmental efficiency as discussed in 6.2.2.

Influent COD concentration and OLR

Influent COD concentration and OLR have strong effect on rate of COD conversion into CH₄ and, therefore, the production of energy (discussed in Section 4.3.6, Section 5.5.1 and Section 5.5.2). The differences between Option 1, Option 2 and Option 3 represented variations in COD concentration and OLR fed to the SABWTP. The effect of COD concentration and OLR are, thus, presented in terms of the three scenarios.

Kilogram of COD fed to SABWTP as reference unit

Figure 6.6 shows the total energy consumed in the SABWTP, total energy recovered from the SABWTP and the net energy recovered for return to the brewery per kilogram of COD fed to the SABWTP. Energy consumption per kilogram of COD decreased from 0.26 kWh/kg-COD in Option 1 (base case reflecting standard operation) to 0.15 kWh/kg-COD in Option 2 (anaerobic co-digestion of spent yeast and wastewater). While energy consumption of the wastewater treatment plant was assumed to be directly proportional to the volume treated, the energy consumption per kilogram of COD for Option 2 was expected to be lower since the volume fed increased by only 0.7% whereas the COD loading increased by 69%. Hence, the energy consumption of Option 2 was about 68% lower than Option 1 as expected. The much greater increase in energy from the 0.26 kWh/kg-COD for Option 1 to 0.32 kWh/kg-COD for Option 3 (case with anaerobic digestion of filtered spent yeast filtrate and wastewater) was due to the contribution of the rotary vacuum filter. At a filtration pressure difference (ΔP_{fil}) of 250 kPa, the energy required to run the filtration was about 29% of the total energy requirement in the scenario. At a filtration pressure difference (ΔP_{fil}) of 250 kPa, the energy required to run the filtration was about 29% of the total energy requirement in the scenario. The results showed a small increase (4%) in energy recovered per kilogram of COD from Option 1 to Option 2. Figure 5.6 showed the linear relationship with a positive slope between the rates of COD reacted, which can be converted to methane production by the yield factor, and COD fed. Thus, the energy produced per unit of time per reactor volume was expected to be higher for Option 2 than Option 1 due to the increased loading rates. However, normalising by dividing by the COD fed resulted in the small increase of energy output per kilogram of COD fed. Similarly with Option 3, the change in energy output per kilogram of COD fed was very small.

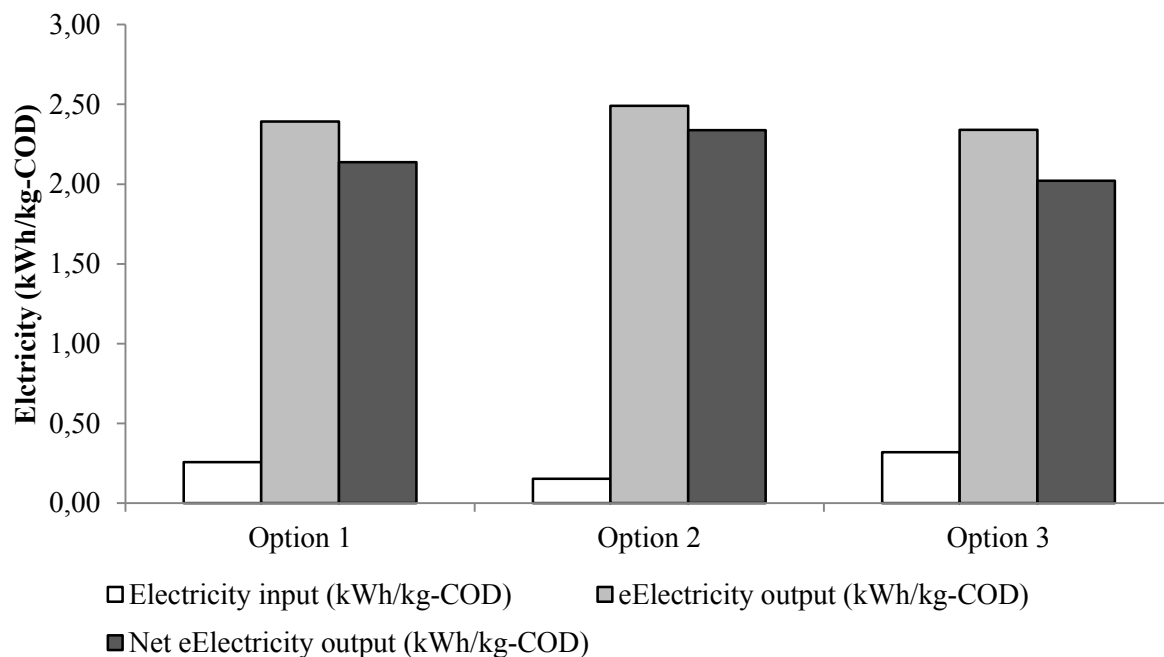


Figure 6.6: Electricity consumption, equivalent electricity produced and net equivalent electricity per kilogram of COD fed to the SABWTP for Option 1, Option 2 and Option 3 at 1.8 day HRT.

Litre of beer produced as reference unit

The effect of the addition of organic loading in on the power saving as shown in Figure 6.6 was not well represented from the brewery perspective. While the representation of energy consumption and recovery per kilogram of COD is useful to demonstrate how the additional feed affects the performance of the SABWTP, it does not account for the fact that the increased COD loading resulted in increased amount of energy produced. When considering the same analysis per volume of beer produced, the results differ significantly (see Figure 6.7). Figure 6.8 (a) and Figure 6.8 (b) show the percentage change in net energy recovered per kilogram of COD and per cubic meter of beer produced respectively, using Option 1 as the starting point (base case). Figure 6.8 (b) shows that the total energy recovered per volume of beer produced for Option 2 is significantly (more than 45%) higher than for Option 1, while the Figure 6.8 (a) shows a smaller increase (less than 10%) in the every recovered per kg-COD. This suggested that increasing the OLR to the on-site anaerobic digester increases the energy output efficiency (energy yield) of the SABWTP-Boiler circuit, but improves the overall energy output of the system by even larger amount. However, the energy output efficiency for Option 3 decreased by 7% (Figure 6.8 (a)), whereas the total energy output efficiency increased by about 6% (Figure 6.8 (b)). The drop in energy production efficiency of Option 3 relative to Option 1 was mainly due to the consumption of power by the filtration system, whereas the smaller increase in energy output relative to that of Option 2 was due to lower COD degradation rates caused by lower OLR and reactor organic concentrations.

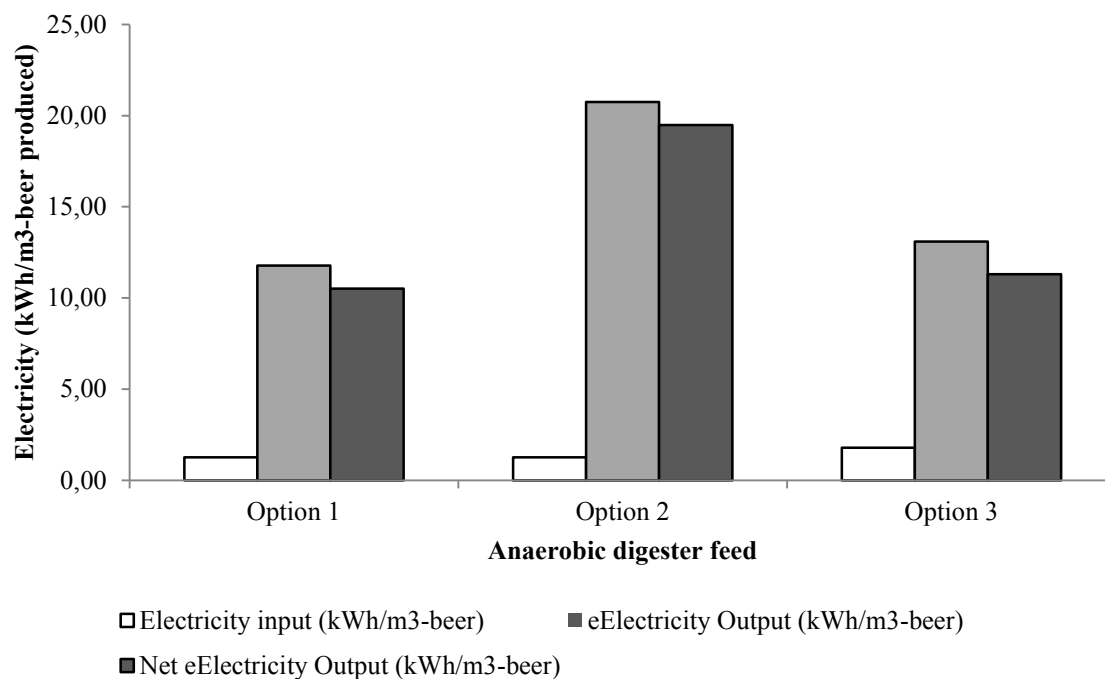


Figure 6.7: Electricity consumption, equivalent electricity produced and net equivalent electricity per volume of beer produced for Option 1, Option 2 and Option 3 at 1.8 day HRT.

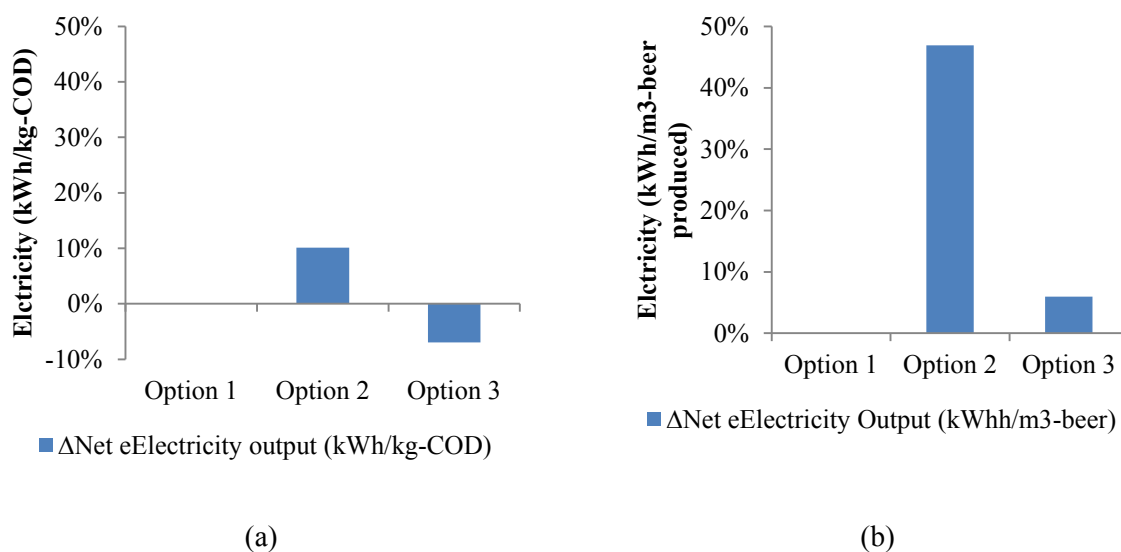


Figure 6.8: Percentage change in net energy recovered using Option 1 as a starting point. (a) Change in net energy recovered per kilogram of COD, (b) change in energy recovered per cubic meter of beer produced.

Methane yield and electric boiler efficiency

A conservative methane yield of $0.272 \text{ m}^3\text{-CH}_4/\text{kg-COD}_{\text{reacted}}$ was assumed for all three scenarios. The methane has a direct impact on the energy output of the system as it directly determines the amount of steam energy produced per COD treated (Equation 6.8). The equivalent electricity output or savings depends on an assumed efficiency of the electric boiler which produces the steam replaced by the biogas

boiler steam according to Equation 6.8. Equation 6.10 represents the net energy output of the system. Equation 6.7 can be expanded further to produce Equation 6.9. Substituting in Equation 6.8 and Equation 6.9 into Equation 6.10 and simplifying according to

$$E_{\text{steam produced}} = H_{\text{vap,H}_2\text{O}} \times Y_{\text{steam/CH}_4} \times Y_{\text{CH}_4/\text{COD}} \times M_{\text{COD reacted}} \quad \text{Equation 6.9}$$

$$E_{\text{net energy}} = e\text{Electricity produced} - E_{\text{consumped}} \quad \text{Equation 6.10}$$

$$E_{\text{net energy}} = \frac{Y_{\text{CH}_4/\text{COD}}}{\xi_{\text{e.boiler}}} Y_{\text{steam/COD}} - E_{\text{consumped}}, \text{ where} \quad \text{Equation 6.11}$$

$Y_{\text{steam/COD}}$ is the energy conversion constant defined according to

$$\text{Equation 6.12}$$

Equation 6.12

$$Y_{\text{steam/COD}} = H_{\text{vap,H}_2\text{O}} \times Y_{\text{steam/CH}_4} \times M_{\text{COD reacted}}$$

Option 2 was chosen to evaluate the effect of the methane yield on energy recovery as this scenario consumes the least energy per each of the functional units. As such, the net energy recovery by Option 2 is least affected by the energy consumption and thus, useful for demonstrating the effect of the above mentioned factors better than Option 1 and Option 3 (i.e. for Option 2, $E_{\text{consumption}}$ is relatively smaller compared to $e\text{Electricity produced}$ (Equation 6.10) such that changes in $Y_{\text{CH}_4/\text{COD}}$ or $\xi_{\text{e.boiler}}$ (Equation 6.11) will demonstrate relatively greater changes in $E_{\text{net energy}}$ as when compared to Option 1 and Option 3.

Figure 6.9 shows the effect of methane yield on change in net equivalent electricity replaced for Option 2. An efficiency value 88% was assumed. Figure 6.10 shows an evaluation of the influence of the electric boiler efficiency on the change in net energy recovered for Option 2. The effect of electric boiler efficiency over this range was not sufficient to cause concern in the final environmental impact assessment and decision making process.

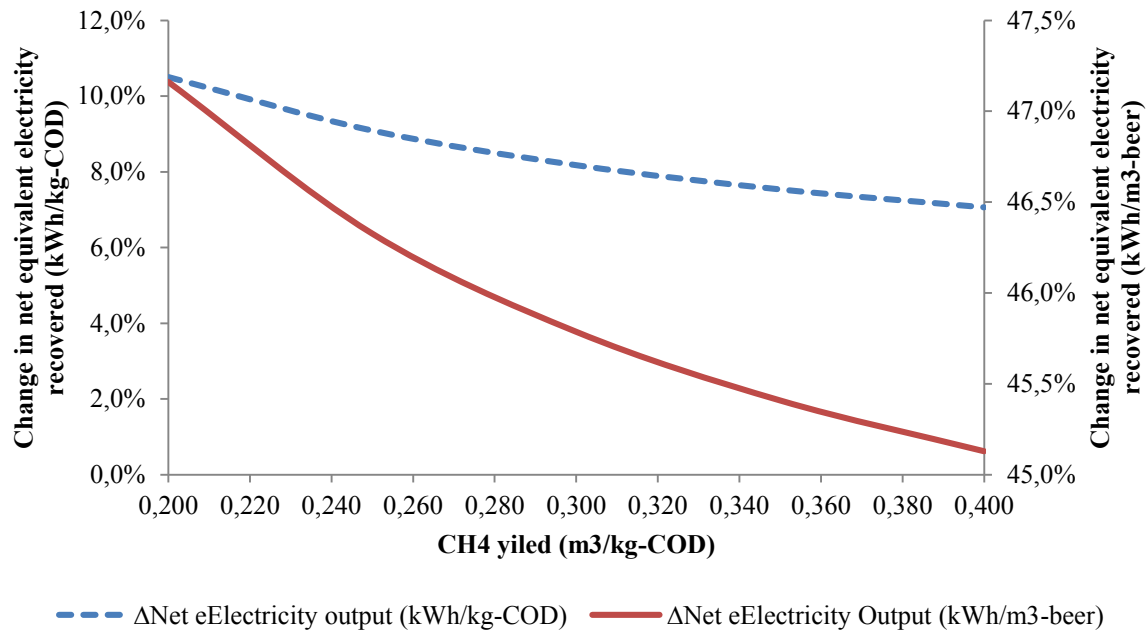


Figure 6.9: The influence of methane yield on the equivalent electricity replaced or recovered by the biogas boiler for Option 2.

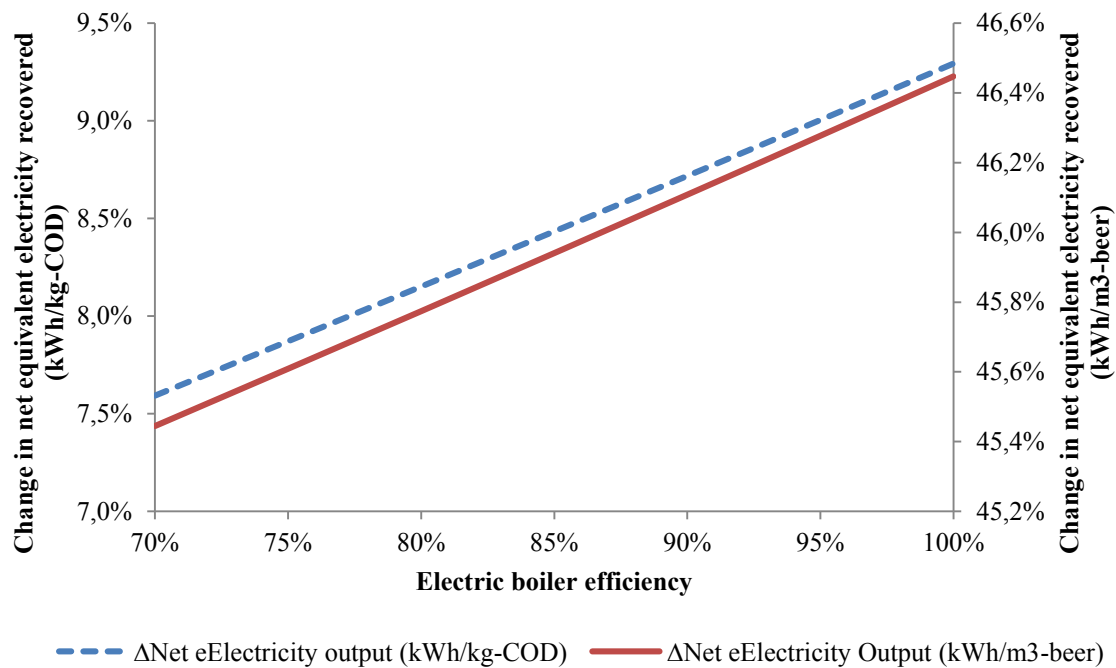


Figure 6.10: The influence of electric boiler efficiency on the equivalent electricity replaced or recovered by the biogas boiler for Option 2.

6.5 Comparing of environmental impacts of the three scenarios using LCA and identifying key contributors

6.5.1 Introduction

This section presents the results and analyses pertaining to the impacts of each of the selected processing options. A general comparison between the three processing options is presented followed by the analysis of each impact category.

6.5.2 Comparison

The results of the comparison of environmental burdens of the three processing options are represented by Figure 6.11 and Figure 6.12. In each category, the results are presented as a percentage relative to the impact score of the option generating the maximum burden or relief. The relative or comparative impact scores were achieved by taking the absolute values of the scores of each scenario and finding the highest number for each impact category, and comparing that with impact scores of the other processes, according to Equation 6.13.

$$RI_i[\%] = \frac{I_i}{\text{maximum}(|I_i|, |I_{j \neq i}|)}, \text{ where} \quad \text{Equation 6.13}$$

RI is the relative (or comparative) impact score of process *i*

I_i is the actual impact score of process *i*

I_{j ≠ i} are impact scores of other processes in the comparison analyses

Figure 6.11 shows the results of the comparison when using **a kilogram of COD entering SAWTP as the functional unit**. Similar trend is observed across all impact categories except *eutrophication*. Option 2 offered the better environmental relief while Option 3 showed the least attractive results cross all impacts categories except for *eutrophication*. The higher *eutrophication* impact by Option 2 was expected since the addition of the spent yeast suspension into the SABWTP resulted in an increase of N and P loading per COD fed. The elevated N and P concentrations in the effluent were thus responsible for the higher *eutrophication* impact of Option 2. While the trends across all the other impact categories are similar to each other, Option 2 showed relatively higher *global warming potential* (GWP) impact relief. The toxicity impact categories (*human toxicity, fresh water ecotoxicity, marine ecotoxicity and terrestrial ecotoxicity*), *abiotic depletion* and *acidification* showed the same characterisation. This suggested that there is one predominant process which contributed to the scores of these impact categories.

Figure 6.12 shows the same comparison using a **litre of beer produced in the brewery as the functional unit**. The results show very little differences between Option 1 and Option 3 at around 55% of Option 2 scores for all impact categories, except for *eutrophication*, *global warming potential* and *photochemical oxidation*. When considering a litre of beer as a functional unit, reductions in environmental burden offered by Option 3 as compared to Option 1 and Option 2 are more pronounced for all impact categories except *eutrophication*. The *eutrophication* burden for Option 2 was the highest, while Option 1 and Option 3 showed similar values at 71% of Option 2. These results suggested that Option 2 is the best choice of the three scenarios when considering all impact categories other than *eutrophication*. Considering Option 3 from the brewery's point of view, it would not offer any appreciable environmental benefits as compared to the current case of operation (Option 1). The difference between Options 1 and 3 across all impact categories lies between 0 and 10%. However, Option 3 has the burden of having to install, run and maintain the yeast filtration equipment. As such, Option 1 remains preferable to Option 3. The choice between Option 1 and Option 2 is not obvious when considering the results presented in Figure 6.11 and Figure 6.12, as Option 2 offers better environmental relief for some impact categories but presents higher burden with respect to *eutrophication* as compared Option 1, thus requiring a trade-off. It was thus, important to consider the possibility of reducing the *eutrophication* impact associated with Option 2. To achieve this, the main source of *eutrophication* had to be identified. Further analyses to track factors contributing to the observed environmental impact scores are presented in subsequent sections.

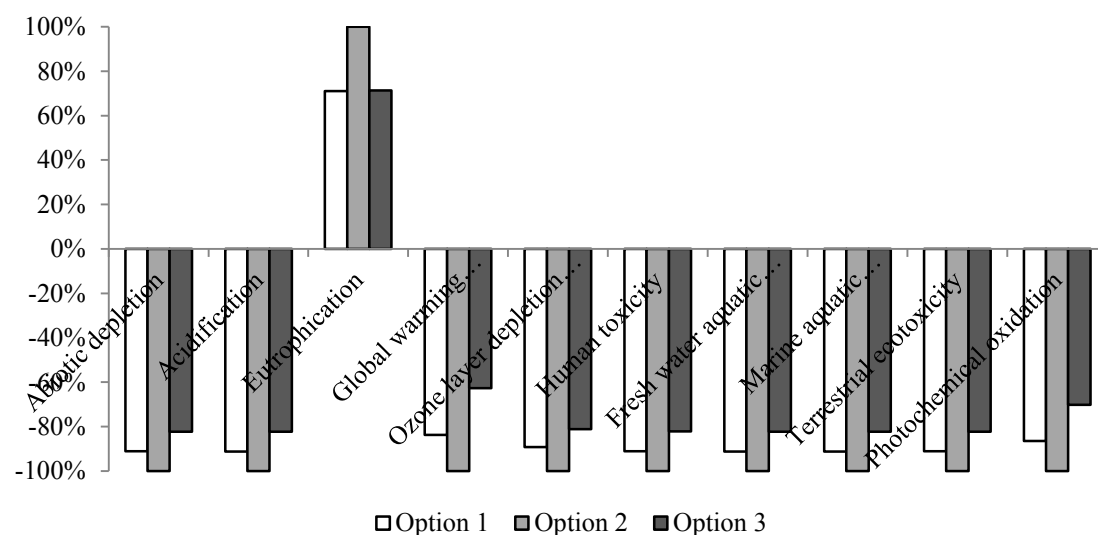


Figure 6.11: Impact assessment based on kg-COD as functional unit: A characterisation comparison of Option 1, Option 2 and Option 3.

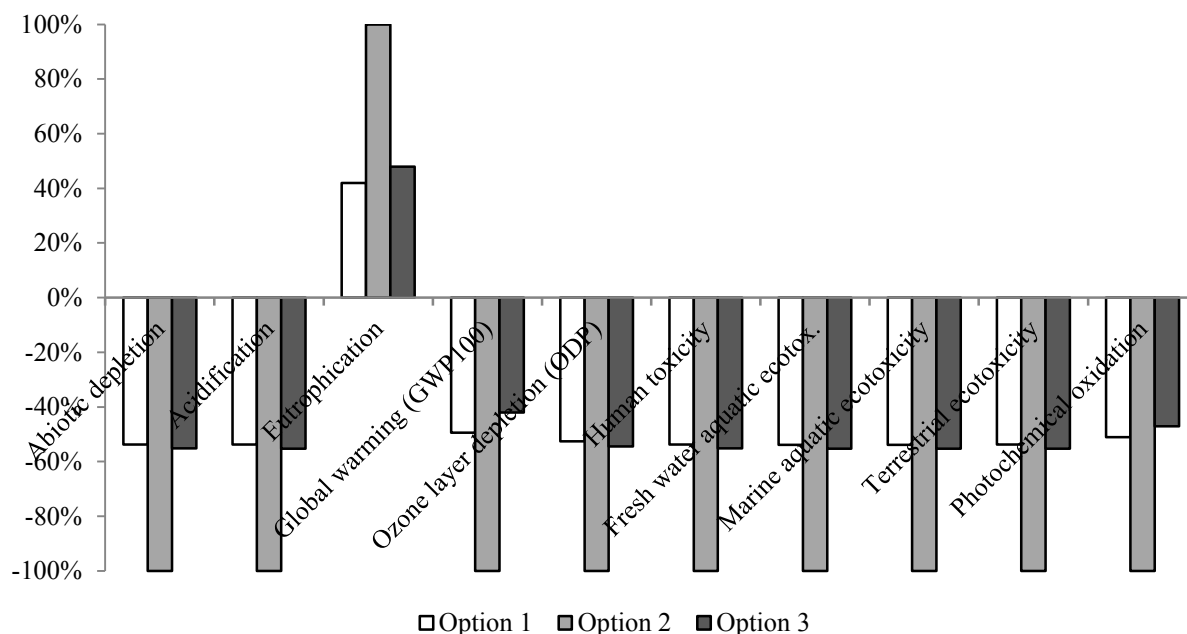


Figure 6.12: Impact assessment based on L-beer as functional unit: A characterisation comparison of Option 1, Option 2 and Option 3.

6.5.3 Contributing processes

Figure 6.13, Figure 6.14 and Figure 6.15 show major contributing processes to each impact category for Option 1, Option 2 and Option 3 respectively. These graphs show the source of the contribution to the impact scores. For example, the contribution of the Eskom Power Mix is caused by the emissions created (e.g. CO₂) and resources used (e.g. coal) by processes within the electricity mix. Similarly, contribution of the AWTW is caused by the emissions created (e.g. NO₃ and CH₄) and resources used (electricity for pumping and water treatment) at the municipal treatment works in Athlone. The SABWTP is the central process and therefore, contributions of other processes to the environmental impacts are incurred in support of this process. Input processes from technosphere include the Eskom Power Mix and the Caustic process, which delivering electricity and calcium carbonate respectively. The Boiler process takes in the gaseous emissions (largely CH₄) from the SABWTP and turns the methane component into energy contained in steam, but described as the equivalent electricity required to generate this steam according to the Eskom Power Mix. The AWTW treats its aqueous effluent removing, for example, N and P.

The net effect of the Boiler process in each scenario is environmental relief due to production of equivalent electricity. This suggested that the benefits of replacing electricity from the Eskom Power Mix outweigh the impacts of emissions exiting the Boiler processes and the impact of AWTW processes. The AWTW processes showed significant contributions to *eutrophication*, *global warming potential* and *photochemical oxidation* impact scores. The AWTW contributions to *eutrophication* were

approximately 92%, 82% and 84% for Option1, Option 2 and Option 3 respectively. However, the process's contributions to *global warming* and *photochemical oxidation* were roughly the same ranging between 22% and 24%, and 28% and 30% respectively across the scenarios.

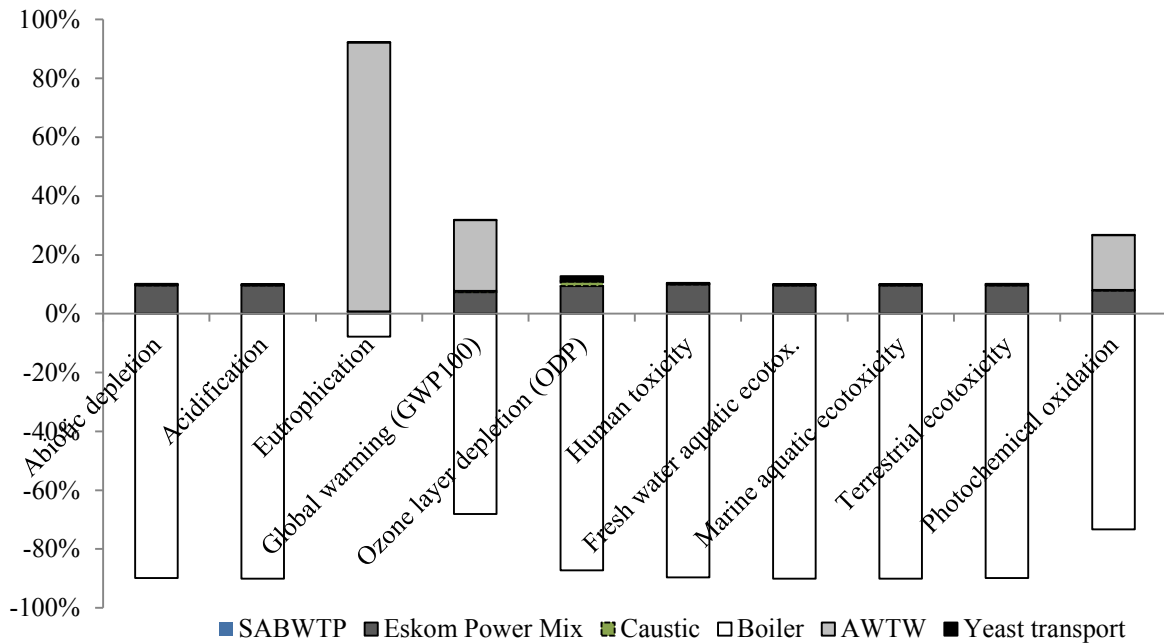


Figure 6.13: Contribution distributions for Option 1, with kilogram of COD as functional unit.

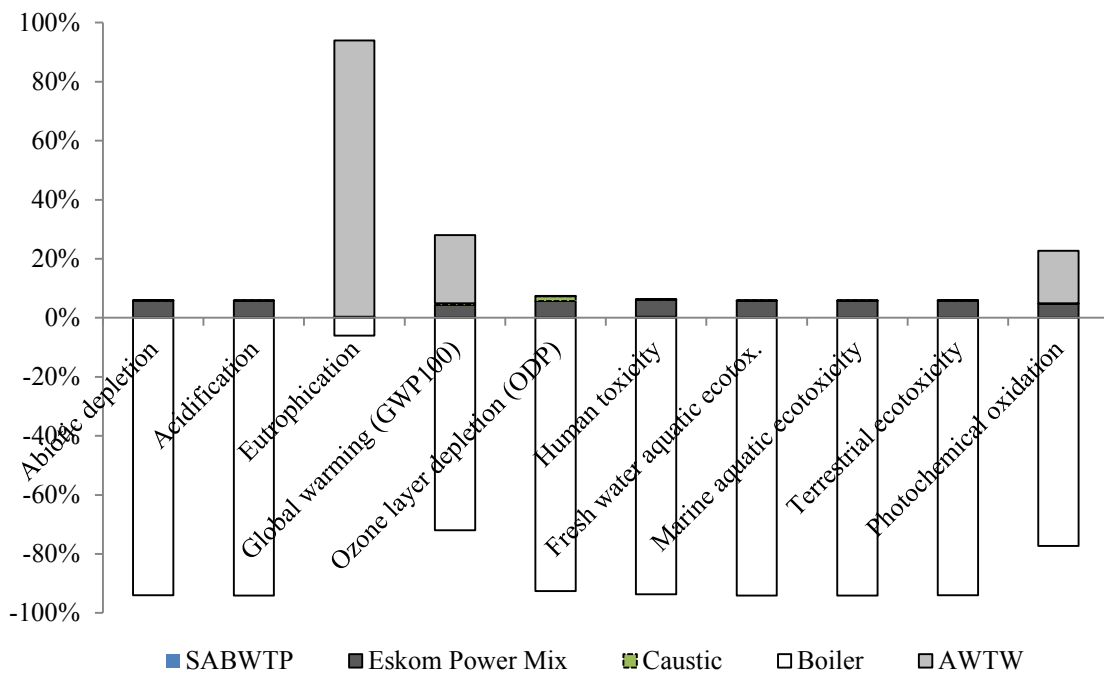


Figure 6.14: Contribution distributions for Option 2, with kilogram of COD as functional unit.

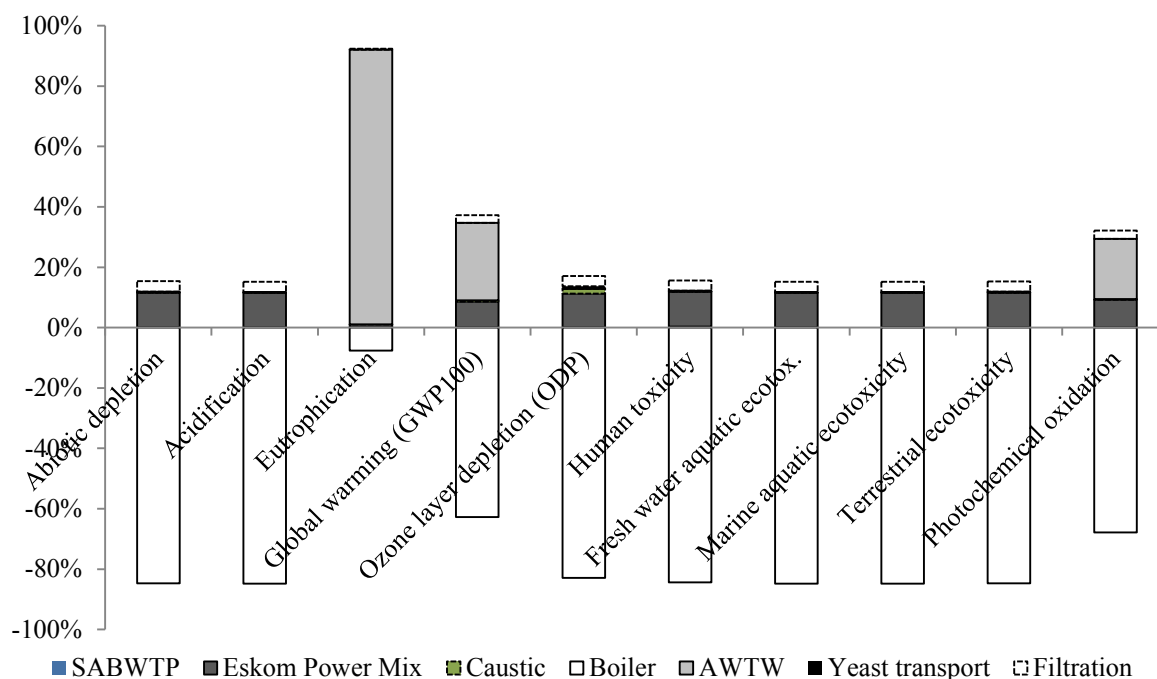


Figure 6.15: Contribution distributions for Option 3, with kilogram of COD as functional unit.

Abiotic depletion, acidification, human toxicity, and fresh water, marine and terrestrial ecotoxicity, and ozone layer depletion

In each impact category, except for *eutrophication*, the impacts associated with electricity were dominant as seen by the contribution of the Boiler and Eskom Power Mix processes. The scores associated electricity contributed almost 100% to *abiotic depletion*, *acidification* and the toxicity impact categories. Within these, the effect of the electricity replaced by the Boiler process is dominant, followed by electricity consumption in the three scenarios. As such, the differences in environmental impact between the three scenarios with respect to these impact categories are entirely attributable to the net energy output of the each scenario.

Since the net energy produced per COD consumed is the same across the scenarios, the variation in net energy produced depended on electricity consumption. The contribution of electricity input (consumption) to the scores of these impact categories for Option 1, Option 2 and Option 3 are 12%, 8% and 18% respectively (per kilogram COD fed). For Option 3, the 18% constituted of 14% and 4% electricity to run the SABWTP and yeast filtration respectively. The higher energy consumption of Option 1 and Option 3 as compared to Option 2 was due to the higher COD concentration of feed for Option 2, in which more COD was fed at volumetric flowrates relatively similar to Option 1 and Option 3. Hence the power consumed, which is related volumetric flowrate, per COD fed was lowest for Option 2. When removing the contribution of the filtration process (4%), the energy contribution of Option 3

remains higher than the 8% of Option 2. This suggested that Option 3 is less favourable than Option 2 even when the impact contribution from filtration is not included, due to the lower COD concentration of the feed. As such, implementation of less energy intensive solid-liquid separation process such as sedimentation would not change the outcome of the comparison between these two scenarios. The *ozone layer depletion* scores showed similar distribution to *abiotic depletion*, and the toxicity impact categories, but with increased contribution by the Quicklime process across all scenarios and Yeast transport processes in the case of Option 1 and Option 3.

Figure 6.16 shows that environmental impact scores for each of the abovementioned categories were predominantly associated with coal-fired generation of electricity with just over 94% contribution for *human toxicity*, 96% for *terrestrial ecotoxicity* and 98% for the remaining categories. Figure 6.17 shows the distribution of processes contributing to impacts associated with coal-based electricity generation, where the Electricity, coal (South Africa) process represents electricity generation at power station and Hard coal mix, at regional storage/UCTE S process represents the process of mining, processing and delivering coal to storage (i.e. coal acquisition process).

Acidification refers to the increase in soil and water acidity. Almost 100% of the *acidification* scores were associated with coal-based electricity with over 90% contribution from coal power plant emissions and the remainder from the coal acquisition processes (see Figure 6.16 and Figure 6.17). Figure 6.18 shows that *acidification* attributable to electricity generation at the coal-fired power station was caused by SO_x followed by NO_x airborne emissions. Coal usually contains sulphur and nitrogen which, upon combustion, turn into gaseous sulphur oxides and nitrogen oxides respectively.

The ozone layer depletion scores comprised impacts from coal-based electricity at about 90%, of which over 80% is attributed to the coal power plant, followed by nuclear-based electricity and hydropower at pumped storage at approximately 7% and 3% respectively (see Figure 6.16). Similarly, the **human toxicity** scores were comprised of impacts from coal-based electricity, nuclear-based electricity and hydropower at pumped storage of about 94%, 4% and 2% respectively. Of the 94% contribution by coal-based electricity, approximately 85% of the impact was generated at the coal acquisition phase with the remaining 15% generated at the coal power plant. The **fresh water aquatic ecotoxicity** and **marine aquatic ecotoxicity** scores were contributed by approximately 98% coal-based electricity of which almost 100% was associated with coal acquisition. However, the coal-based electricity contributed about 97% to the **terrestrial ecotoxicity** scores, of which about 80% was contributed by the power plant with the remainder attributed to the coal acquisition processes. The distributions of substances contributing to *ozone layer depletion* and the toxicity impact scores for the coal power plant and coal acquisition processes are shown by Figure C.4, Figure C.5, Figure C.6, Figure C.7 and Figure C.11 in the Appendix.

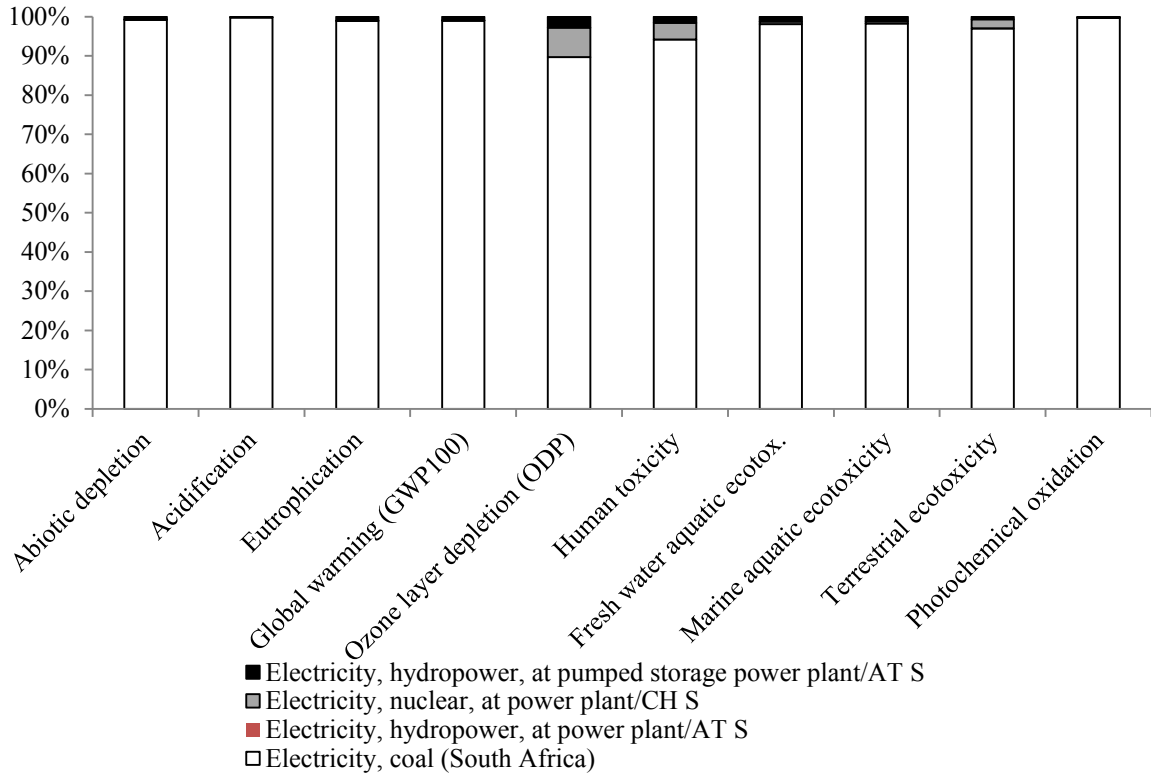


Figure 6.16: Composition of processes contributing to environmental impacts associated with the Eskom Power Mix.

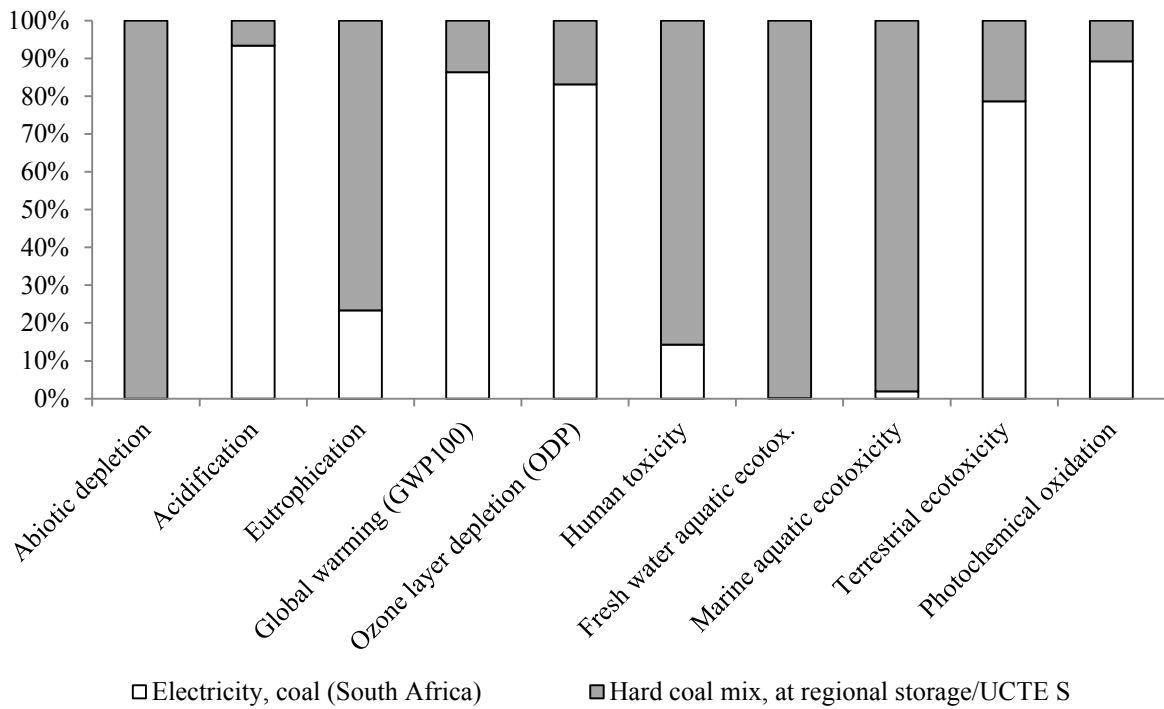


Figure 6.17: Composition of processes contributing to environmental impacts associated with the Electricity, coal (South Africa).

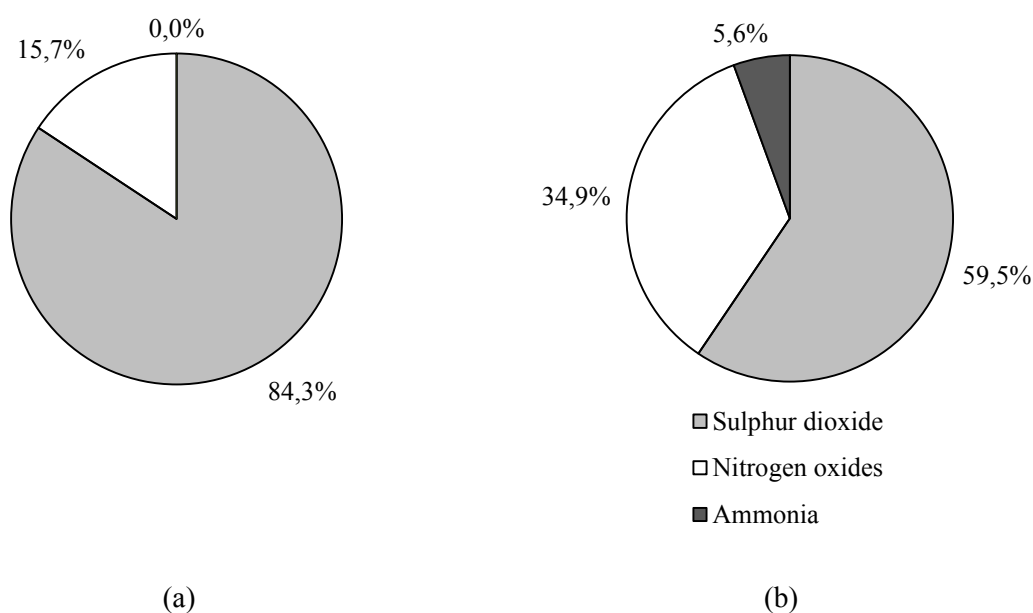


Figure 6.18: Distribution of contributing substances to *acidification* at: (a) the coal power plant (*Electricity, coal (South Africa)*) and; (b) coal acquisition processes (*Hard coal mix, at regional storage/UCTE S*).

Eutrophication

Figure 6.13, Figure 6.14 and Figure 6.15 showed that, for each scenario, the *eutrophication* burden was associated with the AWTW process. Table 6.6 shows the contributions of various processes to the *eutrophication* impact of the AWTW for the three scenarios. In each case, the impact caused by direct emissions from the AWTW contributed entirely to the impact score with the other processes contributing four orders of magnitude less. As such, the *eutrophication* impacts associated with AWTW in the process boundaries resulted directly from emissions of this secondary treatment. Therefore, all *eutrophication-causing* substances emitted from the treatment facility make up the contribution by AWTW. However, it is worth noting that the *eutrophication-causing* substances are not generated at the AWTW, though they are emitted to the environmental at through this facility. As such, further treatment of the SABWTP effluent shifted the environmental burden to the AWTW.

It was previously discussed that the AWTW does not perform the same across the processing scenarios (see Section 6.4.4). Therefore, to further understand the source of the *eutrophication* impact for each processing scenario, how they compare and how it can be mitigated, it was necessary to decouple the impact caused by direct material flows from the SABWTP and the effect of AWTW efficiency in response to the changing nutrient loading. Figure 6.19 shows a comparison of *eutrophication* scores of

the AWTW operating at the three scenarios and the distribution of contributing substances. A litre of wastewater entering the AWTW was used as a basis.

Chemical oxygen demand (COD) contributed the least to the *eutrophication* scores at 2.5%, 1% and 2.2% for Option 1, Option 2 and Option 3 respectively. Nitrogen and phosphorus were significant contributors in various forms. Nitrogen in the form of nitrate in air was the biggest contributor to *eutrophication* making up 53%, 48% and 54% of the scores for Option 1, Option 2 and Option 3 respectively. Nitrogen in soil was the second largest contributor resulting in combined N contributions of 73.6%, 65.3% and 74.8% in air and soil for Option 1, Option 2 and Option 3 respectively. The lower contribution of N for Option 2 corresponded to an increase in phosphorus in water contribution. This highlighted the impact of increased phosphorus from the spent yeast relative to the increase in nitrogen loading.

The increase in phosphorus in soil contribution was attributed to the need to reduce P concentration in water to a compliant 8.7 mg-P/L, thereby removal more than 20% which was the set conversion for Option 1 and Option 2. Since nitrate contribution was an inevitable result of treatment and N to soil was a result of valorisation of the sludge through agriculture, the key focus for improvement of the *eutrophication* impact for Option 2 should be reducing the Phosphorus in water.

Table 6.6: Process contribution to *eutrophication* impact associated with AWTW.

Process	Unit	AWTW - Option 1	AWTW - Option 2	AWTW - Option 3
Total of all processes	kg-PO ₄ eq	0.1079	0.2834	0.1240
AWTW - Option 1	kg-PO ₄ eq	0.1079	x	x
AWTW - Option 2	kg-PO ₄ eq	x	0.2834	x
AWTW - Option 3	kg-PO ₄ eq	x	x	0.1240
Coal acquisition	kg-PO ₄ eq	1.619x10 ⁻⁵	1.619x10 ⁻⁵	1.619x10 ⁻⁵
Electricity, coal (South Africa)	kg-PO ₄ eq	4.940x10 ⁻⁶	4.940x10 ⁻⁶	4.940x10 ⁻⁶
Hydropower, at pumped storage power plant/AT S	kg-PO ₄ eq	2.211x10 ⁻⁷	2.211x10 ⁻⁷	2.211x10 ⁻⁷
Nuclear power	kg-PO ₄ eq	1.384x10 ⁻⁸	1.384x10 ⁻⁸	1.384x10 ⁻⁸
Hydropower, at power plant	kg-PO ₄ eq	4.011x10 ⁻¹⁰	4.011x10 ⁻¹⁰	4.011x10 ⁻¹⁰

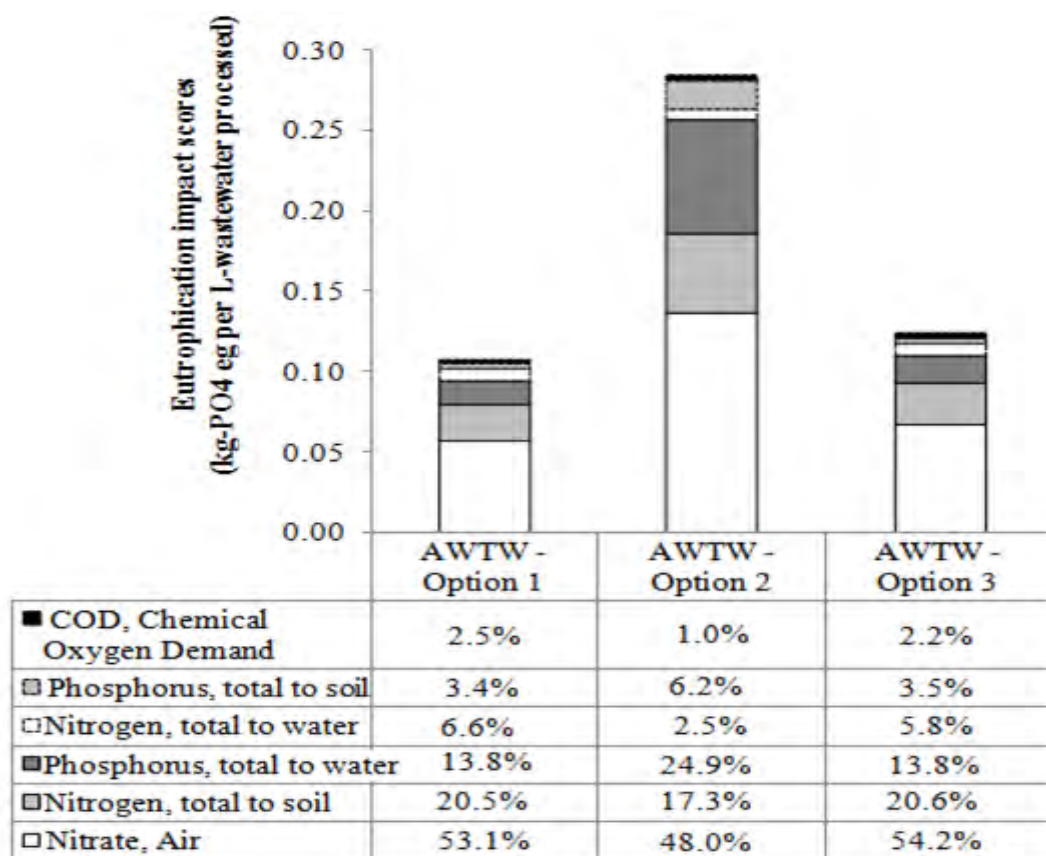


Figure 6.19: Substance contribution to *eutrophication* impact at AWTW with a L-wastewater as unit of basis.

Global warming potential (GWP) and photochemical oxidation

Global warming impact scores are mainly associated with electricity processes with some contribution from the AWTW for all scenarios. The net positive energy production provided by the Boiler resulted in displacement of *global warming* associated with the Eskom Power Mix. In this mix, coal-generated electricity accounted for over 99% of the *global warming* score according to Figure 6.16. Substances emitted at the coal power station accounted for approximately 85% of the impact score with the remaining 15% exerted within the coal acquisition process (see Figure 6.17). The substance contributing to the environmental impact at the coal power stations is CO₂, where as 79% and 21% of the impact score from the coal acquisition process is attributed to N₂O and CO₂ respectively.

The impact scores for *photochemical oxidation* are mainly attributed to the electricity processes followed by AWTW (see Figure 6.13, Figure 6.14 and Figure 6.15). Figure 6.16 showed that over 99% of the *photochemical oxidation* impact scores were attributable to coal based electricity. Figure 6.17 showed that approximately 90% of that was attributable to emissions at the coal power station with the remainder coming from the coal acquisition processes. The contribution of AWTW resulted mainly from the direct air-borne emissions, with a small contribution from electricity consumed by the facility as presented by Table 6.7. As with *global warming* potential, the *photochemical oxidation* impact score

associated with AWTW resulted from the CH₄ emission, despite the lower amount of this substance relative to nitrate (see Figure 6.21).

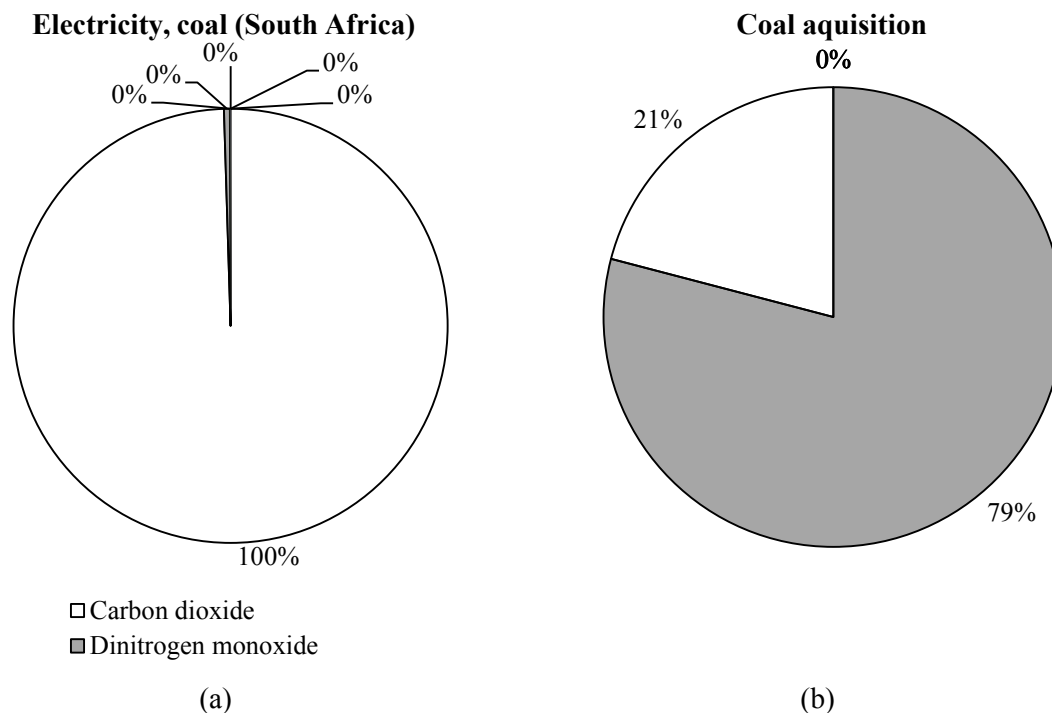


Figure 6.20: Substances contributing to global warming by: (a) coal power station (Electricity, coal (South Africa) and; (b) coal mining and processing (Coal acquisition process).

Table 6.7: Process contribution to photochemical oxidation impact associated with AWTW.

Process	Unit	AWTW - Option 1	AWTW - Option 2	AWTW - Option 3
Total of all processes	kg C ₂ H ₄ eq	5.02x10 ⁻⁴	9.00x10 ⁻⁴	6.54x10 ⁻⁴
AWTW - Option 1	kg C ₂ H ₄ eq	4.97x10 ⁻⁴	x	X
AWTW - Option 2	kg C ₂ H ₄ eq	x	8.96x10 ⁻⁴	x
AWTW - Option 3	kg C ₂ H ₄ eq	x	x	6.50x10 ⁻⁴
Electricity, coal (South Africa)	kg C ₂ H ₄ eq	4.07x10 ⁻⁶	4.07x10 ⁻⁶	4.07x10 ⁻⁶
Coal acquisition	kg C ₂ H ₄ eq	4.92x10 ⁻⁷	4.92x10 ⁻⁷	4.92x10 ⁻⁷
Hydropower, at pumped storage power plant/AT S	kg C ₂ H ₄ eq	1.03x10 ⁻⁸	1.03x10 ⁻⁸	1.03x10 ⁻⁸
Nuclear power	kg C ₂ H ₄ eq	2.00x10 ⁻⁹	2.00x10 ⁻⁹	2.00x10 ⁻⁹
Hydropower, at power plant/AT S	kg C ₂ H ₄ eq	6.26x10 ⁻¹¹	6.26x10 ⁻¹¹	6.26x10 ⁻¹¹

Table 6.8 presents process contribution to *global warming* scores associated with the AWTW. In each scenario, the impact scores were made up almost entirely by air-borne emissions directly from the AWTW. The impact associated with the use of electricity was three orders of magnitude lower than the impact from AWTW emissions. Figure 6.21 illustrates the composition of air-borne emissions from the AWTW and the resulting contribution to the *global warming* potential score. Although significant amounts of CO₂ and NO₃ were emitted relative to the amount of CH₄, the AWTW contribution to the *global warming* impact score resulted dominantly from the production of CH₄ resulting from reduction of COD. This illustrated the impact of CH₄ relative to other air-borne substances. The *global warming* contribution of the AWTW can be reduced significantly (25 fold) by burning the methane to CO₂ since CH₄ is 25 times more harmful than CO₂ in terms of *global warming*. The AWTW contribution to the *global warming* impact (shown in Figure 6.13, Figure 6.14 and Figure 6.15) can be eliminated by recovering the CH₄ as energy. The resulting replacement of electricity will also contribute towards reducing other impacts associated with electricity production.

Table 6.8: Process contribution to *global warming* potential impact associated with AWTW.

Process	Unit	AWTW - Option 1	AWTW - Option 2	AWTW - Option 3
Total of all processes	kg-CO ₂ eq	1.669	2.997	2.177
AWTW - Option 1	kg-CO ₂ eq	1.658	x	x
AWTW - Option 2	kg-CO ₂ eq	x	2.986	x
AWTW - Option 3	kg-CO ₂ eq	x	x	2.166
Electricity, coal (South Africa)	kg-CO ₂ eq	9.38x10 ⁻³	9.38x10 ⁻³	9.38x10 ⁻³
Coal acquisition	kg-CO ₂ eq	1.48x10 ⁻³	1.48x10 ⁻³	1.48x10 ⁻³
Hydropower, at pumped storage power plant/AT S	kg-CO ₂ eq	9.93x10 ⁻⁵	9.93x10 ⁻⁵	9.93x10 ⁻⁵
Nuclear power	kg-CO ₂ eq	6.14x10 ⁻⁶	6.14x10 ⁻⁶	6.14x10 ⁻⁶
Hydropower, at power plant/AT S	kg-CO ₂ eq	3.07x10 ⁻⁷	3.07x10 ⁻⁷	3.07x10 ⁻⁷

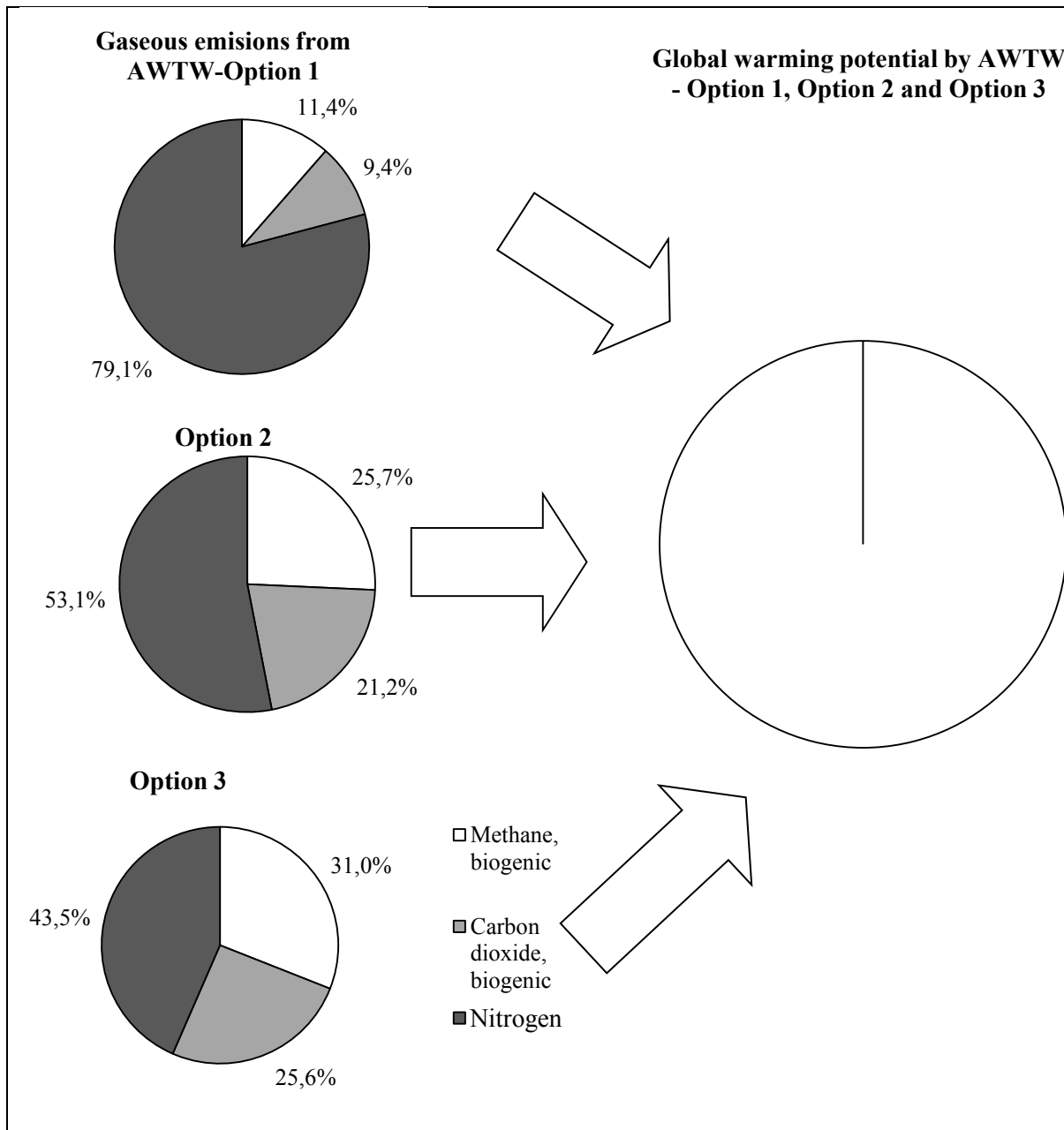


Figure 6.21: Inventory contribution of air-borne emissions (LHS) and their contribution to global warming potential for AWTW (RHS) (i.e. from inventory to impact), when the methane is not flared or used for energy generation.

6.6 Further improvement: Effect of operational changes on environmental impact results

6.6.1 Introduction

Sections 0 and 6.5 presented inventory and environmental assessment based on average values from data acquired over a three year period of operation of the SABWTP. However, operational efficiency varies monthly and seasonally. It was, thus, important to consider seasons of improved operational efficiency as a goal towards which to develop as well as to quantify the potential environmental benefits. Further, the environmental benefits of considering other improvement scenarios such as valorising the methane from the AWTW were useful to consider. In this section, analyses and discussions on various improvement scenarios are presented. The main purpose of the section is to address other possibilities of operational efficiency both by the brewery and at the SABWTP.

6.6.2 Effect of water use efficiency in the brewery

As shown before, the wastewater concentration entering the SABWTP is variable and depends on the water usage efficiency of the brewery. The amount of waste COD produced by the brewery is a function of the production load. However, the volume of wastewater generated depends on the amount of COD to be cleaned from the brewery and the water use efficiency. Therefore, the volume of wastewater generated per amount of COD cleaned is an indication of the water use efficiency. The reciprocal of volume of wastewater per COD cleaned is the concentration of the wastewater stream. Therefore a wastewater stream with a high COD content indicates more efficient use of water by the brewery.

The period between March 2010 and September 2010 represented a period of efficient water use by the brewery, reflected by the higher wastewater concentration of around 3000 mg-COD/L. To investigate the impact of this water efficiency, the data was re-analysed, using the same cumulative COD load, but a water flow equating to a COD concentration of 3000 mg COD/L. The same amount of COD would be generated over the period but water used to get rid of that COD can be improved. Hence the cumulative COD was used with the cumulative volume of water changed to result in the set concentration of 3000 mg-COD/L. This approach increased the hypothetical wastewater concentration in the feed from 2290 mg/L to 3000 mg/L which corresponded to an increase in hydraulic retention time (HRT) from 1.8 days to 2.3 days, while keeping the COD loading constant. Table 6.9 shows the predicted effluent CODs for the two influent concentrations. The longer hydraulic retention time and the higher COD degradation rates allowed by the higher COD per volume wastewater resulted in a lower overall COD loading to the AWTW. This suggested that more COD is valorised into useful energy through use of the methane generated in the Boiler, thereby, reducing the environmental impact associated with electricity production as well as the environmental impact associated with release of methane to atmosphere at AWTW. Hence, environmental scores across all impact categories apart from *eutrophication* were expected to decrease accordingly.

Table 6.9: Improved COD reduction by SABWTP due to increased influent concentration.

Influent COD concentration	HRT	Kg-COD effluent/kg-COD influent		
		Option 1	Option 2	Option 3
2290 mg/L	1.3 day	0.2235	0.1920	0.2405
3000 mg/L	2.8 day	0.1884	0.1623	0.2015

Table 6.10: Inventories for the Municipal wastewater treatment (AWTW) works based on a litre of wastewater processes (L-influent) with SABWTP influent at 3000 mg-COD/L.

		MWT - Option 1	MWT - Option 2	MWT - Option 3
Inputs				
TCOD	kg/L-influent	6.85E-04	1.18x10 ⁻³	8.51x10 ⁻⁴
P	kg/L-influent	8.38x10 ⁻⁶	4.21x10 ⁻⁵	9.74x10 ⁻⁶
SO ₄	kg/L-influent	6.92x10 ⁻⁴	8.39x10 ⁻⁴	7.16x10 ⁻⁴
TKN	kg/L-influent	2.02x10 ⁻⁴	4.74x10 ⁻⁴	2.35x10 ⁻⁴
CaCO ₃	kg/L-influent	1.07x10 ⁻⁴	2.15 x10 ⁻⁴	1.25 x10 ⁻⁴
Inputs from technosphere				
Electricity	kWh/kg-influent	1.46 x10 ⁻⁵	1.46 x10 ⁻⁵	1.46 x10 ⁻⁵
Emissions to air				
CH ₄	kg/L-influent	1.00 x10 ⁻⁴	1.89 x10 ⁻⁴	1.30 x10 ⁻⁴
CO ₂	kg/L-influent	8.28 x10 ⁻⁵	1.56 x10 ⁻⁴	1.07 x10 ⁻⁴
NO ₃	kg/L-influent	4.98 x10 ⁻⁴	1.27 x10 ⁻³	5.92 x10 ⁻⁴
Emissions to water				
TCOD	kg/L-influent	1.25E x10 ⁻⁴	1.25 x10 ⁻⁴	1.25 x10 ⁻⁴
P	kg/L-influent	6.70 x10 ⁻⁶	3.37 x10 ⁻⁵	7.79 x10 ⁻⁶
SO ₄	kg/L-influent	6.92 x10 ⁻⁴	8.39 x10 ⁻⁴	7.16 x10 ⁻⁴
TKN	kg/L-influent	1.70 x10 ⁻⁵	1.70 x10 ⁻⁵	1.70 x10 ⁻⁵
CaCO ₃	kg/L-influent	1.07 x10 ⁻⁴	2.15 x10 ⁻⁴	1.25 x10 ⁻⁴

Emissions to soil, Agriculture				
P	kg/L-influent	1.68×10^{-6}	8.42×10^{-6}	1.95×10^{-6}
TKN	kg/L-influent	7.29×10^{-5}	1.71×10^{-4}	8.47×10^{-5}

Figure 6.22 shows a comparison of environmental impacts between the average wastewater COD concentration case and the higher wastewater COD concentration case across for the three processing scenarios across all impact categories. The comparison of the processing options at the high wastewater concentration showed the same trend as the average concentration case. This suggested that, within the concentration range, the outcome of the analyses to choose between the options would lead to the same conclusion regardless of the wastewater concentration, provided the SABWTP performance is not severely hampered. It was observed that the high wastewater concentration cases showed less environmental burden for *eutrophication* and more environmental relief for the other impact categories. All the increased relief in all environmental impact categories apart from *eutrophication* were due to replacement of the Eskom Electricity Mix by higher energy recovery. This corresponded to the lower *eutrophication* impacts which indicated lower COD disposal. Although the COD content of the wastewater leaving the SABWTP was higher using the influent at 3000 mg/l than at the average influent COD, the overall COD entering the AWTW was lower. This occurred due to the lower volumes of wastewater sent to the AWTW.

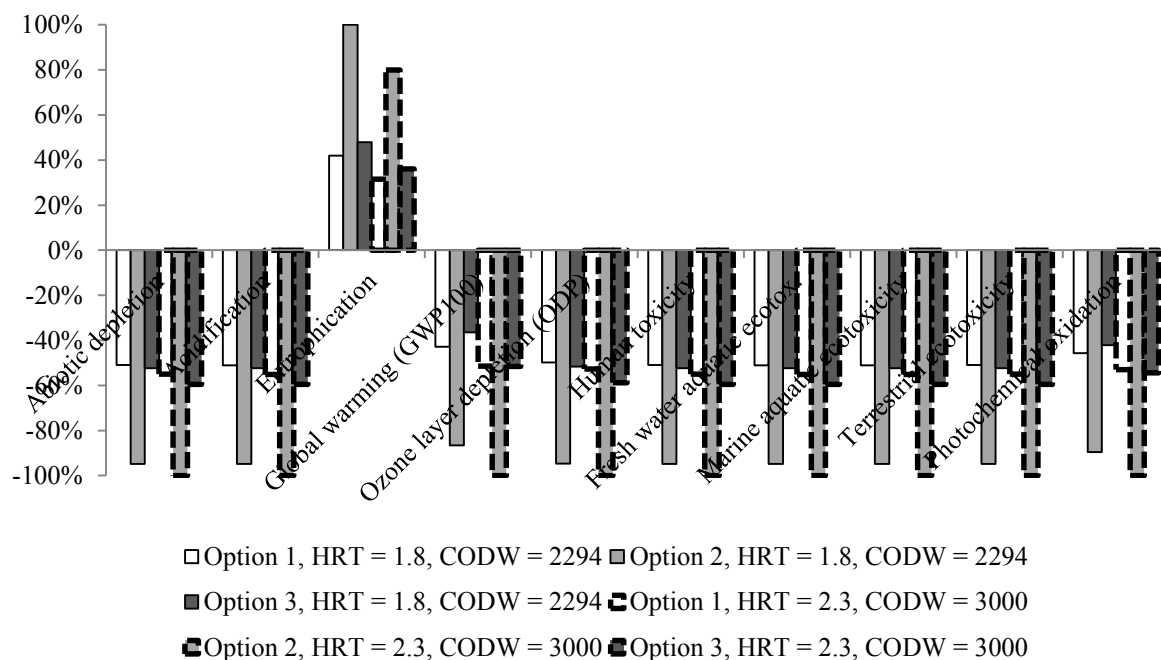


Figure 6.22: Improvement assessment: A predictive comparison between average COD concentration (2290 mg-COD/L) and high COD concentration (3000 mg-COD/L) of wastewater

entering the SABWTP at a HRT of 1.8 days and corresponding HRT of 2.3 days respectively, presented using a functional unit of one L of beer.

6.7 Considering burden associate with substitution of spent yeast as animal feed

6.7.1 Use of spent yeast in farming/ animal production

Brewer's spent yeast used as animal feed serves various functions. It can be considered as a protein source (or nitrogen source), a source of gluten which has various physiological advantages, or as an energy source. However, it is usually blended with a starchy or lignocellulosic material which provides all the energy requirements. In most cases, spent yeast is chosen to deliver protein or nitrogen. Therefore, the spent yeast has been considered as a protein source in subsequent analysis. Wilkins & Jones (2000) studied the alternative sources of protein for ruminants in the context of the United Kingdom and determined that more than 70% of crude protein (CP) consumed by ruminants was from grasslands, while cereals and oil seeds contributed about 14% and 10% respectively. Wilkins & Jones (2000) indicated that the losses in nitrogen (N) when grassland CP is used as ruminant feed are high with only 5% to 20% of the nitrogen fed being recovered as meat or milk. This suggested that the nitrogen retention or nitrogen utilisation efficiency differed depending on the type of feed chosen to deliver it. The complication in calculating the equivalent amount of replacement product, thus, arises from lack of knowledge about the nitrogen utilisation efficiencies and the necessary energy supplementation for both spent yeast and the replacer product.

6.7.2 The replacer product and its characteristics

In South Africa, the traditional energy source for feed for animal production is maize, grown in the northern regions. According to Brand et al. (2003) the costs of transporting maize to the Western Cape have prompted research into alternative feeds for this area. Studies are available on barley grains, naked oats, wheat and triticale with research spanning the mid-1980s to early 2000s. Brand et al. (2003) studied variations in chemical and physical properties of cereal grains produced in the Western Cape area of South Africa. Selected results are presented in Table 6.12 and Table 6.12. The samples were collected from 10 different locations.

Table 6.11: Crude protein and nitrogen composition of grains grown in the Western Cape region of South Africa. Values for Crude protein attained from Brand et al. (2003), values for N estimated as crude protein times 6.25.

Grain type	Crude protein		N	
	(g/kg)	(g/kg)	(g/kg)	(g/g)
Barley	136	21.76		2.2%

Triticale	148	23.68	2.4%
Oats	143	22.88	2.3%

Table 6.12: Chemical composition of grains grown in the Western Cape region of South Africa from Brand et al. (2003).

Grain type	Chemical Composition							
		Dry matter (g/kg)	Ash (g/kg)	Crude protein (g/kg)	Acid detergent fibre (g/kg)	Neutral detergent fibre (g/kg)	Ether extract (g/kg)	Non- structural carbohydrate (g/kg)
2-Row brewer's barley	mean (s.d.)	899 (7)	20 (0.4)	136 (2.6)	68 (11)	246 (43)	24 (4)	575 (52)
	n	103	103	103	103	103	64	64
6-Row feed barley	mean (s.d.)	901 (8)	21 (0.4)	135 (2.4)	92 (15)	289 (41)	24 (4)	527 (48)
	n	336	336	336	336	336	215	215
2-Row naked barley	mean (s.d.)	901 (4)	17 (0.2)	159 (2.5)	26 (4)	166 (23)	23 (2)	631 (31)
	n	36	36	36	35	32	18	16
6-Row naked barley	mean (s.d.)	902 (5)	18 (0.3)	154 (2.8)	31 (4)	185 (26)	25 (3)	617 (36)
	n	36	36	34	34	36	18	18
Wheat/triticale	mean (s.d.)	897 (7)	20 (0.9)	148 (2.3)	43 (10)	188 (52)	22 (2)	614 (65)
	n	39	39	39	39	39	19	19
Wheat	mean (s.d.)	896 (6)	15 (0.7)	148 (2.2)	36 (8)	160 (37)	25 (5)	650 (44)
	n	178	178	178	177	176	107	106
Triticale	mean (s.d.)	894 (7)	18 (0.7)	146 (2.3)	40 (10)	183 (43)	22 (4)	632 (44)
	n	300	300	300	298	298	200	198
Oats	mean (s.d.)	909 (8)	25 (0.5)	143 (2.2)	164 (23)	381 (48)	60 (15)	387 (43)
	n	308	308	308	303	305	192	189
Naked oats	mean (s.d.)	906 (11)	17 (0.4)	158 (2.7)	40 (11)	166 (39)	97 (9)	554 (48)
	n	103	103	103	103	103	66	66
LSD (mean) at P ≤ 0.05		0.2	0.2	0.4	0.4	1.6	0.3	1.8

6.7.3 Methodology

Environmental impact scores of various common nitrogen sources from the SimaPro™ data base were compared. The highest score from each impact category across all the replacer products were noted and taken as the scores of a new hybrid replacer product to be used in subsequent analyses. This approach ensures that the analysis is not biased in favour of feeding yeast to the SABWTP. The calculation procedure is presented by Equation 6.14 to Equation 6.18.

$$M_{N \text{ in SY to SABWTP}} = M_{\text{total N in SY produced}} - M_{N \text{ in SY to farmer}} \quad \text{Equation 6.14}$$

where

M_i [kg/L-beer] refer to mass of component i

∴

$$M_{N \text{ to be replaced at farm}} = M_{N \text{ in SY to SABWTP}} \quad \text{Equation 6.15}$$

but N retention or utilisation efficiency by the animal is different when delivered by spent yeast to when delivered by the spent yeast replacer

$$\therefore \quad \text{Equation 6.16}$$

$$M_{N \text{ to be replaced at farm-actual}} = \frac{M_{N \text{ in SY to SABWTP}}}{\xi}, \text{ where} \quad \text{Equation 6.17}$$

$$\xi = \frac{M_{N \text{ from replacer retained by animal}}}{M_{N \text{ from SY retained by animal}}}$$

\therefore

$$M_{\text{replacer needed}} = \frac{M_{N \text{ in SY to SABWTP}}}{\xi \times \beta}, \text{ where} \quad \text{Equation 6.18}$$

$$\beta \left[\frac{M_{N \text{ in replacer}}}{M_{\text{replacer}}} \right] \text{ is the } N \text{ content of the replacer on a mass basis}$$

$$\text{Combined score} = P + (M_{\text{replacer needed}} \times R), \text{ where} \quad \text{Equation 6.19}$$

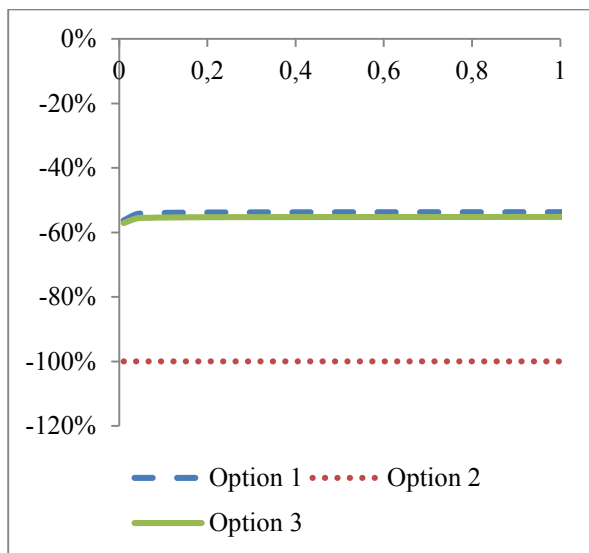
P [per L-beer] is impact score of the process excluding replacer,

$M_{\text{replacer needed}}$ [kg-replacer/L-beer] is the mass of replacer need and,

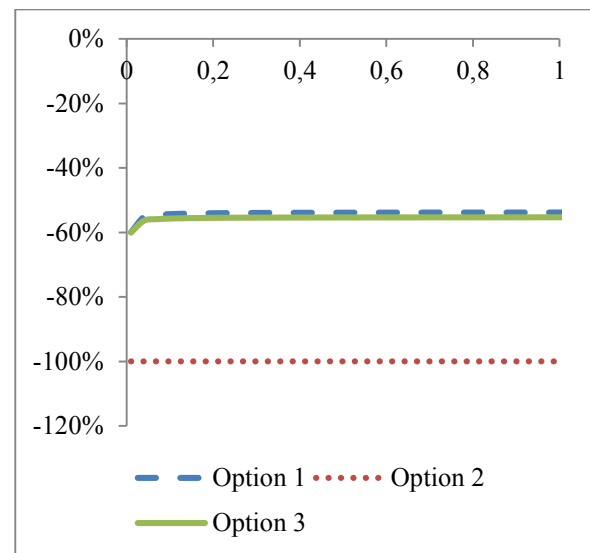
R [per kg-replacer] is the impact score of the replacer

The equations above were used to evaluate the effect of relative nitrogen utilisation efficiency (ξ). In this analysis, environmental scores for all three scenarios across were compared over ξ values ranging between 0.001 and 1 as shown by Figure 6.23 (a) to (j). In general, Option 1 showed lower environmental burdens at low ξ values. Increasing values of ξ resulted in relative improvement of environmental scores for Option 2 for all impact categories other than *eutrophication*. This was expected since an increasing relative efficiency corresponded to a decrease in the amount of replacer needed and therefore, the significance of the impacts associated with the replacer. The results were similar to those of the impact comparison excluding yeast replacement at ξ values above 0.1.

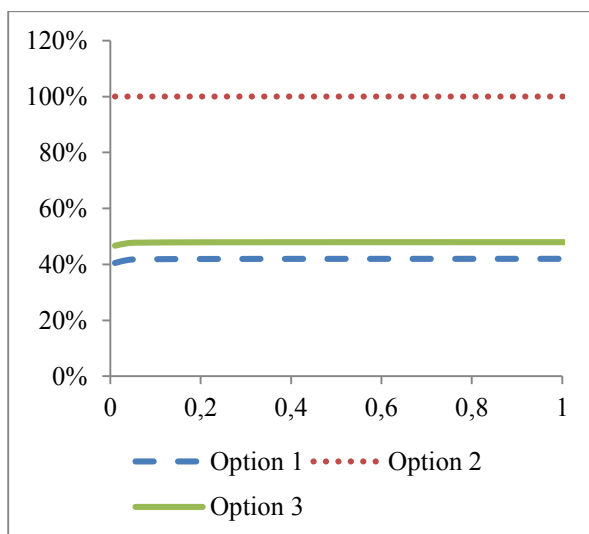
At ξ values greater than 0.1, the replacer did not contribute significantly to the score of the processes across all impact categories. This suggested that nitrogen utilisation from spent yeast would have to be 10 times more efficient than the replacer product for the burden to have any notable effect on the entire process. The burden of the replacer product showed notable effect to the impact scores of Option 1 and Option 3 relative to Option 2 at ξ values lower than 0.1 across the impact categories. For all categories other than *ozone layer depletion* and *terrestrial ecotoxicity*, the relative impacts changed by less than 10% between ξ values of 0.1 to 0.01. This corresponded to an increase in replacer requirement by 10 fold. The environmental relief with respect to *ozone layer depletion* for Option 2 decreased from 1.79 to 0.76 times the relief provided by Option 1. For *terrestrial ecotoxicity* the decrease in ξ values of 0.1 to 0.01 resulted in change of environmental contribution by Option 2 from relief 1.79 times to burden 0.69 times the relief provided Option 1.



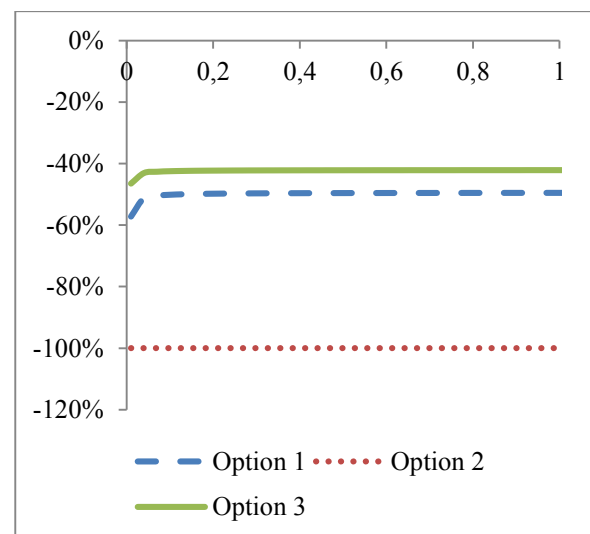
(a) Abiotic depletion



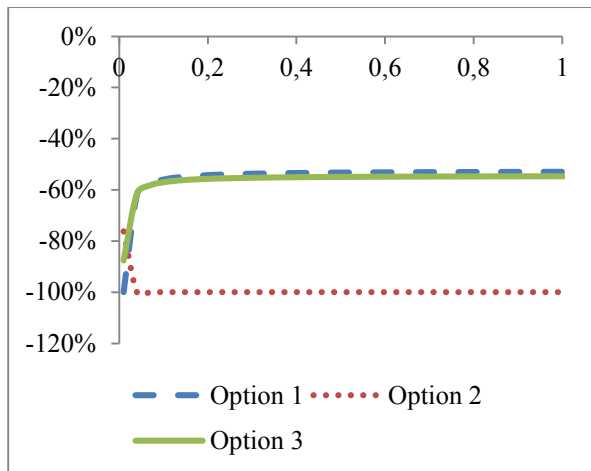
(b) Acidification



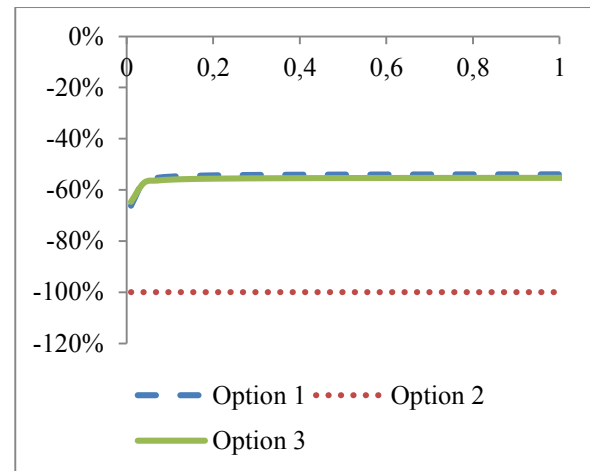
(c) Eutrophication



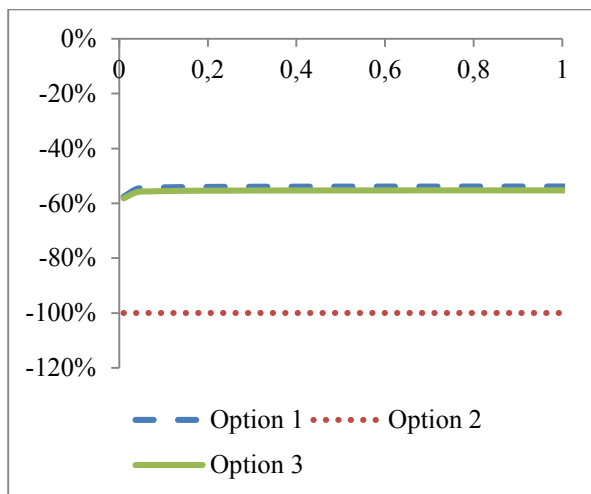
(d) Global warming (GWP100)



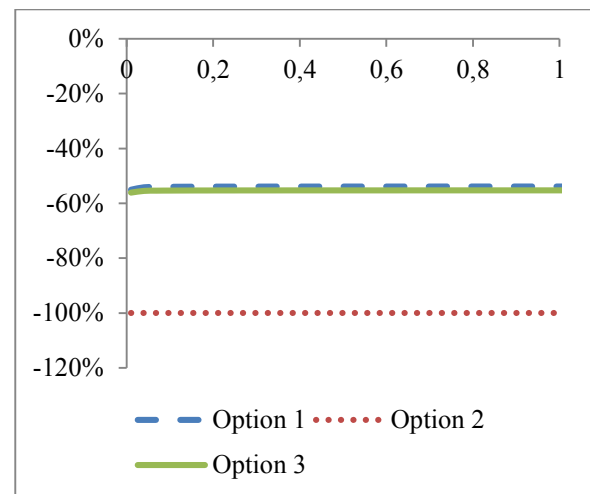
(e) Ozone layer depletion (ODP)



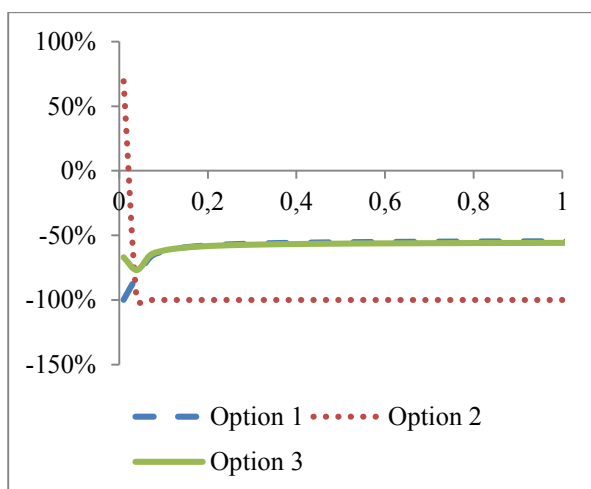
(f) Human toxicity



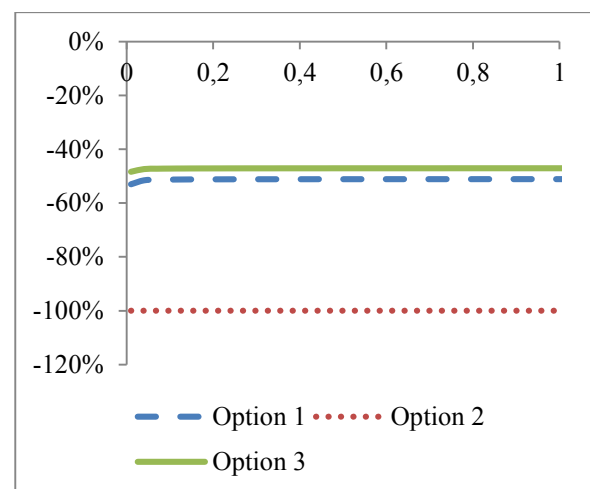
(g) Fresh water aquatic ecotoxicity



(h) Marine aquatic ecotoxicity



(i) Terrestrial ecotoxicity



(j) Photochemical oxidation

Figure 6.23: Illustration of the effect of relative efficiency of replacer product as compared to spent yeast (ξ) on the environmental impact (per litre of beer) of the processes under study. The nitrogen (N) content (β) of the replacer product was assumed to be 2.2wt%.

6.8 Summary of Chapter 6

Chapter 6 presented materials and energy inventories as well as environmental impact analysis of the brewery wastewater management processes. Further, the effect of enhancing the organic load to the anaerobic digester by adding brewer's spent yeast and spent yeast liquor was considered in terms of inventories and environmental impact assessment. Three processing scenarios were compared according to the system boundaries depicted in Figure 6.1, Figure 6.2 and Figure 6.3. Option 1 represented the current scenario in which only wastewater is processed through the SABWTP with brewer's spent yeast sent to the farm as animal feed. In Option 2, brewery wastewater and spent yeast provide the organic load to the SABWTP. In Option 3, following filtration of the spent yeast, its spent liquor, together with brewery wastewater, provided the organic load to the SABWTP, with the spent yeast concentrate sent to the farm as animal feed. This section presents a summary of important findings and conclusions drawn.

The importance of the functional units

Two functional units, kg-COD fed to SABWTP and L-beer produced by brewery, were considered as the bases for comparison. It was concluded that an environmental impact comparison with respect to kg-COD somewhat dampens the effect of increased organic loading, thereby providing insights into the change in environmental efficiency of the wastewater treatment system. On the other hand, analysis per L-beer produced is independent of the processing scenarios and provides a non-changing anchor through which the true effect of change in the amount of COD fed to the SABWTP can be evaluated. Figure 6.11 and Figure 6.12 in Section 6.4.5 show that the differences in the energy recovered by the wastewater management system across the three options was less pronounced when using kg-COD as a basis as compared to analysis per L-beer produced. Analysis of inventories with respect to kg-COD showed about 10% increase in energy recovery between Option 1 and Option 2, and about 7% decrease in energy recovery for Option 3. However, a similar analysis per L-beer produced showed increases in energy recovery of more than 45% and about 6% for Option 2 and Option 3 respectively. The results suggested that the efficiency of energy recovery increased 10% for Option 2 but the amount of energy recovered increased by over 45%, whereas the efficiency of energy recovery for Option 3 decreased by about 7% but the amount of energy recovered increased by 6%. The decrease in energy output efficiency of Option 3 was due to the power consumption of the filtration system which also contributed to the lower increase in total energy output. Lower methane productivity of Option 3 compared with Option 2 was due to the relatively lower OLR and reactor concentrations.

Environmental impacts: Process comparisons and impact contributors

For all three scenarios, net relief in environmental burdens, relative to a system where no energy is produced, was observed for all impact categories other than *eutrophication*. For all impact categories other than *eutrophication*, Option 2 had the best environmental impact scores, followed by Option 1,

with Option 3 showing the least favourable scores of the three per kg-COD fed to SABWTP. When analysing per L-beer produced, the *eutrophication* burden provided by Option 2 was almost twice that of Option 1 and Option 3 while twice the environmental relief was observed for all other impact categories. Relief was mostly the result of replacement of electricity from the Eskom Power Mix, and therefore, its associated emissions. The impact scores of Option 3 were worsened by emissions from electricity required to run the spent yeast filtration system. The impact scores for the three options are almost entirely associated with energy (electricity) for all impact categories other than *eutrophication*, *global warming* potential and *photochemical oxidation*. The high *eutrophication* impact caused by Option 2 was due to increased loading of N and P from the spent yeast. The *eutrophication* impact can be reduced by valorisation of the N and P. Based on the above; Option 2 is the best of the three options in offering reduced environmental impact in the case where N and P are valorised.

7 Conclusions and recommendations

7.1 Introduction

This work set out to evaluate the feasibility of using spent yeast and spent grain as an additional source of organic matter for the on-site anaerobic wastewater treatment plant (SABWTP) at Newlands Brewery (Cape Town, South Africa) to fill its un-used capacity and allow generation of more biogas to enhance energy efficiency of the overall process. The project also sought to analyse and compare the environmental impact of the proposed strategy against the current operation and to identify key contributing factors to environmental impact, towards which focus should be directed to further improve the environmental performance of the brewery wastewater management systems. While the project was conducted in the form of a case study, the approach used allowed generic knowledge and understanding to be built for application to other anaerobic digestion processes integrating waste treatment and energy efficiency. This chapter presents the brief overview of the work done and sets out the overall conclusions drawn from discussions presented in previous chapters. Recommendations for actions to be taken and future studies are also presented.

7.2 Anaerobic digestion of spent grain and spent yeast as feed to SABWTP

Experimental anaerobic digestion studies showed that, in the absence of intensive pre-treatment, spent grain did not degrade within 19 days. Hence its degradation is very slow, compared to the 1.8 day average residence time of the SABWTP operation. This low digestibility of the grains would result in accumulation of solids in the anaerobic digestion reactor, requiring frequent sludge removal, with minimal benefit of extra methane produced. Spent grain is, therefore, not suitable for feeding into the SABWTP, unless pre-treatment is considered.

Anaerobic digestion experiments on spent yeast using fed-batch reactor systems showed that almost all soluble COD and about 62% of the solid COD of the spent yeast was degraded within the first 4 days of batch treatment. The solid COD degraded rapidly until about 62% completion; thereafter it continued at a slower rate. This indicated a switch, on depletion of the readily degradable solids, to the slowly degradable portion of the solid yeast cells. The availability of readily utilisable organic matter did not hinder the hydrolysis of the solid yeast biomass. The average methane yield ($Y_{\text{CH}_4/\text{COD}}$) from digestion of spent yeast was 0.300 mL- $\text{CH}_4/\text{mg-COD}_{\text{reacted}}$. Co-digestion of spent yeast with the wastewater to increase methane productivity would, thus, be feasible.

The methanogenesis process is often the rate determining step in anaerobic digestion of readily digestible dissolved organics. However, hydrolysis may be rate-limiting during degradation of solids. As such, only these two processes are important to consider for the kinetics of anaerobic digestion. The current study, together with published research, has shown that the process of anaerobic digestion could

be modelled by describing the hydrolysis step using first order kinetics and soluble COD degradation using Monod kinetics. The specific rate of methane production was the same as the specific rate of soluble COD removal from the brewery wastewater. The removal of soluble COD during co-digestion of spent yeast and wastewater in the SABWTP was, thus, modelled using the same kinetic expression with the same kinetic constants. The rate of COD removal increases with increasing influent COD concentration. However, when the organic loading is too high, volatile acids may accumulate and reduce the pH of the system. This inhibits methanogenic activity. Efficiency with respect to the amount of methane produced per COD input increased (from 0.139 kg-CH₄/kg-COD_{influent} to 0.145 kg-CH₄/kg-COD_{influent}, corresponding to 78% and 81% conversion respectively) with increase in influent COD (from 2300 mg-COD/L to 3800 mg-COD/L) caused by addition of spent yeast with a reactor operating at the same hydraulic retention time of 1.8 day. However, efficiency with respect to wastewater treatment function of the SABWTP was reduced due to an increase in effluent COD concentration (from 513 mg-COD/L to 741 mg-COD/L) caused by the addition of spent yeast.

In conclusion, anaerobic digestion of spent yeast liquor and spent yeast cells produced methane. Their addition to the on-site anaerobic wastewater treatment plant is expected to intensify methane production per unit bioreactor volume. Higher influent COD concentrations achieved by addition of digestible organic resources, including spent yeast, increased the overall rate of anaerobic digestion, and therefore, the methane productivity (amount of methane per time per unit volume of reactor) of anaerobic wastewater treatment system in a manner limited by the balance in acid generation and methanogenic activity. Increasing influent COD improved space and time utilisation of the anaerobic digestion reactor (i.e. the amount of COD removed in a given space over a given time period), and therefore increased methane production efficiency, but reduced the wastewater treatment efficiency of the anaerobic digestion system. While these changes seem negligibly small, their environmental implications are significant, as discussed in the Section 7.3.

7.3 Effect of brewer's spent yeast feed on environmental performance of the brewery and its wastewater management systems

Environmental impact comparison of three processing options was carried out. Option 1 represented the current operation where only wastewater was fed to the SABWTP (Figure 6.1). In Option 2, the impact of feeding of the entire brewer's spent yeast into the SABWTP was modelled (see Figure 6.2). Option 3 considered separation of the yeast cells from the spent yeast liquor with the latter being fed to the SABWTP (see Figure 6.3). Material and energy inventories were prepared for each scenario. These scenarios were compared across ten environmental impact categories, namely: *abiotic depletion*, *acidification*, *eutrophication*, *global warming potential*, *ozone layer depletion*, *fresh water aquatic ecotoxicity*, *marine aquatic ecotoxicity*, *terrestrial ecotoxicity* and *photochemical oxidation*. The comparisons were represented as the score of the processing scenario relative to the highest score of the compared scenarios for each impact category (Equation 6.13).

Acidification, abiotic depletion, ozone layer depletion, fresh water aquatic, marine aquatic and terrestrial ecotoxicity

Relative impact scores between Option 1, Option 2 and Option 3 were similar across these impact categories in which the scores were dominantly (over 98%) associated with electricity. The environmental relief of the wastewater treatment system per kilogram of COD fed in Option 1 and Option 3 were about 10% and 20% of the relief presented by Option 2, respectively. However, these values changed to 50% for both Option 1 and Option 3 when a litre of beer produced by the brewery is used as a basis for comparison. This indicated that the environmental efficiency of the wastewater treatment system for Option 2 was 1.1 and 1.25 times greater than Option 1 and Option 3, respectively; whereas the actual environmental relief provided by Option 2 was approximately 2 times the relief provided by Option 1 and Option 3. The scores of these energy dependent categories are controlled by the net energy produced. Therefore, electricity use and energy recovery are the most important parameters to consider in terms of reducing environmental burdens with respect to these categories. Increasing organic loading into the on-site anaerobic digestion plant intensified methane production and net energy production by the wastewater management system, thereby reducing *acidification, abiotic depletion, ozone layer depletion, fresh water aquatic, marine aquatic and terrestrial ecotoxicity*.

In conclusion, addition of the brewery's spent yeast improved both environmental efficiency for a given capacity of the wastewater management system and improved the relief presented by the brewery wastewater management system and associated methane valorisation process. Increasing energy production and reducing energy consumption improved the relief presented by the wastewater management system.

Global warming potential and photochemical oxidation

Methane emissions released at AWTW from the residual COD in the SABWTP effluent water contributed the most (more than 99%) to the *global warming* burden of Option 1, followed by the CO₂ released from the coal-generated electricity consumed at SABWTP. Despite the increased COD loading to the AWTW which resulted in increased methane emissions, *global warming* relief was highest for Option 2. This was due to as the higher methane productivity and subsequent valorisation into energy at the SABWTP. The increase in influent COD concentration increased the portion of COD converted to methane (methane yield) at SABWTP and subsequent valorisation of the methane into energy increases *global warming* relief due to increased replacement of coal-based electricity. Similarly, *photochemical oxidation* was caused mainly by methane emissions at the AWTW which contributed more than 99% and was reduced by replacement of coal-based electricity. However, the balance was contributed by sulphur oxides released during coal-based electricity generations.

Eutrophication

Addition of the brewer's spent yeast to the feed stream increased the loading of N and P to the anaerobic digester. This was largely untreated by the anaerobic digestion at SABWTP which is designed to metabolise the organic carbon fraction of the wastewater. While the addition of spent yeast improved environmental performance with respect to the other impact categories, it increased the ***eutrophication*** burden due to this elevated N and P loading. This could be overcome by the post-anaerobic digester treatment of N and P prior to release or re-use of the water.

7.4 Effect of replacing spent yeast as animal feed on environmental performance of the wastewater management systems

Currently, Brewer's spent yeast is used as a source of protein and vitamins in animal feed. Re-purposing the spent yeast as an energy source through the on-site anaerobic digestion plant requires the farms to replace the spent yeast with another source of protein or nitrogen. While the environmental performance of the brewery and its wastewater management systems is independent of the burden associated with the replacement product, the implementation of the strategy affects the overall environmental impact. Burdens associated with the replacement product were included in the system boundary to evaluate the overall environmental impact of the strategy. The amount of the alternative product needed to replace the spent yeast and therefore, the associated impact score, depends on the relative nitrogen utilisation efficiency (ξ , defined as the amount of nitrogen retained by animal when delivered by the replacement over the amount of nitrogen retained by animal when delivered by spent yeast). The effect of the replacement product on the outcome of the environmental impact assessment was evaluated in terms of ξ .

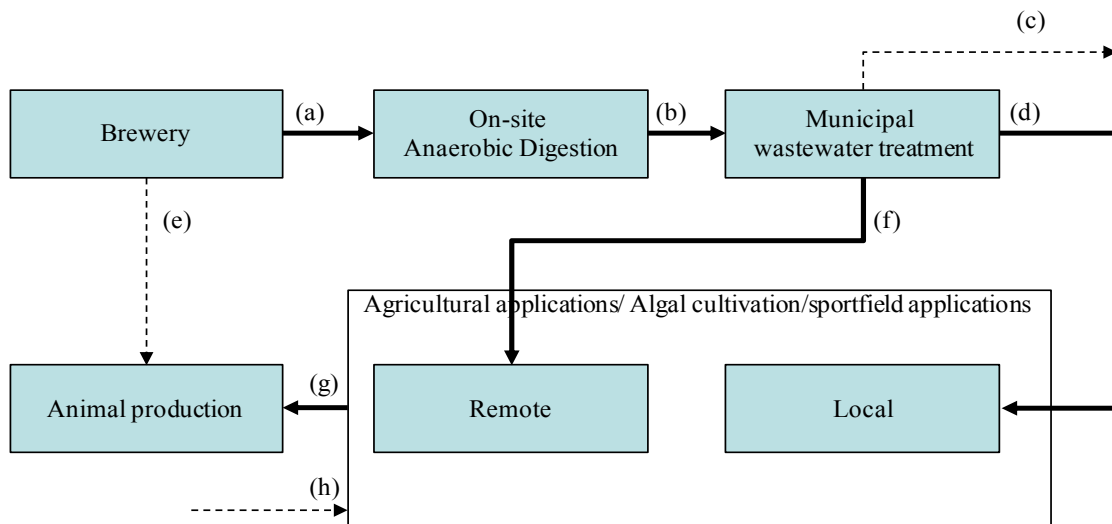
The contribution of the replacement product was negligibly low at ξ values greater than 0.1 across all impact categories, and resulted in less than 10% change in relative impacts when ξ decrease to 0.001 across all impact categories other than ***Ozone layer depletion*** and ***Terrestrial ecotoxicity***. ***Ozone layer depletion*** and ***Terrestrial ecotoxicity*** were more susceptible to the amount of replacement product needed at ξ values less than 0.1. In conclusion, considering the burden associated with replacement of spent yeast does not affect the results of the environmental impact assessment sufficiently to influence the decision making process when considering all impact categories. However, the replacement product burden influenced environmental assessment outcomes when the utilisation efficiency of nitrogen from spent yeast was more than ten times that of the replacement product.

7.5 Recommended further studies

Concerns of imbalances of microbial activities in the anaerobic digester caused by variations in COD loading to the on-site wastewater treatment plant, as well as the potential to utilise the spent yeast to control these variations were raised. It is recommended that the effect of variation in COD loading on

the efficiency ($\text{kg-COD}_{\text{removed}}/\text{L}_{\text{reactor}}\cdot\text{day}$) of the anaerobic digestion system be investigated using continuously operated laboratory reactor systems. This can be done by comparing the performance of a system fed with constant flow of COD against one fed with intermittent amounts of COD. The overall amount of COD fed to both systems should be equal. The possibility of damping spikes in COD loading may also be investigated should the variations in COD loading be detrimental to the digester operation. Other potential effects on the addition of spent yeast to the UASBR may include morphology and size of the sludge granules which affect settleability, and therefore, the allowed upflow velocity. The study may consist of operation of a laboratory scale UASBR and visual (microscopic) analysis of samples to evaluate the change in sludge granule morphology and size over time upon the introduction of spent yeast in the wastewater feed.

Methane emitted at the municipal wastewater treatment facility (AWTW) was found to contribute significantly to *global warming* potential and *photochemical oxidation*. It is recommended that the feasibility of capturing the methane from AWTW for use as an energy source be investigated. Should it not be feasible to valorise, the methane should be flared, converting into the less environmentally harmful carbon dioxide. Another potential valorisation opportunity is for the final effluent wastewater exiting the AWTW or SABWWTP to be used as nutrient-rich irrigation water in agricultural lands and sport fields etc. Figure 7.1 describes a potential system to investigate against the current brewery, farm and wastewater management circuit.



- a) Spent yeast sent to onsite AD plant to increase methane production, resulting in reduced dependency on coal-generated electricity.
- b) The stream to local municipal wastewater treatment works contains elevated amounts of nitrogen (N) and phosphorus (P).
- c) At the municipal treatment works, nitrification to gaseous nitrates is reduced resulting in less Eutrophication impact.
- d) Increased N and P in final effluent, which is valorised through irrigation on local agricultural land. The relative location of the treatment facility and the land is key consideration for energy expenditure due to pumping of the effluent.
- e) Reduced flow of material to farm in favour of increasing onsite AD organic loading
- f) Increased entrapment of N and P in sludge which subsequently applied to agricultural land. Dried sludge allows for longer distant transportation/delivery of N and P as compared to delivery via the effluent water stream (d).
- g) Agricultural products replace the brewery by-products as feed to animal production.
- h) N and P delivered as effluent water and sludge reduce fertiliser input to agricultural land, and thereby reduced impact associated with fertiliser production.

Figure 7.1: Representation and description of the system recommended for further investigation.

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**Maximising energy recovery from the brewery wastewater
treatment system: A case study evaluating the anaerobic
digestion wastewater treatment plant at SAB's Newlands
Brewery**

Appendices

A. Standard curves

A.1. Chemical oxygen demand (COD)

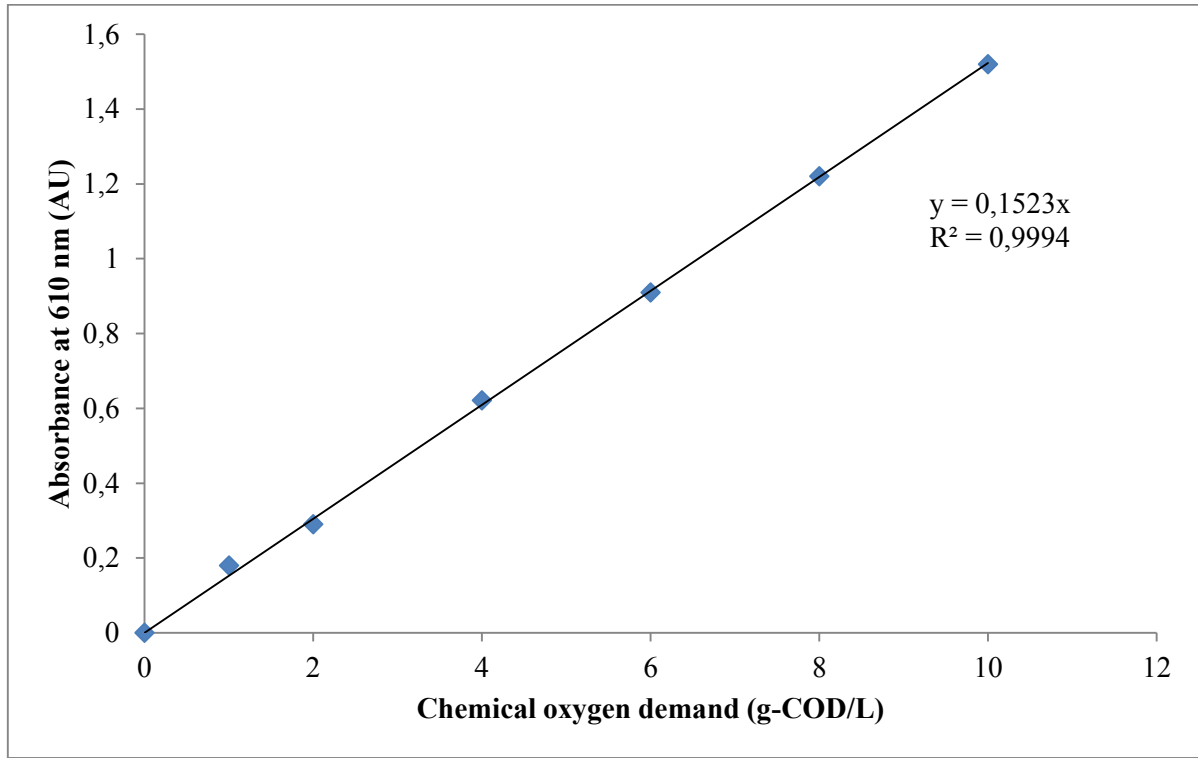


Figure A.1: Standard curve for chemical oxygen demand determination.

B. Development of equations and filtration theory

B.1. Solid COD concentration

$$M_{SY} = \rho_{YS} \times V_{YS},$$

Equation B.1

where

M_{YS} [mg] is the mass of the spent yeast suspension,

$\rho_{SY} \left[\frac{mg}{L} \right]$ is the density of the spent yeast suspension,

V_{SY} [L] is the volume of the spent yeast suspension

$$M_{cell} = x_{cell} \times \rho_{YS} \times V_{YS},$$

Equation B.2

where

M_{cell} [mg] is the mass of yeast cells (or biomass),

$x_{cell} \left[\frac{mg - cells}{mg - YS} \right]$ is the mass fraction of yeast cells in suspension

$$M_{COD,cell} = y \times x_{cell} \times \rho_{YS} \times V_{YS},$$

Equation B.3

where

$M_{COD,cell}$ [mg - COD] is the COD of spent yeast cells,

$y \left[\frac{mg - COD}{mg - dry cell} \right]$ is the COD content of the yeast cells on a dry basis

therefore

$$C_{Solid COD} = y \times x_{cell} \times \rho_{YS},$$

Equation B.4

where

$COD_{SolidCOD} \left[\frac{mg - COD}{L} \right]$ is the COD concentration attributable to the cells,

but

$$M_{SY}(1 - x_{cells}) = M_L = \rho_L \times V_{YS} (1 - c_{cells}),$$

Equation B.5

where

M_L and ρ_L are the mass and density of the spent yeast liquor respectively,

$c_{cells} \left[\frac{V_{dry cells}}{V_{suspention}} \right]$ is the volumetric concentration of cells in the suspension

therefore

$$\frac{M_{SY}}{V_{SY}} = \frac{\rho_L \times (1 - c_{cells})}{(1 - x_{cells})} = \rho_{YS} \quad \text{Equation B.6}$$

after substituting ρ_{YS} in

Equation B.4

Equation B.7

$$C_{SolidCOD} = \frac{y \times \rho_L \times x_{cells} \times (1 - c_{cells})}{(1 - x_{cells})}$$

B.2. Filtration theory

However, the pressure drop needed is a function of the compressibility of the filter cake (n) and the specific filtration resistance (α_0) associated with the cake (Svarovsky 1977). Zhou and Titus (1998) studied the filtration of two yeasts (*Cryptococcus albidus* and *Sacchromyces cerevisiae*) suspensions. They showed that filtration characteristics both yeasts can be adequately described by the correlation proposed by Svarovsky (1977). Values of n and α_0 were found to be 0.89 and 1.1×10^8 respectively (Zhou and Titus 1998).

$$\Delta P = \left(\alpha_0 (1 - n) \mu c \frac{Q^2}{A^2} t \right)^{\frac{1}{1-n}}, \text{ where} \quad \text{Equation B.8}$$

ΔP [kPa] is the pressure drop across the filter medium,

α_0 [mkg^{-1}] is the specific filtration resistance,

n [–] is the compressibility factor of the cake,

μ [Nsm^{-2}] is the dynamic viscosity of the filtrate,

c [kgm^{-3}] is the concentration of solids per volume of filtrate,

Q [$\text{m}^3 \text{s}^{-1}$] is the volumetric flowrate of the filtrate, and

t [s] is the time of filtration

$$P_{filter} = \frac{\Delta P_{filter} u}{\eta}, \text{ where}$$

Equation

B.9

P_{filter} [$kW m^{-2}$] is the power consumed per surface area of filter medium,

u [ms^{-1}] is the linear velocity through the filter medium and

η [–] is the efficiency of filtration which is the ratio of theoretical power needed to the actual power consumed.

The use of a filter aid depends on the porosity of the filter medium or material (Pickering 2005). However, according to Grima et al. (2003), the use of precoat filtration is not acceptable as contamination of the biomass with filter aid cannot be tolerated. Yeast cells have particle size distribution between 1.0 μm and 100 μm .

Energy consumption data for a vacuum filter per volume of slurry entering sourced from Grima et al. (2003) were used. Filtration of yeast for the purpose of dewatering falls under the category of microfiltration due to the particle size distribution of the yeast cells. The default energy per unit volume for microfiltration was reported by Harding (2008) as 7.3 MJ/m³. This agreed with the values quoted by Grima et al. (2003).

C. LCA input data, inventories and results (not presented in thesis)

C.1. Cohen (2006), approach, assumptions and data sets

Data pertaining to the life cycle inventory of Cohen (2006) were collected from a daily performance log of the SAB wastewater treatment plant dating .The data sets included volumetric flowrates, COD, VFA and SS concentrations, pH, volumetric flowrates of the biogas flared, caustic consumption, and power ratings of pumps, stirrers and fans.

The material balance was mainly based on the following assumptions and simplifications:

1. **Wastewater characteristics:** Likely compositions of species in the wastewater were assumed based on the characteristics of anaerobic digestion and brewery effluent as reported in the literature. Nitrogen and phosphorus were modelled as aqueous ammonia and phosphate respectively.

2. **Biogas characteristics:** Biogas compositions reported in literature were initially used, and later adjusted to agree with calculated and measured flowrates to flare and the overall carbon balance.

Cohen (2006) observed discrepancies between calculations of COD treated based on measurements and the AD reactor, and the data presented by SAB. This led to a conclusion that the COD concentration measurements were exclusive of the SS contribution. When using measured COD concentrations, the measured flare flowrates were not reconcilable without accounting for SS contribution. In addition, comparisons with methane yield values reported in literature were not achievable. However, the flare power correlated well with the mass balance, thus increasing confidence in the flare flowrate measurements. The resulting material balances based on initial assumptions were modified subject to the measured VFA, suspended solids concentrations in the SABWTP feed and effluent, in conjunction with the overall carbon balance. Table C.1 shows the resulting inventory. The phosphorus, nitrogen and sulphate concentrations for the aqueous streams were collected from SAB laboratory results log. They were compared and consolidated with data sets from literature sources.

Table C.1: Inventory results from Cohen (2006).

		SABWTP and AWTW		
	Inputs	SABWTP	AWTW	AWTW
COD	(kg/kg-COD)	1.0000	1.0000	1.0000
P	(kg/kg-COD)	0.0244	0.0244	0.0244
SO₄	(kg/kg-COD)	0.1904	0.1904	0.1904
N	(kg/kg-COD)	0.2149	0.2149	0.2149
SS	(kg/kg-COD)	0.1904	0.1904	0.1904
Power	(kWh/kg-COD)	0.1642	0.2464	0.1095
NaOH	(kg/kg-COD)	0.1008	0.1008	0.0000
Outputs				
COD	(kg/kg-COD)	0.0885	0.0009	0.1018
P	(kg/kg-COD)	0.0028	0.0004	0.0031
SO₄	(kg/kg-COD)	0.0656	0.0232	0.0674
N	(kg/kg-COD)	0.0094	0.0005	0.0124
SS	(kg/kg-COD)	0.0656	0.0018	0.0054
CH₄	(kg/kg-COD)	0.0079	0.0272	0.2181
CO₂	(kg/kg-COD)	1.0160	1.0603	0.5001
H₂	(kg/kg-COD)	0.0002	0.0002	0.0004
H₂S	(kg/kg-COD)	0.0069	0.0075	0.0072
water vapour	(kg/kg-COD)	0.5839	0.5944	0.1191
Power	(kWh/kg-COD)	-	-	-
NaOH	(kg/kg-COD)	0.1008	0.1008	0.0000

C.2. LCA Results not presented in Section 6.5.3 Chapter 6: Comparison of processes and substances contributing to environmental impact scores

Abiotic depletion

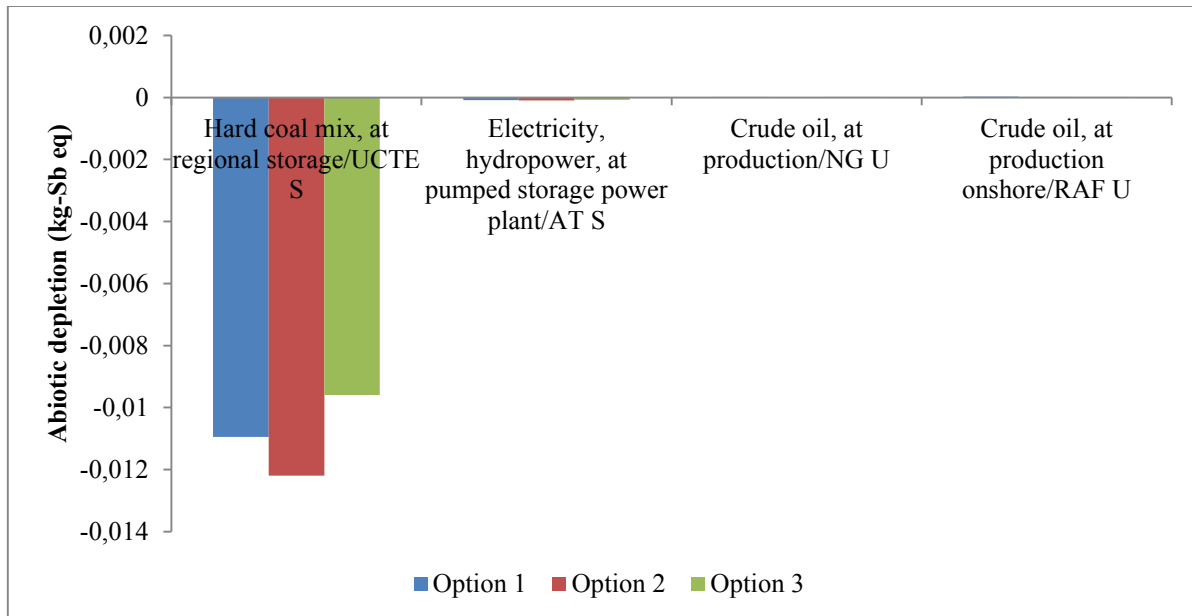


Figure C.1: Abiotic depletion: comparison of contributing processes.

Acidification

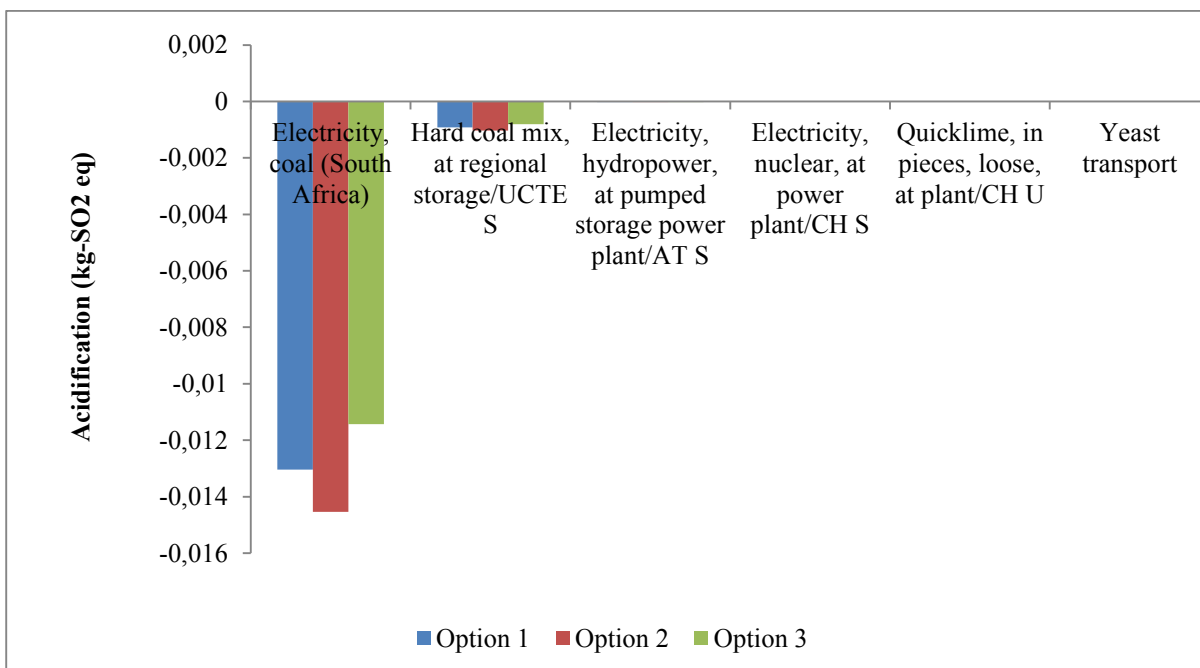


Figure C.2: Acidification: comparison of the contributing processes.

Human toxicity

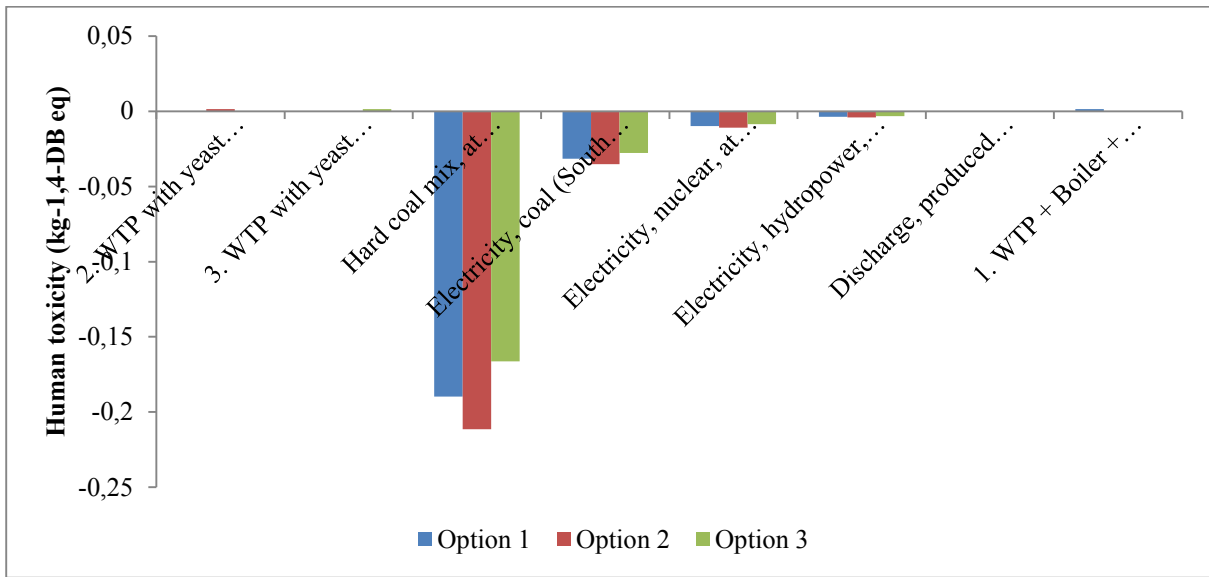


Figure C.3: Human toxicity: contributing processes

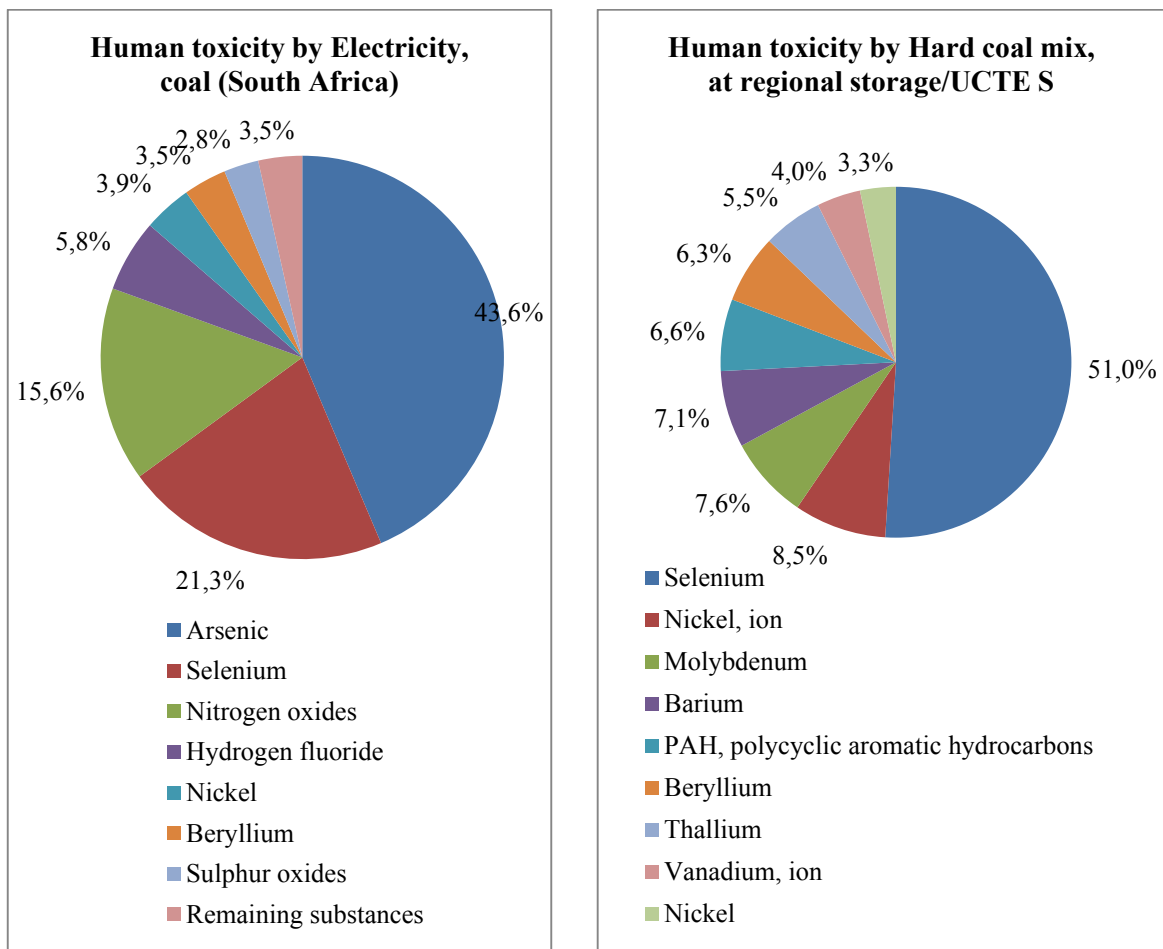


Figure C.4: Distribution of contributing substances to human toxicity at the coal power plant and coal acquisition processes.

Fresh water aquatic ecotoxicity

Contribution to *fresh water aquatic ecotoxicity* is almost entirely attributable to coal acquisition.

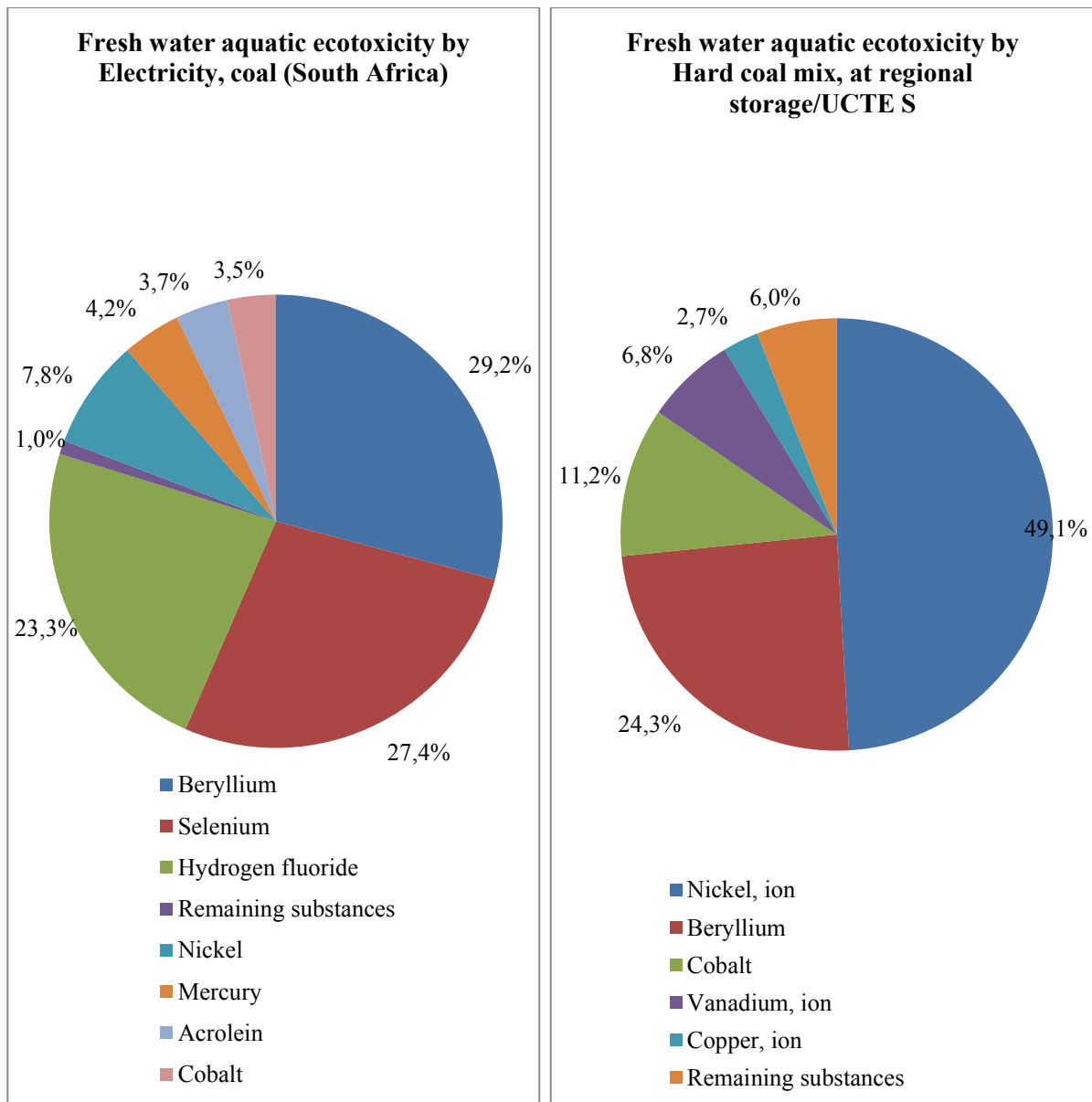


Figure C.5: Distribution of contributing substances to *fresh water aquatic ecotoxicity* at the coal power plant and coal acquisition processes.

Marine aquatic ecotoxicity

Marine aquatic *ecotoxicity* is contributed almost entirely by coal acquisition.

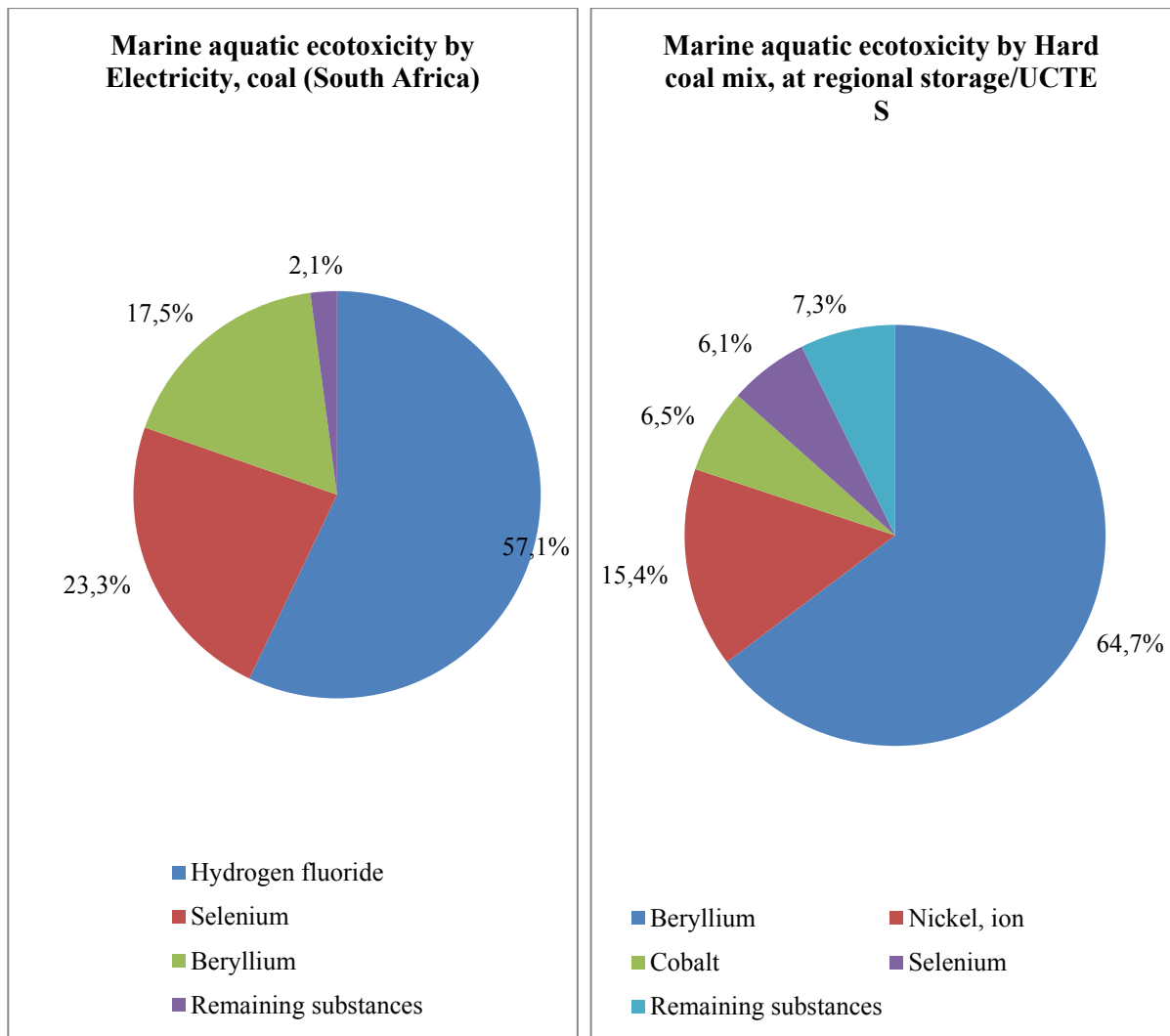


Figure C.6: Distribution of contributing substances to marine aquatic *ecotoxicity* at the coal power plant and coal acquisition processes.

Terrestrial ecotoxicity

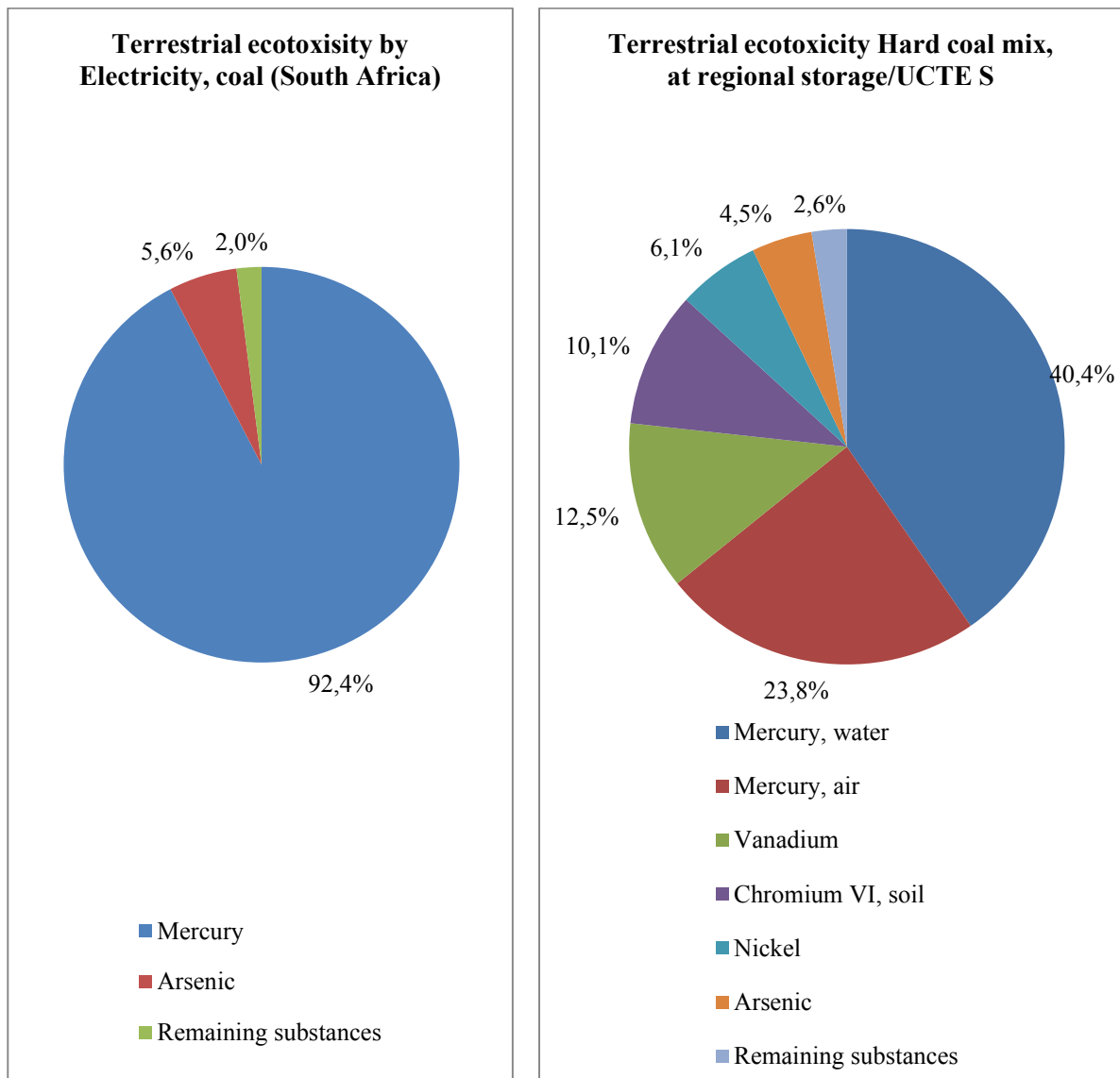


Figure C.7: Distribution of contributing substances to *terrestrial ecotoxicity* at the coal power plant and coal acquisition processes.

Eutrophication

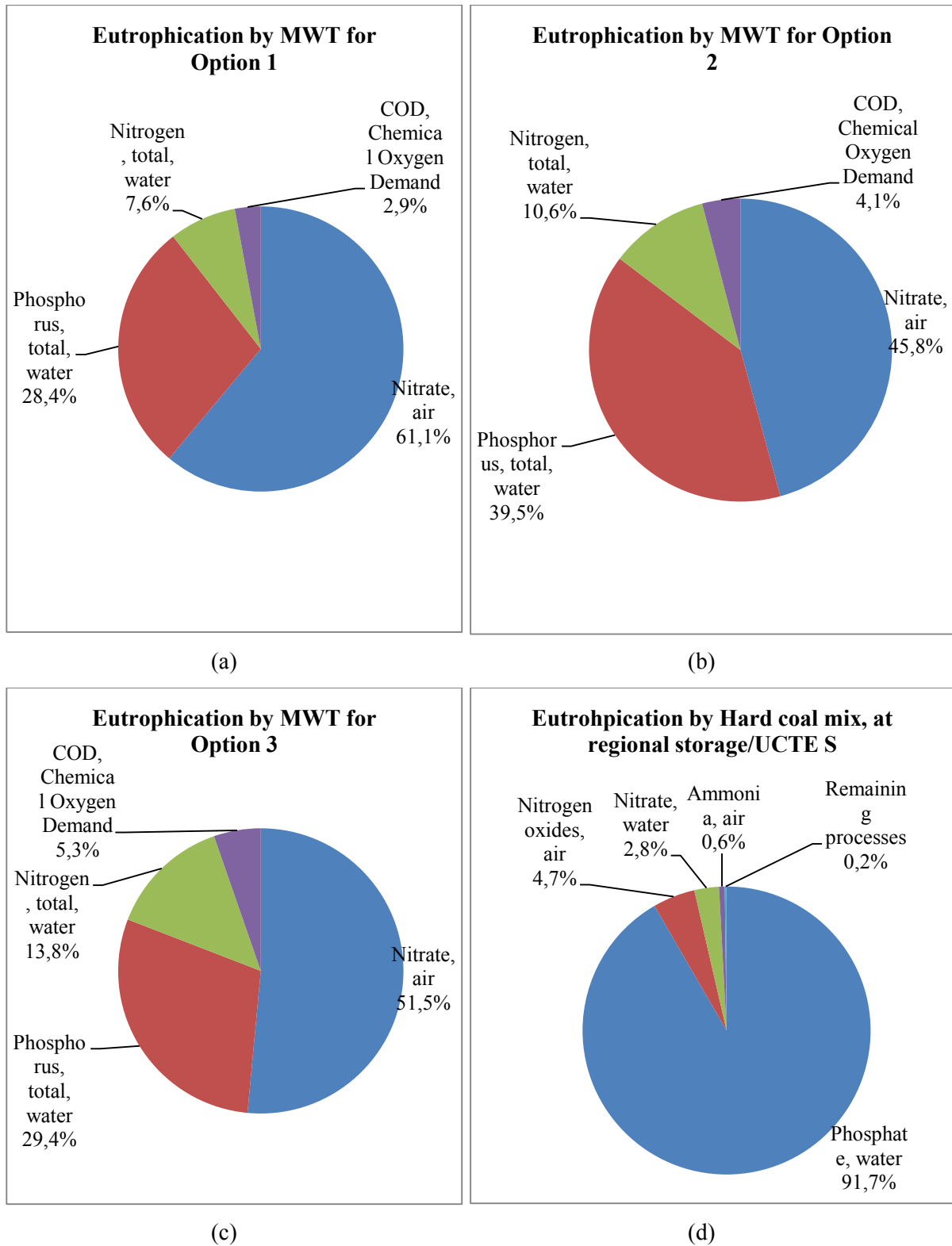


Figure C.8: Composition of the eutrophication scores for: (a) AWTW Option 1, (b) AWTW Option 2, (c) AWTW Option 3, and (d) coal acquisition.

Global warming potential

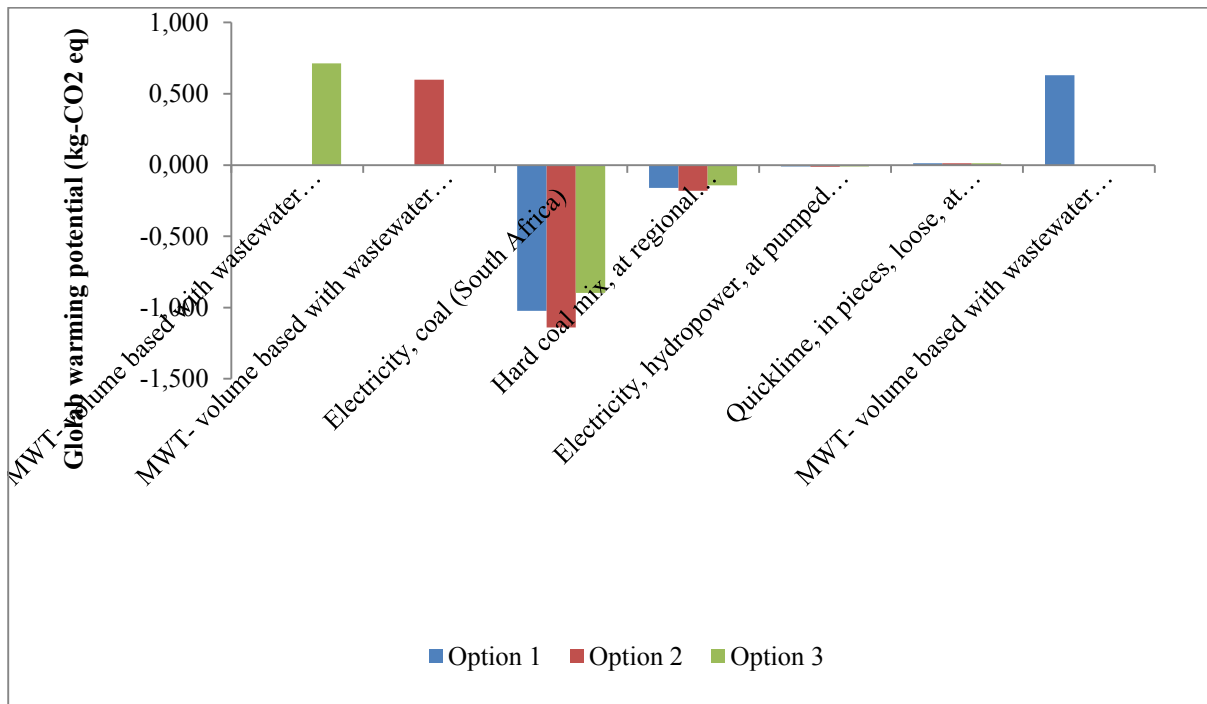


Figure C.9: Global warming potential: Comparison of contributing processes.

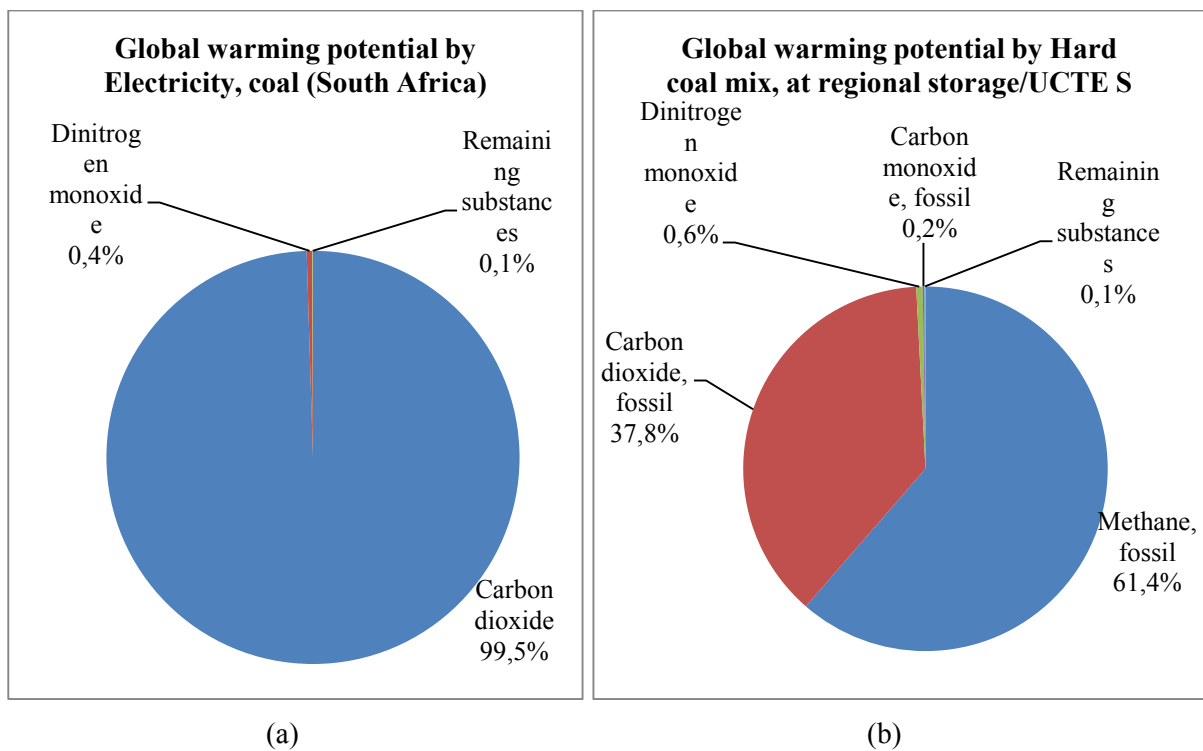


Figure C.10: (a) Composition of the global warming potential score for the coal acquisition process, and (b) Composition of the global warming potential score for coal-fired electricity production.

Ozone layer depletion

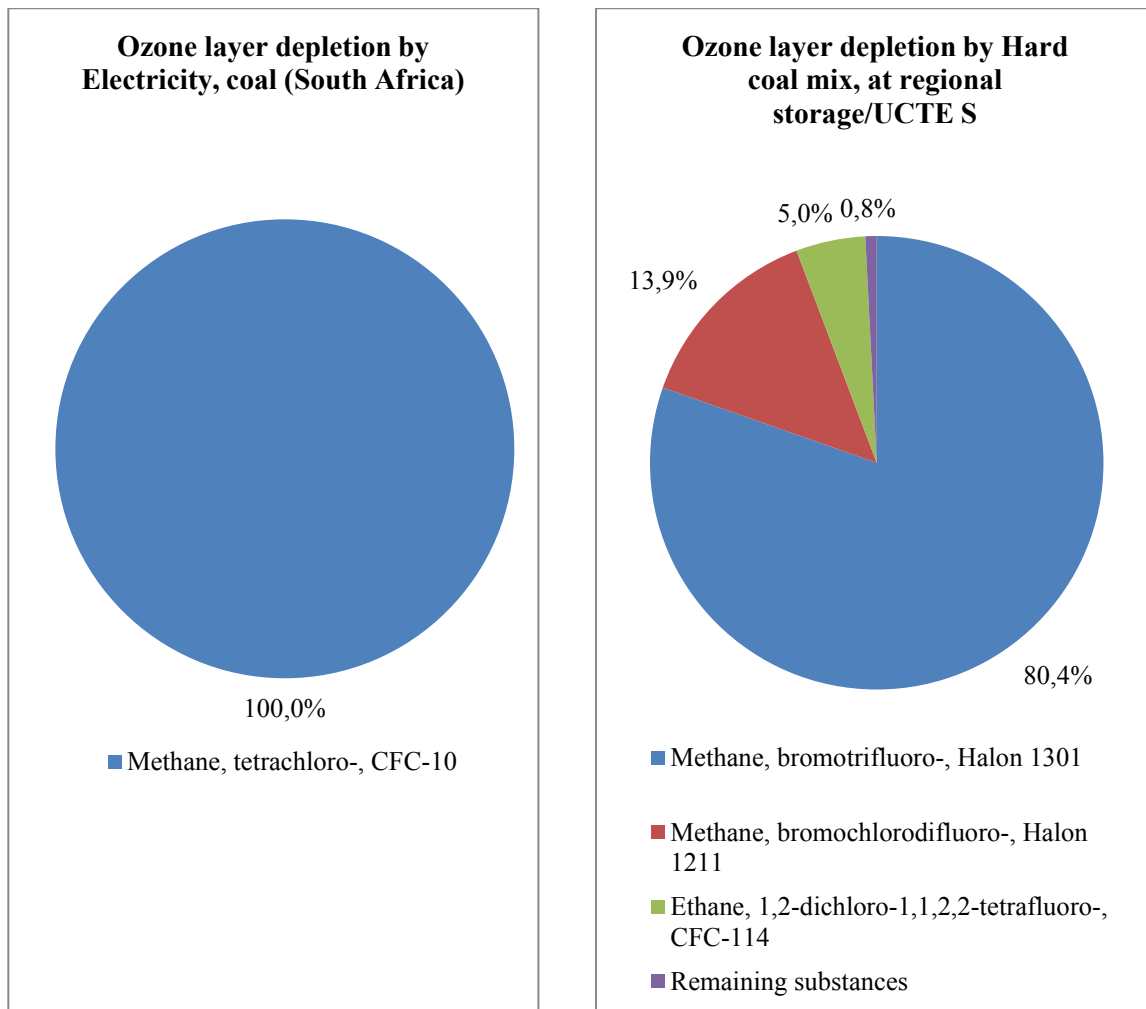


Figure C.11: Distribution of contributing substances to *ozone layer depletion* at the coal power plant and coal acquisition processes.

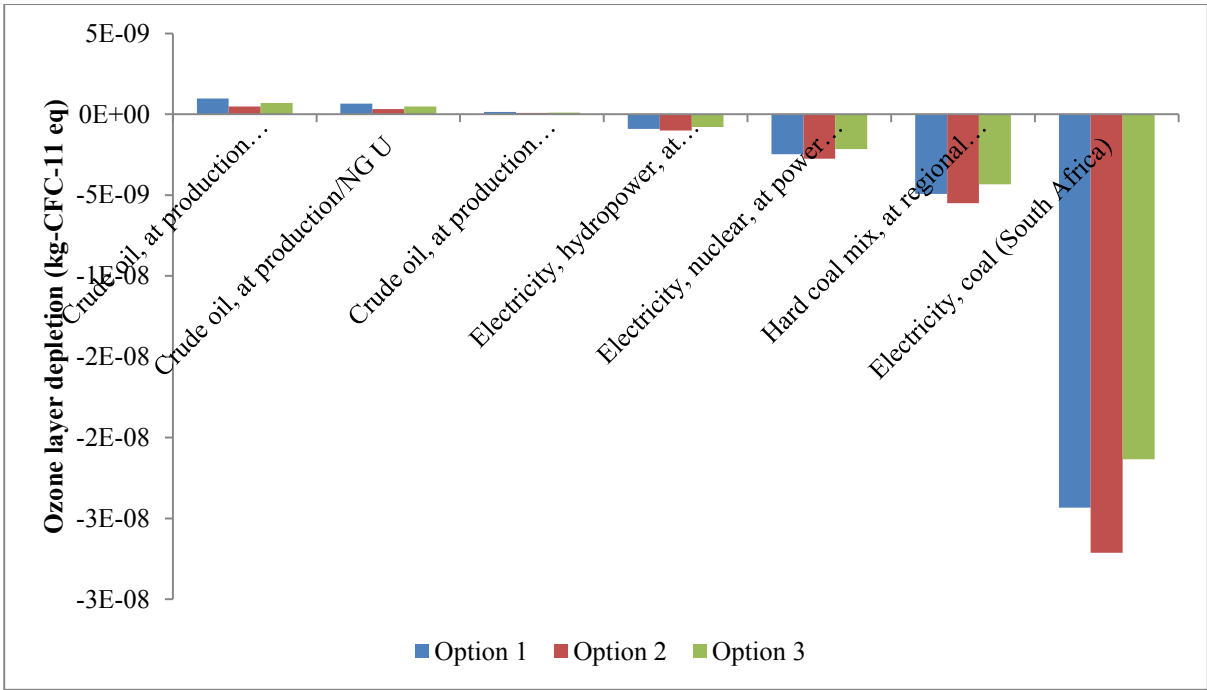


Figure C.12: Ozone layer depletion: Comparison of contributing processes.

Photochemical oxidation

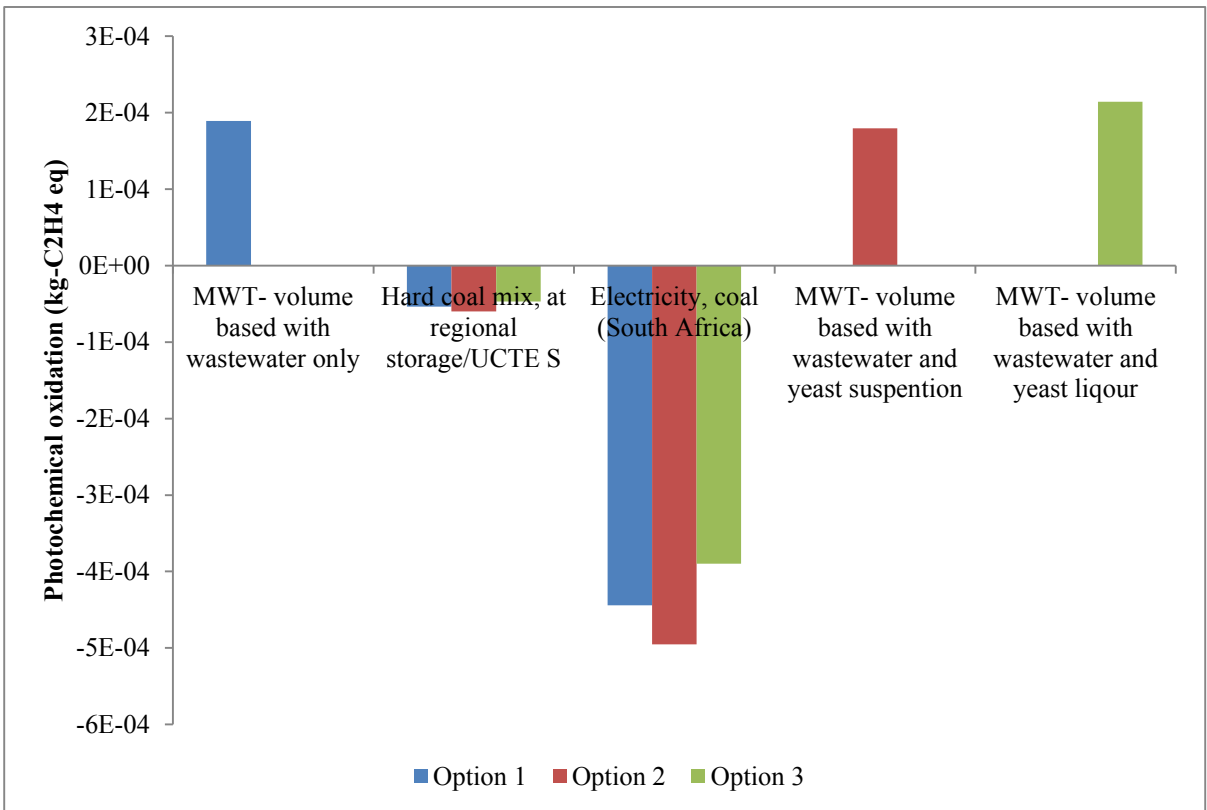


Figure C.13: Photochemical oxidation: Comparison of contributing processes.