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Jochen Sinn

Waste stabilization ponds for water reuse in water scarce regions

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Waste stabilization ponds for water reuse in water scarce regions

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»Du darfst am Guten in der Welt mitarbeiten – Das Wenige, das du tun kannst, ist viel.«

(Albert Schweitzer)

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Darmstadt, November 2022 Jochen Sinn

Abstract

Waste stabilization pond (WSP) systems are widely applied for communal wastewater treatment especially in countries of the global south with warm climates and sufficient available land area. At the same time many of these regions are affected by climate change and recently experience erratic rainfalls which distress the local subsistence agriculture. In these water scarce regions farmers often rely on surface or groundwater sources to irrigate staple foods and fodder or to water their animals. This in turn puts a burden on the water supply of the population. Therefore, reuse of treated WSP effluent, which is often only evaporated, presents a valuable source for irrigation water and at the same time of plant nutrients. But due to rapid population growth and lack of regular operation and maintenance many of these systems are overloaded and the effluents are overflowing into the surrounding environment causing environmental degradation and health risks to humans and animals.

Especially in sub-Saharan Africa there is little documented long-term experience with WSP operation and performance and their potential for water reuse. Therefore, this research presents an overview of the existing situation of nine WSP systems in north-central Namibia which in their current state do not fulfil the national Namibian and the newly published European standard for water reuse. As part of the research project EPoNa one WSP system was enhanced at full scale with different pre- and post-treatment technologies. These included a 250 μm micro sieve as mechanical pre-treatment for the removal of solids and organic carbon, an upstream anaerobic sludge blanket (UASB) as biological pre-treatment also for the removal of solids and organic carbon, sludge removal to restore the original volume of the ponds, floating baffles to improve flow conditions in the facultative pond and a rock filter as post-treatment in the final maturation pond for algae and pathogen reduction. The effects of these enhancements were compared with a second, parallel treatment train operated with its original setup. Compliance with the national and European reuse requirements was evaluated regarding the physical and biological wastewater parameters and further the microbial community was analysed.

The main results of this dissertation and relevant aspects for further applications are the following:

- In their current state, none of the researched WSP systems in north-central Namibia adhere with the Namibian and European reuse standards, which is mainly due to total organic carbon concentrations above 100 mg/L caused by high algal fractions in the particulate organic carbon.
- The algae related chlorophyll-*a* concentrations correlate linearly with the particulate organic carbon and this correlation can be used to fractionate the total organic carbon for further judgement.
- The microbial community is divers with different dominating genera in the influent than in the effluent.
- The mechanical pre-treatment with micro sieve (MS) (250 μm) and the anaerobic biological pre-treatment with an upstream anaerobic sludge blanket (UASB) reactor are both operational under the local conditions and can be implemented on large scale to reduce organic carbon, suspended solids and partially pathogens.

- The UASB achieves better average removal of chemical oxygen demand (50 %) and total suspended solids (57 %) but the MS is more flexible in handling changing inflow patterns and has a much smaller footprint. A maximum particulate chemical oxygen demand reduction of 89 % is reached with the UASB and 72 % with the MS.
- With the pre-treatment there is only limited nitrogen and phosphorus reduction which therefore remain as nutrients in the water and are valuable for further irrigation purposes.
- After one year of operation the rock filter as post-treatment reduces only 5 % of chlorophyll-*a* and shows no additional removal of algae compared to the original treatment train.
- Algae concentrations are best reduced with pre-treatment, sludge removal and baffles in the facultative pond.
- With enhancements *E. coli* concentrations are reduced down to the new EU water reuse standard of 1,000 MPN/100 mL for fodder irrigation whilst concentrations of *P. aeruginosa* stagnate and *Enterococci* levels increase. With this divergence the function of *E. coli* as indicator for broader pathogen reduction is questioned. Main pathogen reduction happens during pre-treatment and in the facultative pond with baffles and not as expected in the maturation ponds.
- Due to high carbon and nitrogen concentration the effluent does not meet the Namibian and European reuse standards but the high algal content would add valuable biomass and fertilizer to the barren soil. Therefore, a review of the standards considering particularly WSP effluents is suggested.

For the first time WSP in north-central Namibia are comprehensively analysed and the reuse potential of their effluents compared. At one pilot plant different enhancement technologies are tested at large scale and the results evaluated with regards to their applicability under similar conditions. Therefore, this dissertation contributes valuable information for the upgrade of existing WSP to improve the environmental situation and produce irrigation water in water scarce regions.

Kurzfassung

In großem Umfang werden Abwasserteichanlagen besonders in Ländern des globalen Südens mit warmen Klimazonen und ausreichend verfügbarer Landfläche für die kommunale Abwasserbehandlung eingesetzt. Gleichzeitig sind viele Menschen in diesen Regionen vom Klimawandel betroffen. Sie erleben unregelmäßige Regenfälle, die insbesondere die lokale Subsistenzlandwirtschaft belasten. In diesen durch Wasserknappheit betroffenen Gebieten sind die Landwirte besonders auf Oberflächengewässer oder Grundwasser angewiesen, um Grundnahrungsmitteln und Futtermitteln zu bewässern oder ihre Tiere zu tränken. Im Gegenzug belastet dies die Trinkwasserversorgung der Bevölkerung. Aus diesem Grund bietet die Wiederverwendung von gereinigtem Wasser aus Abwasserteichanlagen, das oft nur verdunstet, eine wertvolle Ressource für Bewässerungswasser sowie für Pflanzennährstoffe. Viele dieser Teichsysteme sind jedoch durch schnelles Bevölkerungswachstum als auch durch mangelnden Betrieb und unregelmäßiger Wartung überlastet. Somit führt das ungenügend gereinigte Ablaufwasser zu Umweltverschmutzungen und zu erhöhten Gesundheitsrisiken für Menschen und Tiere.

Insbesondere in afrikanischen Ländern südlich der Sahara gibt es nur wenige veröffentlichte Langzeitstudien zur Betriebserfahrung und Leistung von Abwasserteichanlagen sowie zu deren Potential für die Wasserwiederverwendung. Somit bietet diese Forschungsarbeit einen Überblick über neun existierende Teichkläranlagen im zentralen Norden Namibias. In ihrem aktuellen Ausbauzustand erfüllt keine dieser Anlagen die nationalen namibischen Anforderungen noch die neuen Standards der Europäischen Union zur Wasserwiederverwendung. Im Rahmen des Forschungsprojektes EPoNa wurde eine Abwasserteichanlage durch verschiedene Vor- und Nachbehandlungstechnologien in großem Maßstab ertüchtigt. Dazu gehörte ein Mikrosieb mit einer Maschenweite von 250 μm als mechanische Vorbehandlung und ein UASB-Anaerobreaktor als biologische Vorbehandlung. Beide dienten zur Reduzierung abfiltrierbarer Stoffe und von organischem Kohlenstoff. Die Entschlammung der Teiche ermöglichte die Wiederherstellung des ursprünglichen Volumens und schwimmende Leitwände im Fakultativteich verbesserten die Strömungsbedingungen. Der im letzten Schönungsteich als Nachbehandlung eingebaute Steinfilter sollte Algen und pathogener Keime weiter reduzieren. Die Auswirkungen der Ertüchtigungsmaßnahmen auf der Pilotanlage wurden einer zweiten, im Originalzustand parallel betriebenen Behandlungsstraße gegenübergestellt. Die Auswertung biologischer und physikalischer Abwasserparameter diente zum Vergleich mit den nationalen namibischen sowie den europäischen Anforderungen an die Wasserwiederverwendung. Darüber hinaus wurde auch die DNA der vorhandenen mikrobiellen Gemeinschaften analysiert.

Die Hauptergebnisse dieser Forschungsarbeit und relevante Aspekte für weitere Anwendungen sind die folgenden:

- In ihrem aktuellen Zustand erfüllt keine der im zentralen Norden Namibias untersuchten Abwasserteichanlagen die namibischen und die europäischen Anforderungen zur Wasserwiederverwendung. Dies ist auf den chemischen Sauerstoffbedarf (CSB) von über 100 mg/L zurückzuführen, der durch hohe partikuläre Algenanteile verursacht wird.

- Die durch Algen hervorgerufenen Chlorophyll-*a* Konzentrationen korrelieren linear mit dem partikulären CSB. Diese Korrelation kann für die Fraktionierung und Beurteilung des CSB genutzt werden.
- Es liegt eine sehr vielfältige mikrobielle Gemeinschaft vor, die im Zulauf der Anlagen andere dominierende Gattungen als im Ablauf aufweisen.
- Die mechanische Vorbehandlung mit einem Mikrosieb (250 μm) und die anaerobe biologische Vorbehandlung mit einem UASB-Reaktor sind beide unter den gegebenen lokalen Voraussetzungen und in großem Maßstab einsetzbar. Sie reduzieren zuverlässig organischen Kohlenstoff, abfiltrierbare Stoffe und teilweise pathogene Keime.
- Der UASB-Reaktor erreicht eine bessere durchschnittliche Reduktion des chemischen Sauerstoffbedarfs (50 %) und der abfiltrierbaren Stoffe (57 %) während das Mikrosieb flexibler auf schwankende Zuflüsse reagiert und einen viel kleineren Platzbedarf aufweist. Die maximale Reduktion des chemischen Sauerstoffbedarfs wird mit 89 % durch den UASB-Reaktor und mit 72 % durch das Mikrosieb erreicht.
- In der Vorbehandlung werden Stickstoff und Phosphor nur in geringem Maße reduziert. Somit verbleiben sie als wertvolle Pflanzennährstoffe im Bewässerungswasser.
- Nach einem Jahr Betrieb entfernt der Steinfilter in der Nachbehandlung nur 5 % des Chlorophyll-*a* und zeigt keine zusätzliche Entfernung im Vergleich zum ursprünglichen Zustand.
- Algenkonzentrationen werden am wirkungsvollsten durch Vorbehandlung, Schlammentfernung und Leitwände im Fakultativteich minimiert.
- Durch die Ertüchtigungsmaßnahmen ist es möglich, die *E. coli* Konzentrationen unter den von der EU geforderten Standard für die Bewässerung von Futtermitteln in Höhe von 1.000 MPN/100 mL zu reduzieren. Gleichzeitig stagnieren die Werte für *P. aeruginosa.* Für *Enterococci* steigen sie sogar an. Durch dieses Auseinanderdriften der Werte stellt sich die Frage, ob *E. coli* als alleiniger Indikator zur generellen Reduktion pathogener Keime geeignet ist. Die größte Keimreduktion findet, wider Erwarten nicht in den Schönungsteichen, sondern in der Vorbehandlung und durch die Leitwände im Fakultativteich statt.
- Aufgrund der hohen Kohlenstoff- und Stickstoffkonzentrationen entsprechen die Ablaufwerte weder den namibischen noch den europäischen Normen zur Wasserwiederverwendung. Gleichzeitig würde der hohe Algenanteil wertvolle Biomasse und Nährstoffe in die kargen Böden einbringen. Deshalb wird eine Überprüfung der Wasserwiederverwendungsstandards insbesondere in Bezug auf Abwasserteichanlagen vorgeschlagen.

Diese Arbeit untersucht zum ersten Mal umfassend Abwasserteichanlagen im zentralen Norden Namibias und ermittelt das Wiederverwendungspotential der Abläufe. Auf einer Pilotanlage wurden verschiedene Ertüchtigungsmaßnahmen in großem Maßstab erprobt und die Ergebnisse hinsichtlich ihrer Übertragbarkeit auf andere Standorte unter ähnlichen Bedingungen ausgewertet. Somit trägt diese Dissertation wertvolle Informationen zur Ertüchtigung existierender Teichkläranlagen, zum Schutz der Umwelt und zur Produktion von Bewässerungswasser in Regionen mit knappen Wasserressourcen bei.

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Abbreviations

AP	Anaerobic Pond
BMBF	Bundesministerium für Bildung und Forschung – German Federal Ministry of Education and Research
BOD	Biological Oxygen Demand
COD	Chemical Oxygen Demand
DIN	Deutsche Institut für Normung – German Institute for Standardization
DNA	Deoxyribonucleic Acid
DO	Dissolved Oxygen
DWAF	Department of Water Affaires and Forestry
EC	Electrical Conductivity
E. coli	*Escherichia coli*
EP	Evaporation Pond
EPoNa	Enhancement of Ponds in Namibia
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FP	Facultative Pond
GRN	Government of the Republic of Namibia
GS	General Standard
HDPE	High Density Polyethylene
HRT	Hydraulic Retention Time
IDM	Electromagnetic Flowmeter
ISO	International Organization for Standardization
IWRM	Integrated Water Resource Management
LOQ	Limit of Quantification
MP	Maturation Pond
MPN	Most Probable Number
MS	Micro Sieve
n	Number of measurements
OTC	Outapi Town Council
pCOD	Particulate Chemical Oxygen Demand
PE	Population Equivalent
PreT	Pre-treatment
PostT	Post-treatment
P. aeruginosa	*Pseudomonas aeruginosa*
RBC	Rotating Biological Contactor
rRNA	Ribosomal Ribonucleic Acid
sCOD	Soluble Chemical Oxygen Demand
Sd	Standard deviation
SDG	Sustainable Development Goals
SS	Special Standard
tCOD	Total Chemical Oxygen Demand
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TP	Total Phosphorous

TS	Total Solids
TSS	Total Suspended Solids
TVS	Total Volatile Solids
UASB	Upstream Anaerobic Sludge Blanket
UN	United Nations
VSS	Volatile Suspended Solids
WHO	World Health Organization
WSP	Waste Stabilization Pond
WWTP	Wastewater Treatment Plant

1 Introduction

1.1 Challenges of wastewater treatment in sub-Saharan Africa

Access to sanitation, sewer connections and wastewater treatment is still very limited in sub-Saharan Africa (WWAP, 2019). It is estimated that in these countries overall only 1 % of all wastewater is treated (Bahri et al., 2008). In recent years water utilities are mainly focusing on water supply development and are struggling with the development and operation of piped sewer systems as well as wastewater treatment plants (WWTP) causing environmental pollution and public health risks (Werchota, 2020).

Most common wastewater treatment technologies in the region are waste stabilization ponds (WSP) followed by activated sludge systems and some trickling filters (Nikiema et al., 2013). Many of these WWTPs are worn out (Nikiema et al., 2013), dysfunctional, overloaded, poorly operated and maintained and their discharge is contaminating surface waters (Swana et al., 2020). Common challenges are high organic loads due to low water consumption combined with increasing flow rates due to rapid population growth, uncontrolled solid waste disposal, high energy costs, unreliable power supply as well as lack of re-investments (Nikiema et al., 2013).

At the same time sustainable maintenance mechanisms such as sludge removal from WSP are not established with the design of the WWTPs and qualified managers and operators are missing to ensure proper operation and maintenance (Wang et al., 2014). Often it is difficult to motivate and encourage employees because of administrative procedures, lack of funds for spare parts, lack of maintenance planning and missing recognition of personal performance (Nikiema et al., 2013). Additionally, local laboratories for water analyses, if they exist at all, are poorly equipped and miss consumable supplies so that water quality monitoring is often neglected (Wang et al., 2014).

1.2 Research questions and objectives

The main objective of this dissertation is to provide additional information on existing WSP systems in Africa and particularly in Namibia. This is supplemented by practical research on rehabilitation, extension and upgrade of one specific wastewater pond system to improve its effluent quality and protect the surrounding environment. Additionally, the potentials of WSP to generate irrigation water for agriculture is evaluated. Therefore, this dissertation contributes directly to the United Nations Sustainable Development Goals (SDG) (UN, 2015) and specifically to goal 6.3 on sustainable reuse and goal 6.a on international cooperation.

The research questions and the corresponding objectives are:

1. What is the current state and performance of WSP in north-central Namibia and what is their potential for water reuse?
    - Characterise and evaluate for the first time existing WSP systems in north-central Namibia
    - Analyse and compare microbial communities including the composition of pathogens and cyanobacteria in the influent and effluent of WSP
    - Establish seasonal effects on effluent water quality of WSP

2. With which biological and/or physical enhancement measures can WSP be upgraded to produce irrigation water?
    - Compare the impact of anaerobic biological and mechanical pre-treatment technologies in the local context
    - Analyse the performance of WSP with baffles and rock filter as additional treatment technologies
    - Examine the change in composition and concentrations of pathogens as well as the microbial community by different operation scenarios and enhancements

3. How can traditional or upgraded WSP comply with the Namibian and European water reuse standards for irrigation?
    - Elaborate the impact of algae on the particulate COD
    - Review Namibian and European water quality standards for water reuse in agricultural irrigation and their applicability on WSP
    - Identify obstacles for agricultural water reuse as well as operation and maintenance of existing WSP

1.3 Outline of this dissertation

This dissertation was conducted at the chair of Water and Environmental Biotechnology, Institute IWAR, Technische Universität Darmstadt, under the joint research project Enhancement of Ponds in Namibia (EPoNa). It was financed by the German Federal Ministry of Education and Research (BMBF) and its funding measure Future-oriented Technologies and Concepts to Increase Water Availability by Water Reuse and Desalination (WavE) from September 2016 till August 2020.

The main chapters of this dissertation are based on three peer-reviewed articles in scientific journals:

- Sinn, J., Agrawal, S., Orschler, L., and Lackner, S. (2022). “Characterization and evaluation of waste stabilization pond systems in Namibia” has previously been published in the H2Open Journal (2022), volume 5 (2): 365-378. It characterizes and evaluates for the first time nine operational WSP systems in north-central Namibia. Their specificities and especially influent and effluent characteristics are discussed together with the analysis of their algal biomass as well as their microbial ecology.

- Sinn, J., and Lackner, S. (2020): “Enhancement of overloaded waste stabilization ponds using different pre-treatment technologies: a comparative study from Namibia” was published in the Journal Water Reuse and Desalination (2020), volume 10 (4): 500-512. It presents the results of the implemented pre-treatment enhancements at the full-scale pilot plant, with the comparison of an anaerobic biological and mechanical technology.

- Sinn, J., Agrawal, S., Orschler, L., Schubert, S. and Lackner, S. (2022) “Upgrade of waste stabilization ponds to improve effluents for reuse purposes” has been submitted for publication in the H2Open Journal (2022). It presents the long-term results of all enhancement, pre-treatment as well as post-treatment with a focus on the seasonal effects and the influence of the hydraulic retention time as well as on the natural disinfection and the bacterial community. The impact of the additional treatment technologies as well as the effects on the microbial community and the algae development are discussed.

2 Background

Worldwide, the importance of water shortage is growing every year. Two-thirds of the world population (4.0 billion people) experience severe water scarcity at least during one month per year and half a billion during the whole year (Mekonnen and Hoekstra, 2016). This situation is exacerbated by climate change, population growth, economic development and urbanization, and affects more and more regions (WHO, 2006). At the same time the water demand for drinking will increase and even double for food production by 2050 (Assouline et al., 2015). An alternative resource, especially for agricultural irrigation, is the reuse of treated wastewater (WWAP, 2017). Particularly in arid areas with available land, WSP systems are an important treatment technology (Fuhrmann, 2014; Mara, 2009) and are also widely implemented in Namibia.

2.1 Waste stabilization ponds (WSP)

The technology of WSP dates back to the year 1901 (Ho and Goethals, 2020). They represent a simple form of wastewater treatment and offer a cost effective alternative to activated sludge systems or trickling filters, especially in countries with sufficient land availability and favourable climatic conditions (Mara, 2004; von Sperling, 2007a). Due to the simplicity of construction, operation and low costs, WSP are also desirable in developing countries and provide a sustainable way of wastewater treatment in rural areas (Ho et al., 2017). Furthermore, well planned WSP require less maintenance than conventional WWTP as no control systems are required (DWA, 2005). Nowadays, WSP are increasingly used in the field of water reuse for agricultural purposes. This not only lowers the operating costs for agricultural cultivation but also reduces the exploitation of water resources (Pivelli et al., 2008; U.S. EPA, 2011).

WSP are operated worldwide. At the beginning of the 21st century about 2,000 plants were operational in rural communities in Germany (Fuhrmann, 2014) and about 2,500 in France (Mara, 2009). In Greece 8 % of all urban wastewaters are treated with WSP (Chalatsi and Gratziou, 2014) and in the USA more than 8,000 WSP represent 50 % of all WWTP (Ho et al., 2017). Also in Mexico 16 % of WWTP are WSP (Hernandez-Paniagua et al., 2014). Especially in South Asia and Africa WSP not only exist in small municipal areas but also in metropolitan agglomerations (Ho and Goethals, 2020) such as Nairobi with 80 % of wastewater treated in the city (Wang et al., 2014).

Being a nature based solution WSP in their basic form do not rely on electromechanical treatments (Mara, 2004; Mara and Horan, 2003). The main principles are sedimentation and biological treatment without energy or chemical input (Crites et al., 2014). They are most effective in regions with high temperature all year-round and high solar radiation (von Sperling, 2007a). These are relevant conditions for algae growth and their products, such as oxygen, needed for other microorganisms (Wallace et al., 2015). However, depending on the size of the population large land areas are required (Mara, 2004).

The typical setup of a WSP consists of anaerobic ponds (AP), facultative ponds (FP) and a series of maturation ponds (MP). If there is no AP the first pond is called primary FP (Shilton, 2005; Verbyla et al., 2017). Finally, evaporation ponds (EP) are used as storage or buffer for treated wastewater if there is no alternative of discharge into surrounding water bodies (Asano et al., 2007).

The first biological treatment stage in WSP systems are AP to treat high organic loadings (> 100 g BOD/m^3/d) (Mara, 2004). Their primary function is the removal of BOD, COD and TSS from the raw wastewater under anaerobic conditions (Alexiou and Mara, 2003). Compared to the other pond types AP have a smaller surface area and are typically 3 – 5 m deep (von Sperling, 2007a). To allow anaerobic digestion and sedimentation hydraulic retention times (HRT) can be as short as one day and still > 60 % BOD removal at 20 °C (Mara, 2004) is achieved. At lower temperatures, removal efficiency is limited to sedimentation of particulate organic matter (DWA, 2016).

Whilst AP are only recommended, FP are inevitable. They are the most common type of pond in WSP systems (Varón and Mara, 2004). Figure 2.1 shows their upper layer as aerobic and bottom layer as anaerobic with their position varying during the day depending on the sun radiation and related algal activity (Gloyna, 1971). Shilton (2005) identified an anoxic zone above the anaerobic sludge. With 1 – 2 m depth and 17 – 33 days HRT in tropical areas up to 95 % of the original wastewater BOD can be removed depending on the temperature (Gloyna, 1971). However, BOD concentrations in the effluent of FP remain high with 60 – 90 % consisting of particulate matter from algae and cyanobacteria (von Sperling, 2007a).

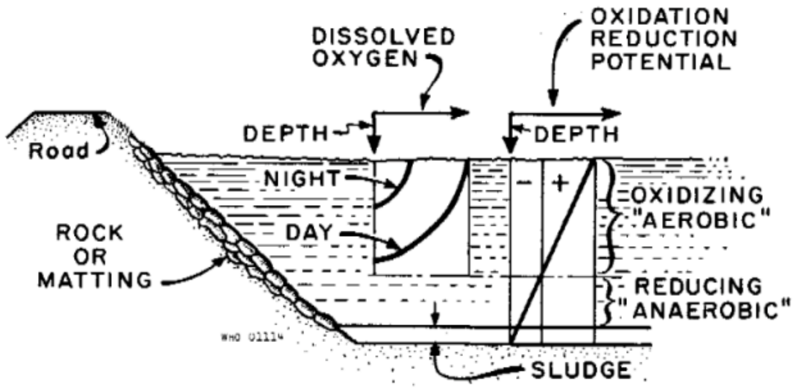

Figure 2.1: Dissolved oxygen and oxidation-reduction potential of typical facultative ponds (Gloyna, 1971)

The final stage of WSP systems consists of several MP in series with the main task of removing pathogens, nitrogen and phosphorus (Mara et al., 1992; Varón and Mara, 2004). Pathogens are eliminated by direct UV radiation in combination with high oxygen concentrations and high pH values around 9 or higher (Curtis et al., 1992; Mara and Pearson, 1998) as well as higher water temperatures (above 26 °C) (Ouali et al., 2013). For best disinfection effects MP are typically shallow (< 1 m) and HRT are 7 – 10 days (Gloyna, 1971). Due to lower BOD loads there is a higher algae diversity and 70 – 90 % of BOD result from algae (Mara, 1997).

EP perform best in regions with favourable climatic conditions especially with high solar radiation (Asano et al., 2007) and in Namibia they are widely applied because of missing perennial receiving water bodies. The aim is to reduce the risk of surface or groundwater contamination and therefore the national code of practice for pond systems (DWAF, 2008) requires all water from WSP to be evaporated. The design of EP is based on sewage volume, annual precipitation and evaporation rates (DWAF, 2008).

2.2 Reuse of treated wastewater in agriculture

Population growth and economic development have and will further increase the global water and food demand with agriculture accounting for 70 % of global water withdrawals (WWAP, 2018). For this reason, reuse of treated wastewater is becoming more and more recognised worldwide (WWAP, 2017). It reduces the demand on freshwater (WHO, 2006) and can contribute to global food security (WHO, 2006; Zimmermann and Neu, 2022). According to Maeseele and Roux (2021) 80 % of wastewater reuse is implemented in semiarid and arid regions with growing water scarcity. Especially in areas with erratic rainfalls it enables continuous agricultural production for the local communities (Jiménez et al., 2010) and contributes to their sustainable circular economies (EU, 2020; Lahlou et al., 2021). In these regions existing WSP systems provide an important water source (Pivelli et al., 2008). Properly managed and enhanced WSP have the potential to comply with national and international reuse standards for the irrigation of fodder crops (Ayers and Westcot, 1985; Bansah and Suglo, 2016; Chalatsi and Gratziou, 2014; Mara, 2009; Verbyla et al., 2016). Compliance with standards also increases acceptance amongst the population. In Tanzania, Msaki et al. (2022) found that a majority does not accept water reuse for domestic applications but are willing to use treated wastewater for irrigation of forests and farming of fodder crops.

Not only the water but also nutrients such as nitrogen, phosphorous and potassium are important resources to be recovered and reduce costs for chemical fertilizers (Jiménez et al., 2010; Lahlou et al., 2021). Doorenbos et al. (1979) indicate that fertilizer requirements vary with the production level and depend on each crop. Typical fodder crops are alfalfa, maize and sorghum. On average alfalfa needs 12,000 m^3/ha of water per growing period (about 90 days) and 3 mg/L total nitrogen (TN), 5 mg/L total phosphorous (TP) and 7 mg/L potassium (K) (Doorenbos et al., 1979). Maize needs about 6,500 m^3/ha of water per growing period (about 125 days) and 31 mg/L TN, 10 mg/L TP and 12 mg/L K (Doorenbos et al., 1979). Sorghum requires on average 5,500 m^3/ha of water per growing period (about 120 days) and 33 mg/L TN, 6 mg/L TP and 11 mg/L K (Doorenbos et al., 1979). Untreated domestic wastewater consists of 23 – 69 mg/L TN, 4 – 11 mg/L TP and 11 – 32 mg/L K (Tchobanoglous et al., 2014). In a previous study the urban centre of this research had average wastewater concentrations of 58 mg/L TN and 10 mg/L TP with an average water consumption of 61 L/person/day (Müller, 2017). According to von Sperling (2007a) a typical WSP system of AP, FP and MP can remove at least 50 % of TN and TP. Therefore, additional costs can be saved through the adaptation of WWTP if for example nutrient levels for irrigating fodder crops are easier to reach than for discharge into the environment (Lahlou et al., 2021). Nevertheless, protection of the receiving plants and water body prevails and with high nutrient levels a mixture with surface water can be applied (Leonel and Tonetti, 2021).

Besides the treatment of the wastewater the irrigation technology has to be adapted. Sprinkler and furrow irrigation should be replaced by drip irrigation to avoid aerosols and direct contact with fruits and crops (Fonseca et al., 2011; Lahlou et al., 2021; Ofori et al., 2021). To assure the highest health protection Carr et al. (2004) as well as Mohr et al. (2020) propose a "multi barrier" approach including wastewater treatment, crop restrictions, irrigation technique, human exposure control as well as vaccination of field workers.

For the reuse of treated wastewater different national and international regulations, frameworks, standards or guidelines exist to protect public health (Deviller et al., 2020; Helmecke et al., 2020; Janeiro et al., 2020; Müller and Cornel, 2017; Ofori et al., 2021; Reynaert et al., 2021). Depending on the guidelines' origin and the existing background conditions their requirements on monitoring and measurements vary considerably. The guidelines for water reuse in the US suggest continuous monitoring of turbidity and chlorine, daily analysis of faecal coliforms as well as pH and BOD on a weekly basis (U.S. EPA, 2012).

In order to reduce public health risks the EU regulation on water reuse requires effective risk barrier systems and disinfection prior to crop application (EU, 2020). Therefore, Ofori et al. (2021) suggest the use of UV disinfection to avoid by-products or residues caused by other technologies. Within the European context it is even further discussed to include a systematic science-based monitoring of suitable indicator substances for micro-contaminants (Helmecke et al., 2020). However, Reynaert et al. (2021) state clearly that such extensive monitoring will not everywhere be possible for economic reasons especially for small-scale water reuse schemes and also for areas with limited laboratory equipment. Therefore, it is suggested to develop localized or country-specific reuse guidelines and regulations (Ofori et al., 2021) as well as a multiple barrier approach throughout the total supply chain of the crops produced and not only on the wastewater treatment (Janeiro et al., 2020).

The first WHO water reuse guidelines were published in 1973 and revised in 1989 based on restrictive effluent criteria (Lazarova and Bahri, 2005). The next revision in 2006 (WHO, 2006) no longer requires specific treatment technologies such as disinfection but considers health based targets for the whole supply chain of the products irrigated with treated wastewater (Drechsel et al., 2008; Janeiro et al., 2020). To achieve the health based target for water-related illnesses an overall *E.coli* reduction between 2 and 7 $\log_{10}$ units is recommended depending on the irrigation method and the planted crops (WHO, 2006).

In order to reflect the local appropriateness and the latest international developments this research focuses on the Namibian "Code of practice: Volume 6 Wastewater Reuse" published in 2012 (DWAF, 2012) and the most recent EU Regulation No. 2020/741 on the minimum requirements for water reuse from 2020 (EU, 2020).

2.2.1 Namibian national code of practice

Sustainable water resource management is the main objective of the Namibian Water Resource Management Act, 2013 (GRN, 2013). Its fundamental principles are based on protection, sustainable development and management of all water resources. This specifically includes access to safe drinking water and wastewater treatment as well as the reuse of treated effluents for domestic, commercial, industrial or agricultural purposes (GRN, 2013). All these aspects were also addressed within the previous act from 2004. Based on this the Code of practice: Volume 6 – Wastewater Reuse has been developed by the former Ministry of Agriculture, Water and Forestry and its Department of Water Affaires and Forestry (DWAF, 2012).

The requirements and quality standards depend on the origin of the wastewater: greywater, domestic or industrial effluents as well as the purpose of the reused water. Figure 2.2 shows the required treatment technologies to fit the allowed reuse purposes. Raw sewage and effluents from systems with only mechanical (primary) treatment as well as WSP with low HRT in MP are excluded for irrigation. The Minimum requirement for the irrigation of fodder crops are WSP with at least a 40 days HRT in MP. For all other purposes, secondary, tertiary or advanced treatment is required which cannot be reached by WSP. For fodder crops the effluent water quality has to comply with the general standard. The special standard is applied for the irrigation of vegetables and crops consumed raw by humans.

Table 2.1 shows an extract of the water quality required by the general and special standard.

Table 2.1: Extract of the general and special water quality standards for effluents (DWAF, 2012)

	Turbidity [NTU]	**pH [-]**	**TSS [mg/L]**	**COD [mg/L]**	**NH_4-N [mg/L]**	**TKN [mg/L]**
General Standard	12	6.5 - 9.5	100	100	10	33
Special Standard	5	6.5 - 9.5	40	55	1	5

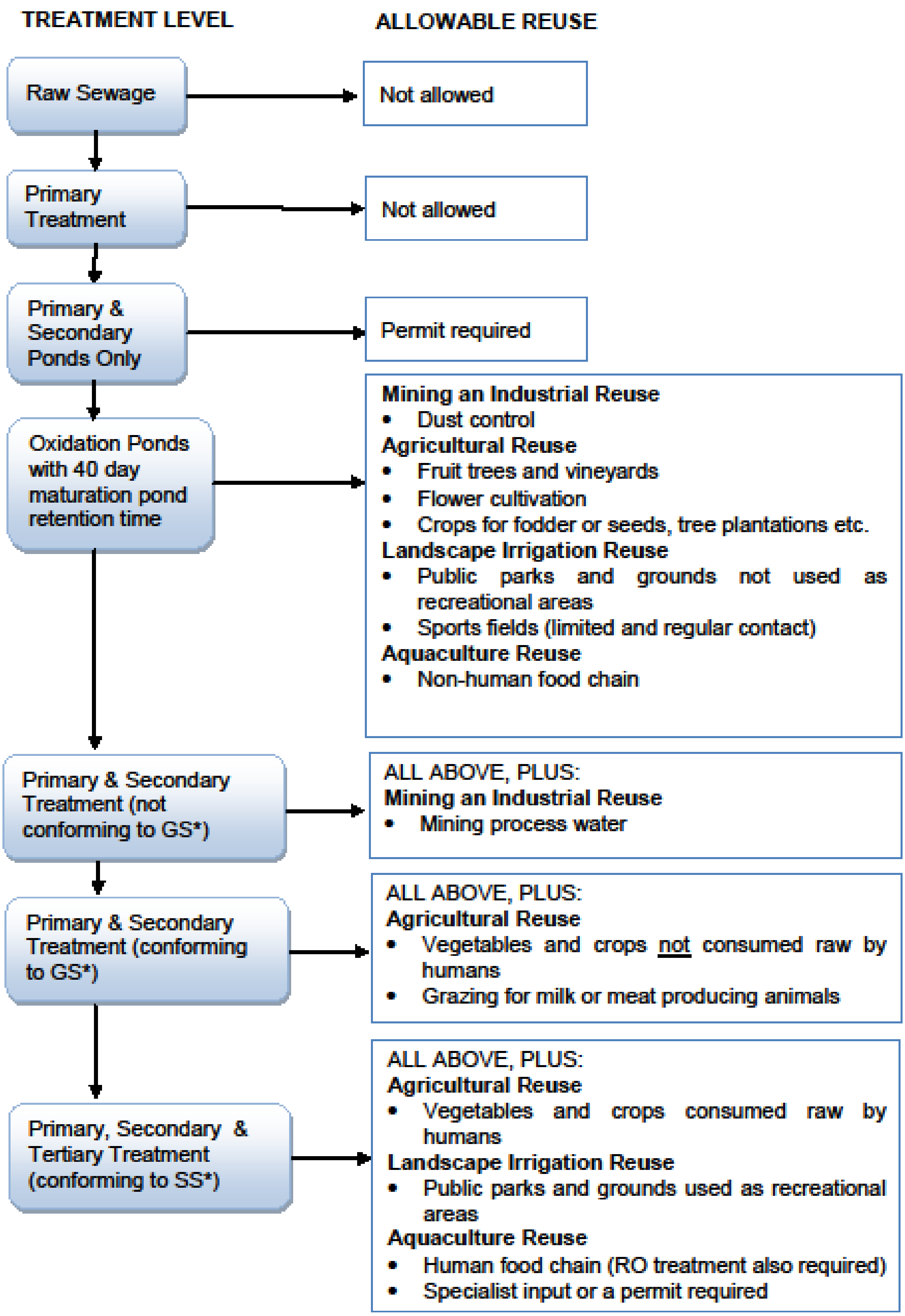

Figure 2.2: Reuse Flowchart showing allowed reuse purposes in relation to the achieved treatment level in Namibia (GS= General Standard for effluent, SS= Special Standard for effluent) (DWAF, 2012)

2.2.2 European reuse standards

The regulation 2020/741 of the European Parliament and of the Council of 25 May 2020 on minimum requirements for water reuse contains the first standard for the reuse of treated wastewater especially for agricultural irrigation in the European Union (EU, 2020). This regulation is based on the Council Directive 91/271/EEC which requires that treated wastewater is reused whenever appropriate (EU, 1991) and will be applicable from 26 June 2023. All wastewater has to be treated in municipal wastewater treatment plants and the required effluent values (Table 2.2) are based on the guidelines for treated wastewater use for irrigation projects by the International Organization for Standardization (ISO, 2015) and on the WHO Guidelines for the safe use of wastewater, excreta and greywater by the World Health Organization (WHO, 2006).

Table 2.2: Extract of the reclaimed water quality requirements for agricultural irrigation by the EU-regulation 2020/741 (EU, 2020). The first TSS value relates to large plants of more than 10,000 people equivalents. The second value relates to small plants between 2,000 and 10,000 people equivalents.

Reclaimed water quality class	**Indicative technology target**	***E. coli*** **[1/100mL]**	**BOD_5 [mg/L]**	**TSS [mg/L]**	**Turbidity [NTU]**
A	Secondary treatment, filtration, and disinfection	≤ 10	≤ 10	≤ 10	≤ 5
B	Secondary treatment, and disinfection	≤ 100	≤ 25	≤ 35 / ≤ 60	-
C	Secondary treatment, and disinfection	≤ 1,000	≤ 25	≤ 35 / ≤ 60	-
D	Secondary treatment, and disinfection	≤ 10,000	≤ 25	≤ 35 / ≤ 60	-

2.3 Enhancement with pre- and post-treatment in this study

With fast growing populations existing WSP systems are reaching their treatment capacity and therefore need to be upgraded (Dias et al., 2017a). Either additional ponds are constructed or alternatively they are equipped with pre- or post-treatment. Micro sieves (MS) (Walder et al., 2015) and an upflow anaerobic sludge blanket (UASB) reactor (Cavalcanti, 2003) are possible additional pre-treatment (PreT) technologies whilst sludge can be removed and baffles installed in the existing FP as well as a rock filter as post-treatment (PostT) in the MP (Middlebrooks, 1995).

2.3.1 Micro sieve for pre-treatment of waste stabilization ponds

Within wastewater treatment MS are mainly installed as PreT to reduce TSS. In some applications they have also been researched as post-treatment to eliminate helminths (Müller, 2017). The elimination is fully based on the physical process of sieving with a drum (Figure 2.3) or disc sieve. Both types are fed from the inside where the solids are retained on the mesh and once a specific water level is reached the mesh is sprayed from the outside. The screening is collected in a trough, dewatered and discharged into a collection container. Mesh widths between 100 und 300 μm made of stainless steel or polyester screen cloth (Ljunggren, 2006; Tchobanoglous et al., 2014) are applied in municipal wastewater treatment of raw water (Gikas and Tsoutsos, 2015). However, depending on the particle size distribution of the TSS even smaller fractions can be retained due to the filter cake on the sieve (Ljunggren, 2006; Rusten and Ødegaard, 2006; Shea and Males, 1971).

Figure 2.3: Micro sieve with sieving drum (left) and screening discharge (right)

The discharged screenings can reach a dry matter content of 6 % (Wid and Horan, 2016) and when coupled with an anaerobic digestion process biogas can be produced. Depending on the national legislation the contained phosphorous in the digested sludge can be used in agriculture as valuable fertilizer (Marti et al., 2008). At laboratory scale TSS retention of up to 60 % and COD reduction of up to 30 % can be reached with TSS effluent concentrations of 34 mg/L (Ljunggren, 2006; Rusten and Ødegaard, 2006).

MS installed as tertiary treatment with smaller mesh widths of 10 – 20 μm can reach TSS effluent concentrations below 5 mg/L (Grau et al., 1994). With WSP they can also be used to separate algae (Hendricks, 2011; Kothandaraman and Evans, 1972).

2.3.2 Upflow anaerobic sludge blanket reactors as pre-treatment of waste stabilization ponds in warm climates

The upflow anaerobic sludge blanket (UASB) reactor is a wastewater treatment stage based on anaerobic digestion within a sludge blanket of high microbial activity. The raw wastewater is flowing vertically from bottom to top through the reactor. The profile of the settleable solids within the reactor are graded from densely packed or granular particles at the bottom (sludge blanket) up to light sludge with dispersed solids at the reactor head (Chernicharo, 2007; Lettinga et al., 1980). Anaerobic microorganisms are digesting the organic material throughout the reactor. Constant flow from the bottom and rising gas bubbles assure their continuous circulation. At the top, the separation of solids, gas and liquids is implemented with a sharp-edged overflow in the phase separator (Figure 2.4) (Cavalcanti and van Haandel, 1996; Chernicharo, 2007; Tiwari et al., 2006).

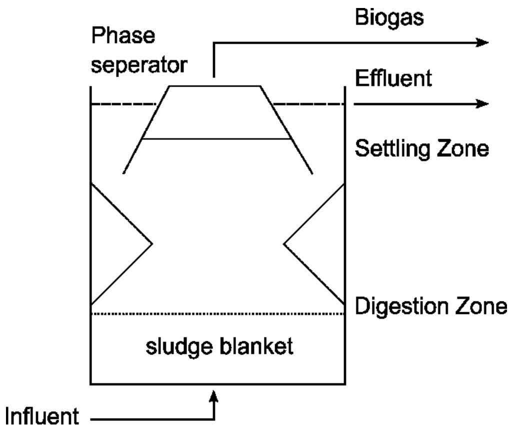

Figure 2.4: Schematic representation of an UASB reactor as anaerobic pre-treatment process (Cavalcanti and van Haandel, 1996)

Originally the UASB has been developed for wastewaters from industry with high COD concentrations (Lettinga et al., 1980). However, due to its robustness, easy operation and maintenance as well as reduced costs it is also implemented in countries with warm climates (Chernicharo, 2007; Mara, 2004). Its main purpose is the reduction of organic material and partially the reduction of pathogens whilst nutrients such as nitrogen and phosphorous concentrations are only slightly reduced (Tian et al., 2015). The reduction of COD strongly depends on the water temperature (Lew et al., 2004; Seghezzo et al., 1998; Uemura and Harada, 2000) and the HRT (Alaerts et al., 1993). The HRT ranges between 6 and 12 hours and is therefore shorter than the HRT of an anaerobic pond with 2 – 9 days (Mara, 2004). In contrast, the sludge age in UASB reactors usually exceeds 30 days, leading to stabilisation of the excess sludge removed from the system (Chernicharo, 2007). Most UASB reactors are operated in a mesophilic temperature range between 25 °C and 38 °C and are therefore particularly suitable for warm climates (von Sperling, 2007a).

In combination with WSP an UASB as PreT can replace an AP and therefore less land area and an up to 30 times smaller volume is required (Cavalcanti and van Haandel, 1996). With a sufficient HRT even the FP can be replaced and a much lower sludge accumulation rate is reached (Dias et al., 2014). However, according to Symonds et al. (2014) better pathogen reduction is reached with FP and two MP. With the UASB reactor about 70 % of TSS can be reduced and thus lower turbidity allows for higher light penetration in the subsequent ponds and stimulates photosynthesis of the algae (Dias et al., 2014).

2.3.3 Sludge removal

Sludge accumulation at the bottom of WSP occurs due to sedimentation of suspended solids as well as microbial growth of anaerobic bacteria (Effebi et al., 2011). It is greatest in primary ponds and its distribution depends strongly on their size as well the position and number of their inlets (Nelson et al., 2004). Nelson et al. (2004) found uniform sludge distribution in an AP with multiple inlets and a two days HRT, whilst in FP with singe inlets and higher HRT (9 – 41 days) most of the sludge accumulated directly in front of their inlet and in the corners.

Average sludge accumulation rates strongly depend on local conditions. Highest rates were measured in Australia with 0.12 – 0.23 m^3/person/year (Coggins et al., 2017). In France Picot et al. (2005) observed 0.04 – 0.148 m^3/person/year whilst in Brazil and Mexico only up to 0.036 m^3/person/year (Gonçalves, 1999; Nelson et al., 2004) were reported.

The effect of sludge accumulation in WSP is twofold on the hydraulics: at first the HRT is reduced due to volume reduction and secondly a flow channel is formed with higher sludge layers at the sides of the ponds because of lower flow velocity, both change the treatment efficiency (Coggins et al., 2017). Sludge accumulation not only impacts the available pond volume, but also effects carbon dioxide (CO_2) and methane (CH_4) emissions. With thick sludge layers Ho et al. (2021) observed extensive greenhouse gas emissions. Therefore, sludge management of WSP is important to improve effluent values as well to reduce their carbon footprint (Ho et al., 2021).

Figure 2.5: Removal of sun dried sludge from a facultative pond (left) and a maturation pond (right)

Desludging is recommended with a filling rate above 30 % or 2 – 5 years for AP (Oakley et al., 2012) and in primary FP after 15 years (Picot et al., 2005). Secondary FP can be desludged after more than 20 years (von Sperling, 2007b). However, in Botswana sludge was already removed after seven years due to high loads (Letshwenyo, 2021).

The sludge is either extracted by suction from below the water surface, which is more expensive and needs special equipment, or after emptying of the ponds, which needs a parallel bypass (Nelson et al., 2004; Picot et al., 2005). After drying this sludge can be easily removed with heavy equipment such as excavators (Oakley et al., 2012) or also manually (Figure 2.5). This periodic sludge removal needs to be considered already during the design and must be an integral part of the overall operation and management of the treatment process but is often missing in developing regions (Oakley et al., 2012).

2.3.4 Baffles to improve flow regime

Most WSPs are hydraulically inefficient, and pond hydraulics are further impaired by sludge accumulation and short-circuiting (Coggins et al., 2017). Besides the above mentioned regular removal of sludge it is also possible to increase the HRT (Verbyla et al., 2016). The HRT distinguishes the time water spends within a specific pond and is calculated with the inflow and the pond volume. However, the real retention time is often much shorter, especially with inefficient pond geometry or poor positioning of inlet and outlets. Tracer tests in WSP showed real retention times only reaching 45 % of the calculated HRT (Crites et al., 2014). Especially disinfection can be improved with longer HRT and therefore WSP can be enhanced with baffles that improve the flow pattern (Curtis, 2003).

In addition to the improvement of the hydraulic flow pattern baffles add further submerged surface to which microorganisms can attach themselves and increase their concentration (Muttamara and Puetpaiboon, 1997) and additionally contribute to the reduction of nitrogen and carbon compounds (Pearson, 2005). Baffles can be installed at FP as well as MP, especially if a higher length-to-width ratio is needed to improve pathogen reduction (Verbyla et al., 2017). Organic matter and pathogen concentrations following first-order kinetics are best reduced under plug-flow conditions, which can only be approximated with high length-to-with ratios. According to von Sperling (2007a) these should be larger than 10 in a single MP, between 1 and 5 in a series of more than 3 MP, between 2 and 4 in a primary FP and between 1 and 3 in an AP.

Figure 2.6: Preparation of floating baffle 1 (left) and baffle 2 installed in a facultative pond (right)

Ideally baffles are included with the original design of the pond and can be constructed as concrete or earth walls, but it is also possible to retrofit floating baffles (Figure 2.6) to avoid short-circuiting (Shilton and Harrison, 2003). Modelling of water flows in WSP is more widely applied, but these models are mainly validated by hydrodynamic or laboratory models and seldom at real pond scale (Coggins et al., 2018). The practical example of Coggins et al. (2018) in Australia proved a better hydraulic performance due to decreased short-circuiting and increased retention time by at least 20 %. According to Shilton and Harrison (2003) one baffle has no effect, but two or three baffles are technically and financially most effective and should be spread evenly across the pond. They achieved a pathogen reduction of up to 5 log units with the installation of two baffles over 65 % of the length of the pond.

2.3.5 Rock filter for removal of algae

WSP systems are an effective technology to treat wastewater but the presence of algae, which provide the major source of oxygen, is responsible for high concentrations of organic matter and suspended solids in the effluent (Dias et al., 2017b; Shelef and Azov, 2000). These solids originate mainly from organic algae matter and other pond debris but not from the original wastewater (Middlebrooks, 1995). According to von Sperling (2007a), 90 % of the TSS in the effluent consist of algae. When the treated wastewater is used for irrigation, the solids can lead to blockages of the small emitter orifices (0.1 - 2 mm) in the irrigation system (Lazarova and Bahri, 2005).

For the removal of algae, rock filters (Figure 2.7) offer a cost-effective and efficient alternative for post-treatment (Juanicó and Milstein, 2004; Mara and Johnson, 2007; Shelef and Azov, 2000; U.S. EPA, 2011). Rock filter operate by wastewater flowing through a submerged porous rock bed, causing algae to settle on the rock surfaces as the fluid flows through the cavities (Saidam et al., 1995). The collected algae are then biologically degraded and additionally pathogens reduced (Crites et al., 2014; Dias et al., 2017a). In some cases, rock filters have even replaced MP to polish WSP effluents (Mara and Johnson, 2007).

Figure 2.7: Rock filter installed in a maturation pond

Typical depths of rock filters are between 1.5 and 2 metres with rock diameters of approximately 10 – 12 cm (Shelef and Azov, 2000). However, von Sperling et al. (2007) proved that smaller rock sizes (3 – 10 cm) perform better than larger rocks (8 – 20 cm) whilst Dias et al. (2017a) have reached best results with three decreasing grain sizes. According to Mara (2004), the porosity of the rock filter medium can be assumed as 0.4 and the best hydraulic loading rate was identified with 0.5 $m^3/m^2/d$ (von Sperling et al., 2007).

So far there are no cleaning procedures implemented (Mara, 2004; von Sperling, 2007a), but several rock filters have been operating for 10 – 20 years without blockage (Dias et al., 2014; Middlebrooks, 1995) and negligible head losses (von Sperling et al., 2007). With a rock filter up to 60 % of suspended solids (Saidam et al., 1995), up to 70 – 90 % of BOD_5 (Fuhrmann, 2014; Shilton, 2005) and up to 90 % of algal concentration (Shelef and Azov, 2000) can be decreased. At the same time it remains a low-tech system without power, mechanical or chemical requirements as well as low construction and maintenance costs (von Sperling et al., 2007).

2.4 Study area in Namibia

North-central Namibia with its administrative regions Ohangwena, Omusati, Oshana and Oshikoto covers about 15 % of the national land area and in 2016 was home to 42 % of the total population (NSA, 2017). The climate is semi-arid and characterised by highly variable climatic conditions and seasonal rainfall falling mostly from November to April with spatial heterogenic rainfall patterns (Mendelsohn et al., 2002). The precipitation rate varies throughout the region, decreasing from east to west and from north to south with an annual average rainfall in the Omusati region from 350 to 500 mm/a. The evaporation rate is relatively high with an annual average of 1,820 – 2,240 mm/a, by far exceeding the precipitation. The average temperature of the region ranges from 25 °C in the summer to 17 °C in the winter (Mendelsohn et al., 2002). Between 2018 and 2019 only 50 % of the average annual rainfall was measured at the local weather station and increased aridity and evaporation is predicted (Angula and Kaundjua, 2016).

57 % of the rural population relies on subsistence farming (NSA, 2011) with rainfed crop and livestock production which is strongly affected by drought and high temperatures (Angula and Kaundjua, 2016). Especially in 2013 farmers and pastoralists experienced a major drought and again in the 2014 - 2015 season, which started well in October/November but in February/March was the driest in 27 years causing 30 % crop reduction and a strong decrease in grassland biomass as well as depletion of surface water bodies used for cattle drinking (Kerdiles et al., 2015). Ecosystem and land degradation is also aggravated by population growth and increased livestock keeping for prestige, wealth, manure as well as for meat and income generation (Kangombe, 2010).

2.4.1 Water and sanitation in neighbouring communities

Fresh water is not only limited due to high evaporation rates in north-central Namibia but also because of groundwater bodies with high salinity (Mendelsohn et al., 2002). Therefore, since 1974 water supply and also irrigation is based on surface water from the Kunene River at the Angolan border which is transported over 160 km in an open canal from Calueque to Oshakati (Kluge et al., 2008).

At the same time the first WSPs have been built in towns and most were extended after 10 to 20 years (Table SI 8.1.1) due to population growth. Urban household sanitation is mainly based on septic tanks and separate sewerage systems evacuating in WSPs, whilst rain water is evacuated in open drainage system (Liehr et al., 2018). In semi-urban areas with simple houses or shacks pit latrines are utilised as well as open defecation (Deffner and Mazambani, 2010).

So far the treatment of wastewater was only a minor aim, whilst the main purpose of WSP was evaporation and therefore they were often poorly managed. The potential for water reuse in agriculture has only been discussed after recent droughts especially with the focus on integrated water resource management (IWRM) (Kluge et al., 2008).

2.4.2 Urban centre with pilot plant

The pilot plant was established at the existing WSP system in Outapi (Figure 2.8), since 1998 capital of Omusati region. In 2001 Outapi was the smallest town in Namibia but as a commercial hub it grew quickly with 6,437 inhabitants in 2011 (NSA, 2011) and with a growth rate of 9.3 % to about 11,000 in 2018 (Mwinga et al., 2018). This rapid growth is a challenge to planning and implementation of sanitation infrastructure, sewage collection and wastewater treatment (Liehr et al., 2018; Müller, 2017). Due to the flat topography the wastewater is pumped with 9 pump stations to the WSP built in 2004 and designed for about 5,000 people. The WSP consist of two parallel treatment trains each with one primary FP and three MP followed by an EP which was constructed later. Especially during the rainy season surface water is entering the sewer system and overloading the WSP so that the EP is overflowing into the surrounding area.

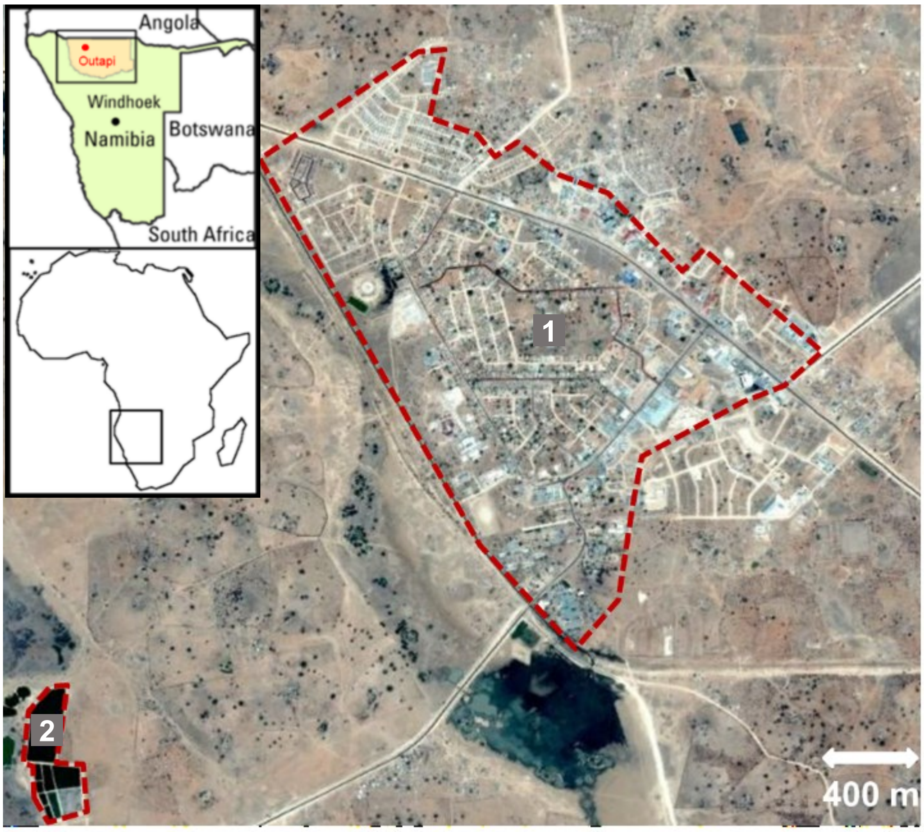

Figure 2.8: Location of sewer catchment (1) and waste stabilisation ponds (2) (Google-Earth (2016); Price and Hegnauer (2016), modified)

Given the need for irrigation water in north-central Namibia and challenges with existing WSP the research project EPoNa was implemented from September 2016 until August 2020. The first aim of the technical approach was to establish the current state and performance of WSP in the region, as well as their potential for water reuse. This included the characterization and evaluation of nine existing WSP systems, the analysis of their microbial communities as well as the composition of pathogens and cyanobacteria in their influent and effluent (chapter 3).

The second aim was to research different biological and physical enhancements at the existing WSP pilot plant to improve the environmental situation and to generate irrigation water. Hence, an anaerobic biological and a mechanical PreT technology were compared and evaluated (chapter 4). Additionally, the FP was equipped with floating baffles to improve flow conditions and the last MP was turned into a rock filter as PostT to retain algae. The performance of the baffles and the rock filter especially with regards to algae retention, pathogen reduction and their effect on the microbial community were examined (chapter 5).

The last aim was to establish the compliance of traditional and upgraded WSP with the Namibian and the European reuse guidelines. Therefore, the impact of algae on the particulate COD was elaborated (chapter 3). The results of the pilot plant were compared with the standards and further obstacles for agricultural water reuse as well as operation and maintenance of existing WSP discussed (chapter 5).

3 Characterization and evaluation of waste stabilization pond systems in Namibia

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3.1 Abstract

Waste stabilization ponds (WSP) exist worldwide to treat wastewater, especially in warm climates. They are characterized by simple operation and maintenance and over 50 years many WSP were built in urban communities in Namibia. This study characterized and evaluated nine of these WSP systems in terms of their influent and effluent water quality and compared them with the requirements for water reuse in agriculture. In their current state none of them adhered with the Namibian or the new European reuse standards, especially due to tCOD concentrations above 100 mg/L caused by high algal fractions in the pCOD. The algae related chlorophyll-*a* concentrations correlated linearly with the pCOD and this correlation can therefore be used to fractionate the tCOD for further judgement. Additionally, microbial community analyses determined the composition of pathogens in the WSP influent and effluent, this helped to assess potential risks and distinguish between potentially toxic and non-toxic cyanobacteria. The EU requirement of less than 1,000 *E. coli* per 100 mL for fodder crop irrigation of was only achieved with one WSP system which was enhanced with additional pre- and post-treatment. This research delivers a first overview of the current situation and can be used as basis to establish possible enhancement measures for existing WSP as well to investigate possible effluent application in agricultural irrigation.

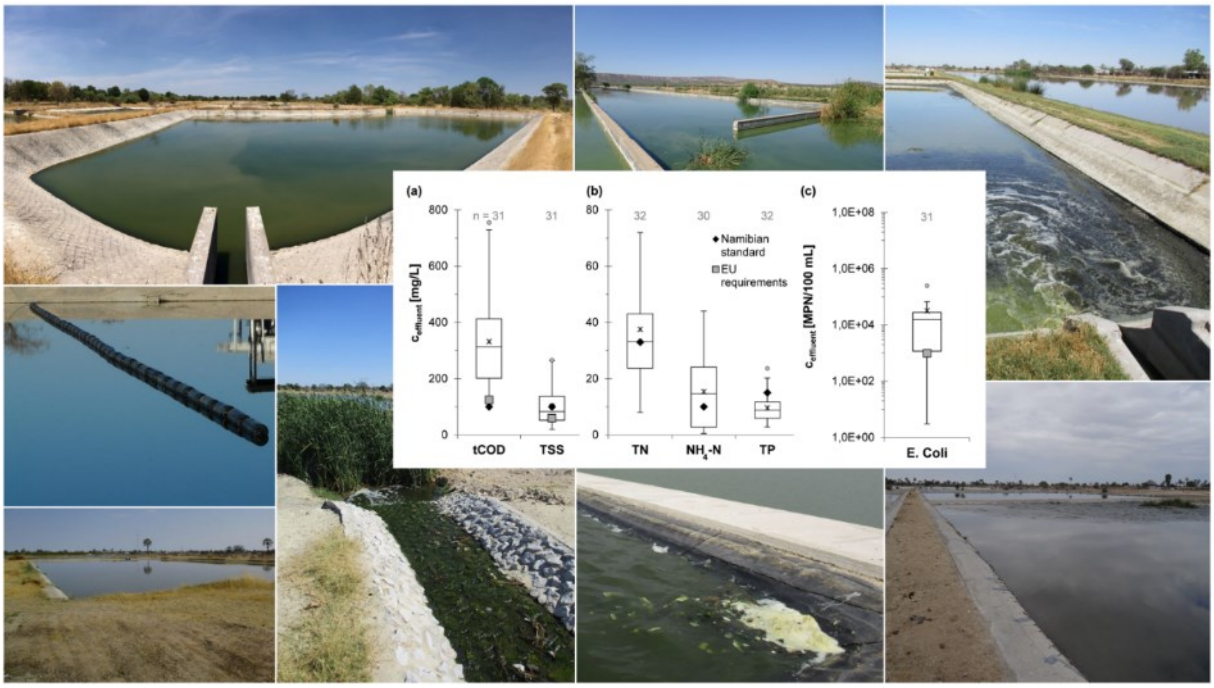

Figure 3.1: Graphical abstract: Characterization and evaluation of waste stabilization pond systems in Namibia

3.2 Introduction

Waste stabilization ponds (WSP) are a cost-effective treatment option for towns in water scarce areas with only few disadvantages such as large land requirements, high methane emissions and high concentrations of chemical oxygen demand (COD) and total suspended solids (TSS) in the effluent due to algae (Alves et al., 2020; Mara, 2004). Conventional pond system design consists of an anaerobic pond (AP) followed by a facultative pond (FP) and several maturation ponds (MP) (von Sperling, 2007a).

In many African countries WSP are most common for wastewater treatment (Bansah and Suglo, 2016; Edokpayi et al., 2021; Janeiro et al., 2020; K'Oreje et al., 2020; Kihila et al., 2014; Nikiema et al., 2013; Zacharia et al., 2019). But compared to Latin America (Alves et al., 2020; Dias et al., 2017b; Hernandez-Paniagua et al., 2014; Nelson et al., 2004; Verbyla et al., 2016) and Australia (Buchanan et al., 2018; Gruchlik et al., 2018; Rose et al., 2019) limited information is available about their performance and potential for water reuse in agriculture. Additionally, the occurrence and composition of the algal biomass in WSP has been investigated worldwide (Eland et al., 2018; Liu et al., 2020; Pham et al., 2014; Wallace et al., 2015) but not much in Africa.

In regions without perennial receiving water bodies such as Namibia, WSP concepts also include an evaporation pond (EP)(DWAF, 2008) to facilitate complete evaporation. Alternatively, reuse of the effluent, e.g. for irrigation, has a twofold benefit: the treated wastewater is put to use and provides an important source of water and nutrients (Mara, 2009). But WSP are often overloaded due to high population growth and effluent values often exceed the required quality standards for water reuse (Ho and Goethals, 2020).

Water reuse requires certain quality standards, with focus on COD removal and reduction of pathogens. Nutrients such as nitrogen, phosphorous and potassium can remain in the water and add additional value for the irrigation of plants as they complement fertilization. COD threshold values range from 100 mg/L in Namibia (DWAF, 2012) to 125 mg/L under the new EU regulation (EU, 2020) (Table 3.1) but are often exceeded due to the formation of algae biomass (Alves et al., 2020) that is not well retained in WSPs and thus detected in the effluent. Algae are not necessarily harmful and might even be beneficial for irrigation purposes, with the exception of potentially toxic species (Eland et al., 2018).

Depending on the reuse application, i.e. irrigation of green space, fodder crops or vegetables, different levels of reduction for pathogens are required. WSP rely on natural UV disinfection and the efficiency depends on the appropriate hydraulic retention times. Thus effluent quality has to be monitored carefully.

For Africa and particularly in Namibia only limited information about the water quality attained by WSP of different design, state of operation and general condition is available. Therefore, this study aims to provide a first data set of nine WSP systems in the North of Namibia, focusing on physical-chemical parameters and the microbial community. In particular, we investigated the microbial community composition with a focus on pathogens and cyanobacteria using modern sequencing methods. Such data has not yet been published and therefore adds new insights into the diversity of the microbial community in addition to standard indictor organisms.

Additionally, the WSP systems were compared by means of standard physical-chemical wastewater parameters to generate an overview of inflow and effluent values of mostly overloaded WSP with various configurations. This also includes the calculation of total COD effluent values without algal matter in order to compare the effluents with the national and international COD requirements which could otherwise only be reached with high technological input.

Table 3.1: Effluent water quality and performance comparison: Load reduction and effluent concentrations judged against the Namibian (DWAF, 2012) and EU (EU, 2020) standards. The WSP are grouped according to their system setup. Green indicate WSP that fulfil the requirements, yellow show concentrations up to 20 % above the requirements or values for slight to moderate irrigation restrictions (Ayers and Westcot, 1985) and in red are all the WSP that are more than 20 % above the requirements.

System	WSP	#	TSS	tCOD	tCOD $_{(w/o)}$	TP
FP+MP	D, G1, H	removal:	23 -85%(load)	38 - 77% (load)	-	17 - 53% (load)
		effluent value:	57 - 253 mg/L	182 - 568 mg/L	85 -303 mg/L	4.2 - 10.3 mg/L
AP+FP+MP	A, B, E, F	removal:	23 -91% (load)	21 - 85% (load)	-	16 - 70% (load)
		effluent value:	49 - 266 mg/L	198 - 647 mg/L	66 - 327 mg/L	5.1 - 23.7 mg/L
AP+FP+MP	C, I	removal:	no effluent	no effluent	-	no effluent
w/o effluent		last pond value:	26 - 188 mg/L	116 - 425 mg/L	75 - 95 mg/L	2.8 - 9.7 mg/L
PreT+FP+MP	G2	removal:	48 - 71% (load)	55 -71% (load)	-	0 - 13% (load)
		effluent value:	93 - 144 mg/L	371 - 466 mg/L	131 - 338 mg/L	11.1 - 12.8 mg/L
PreT+FP+MP+PostT	G3	removal:	86 - 96% (load)	64 - 84% (load)	-	64% (load)
		effluent value:	20-48 mg/L	173 - 202 mg/L	59 - 144 mg/L	4.8 - 13.5 mg/L
national and international reuse values						
Namibian Reuse Standard (DWAF, 2012)			< 100 mg/L	< 100 mg/L	-	< 15 mg/L
EU Reuse Regulation (EU, 2020)			< 60 mg/L	< 125 mg/L	-	-
FAO (moderate) (Ayers and Westcot, 1985)			< 100 mg/L	-	-	< 13 mg/L

System	WSP	#	TN	NH_4-N	EC	E.Coli
FP+MP	D, G1, H	removal:	25 - 57% (load)	49 - 99% (load)	0 - 27% increase	2 - 4 $\log_{10}$ units
		effluent value:	26 - 89 mg/L	1 - 38 mg/L	440 - 2700 µS/cm	1.0E+03 - 1.7E+05 (MPN/100mL)
AP+FP+MP	A, B, E, F	removal:	32 - 87% (load)	18 - 99% (load)	2 - 109% increase	2 - 5 $\log_{10}$ units
		effluent value:	19 -53 mg/L	1 - 31 mg/L	635 - 3150 µS/cm	3.0E+00 - 2.8E+04 (MPN/100mL)
AP+FP+MP	C, I	removal:	no effluent	no effluent	0 - 50% increase	2 - 4 $\log_{10}$ units
w/o effluent		last pond value:	8-41 mg/L	1 -20 mg/L	754 - 1352 µS/cm	1.3E+03 - 2.5E+05 (MPN/100mL)
PreT+FP+MP	G2	removal:	46 - 51% (load)	49 -74% (load)	0	3 - 4 $\log_{10}$ units
		effluent value:	40 - 56 mg/L	16 -36 mg/L	818 - 1062 µS/cm	1.0E+03 - 5.6E+04 (MPN/100mL)
PreT+FP+MP+PostT	G3	removal:	45 -85% (load)	55 - 98% (load)	2 - 32% increase	5 - 6 $\log_{10}$ units
		effluent value:	15 - 31 mg/L	2 - 22 mg/L	724 - 1072 µS/cm	1.5E+01 - 3.0E+02 (MPN/100mL)
national and international reuse values						
Namibian Reuse Standard (DWAF, 2012)			< 33 mg/L	< 10 mg/L	-	-
EU Reuse Regulation (EU, 2020)			-	-	-	< 1.0E+03 (MPN/100mL)
FAO (moderate) (Ayers and Westcot, 1985)			< 30 mg/L	< 5 mg/L	< 3000 µS/cm	-

AP = anaerobic pond, FP = facultative pond, MP = maturation pond, PreT = pretreatment, PostT = post treatment, WSP = waste stabilization pond system A – I, TSS = total suspended solids, tCOD = total chemical oxygen demand, tCOD (w/o) = tCOD without algae, TP = total phosphorous, NH_4-N = ammonia, EC = electrical conductivity, E. coli = Escherichia Coli

3.3 Material and Method

3.3.1 Data collection at nine WSP in North Namibia

This study characterized and evaluated nine WSP in North Namibia, where the majority of the country's population resides. Background information about each WSP was collected through semi-structured interviews with the local operation managers. The water quality was analysed between 2017 and 2020. Eight of the WSP (A – F, H and I) remained in their original setup over the whole research period. They were anonymized to ensure confidentiality. One plant (G) has been equipped with a mechanical and anaerobic pre-treatment, i.e. a micro sieve and an upflow anaerobic sludge blanket (UASB) reactor, in 2018 to reduce COD and TSS. In 2019 a post-treatment employing a rock filter to further decrease COD, algae and pathogens (Sinn and Lackner, 2020) was installed at WSP G. At this full-scale installation further enhancement measures such as sludge removal and baffles were also installed and investigated. G1 refers to samples from 2017 before the enhancement, G2 to samples that were taken after the installation of the pre-treatment in 2018 and G3 to the samples after the start-up of the post-treatment in 2019.

The daily evaporation was measured at WSP G with an iMetos SD weather station (Pessl Instruments, Austria) over four years. Due to similar climate conditions at all nine communities, an average evaporation rate of 5.4 mm/d was applied to calculate the theoretical water loss for each WSP. The surface area of the ponds was either provided during the interviews by the responsible person or estimated from aerial images. There were no design documents available, and therefore, no information regarding design values with respect to population or related loading rates are provided.

3.3.2 Sampling and analysis

1 L grab samples were taken during dry weather seasons at the influent of the WSP system and at the outflow of each WSP system once in October 2017 and twice in May and June 2019. Due to the local situation and lab availabilities, only a limited number of samples could be analysed. Long distances and local road conditions resulted in different sampling times, mostly during morning hours between 10 and 12 am, but some also in the early afternoon (2 till 4 pm). Samples were transported to the laboratory with a cooler box and analysed within 24 h. COD analyses were performed from homogenized (homogenizer: T 25 digital Ultra-Turrax, IKA, Germany) (total COD (tCOD)) and filtered (0.45 μm, Whatman membrane filters, ME 25) water samples (soluble (sCOD)). The particulate fractions of the COD (pCOD) were calculated as subtraction of the sCOD from the tCOD. The following parameters were analysed with Hach cuvette tests using a spectral photometer (DR 2800, Hach Lange, Germany): tCOD, total nitrogen (TN) and total phosphorus (TP) from homogenized samples as well as sCOD, ammonium (NH_4-N), nitrite (NO_2-N), nitrate (NO_3-N), phosphate (PO_4-P) and potassium (K^+) from filtered samples. Additionally, chlorophyll-*a* concentrations were determined according to German standard methods for the examination of water, wastewater and sludge (DIN 38409-60)(DIN, 2019).

TSS and volatile suspended solids (VSS) were measured by German standard methods (DIN 38409-2, 1987)(DIN, 1987) using Whatman 934-AH glass microfiber filters. Electrical conductivity (EC), pH, temperature and dissolved oxygen (DO) were analysed with a WTW multimeter 3410 (Xylem Analytics, Germany). Additionally, the concentrations of total coliforms and *Escherichia Coli* (Colilert-18), *Enterococci* (Enterolert) and *Pseudomonas aeruginosa* (Pseudalert) were measured with an IDEXX system employing a Quantiy-Tray/2000 (IDEXX, Germany).

3.3.3 16S rRNA amplicon sequencing to determine the composition of cyanobacteria

Samples were collected as biological triplicates at six (A, B, D, G, H and I) of the nine WSP in 50 mL centrifuge tubes to determine the composition and abundance of cyanobacteria and pathogens. These tubes were centrifuged at 8,000 g and 4 °C for 25 minutes. The supernatant was discarded and pellets were stored at 4 °C overnight and brought to Germany for further downstream analysis. Total genomic DNA was extracted using the Fast DNA Spin kit for soil (MP Biomedicals, Germany) according to a modified manufacturer's protocol (Orschler et al., 2019). DNA concentration was analysed using Qubit 3.0 Fluorometer with Qubit dsDNA HS kit (Thermo Fisher Scientific, Germany). Further, the DNA was used to perform 16S rRNA amplicon sequencing according to the method by Agrawal et al. (2020). Then the raw data was filtered for the sequences associated with the phylum cyanobacteria and the composition of the cyanobacteria was determined in R using ggplot. The abundance of genera associated with pathogenic bacterial species, was determined according to a previous study (Agrawal et al., 2020).

3.3.4 Data processing

The collected data was analysed with conventional statistical methods. Concentrations below the limit of quantification (LOQ) were not considered for evaluation. COD measurements influenced by chloride concentrations above 1,500 mg/L were also discarded. The performance of the WSP systems was evaluated by the water quality at the inflow in comparison to the effluent quality. Additionally, the performance was related to the code of practice for wastewater reuse of Namibia (DWAF, 2012), the regulation on minimum water requirements for water reuse in the European Union (EU, 2020) as well as the Food and Agriculture Organization (FAO) water quality standards (Ayers and Westcot, 1985). In Table 3.1 values shown in green indicate WSP that fulfil the requirements, WSP in yellow indicate concentrations of not more than 20 % above the requirements and all WSP in red are not even close to the standards (>> 20 % above the requirements).

3.3.5 Semi-structured interviews

Semi-structured interviews with operation staff in 2017 gave important background information on each WSP system especially with regards to the total population and the population connected to the sewers and wastewater treatment plants. This information is presented in Table SI 8.1.1. Only in a few towns technical drawings were available. Data from flow meters existed from WSP A, B and G. For the others, values were estimated by the operators (WSP C, D, E, F, H and I).

3.4 Results and Discussion

3.4.1 Waste stabilization pond systems – introduction of the study sites

This is the first comprehensive evaluation of WSP in Northern Namibia. These systems have been built since the 1970's. Almost every ten to fifteen years extensions were implemented to accommodate the constant urbanisation and population growth. The size of the towns connected to these WSP ranged from 2,300 to 50,000 inhabitants; with connection rates from 30 to 70 % (Table SI 8.1.1). Therefore, further upgrades or enhancements would be necessary to accommodate the total population. According to the Namibian code of practice for pond systems, WSP should only be designed for up to 5,000 population equivalents (PE) (DWAF, 2008). In consequence, eight out of nine communities would need new treatment systems. This is not possible, neither financially nor in time. Additionally, there is no local experience in operation and maintenance of activated sludge systems or trickling filters. Therefore, WSP remain the only reasonable solution under the given circumstances.

All available details from the operators, inflow quantities and information obtained from satellite images are summarized in Table SI 8.1.1. The surface areas of most WSP ranged between 7,000 and 50,000 m^2 while two plants considerably exceeded these values with a surface areas of 200,000 and 280,000 m^2. At the same time, the per capita land requirements varied from 1 to 26 m^2/cap. von Sperling (2007a) recommends values of 3 to 5 m^2/cap, which means that six plants were over designed and should have the capacity to serve more households. Two WSP were already too small and would need upgrades. The depth and consequently the pond volume was available from five WSP. For the remaining four WSP the volume was calculated with an average depth of 3.5 m for AP and 1.5 m for FP (von Sperling, 2007a). The volumes ranged from 8,300 to 360,000 m^3. The theoretical hydraulic retention times (HRT) were between 12 and 302 days as calculated from either the measured inflow or estimated values and the corresponding pond volumes. By these estimates, three of the plants (B, D, H) were already below the 40 days minimum HRT of MP required for water reuse in Namibia (DWAF, 2012). This calculation does not take into account short-circuiting and the reduced volume due to sludge accumulation over the years. At one plant, after 15 years of operation, the sludge layer in the primary facultative pond was between 20 and 70 cm thick. So roughly 1/3 was filled with sludge. However, in the subsequent ponds the sludge layers were much thinner. Thus, the available volumes are probably less and the real HRTs shorter particularly in the first ponds. However, as the sludge layers could not be determined for the other pond systems and for comparison purposes the theoretical HRT was calculated as a first estimate.

The nine WSP not only differed by size and HRT but also by the design of the different ponds. Four plants had one treatment train, five had two parallel trains. Three treated the raw sewerage directly in FP without AP. The number of MP varied between one at the smallest plant and up to seven at the oldest plant (D) constructed in the 1970s. All but one WSP had one or two EP at the end (Table SI 8.1.1).

3.4.2 Influent characteristics

The water quantity entering the different plants was between 83 and 2,160 m^3/d depending on the town size and the number of people connected. At three plants the actual inflow was measured with flow meters whilst at the others an average inflow of 120 L/cap/d (Table SI 8.1.1) was estimated. This inflow was confirmed by the local operators but lay below the average consumption of 163 L/cap/d in Windhoek calculated by Uhlendahl et al. (2010). It, however, reflected the local living standard and accounts for water losses in the systems and also from households with standpipes but no sewer connection. Based on this information the PE loads entering each plant were calculated and compared with typical values in Europe (Germany) and Africa (Uganda) (Tchobanoglous et al., 2014) as well as with a wastewater treatment plant using UASB and rotating biological contactors (RBC) in Outapi, Namibia (Müller, 2017) (Figure 3.2 and Table SI 8.1.1). The total COD (tCOD) was between 21 and 160 g/(PE·d) with an average of 88 g/(PE·d). These values were lower than in Germany or in Uganda with 123 g/(PE·d) (calculated as twofold the biological oxygen demand (BOD) (Tchobanoglous et al., 2014) but only slightly higher than the 75 g/(PE·d) of the Namibian comparison. The TSS values of 10 to 50 g/(PE·d) and 27 g/(PE·d) on average, were at a similar level as the study in Namibia with 29 g/(PE·d), but lower than in Uganda (48 g/(PE·d)) and Germany (89 g/(PE·d)). The PE loads for TN and TP were 5 to 13 g/(PE·d) and 0.5 to 1.7 g/(PE·d), respectively. The TN values were lower than in Germany (11 – 16 g/(PE·d)) but similar to Uganda (8 – 14 g/(PE·d)) and Namibia with 6 g/(PE·d) on average. The TP load in this study was within the same range as in Germany and Namibia, but more than double the load in Uganda. This indicates that the consumption patterns in Namibia were closer to Germany than to Uganda. Overall the loads were within the given ranges and therefore the inflow estimations were reasonable.

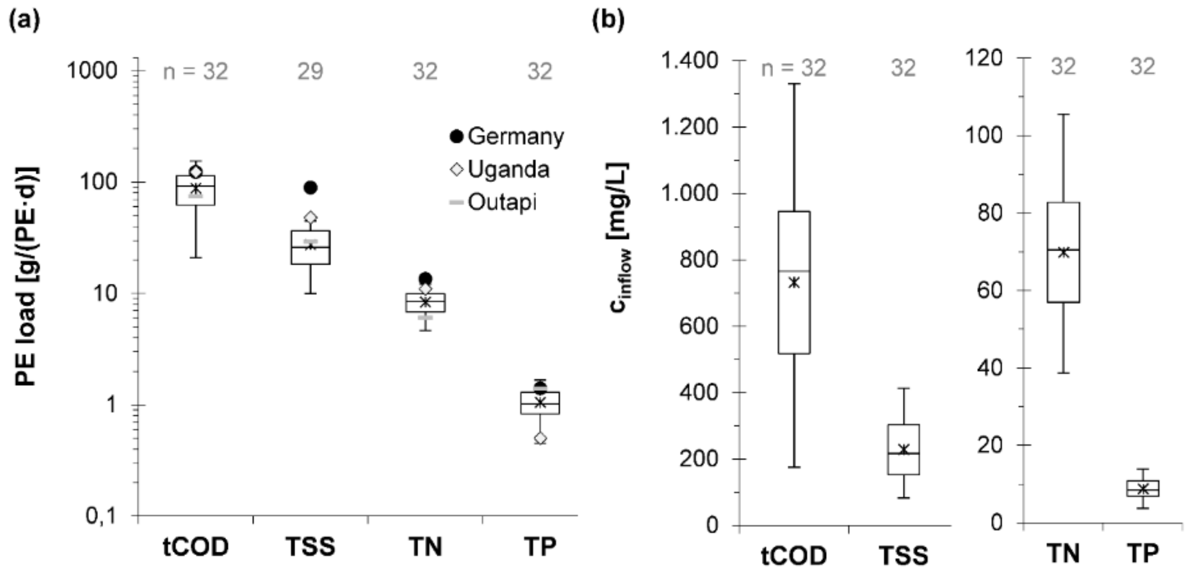

Figure 3.2: Inflow parameters as (a) population equivalent (PE) load of the nine waste stabilization ponds (WSP) in comparison with typical values in Germany and Uganda (Tchobanoglous et al., 2014) as well as with a biological treatment plant in Namibia (Müller, 2017) and (b) inflow concentrations (c_{inflow}) of the nine WSP for total chemical oxygen demand (tCOD), total suspended solids (TSS), total nitrogen (TN) and total phosphorus (TP).

The inflow concentrations of all plants varied considerably (Figure 3.2 and Table SI 8.1.2). As all samples were grab samples over a period of two years there were variations within one WSP as well as between all systems. Especially the tCOD of the single samples covered a wide range from 175 mg/L (plant I) to 1,331 mg/L (plant E) with an average of 733 mg/L. The TSS values were 83 to 413 mg/L and 229 mg/L on average. Nutrient concentrations were between 39 and 106 mg/L for TN and between 3.8 and 13.9 mg/L for TP. Inflow concentrations of soluble COD (sCOD), particulate COD (pCOD), NH_4-N, PO_4-P, total coliforms, *E. coli*, *Enterococci*, pH and EC are presented in Figure 3.3. This data provides a first glance at the wastewater composition in Namibia and is valuable for further research on WSP.

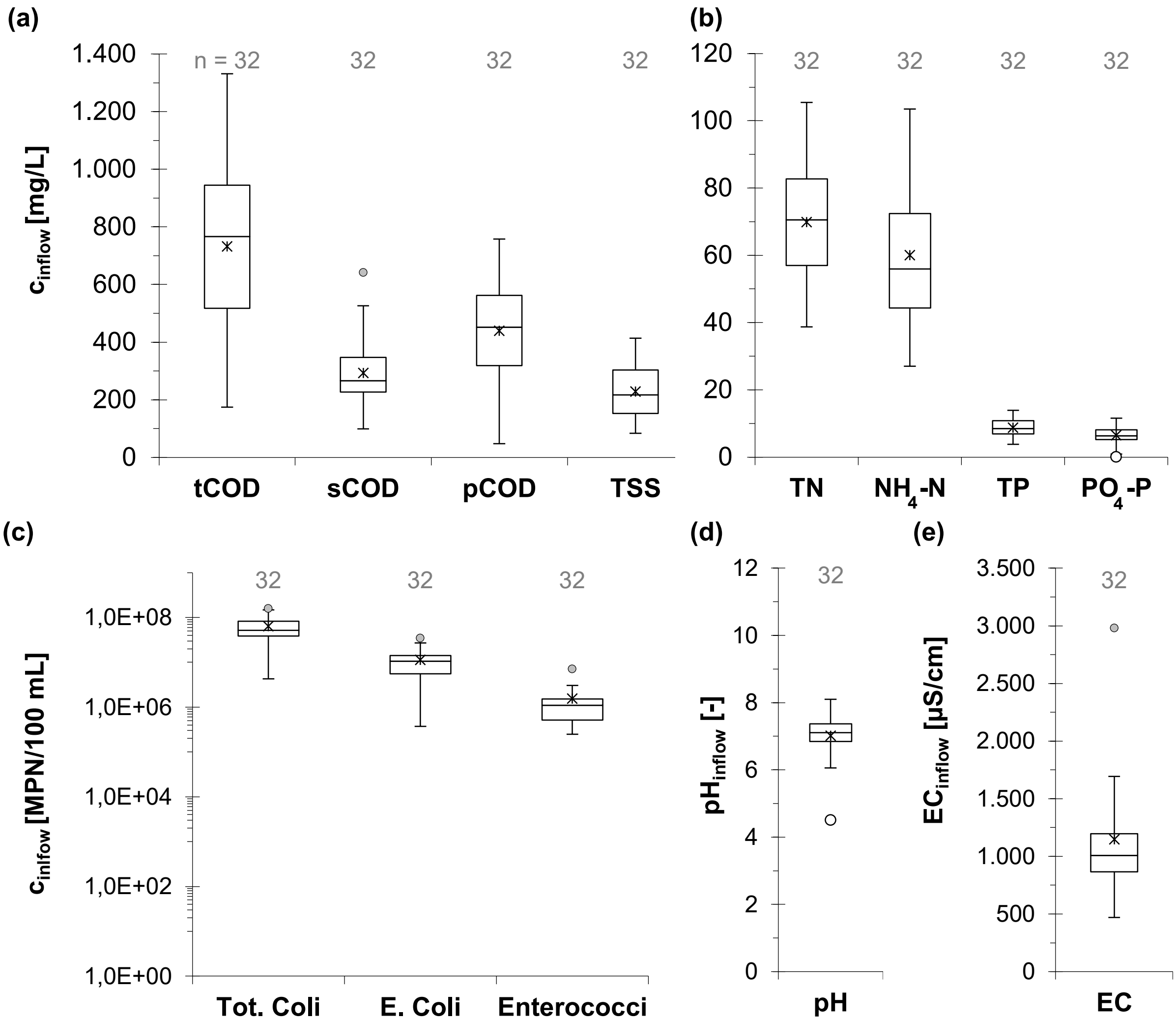

Figure 3.3: Inflow concentrations (c_{inflow}) of nine waste stabilization ponds (WSP): (a) total chemical oxygen demand (tCOD), soluble chemical oxygen demand (sCOD), particulate chemical oxygen demand (pCOD), total suspended solids (TSS), (b) total nitrogen (TN), ammonia (NH_4-N), total phosphorus (TP), phosphate (PO_4-P), (c) total coliforms (Tot. Coli), *Escherichia Coli* (*E. coli*), *Enterococci*, (d) pH and (e) electrical conductivity (EC).

3.4.3 Effluent characteristics

The tCOD concentrations, taken at the overflows to the EP or from the last MP of each WSP, ranged between 116 and 755 mg/L and were all above the Namibian standard of 100 mg/L and mostly also above the EU standard of 125 mg/L (Figure 3.4 and Table SI 8.1.2). This reflected very well the finding of Alves et al. (2020) in Bolivia for similar size WSP. The TSS concentrations showed an average concentration of 103 mg/L and thus almost met the local standard of 100 mg/L. They were however above the EU requirements (EU, 2020) of 60 mg/L. There are no requirements from the EU (EU, 2020) for nutrient concentrations as they pose no harm for water reuse and are even considered necessary for plant growth. On the contrary, in Namibia (DWAF, 2012) there are standards for TN, and NH_4-N (33 mg/L and 10 mg/L, respectively) which were just below the measured average effluent values of 38 mg/L and 15 mg/l. The TP concentration of 15 mg/L in the Namibian standard was above the measured values (average 10 mg/L) and not critical. Further removal might be required for TN and NH_4-N though.

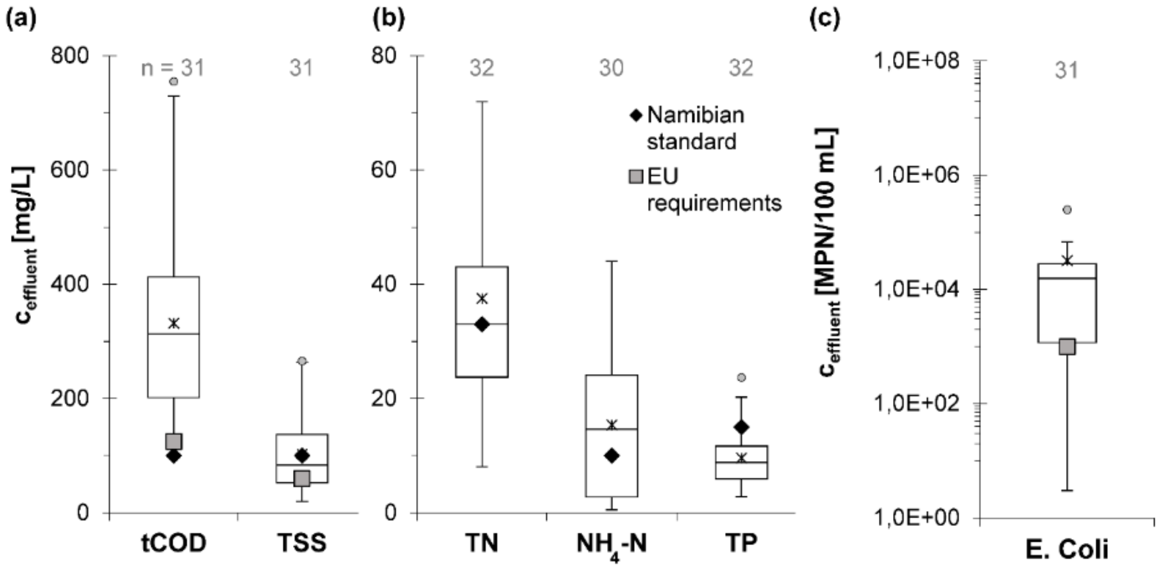

Figure 3.4: Effluent quality ($c_{effluent}$) in comparison with the Namibian Reuse Standard (DWAF, 2012) and the EU requirements (EU, 2020) for (a) total chemical oxygen demand (tCOD), total suspended solids (TSS), (b) total nitrogen (TN), ammonia (NH_4-N), total phosphorus (TP) and (c) *Escherichia Coli* (*E. coli*).

In Figure 3.5, the WSP were grouped according to their system setup: WSP A, B, E and F had the traditional setup of AP, FP and MP and were all overflowing. WSP C and I had the same setup but had no effluent. WSP D, G, and H formed the third group and represented the systems without AP. The systems without AP had higher effluent values for tCOD and TSS, whilst the group without effluent was close to the required standard due to the long HRT. With regards to the volumetric loading rate of the AP all plants were below the design parameters of 0.10 – 0.35 kg $BOD_5/m^3/d$ suggested by von Sperling (2007a) (Table SI 8.1.1). At two of the plants without AP the surface loading rate of 140 kg BOD_5/ha/d of the primary FP was at the lower end of the design values (von Sperling, 2007a). The surface loading rate of plant H was almost double the design value. The concentrations of *E. coli* were at a similar level in all groups. Altogether there was a wide divergence between all nine systems with hardly any compliance to national or international standards. Within the African context soil salinization needs consideration as well as salt acceptance of the irrigated plants. All the WSP showed EC values between 440 and 3,150 μS/cm which were within the slight to moderate restrictions of the FAO (Ayers and Westcot, 1985) and therefore allow for fodder irrigation.

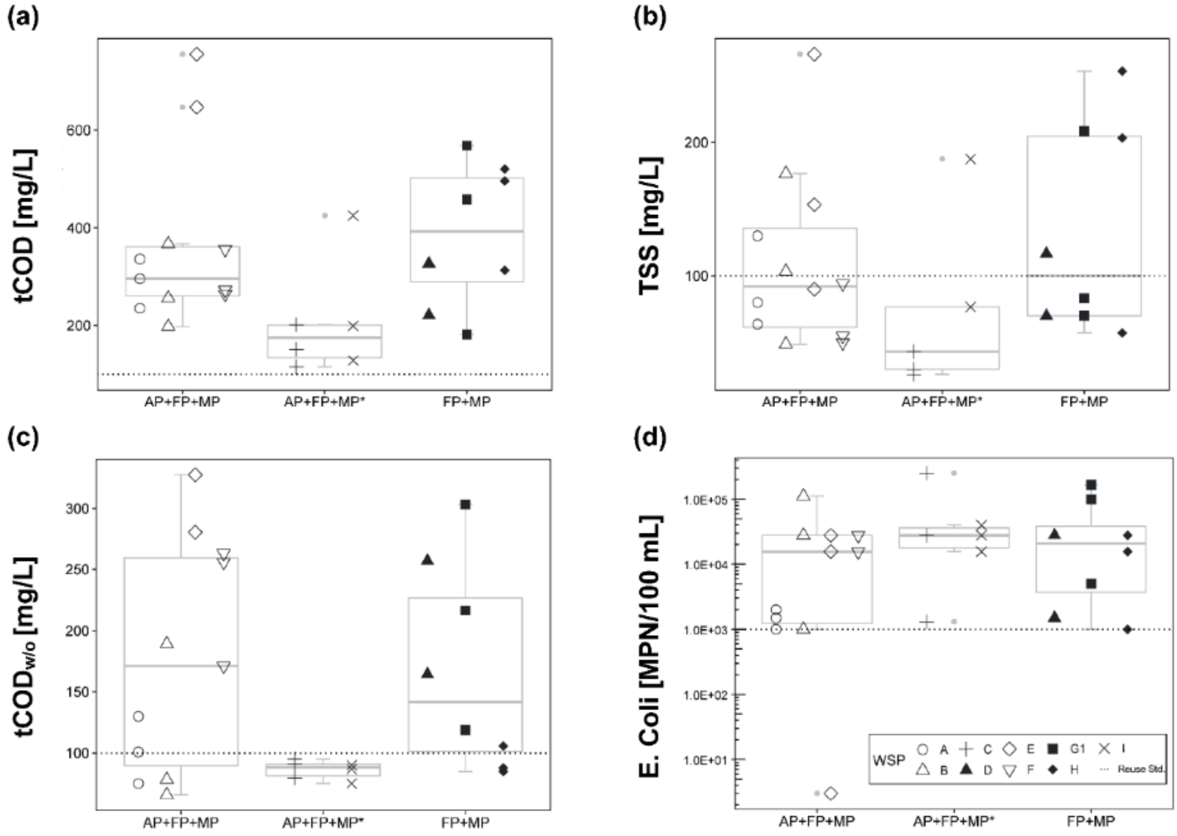

Figure 3.5: Effluent quality grouped according to the waste stabilization pond (WSP) system setup: AP = anaerobic pond, FP = facultative pond, MP = maturation pond, * = pond systems without effluent. The parameters total chemical oxygen demand (tCOD) (a), total suspended solids (TSS) (b), total chemical oxygen demand without algae (tCOD$_{w/o}$) (c), *Escherichia Coli* (*E. coli*) (d) are compared with the respective reuse standard (···).

3.4.4 Algal biomass

An important aspect to consider with regards to high tCOD effluent concentrations are algae. In WSP they find ideal growth conditions, especially in warm climates, and thus contribute considerable amounts of biomass to the pCOD in the effluent of MP. Measurements of chlorophyll-*a* can provide a first indication of the algae fraction in the pCOD. For the nine WSP, the chlorophyll-*a* concentrations ranged from 58 to 1,675 μg/L and correlated linearly with the pCOD. These values were in the same range as in Bolivia (Alves et al., 2020). Based on this relation, the pCOD due to algae was deducted from the tCOD and an adapted tCOD without algae (tCOD$_{w/o}$) was estimated resulting in values between 59 and 339 mg/L and an average of

162 mg/L (Figure 3.6). These values did not yet reach the required effluent standards of the EU (125 mg/L) and Namibia (100 mg/L), but gave an indication that either algae have to be removed or there is the need for different standards if the reuse water originates from WSP. In situations where chlorophyll-*a* analysis is not possible, the sCOD can also provide a first approximation. These concentrations will indicate whether the given standards for tCOD are reachable with removal of the algae. Too high values are an indication of an overloaded WSP or of possible non-biodegradable COD. In such cases, further enhancement measures have to be considered. For the nine plants the average sCOD concentration was exactly 100 mg/L with values between 47 and 251 mg/L.

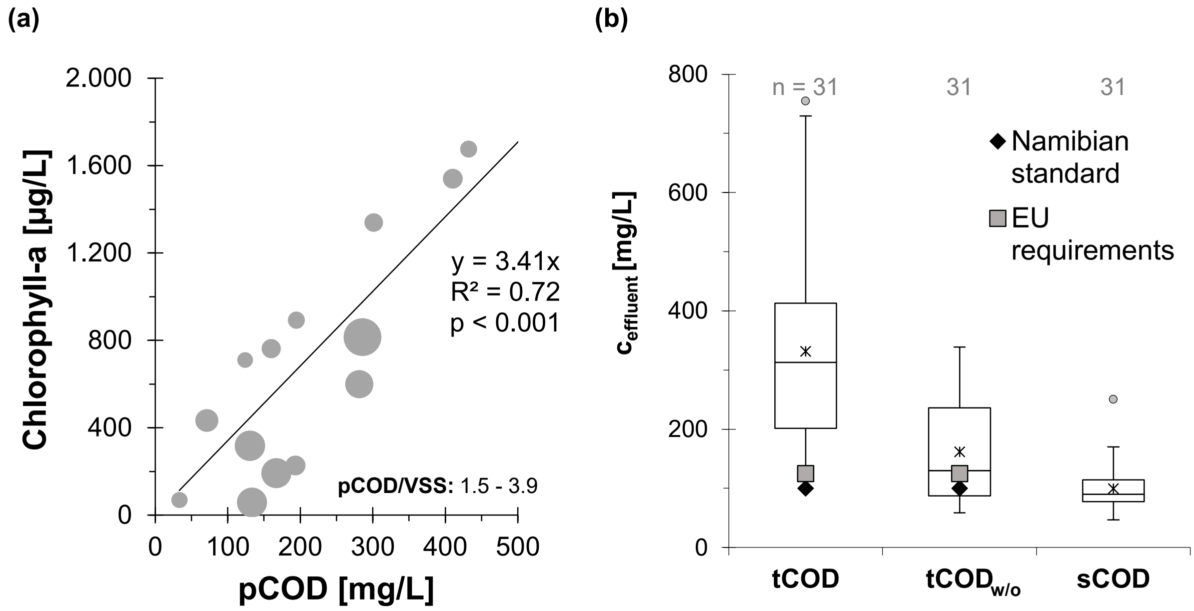

Figure 3.6: (a) Effluent ratio chlorophyll-*a* versus particulate chemical oxygen demand (pCOD) with the size indicating the relation of pCOD to volatile suspended solids (VSS). (b) chemical oxygen demand (COD) effluent concentrations ($c_{effluent}$) as total COD (tCOD), calculated tCOD without algae content ($tCOD_{w/o}$) and soluble COD (sCOD) in comparison with the Namibian Reuse Standard (DWAF, 2012) and the EU requirements (EU, 2020).

3.4.5 Microbial ecology

Microbiological parameters are also important for water reuse and water quality. Typically, the water quality is judged by cultivation methods capturing indicator bacteria, with total coliforms and *E. coli* being the most popular indicators (Liu et al., 2020). Total coliforms reached an average concentration of 3.3×10^6 MPN/100mL at the outflows which resulted in a reduction of only 2 log values. A slightly higher reduction of 3 log values was observed for *E. coli* and *Enterococci* with average concentrations of 3.2×10^4 MPN/100mL and 1.3×10^4 MPN/100mL, respectively. The EU (2020) stipulates less than 1,000 *E. coli* per 100 mL for the irrigation of fodder crops. Only one plant with a rock filter as post-treatment reached this value. Therefore, it needs to be evaluated if the natural disinfection processes in the MP can be improved or if other measures such as filters or technical disinfection have to be implemented. Further supporting parameters are presented in Figure 3.7.

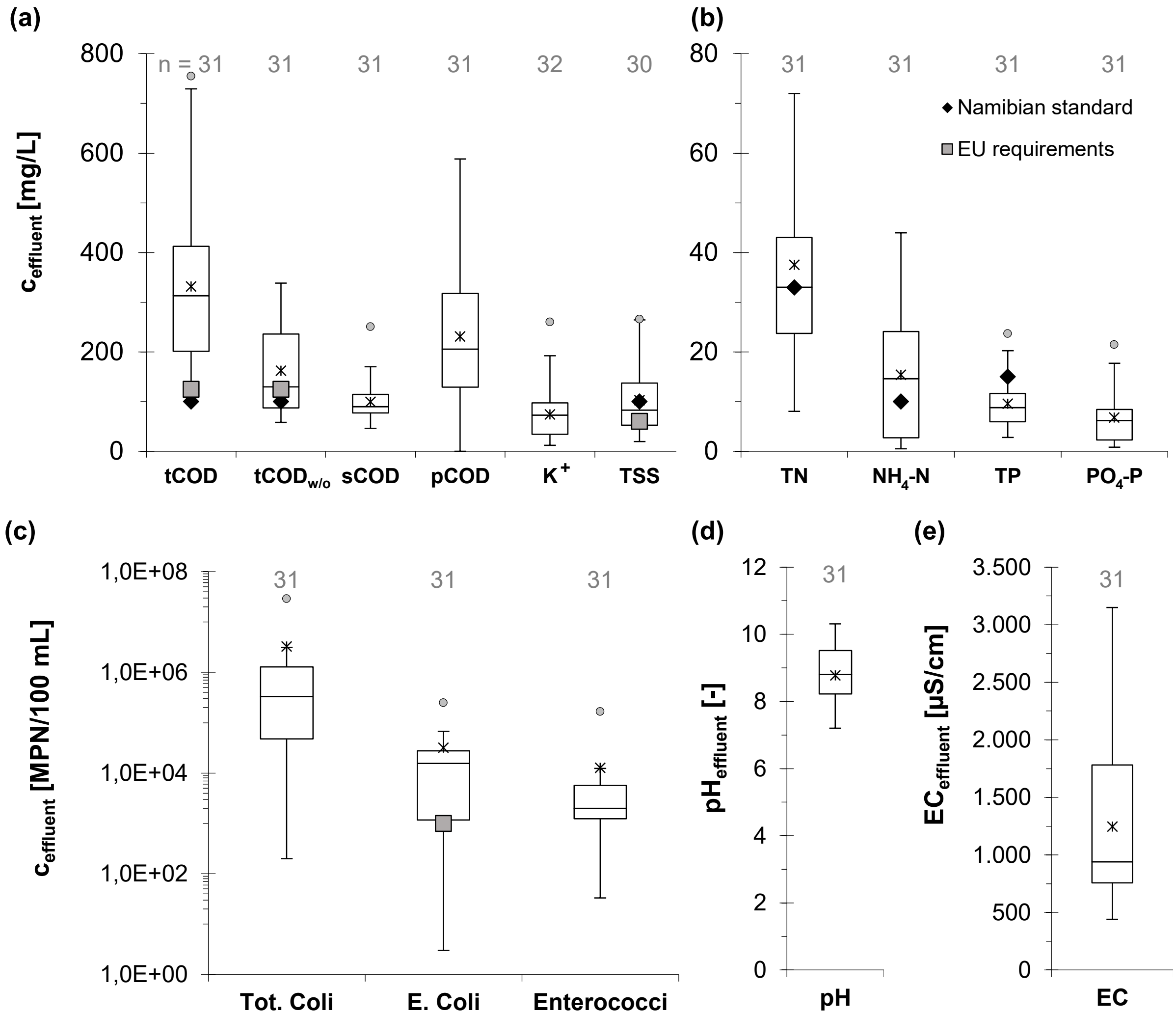

Figure 3.7: Effluent concentrations ($c_{effluent}$) of nine waste stabilization ponds (WSP): (a) total chemical oxygen demand (tCOD), soluble chemical oxygen demand (sCOD), particulate chemical oxygen demand (pCOD), potassium (K^+), total suspended solids (TSS), (b) total nitrogen (TN), ammonia (NH_4-N), total phosphorus (TP), phosphate (PO_4-P), (c) total coliforms (Tot. Coli), *Escherichia Coli* (*E. coli*), *Enterococci*, (d) pH and (e) electrical conductivity (EC).

The determination of faecal contamination through indicator bacteria is a common procedure, it may however not provide an accurate or complete picture of the pathogens present in a water sample. Therefore, a sequencing approach was also used to gain an overview over the diversity and dynamics of pathogenic genera without the bias of culturability. The composition and abundance of the genera consisting of pathogenic species, called "pathogenic genera", was thus analysed in the influent and effluent samples of six of the WSPs (Figure 3.8). Among the influent samples, the relative abundance of these pathogenic species ranged between 8 to 42 %, whereas for effluent samples it was between 1 to 20 %. In WSP G, the fraction of pathogenic genera was highest in both influent and effluent. Overall, we observed a reduction in the abundance of pathogenic genera in the effluent of all WSP. In total, 79 genera were found across all samples. Eighteen pathogenic genera were shared among the influent samples and four among the effluent samples (Figure 3.9). Although the cumulative abundance of low abundant pathogenic genera (i.e. having less than 0.1 % abundance) was highest in most samples, Acinetobacter was the most dominant across the influent samples (accounting for up to 16 % of the total microbial abundance) and Mycobacterium was dominant (accounting for up to 14 % of the total microbial abundance) across the effluent samples (Figure 3.8).

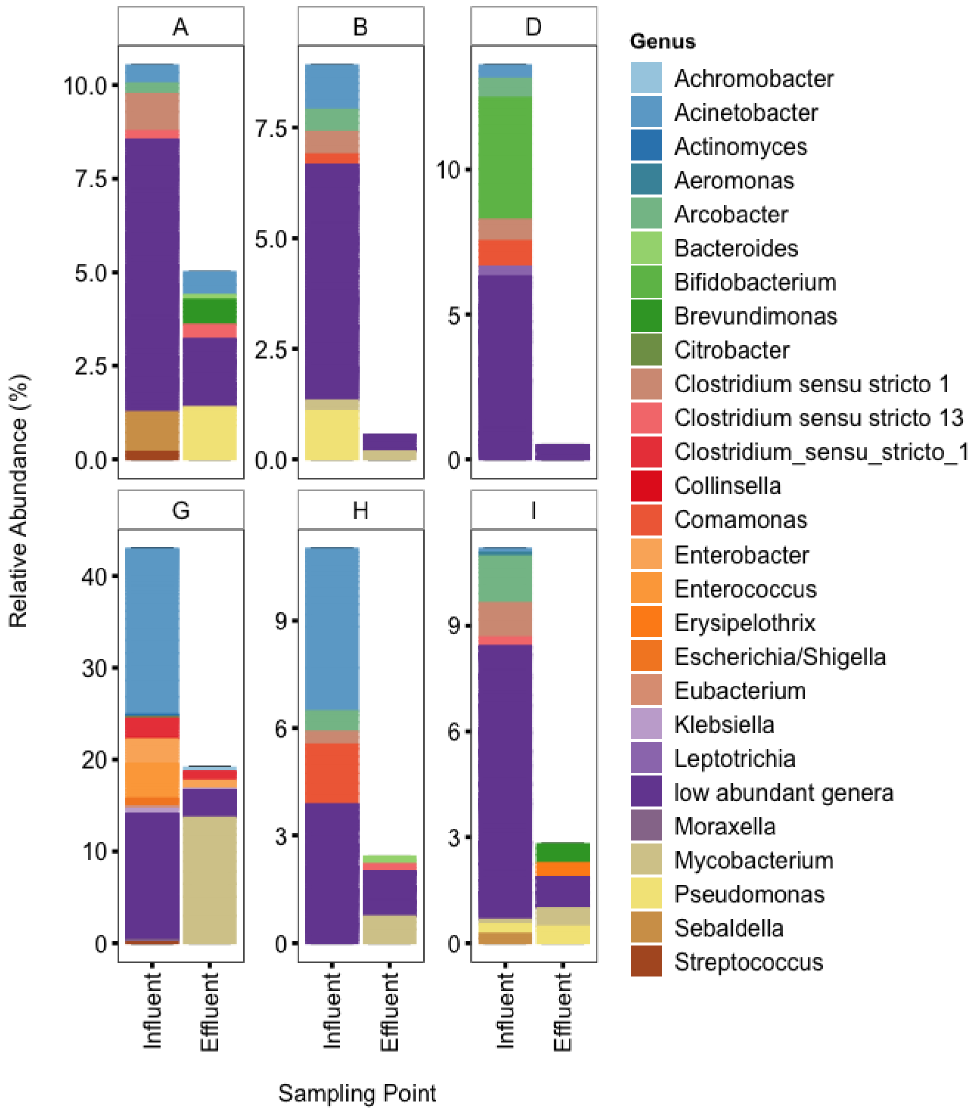

Figure 3.8: Bar plot showing the relative abundance of pathogenic genera found in each sample. Pathogenic genera having abundance < 0.1 % were categorized as “low abundant genera”.

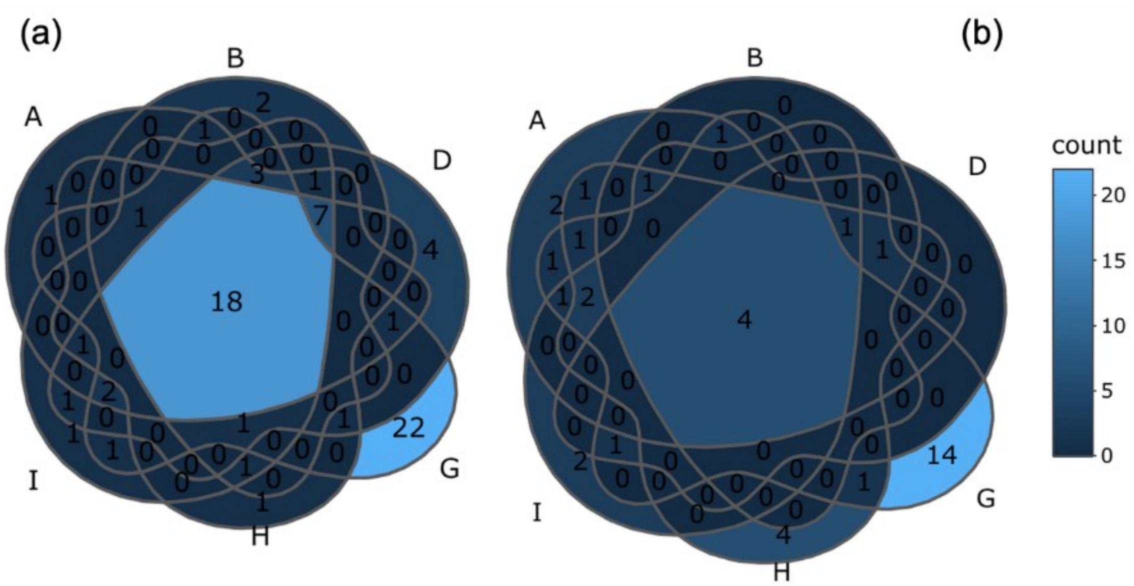

Figure 3.9: Venn diagram showing the number of pathogenic genera shared among (a) influent samples, (b) effluent samples of the WSPs.

Additionally to the pathogens, sequencing analysis can also provide information about the diversity of cyanobacteria. All WSP showed higher overall abundance of cyanobacteria in the effluent compared to the influent (Figure 3.10 a). But in the effluent their relative abundance (in relation to all bacteria) varied significantly from 0.45 (WSP I) to 25 % (WSP H). Figure 3.10 a shows the different composition of the biomass samples within the cyanobacteria at genus levels for the six WSP. Overall, *Synechococcus* was most dominant (i.e. up to 45 %), followed by *Cyanobium* (approx. 7 %) and *Chlorella* (approx. 5 %); others showed a relative abundance lower than 5 %. The identification of the respective genera is particularly important with regard to potential toxicity during the decomposition of these bacteria. A detailed heatmap with potentially toxic and non-toxic genera is presented in Figure 3.10 b.

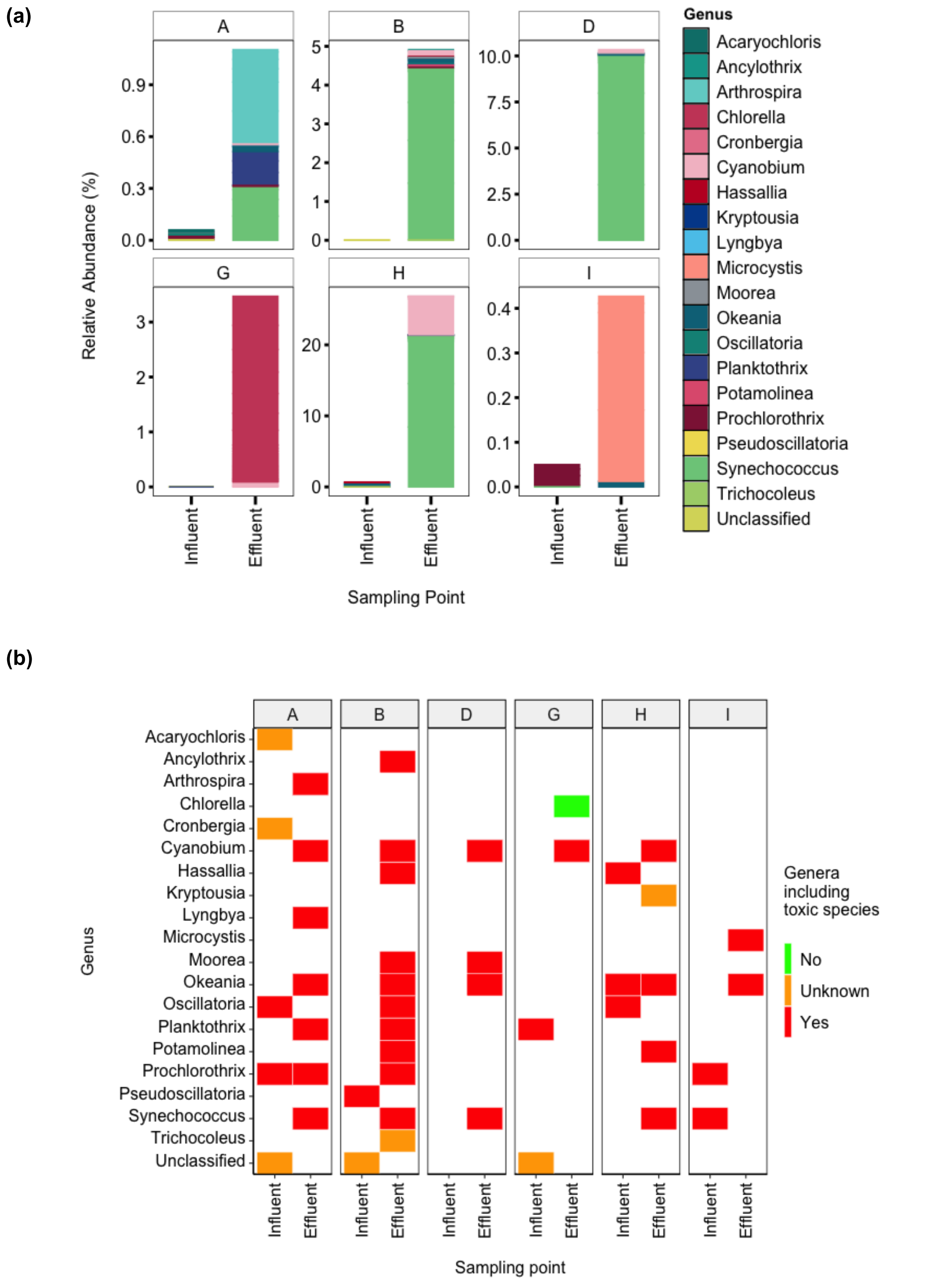

Figure 3.10: (a) Bar plot showing the relative abundance of the genera associated with cyanobacteria found in the waste stabilization pond systems (A, B, D, G, H and I). (b) Heatmap showing the composition of the cyanobacteria at genus level. The colour designates the difference between genera having toxic and non-toxic species.

3.4.6 Performance efficiency

In order to judge the performance of each WSP not only effluent concentrations needed to be considered but also removal efficiency. This is especially important as all WSP experienced high evaporation due to large surface areas. Three of the plants had water losses below 12 %, four between 26 to 49 % and two lost 100 % of their water (Figure 3.11), which is the original aim according to the code of practice in Namibia (DWAF, 2008).

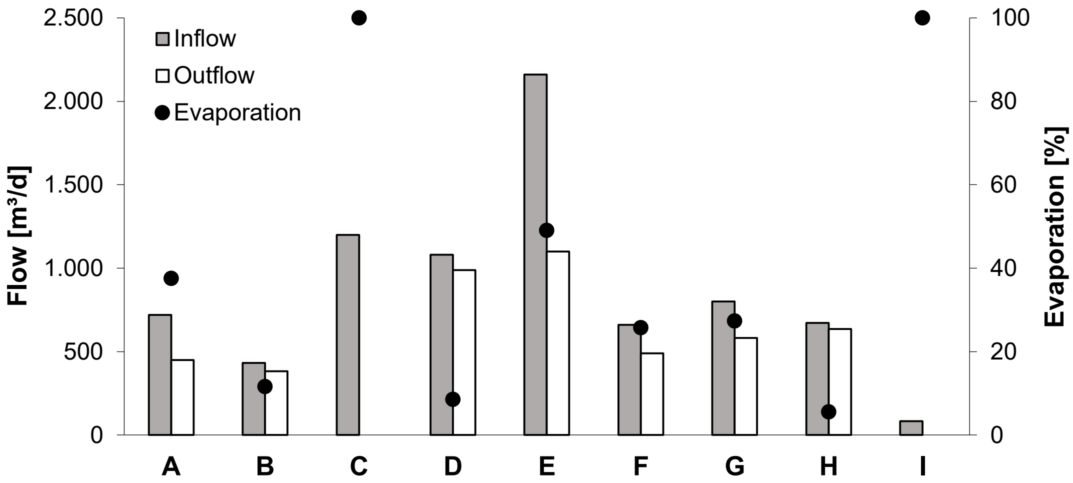

Figure 3.11: Inflow, outflow and evaporation of the nine waste stabilization ponds (WSP) A – I. Calculated with the average measured evaporation at WSP G and the corresponding surfaces given during interviews or estimated from aerial images.

The calculated load removal is presented in Table 3.1. The WSP are grouped according to their system setup. WSP D, G1 and H were only composed of FP and MP whilst WSP A, B, E and F were also designed with an AP upfront. WSP C and I evaporated all water and therefore there was no effluent load. Possible enhancements are reflected in WSP G2 with pre-treatment measures, FP and MP. WSP G3 had an additional post-treatment stage.

As mentioned before, none of the configurations reached the effluent requirements for tCOD when algae were included. The plants with the traditional design of FP, MP and some with AP were not able to reach the standards for TSS, tCOD, NH_4-N and *E. coli*. WSP with large surface areas and no effluent reached better values for TSS and $tCOD_{w/o}$, but still did not meet the required NH_4-N and *E. coli* values. With 100 % evaporation they present no viable solution as no water is available for irrigation. The WSP with enhancement measures did not show much improvement when only equipped with pre-treatment (WSP G2). In combination with post-treatment (WSP G3) the requirements were met for TSS, TP, TN and *E. coli*. For NH_4-N and $tCOD_{w/o}$ WSP G3 was just above the standards. Especially for the hygienic parameter *E. coli* it was the only plant setup with acceptable values.

3.4.7 Challenges for reuse of WSP effluent

All communities experience an increasing need of irrigation water for year-round fodder production due to changing rainfall patterns. Additionally, solutions are needed to accommodate for fast population growth, reuse of water as well as nutrients, production of crops and reduction of greenhouse gas emissions (Hernandez-Paniagua et al., 2014; Shelef and Azov, 2000). With small improvements WSP can reach a higher capacity and better effluent quality which is particularly essential for fast growing communities with the need of agricultural reuse and without receiving waters (Butler et al., 2017).

The results from all WSP showed considerable differences in the effluent quality (Table 3.1 and Table SI 8.1.2). In their original stages none of the WSP adhered fully to the required standards for water reuse. Either all water was evaporated or TSS, tCOD, nutrients and bacteria were not adequately removed. Algae also need consideration as they cause high tCOD. On the other hand, this biomass presents an important soil enhancer that improves the water-holding capacity of the soil (Mara, 2004). Other WSP for example in Brazil also reached average COD effluent values of 100 to 150 mg/L even with rock filters (von Sperling et al., 2007) and the water was still recommended for reuse. In the case of agricultural irrigation there was no need to remove algae in coarse filters (von Sperling and De Andrada, 2006), so the COD would be even higher. Juanicó and Milstein (2004) also found high COD effluents during a study in Israel but stated clearly that the organic matter formed by algae growth has no relationship with the solids originally present in the sewage. Also in South Europe countries are struggling to meet the water reuse regulation with constructed wetlands, especially for microbial parameters, and therefore, the reuse purpose has to be adopted to the water quality (Lavrnić and Mancini, 2016). A further option is a classification with crop restrictions and obligations for irrigation methods (EU, 2020) to ensure a multi-barrier approach.

3.5 Conclusions

According to the feedback of the WSP operators the main purpose of WSP in Namibia is seen in evaporation whilst the wastewater treatment itself is only secondary, especially because so far no reuse projects have been implemented. Therefore, operation and maintenance are mostly neglected. Overall WSP operation and management have to be revised focusing on reuse rather than on evaporation. Especially, as all towns struggle with fast growing populations and their WSP are overflowing into the surrounding environment with no perennial streams. As basis for further in-depth studies on WSP this research presents a first systematic evaluation of nine WSP in North Namibia:

- Pond surface areas vary considerably between 1 to 26 m^2/cap and HRT from 12 and 302 days depending on the design, date of construction, current population and surface extensions. Due to a multitude of local influences the effluent values do not correlate with the surface area or HRT.
- PE loads in the raw sewerage are within the literature range and therefore inflow estimations between 83 and 2,160 m^3/d are reasonable.
- Removal efficiencies of up to 85 % of tCOD, 64 % TP and 87 % of TN are reached by single WSP.

- The EU requirement of less than 1,000 *E. coli* per 100 mL for the irrigation of fodder crops is only reached with enhancement measures.
- Acinetobacter is the most dominant pathogen across the influent samples (accounting for up to 16 % of the total microbial abundance) and Mycobacterium is dominant (accounting for up to 14 % of the total microbial abundance) across the effluent samples.
- Relative abundance of cyanobacteria (in relation to all bacteria) varied significantly from 0.45 to 25 %. Overall, *Synechococcus* dominated (i.e. up to 45 %), followed by *Cyanobium* (approx. 7 %) and *Chlorella* (approx. 5 %).
- Currently none of the plants fulfils the national reuse standard in their original design and the effluents are not fit for reuse purposes yet, especially due to tCOD concentrations above 100 mg/L.
- Chlorophyll-*a* concentrations provide a first indication of the algal fraction in the pCOD, as they correlate linearly with the pCOD.
- The reuse standards need to be adapted specifically for WSP, either to allow for higher tCOD concentrations caused by algae or to use the sCOD as indicator.

4 Enhancement of overloaded waste stabilization ponds using different pre-treatment technologies: a comparative study from Namibia

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4.1 Abstract

Waste stabilization ponds (WSP) are a well-established wastewater treatment technology in Namibia. But they are often overloaded and we still lack concepts and technologies for improvement. Therefore, this study presents the full scale implementation of two pre-treatment technologies to reduce the inflow of organic and solid loads into a facultative pond. We specifically compared the effects of anaerobic biological and mechanical pre-treatment by an upstream anaerobic sludge blanket (UASB) reactor and a 250 μm micro sieve. Not only in Namibia but also in most sub-Saharan countries there is little experience with these technologies for the treatment of municipal wastewater in small and fast growing local communities. Both technologies were tested in parallel for a period of 17 months and proved operational. Whilst the UASB achieved better removal results with respect to chemical oxygen demand (COD) and suspended solids (TSS), the micro sieve was more flexible in handling changing inflow patterns and had a much smaller footprint. The average total COD reductions of the micro sieve and the UASB were 22 % and 50 %, respectively. TSS were removed by 45 % with the micro sieve and by 57 % with the UASB reactor. Therefore, UASB and micro sieve are viable options for the enhancement of existing WSP to reach better effluent values of the facultative pond.

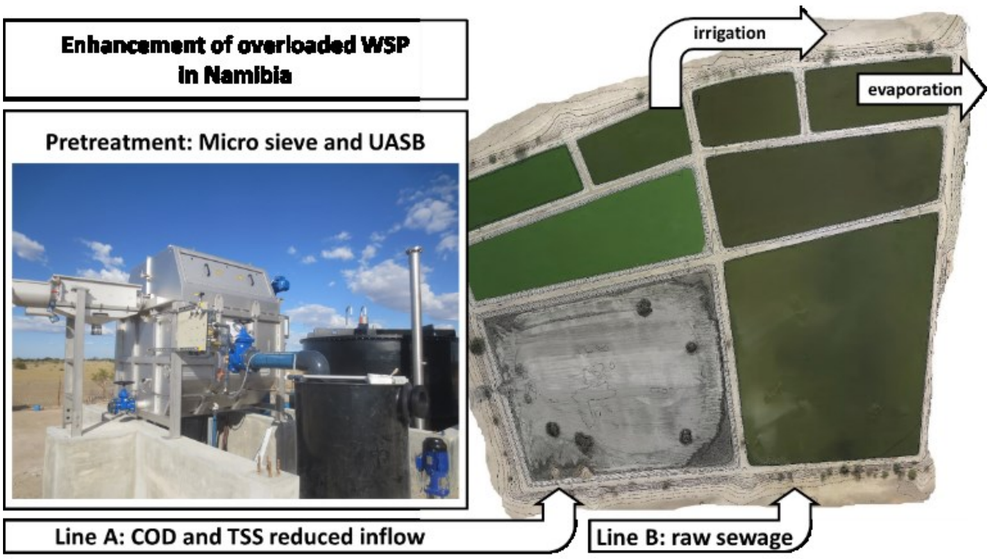

Figure 4.1: Graphical abstract: Enhancement of overloaded waste stabilization ponds using different pre-treatment technologies

4.2 Introduction

Water scarcity has become an increasing struggle in many regions worldwide. Recent variable weather conditions in sub-Saharan Africa show direct effects of climate change. Most affected by variable rainfall or lack of any precipitation are farmers who need the precipitation for their crops. Therefore, water reuse of treated wastewater is becoming an ever more important mitigation measure. Mara (2009) already postulated the increasing need for reuse of treated municipal wastewater, from e.g. waste stabilization ponds (WSP), in water scarce areas. However, in many towns with sewer systems the only wastewater treatment systems, if any, are WSP. They require low maintenance and power requirements, but are often too small and overloaded due to the very high population growth.

With the combination of extended drought periods due to climate change and increasing water demands, the focus is shifting more and more towards water reuse, e.g. for irrigation. Thereby, WSP face new challenges. They have to fulfil national or international water quality standards, especially for hygienic parameters but also for COD removal. Not only the increasing population and the related need for more irrigation water with valuable nutrients, but also the reduction of greenhouse gases impose a growing burden on WSP (Hernandez-Paniagua et al., 2014; Shelef and Azov, 2000). A typical strategy for capacity enhancement is volume extension (number and or size of the ponds) which is accompanied by growing land requirements. Other options may present themselves through upgrades of existing WSP with more advanced treatment technologies, such as anaerobic biological or mechanical treatment units.

This study compares an upstream anaerobic sludge blanket (UASB) reactor to a micro sieve (MS) as potential pre-treatment technologies in order to reduce the load of organic carbon (measured as chemical oxygen demand, COD) and suspended solids entering an existing overloaded WSP in Northern Namibia. The potential of these technologies to reduce COD and solids has been reported by Lazarova and Bahri (2005) with a wide range of values of 40 - 70 % and 20 - 75 % for UASB and MS, respectively. Also, the effect of such measures on the performance of consecutive ponds has not been monitored in depth in any sub-Saharan country.

Due to high COD and TS loads into primary facultative ponds sludge accumulation is also an issue and reduces the treatment capacity. A typical WSP setup would include an anaerobic pond to reduce COD and TSS loads into the facultative pond (Shilton, 2005; von Sperling, 2007a). However, this would not reduce potential methane emissions and would not solve the problem of the quickly accumulating sludge. In order to address this, UASB reactors have already been installed as anaerobic biological pre-treatment of municipal wastewater in front of WSP in other countries (e.g. in Brazil or India) with similar climate conditions (Bressani-Ribeiro et al., 2019; Cavalcanti and van Haandel, 1996; Dias et al., 2017b; Khan et al., 2011; Vassalle et al., 2020). Only Müller (2017) already examined UASB reactors before rotating biological contactors (RBC) in Namibia.

At the same time mechanical pre-treatment with micro sieves is becoming increasingly more important for energy recovery (Hey et al., 2016; Jahn et al., 2017; Paulsrud et al., 2014; Prösl et al., 2013; Rusten and Ødegaard, 2006; Walder et al., 2015). In Europe, Jahn et al. (2017) and Walder et al. (2015) researched a MS within the Austrian context. In Namibia micro sieves have so far been implemented as pre-treatment for an industrial wastewater treatment plant followed by membrane bio-reactors (Prösl et al., 2013) and as post-treatment for a municipal wastewater treatment plant (Müller, 2017). A MS for the pre-treatment before a WSP, however, has been implemented for the first time in this research.

This is the first study which compares an UASB reactor (biological pre-treatment) with a MS (mechanical pre-treatment) to relieve overloaded WSP, with Namibia as an example. The results are especially important for fast growing communities in warm climates with the need of water reuse for irrigation and for regions without perennial streams (Butler et al., 2017).

4.3 Material and Methods

4.3.1 Waste stabilization ponds (WSP) and pre-treatment technologies

This project was conducted at an existing WSP system in a regional capital in Northern Namibia. The original WSP system as planned in 2004, consisted of two parallel trains (train A and train B) with four ponds each (Figure 4.2): one primary facultative pond (1) and three maturation ponds (ponds 2 - 4) (Figure 4.3). According to the design, the feed rhythm alternated between the two trains. The two facultative ponds have the largest volume of 16,000 m^3 each and a surface of 11,000 m^2. Both are followed by three maturation ponds with smaller surfaces and volumes. The total water surface of the WSP is 40,500 m^2 with a total volume of 53,000 m^3. The water level decreases from 1.5 m in the first pond to 1.1 m in the last pond (Table SI 8.2.1). Due to a high ground water table and a nearby ephemeral stream the earth dams constructed above the natural surface are covered with concrete whilst the ground of the ponds is lined with clay.

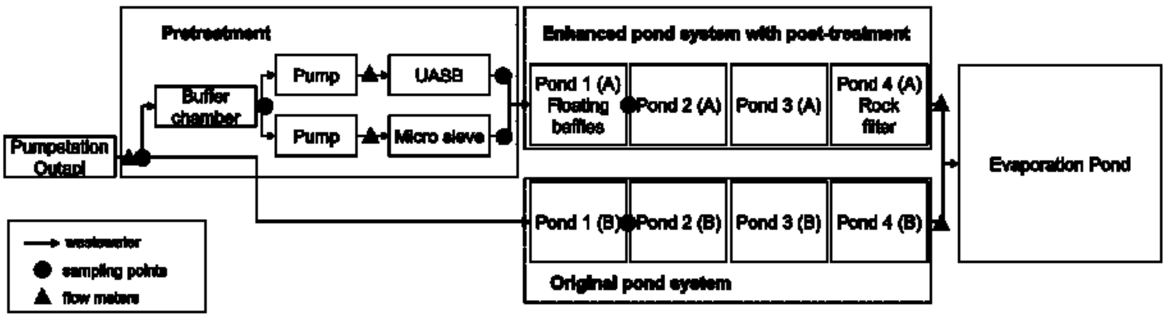

Figure 4.2: Schematic drawing of the water reuse project, the black dots represent sampling points

Two pre-treatment technologies were tested: a MS and a UASB reactor. Upfront, a coarse screen with 0.01 m bar spacing followed by a 56 m^3 open collection chamber to buffer peak inflows was installed (Figure 4.2). The pre-treatment technologies for the upgrade were installed in train A. The UASB reactor had a volume of 42 m^3 and was continuously fed with 3 - 4 m^3/h inflow of raw wastewater, resulting in a hydraulic retention time of 10 – 14 h. In parallel, a MS (Noggerath Rotary Drum Screen RSJ-MG®) with a 250 μm monodur polyamide mesh was installed. Its drum diameter was 1.6 m, with a length of 1.5 m, resulting in a total area of 7.5 m^2. Aside from the standard monitoring, a two-week intensive testing phase was carried out from day 265 to 275. During this period, different flows and operational settings were applied. These included an increasing inflow from 28 m^3/h to 60 m^3/h (maximum capacity of the feed pump: 75 m^3/h), increase of the impounding depth from 10 to 18 cm and chemical cleaning with a 2 % potassium hydroxide solution at 60 °C.

Additionally to the pre-treatment, pond A1 was renewed by emptying out the settled sludge and installing two floating baffles made of 2 mm thick HDPE sheets down to the full depth over 2/3 of its length. This improvement now avoids short circuiting and enables better plug flow conditions. The overall project also includes the installation of a rock filter in pond A4 as post-treatment. The focus of this work was, however, on the pre-treatment technologies only. These were continuously monitored for 17 months. Due to pump failure after heavy rainfalls with inundations and power failures the MS had an 88 days standstill time from day 110 to day 198.

4.3.2 Sampling and analyses

Electromagnetic flowmeters (IDM) measured the inflow to the plant, the inflow to the UASB reactor and the MS, as well as the outflow of ponds A4 and B4. Wastewater quality was monitored with portable probes for pH, dissolved oxygen and conductivity using a WTW multimeter 3410 (Xylem Analytics Germany) at different sites: untreated wastewater, the effluents of UASB, MS, A1 and B1 (Figure 4.2).

Due to high retention times in the sewer system with 15 pump sumps and in the ponds, grab samples were taken at different times of the day. For the untreated wastewater also 2 h mixed samples were analysed for comparison. The samples were analysed regularly with Hach Lange LCK cuvette tests for COD, total nitrogen (TN) ammonium, nitrite, nitrate, total phosphorus (TP) and phosphate with a spectrophotometer DR 2800 (Hach Lange GmbH, Germany). Indicator bacteria were examined with IDEXX Colilert-18 (enumeration of total coliforms and *Escherichia coli*) and Enterolert (*Enterococci*) employing Quanti-Tray/2000 (IDEXX Germany). To obtain total solids (TS) and total suspended solids (TSS) the samples were dried at 105 °C and for TSS also filtered with glass microfiber filters (Whatman 934-AH). Total volatile solids (TVS) and volatile suspended solids (VSS) were measured after ignition at 550 °C. The UASB reactor had five outlets to determine the level of the sludge bed.

4.4 Results and Discussion

4.4.1 Site development in Outapi

Originally about 2,500 of 3,000 inhabitants were connected to the sewer system, and the pond system treated their wastewater. By 2018 the connected population had almost tripled up to nearly 7,000 people. However, due to the high population growth rate of 9.3 % per annum, the total population in 2018 was estimated at 12,000 (Mwinga et al., 2018). This resulted in only 58 % of the total population discharging their wastewater to the sewer and wastewater treatment system, whilst 69 % were connected to the town's water supply system (Mwinga et al., 2018).

The WSP were designed with no overflow to the surrounding ephemeral water course so that all water was supposed to evaporate. Already at the early stage of operation, an additional evaporation pond with a surface area of 41,000 m^2 and a volume of 20,500 m^3 (Figure 4.3 and Table SI 8.2.1) was built with simple earth dams. But due to higher flow rates, especially during the rainy season, the evaporation pond was overflowing regularly, posing a potential health risk to humans and grazing animals.

Figure 4.3: Aerial photo of the pond system with two trains (pond A1 – pond A4 and pond B1 – B4) and evaporation pond (Google-Earth (2019), modified)

Within this study, enhancement measures at the existing WSP with simple pre- and post-treatment technologies were introduced and tested in order to improve the effluent quality and thereby ensure safe water reuse for irrigation of fodder crops. For comparison, one train (train B) kept its original status of 2004 whilst the other train (A) was enhanced.

4.4.2 Inflow characteristics

In this study, wastewater characteristics were monitored extensively for a long period of time. Such detailed information has so far not been available for the sub-Saharan context. The collected data included water quality parameter as well as flow patterns.

The mean total inflow to the WSP was 802 (± 177) m^3/d during the observation period, with a peak inflow of 1,330 m^3/d on day 267 (Figure 4.4). The maximum inflow was reached after rainfall events due to surface water entering the sewer system. Compared to preceding years, there was hardly any rain in the region during summer. This is also an indicator for changing rain patterns. In comparison, previous years recorded the highest hydraulic inflow to the plant with almost 2,000 m^3/d after heavy rainfalls.

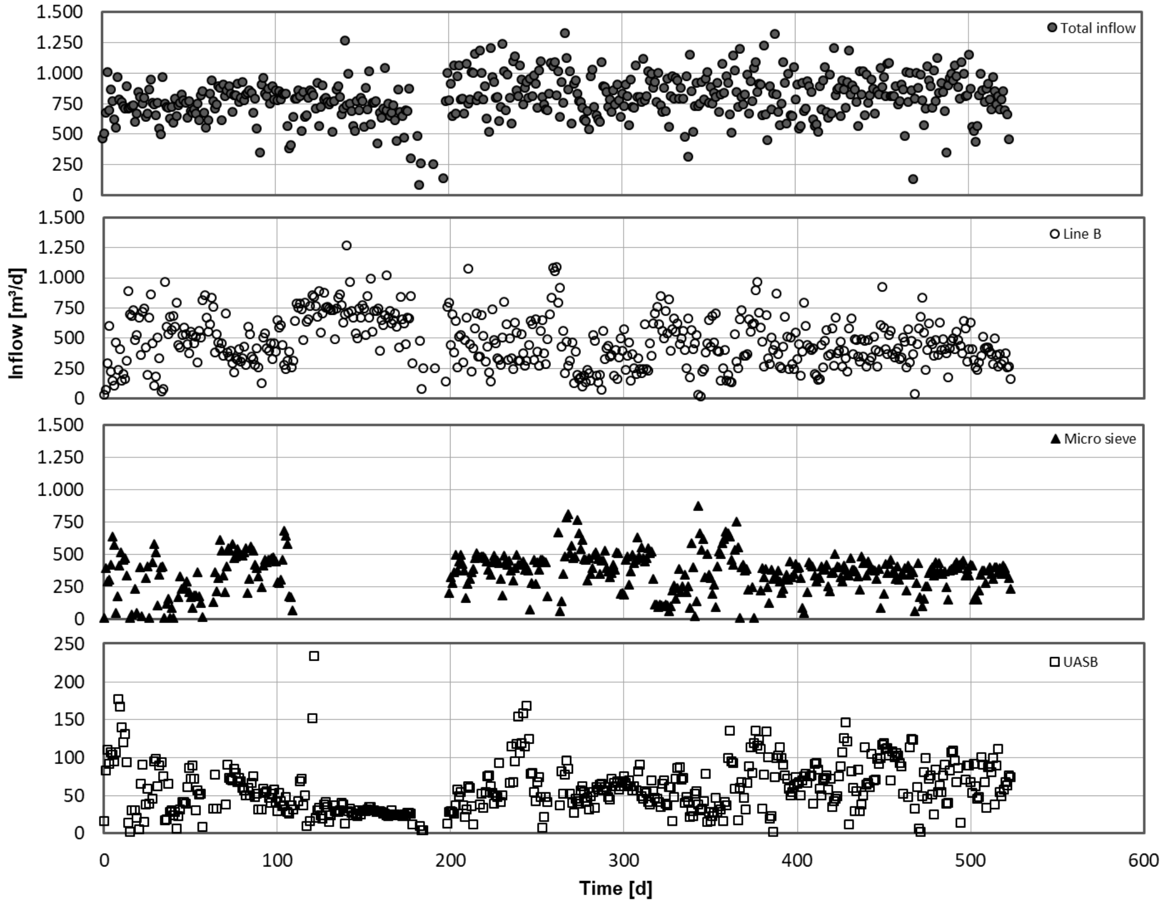

Figure 4.4: Flow rates of the total inflow to the pond system, inflow to unimproved train B and to the pre-treatment technologies micro sieve and UASB

Over the past years the mean daily inflow has increased from 710 m^3/d in 2016 to 811 m^3/d in 2018. This 14 % increase over two years very well reflects the growth of the town and more connections to the sewer system. The peak inflow to the WSP was in the morning between 6 am and 11 am. Three typical inflow patterns are presented in Figure 4.5. After the peak in the morning there was a second increase towards the early evening followed by reducing values during the night, showing rather typical daily variations in the inflow.

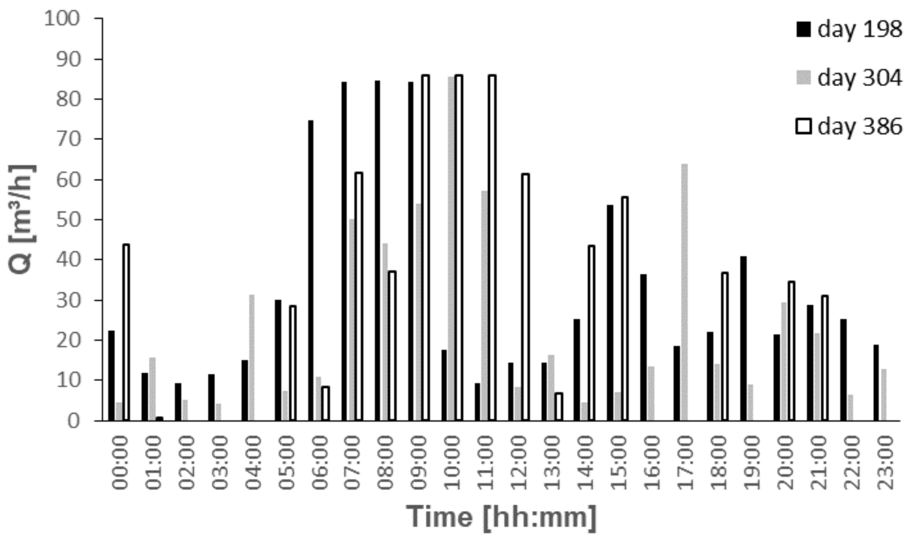

Figure 4.5: Typical daily inflow to WSP at three weekdays Wednesday (day 198), Thursday (day 304) and Tuesday (day 386)

The inflow was divided equally between both trains for the purpose of good comparison between the enhanced train A and the train at its original state (train B). Train A received a mean inflow of 345 (± 191) m^3/d pre-treated wastewater, the inflow to train B without pre-treatment was 472 (± 211) m^3/d (Figure 4.6).

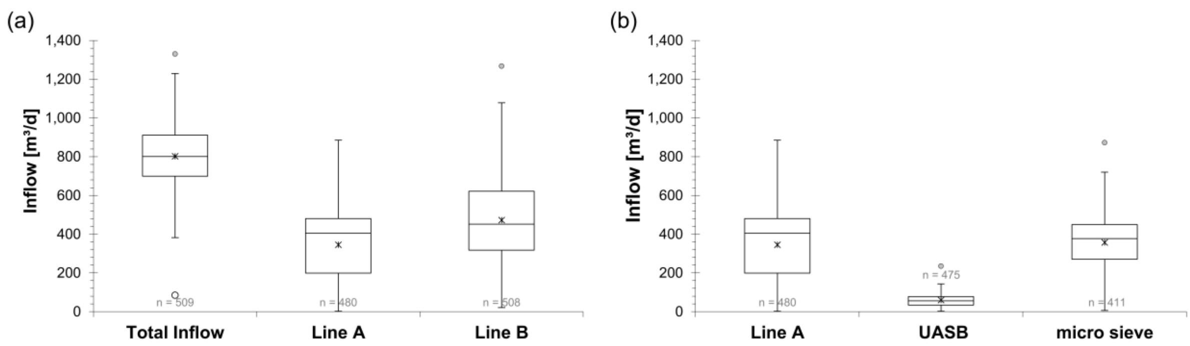

Figure 4.6: Daily flow rates (total inflow, train A and B, UASB, micro sieve) from day 1 - day 527

The water quality of the raw wastewater is included in Table SI 8.2.2. Mean loads and concentrations of 606 (± 149) kg/d and 749 (± 153) mg/L for tCOD, 61.7 (± 15.4) kg/d and 76.0 (± 12.5) mg/L for TN, 8.0 (± 1.7) kg/d and 9.8 (± 1.3) mg/L for TP, and 174 (± 67) kg/d and 220 (± 81) mg/L for TSS were observed. These values are within a common range for municipal wastewater (Tchobanoglous et al., 2014).

4.4.3 Pre-treatment technologies

Two different pre-treatment technologies, a biological system relying on anaerobic degradation processes (UASB) and a mechanical treatment system (MS) were monitored in parallel to evaluate their potential for reducing the COD and solids load into the WSP. Besides the settling of the pCOD and the TSS in the UASB, anaerobic microorganisms also reduced sCOD, nutrients and pathogens. In comparison, the MS used a solely mechanical sieving process to reduce pCOD and TSS. The continuous operation of the UASB required a larger buffer volume to compensate for peak inflows whilst the MS was very flexible with changing flow patterns.

Over the course of the research period the MS received on average 357 (± 152) m^3/d and the UASB 60 (± 33) m^3/d. Changing treated volumes depended on the total inflow fluctuation and peak times as well as on the pump capacities. Peak inflows were 235 m^3/d for the UASB and 873 m^3/d for the MS.

4.4.4 Micro sieve

The different operation phases of the micro sieve are indicated in Figure 4.7. The commissioning (day 1 – day 43) with instable COD effluent values was followed by a stable operation phase. Effluent concentrations of the MS from day 77 to day 109 were: tCOD 676 (± 24) mg/L, pCOD 370 (± 26) mg/L and sCOD 304 (± 14) mg/L. After the standstill (day 110 – day 199), operation was less stable but improved towards the end of the study period with concentrations for tCOD of 708 (± 132) mg/L, pCOD = 404 (± 133) mg/L and sCOD = 293 (± 68) mg/L. Over the whole study period (day 1 – day 527) the mean sCOD concentrations with 306 (± 58) mg/L were more constant than the tCOD concentrations with 740 (± 142) mg/L and pCOD = 431 (± 128) mg/L.

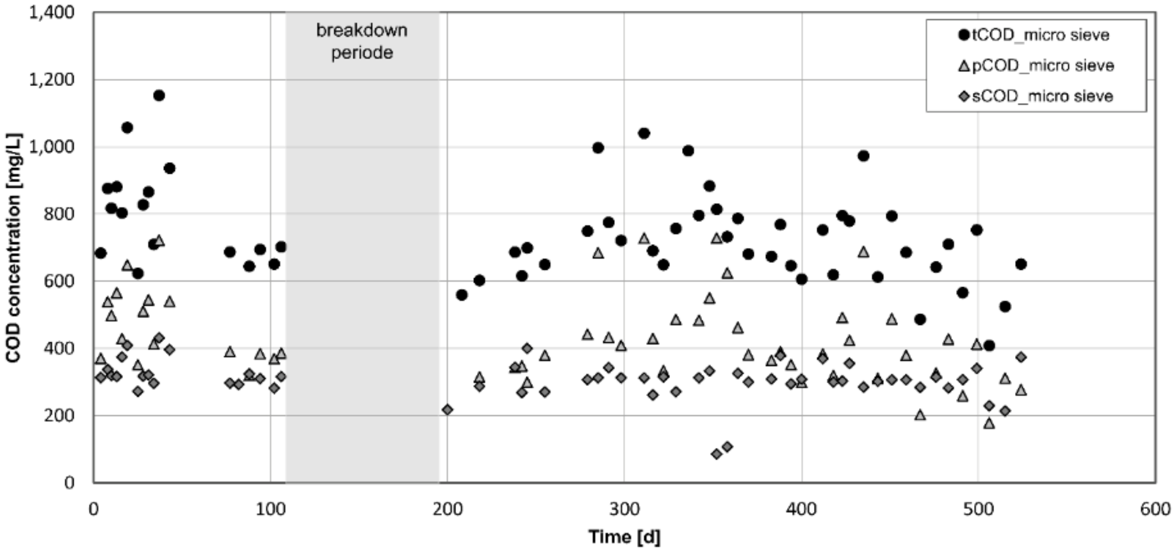

Figure 4.7: COD effluent concentrations of the micro sieve (tCOD: total COD, pCOD: particulate COD, sCOD: soluble COD). The grey area indicates the breakdown period.

Over the whole study period the tCOD removal was 22 (± 18) % and pCOD removal was 28 (± 24) %. Therefore, the installed MS reached good percentage removal which were within the range of others, e.g. the 15 - 25 % tCOD removal reported by Lazarova and Bahri (2005) or the 25 % removal reported by Prösl et al. (2013). Elimination efficiencies of up to 40 % tCOD seem only reachable with flocculation (Jahn et al., 2017; Walder et al., 2015). Peak tCOD and pCOD removal reached up to 57 % and 72 % respectively. At the same time the sCOD was hardly reduced as expected.

On the contrary, TSS removal reached 45 (± 27) % over the whole study period. The removal of VSS was in the same range with 48 (± 22) %. These values are better than at an industrial wastewater treatment plant in Windhoek. There, the two MS for the mechanical pre-treatment also operate with 250 μm sieves and remove only 30 % of TSS (Prösl et al., 2013). Sieves in Austria reached a removal of 65 % with flocculation (Jahn et al., 2017; Walder et al., 2015). Our MS reached similar levels as the ones in Austria only during three days for its peak performance, which showed a maximum removal of 80 % (TSS) and 82 % (VSS), respectively. Therefore, the original design value of the manufacturer with an average of 60 % TSS removal can only be reached with flocculation.

The intensive two-week testing phase revealed, that with the increasing inflow the spray time over the cycle time in-between two cleaning events increased linearly (Figure 4.8 a). Hence, the maximum theoretical inflow of the tested MS would be about 100 m^3/h. This, however, would require a constant spray operation which is not practical. Therefore, a maximum spray time of 80 % of the cycle time was chosen which yielded a maximum inflow of 83 m^3/h. This compares very well with the maximum design value of 82 m^3/h. Based on the total sieve surface of 7.5 m^2 the maximum hydraulic loading rate of 11 $m^3/(m^2 \cdot h)$ can be reached. This value is within the same range of 6 - 12 $m^3/(m^2 \cdot h)$ as observed in Austria (Jahn et al., 2017).

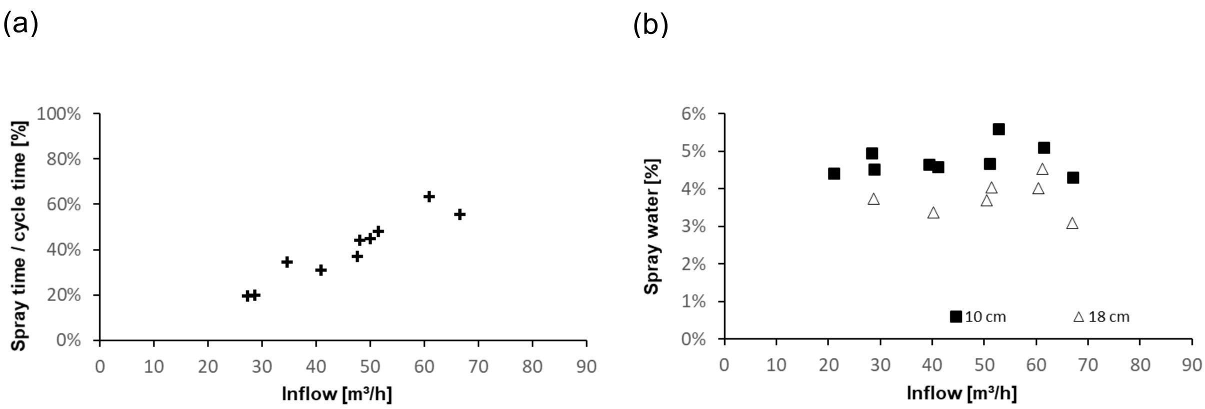

Figure 4.8: Micro sieve spray time (a) in % of the cycle time and spray water consumption (b) in % of the treated water

Another important aspect of the MS was the spray water consumption in relation to the treated wastewater flow. For a sustainable operation process, water from the MS effluent was used to spray the sieve itself instead of using valuable, high quality tap water. Nevertheless, spray water use should be as low as possible to reduce energy consumption for the process water pump. This study shows that the percentage of spray water did not depend on the inflow but on the impounding depth. With an impounding depth of 10 cm, there was an average need of just below 5 % whilst with a depth of 18 cm less than 4 % were needed (Figure 4.8 b). This is almost double the 2 - 3 % measured by Walder et al. (2015) in Austria who have cleaned their sieve with water and a combined air injector.

4.4.5 UASB reactor

For the first two months after commissioning of the pre-treatment stage there was only a small COD reduction in the UASB reactor. Total COD was reduced by only 18 % from 838 mg/L to 685 mg/L whilst the sCOD remained constant at around 400 mg/L. After the inoculation of the UASB reactor with anaerobic sludge from the bottom of pond B1 on day 65, the tCOD concentration reduced over three months down to 234 mg/L and sCOD to 102 mg/L (Figure 4.9). This improvement of the performance was partially attributed to this inoculation but also to a rise in water temperatures. The temperature increased from 23.5 °C in winter (day 1 – day 30) up to 28.5 °C in summer (day 75 – day 145). The temperature influence was also evident during later periods (day 200 – day 527). During the cold season the highest tCOD effluent was measured with 720 mg/L (day 370). But this was also influenced by the accumulating sludge in the UASB. With regular sludge removal every 4 to 6 weeks from day 380 onwards, the tCOD effluent improved and reduced back down to 274 mg/L. Whilst the tCOD was influenced by temperature and sludge accumulation, the sCOD remained constant with an average concentration of 108 mg/L after the inoculation.

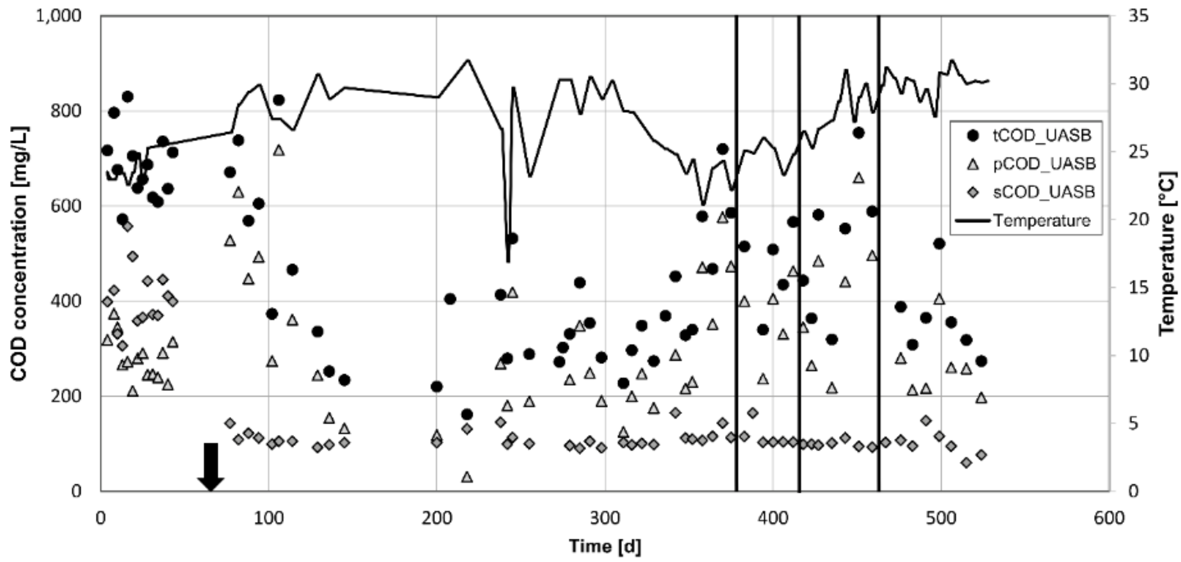

Figure 4.9: COD effluent concentrations of the UASB reactor (tCOD: total COD, pCOD: particulate COD, sCOD: soluble COD) and effluent temperature. The arrow indicates the inoculation (day 65), the black lines represent the sludge removal

The removal of tCOD was 50 (± 21) % during the whole study period. The peak removal was reached on day 311 with 84 %. This is lower than the measured average values in Brazil of 63 % (Dias et al., 2017b), but close to the 57 % of Vassalle et al. (2020). At the same time the mean sCOD removal was 54 (± 28) %, pCOD was removed by 45 (± 29) %. Peak removals were 80 % for sCOD on day 524 and 89 % for pCOD (day 311).

After the inoculation of the UASB reactor with the anaerobic sludge from pond B1 on day 65, the solids in the UASB settled constantly over the following three months. In the beginning, the TS concentration at the top level reduced from 11 g/L to 1.0 g/L and TVS from 7 g/L down to 0.4 g/L. At the same time, the TS concentration at the bottom level increased from 43 g/L to 62 g/L, the TVS concentration from 30 g/L to 38 g/L. After a continuous feeding period of one year the first sludge was removed from the bottom layer (Figure 4.10). During the following months the sludge bed was monitored and when it reached a height of 2.5 m in the reactor, between the second and third sampling level, about 5 m^3 were removed. This was three times during the study period. The mean TS concentration at the bottom level was 58 g/L with a TVS content of 66 %, whilst the top level had an average TS concentration of 3.8 g/L with a TVS content of also 66 % (Figure 4.11).

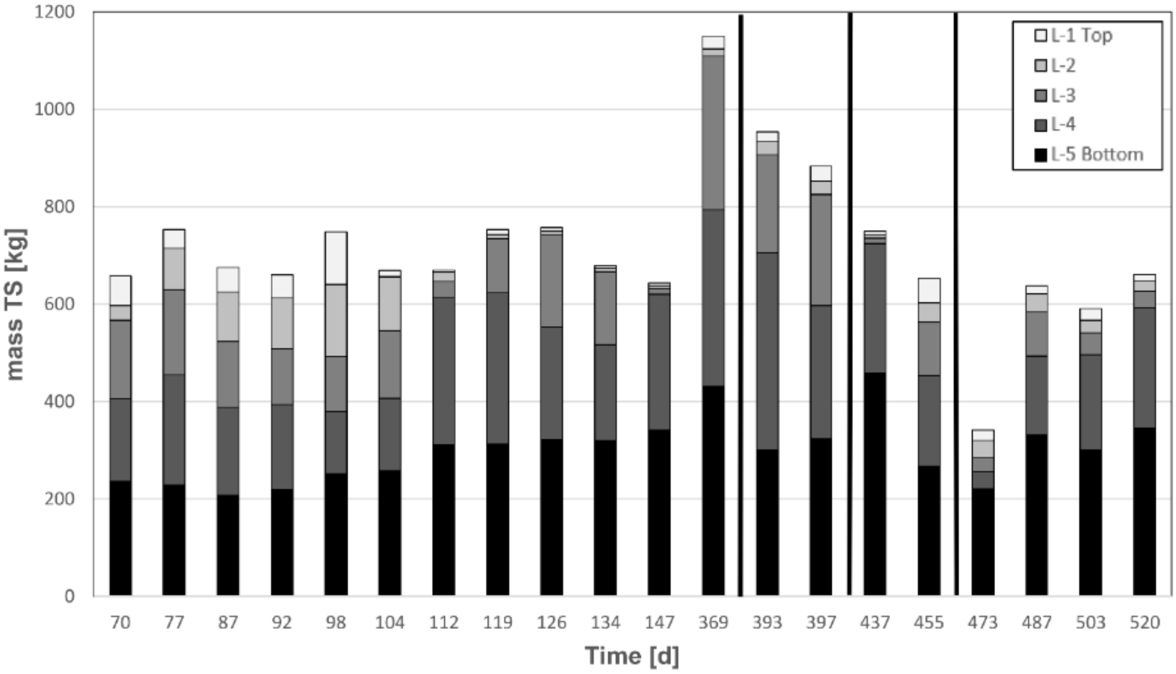

Figure 4.10: Mass of total solids (TS) in the UASB in each layer L-1 Top, L-2, L-3, L-4, L-5 Bottom with the black lines representing sludge removal

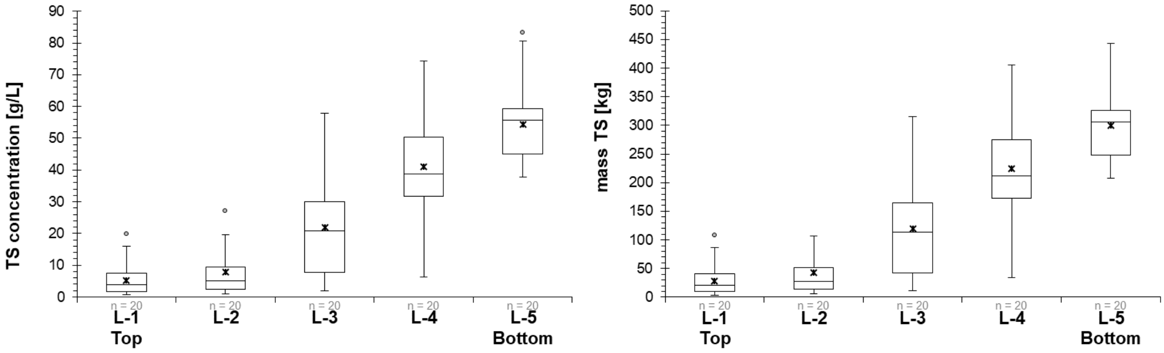

Figure 4.11: Total solids (TS) concentration and mass of different sludge levels in the UASB reactor

4.4.6 Water quality comparison

Both, UASB reactor and MS, are valuable pre-treatment technologies that have been implemented in different contexts. Within the local context of this study, the direct comparison of the effluent values showed that the percentage removal of the UASB were often higher than the ones of the MS. With regards to the tCOD removal, the UASB mostly came close or above 50 % whilst the MS seldom achieved over 50 % (Figure 4.12 a). This was mainly a result of the sCOD reduction in the UASB that did not exist in the MS. For the pCOD removal the MS delivered more values above 50 % but never above 75 % like the UASB did (Figure 4.12 b). The average tCOD reduction of the MS was 22 % versus 50 % of the UASB. TSS were removed by 45 % with the MS and by 57 % with the UASB reactor.

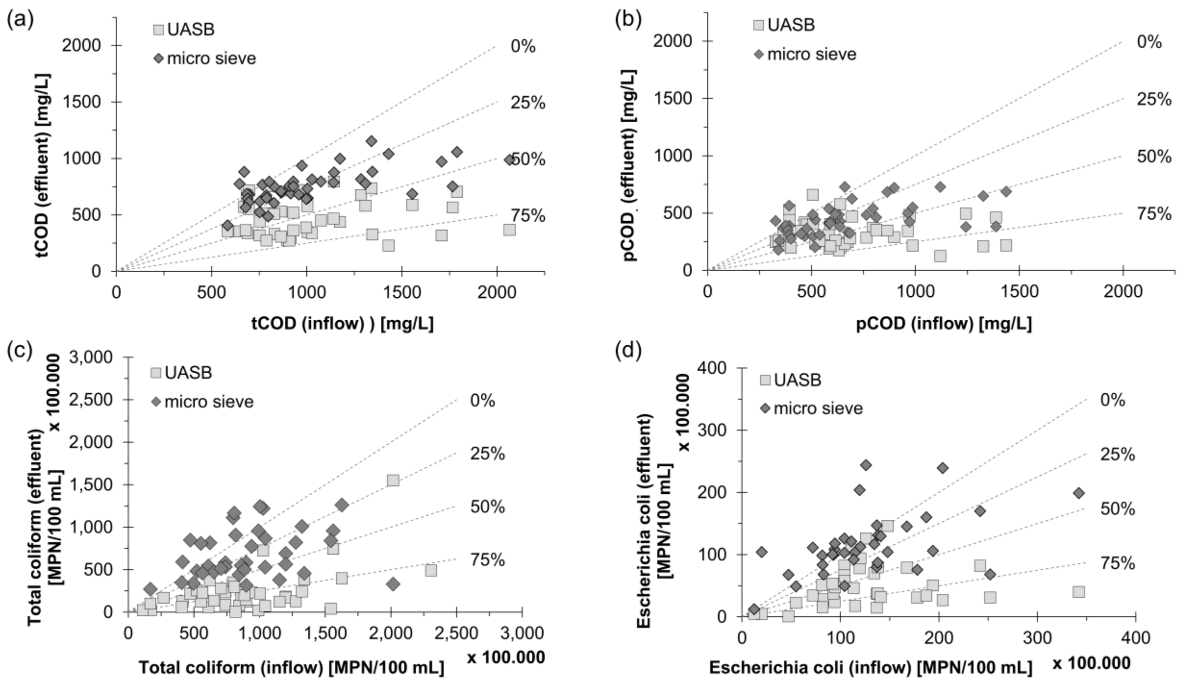

Figure 4.12: Comparison of the removal percentages of micro sieve and UASB reactor for tCOD (a), pCOD (b), total coliforms (c) and *E. coli* (d).

Even though the main purpose of both technologies was to reduce COD and TSS loads, they also had a visible effect on the microbiology. Whilst the MS removed up to 50 % of total coliforms and *Escherichia coli*, the UASB reached even better removal efficiencies of 50 % to 75 % for *E. coli* and even above 75 % for total coliforms (Figure 4.12 c and Figure 4.12 d). Negative percentage removal in all cases can be explained by washing out of biofilm that accumulated on the walls of the UASB and the MS. The UASB was less affected by detaching biofilm as this biomass generally settles with the pCOD and is removed via sludge withdrawal. For the MS detached biofilm was easily washed out with the effluent, and was therefore more often detected.

On average, the UASB reactor reached tCOD effluent concentrations of 477 (± 174) mg/L compared to 740 (± 142) mg/L by the MS (Figure 4.13 a). This was mainly due to the additional sCOD removal of the UASB. Soluble COD effluent values for UASB and MS were 171 (± 126) mg/L and 306 (± 58) mg/L, respectively (Figure 4.13 b). But also for the pCOD the UASB had a better removal of 45 % with an outflow of 313 (± 136) mg/L compared to 28 % and 431 (± 128) mg/L for the MS (Table SI 8.2.2).

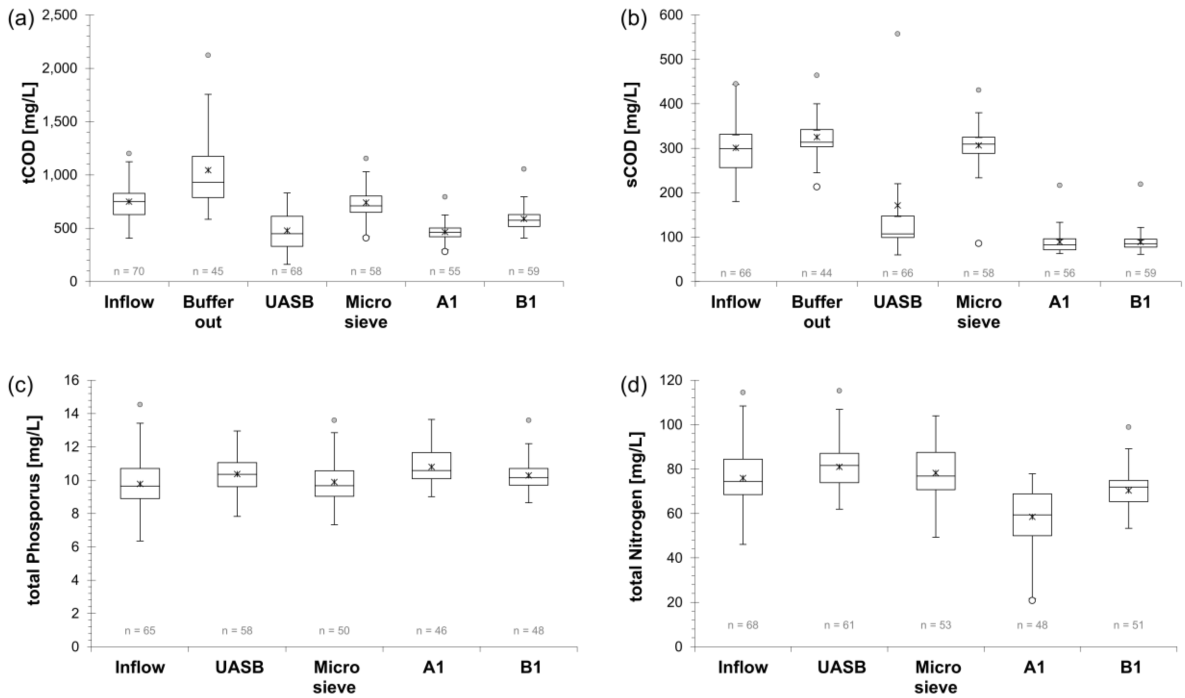

Figure 4.13: Total (a) and soluble (b) COD effluent concentrations, and data for total phosphorous (TP) (c) and total nitrogen (TN) (d) concentrations of the UASB reactor, the micro sieve, the secondary facultative pond A1, and the primary facultative pond B1

With regard to the potential water reuse purpose for irrigation of fodder crops nutrient removal or conservation is another important aspect to be considered. As shown in Figure 4.13 c and Figure 4.13 d there was almost no removal of phosphorus and nitrogen. The effluent values for TN were very close with 81 (± 10) mg/L for the UASB and 79 (± 12) mg/L for the MS. The same applied for TP with 10.4 (± 1.1) mg/L and 9.9 (± 1.3) mg/L, respectively. This corresponds well with findings of Dias et al. (2017) who measured even slight increases in TKN concentrations after a UASB. Jahn et al. (2017) and Walder et al. (2015) reached 29 % of TP removal and 6 % of TN with their MS. However, they also implemented a flocculation process which certainly influenced the phosphorus removal.

4.4.7 Facultative ponds

Both first ponds were originally designed as primary facultative ponds, whilst pond A1 was transferred into a secondary facultative pond with the installation of the pre-treatment (Shilton, 2005; von Sperling, 2007a). The floating baffles were installed into pond A1 on day 64 as measure to optimize the flow conditions instead of potential short circuiting existing in B1, as last measure of three improvements. The other two main differences of this pond (A1) were the reduced inflow loads of COD and TS (due to the pre-treatment) and the full available pond volume due to initial sludge removal. The total inflow into pond A1 was the combined effluent from the UASB reactor (16 %) and the MS (84 %) whilst pond B1 received 100 % raw sewage.

The tCOD concentration in the effluent of pond A1 was 21 % lower than that of B1 with 465 (± 93) mg/L versus 588 (± 123) mg/L, respectively (Figure 4.13 a). Also the pCOD concentrations were lower in A1 with 380 (± 92) mg/L than in B1 with 461 (± 125) mg/L. However, the sCOD concentrations with 89 (± 28) mg/L for A1 and 89 (± 22) mg/L for B1, were the same (Figure 4.13 b). This clearly showed that the enhancement of the pond system had its main impact on the removal of the pCOD.

At the same time both ponds, A1 and B1, recorded impacts on the microbial parameters. All measured parameters indicated further but low removal. A1 had maximum one $\log_{10}$ unit better effluent values than pond B1. For *E. coli* it reached an average of 4.4×10^5 ($\pm 6.6 \times 10^5$) MPN/100mL with B1 of 1.6×10^6 ($\pm 3.2 \times 10^6$) MPN/100mL (Figure 4.14). Total coliforms were almost at the same level with 1.2×10^7 ($\pm 1.3 \times 10^7$) MPN/100mL and 2.6×10^7 ($\pm 3.8 \times 10^7$) MPN/100mL, respectively. Also for *Enterococci*, there was one $\log_{10}$ unit difference with A1 being 8.6×10^3 ($\pm 1.4 \times 10^4$) MPN/100mL and B1 being at 6.3×10^4 ($\pm 4.4 \times 10^4$) MPN/100mL. Even with those low removal values it has to be stated that disinfection was not the main task of facultative ponds. This had to be achieved by the following maturation ponds (Shilton, 2005; von Sperling, 2007a).

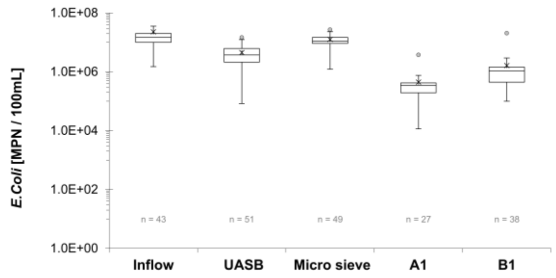

Figure 4.14: *Escherichia Coli* concentrations of the inflow, the UASB reactor, the micro sieve, train A and train B (MPN – most probable number)

In order to judge the treatment capacity of the ponds, the effluent concentrations were one aspect as they are relevant for the intended irrigation purpose. However, due to high evaporation losses, those concentrations can be misleading in terms of the functionality of the system. Therefore, we also considered loads and their reduction. The mean tCOD loads at the outflow of pond A1 were 158 (± 69) kg/d with a removal efficiency of 49 % compared to the inflow into train A. At the same time the removal after pond B1 was only 36 % with an average load of 211 (± 117) kg/d (Figure 4.15). For the sCOD the removal in both trains was 77 % with similar load values of 29 (± 15) kg/d for A1 and 32 (± 18) kg/d for B1. Also for TP, there was only a slightly reduced load for A1 with 3.4 (± 1.5) kg/d compared to 3.8 (± 2.0) kg/d at B1. On the contrary, the TN loads were further reduced in train A than in train B with 19 (± 9) kg/d and 26 (± 14) kg/d, respectively.

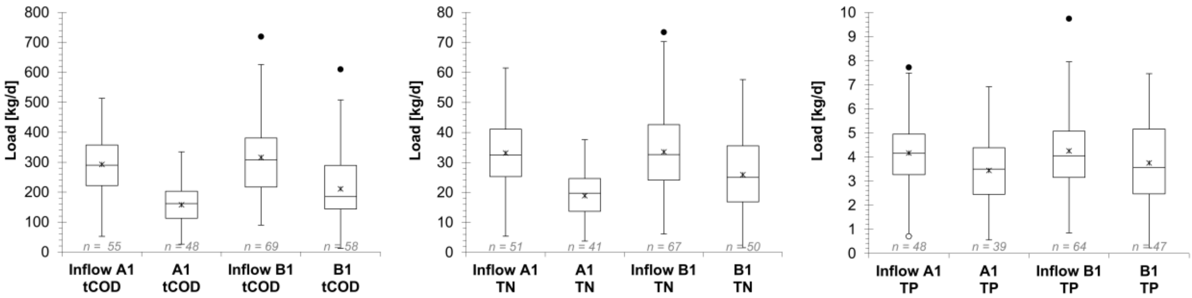

Figure 4.15: Reduction of tCOD, TN and TP loads over the treatment train

4.4.8 Challenges for large scale implementation

Both technologies proved to be viable options for the pre-treatment of raw sewage upfront of WSP in the sub-Saharan context. However, for the large scale implementation of only one technology there are certain issues that have to be considered. The UASB had better effluent values but only treated a small fraction of the total inflow. For further improvement of the inflow to pond A1 by treating the entire inflow volume with UASBs they would have to be much larger. Either a much bigger reactor or several reactors in parallel, which would be more complicated to feed and operate, have to be installed. In the local situation power supply was needed to operate the feed pump of the UASB. Depending on other situations it might be possible to feed the UASB with the last sewer pump in town. In this case proper waste removal has to be ensured at the last pump station. Besides the reactor itself dry beds are needed for the handling of excess sludge. They have to be designed according to the size of the reactor and the expected sludge accumulation. In order to reduce climate gas emissions, methane has to be transformed to CO_2. This can be either achieved with a flare as in this study or, if higher volumes of CH_4 are produced, with a combined heat and power system. This would also have a positive effect on the energy consumption of the plant. Another consideration for the implementation of a system with UASB reactors is the local temperature. During hot seasons the removal efficiencies were considerably better, and therefore, it has to be ensured that also during colder periods the effluent values are reached. In areas with long cold periods the pre-treatment with a MS might be a better option as temperature has no influence on the MS. However, constant power supply is necessary for operation of the MS as well as for the spray water pump. If process water is used for spraying, a buffer tank is required, so that tap water consumption can be reduced. At the same time the MS was flexible with changing inflow peaks and volumes. The one MS would be able to treat the full inflow as the only pre-treatment technology. Nevertheless, regular maintenance such as cleaning of the sieves and replacement of spare parts are necessary, which can be difficult to acquire in the region. Also further treatment of the sieving residue is required. Ideally, this is done with a fermentation process, so that the produced biogas can be transferred into power. Alternatively, the residue can be composted and after stabilization and drying used as fertilizer.

During the research period operation and maintenance was conducted jointly by project partners, students and the local Namibian plant operators. They were trained during the implementation of the project and are afterwards capable to sustain the required daily works.

4.5 Conclusions

In this study, two different pre-treatment technologies were compared for the enhancement of overloaded WSP in Africa, also providing a detailed dataset of wastewater characteristics and effluent qualities. This full scale research proved, that both, UASB-reactor as well as micro sieve, are capable of reducing COD and TSS from the raw wastewater in the local context of warm climates, especially if there are space restrictions for further extensions. However, with the technical enhancement of existing WSP, the requirements increase. This means, that regular power supply is needed for the micro sieve, and it would be a benefit for the operation of the UASB reactor. Secondly, more maintenance for pumps and machinery will be necessary. At the same time this improves the approach of the operators, who might have interpreted WSP as maintenance-free. Thirdly, spray water is necessary to clean the sieve. Ideally this is implemented through a recirculation system of process water. Water consumption could be further reduced with a combined air and water cleaning.

The reduction of COD and solid loads into the first pond was achieved by both technologies with better effluent values of the UASB. However, the MS was more flexible with changing inflows and large volumes. The UASB further reduced sCOD, pathogens and small amounts of nutrients, which is beneficial for further treatment with ponds but not strictly required. In contrast, phosphorus and nitrogen are valuable nutrients for the projected irrigation. Little to no removal of TN and TP was observed with the pre-treatment, only later small amounts were consumed by algae and therefore remained in the system and were available for further irrigation purposes.

A positive effect of sludge removal, pre-treatment and baffles in train A was evident by better effluent values from A1 than with the unimproved pond B1. A further benefit of the pre-treatment will be lower sludge accumulation rates and therefore longer removal intervals. Further research will focus in detail on the effluent values of pond A4 and B4 and the suitability of the water quality for irrigation purposes. Such research will also include the maturation ponds and especially the influence of algae on the irrigation water as well as the disinfection and reduction capacity for pathogens. Long term performance and operation stability data of UASB and MS will also be available for this local context.

5 Upgrade of waste stabilization ponds to improve effluents for reuse purposes

This chapter has been submitted for publication in the H2Open Journal (2022) with S. Agrawal, L. Orschler, S. Schubert, S. Lackner.

5.1 Abstract

Waste stabilization pond (WSP) systems are operated in many countries. They are especially effective in warm climates and regions without space limitations. Due to fast growing populations many of those WSP systems are overloaded and at the same time their countries are affected by changing climate and irregular rainfalls which cause an increasing demand for irrigation water. There is still little long-term experience with WSP especially in Africa and thus this study provides a comprehensive investigation of a WSP in North Namibia and the effect of different factors on treatment performance. One treatment train A of this WSP system was equipped with mechanical and anaerobic pre-treatment as well as post-treatment and was compared to a second train B without enhancement. The results show positive effects on the removal of COD, TSS and to some extend pathogens with a micro sieve and UASB as pre-treatment. In contrary, the rock filter as post-treatment did only reduce 5 % of chlorophyll-*a* and did not show any additional removal of algae compared to the original train. Algae concentrations were best removed with pre-treatment, sludge removal and baffles in the facultative pond.

With regards to pathogens the enhancement measures showed varying effects. *E. coli* were reduced to the new EU water reuse standard of 1,000 MPN/100mL for fodder irrigation, *P. aeruginosa* stagnated and *Enterococci* levels increased. The main pathogen reduction happened during pre-treatment and in the facultative pond with baffles and not in the maturation ponds. For total coliforms no clear trend was visible. Among the top 20 genera found *Pseudomonas* was dominant at most sampling points. The research showed that different enhancement measures at WSP can improve the effluent and help to reach the reuse requirements especially for restricted irrigation of fodder crops. At the same time, high tCOD and TN effluent values did not meet Namibian and European reuse standards. But as large portions of the tCOD, consisting of algae, add valuable biomass and TN valuable fertilizer to the barren soil, it raised the question if all the parameters set in the different standards are directly applicable for WSP or have to be adapted to their specificities.

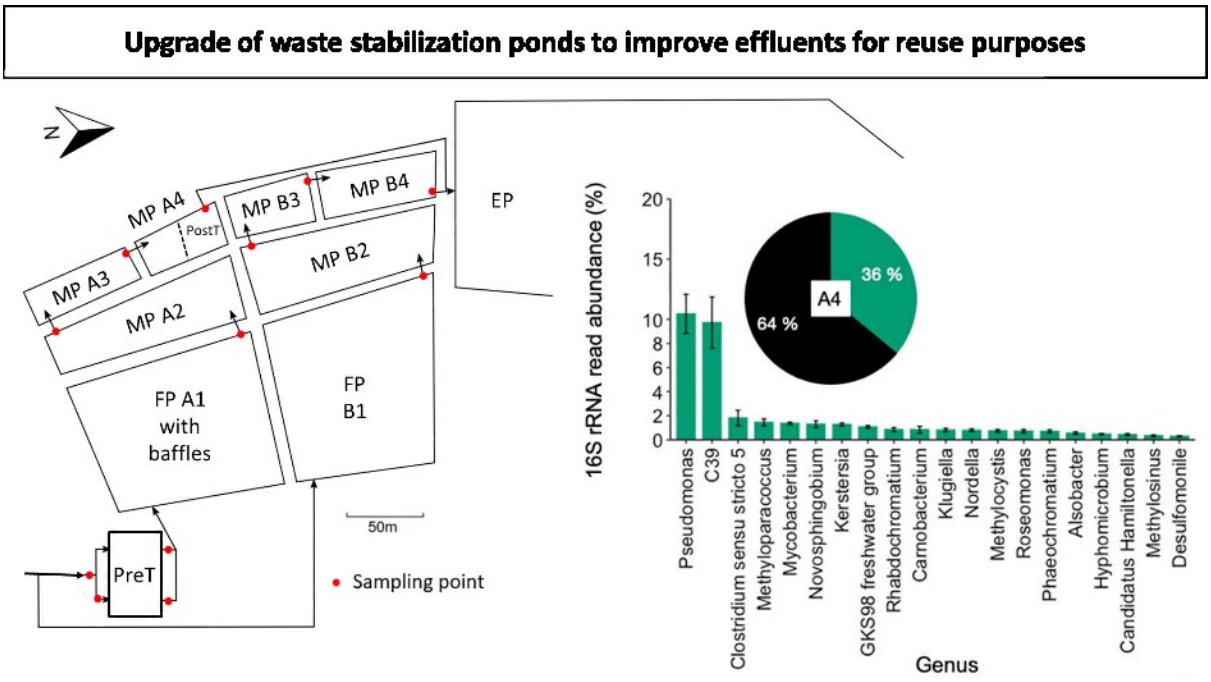

Figure 5.1: Graphical abstract: Upgrade of waste stabilization ponds to improve effluents for reuse purposes

5.2 Introduction

Waste stabilization ponds (WSP) are common wastewater treatment options in water scarce areas especially in Africa but also on other continents (Ho and Goethals, 2020; Janeiro et al., 2020). Treated wastewater presents a valuable resource of water as well as nutrients, however, quality assurance is required, esp. regarding pathogens. In Namibia, the effluent water quality of WSP often does not fulfil the national and international requirements for water reuse. At the same time there is a high demand on water for irrigation purposes.

Traditional WSP systems consist of anaerobic ponds (AP), facultative ponds (FP) and maturation ponds (MP) (Shilton, 2005; Verbyla et al., 2017; von Sperling, 2007a). If there is no receiving water body an evaporation pond (EP) is also needed. The main purpose of AP and FP is the removal and stabilization of organic matter whilst MP are designed to remove pathogens (Verbyla et al., 2017).

Operators of WSP systems are facing fast growing urban populations with rapidly increasing inflows to their WSP, which leads to overloading and overflowing (Verbyla et al., 2013). The classical way of meeting this challenge is to increase the number of ponds and their surface. However, the related raising evaporation is contradictory to the aim of water reuse. Therefore, sustainable solutions are needed to achieve sufficient water quality and at the same time reduce evaporation losses. WSP have been studied extensively in Latin America (Dias et al., 2017b; Pham et al., 2014; Verbyla et al., 2016; von Sperling and De Andrada, 2006; von Sperling et al., 2007), Australia, New Zealand and the USA (Guieysse et al., 2012; Powell et al., 2011) but there is only punctual research in sub-Saharan countries such as Burkina Faso, Ghana, Malawi and Tanzania (Bansah and Suglo, 2016; Kihila et al., 2014; Konaté et al., 2013; Maiga et al., 2009; Ngoma et al., 2020; Zacharia et al., 2019) on WSP, particularly on reuse for the irrigation of fodder crops.

Algae and cyanobacteria are indispensable for WSP. Within the algal-bacterial mutualism they produce oxygen, and consume nutrients and CO_2. They also play an important role on the removal and inactivation of pathogens (Liu et al., 2020). However, only little information is available on the microbial communities in WSP. Eland et al. (2018) quantified cyanobacterial and eukaryotic communities in two WSP in Brazil and Wallace et al. (2015) evaluated algae and macrophyte species distributions in three WSP in Canada. Due to high algae contents the effluent from WSP presents an opportunity to add biomass into barren soil and improve its water-holding capacity (Mara, 2004). But negative effects of algal toxins need to be considered (Ho and Goethals, 2020). Algae contribute considerable amounts of pCOD to the effluent and are therefore important regarding reuse requirements.

According to Verbyla et al. (2017) the most important factor for pathogen reduction in WSP is sunlight. Temperature, turbidity, dissolved oxygen (DO), pH, sedimentation and hydraulics are also important but have varying effects on the removal of different pathogens. Most of these factors are also influenced by the amount and activity of the algae. So far, there have only been short term investigations and literature reviews on pathogen reduction in WSP (Ho and Goethals, 2020; Janeiro et al., 2020; K'Oreje et al., 2020; Nikiema et al., 2013) but no long-term studies have been conducted on WSP in sub-Sahara Africa.

This research presents an extensive evaluation of factors affecting the treatment performance and effluent quality of WSP. A WSP system in Namibia was monitored over four years under several operational conditions, i.e. overloading of one treatment train, load reduction scenarios and additional pre- and post-treatment for enhancement. These scenarios were assessed regarding the removal of the chemical oxygen demand (COD), total suspended solids (TSS) and selected pathogens. Additionally, nutrient levels and algae content were considered, as important parameters for water reuse.

The following research questions were addressed: What are the seasonal effects on the effluent water quality of WSP comparing an overloaded with an enhanced treatment train? How does a reduced hydraulic load influence the performance? What is the impact of additional treatment technologies on the performance of a WSP? How are composition and concentrations of pathogens as well as the microbial community in general affected by these operation scenarios and improvements? Which enhancements have the highest removal potential? Can enhanced WSP fulfil Namibian and European water reuse standards?

5.3 Material and Methods

5.3.1 Configuration of the waste stabilization pond system

The investigated WSP system (Figure 5.2) is located in North Namibia and consisted of two parallel treatment trains A and B, each with one primary FP and three MP with almost identical pond surface and volumes (Further information in chapter 4.3.1). In order to enable water reuse for irrigation treatment train A was enhanced with a micro sieve (MS) and in parallel with an upstream anaerobic sludge blanket (UASB) reactor as pre-treatment (PreT) to reduce TSS and organic carbon (measured as COD) and floating baffles in pond A1 to approach plug flow conditions. As post-treatment (PostT) a rock filter was constructed to improve the effluent quality and reduce algae concentrations. For further information on the PreT and the PostT it is referred to Sinn and Lackner (2020) and Rudolph et al. (2020), respectively. The almost identical treatment train B remained in its original setup as comparison. Operation was divided into three phases. During phase I (day 1 – day 675) the PreT was constructed and during phase II the PostT. With the start of phase III on day 1,012 the whole plant was operational (Table SI 8.3.1). For phases I+II train B received 800 m^3/d inflow, during phase III 430 m^3/d. After commissioning, train A received on average 350 m^3/d in phase III. The inflow into pond B1 was raw wastewater whilst in A1 it was pre-treated. During the one year of full operation there was a 50 days suspension of the PreT after lightning stroke on day 1,275. Train A was back to full capacity on day 1,362 (Figure 5.5 c).

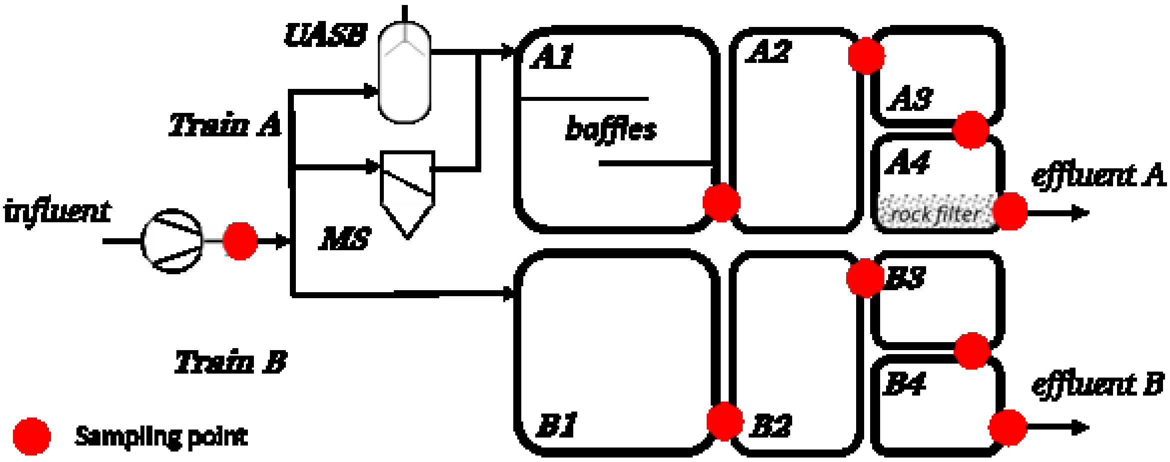

Figure 5.2: Flow chart of the waste stabilization pond system with the two treatment trains A and B. Both trains consist of four ponds each. Train A was equipped with additional pre-treatment (UASB – upstream anaerobic sludge blanket and MS – micro sieve) followed by floating baffles in pond A1 as well as a rock filter as post-treatment in pond A4. The red points are the sampling points for the water quality analyses.

5.3.2 Sampling and analyses

During four years of operation one-litre grab samples were taken regularly at the inflow and overflows of each of the eight ponds. Electrical conductivity (EC), pH, temperature and DO were analysed directly on site with a WTW multimeter 3410 (Xylem Analytics, Germany) and samples were cooled and transported to the laboratory. The samples were analysed after homogenizing or filtration (0.45 μm, Whatman membrane, ME 25) with Hach cuvette tests using a spectral photometer DR 2800 (Hach Lange, Germany) for: COD, total nitrogen (TN), ammonium (NH_4-N), nitrite (NO_2-N), nitrate (NO_3-N), total phosphorus (TP), phosphate (PO_4-P) and potassium (K^+). According to standard methods (DIN 38409-2) (DIN, 1987) TSS and volatile suspended solids (VSS) were measured with glass microfiber filters (Whatman 934-AH). Further, chlorophyll-*a* concentrations were analysed based on DIN 38409-60 (DIN, 2019).

The following indicator bacteria were determined using an IDEXX system with Quanti-Tray/2000 (IDEXX, Germany): total coliforms and *E. coli* (Colilert-18), *Enterococci* (Enterolert) and *Pseudomonas aeruginosa* (Pseudalert). For a selected number of samples, biomass was collected in biological triplicates and centrifuged in 50 mL tubes at 8,000 g and 4 °C for 25 minutes. After discharge of the supernatant the pellets were stored at 4 °C prior to further downstream analysis. Total genomic DNA was extracted with the Fast DNA Spin kit for soil (MP Biomedicals, Germany) based on a modified manufacturer's protocol (Orschler et al., 2019). The DNA concentrations were determined by a Qubit 3.0 Fluorometer with Qubit dsDNA HS kit (Thermo Fisher Scientific, Germany) and DNA was subsequently used for 16S rRNA gene amplicon sequencing. We targeted multiple hypervariable regions of 16S rRNA genes with the 16S Ion Metagenomics Kit™ (Thermo Fisher Scientific, Germany) by two separate PCR reactions, amplifying the V2, V4, V8 and V3, V6-7, and V9 hypervariable regions, according to the kit protocol, as described in Agrawal et al. (2020). Sequencing was performed on an Ion Torrent (ION Torrent Ion S5) using the 400-bp kit and 530 chip. Base calling and run demultiplexing were conducted by Torrent Suite version 4.4.2 (Thermo Fisher Scientific, Germany) with default parameters. DADA2 (v 1.14.1) was implemented for separating sequences for each sample; filtering the low quality and limit the length of sequences > 260 bp; filtering sequences with potential chimera. The sequences were classified based on the taxonomy in the Silva database (97 % confidence threshold, version 138). Diversity and compositional analysis was performed in R (http://www.R-project.org/), using phyloseq (1.30.3) and ggplot2 (3.3.0) packages.

The performance of the enhanced train A during phase III was compared with the water quality of train B in its original setup. The effluent quality was also evaluated in comparison with the code of practice for wastewater reuse in Namibia (DWAF, 2012), the newly published regulation on minimum water requirements for water reuse in the European Union (EU, 2020) as well as the necessary water quality for agriculture by the Food and Agriculture Organization (FAO) (Ayers and Westcot, 1985).

In each train the MP HRT was calculated with the water flow divided by the related pond volume. Given the strong variations of the wastewater flow the HRT was averaged over 21 days. This included the ten days before and after each measurement.

5.3.3 Richness and Shannon Index

The microbial community was analysed to estimate the richness measured as Chao1 index. This index refers to the number of amplicon sequencing variants (ASVs) expected in the samples based on the number of ASVs found in the samples (Chao, 1984). Additionally, the Shannon index was measured to estimate the microbial community diversity in the samples (Schloss and Handelsman, 2006), which is based on the diversity (means the number of different ASVs) as well as the evenness (it describes the abundance of each ASV, lower difference between the abundance of each ASV suggest higher evenness) in the samples (Table SI 8.3.3).

5.4 Results

5.4.1 Seasonal effects and influence of HRT

The biological wastewater treatment of the WSP strongly depended on the local climate. The effects of rain and temperature are presented in Figure 5.3 for train B for all phases. During summer months (October till March) the highest water temperatures of up to 35 °C and the highest precipitation of over 65 mm/d were measured. Even with the separate sewer system, storm water was evacuated through incorrectly connected rain gutters and untight sewers. This dilution was visible at the outflow of B4 after 200 days at the end of the first summer with the lowest EC of 400 μS/cm as well as concentrations of tCOD below 200 mg/L, TN of 20 mg/L and TP of 4 mg/L.

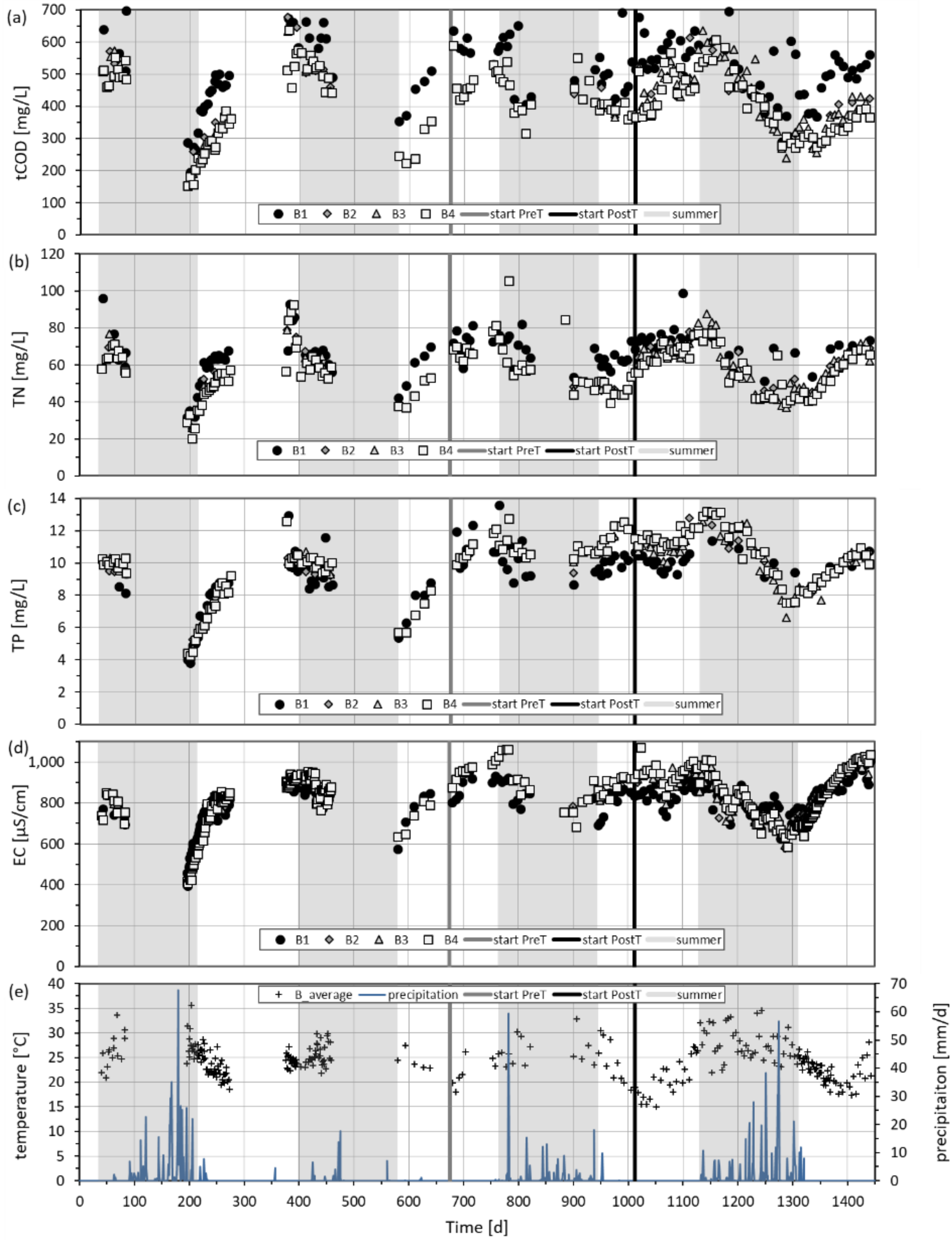

Figure 5.3: Water quality of each pond B1 – B4 over four years for the parameters: total chemical oxygen demand (tCOD) (a), total nitrogen (TN) (b), total phosphorous (TP) (c), conductivity (d), water temperature and precipitation (e). The grey areas indicate the rainy seasons or summer periods (October – March) and the grey line represents the start of the PreT (phase II) and the black line the start of the PostT (phase III) in train A.

The second summer was dry with minor rainfalls. Therefore, this period represented the influence of higher temperatures without dilution effects. All effluent values reached their low point at the end of this season but did not drop as much as the year before. During the third summer there were small rainfalls of 207 mm in total with high water temperatures of up to 35 °C. But the lowest effluent values remained higher than the two years before. EC was reduced to 700 μS/cm, tCOD to 400 mg/L, TN to 40 mg/L and TP remained above 10 mg/L in the outflow of B4.

The microbial parameters showed variable effects over the four years. For total coliforms (Figure 5.4 a) the outflow concentrations of B1 remained between six and eight log values at the same level without any seasonal influence. At the outflow of B4 there was more variation but hardly any removal. The lowest effluent values were reached after heavy rainfalls at the end of the first summer season.

On the contrary, the *E. coli* concentrations (Figure 5.4 b) showed different behaviour. The outflow of B1 remained at a constant level between five and seven log values over the whole research period whilst the values at the effluent of B4 showed a clear decrease after the start of phase III and the increased HRT. *Enterococci* (Figure 5.4 c) concentrations after B1 were constantly between four and five log values whilst *Pseudomonas aeruginosa* (*P. aeruginosa*) (Figure 5.4 d) concentrations were slightly increasing over the four years from five to seven log values. The effluent concentrations at B4 remained at similar levels for *P. aeruginosa* even with the reduced inflow from day 1,012 onwards. For *Enterococci* the effluent concentrations of B4 increased from below 1,000 MPN/100mL to about 10,000 MPN/100mL after the commissioning of train A.

During phase III train B received only half the daily plant inflow and still the yearly variations were visible (Figure 5.3). With rainfalls similar to the first year the dilution effect appeared but was not as distinct. The low point of EC was 600 μS/cm whilst tCOD was 250 mg/L, TN at 40 mg/L and TP at 7 mg/L. With the last winter season all effluent values increased. EC reached its high point with over 1,000 μS/cm whilst TN and TP stayed within their general fluctuation. Only tCOD remained below 400 mg/L which could be a first indication of reduced inflow loads and higher hydraulic retention times (HRT).

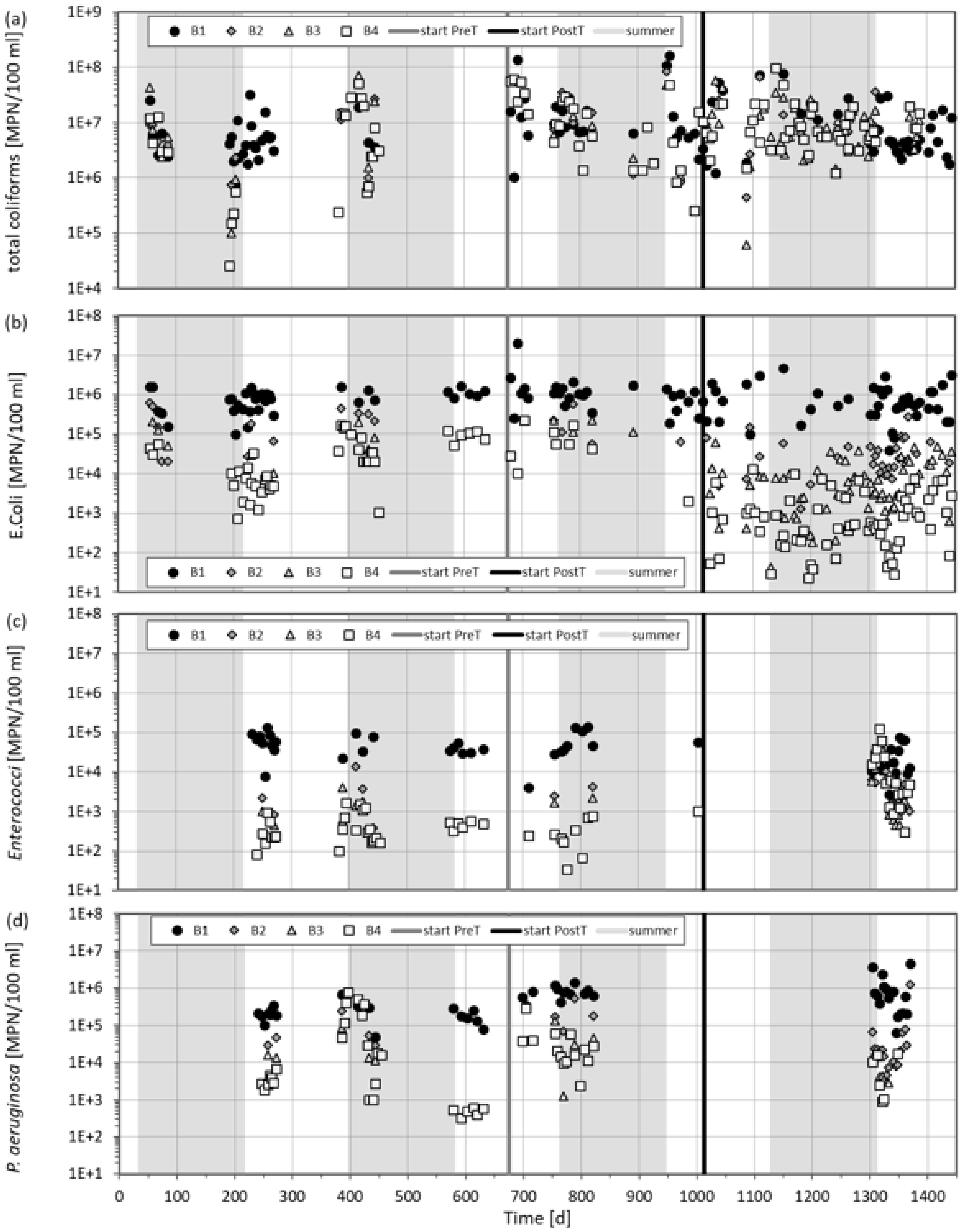

Figure 5.4: Water quality parameters of train B. Data is shown for each pond (B1 – B4) over four years for the parameters: total coliforms (a), *Escherichia Coli (E. coli)* (b), *Enterococci* (c) and *Pseudomonas aeruginosa (P. aeruginosa)* (d). The grey areas indicate the rainy seasons or summer periods (October – March) and the grey line represents the start of the PreT (phase II) and the black line the start of the PostT (phase III) in train A.

5.4.2 Upgrade: pre-treatment and post-treatment in train A

With the commissioning of the enhanced train A tCOD (Figure 5.5 a) concentrations in the effluent of pond A4 started with 300 mg/L, increased to almost 500 mg/L in the middle of the summer and fell below 200 mg/L at the end of the summer. Afterwards it remained stable around 150 mg/L.

Whilst the tCOD showed seasonal effects the sCOD concentration (Figure 5.5 b) had only little variations of 20 to 40 mg/L. The effluent of pond A1 had the lowest sCOD concentrations between 60 and 80 mg/L. The final concentrations in the effluent of A4 had a higher variation between 60 and 100 mg/L. Nutrients followed a similar pattern as tCOD. TN concentrations (Figure 5.5 e) in A1 started out at 60 to 70 mg/L and decreased to 28 mg/L with the dilution during rainfall and the interruption, but rose again over 70 mg/L. A4 effluent concentrations were between 40 and 60 mg/L, and reduced to 30 mg/L in summer and to 10 mg/L during the interruption period (Figure 5.5 e). After reaching full capacity again the values remained at 10 mg/L and then started raising again with lower temperatures to 30 mg/L. During the whole research period the NO_3-N concentrations remained below 1 mg/L.

The NH_4-N concentrations (Figure 5.5 g) experienced a different effect with the power cut on day 1,275. The effluent concentrations of A1 were about 40 mg/L at the beginning and dropped to 22 mg/L at the end of the summer but then returned to the original level. During summer, the NH_4-N concentrations almost completely reduced in pond A3 and remained around 1 mg/L during the interruption. With full inflow it started raising again. At the same time there was a different effect visible with pond A4. The concentrations dropped to about 5 mg/L during summer but with the interruption increased again to 25 mg/L and reduced to 1 mg/L before regular inflow stared again.

There was hardly any phosphorous removal and the TP concentrations (Figure 5.5 f) between A1 and A4 were very similar. They were between 10 and 12 mg/L and with the beginning of the summer increased up to 14 mg/L followed by a constant drop down to 5 mg/L during the interruption period. After repairs the values returned to 11 mg/L which was at the same level as at the beginning.

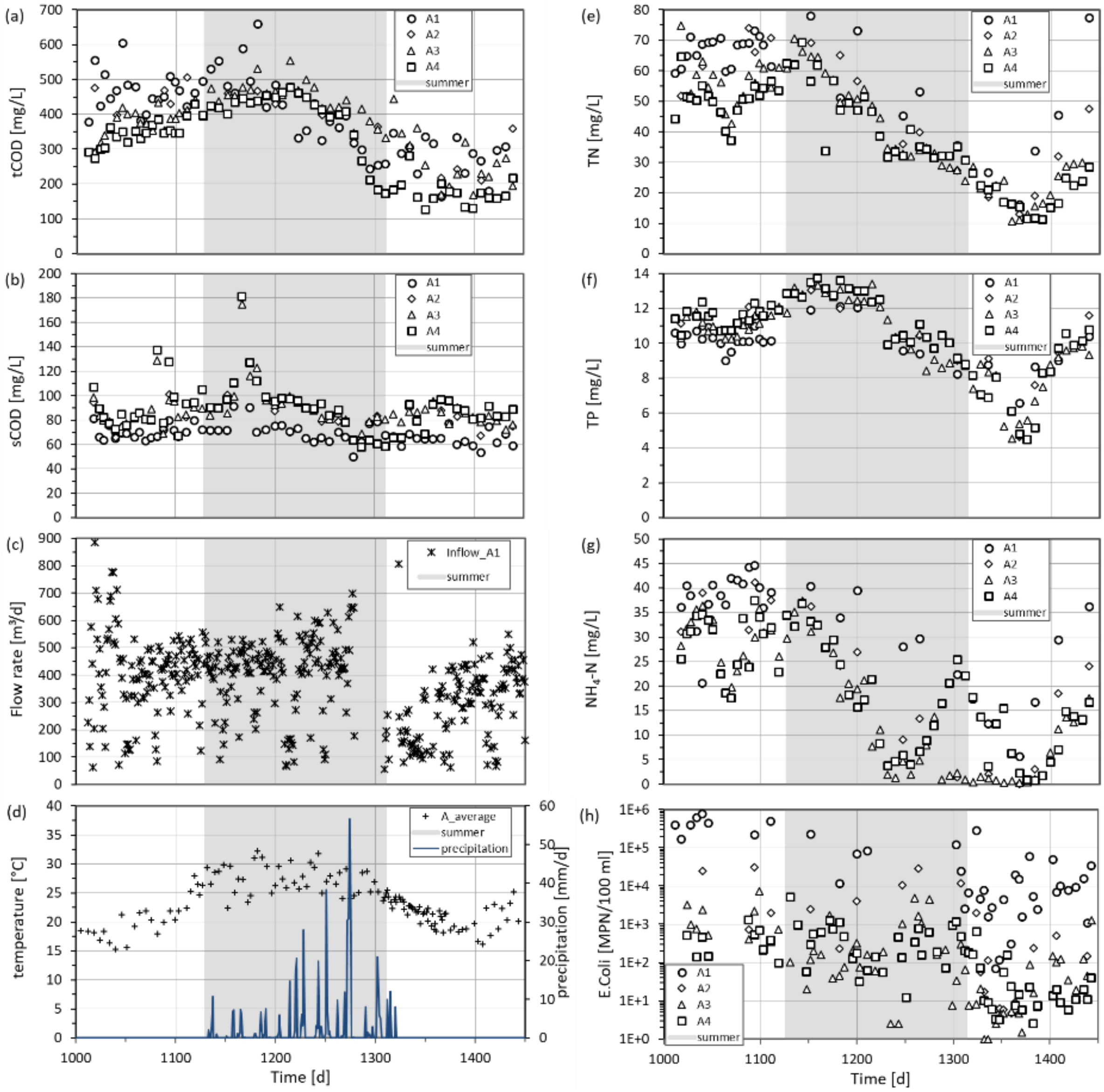

Figure 5.5: Water quality of each pond A1 – A4 over one year after the commissioning of the post-treatment (PostT) on day 1,012 for the parameters: total chemical oxygen demand (tCOD) (a), soluble chemical oxygen demand (sCOD) (b), inflow into train A (c), average water temperature and precipitation (d), total nitrogen (TN) (e), total phosphorous (TP) (f), ammonium (NH_4-N) (g) and *Escherichia Coli (E. coli)* (h). The grey area indicates the rainy season or summer period (October – March).

5.4.3 Gradients along the treatment trains

With the commissioning of the rock filter in pond A4 on day 1,012, phase III started. It was then possible to compare both trains under the same climatic conditions. The influent into pond B1 was raw wastewater whilst A1 received pre-treated water. Additionally, two floating baffles improved the hydraulic flow in A1. The tCOD (Figure 5.6 a) at the inflow was 764 (± 223) mg/L. This concentration was reduced by 48 % in A1 to 401 (± 121) mg/L and by 29 % in B1 to 544 (± 126) mg/L. At the effluent of the WSP, train A reached a total removal of 59 % with an average concentration of 317 (± 108) mg/L and train B reached 46 % with 415 (± 100) mg/L.

In the context of WSP the measured tCOD not only consists of components originating from wastewater but also from biomass due to algae growth. Therefore, the sCOD also needs to be considered. At the inflow the sCOD (Figure 5.6 b) had an average concentration of 315 (± 68) mg/L. At the overflow of A1 the removal was 78 % with 71 (± 16) mg/L. For B1 it was 75 % with 80 (± 25) mg/L. Afterwards, the sCOD did not further decrease in both trains and remained at 89 (± 20) mg/L in A4 and 85 (± 21) mg/L in B4. This trend was different for the removal of nutrients. TN (Figure 5.6 c) had an inflow concentration of 79 (± 12) mg/L and was reduced by 24 % to 60 (± 15) mg/L at the overflow of A1 and by 10 % to 71 (± 8) mg/L in B1. In both trains the removal continued up to the last ponds and reached a total reduction of 50 % (39 ± 15 mg/L) in A4 and 25 % (59 ± 11 mg/L) in B4. For TP (Figure 5.6 d) the average inflow concentration was 9.5 (± 1.4) mg/L and at the outflow of A4 and B4 it reached 10.6 (± 2.2) mg/L and 10.7 (± 1.5) mg/L, respectively. With regards to the removal of COD and nutrients in the WSP high evaporation has to be considered and therefore not only concentrations but also load reductions are presented in Figure 5.7.

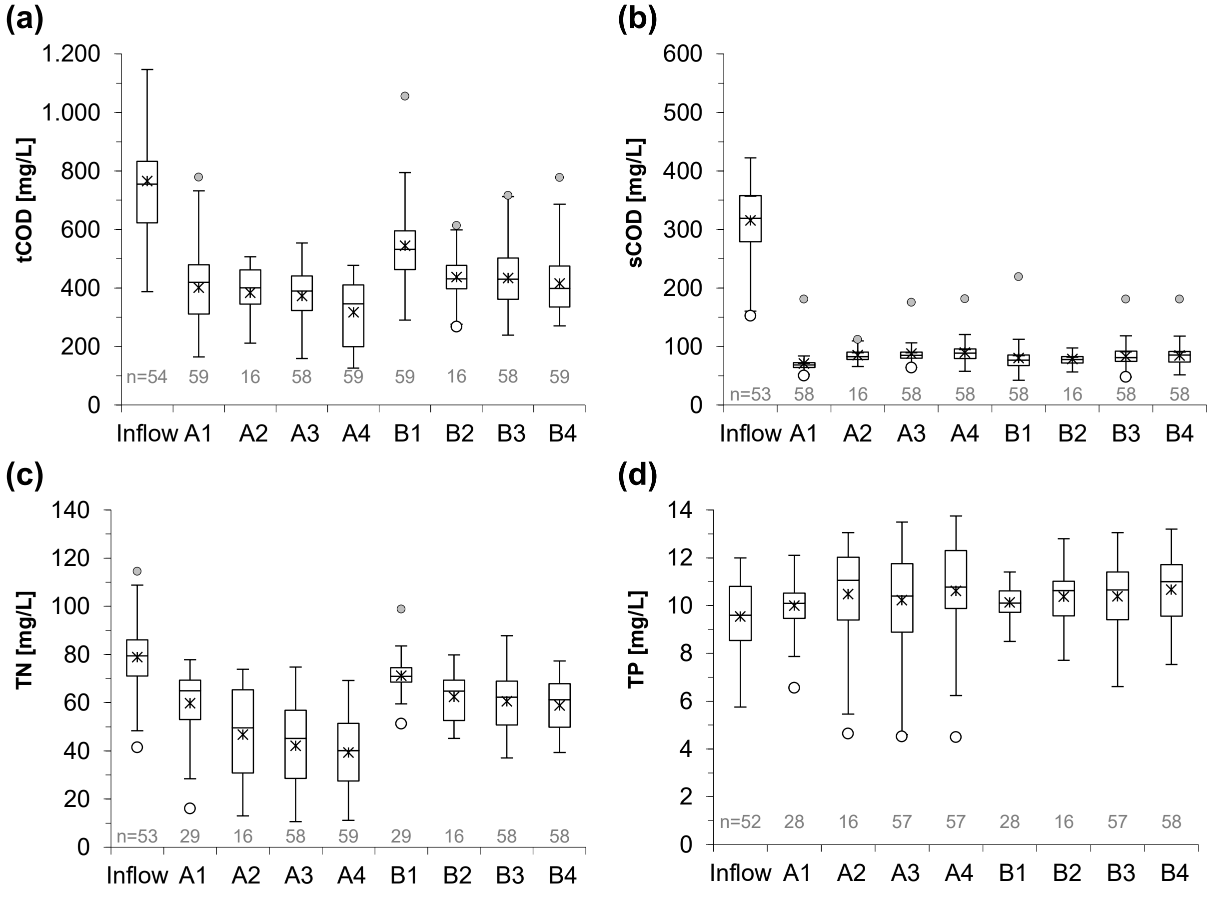

Figure 5.6: Inflow compared to outflow concentration from each pond A1 – A4 and B1 – B4 for the parameter: total chemical oxygen demand (tCOD) (a), soluble chemical oxygen demand (sCOD) (b), total nitrogen (TN) (c), total phosphorous (TP) (d).

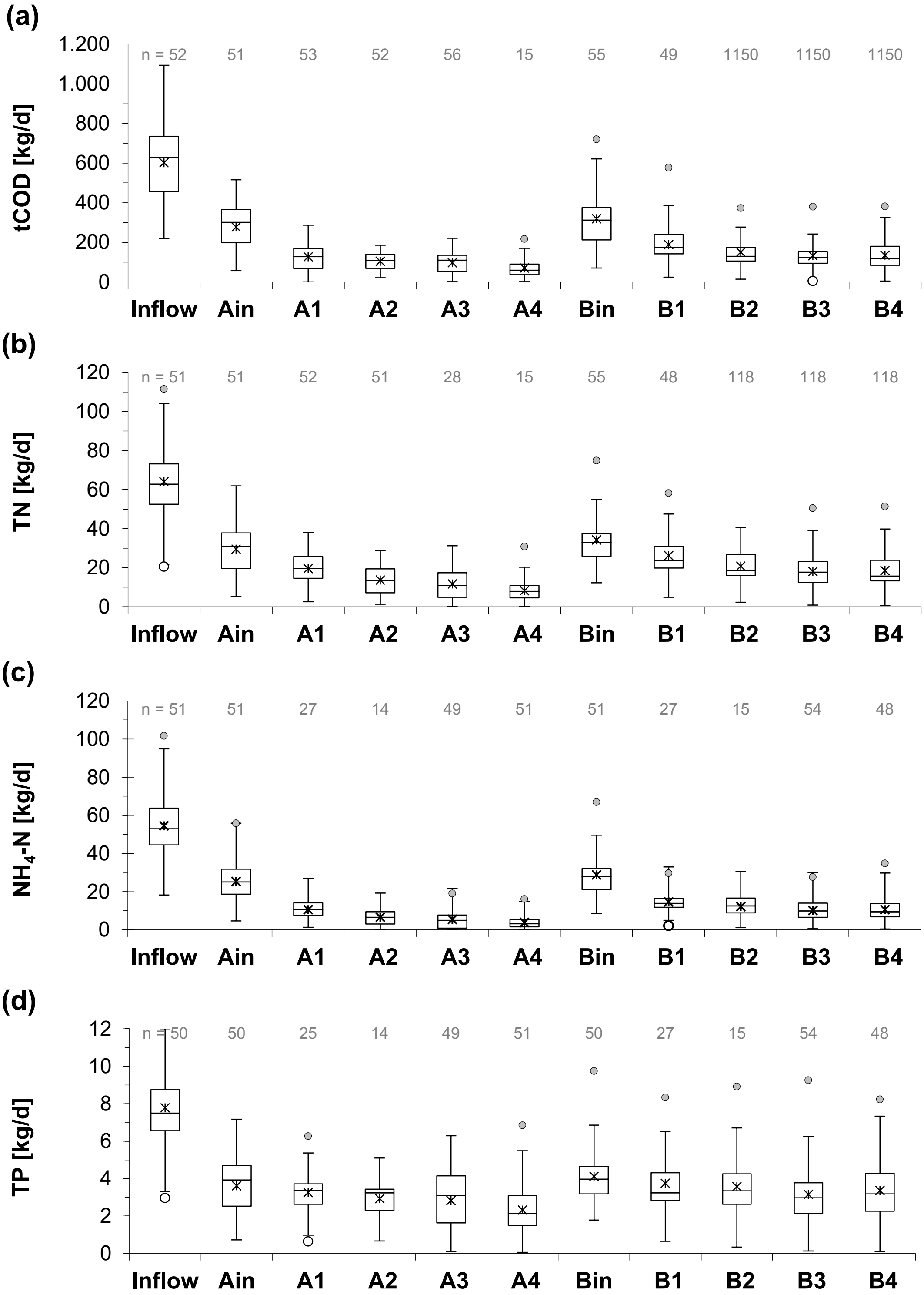

Figure 5.7: Loads for both treatment trains, including inflow to the WSP, inflow into A1 after pre-treatment (Ain) and effluent values for each pond, A1 – A4 and B1 – B4, for the parameters: total chemical oxygen demand (tCOD) (a), total nitrogen (TN) (b), ammonium (NH_4-N) (c) and total phosphorous (TP) (d).

For microbial indicators different trends were measured. Total coliforms inflow concentration (Figure 5.8 a) was on average 7.3 x 10^7 (± 4.2 x 10^7) MPN/100mL. After pond A1 this was reduced by 1.1 log values to 5.7 x 10^6 (± 1.1 x 10^7) MPN/100mL and after B1 by 0.7 log values to 1.3 x 10^7 (± 1.8 x 10^7) MPN/100mL. Over the following ponds in train B there were only small changes with a final effluent reduction in B4 of 0.8 log values to 1.2 x 10^7 (± 1.5 x 10^7) MPN/100mL. In train A there was a small up and down over A2 and A3 and the final effluent concentration from A4 was higher than of A1. Compared to the inflow concentration a 0.9 log values reduction was observed down to 8.9 x 10^6 (± 1.2 x 10^7) MPN/100mL.

E. coli showed a different trend (Figure 5.8 b), as the average inflow concentration to the plant was 1.2 x 10^7 (± 8.1 x 10^6) MPN/100mL. This value was reduced through the PreT and the baffles by 2.0 log values down to 1.2 x 10^5 (± 1.9 x 10^5) MPN/100mL after A1. In B1 the reduction was 1.1 log values down to 9.4 x 10^5 (± 9.4 x 10^5) MPN/100mL. In the following ponds B2, B3 and B4 there was a continuous removal by overall 3.8 log values down to 1.8 x 10^3 (± 2.6 x 10^3) MPN/100mL. At the outflow of ponds A2 and A3 there was a removal of 3.4 and 4.3 log values, respectively. It levelled out with pond A4 and a reduction of 4.5 log values to 3.6 x 10^2 (± 6.8 x 10^2) MPN/100mL was achieved overall.

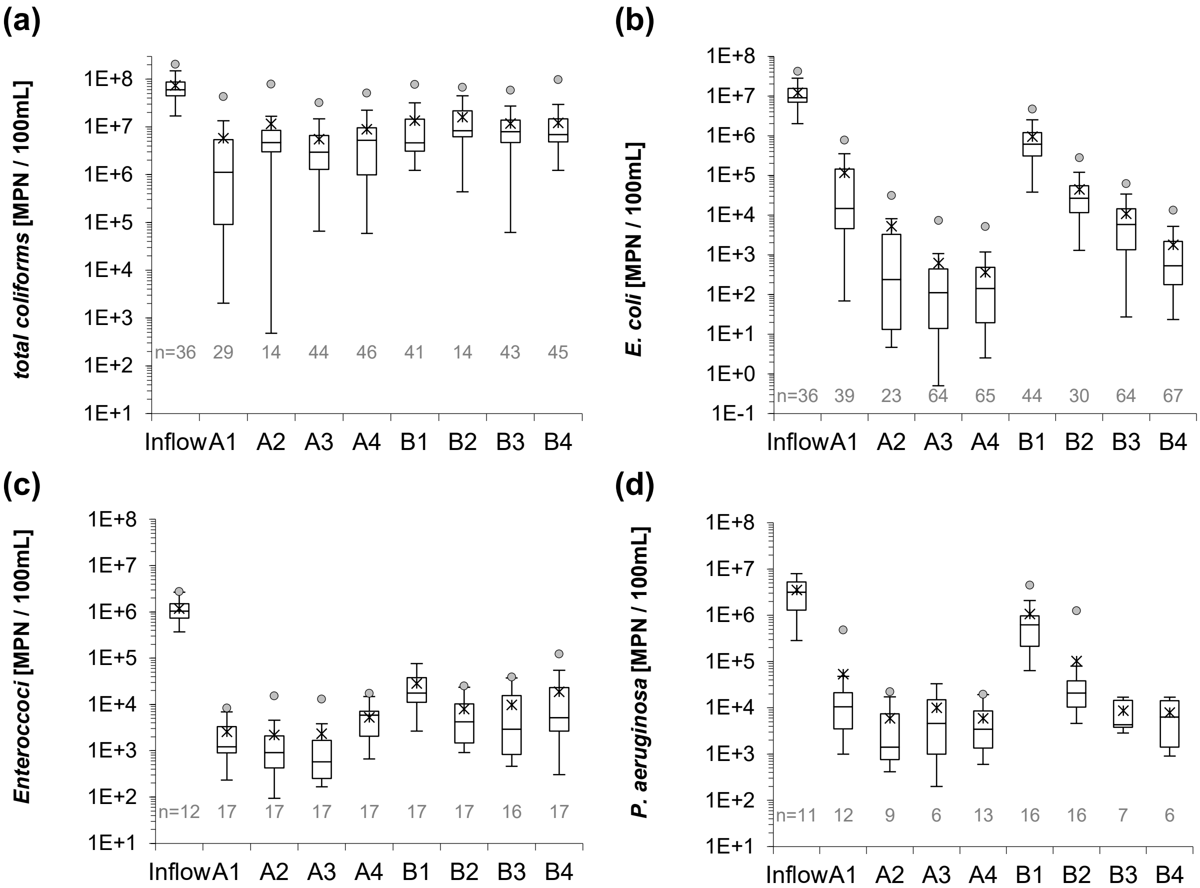

Figure 5.8: Inflow compared to outflow concentrations of each pond, A1 – A4 and B1 – B4 for the parameters: total coliforms (a), *Escherichia Coli (E. coli)* (b), *Enterococci* (c) and *Pseudomonas aeruginosa (P. aeruginosa)* (d).

The behaviour of the *Enterococci* concentrations was different (Figure 5.8 c). From the inflow with 1.2×10^{6} ($\pm 6.6 \times 10^5$) MPN/100mL train A reduced their concentration by 2.7 log values after A1. In comparison train B showed 1.6 log values in B1. The final overall log reduction in B4 was 1.8 to 1.9×10^4 ($\pm 3.1 \times 10^4$) MPN/100mL. In train A, no significant further reduction compared to the inflow concentration was measured. In contrary, it increased again between A3 and A4 with a total overall log reduction of 2.3 to 5.3×10^3 ($\pm 4.0 \times 10^3$) MPN/100mL.

For *P. aeruginosa* (Figure 5.8 d) there was a decrease in the concentration from 3.5×10^6 ($\pm 2.4 \times 10^6$) MPN/100mL at the inflow of 1.8 log values in A1 and 0.5 log values in B1. In train B, there was a significant further reduction from B1 to B2 to B3, but then it levelled out with overall 2.6 log values reaching 7.9×10^3 ($\pm 6.8 \times 10^3$) MPN/100mL. In contrary, further along train A there was only little more reduction of 2.8 log values to a value of 6.0×10^3 ($\pm 5.7 \times 10^3$) MPN/100mL in A4.

Even with stagnant concentrations there was a continuous load reduction for all parameters (Figure 5.7). The inflow load was distributed with 47 % into train A and 53 % into train B. The tCOD (Figure 5.7 a) at the inflow was 601 (± 186) kg/d, and reductions of 54 % and 41 % at the outflows of A1 and B1 were observed, respectively. The maximum removal of 75 % was reached at A4 with 70 (± 54) kg/d. In comparison B4 had almost double the effluent load of A4, with 135 (± 77) kg/d and a total removal in train B of 58 %.

This trend was also visible for TN (Figure 5.7 b) and NH_4-N (Figure 5.7 c) with the highest load removal. The total incoming TN load of 64 (± 18) kg/d was distributed to train A with 29 (± 12) kg/d and to train B with 34 (± 14) kg/d. At the outflow of A1 there was a 33 % removal to 20 (± 9) kg/d and 23 % removal at B1 to 26 (± 11) kg/d. Further removal of up to 72 % was measured in A4 with 8 (± 6) kg/d and 46 % in B4 with 19 (± 10) kg/d.

The highest load-based removal was observed for NH_4-N (Figure 5.7 c). The inflow load of 55 (± 17) kg/d was divided with 25 (± 11) kg/d into train A and 29 (± 12) kg/d in train B. In train A the removal increased from 57 % in A1 to 85 % in A4 with 11(± 5) kg/d and 4 (± 3) kg/d respectively. At the same time the removal was 49 % in B1 and 63 % in B4 with 15 (± 6) kg/d and 11(± 6) kg/d respectively.

Only looking at the rising TP concentrations (Figure 5.6 d) there could be the impression that there was no removal at all. This, however, was not the case and can be shown with the daily TP loads leaving each pond (Figure 5.7 d).

5.4.4 Natural disinfection – Reduction of pathogens

The average HRT for the original operation of train B during phase I and II (B – I+II) was 19.1 days with a standard deviation of 7.9 days (Figure 5.9). After the commissioning of the PostT (phase III) the HRT in train B (B – III) increased up to 42.8 (± 20.3) days. This large deviation of 20 days is caused by the breakdown period when the PreT and pumps were not operational due to power failure. Train A (A – III) had a 21-days-mean HRT of 47.8 (± 13.9) days. Given similar volumes of both trains this indicates a slightly lower inflow into train A compared to train B.

With regards to the corresponding log reductions of pathogens in all three operating stages (B – I+II; B – III; A – III) there was only a small difference for the reduction of total coliforms (Figure 5.9 a). During the operation of B – I+II the average log reduction was 1.1 (± 0.6). After the commissioning, A – III achieved a reduction of 1.4 log values and the same deviation of ± 0.6. During stage B - III only a reduction of 1.0 (± 0.5) log values occurred. However, there was an effect on the reduction of *E. coli* (Figure 5.9 b). During B – I+II the average log reduction was 3.1 (± 1.5). This reduction increased to 3.9 (± 0.6) log values during B – III. And during stage A – III an even better value with a log reduction of 5.1 (± 0.8) log values was achieved.

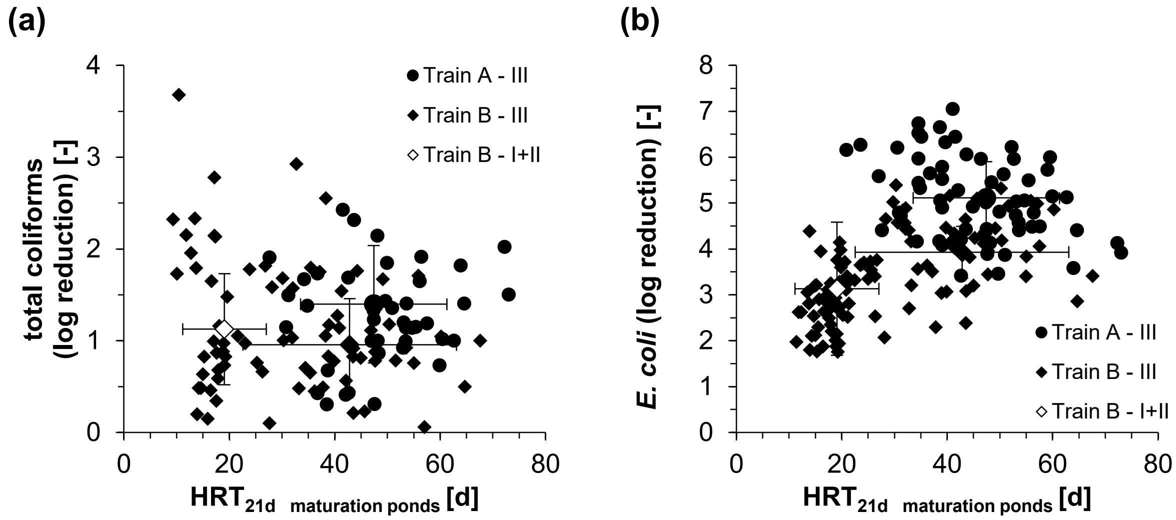

Figure 5.9: Comparison of total coliforms (a) and *Escherichia Coli (E. coli)* (b) log reductions based on the 21 days averaged hydraulic retention time (HRT) of the maturation ponds between the original train B in phase I and II, train A and train B after enhancement (phase III). The black lines indicate the standard deviation.

Correlations of these log reductions with chlorophyll-*a*, turbidity, temperature, precipitation, solar radiation and inflow concentrations were evaluated and no direct singular dependencies were found. Therefore, the log reductions seem to be influenced by a combination of several parameters. The T-test (Table SI 8.3.2) indicated that there was also no significant correlation between precipitation and log reduction. Between chlorophyll-*a* and log reduction there was a good significance correlation ($p < 0.01$) and with all other parameters there was a high significance ($p < 0.001$).

5.4.5 Bacterial community composition and algae

The overall microbial community composition varied between inlet samples and effluent samples from both trains (Figure 5.10). Due to PreT steps introduced in train A before sampling point A1, in comparison to train B, the samples from A1 and B1 clustered separately. Also, the samples of A3 and A4 were in different clusters. The effluent samples from both trains clustered together, except sample from day 1,312 and 1,328. However, no significant difference between the richness and Shannon's diversity indices was observed (Table SI 8.3.3).

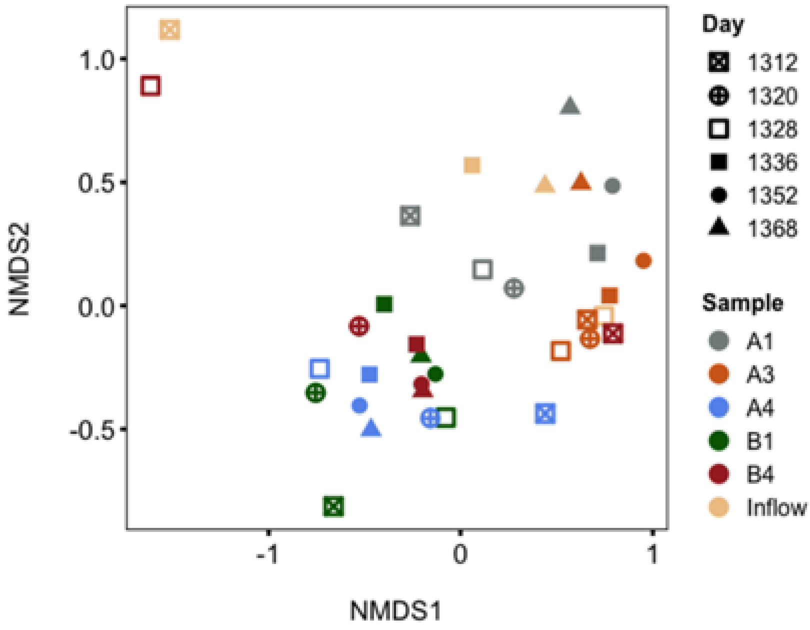

Figure 5.10: NMDS (nonmetric multidimensional scaling) plot representing the Bray–Curtis (dis)similarity of the microbial communities in the samples from train A (modified) and train B, at different sampling points and at different days.

The phyla Proteobacteria and Actinobacteriota dominated in all samples (Figure 5.11). In the inflow, A1, and A3, Actinobacteriota accounted for 47 %, 49 %, and 40 % of the relative abundance, respectively, followed by Proteobacteria (34 %, 32 %, 30 %, respectively). In A4, B1, and B4, the average relative abundance of Proteobacteria (i.e. 50 %, 50 %, 44 %, respectively) was higher than Actinobacteriota (i.e. 31 %, 27 %, 32 %, respectively).

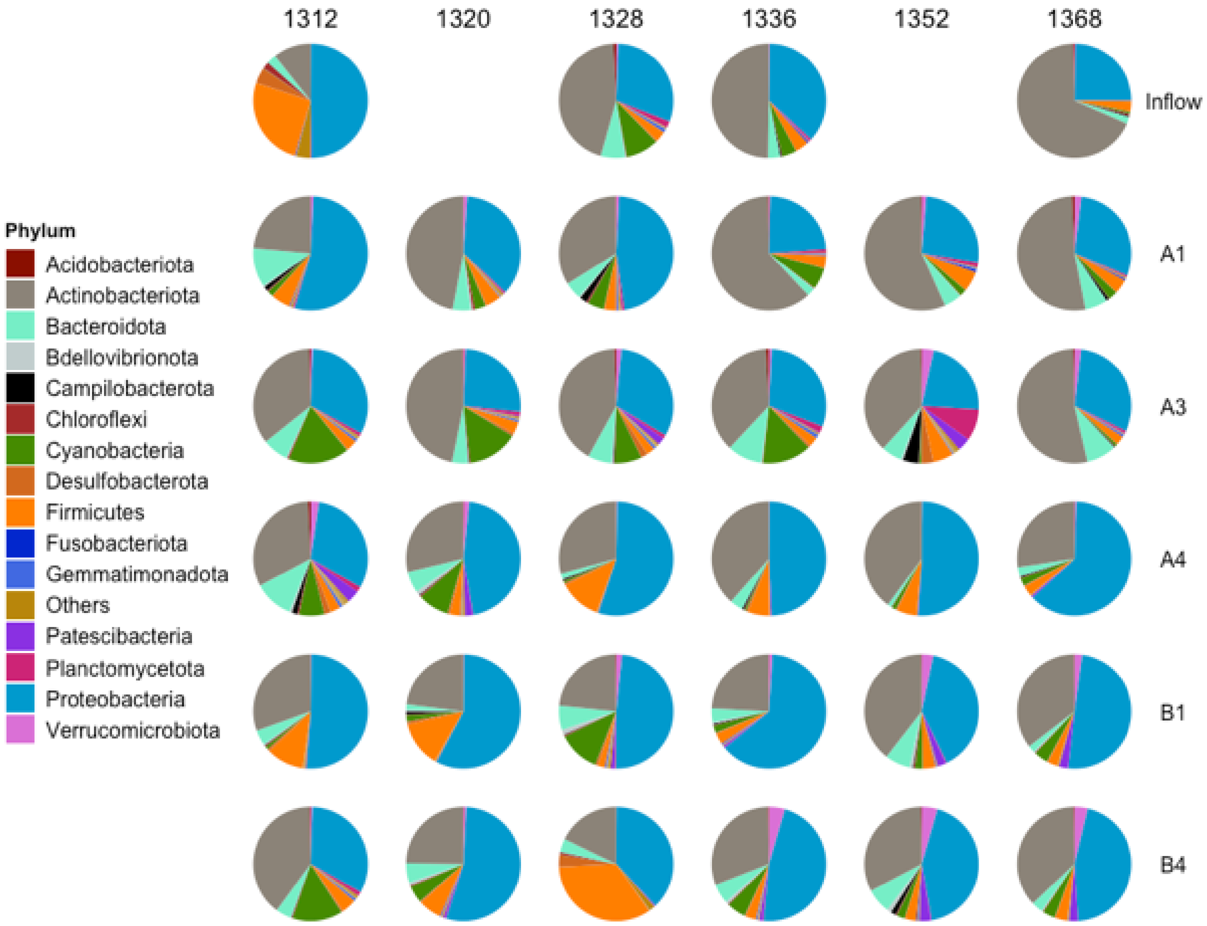

Figure 5.11: Relative abundance of the phyla found in the samples during phase III of the study. Phyla accounting for less than 5 % relative abundance are grouped as Others.

Among the top 20 genera found in all the samples from each sampling point, *Pseudomonas* was dominant at all sampling points except A3 (Figure 5.12, Figure 5.13). However, the average relative abundance of *Pseudomonas* was lower in train A (approx. 9 % for A1 and 10 % for A4) than train B (approx. 14 % for B1 and B4). In case of other genera, differences between sampling points were observed (Figure 5.12). The inflow shared 13 genera with A1, 7 with A4, and 8 with B1 and B4. In train A, 10 genera were common between A1 and A4, whereas 19 genera were common between B1 and B4. A known toxin producing Cyanobacterium, *Cyanobium PCC-6307* was detected in samples from A1 and B1 (approx. 2 % average relative abundance), however, it behaved differently in train A and train B (Figure 5.12, Figure 5.13).

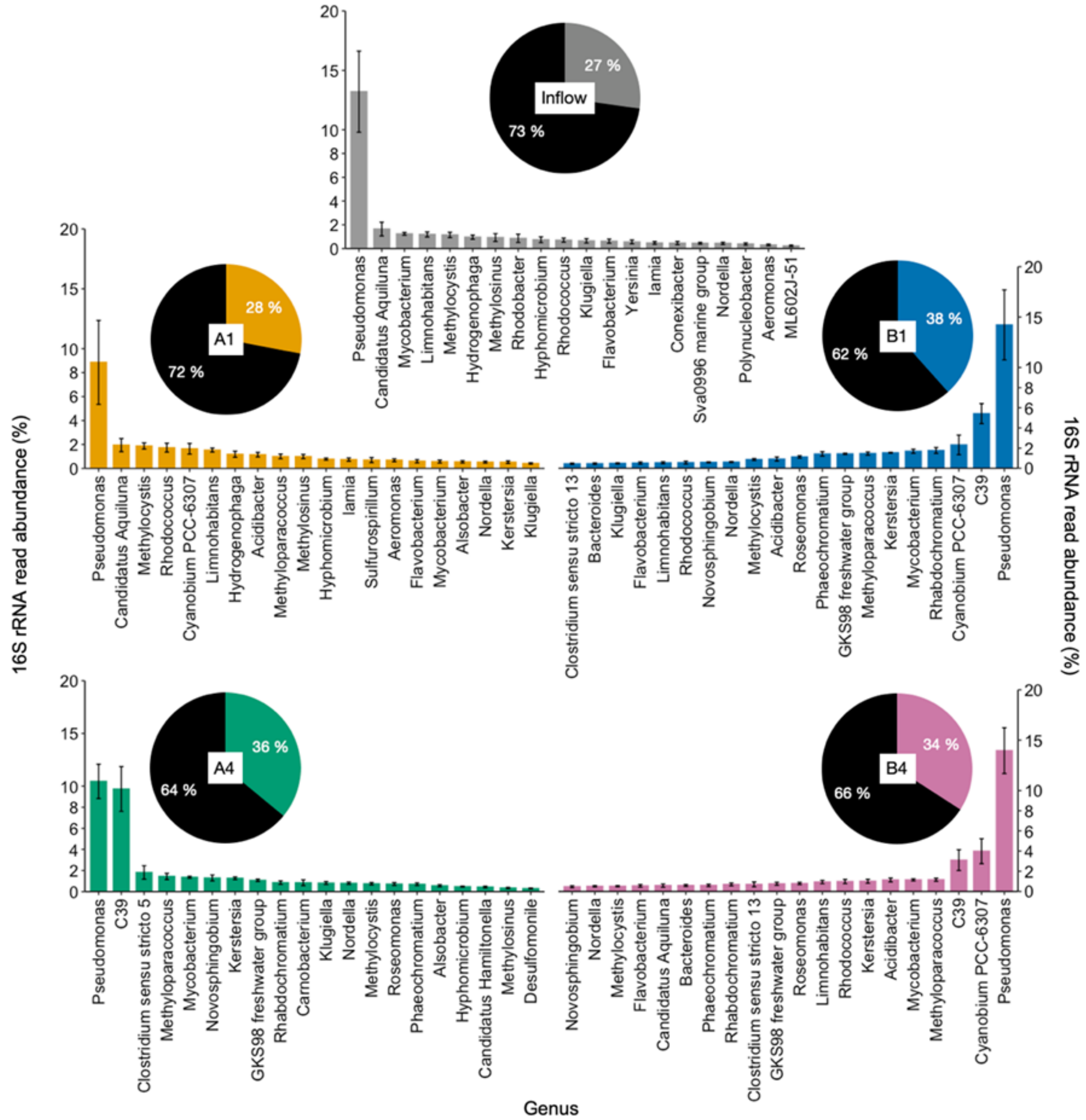

Figure 5.12: Bar plot represent top 20 genera found at each sampling point. The standard deviation across different sampling days is shown as black line extending to the minimum and maximum deviation while intersecting the bar at the mean value. Pie charts represent the sum abundance of top 20 genera (same colour as the bar plot, respectively) and abundance of the others (black).

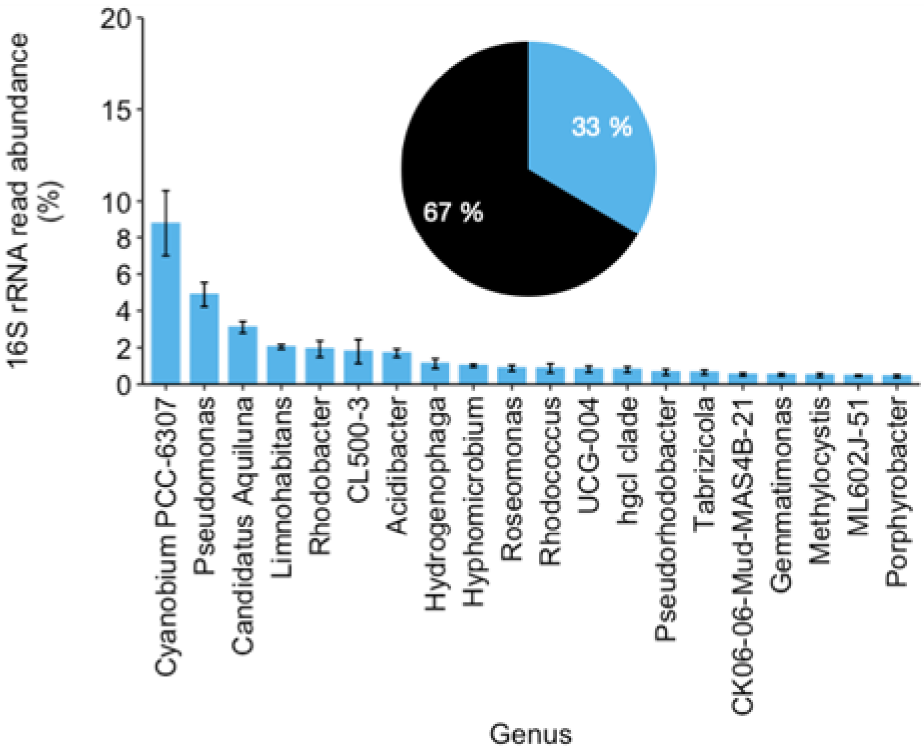

Figure 5.13: Bar plot representing the top 20 genera found in samples from pond A3. The standard deviation across different sampling days is shown as black line extending to the minimum and maximum deviation while intersecting the bar at the mean value. Pie charts represent the sum abundance of top 20 genera (same colour as the bar plot, respectively) and abundance of the others (black).

The results in Figure 5.14 a show a positive effect on the removal of chlorophyll-*a* in the enhanced train A. The average effluent concentration of 1,304 μg/L in A4 was 34 % lower than in B4. The main removal in train A had already taken place by the outflow of A3 reaching 1,375 μg/L. Pond A4 with its rock filter only decreased the chlorophyll-*a* by another 5 % which was lower than the 15 % removal in B4.

Within the biomass sample of the influent only an exiguous percentage of Planktothrix and unclassified species were found (Figure 5.14 b). At the effluent the relative abundance in the bacterial biomass increased to over 3 % with *Chlorella* being the dominant genus. Only a small percentage of biomass consisted of potentially toxic *Cyanobium* (Figure 5.14 c).

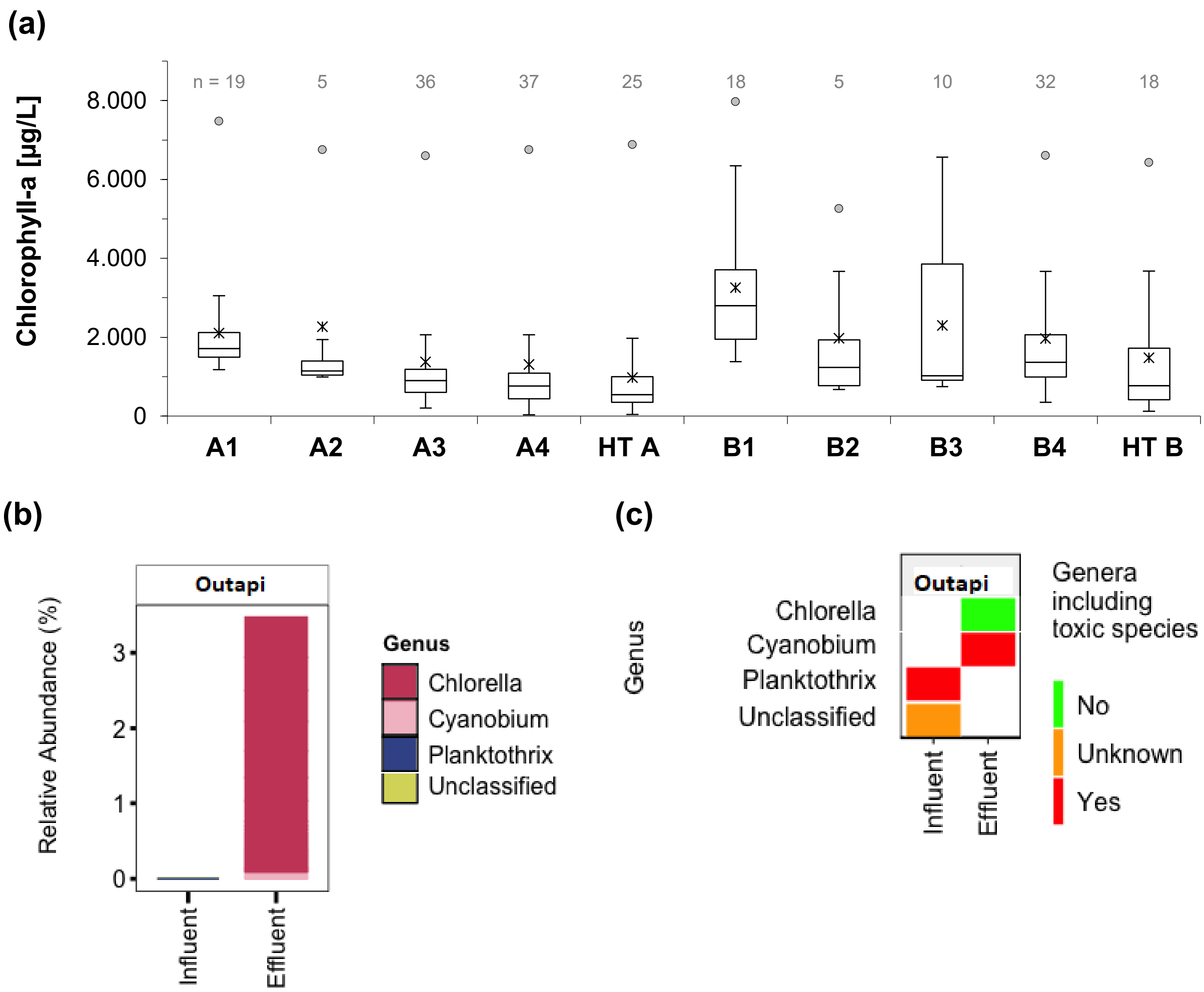

Figure 5.14: Chlorophyll-*a* concentrations (a) over both treatment trains (A1 – A4 and B1 – B4) and in the high tanks (HT A and HT B). Relative abundance and genus specification (b) of algae in the influent and effluent of the ponds. Potential toxic algae species (c) in the influent and effluent of the plant. Graphical abstract: Effluent quality of WSP for reuse purposes – impact of enhancement measures.

5.5 Discussion

5.5.1 Effluent quality - seasonal effects and influence of hydraulic load

Effluent values are influenced not only by the treatment technology but also by the surrounding environment and especially the local climate (Maiga et al., 2009; Shilton, 2005; von Sperling, 2007a). Therefore, the seasonal effects and the influence of the reduced hydraulic load were examined by comparing train B during phase I and II (B – I+II) with phase III (B – III). This information allows to better judge the effectiveness of train A and to distinguish if the lower effluent values were caused by the enhancement measures, seasonal effects or increased HRT.

Over four years, regular seasonal effects were visible in train B for different parameters (Figure 5.3). The lowest effluent values e.g. for tCOD or EC were reached at the end of the summer season. This was partially due to the highest temperatures and therefore highest microbial activity, but also due to dilution effects from rainfalls, mainly towards the end of the summer season. These effects were happening independently of the enhancement measures.

Therefore, the decline in COD concentrations in train A from day 1,200 can also be attributed to microbial and dilution effects as it coincided with the highest temperatures and towards the end of the summer with heavy rainfalls (Figure 5.5 d). Secondly, the suspension of the PreT reduced the inflow into train A from day 1,275 and resulted in a longer HRT. However, after being back to full operation on day 1,362 effluent concentrations remained low. Especially tCOD and TN were increasing slowly towards day 1,444 to only half the effluent concentrations compared to the beginning. Whether this was related to the rock filter as suggested by Rudolph et al. (2020) or it was caused only by the higher HRT cannot be finally concluded and has to be subject of future research.

The NH_4-N concentration was strongly affected by rising temperatures which became visible by the reduction of the concentrations in pond A3 during summer and their stagnation at around 1 mg/L during the interruption in phase III (Figure 5.5 g). With full inflow and decreasing temperatures, the concentrations started rising again. At the same time, the behaviour in A4 was different. The concentrations dropped to about 5 mg/L during summer but with the interruption and stagnation from day 1,275 onwards, they increased up to 25 mg/L and later reduced again to 1 mg/L before regular inflow started again on day 1,362. This indicates that within pond A4 and especially in the rock filter NH_4-N re-dissolved during stagnation periods due to anaerobic conditions. So ideally stagnation should be avoided within the rock filter. Overall, train A with the enhancements shows a 64 % better reduction of NH_4-N in the effluent (Figure 5.7 c) suggesting also a significant contribution of the enhancements to the increased performance.

The increased HRT also has a positive effect on the algae growth and pathogen reduction. According to Liu et al. (2020) varying reduction effects on different pathogens are typical and were also observed in this study. For total coliforms, hardly any change in log reduction was measured between stages B - I+II and B - III or A - III (Figure 5.9 a and Figure 5.15 a) whilst at the same time *E. coli* concentrations were further reduced. One log value was reduced with increased HRT in train B - III and one additional log value with the enhancements in train A - III (Figure 5.9 b). However, those reductions are due to the PreT and not to the rock filter (Figure 5.15 b). In contrary, in pond A4 further reduction was hindered compared to B4. For *Enterococci* the final effluent concentration was even higher with the increased HRT than with the shorter HRT. This negative effect was similar in train A and train B (Figure 5.15 c). *P. aeruginosa* concentrations were about one log value lower with the increased HRT during phase III. The main reduction was visible in the FP and the first MP (Figure 5.15 d).

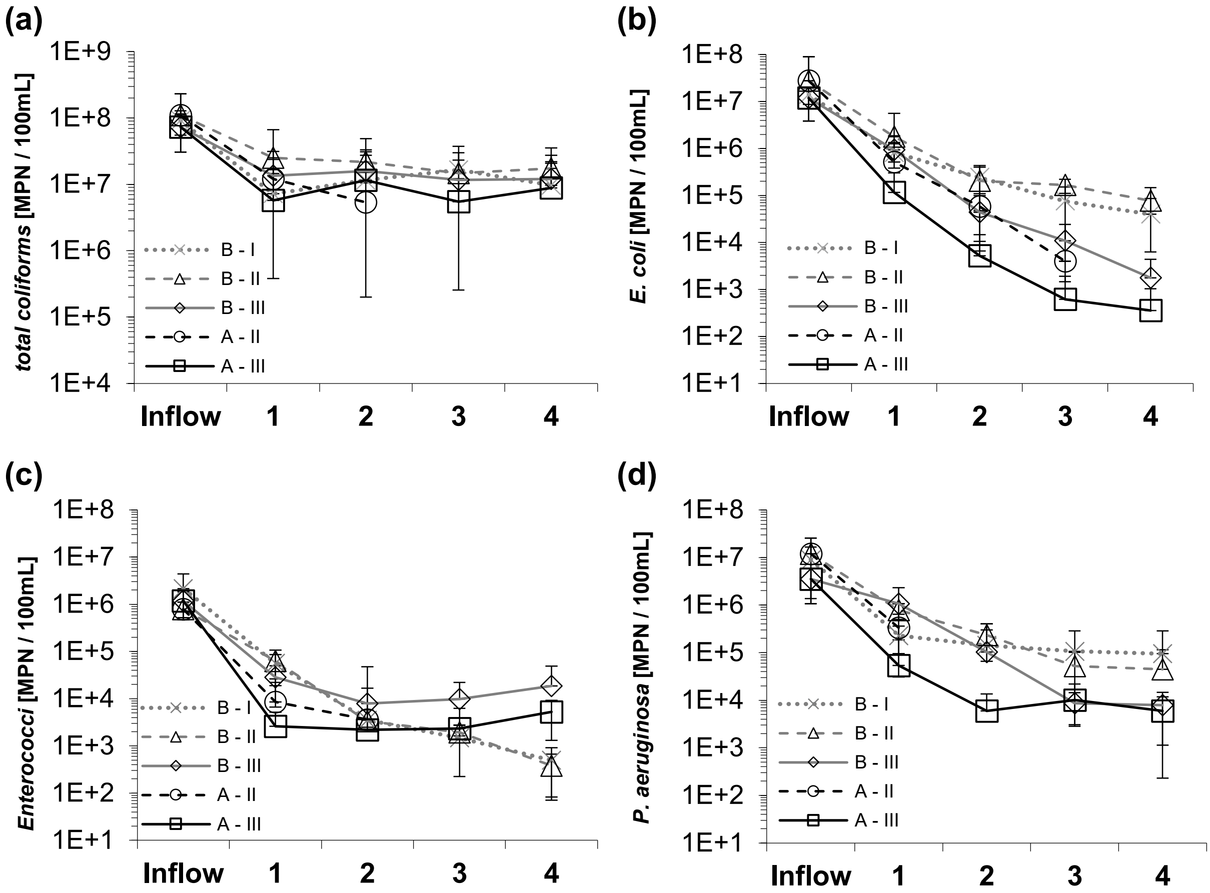

Figure 5.15: Comparison of train A and train B for each operation phase I, II and III with regard to total coliforms (a), *Escherichia Coli (E. coli)* (b), *Enterococci* (c) and *Pseudomonas aeruginosa (P. aeruginosa)* (d) at the inflow as well as ponds 1, 2, 3 and 4 of each train.

Varying effects on pathogens were also shown by Maiga et al. (2009). In their case increased solar radiation, was more harmful to *Enterococci* than to *E. coli*. For our WSP increased HRT would be associated with longer solar radiation and therefore better removal of *Enterococci*. In our system the increased HRT had better removal of *E. coli* and negative effects on *Enterococci*. Liu et al. (2020) showed the persistence of *Enterococci* with low DO whilst *E. coli* further reduced. This trend was similar in our system.

5.5.2 Impact of additional treatment technologies

The additional PreT and PosT technologies in train A showed positive impacts on the treatment performance compared to the original train B, but still did not fulfil all reuse standards. For tCOD a concentration of 100 mg/L is required in Namibia (DWAF, 2012) and 125 mg/L in Europe (EU, 2020). The best average effluent value of 317 mg/L tCOD was reached in A4 during phase III with minimum values of 127 mg/L just near the reuse standard of the EU. The increased HRT only resulted in an average outflow concentration in B4 of 415 mg/L. However, tCOD load reductions indicate an important effect of the enhancement measures in train A. The PreT and baffles already led to a 13 % better result and with the rock filter this improved to 17 %. This is considerably lower than the reported reduction of 15 – 25 % with a MS (Lazarova and Bahri, 2005; Prösl et al., 2013) and more than 60 % for a UASB (Dias et al., 2017b; Vassalle et al., 2020). However, those studies do not consider algae growth in the following ponds which add considerable new COD. If all particulate COD (pCOD) is assumed to be related to algae, the sCOD would be an alternative indicator for COD reduction. For the given plant both trains delivered a sCOD effluent quality below 90 mg/L and therefore below both reuse standards.

With regard to nutrient concentrations the Namibian reuse standard (DWAF, 2012) requires a concentration of 33 mg/L TN, which was originally aimed for effluents in areas with potential drinking water sources. In the EU there are no specific limits for nitrogen and phosphorous as these nutrients are supposed to be returned into the biological cycle (EU, 2020). In order to protect crops and soil Ayers and Westcot (1985) require severe restrictions for values above 30 mg/L TN. With average effluent values above 55 mg/L in train B no positive effect could be attributed to the increased HRT. But clear advantages of the enhancements were visible in train A with 39 mg/L TN and minimum values of 11 mg/L. This was also visible for TN effluent loads. With the PreT and the baffles, train A reached a 10 % better load reduction than train B and with the rock filter it increased to 16 %. In total, A4 had a 34 % better TN effluent concentration than B4.

For irrigation purposes the Namibian code of practice (DWAF, 2012) allows a maximum of 15 mg/L TP which was reached by both trains with 11 mg/L. But these values were even higher than the average inflow concentration of 10 mg/L. This is due to the effect of high evaporation and proves Buchanan et al. (2018) who showed, that there is only limited phosphorous removal in WSP. So only looking at the rising TP concentrations (Figure 5.6 d) there could be the impression that there was no removal at all. But this is clearly not the case and can be shown with the daily TP loads leaving each pond (Figure 5.7 d). The total inflow load was 7.8 (± 2.3) kg/d and was equally distributed between trains A and B with 3.6 (± 1.5) kg/d going into train A and 4.1 (± 1.6) kg/d to train B. The first ponds in each train showed an almost parallel removal with 10 % in A1 and 9 % in B1 as well as 22 % in A3 and 23 % in A4. But the final effluent turned, whilst A4 improved up to 36 % and 2.3 (± 1.5) kg/d B4 only reached 18 % with 3.4 (± 1.6) kg/d.

5.5.3 Composition of pathogens and their reduction in WSP

The different treatment technologies have also various effects on the pathogens. Their composition and concentrations were affected in various ways. Over the whole WSP total coliforms were less dynamic than COD and nutrients. They were hardly reduced and there was no significant difference between the effluent of the enhanced train A and the original train B. In train A the rock filter even had a negative effect with a slight recontamination of total coliforms.

Other pathogens showed a different behaviour. *E. coli* were constantly reduced, in B4 to 4.0×10^4 MPN/100mL during phase I and 1.8×10^3 MPN/100mL during phase III, and to 3.6×10^2 MPN/100mL in A4. Therefore, the increased HRT had a first positive effect on *E. coli*, resulting in a reduction of one log value and further enhancement measures improved this value by another log value. However, the observed reduction of *E. coli* was more related to the effects of the ponds in train A rather than the rock filter. In A4 the rock filter had no additional positive effect on *E. coli* concentrations. In contrary, it hindered further reduction as it happened from B3 to B4. Dias et al. (2017b) also reported the best *E. coli* removal of 2.2 log values by ponds, however, their granular rock filter still achieved a removal of 1.0 log value. Nevertheless, the reductions that were obtained here reached the required EU effluent value of 1,000 MPN/100mL (EU, 2020) for fodder irrigation with train A and were just missed with train B.

This study additionally evaluated the removal of *Enterococci* and *P. aeruginosa*. Both pathogens behaved quite differently compared to the standard indicator *E. coli*. *Enterococci* almost continuously reduced over the different ponds during phase I down to a concentration of 5.0×10^2 MPN/100mL in B4. During phase III, the steepest decrease occurred in the FP in both treatment trains and there was hardly any change in the MPs. Also, there was no significant difference between the two trains with B4 reaching 1.9×10^4 MPN/100mL and A4 5.3×10^3 MPN/100mL. *Enterococci* are best reduced with the PreT and the baffles in A1. It seems that the reduction occurs mainly through sedimentation rather than sunlight. Differences in the inactivation of *Enterococci* compared to the traditional indicator *E. coli* were also reported by Liu et al. (2020). *P. aeruginosa* followed a similar pattern with the difference that during phase III there was some further reduction in the first MP (up to the effluents of A2 and B2) before concentrations stagnated. Also, the final concentrations of *P. aeruginosa* in the effluent were almost identical between A4 and B4. A positive effect of the PreT and the baffles on the reduction of *P. aeruginosa* was also visible. So far, there is only limited research on the behaviour of *P. aeruginosa* in WSP and thus this study provides first novel insights into the behaviour of these bacteria in WSP. Søberg et al. (2019) reported that the trends for *P. aeruginosa* correlated strongly with *Enterococci* in storm water bioretention systems. This could not be confirmed with this research.

The effects of PreT, baffles and rock filter vary depending on each specific parameter, as train A has better effluent values due to the PreT and baffles. For the rock filter in A4 there is no evidence of any positive effect. In contrary for *E. coli* and *Enterococci* it has a stagnant or negative effect. This research supports the findings of Liu et al. (2020) that *E. coli* should not be the only indicator organism to evaluate pathogen removal.

5.5.4 Microbial community composition and algae development

Not only the concentrations of specific pathogens, but also the whole microbial community and algae can be affected by the different treatment technologies. Especially algae compose a considerable part of the pCOD. Given the local circumstances the best way to estimate the algae content in the ponds was through the concentration of chlorophyll-*a*. This does not cover all types of algae but mainly cyanobacteria that could potentially be toxic (Vidal et al., 2021). Whilst the average concentrations in train A were between 1,000 and 2,000 μg/L, train B showed higher values of up to 3,300 μg/L of chlorophyll-*a*. Besides the positive effect of added biomass to the soil, algae also increase the tCOD (section 5.5.2). Another negative effect is the risk of blockages in drip irrigation systems. Therefore, a rock filter was investigated for its removal of the algae content (Rudolph et al., 2020). However, no significant chlorophyll-*a* removal was visible when comparing chlorophyll-*a* concentrations between A3 (1,375 μg/L) and A4 (1,305 μg/L). The main effects on the algae were related to the PreT, the floating baffles and the removed sludge, which all happened upfront or in pond A1. Only long-term observations will show if the rock filter develops a positive algae removal with increasing growth of biofilm on the rock surface.

Another important aspect for reuse of algae containing water is their toxicity. The results of this study show that only a very small percentage of the cyanobacteria were potential toxin producers. *Cyanobium PCC-6307* was not dominant in samples from A4 but the second most dominant genus in samples from B4. Interestingly, *Candidatus Aquiluna*, a photoheterotroph (Kang et al., 2012), seemed to be replaced from the second most abundant in the inflow by either *Cyanobium PCC-6307* or *C39* genus of the *Rhodocyclaceae* in the samples of A4 and B4. *C39* has been mainly detected in freshwater bodies (Cannon et al., 2017; Carney et al., 2015). A high amount of algae was observed in samples from A4 and B4, which might explain the high abundance of genus *C39*. A previous study emphasized a plausible association between other algae and genus *C39* (Cannon et al., 2017). However, whether toxins are released depends the environment and stress (Vidal et al., 2021). A more detailed evaluation of potential toxins was not possible and should be further examined in future research.

During the course of operation of both trains, sequential change in the microbial community occurred, however, no change in the dominant phyla (i.e. Proteobacteria and Actinobacteriota) was observed (Figure 5.10, Figure 5.11). Both of these phyla include microorganisms (such as *Pseudomonas*, *Actinobacteria*) known for enhanced phosphorous removal (Lee et al., 2002), which might explain the reduction of the TP load in both trains. There also seemed to be a positive effect of PreT on the reduction of *Mycobacterium* (Figure 5.12), which is a re-emerging pathogen (Taylor et al., 2001) but no impact on *Pseudomonas*. This underlines the persistent nature of *Pseudomonas* in the pond system.

5.6 Conclusions

This research detailed different effects and operating conditions on the behaviour and effluent quality of a WSP system in Namibia over a four-year period. The main findings are:

- Regular seasonal effects on different parameters are visible in the overloaded and enhanced train over four years of operation.
- Increasing HRT and reduced hydraulic load have a positive effect on the removal of *E. coli* and the required 40 days HRT is achieved with both trains in operation.
- Mechanical and anaerobic biological PreT such as micro sieve and UASB have a positive effect on the removal of COD, TSS and to some extent pathogens.
- Within the first year of operation the rock filter did not show any additional removal of algae compared with the original train. Long-term operation and monitoring will show if more biofilms are growing on the rocks and if they will improve removal.
- Best measures to reduce the algae concentrations are emptying of the sludge in the first pond, installing PreT and baffles in the FP.
- The studied pathogens exhibit different behaviour: *E. coli* were reduced, *P. aeruginosa* stagnated and *Enterococci* levels increased. The main pathogen reduction happened during PreT and in the first pond (FP enhanced with baffles) and not in the maturation ponds.
- Future detailed research is needed into the pathogen removal capacity of WSP as *E. coli* do not seem to be the best indicator for broader pathogen reduction.
- With enhancement WSP reach the new EU water reuse standard for *E. coli*.
- High tCOD and TN effluent values currently do not meet Namibian and European reuse standards. But a large portion of the tCOD consists of algae which add, as long as they are not toxic, valuable biomass to the barren soil. Also TN is a valuable fertilizer and depending on the selected crop this can be an important asset especially if additional water from other sources with low nutrients is used.

6 Final conclusions and outlook

6.1 Final conclusions

The results of this dissertation proved that WSP systems can treat municipal wastewater and their effluent is fit for the purpose of agricultural irrigation, especially with slight or moderate restrictions and applying a multi barrier approach. Enhanced WSP can partially fulfil Namibian (DWAF, 2012) and the newly published European (EU, 2020) reuse standards.

Depending on the different technologies, effects vary considerably. The PreT and the baffles have a strong positive effect on the removal of tCOD, TN and NH_4-N loads. For the rock filter, load reduction of tCOD and TN are also positive but for some microbiology contaminants such as *E. coli* and *P. aeruginosa* the effect on the reduction of their concentrations was minimal or even negative. Similar observations were made for *Enterococci*.

All these technical enhancements have to be judged against the traditional setup with anaerobic ponds that include previsions (e.g. ramps) for sludge removal. The advantage of such a PreT is the simplicity in handling and that it does not require external power supply. However regular sludge removal has to be ensured (Letshwenyo et al., 2020) and climate gas emissions are not reduced (Hernandez-Paniagua et al., 2014).

Alternatively to the researched technologies, further enhancements must be considered. For higher COD removal, especially sCOD, and nutrients, especially nitrogen, aeration is an option, but would need either solar or conventional power supply. Additional environmental improvement would be achieved with a combined heat and power unit if the capacity of the UASB was extended and more biogas produced. According to the Namibian reuse standard (DWAF, 2012) an HRT of 40 days is required for pathogen removal. At the same time this results in high evaporation losses and hinders lower concentrations for other parameters. Therefore it is better to allow shorter HRT and to focus on specific removal rates for example *E. coli* as with the new water reuse standards of the EU (2020).

The main research outcomes according to the research questions and objectives are summarised as follows:

1. WSP in north-central Namibia are in different states with regard to layout, design, operation and maintenance, microbial communities and due to their nature-based approach depend largely on seasonal effects:

    - The existing WSP in north-central Namibia are all based on the typical layout of FP and MP followed by EP and no water reuse scheme has so far been researched and applied. Depending on the design, connected population and date of construction the surface areas per capita and HRT differ substantially with no correlation. Operation and maintenance have been neglected with all systems and therefore none of the WWTP currently complies with the national reuse standard mainly due to high tCOD concentration which originate from algae.

- The microbial communities in the various WSP develop over the different treatment trains. Amongst the pathogens *Acinetobacter* dominate the inflow and *Mycobacterium* the outflow samples. In relation to all bacteria Cyanobacteria accounted only for up to 25 % with *Synechococcus* dominating. The genera found included also potentially toxic species which could have negative effects during irrigation.
- Seasonal effects are clearly visible over the four years of the research. Best effluent values reached are at the end of the summer season. This is mainly related to rising temperature and higher microbial activity. The increased HRT also has a positive effect on the algae growth and *E. coli* reduction whilst there is hardly an effect on *Enterococci*.

2. The implemented pre-treatment and post-treatment enhancements have various effects on the biological, chemical and physical parameters of the irrigation water:
    - The anaerobic biological and mechanical pre-treatment, introduction of baffles and sludge removal have their main impact on the removal of pCOD and TSS. Another positive effect shows the reduction of pathogens, *E. coli* and *Enterococci* had one log_{10} unit better effluent values than without the enhancement. This is particularly interesting as disinfection is not described in literature as task of facultative ponds.
    - For the performance of the rock filter longer research is necessary. At a first glance it seems responsible for the reduction of tCOD and TN in the last pond but this could also be due to higher retention time and therefore needs further research. During summer months the NH_4-H concentrations are reduced in other ponds but the contrary is visible during a stagnation period in the last pond with the rock filter. Due to anaerobic conditions NH_4-H re-dissolves, so water stagnation should be avoided within the rock filter and constant flow assured. Also, no significant chlorophyll-*a* reduction is visible with the effluent of the filter. The main effects on algae are related to the PreT, baffles and sludge removal in the FP.
    - Different enhancement technologies have various effects on the pathogen composition as well as their concentrations. Total coliforms are hardly reduced and there is no significant difference between the original and the enhanced train. *E. coli* in contrary are mainly reduced with the PreT, FP and MP whilst the rock filter shows no further reduction. *Enterococci* are best reduced with the PreT and the baffles in the FP but then there is hardly any further reduction in the MP. As *P. aeruginosa* again behave differently it is suggested that *E. coli* should not be used as the only indicator organism to evaluate pathogen removal.

3. Compliance with the Namibian and European water reuse standards is difficult, mainly due to high tCOD concentration which are clearly caused by algae:

    - High tCOD concentrations in the effluents are directly related to algae measured by the chlorophyll-*a* concentration which correlate linearly with the pCOD. After deduction of the algae concentration tCOD without algae (59 – 339 mg/L) consist mainly of sCOD (47 – 251 mg/L).
    - The main obstacle for water reuse in relation to the Namibian and European water quality standards are the required tCOD concentrations and the pathogen reduction. In their original state none of the WSP systems in north-central Namibia corresponded to the *E. coli* requirements but with the enhancements this was possible. With the PreT and PostT measures the maximum value of tCOD without algae (144 mg/L) almost reached the EU standard (125 mg/L). Therefore, it should be considered to adapt the maximum tCOD if water is to be reused from WSP.
    - Operation and maintenance are neglected with all existing WSP in north-central Namibia. Even simple tasks such as site inspection with cutting of vegetation or control of fencing are randomly implemented, not to mention sludge removal especially from anaerobic ponds. Therefore, the enhancement with higher technology requires a regular presence on site and also the necessary skills and materials such as tools and spare parts. This can be a chance and at the same time a risk for WSP if they are upgraded to produce reuse water for agricultural irrigation.

6.2 Future work and outlook

At the pilot plant in Outapi it was possible to research different enhancement measures for PreT and PostT which were adapted to the local conditions of the existing WSP system. One important aspect was the availability of power from the local supply grid. In other locations this is not possible and therefore different technologies have to be considered, either without power requirements such as AP or for local power generation such as photovoltaic or a generator using locally produced biogas. For the PreT well-designed anaerobic ponds would allow reduction of suspended solids and organic matter without power requirements. However, access ramps, awareness amongst operators for regular sludge removal and sludge treatment have to be an integral part of the design. This would overcome the need for power but would not change the greenhouse gas emissions. Further applied research is needed to capture greenhouse gas emissions considering the local climatic conditions.

Another technology, aeration of WSP, has so far not been applied on WSP in sub-Saharan countries. With solar aerators this would be possible even without grid connection. The application of this technology needs to be field tested under local conditions and technical as well as operational requirements for local staff established. An important aspect will be to test the durability and the supply chain of spare parts.

Other PostT technologies to remove suspended solids and algal matter should be compared with the installed rock filter. On the one hand long term effects of the rock filter have to be measured and evaluated and on the other hand other technologies such as sand filter or pile cloth media filtration can be compared to further reduce algae and pathogen concentrations.

Instead of removing the algae, their use as soil enhancement by adding biomass and improving the water holding capacity should also be considered. However, this requires further research in drip irrigation techniques so that pipes and drippers are not blocked or washed out without chemical additives. At the same time, it has to be assured that no toxic algae or cyanobacteria are applied. It is therefore necessary to develop low-cost methods for microbial analytics which can be used either by local personnel on site or in close by laboratories.

All the above mentioned technologies need practical research in full scale plants but at the same time recent advancements in modelling allow simulation of hydraulics and microbial processes in the WSP as well as the different PreT and PostT technologies. Therefore, simulation and modelling should be included and validated with further projects on WSP.

With regard to the national Namibian reuse standards it became evident that they are mainly focused on effluents from advanced WWTP such as activated sludge systems, which exist only in large cities. So, a review of the national code of practice is needed to include reuse from pond systems as an integral part and to adapt design recommendations accordingly. On an international level monitoring on the implementation of the new EU regulation and comparison of the transfer into national legislation in all EU member countries is required.

Especially with growing urban development the commercial and industrial wastewater components are becoming more relevant and therefore their specific or separate collection and treatment has to be considered to avoid contamination of the reuse water. As research on and analyses of contaminants of emerging concern advances this should be followed closely, but not prevent further WSP enhancements.

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8 Appendix

8.1 Supplementary information paper I

Table SI 8.1.1: Total population, connected population, pond sizes, hydraulic retention time (HRT) and pond types of the nine towns in northern Namibia with their population equivalent inflow loads as well as their surface loading rates of the first pond and volumetric loading rates of the anaerobic pond.

Town		A	B	C	D	E	F	G	H	I
population (1)	[-]	10,000	7,000	27,000	19,000	50,000	8,000	12,000	8,000	2,300
estimated connection	[%]	60 (2)	50 (2)	40 (1)	50 (1)	40 (1)	70 (1)	60 (2)	70 (1)	30 (1)
estimated inflow	[m³/d]	720(2)	432(2)	1,200 (1)	1,080 (1)	2,160 (1)	660 (1)	800(2)	672 (1)	83 (1)
constructed (1)	[yyyy]	1999	1999	1980's	1970's	2007	1983	2005	1974	2015
upgraded (1)	[yyyy]	2015	2008	2015	-	2012	2015	2018	2008	-
pond surface area (3)	[m²]	50,000	9,300	280,000	17,000	200,000	31,000	41,000	7,000	17,000
pond surface area per capita connected	[m²/cap]	8	3	26	2	10	6	6	1	25
pond volume	[m³]	60,000(1)	17,000(1)	360,000(1)	25,500(4)	294,000(4)	51,000(4)	55,000(1)	8,300(1)	25,000(4)
estimated HRT	[d]	83	39	300	24	136	77	69	12	302
population equivalent (PE) inflow load										
tCOD	[g/PE/d]	121	56	101	53	127	86	92	76	60
sCOD	[g/PE/d]	53	23	39	28	51	32	32	37	25
TSS	[g/PE/d]	28	16	25	23	41	26	32	15	27
TN	[g/PE/d]	12	7	8	6	12	10	9	8	6
NH_4-N	[g/PE/d]	10	5	7	5	12	9	6	7	4
TP	[g/PE/d]	1.6	0.7	0.9	0.8	1.4	1.3	1.1	1.0	0.7
PO_4-P	[g/PE/d]	1.2	0.6	0.7	0.6	1.2	0.9	0.8	0.7	0.4
surface loading rate of first pond (FP or AP)										
tCOD	[kg/ha/d]	3014	699	403	280	715	2498	268	1161	413
BOD_5(5)	[kg/ha/d]	1507	349	202	140	357	1249	134	580	207
volumetric loading rate of anaerobic pond										
tCOD	[kg/m³/d]	0.09	0.02	0.01	-	0.02	0.07	-	-	0.01
BOD_5(5)	[kg/m³/d]	0.04	0.01	0.01	-	0.01	0.04	-	-	0.01
WSP										
treatment trains		1	2	2	1	2	2	2	1	1
Anaerobic (AP) (6)		2	2	2	0	4	2	0	0	1
Facultative (FP) (6)		1	2	5	1	2	2	2	1	1
Maturation (MP) (6)		3	4	6	7	2	4	6	4	1
Evaporation (EP) (6)		1	1	1	1	0	2	1	1	1

(1) Based on operator estimation/ information
(2) Based on inflow data
(3) Calculated by satellite images
(4) Calculated with average depth (AP: 3.5 m; FP: 1,5 m) (von Sperling, 2007a)
(5) Calculated with a BOD_5/COD ratio of 0.5 (Tchobanoglous et al., 2014)
(6) Total number of ponds at the respective site

Table SI 8.1.2: Average physical, chemical and biological parameters, presented as mean values, of nine waste stabilization pond system in Namibia. WSP A – WSP I are referring to different towns.

		Physical parameters				Chemical parameters								Biological parameters			
	Parameter	EC	Turbidity	TS	TSS	pH	tCOD	sCOD	pCOD	TN	NH_4^+-N	TP	PO_4^{3-}-P	*E. Coli*	t. Coliforms	Enterococci	Chlorophyll-*a*
WSP	Unit	µS/cm	FNU	mg/L	mg/L	-	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	MPN/100 mL	MPN/100 mL	MPN/100 mL	µg/L
A	influent ∅ (n)	2004 (3)	384 (3)	1505 (2)	233 (2)	7.2 (3)	1005 (3)	440 (3)	565 (3)	98.1 (3)	86.8 (3)	13.1 (3)	9.8 (3)	6.6E+06 (3)	3.9E+07 (3)	1.5E+06 (2)	n.a.
	effluent ∅ (n)	1842 (3)	129 (3)	1463 (2)	91 (3)	9.1 (3)	289 (3)	102 (3)	187 (3)	21.7 (3)	2.2 (3)	10.5 (3)	10.3 (2)	1.5E+03 (3)	8.4E+05 (3)	1.5E+03 (2)	828 (2)
B	influent ∅ (n)	691 (3)	173 (3)	433 (2)	133 (3)	6.2 (3)	466 (3)	191 (3)	274 (3)	55.3 (3)	42.5 (3)	6.1 (3)	4.8 (3)	1.4E+07 (3)	5.7E+07 (3)	8.3E+05 (2)	n.a.
	effluent ∅ (n)	749 (3)	124 (3)	521 (2)	110 (3)	8.7 (3)	274 (3)	66 (3)	208 (3)	46.8 (3)	28.8 (3)	8.7 (3)	3.7 (3)	4.7E+04 (3)	4.9E+05 (3)	3.6E+03 (2)	784 (2)
C	influent ∅ (n)	975 (3)	328 (3)	676 (2)	210 (3)	7.1 (3)	840 (3)	322 (3)	518 (3)	67.9 (3)	61.5 (3)	7.7 (3)	6.1 (3)	8.7E+06 (3)	9.2E+07 (3)	2.1E+06 (2)	n.a.
	effluent ∅ (n)	1249 (3)	39 (3)	878 (2)	33 (3)	7.7 (3)	156 (3)	84 (3)	72 (3)	21.3 (3)	18.0 (2)	9.0 (3)	8.7 (3)	9.3E+04 (3)	9.5E+04 (3)	4.1E+03 (2)	251 (2)
D	influent ∅ (n)	2555 (2)	203 (2)	1779 (2)	193 (2)	7.6 (2)	440 (2)	235 (2)	206 (2)	47.8 (2)	40.5 (2)	6.3 (2)	4.7 (2)	1.7E+07 (2)	2.8E+07 (2)	1.0E+06 (2)	n.a.
	effluent ∅ (n)	2505 (2)	153 (2)	1987 (2)	93 (2)	9.0 (2)	274 (2)	88 (2)	186 (2)	29.6 (2)	9.3 (2)	4.7 (2)	2.1 (2)	1.5E+04 (2)	1.5E+05 (2)	7.3E+03 (2)	193 (1)
E	influent ∅ (n)	1319 (3)	470 (3)	972 (2)	345 (3)	7.1 (3)	1059 (3)	424 (3)	636 (3)	81.2 (3)	98.1 (3)	11.9 (3)	9.8 (3)	3.9E+06 (3)	3.1E+07 (3)	2.1E+06 (2)	n.a.
	effluent ∅ (n)	3150 (1)	251 (3)	7033 (2)	170 (3)	10.0 (3)	701 (2)	209 (2)	492 (2)	57.4 (3)	2.8 (2)	18.7 (3)	15.8 (3)	1.4E+04 (3)	1.5E+04 (3)	8.5E+04 (2)	403 (1)
F	influent ∅ (n)	1148 (3)	361 (3)	866 (2)	214 (3)	7.2 (3)	719 (3)	269 (3)	451 (3)	80.1 (3)	71.9 (3)	11.0 (3)	7.7 (3)	6.3E+06 (3)	8.8E+07 (3)	3.8E+06 (2)	n.a.
	effluent ∅ (n)	1991 (3)	117 (3)	1485 (2)	66 (3)	8.8 (3)	298 (3)	136 (3)	162 (3)	29.5 (3)	14.0 (3)	11.4 (3)	9.0 (3)	2.2E+04 (2)	8.1E+05 (3)	4.6E+03 (2)	188 (2)
G1	influent ∅ (n)	936 (3)	376 (3)	n.a.	305 (3)	6.5 (3)	732 (3)	227 (3)	504 (3)	71.0 (3)	50.3 (3)	8.1 (3)	5.8 (3)	9.8E+06 (3)	1.0E+08 (3)	1.4E+06 (2)	n.a.
	effluent ∅ (n)	770 (3)	335 (3)	n.a.	121 (3)	8.0 (3)	403 (3)	87 (3)	315 (3)	58.5 (3)	29.8 (3)	8.2 (3)	2.7 (3)	9.0E+04 (3)	1.5E+07 (3)	1.0E+03 (2)	n.a.
G2	influent ∅ (n)	999 (3)	418 (3)	693 (2)	216 (3)	7.1 (3)	820 (3)	303 (3)	518 (3)	76.6 (3)	56.2 (3)	9.6 (3)	7.9 (3)	1.8E+07 (3)	5.9E+07 (3)	9.2E+05 (2)	n.a.
	effluent ∅ (n)	934 (3)	278 (3)	732 (2)	119 (3)	8.6 (3)	409 (3)	96 (3)	313 (3)	52.6 (3)	31.9 (3)	11.8 (3)	1.9 (3)	2.0E+04 (3)	1.2E+07 (3)	5.2E+02 (2)	616 (3)
G3	influent ∅ (n)	864 (3)	375 (3)	n.a.	277 (3)	6.7 (3)	741 (3)	278 (3)	463 (3)	66.2 (3)	53.9 (3)	8.6 (3)	5.5 (3)	1.7E+07 (3)	5.7E+07 (3)	8.3E+05 (2)	n.a.
	effluent ∅ (n)	871 (3)	143 (3)	n.a.	31 (3)	7.8 (3)	258 (3)	84 (3)	174 (3)	34.1 (3)	19.2 (3)	9.0 (3)	6.0 (3)	1.7E+02 (3)	2.3E+06 (3)	3.5E+03 (2)	610 (3)
H	influent ∅ (n)	914 (3)	288 (3)	451 (2)	125 (2)	7.6 (3)	635 (3)	311 (3)	324 (3)	67.6 (3)	57.4 (3)	7.9 (3)	6.1 (3)	1.5E+07 (3)	5.1E+07 (3)	6.4E+05 (2)	n.a.
	effluent ∅ (n)	560 (3)	314 (3)	907 (2)	171 (3)	9.2 (3)	443 (3)	93 (3)	350 (3)	35.2 (3)	5.6 (3)	6.3 (3)	2.9 (3)	1.5E+04 (3)	3.6E+06 (3)	2.8E+04 (2)	1608 (2)
I	influent ∅ (n)	686 (3)	195 (3)	610 (2)	222 (2)	6.9 (3)	504 (3)	205 (3)	299 (3)	49.4 (3)	35.2 (3)	5.6 (3)	2.9 (3)	1.1E+07 (3)	9.0E+07 (3)	1.9E+06 (2)	n.a.
	effluent ∅ (n)	769 (3)	453 (3)	609 (2)	132 (2)	9.7 (3)	251 (3)	84 (3)	229 (3)	23.6 (3)	2.5 (3)	5.7 (3)	4.9 (2)	2.8E+04 (3)	1.7E+05 (3)	1.5E+03 (2)	448 (2)

∅ = mean, n = number of measurements, n.a. = not analysed, EC = electrical conductivity, TS = total solids, TSS = total suspended solids, tCOD = total (homogenized) chemical oxygen demand, sCOD = soluble (0,45µm filtered) COD, pCOD = particulate COD, TN = total nitrogen, TP = total phosphorous, MPN most probable number.

8.2 Supplementary information paper II

Table SI 8.2.1: Details about the town Outapi and its original wastewater pond system

population (2018)	12,000	
annual population growth	9.3 %	
sewer system	gravity sewer	
people connected	about 58 %	
design load	2,500 PE	
load connected	7,000 PE	
treatment technologies	pond system with two treatment lines (á 1 facultative pond and 3 x maturation pond)	
average daily inflow	800 m³/d (dry weather)	
surface area	4.05 ha for treatment	
	line A [m²]	line B [m²]
facultative pond (1)	11,400	11,400
maturation pond (2)	5,200	5,300
maturation pond (3)	1,800	1,600
maturation pond (4)	1,600	2,300
evaporation pond	40,900	
depth	line A [m]	line B [m]
facultative pond (1)	1.5	1.5
maturation pond (2)	1.3	1.3
maturation pond (3)	1.5	1.3
maturation pond (4)	1.3	1.1
evaporation pond	0.5	
volume	line A [m³]	line B [m³]
facultative pond (1)	16,100	16,000
maturation pond (2)	6,300	6,100
maturation pond (3)	2,400	1,900
maturation pond (4)	1,700	2,100
evaporation pond	20,400	

Table SI 8.2.2: Physical, chemical and biological parameters of the wastewater pond system in Namibia. Values (day 1 – day 527) present the two treatment trains: Train A with pre-treatment and train B in its original stage.

		Monitoring data enhanced train												Monitoring data original train					
		Buffer effluent			UASB effluent			Micro sieve effluent			A1 effluent			Untreated wastewater			B1 effluent		
Parameter	**Unit**	**mean**	**sd**	**n**	**Mean**	**sd**	**n**	**Mean**	**sd**	**n**	**Mean**	**Sd**	**n**	**Mean**	**sd**	**n**	**Mean**	**sd**	**n**
Physical characteristics																			
EC	µS/cm	968	95	52	1070	76	81	962	92	68	888	68	68	956	110	78	843	58	71
Turbidity	FNU	484	193	49	266	96	75	365	112	65	337	100	63	351	120	75	478	129	68
TS	mg/L	995	377	23	615	143	25	690	113	25	693	56	21	694	98	27	728	72	21
TSS	mg/L	483	319	39	152	89	57	198	71	50	171	68	35	220	81	56	197	57	40
Chemical characteristics																			
pH	-	6.8	0.7	53	6.7	0.6	82	6.9	0.6	69	8.4	0.6	69	7.0	0.4	79	8.1	0.4	69
tCOD	mg/L	1044	376	45	477	174	68	740	142	58	465	93	55	749	153	70	588	123	59
sCOD	mg/L	325	47	44	171	126	66	306	58	58	89	28	56	301	59	66	89	22	59
pCOD	mg/L	671	291	43	313	136	64	431	128	56	380	92	54	444	133	66	500	113	58
TN	mg/L	83.4	13.0	39	81.0	10.0	61	78.3	11.9	53	58.5	11.4	48	76.0	12.5	68	70.4	7.7	51
NH_4^+-N	mg/L	67.6	8.0	36	72.8	7.2	58	65.6	7.7	50	33.6	7.5	45	64.7	10.8	66	38.1	4.2	49
TP	mg/L	10.9	1.8	36	10.4	1.1	58	9.9	1.3	50	10.8	1.0	46	9.8	1.3	65	10.3	0.9	48
PO_4--P	mg/L	7.9	1.4	33	8.8	1.1	55	7.8	1.2	47	1.7	2.3	40	7.7	1.3	64	1.1	1.4	42
Biological characteristics																			
Escherichia coli	MPN/ 100 mL	1.4×10^7	8.0×10^6	37	4.5×10^6	3.3×10^6	51	1.3×10^7	5.7×10^6	49	4.4×10^5	6.6×10^5	27	2.3×10^7	5.1×10^7	43	1.6×10^6	3.2×10^6	38
Total Coliforms	MPN/ 100 mL	6.9×10^7	3.6×10^7	37	2.4×10^7	2.5×10^7	55	6.5×10^7	2.7×10^7	49	1.2×10^7	1.3×10^7	34	1.0×10^8	1.0×10^8	44	2.6×10^7	3.8×10^7	37
Enterococci	MPN/ 100 mL	4.0×10^5	1.5×10^5	2	7.4×10^4	8.4×10^4	9	6.6×10^5	4.0×10^5	8	8.6×10^3	1.4×10^4	7	7.9×10^5	3.2×10^5	12	6.3×10^4	4.4×10^4	10

n = number of measurements, sd = standard deviation, EC = electrical conductivity, TS = total solids, TSS = total suspended solids, tCOD = total (homogenized) chemical oxygen demand, sCOD = soluble (0.45 µm filtered) COD, pCOD = particulate COD, TN = total nitrogen, TP = total phosphorous, MPN most probable number.

8.3 Supplementary information paper III

Table SI 8.3.1: Operation phases of the WSP depending on the installation of the pre-treatment (PreT) and post-treatment (PostT) with the corresponding outflow of train A (A4) and train B (B4).

Phase	Operation	Start day	End day	Effluent
I	Total inflow in train B (planning and construction of PreT and PostT)	1	675	Pond B4
II	Total inflow in train B (PreT in operation)	676	1,011	Pond B4
III	Inflow shared between train A and B (PreT and PostT in operation)	1,012	1,444	Pond A4 and Pond B4

Table SI 8.3.2: T-Test for the significance of the *E. coli* log reduction and turbidity, solar radiation, chlorophyll-*a*, temperature, inflow concentration, precipitation and between the *E. coli* log reductions of the different treatment trains. NA = not analysed.

Parameter	*E. coli* log reduction		
	A – III	B – III	B – I+II
turbidity	0.0000	0.0000	0.0000
solar radiation	0.0000	0.0000	0.0000
chlorophyll-*a*	0.0090	0.0013	NA
temperature	0.0000	0.0000	0.0000
inflow concentration	0.0000	0.0000	0.0000
precipitation	0.5983	0.5708	0.2245
E. coli log reduction A – III	-	0.0003	-
E. coli log reduction B – III	0.0003	-	0.0000
E. coli log reduction B – I+II	-	0.0000	-

Table SI 8.3.3: Microbial community analysis: Richness (Chao, 1984) and Shannon's (Schloss and Handelsman, 2006) diversity indices estimates for each sampling point. sd = standard deviation.

Sampling point	Chao1		Shannon	
	indices	sd	indices	Sd
Inflow	2,253	344	6.0	0.5
A1	1,799	486	5.9	0.3
A3	2,668	691	6.0	0.3
A4	2,517	584	5.8	0.6
B1	2,140	574	5.7	0.4
B4	2,370	683	6.0	0.4

Table SI 8.3.4: Physical, chemical and biological parameters of the waste stabilization pond system in Namibia. Inflow values are from phase I, II and III, effluent values (phase III) present the two treatment trains: Train A with pre-treatment and post-treatment whilst train B in the original setup. For comparison the water quality objectives by the FAO (Ayers and Westcot, 1985), Müller (2017) and WHO (2006) are presented. Three "degrees of restrictions on use" (1 = none, 2 = slight to moderate, 3 = severe) are applied as suggested by the FAO (Ayers and Westcot, 1985).

		Untreated wastewater			Monitoring data enhanced train A - III (phase III)												Monitoring data original train B - III (phase III)												Water quality objectives			
		(phase I, II and III)			A1 effluent			A2 effluent			A3 effluent			A4 effluent			B1 effluent			B2 effluent			B3 effluent			B4 effluent			Degree of restriction use			
Parameter	Unit	Mean	sd	n	mean	sd	n	Mean	sd	n	Mean	sd	n	Mean	sd	n	Mean	sd	n	Mean	sd	n	Mean	sd	n	Mean	sd	n	1	2	3	source
Physical characteristics																																
EC	µS/cm	943	143	272	755	119	123	721	132	74	750	130	122	832	119	123	825	75	124	834	96	74	835	105	123	837	120	124	<700	700 - 3,000	>3,000	FAO 1985
Turbidity	FNU	355	121	257	243	140	110	181	105	66	220	114	108	192	129	109	417	131	110	306	104	66	292	104	109	266	108	109	$<21^{a)}$	$23 - 43^{a)}$	$>43^{a)}$	Müller 2017
TS	mg/L	669	101	38	702	47	11	752	51	5	711	31	17	718	47	18	753	51	11	755	50	5	729	42	11	743	36	17	-	-	-	-
TSS	mg/L	245	96	118	123	56	25	97	48	16	79	34	57	49	36	58	187	61	23	111	47	16	112	39	52	98	41	57	$<50^{a)}$	$50 - 100^{a)}$	$>100^{a)}$	FAO 1985
Chemical characteristics																																
pH	-	6.8	0.7	273	8.3	0.6	124	8.8	0.7	74	8.7	0.7	123	7.7	0.8	124	7.8	0.5	124	7.9	0.4	74	7.9	0.5	123	7.9	0.5	124	"normal range": 6.5 - 8.4			FAO 1985
tCOD	mg/L	749	158	142	401	121	59	384	91	16	372	94	58	317	108	59	544	126	59	437	87	16	433	103	58	415	100	59	according to BOD/TCOD ratio			Müller 2017
sCOD	mg/L	294	60	138	71	16	58	85	12	16	88	16	58	89	20	58	80	25	58	78	10	16	83	20	58	85	21	58				
pCOD	mg/L	453	140	137	331	118	58	299	88	16	284	90	58	228	102	58	465	109	58	359	81	16	351	90	58	332	84	58				
TN	mg/L	75.9	13.1	140	59.8	15.3	29	46.7	19.9	16	42.1	17.1	58	39.3	17.1	58	71.2	7.9	29	62.4	10.5	16	60.5	12.4	58	58.9	11.3	58	<5	5 - 30	>30	FAO 1985
NH_4^+-N	mg/L	62.9	12.1	131	33.0	10.0	29	21.2	14.2	16	16.6	12.9	58	19.4	11.1	58	38.7	3.5	28	36.4	7.0	16	33.9	8.2	57	32.8	9.2	58				
TP	mg/L	9.7	1.5	137	10.0	1.1	28	10.5	2.1	16	10.2	2.2	57	10.6	2.2	57	10.1	0.7	28	10.4	1.3	16	10.4	1.5	57	10.7	1.5	58	<3.5	3.5 - 13	>13	Müller 2017
PO_4^{3-}-P	mg/L	7.5	1.5	137	1.9	2.5	24	2.9	2.3	17	3.2	2.2	57	6.1	2.2	58	1.7	1.9	24	1.7	1.8	16	1.6	1.9	54	1.9	2.0	55				
Biological characteristics																																
Escherichia coli	MPN/ 100 mL	1.7E+07	3.3E+07	108	1.2E+05	1.9E+05	39	5.2E+03	9.4E+03	23	6.1E+02	1.3E+03	64	3.6E+02	6.8E+02	65	9.4E+05	9.4E+05	44	1.1E+04	1.3E+04	64	1.2E+05	1.9E+05	39	1.8E+03	2.6E+03	67	case specific			WHO 2006
Total Coliforms	MPN/ 100 mL	9.0E+07	7.4E+07	103	5.7E+06	1.1E+07	29	1.2E+07	2.0E+07	14	5.5E+06	6.7E+06	44	8.9E+06	1.2E+07	46	1.3E+07	1.8E+07	41	1.6E+07	1.7E+07	14	1.2E+07	1.1E+07	43	1.2E+07	1.5E+07	45	-	-	-	-
Enterococci	MPN/ 100 mL	1.6E+06	1.8E+06	54	2.6E+03	2.6E+03	17	2.2E+03	3.6E+03	17	2.4E+03	3.9E+03	17	5.3E+03	4.0E+03	17	2.8E+04	2.2E+04	17	7.9E+03	8.8E+03	17	9.9E+03	1.2E+04	16	1.9E+04	3.1E+04	17				
Pseudomonas aeruginosa	MPN/ 100 mL	8.7E+06	9.6E+06	54	5.4E+04	1.3E+05	12	5.9E+03	7.7E+03	9	1.0E+04	1.2E+04	6	6.0E+03	5.7E+03	13	1.1E+06	1.3E+06	16	1.0E+05	3.0E+05	16	8.7E+03	5.8E+03	7	7.9E+03	6.8E+03	6				

n = number of measurements, sd = standard deviation, EC = electrical conductivity, TS = total solids, TSS = total suspended solids, tCOD = total (homogenized) chemical oxygen demand, sCOD = soluble (0.45 µm filtered) COD, pCOD = particulate COD, TN = total nitrogen, TP = total phosphorous, MPN = most probable number.

Zum Autor:

Jochen Sinn, geboren 1978 in Heilbronn, absolvierte an der Universität Karlsruhe sein Diplomstudium in Bauingenieurwesen und den Vertiefungen Wasser und Umwelt, Umwelttechnologie und Siedlungswasserwirtschaft. Für seinen Master im Bereich International Rural Development mit den Schwerpunkten Development Economics, International Development, Project Planning and Management sowie Sustainability studierte er an der Lincoln University in Neuseeland. Nach seinen Abschlüssen arbeitete Jochen Sinn für acht Jahre in der internationalen Entwicklungszusammenarbeit mit dem Fokus auf kommunale Wasser- und Abwasserinfrastruktur in Afrika. Daran anschließend wechselte er ans Fachgebiet Abwassertechnik, später Wasser und Umweltbiotechnologie, des Instituts IWAR an der TU Darmstadt und betreute zwei Projekte zur landwirtschaftlichen Wasserwiederverwendung und Abwasserteichen in Namibia. In diesem Rahmen entstand die vorliegende Dissertation.

Zum Inhalt:

Abwasserteichanlagen werden besonders in Ländern des globalen Südens für die kommunale Abwasserbehandlung eingesetzt. Gleichzeitig sind viele Menschen in diesen Regionen vom Klimawandel und damit einhergehender Wasserknappheit betroffen. Aus diesem Grund bietet die Wiederverwendung von gereinigtem Wasser aus Abwasserteichanlagen eine wertvolle Ressource für Bewässerungswasser sowie für Pflanzennährstoffe. Viele dieser Teichsysteme sind jedoch durch schnelles Bevölkerungswachstum überlastet. Somit führt das ungenügend gereinigte Ablaufwasser zu Umweltverschmutzungen und zu erhöhten Gesundheitsrisiken für Menschen und Tiere. Diese Forschungsarbeit untersucht zum ersten Mal umfassend neun Abwasserteichanlagen im zentralen Norden Namibias und ermittelt das Wiederverwendungspotential zur Bewässerung von Futtermitteln. Außerdem werden auf einer Pilotanlage verschiedene Ertüchtigungsmaßnahmen in großem Maßstab erprobt und die Ergebnisse hinsichtlich ihrer Übertragbarkeit auf andere Standorte unter ähnlichen Bedingungen ausgewertet. Zu den mechanischen und biologischen Vorbehandlungstechnologien gehört ein Mikrosieb sowie ein UASB-Anaerobreaktor. Weiterhin ermöglicht die Entschlammung der Teiche die Wiederherstellung des ursprünglichen Volumens und schwimmende Leitwände im Fakultativteich verbessern die Strömungsbedingungen. Im letzten Schönungsteich wird als Nachbehandlung ein Steinfilter zur Algen- und Keimreduzierung eingebaut. Die Auswirkungen der Ertüchtigungsmaßnahmen werden einer zweiten, im Originalzustand parallel betriebenen Behandlungsstraße gegenübergestellt. Die untersuchten biologischen und physikalischen Abwasserparameter ermöglichen einen Vergleich mit der nationalen namibischen sowie der europäischen Anforderung an die Wasserwiederverwendung. Darüber hinaus wird auch die DNA der vorhandenen mikrobiellen Gemeinschaften analysiert. Somit trägt diese Dissertation wertvolle Informationen zur Ertüchtigung existierender Teichkläranlagen, zum Schutz der Umwelt und zur Produktion von Bewässerungswasser in Regionen mit knappen Wasserressourcen bei.

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