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- Masterarbeit -

Comparative life-cycle assessment of biological and membrane-based technologies for wastewater treatment in a brewery

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Weiterhin versichere ich, dass diese Arbeit noch keiner anderen Prüfungsbehörde vorgelegt wurde.

Berlin, am 09.03.2023 ....

Thomas Exner

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Abstract

A new generation of integrated fixed-film activated sludge (IFAS) systems, merging the biofilm of the root zone from aquatic plants into the activated sludge process, has increasingly gained attention in recent years as a potential alternative to conventional wastewater treatment systems. However, there is a lack of understanding of the broader environmental impact of this emerging technology and how it compares to traditional concepts of wastewater treatment. In this research, we address this gap by conducting a comparative Life Cycle Assessment (LCA) with three reference scenarios, based on design simulations in seven midpoint impact categories. The entire novel wastewater treatment system at a small to medium-sized brewery in the Netherlands, including sludge disposal, resulted in net values of 29.2 MJ, 1.9 kg CO_2-eq., 3.4 g NO_X-eq., 0.1 mg CFC_{11}-eq., 4.0 g SO_2-eq., 0.3 g P-eq., and 1.9 N-eq. per m^3 wastewater treated, under categories CED, GWP, POFP, ODP, TAP, FEP, and MEP, respectively. Compared to aerated SBR systems, the new system demonstrated higher environmental burdens in CED (120%), GWP (122%), POFP (125%), ODP (123%), and TAP (133%). This study provides evidence that these impacts on the environment mainly depend on the technology's current electricity demand, while additional improvements can also be achieved by lowering the chemical and nutrient demand of the system. The comparison to a potential anaerobic treatment opportunity for the brewery wastewater with an EGSB reactor, exacerbated the previously identified shortcomings of the new technology, since the crediting of biogas allowed a complete offset of the total environmental impact measured by the GWP, CED, and ODP. Our findings suggest that additional water recovery concepts with subsequent nanofiltration systems, aimed at preserving natural water resources, may offer no competitive advantage for the GWP, CED, POFP, OPD, and TAP, if the electricity demand (1.17 kWh per provided m^3 reused water) surpasses the benefit of water reuse. However, it is important to note that the new technologies provide their own set of benefits, such as a reduced impact on freshwater and marine eutrophication, due to the high nutrient uptake capability. Our research provides implications for practitioners and researchers seeking to understand the environmental impact associated with plant root equipped IFAS, while implicit design assumptions may limit the ability to generalise findings on real-world scenarios.

Deutsche Zusammenfassung

Eine neue Generation von IFAS-Systemen (Integrated Fixed-Film Activated Sludge) in der Abwassertechnik, die den Biofilm der Wurzelzone von Wasserpflanzen in ein kaskadiertes Belebtschlammverfahren integrieren, haben in den letzten Jahren zunehmend an Aufmerksamkeit gewonnen. Es mangelt jedoch an Kenntnissen über die weiterreichenden Umweltauswirkungen dieser neuen Technologie und darüber, wie sie im Vergleich zu herkömmlichen Konzepten der Abwasserbehandlung abschneidet. In dieser Forschungsarbeit behandeln wir diese Wissenslücke, indem wir eine vergleichende Ökobilanz für das neuartige Abwasseraufbereitungssystem in einer kleinen bis mittelgroßen Brauerei in den Niederlanden und drei modellierten Referenzszenarien durchführen. Für das Abwasseraufbereitungssystem und seine indirekten Auswirkungen durch Produktionsketten und externer Schlammentsorgung, konnten in sieben Umweltwirkungskategorien (KEA, GWP, POFP, ODP, TAP, FEP, MEP) Ergebnisse erzielt werden (Bezogen auf m^3 Abwasser: 29.2 MJ, 1.9 kg CO_2-eq., 3.4 g NO_X-eq., 0.1 mg CFC_{11}-eq., 4.0 g SO_2-eq., 0.3 g P-eq., and 1.9 N-eq). Die Umweltauswirkungen hängen in erster Linie vom Strombedarf der Technologie ab. Potenzielle Verbesserungsmöglichkeiten bestehen in einer Senkung des Energie-, Chemikalien- und Nährstoffbedarfs. Im Vergleich zu Modellen belüfteter SBRs wies das mit Wasserpflanzen bestückte IFAS-System höhere Umweltbelastungen im KEA (120%), GWP (122%), POFP (125%), ODP (123%) und TAP (133%) auf. Der Vergleich mit einer zusätzlichen anaeroben Behandlungsmöglichkeit, simuliert durch Literaturwerte von EGSB-Reaktoren, verschärfte die Aussagen der zuvor genannten Mängel der neuen Technologie, da bei anaerober Behandlung eine Gutschrift für Biogas zu vollständigem Ausgleich der Umweltauswirkungen des KEA, GWP und ODP führte. Ein untersuchtes Wasserrückgewinnungskonzept mit nachgeschalteter Nanofiltrationsanlage, die auf die Schonung der natürlichen Wasserressourcen abzielt, zeigte keinen Vorteil für die Kategorien KEA, GWP, POFP, OPD und TAP, da der Strombedarf (1,17 kWh pro bereitgestelltem m^3 Wasser) den Nutzen der Wasserwiederverwendung übersteigt. IFAS mit den Pflanzen zeigten jedoch auch Vorteile, z. B. eine gute Ablaufqualität, die im Vergleich zu Referenzszenarien nur in geringerem Maße zur Eutrophierung beitragen. Unsere Forschungsergebnisse sollen Praktikern und Forschern helfen, die Umweltauswirkungen von mit Pflanzensystemen ausgestatteten IFAS besser zu verstehen. Implizierte Modellannahmen können jedoch die Deutung und Verallgemeinerung der Ergebnisse einschränken.

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Icon	Meaning	Unit
A	area	m²
B_V	volumetric organic loading rate	kg COD · m^{-3} · d^{-1}
b	deday constant	d^{-1}
C	concentration	mg · l^{-1}
C_{COD}	chemical oxygen demand (concentration)	mg · l^{-1}
$C_{COD,dos}$	C-Dosing concentration of COD	mol · l^{-1}
C_S	saturation concentration of constituent in liquid	mol · l^{-1}
C_{SS}	saturation concentration (temperature dependent)	mol · l^{-1}
C_{COD_S}	soluble chemical oxygen demand (concentration)	mg · l^{-1}
C_{TN}	concentration of total nitrogen	mg · l^{-1}
C_{TP}	concentration of total phosphorus	mg · l^{-1}
C_X	required oxygen concentration	mg · l^{-1}
d	diameter	cm
DM	dry matter	kg or %
$E_{H,Credit}$	Energy credit for heat production	kWh · d^{-1}
$F_{SP,EGSB}$	dry matter of sludge mass production by EGSB	kg DM · d^{-1}
$F_{SP,SBR}$	dry matter of sludge mass production by SBR	kg DM · d^{-1}
$F_{SP,M}$	sludge mass production	kg SS(B)/d
$F_{SP,V}$	volumetric sludge production	kg SS(B)/m3
F_T	factor for temperature correction	-
f_a	inert part of the particulate COD (relative)	- or %
f_b	inorganic content of TSS (relative)	- or %
f_{IR}	ignition residue	- or %
f_{OS}	factor of oversaturation	-
$f_{VSS,N}$	nitrogen content in the VSS	- or %
$f_{VSS,P}$	phsophorus content in the VSS	- or %
g	gravitational constant	m · s^{-2}
h	height	m
H_S^{cp}	henry's law solubility constant	mol · m^{-3} · Pa^{-1}
h_a	water height of the reactor	m
M	molar mass	u
M_N	molar mass of nitrogen	u
m	mass	kg
$\boldsymbol{m_C}$	SBR operation cycles per day	-
$m_{TS,CAS}$	required amount of sludge for CAS treatment	kg
$m_{TS,SBR}$	required amount of sludge for SBR treatment	kg

$\dot{m}$	load	$kg \cdot d^{-1}$
$\dot{m}_{AM_{press}}$	Polymer dosing of the belt filter press as AM	$kg \cdot d^{-1}$
$\dot{m}_{COD_{ele}}$	eliminated load of chemical oxygen demand	$kg \cdot d^{-1}$
$\dot{m}_{COD_{in}}$	influent load of the chemical oxygen demand	$kg \cdot d^{-1}$
$\dot{m}_{COD_{out}}$	effluent load of the chemical oxygen demand	$kg \cdot d^{-1}$
$\dot{m}_{COD,SP}$	mass of organic part of produced excess sludge	$kg \cdot d^{-1}$
$m_{loss,SP}$	mass lost by pre-treatment or sludge production	$kg \cdot d^{-1}$
$\dot{m}_{NO_{3,in}^{-}}$	influent load of nitrate	$kg \cdot d^{-1}$
$\dot{m}_{S_{COD,in}}$	influent load of the soluble COD	$kg \cdot d^{-1}$
$\dot{m}_{S_{COD,I,in}}$	influent load of the inert soluble COD	$kg \cdot d^{-1}$
$\dot{m}_{TN,BM}$	total nitrogen of the biomass	$kg \cdot d^{-1}$
$\dot{m}_{TN,Dos,min}$	dosing requirement for nitrogen	$kg \cdot d^{-1}$
$\dot{m}_{TN_{ele}}$	mass of eliminated total nitrogen per day	$kg \cdot d^{-1}$
$\dot{m}_{TN_{ele,SP}}$	mass of eliminated nitrogen by assimilation per day	$kg \cdot d^{-1}$
$\dot{m}_{TN_{ele,d}}$	mass of eliminated nitrogen by denitrification per day	$kg \cdot d^{-1}$
$\dot{m}_{TN_{in}}$	influent load of total nitrogen	$kg \cdot d^{-1}$
$\dot{m}_{TP,BM}$	total phosphorus of the biomass	$kg \cdot d^{-1}$
$\dot{m}_{TP,Dos,min}$	dosing requirement for phosphorus	$kg \cdot d^{-1}$
$\dot{m}_{TP_{ele}}$	mass of eliminated total phosphorus per day	$kg \cdot d^{-1}$
$\dot{m}_{TP_{in}}$	influent load of total phosphorus	$kg \cdot d^{-1}$
$\dot{m}_{TSS_{ele}}$	mass of eliminated total susp. solids per day	$kg \cdot d^{-1}$
$\dot{m}_{TSS,in}$	influent load of total suspended solids	$kg \cdot d^{-1}$
$\dot{m}_{X_{COD,BM}}$	net biomass production	$kg \cdot d^{-1}$
$\dot{m}_{X_{COD,in}}$	influent load of the particulate COD	$kg \cdot d^{-1}$
$\dot{m}_{X_{COD,I,BM}}$	production of inert parts from biomass decay	$kg \cdot d^{-1}$
$\dot{m}_{X_{COD,I,in}}$	influent load of the inert particulate COD	$kg \cdot d^{-1}$
$\dot{m}_{X_{TN,in}}$	influent load of the particulate TN	$kg \cdot d^{-1}$
NCV_{CH_4}	net calorific value of methane	$kWh \cdot Nm^{-3}$
n	number of reactors	-
n_{boiler}	efficiency of the boiler in the brewery	- or %
OV_C	oxygen requirement for COD elimination	$mol \cdot l^{-1}$
$OV_{d,C}$	oxygen mass requirement for COD elimination	$kg \cdot d^{-1}$
P	pressure of gas	Pa
P_0	pressure of gas at standart conditions	Pa
P_g	partial pressure of gas	Pa
p_g	mole fraction of gas in air	-
P_T	total pressure	Pa
pH	negative logarithmic C of hydrogen ions	-

$Q_{d,av}$	av. volume of wastewater per day over the year	$m^3 \cdot d^{-1}$
$Q_{d,max}$	av. volume of wastewater per day over the year	$m^3 \cdot d^{-1}$
r	radius	m
SAE	aeration efficiency (SAE-value)	$kg \cdot kWh^{-1}$
SRT	Sludge retention time	d
SVI	Sludge volume index	$l \cdot kg^{-1}$
SS(B)	biomass or sludge measured as suspended solids	kg SS(B)
T	temperature	K
TS_R	concentration of total solids in the reacor	$mol \cdot l^{-1}$
TSS	total suspended solids (concentration or load)	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
t_C	time of one SBR operation cycle	h
V	volume	m^3 (or l or hl)
$V_{biogas,av}$	average volume of produced biogas	$Nm^3 \cdot d^{-1}$
V_{gas,CH_4}	volume of methane in the biogas	$Nm^3 \cdot d^{-1}$
V_{loss}	lost volume of the flow rate	$m^3 \cdot d^{-1}$
V_{loss,CH_4}	lost volume of methane	$Nm^3 \cdot d^{-1}$
V_{min}	minimum reactor volume	m^3
V_R	theoretical reactor volume	m^3
$V_{SP,M}$	produced volume of excess sludge	$m^3 \cdot d^{-1}$
V_{total,CH_4}	total methane gas production	$Nm^3 \cdot d^{-1}$
VSS	volatile suspended solids (concentration or load)	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
$X_{COD,BM}$	particulate COD as biomass (concentration)	$mg \cdot l^{-1}$
$X_{CSB,I,BM}$	inert COD from biomass decay (concentration)	$mg \cdot l^{-1}$
x_{DM}	dry matter content	%
x_{GF}	gas content burned in the gas flare	- or %
Y_{hetero}	yield coefficient for sludge production (heterotroph)	kg TS · $(kg\ COD_{ele})^{-1}$
$Y_{COD,dos}$	dosing factor of COD for denitrification	-
y_{real}	yield coefficient for sludge production	kg TS · $(kg\ COD_{ele})^{-1}$
ΔV_{max}	exchange volume of the SBR	m^3
α	alpha factor for aeration efficiency	-
α_{SAE}	alpha factor for aeration efficiency (municipal)	-
αOC	mass of required oxygen (dep. on wastewater)	$kg \cdot d^{-1}$
ρ	density	$kg \cdot m^{-3}$
ρ_{CH_4}	density of methane	$kg \cdot m^{-3}$
ρ_{H_2O}	density of water	$kg \cdot m^{-3}$

Acronyms

Icon	Meaning	Common unit
BOD	Biological Oxygen Demand	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
CAS	Conventional Activated Sludge	
CED	Cumulative Energy Demand	MJ
CH_4	Methane	ppm
$C_6H_{12}0_6$	here: Glucose	mol
C_2H_5OH	Ethanol	mol
CHP	Combined Heat and Power system	
Cl^-	Chloride	
Ca^{2+}	Calcium ion	
CO_2	Carbon dioxide	ppm
CO_3^{2-}	Carbonate ion	
COD	Chemical Oxygen Demand	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
CSTR	Continuous Stirred Tank Reactor	
Cu^2	Cupper Ion	
DAF	Dissolved Air Flotation	
EGSB	Extended Granular Sludge Bed (reactor)	
EIA	Environmental Impact Assessment	
ERA	Environmental Risk Assessment	
FB	Fluidised Bed anaerobic reactor	
$FeCl_3$	Iron(III)chloride	$mg \cdot l^{-1}$
FEP	Freshwater Eutrophication Potential	kg P-eq.
FBBR	Fluidized Bed Biofilm Reactor	
GWP	Global Warming Potential	kg CO_2-eq.
H_2S	Hydrogen sulfide	$mg \cdot l^{-1}$
H_2SO_4	Sulfuric acid	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
IC	Internal Circulation (reactor)	
IFAS	Integrated Fixed-film Activated Sludge (system)	
K^+	Potassium ion	
LCA	Life Cycle Assessment	
LCIA	Life Cycle Impact Assessment	
LCI	Life Cycle Inventory	

MBBR	Moving Bed Biofilm Reactor	
MBR	Membrane Bioreactor	
MEP	Marine Eutrophication Potential	kg N-eq.
MFA	Material Flow Analysis	
NaOCl	Sodium hypochlorite	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
NaOH	Sodium hydroxide	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
Mg^{2}	Magnesium ion	
N	Nitrogen	
Na^{+}	Sodium ion	
NF	Nanofiltration	
NH_4^{+}	Ammonium	
NO_3^{-}	Nitrate	
N_2O	Nitrous oxide	ppm
ODP	Ozone Depletion Potential	kg CFC-11-eq.
OH^{-}	Hydroxide ion	
P	Phosphorus	
POFP	Photochemical Oxidant Formation Potential	kg NO_X-eq.
RBC	Rotating Biological Contactors	
Root IFAS	Plant root enhanced IFAS (see IFAS)	
RQ	Research Question	
SAGB	Submerged Attached Growth Bioreactor	
SBR	Sequencing Batch Reactor	
SO_4^{2-}	Sulfate	
T	Temperature	K
TAP	Terrestrial Acidification Potential	kg SO_2-eq.
TC	Total Carbon	$g \cdot (kg\ VSS)^{-1}$
TKN	Total Kjeldahl Nitrogen	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
TN	Total Nitrogen	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
TP	Total Phosphorus	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
TSS	Total Suspended Solids	$mg \cdot l^{-1}$ or $kg \cdot d^{-1}$
UASB	Upflow Anaerobic Sludge Blanket (reactor)	
WWT	Wastewater Treatment	
WWTP	Wastewater Treatment Plant	

1. Introduction

More than 342 million hectolitres of sales-quality beer have been produced in 2021 in the European Union (The Brewers of Europe, 2022). Efficient water and wastewater management constitutes a critical challenge faced by the food and beverage industry, and in particular, the brewing sector (Fillaudeau, 2006). Breweries generate usually between 0.15 and 0.50 m^3 industrial wastewater per hl sales-quality beer production (European Commission, 2019) by a variety of departments and processes related to boiling and lautering, fermentation, filtering and storing, maturing, packaging, cleaning, and bottle washing (Giner-Santonj, 2019). The amount and composition of brewery WW is highly variable, both intra-brewery and inter-brewery, with fluctuations that strongly impact the nature of the effluent (DWA, 2010).

The general aim of wastewater treatment (WWT) is to reduce water pollution and by that to protect the quality and quantity of local natural water resources. This is essential for public health, the protection of the environment, and the sustainability of society and production (UN-Water, 2017). Effective WWT involves a series of physical, chemical, and biological treatment steps, depending on the specific characteristics of the wastewater (Metcalf & Eddy, 2013). In addition, the application of advanced water management strategies for the brewery industry has given rise to a growing interest in new technologies aimed at water reuse such as nanofiltration membrane systems (Götz, 2013).

Different technological possibilities of WWT differ in energy and resource demand and corresponding emissions that occur at different stages of the life cycle, including indirect emissions from upstream and downstream processes like material and electricity production (DWA, 2022a). In order to quantify the potential environmental impacts of these emissions, there has been a growing interest of utilizing and improving the methodology of Life Cycle Assessment (LCA) in the scientific research on WWTPs over the last decades (Corominas, 2020).

The LCA method also reflects the shift in the field of WWT from a paradigm of pure water pollution control to the further emphases on energy efficiency, resource recovery, and circular economy (Guest, 2009). In this context the emissions can be partially mitigated or completely offset due to savings of electricity or process water and the recovery of valuable refined by-products of wastewater and sludge treatment processes that can substitute primary produced industrial goods like mineral fertilizers or natural gas (Chen, 2013). Therefore, the LCA method offers the potential to become a key component for decision-making in the water and wastewater industry (Byrne, 2017) and is also used in particular for the purpose to compare new processes and emerging technologies to conventional and state of the art technologies (Corominas, 2013).

As expounded in the following chapters, this study investigates a new hybrid technology, between systems of attached and suspended biomass, in a Dutch brewery and a potential expansion with a hollow fibre nanofiltration system for water reuse by the comparative LCA method.

In order to provide first data on the environmental impact of the new WWT technology, the European research project “NextGen” (www.nextgenwater.eu), anchored in the framework program “Horizon 2020” (European Commission, 2022), executed a case study based on an industrial scale plant in Koningshoeven (NL). The aim of NextGen is to examine and promote resource recovery systems and the concept of circular economy in the water and wastewater treatment sector. Such task has to be supported by continuous stakeholder participation (Guest, 2009). The case study was accompanied by regular meetings of research centres, local wastewater management authorities, technology providers, and academia.

The new technology has no established generic scientific term but can be categorized to the group of fixed-film activated sludge (IFAS) systems, and integrates in addition the microbiome of the rhizosphere from higher plants in a cascaded biological treatment phase (Poór-Pócsi, 2021). In this study, the term “plant root enhanced IFAS” (root IFAS) will be used. It can be seen as an emerging technology used in several sewage treatment projects worldwide (Fan, 2022).

In addition, a pilot-scale hollow fibre nanofiltration membrane system was designed and tested by “SEMiLLA IPstar Circular Systems.” for the case study. This system provides the possibility of enhanced grey water treatment mitigating the freshwater demand of the brewery and alleviating the strain on the natural groundwater sources.

The principal aim of this thesis is to evaluate the broader environmental impact of the root IFAS system in Koningshoeven, its comparative efficacy vis-à-vis conventional wastewater treatment (WWT) designs, and the potential for augmenting the system's proficiency by integrating an auxiliary membrane-based technology. The objective can be framed by three research questions (RQ).

- RQ 1: What are the key contributing factors, that warrant attention in light of the potential environmental impact of the root IFAS operation with focus on industrial brewery wastewater treatment in Koningshoeven?

- RQ 2: How does the environmental impact of the WWTP in Koningshoeven compare to conventional WWT technologies, and which shifts, burdens, and trade-offs exist regarding the different types of environmental impacts and their mitigation and offsetting opportunities?

- RQ 3: What is the potential environmental impact resulting from the incorporation of a subsequential nanofiltration membrane system as a water reuse strategy in the context of the industrial brewery wastewater treatment process in Koningshoeven?

In order to provide answers to the research questions, various environmental impacts of the new root IFAS technology are investigated. Therefore, the method of LCA is chosen and explained together with the specifics of brewery wastewater and relevant WWT technologies in the theory part. However, this study cannot address special requirements for comparative approaches, critical reviews, and reporting and therefore not achieve ISO-confirmation (ISO 14044, 2006).

For RQ2 reference scenarios related to the situation of the brewery have to be identified and simulated. Therefore, a common aerobic WWTP and an advanced treatment option with anaerobic technology can be compared. Hence, two additional WWT scenarios are designed in this thesis in order to conduct a comparative assessment and to emphasize the potential strengths and weaknesses of the current WWT processes of the brewery. As a fourth scenario, RQ3 is addressed by the implementation of a hollow fibre nanofiltration membrane system with the goal of water reuse, based on pilot-scale projects by SEMiLLA (2022).

2. Theory Part

This chapter presents a background on the theory related to this study. In the first part the initial local background of the specific case is explained. Since data availability from a single brewery is limited and parts of the used theories and guidelines mainly focus on municipal WWT, additional assumptions and corrections have to be made. Therefore, a short explanation of the wastewater sources and resulting raw wastewater characteristics for brewery effluent is given in the second part. In the third part, an overview of the different WWT technologies is provided, along with their general classification and background information to reactor types, that are relevant in this study. In the fourth part, the LCA method is presented and summarized in general and in detail for the evaluation of WWTPs.

2.1. Background of the Location

The study is related to a small to medium-sized brewery located at the "Our Lady's Abbey" of Koningshoeven (Dutch: *Abdij Koningshoeven, Abdij Onze Lieve Vrouw van Koningshoeven*) near Tilburg (NL) and the A58 motorway (Figure 1).

Figure 1: Location of the abbey and brewery in Koningshoeven (Google maps, 2022).

The brewery was founded by monks of the Order of Cistercians of the Strict Observance in the 19th century and taken over by the "Royal Swinkels Family Brewers Holding N.V.", the market leader of beer production in Nord Brabant, in the years of 1998 (Kreuzen, 2010) and 1999 (Walsh, 2012). The yearly beer production can be estimated around 100,000 hl for 2022 (Brewery of Koningshoeven, personal communication, 24th June 2022). The production process relies on freshwater sourced from an underground aquifer via a groundwater well. Its assortment contains a variety of different regional and seasonal beer products, resulting in shifting production cycles and compositions, which are leading to strong fluctuations in the brewery's wastewater effluent quality and quantity parameters. Since 2018 the root IFAS operation has been installed, that is suitable to adapt to these varied demands and wastewater characteristics. It was designed for small footprints (STOWA, 2017), urban environments, and the possibility of social functions like education (Hetem, 2016). In the years 2020, 2021, and 2022 the COVID-19 pandemic led to an overall lower beer production and consumption in the Netherlands compared to 2018 and 2019 (The Brewers of Europe, 2022). As a result of lowered and highly discontinuous production cycles, the optimization of the WWTP in Koningshoeven has been ongoing and reliable long-term data for the assessment has not been available. Modelling data of 2022 and a sampling campaign during April 2022 have been extrapolated as reference data for flow rate, COD and nutrient values (TN and TP). The flow rate of 2022 can be estimated about 150 m^3 per day with 350 yearly operation days (De Dommel, 2022). The sludge is dewatered by a belt filter press unit to archive dry matter (DM) about 22% and is transported (10 km) to an external sludge digestion.

2.2. Industrial Wastewater from Breweries

In this chapter an explanation and overview of generic sources and characteristics of brewery wastewater is presented.

The main ingredients of beer, other than water, are sources of carbohydrates, starch in particular (Fuchs, 2014). The carbohydrates are converted to simple sugars and fermented to ethanol and CO_2 according to the following reaction (1):

$$C_6H_{12}0_6 \rightarrow 2 \cdot C_2H_5OH + 2 \cdot CO_2 \tag{1}$$

The raw materials, which may include cereal adjuncts, undergo initial storage, cleaning, and wet or dry milling processes before being exposed to hot water (Giner-Santonja, 2019). The following beer production alternately involves three chemical and biochemical reactions (mashing, boiling, fermentation and maturation) and three solid-liquid separations (wort separation or lautering, wort clarification, rough beer clarification) (Fillaudeau, 2006). As described by the European Best Available Techniques (BAT) (Giner-Santonja, 2019), beer production is associated with input and output by-products, including barley, cereals, water, hops, yeast, brewers' grain, turb, yeast, and diatomaceous earth, that can be released into the wastewater as illustrated in Figure 2.

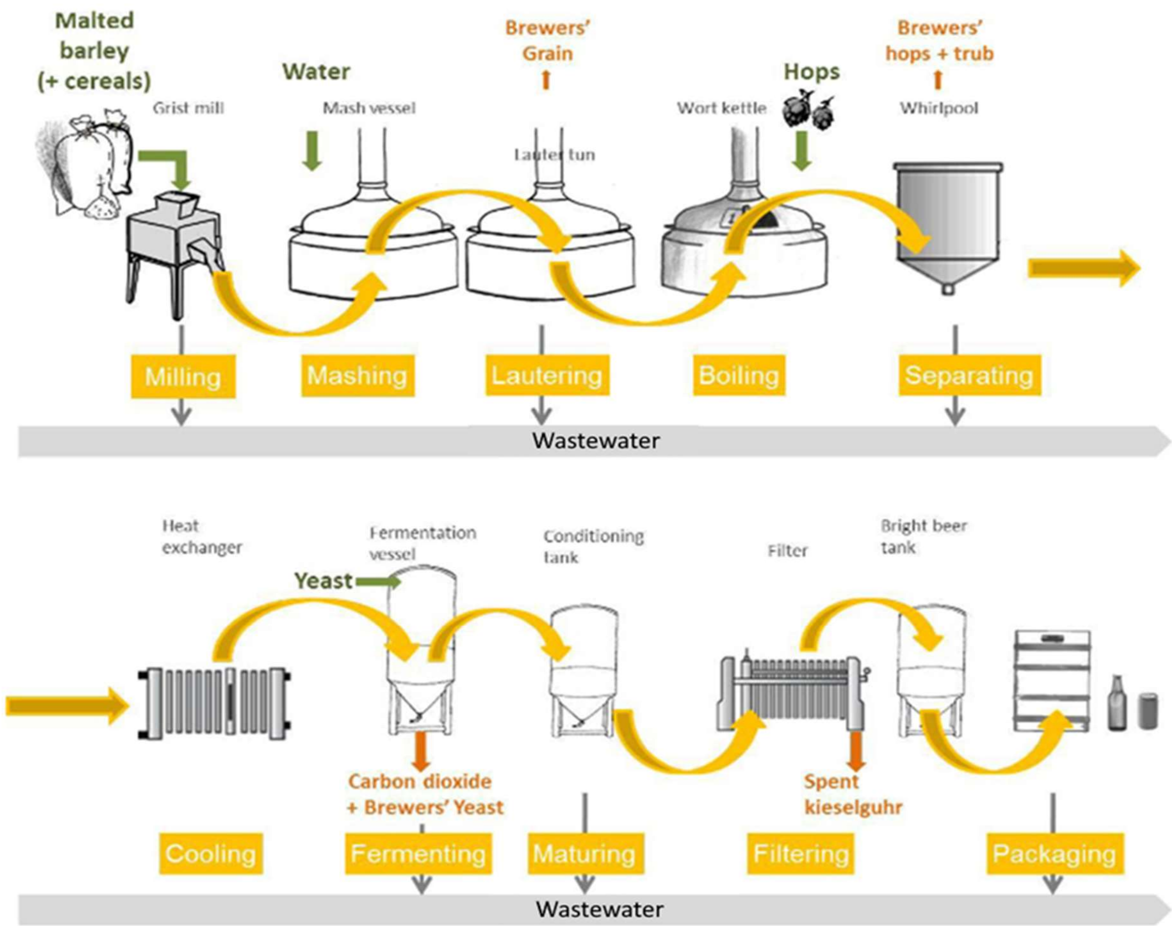


Figure 2: Wastewater production by breweries' processes (Giner-Santonja, 2019).

Other substances that can impact the wastewater of breweries include regular cleaning agents of tanks, such as caustic or acid substances to prevent contamination and scaling, processing aids for the treatment of yeast through sieving or acidic washing, the use of clarifying agents like perlite, cellulose, polymers, isinglass, albumin, tannins, bentonites, silica gel, or woodchips, and the use of various materials for packaging, such as glass, cork, paper, and metals (Giner-Santonja, 2019).

According to the Technology and Engineering Forum of the European Brewery Convention (EBC, 2003) the wastewater production of breweries amounts 3–12 hl · (hl sales beer)$^{-1}$, and is related to different departments of the brewery like the brewhouse, fermentation and maturing, filtration, and packing (Figure 3, p. 7). The process of bottle-cleaning is usually the major contributor (0.1–2.8 hl · hl^{-1}) of industrial brewery wastewater and can be characterised by high temperatures (80 °C) and caustic washing agents (Giner-Santonja, 2019). The large variations in pH, T, organic load, and flow rate associated with the different wastewater streams usually require an equalisation or buffer tank before the biological stage (DWA, 2010).

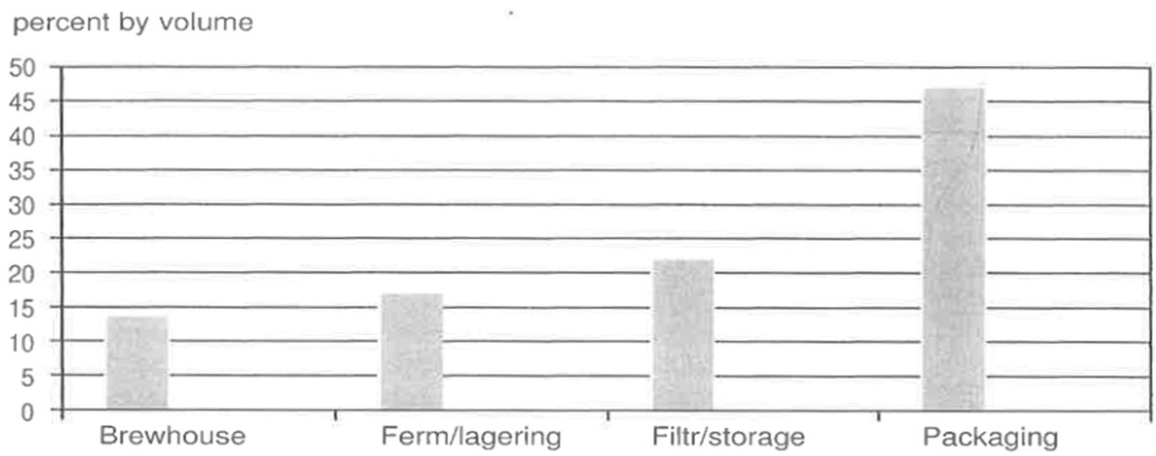


Figure 3: Representative example of relative volumes of wastewater discharges from different production areas (EBC, 2003).

The hourly maximum flow rate occurs during the cleaning period and is in the order of 2.5–3.5 times of the daily average flow rate (Giner-Santonja, 2019). Representative average, minimum, and maximum values for water quality parameters of brewery effluents are presented in Table 1.

Table 1: Typical water quality parameters for industrial brewery wastewater (EBC, 2003).

Parameter	unit	average	min	max
C_{COD}	$[mg \cdot l^{-1}]$	3094	1469	6069
TKN	$[mg \cdot l^{-1}]$	53	39	85
NO_3^-	$[mg \cdot l^{-1}]$	17	8	27
NH_4^+	$[mg \cdot l^{-1}]$	11	3	12
TP	$[mg \cdot l^{-1}]$	17	8	28
TSS	$[mg \cdot l^{-1}]$	688	30	4200
pH		8	4	11
T	K	301.15	293.15	733.15

Many values, including organic pollution, are difficult to estimate, because many materials are unknown and seasonal variations occur during the production (EBC, 2003). The COD of industrial brewery wastewater is readily biodegradable with a COD to BOD ratio of 1.5–1.8 (DWA, 2010). It refers to a high content of sugar, soluble starch, ethanol, and volatile fatty acids (VFA) (Driessen, 2003) and the

BOD to COD ratio in by-products like grain, yeast, trub and kieselguhr (Table 2). The by-products, dirt, and label pulp can create relatively high TSS mass loading rates (Table 1, p. 7), despite mechanical pre-treatment stages, including screens for glass and stones, and additional drum sieves (2 mm diameter) (DWA, 2010).

Table 2: BOD and COD of beer and typical brewery by-products (EBC, 2003).

Component	BOD [mg · l^{-1}]	COD [mg · l^{-1}]
Beer (lager)	80,000 - 120,000	150,000
Spent grain	16,000	24,000
Yeast, fermentation	140,000	210,000
Yeast, lager	120,000	180,000
Trub	110,000	165,000
Kieselguhr (used)	11,000	165,000

Nitrogen originates mainly from yeast, turb, grain losses, and detergents used for tank cleaning (Giner-Santonja, 2019). Phosphorus results from cleaning and descaling agents, beer losses, and organic materials (EBC, 2003).

However, relatively low nutrient vales can lead to shortages or additional dosing requirements in the biological treatment stage (DWA, 2010). According to Henze et al. (2000) the minimum COD/TN/TP ratio for the aerobic activated sludge process is 100/5.5/0.7 (Table 3). For brewery wastewater the N and P content is estimated around 7%, and 1.5% of the COD of the sludge, respectively (Henze, 2000). This equates 10.00% and 2.14% of the volatile suspended solids (VSS). Stronach at al. (1986) reported similar nitrogen content values for anaerobic bacteria, while phosphorus values are estimated close to 1.5% or 2% of the VSS.

Table 3: Carbon and nutrient contents in heterotrophic micro-organisms (Henze, 2000)

Parameter	% per kg VSS	% per kg COD
TC	40.0–60.0	30.0–40.0
TN	8.0–12.0	5.5–8.5
TP	1.0–2.5	0.7–1.8

Micropollutants in brewery wastewater include heavy metals like zinc and nickel, related to glue, and the wear of machines, especially conveyors in packaging lines (EBC, 2003). In higher concentrations, these micropollutants can inhibit anaerobic processes (Stronach, 1986).

2.3. Wastewater Treatment Technologies

This chapter provides an overview of biological wastewater treatment technologies in order to understand the classification of the new root IFAS system. In addition, related wastewater treatment technologies of this study are presented in detail. Anaerobic as well as aerobic biological treatment processes are common for the treatment of industrial brewery wastewater (Rosenwinkel, 2000).

2.3.1. Technical aerobic Wastewater Treatment with Biomass Retention

Technical biological reactors for aerobic WWT with biomass retention can be classified in processes with suspended biomass growth and systems with immobilised biomass growth (Figure 4). Important aerobic variants of wastewater treatment include Conventional Activated Sludge processes (CAS), Sequencing Batch Reactor processes (SBR) and Membrane Bioreactors (MBR) (Grady Jr., 2011). Systems with immobilised growth are characterised by a rigid surface or carrier system to attach biomass. The later refers mainly to Rotating Biological Contactors (RBC), Tickling Filters (TF) and the group of Submerged Attached Growth Bioreactors (SAGB). SAGB include for example the packed bed bioreactors, Fluidized Bed Biofilm Reactors (FBBR), Expanded or Moving Bed Biofilm Reactors (MBBR) and the Integrated Fixed-Film Activated Sludge systems (IFAS) (Grady Jr., 2011).

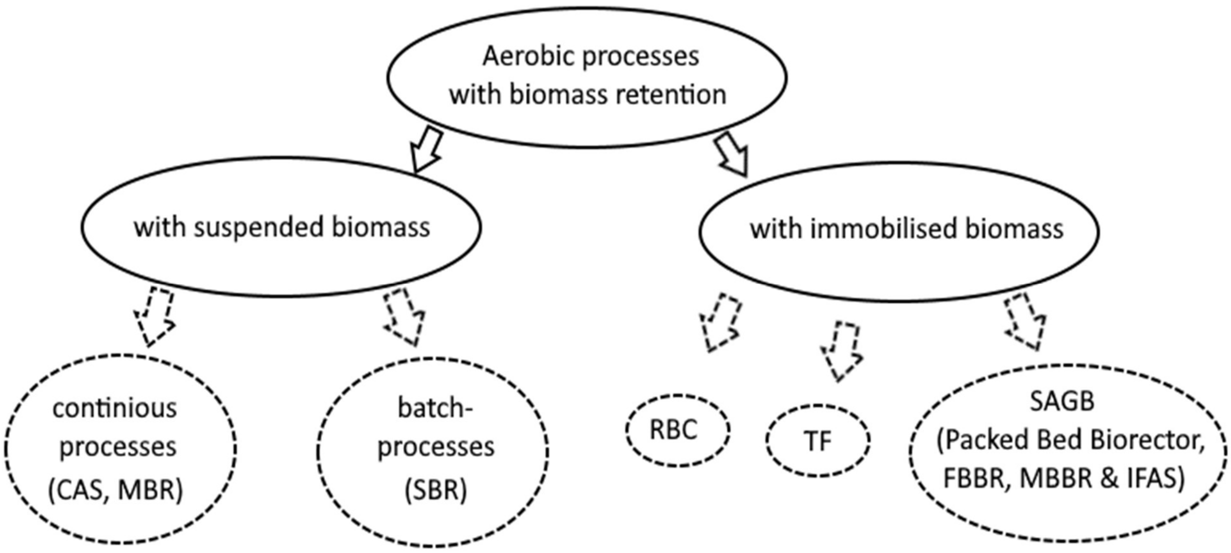


Figure 4: Classification of technical aerobic processes for biological wastewater treatment (own illustration, based on Grady Jr., 2011)

Important design parameters for aerobic systems with suspended biomass include the minimum, medium, and maximum Temperature (T), the determining average flow rate, and the COD, TP, TN, TSS mass loading rates (DWA, 2016). The organic loading rate is a crucial factor in WWTP design, affecting reactor size, oxygen demand, and sludge production (DWA, 2016). Therefore, detailed information on the COD is required to distinguish between suspended and particulate, as well as biodegradable and inert fractions (Figure 5). Parts of the biodegradable fraction get transformed into biomass during the treatment process.

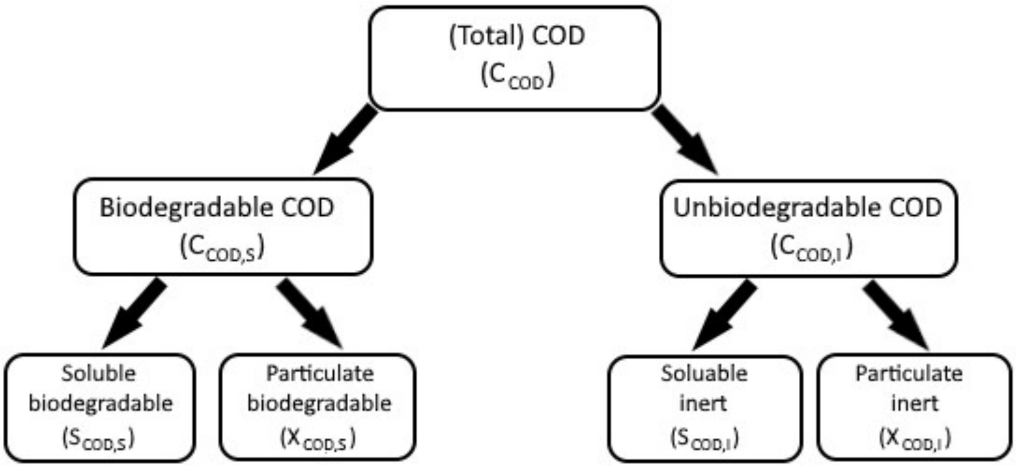


Figure 5: COD divided in soluble, particulate, biodegradable, and inert fractions (own illustration; acronyms like in DWA, 2016)

2.1.3.1. Sequencing Batch Reactor (SBR)

In the field of WWT, a Sequencing Batch Reactor (SBR) describes a biological reactor, that operates without continuous flow and follows a sequence of events or periods. As illustrated in Figure 6, the periods of an SBR cycle include (a) fill (inlet of raw wastewater), (b) react (mixing, aerating), (c) settle (sedimentation), (d) draw (discharge or decanting of treated wastewater and removal of excess sludge), and (e) idle (waiting or resting) (Irvine, 1979).

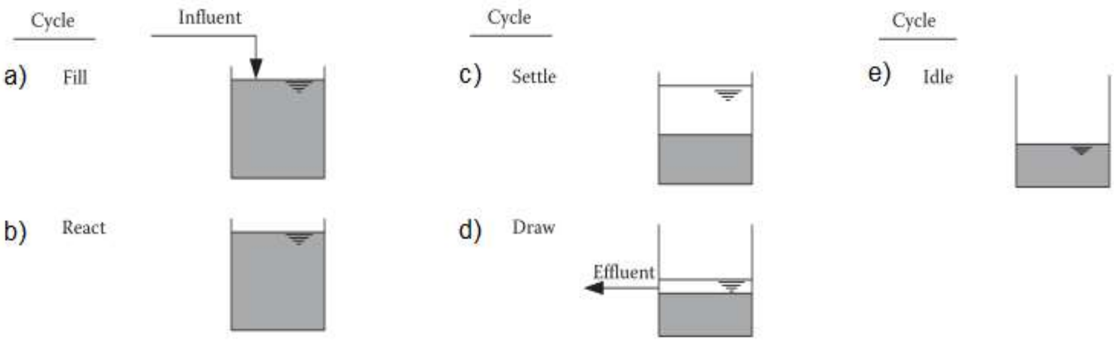


Figure 6: Operational cycle of the SBR activated sludge process (from a to d) (Grady Jr., 2011).

While continuous systems manage WWT more related to the space, the SBR can provide the treatment by a regulated and configurable time and degree of mixing, that allows more flexibility and the control of reaction periods (Grady Jr., 2011). Anaerobic, anoxic, and aerobic phases can be created independent from the hydraulic demands of continuous systems and allow intermediate denitrification and advanced phosphorus removal (DWA, 2009). The inlet begins with a relatively high organic loading rate, followed by a limited aeration span, what results into relatively beneficial sludge settling abilities and can prevent the risk of sludge bulking (DWA, 2009). Due to the settling phase, an external secondary clearer is not needed for SBRs. The main advantage of SBRs is the improved opportunity to deal with variations of the influent and to adjust reaction time and filling grade accordingly (Grady Jr., 2011). SBRs are especially recommended for industrial wastewaters with a high content of carbohydrates (Wilderer, 1989). Difficulties can occur for the cultivation strategy of preferred microorganisms, since facultative anaerobic microorganisms can compete with denitrifiers and release unwanted products like nitrate (Wilderer, 1978).

SBRs can be supported by a pre-storage, process monitoring and control instrumentation, and a cycle strategy, often with two or more parallel, phase shifted tanks (DWA, 2009). The current state-of-the-art for sequencing batch reactors (SBRs) involves reactor heights ranging from 4 to 7 meters, and regulatable aeration and decant systems, that have to be protected from plugging (DWA, 2009). The environmental impact can be seen in a similar range than of continuous CAS systems (DWA, 2009).

2.1.3.2. Integrated fixed-Film Activated Sludge (IFAS)

Integrated fixed-film activated sludge (IFAS) processes refers to wastewater treatment systems that combine suspended and attached biomass growth techniques in one reactor. The attached biomass can be described by systems with stationary media to which microorganism can attach, as well as for reactors with moving or free-floating growth media carriers (Grady Jr., 2011). Biofilm systems allow the development of heterogenic, synergetic, and protected habitats for bacteria (Teichmann, 1997), including anoxic zones for denitrification and additional phosphorus uptake (Jabari, 2014). Biological phosphorus elimination can be supported by recirculation of suspended biomass (Grady Jr., 2011). Competition for space is crucial in determining the distribution of competing species in a biofilm (Grady Jr., 2011). In a single species biofilm, a concentration gradient exists due to the limited diffusion of substrate into the biofilm as illustrated by Figure 7. A relatively long solid retention time (SRT) and advanced removal rates for anthropogenic composites and nutrients can be achieved the biofilm (Arias, 2018; Jabari, 2014).

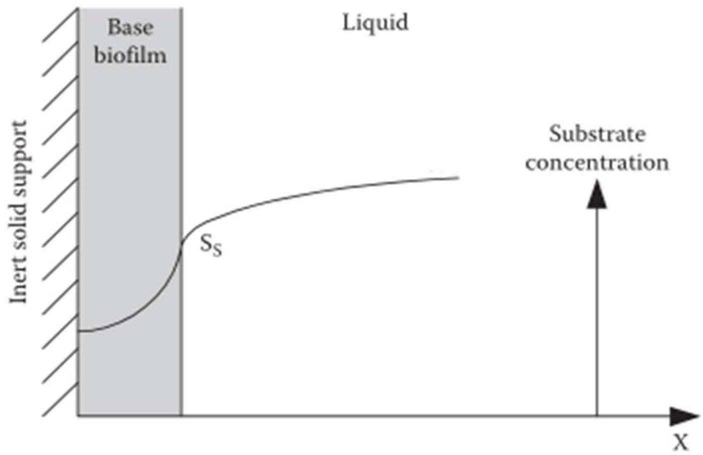


Figure 7: Traditional conceptualization of a base biofilm growing on inert solids a substrate concentration profile (Grady Jr., 2011).

In contrast to traditional biofilm technologies, IFAS allow the microbial growth of two sperate populations to act synergistically, with the mixed liquor suspended solids degrading most of the organic load and the biofilm for efficient simultaneous nitrification and denitrification processes (Waqas, 2020). Similar to CAS, the IFAS also integrates a solid-liquid separation phase and the possibility of recirculation in order to provide biomass retention. As illustrated in Figure 8, the influent gets treated in an aerated tank by suspended and media attached growth, followed by a secondary clarifier, where the Mixed liquor (ML) gets separated, Return Activated Sludge (RAS) recirculated and the excess sludge or Waste Activated Sludge (WAS) removed.

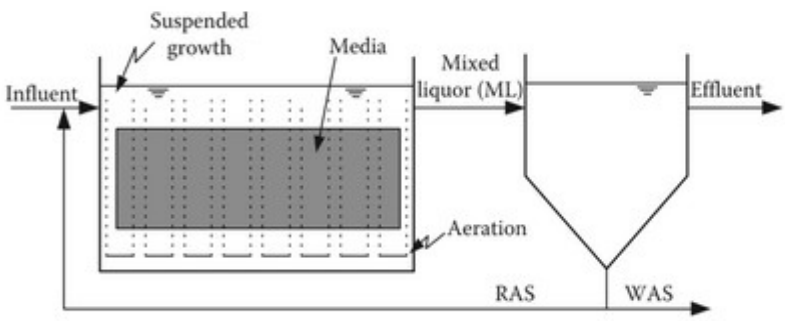


Figure 8: Flow diagram of an Integrated fixed film activated sludge process (Grady Jr., 2011).

The efficiency of the IFAS is highly dependent on the material of the carriers in terms of biological activity and the specific surface area (Teichmann, 1997; Felföldi, 2015). Limited full-scale applications and no well-established process design bases exist for IFAS (Grady Jr., 2011). IFAS achieve over 90% removal rates for COD and ammonia while having 40–60% smaller footprints, and better sludge settling properties and process stability than CAS systems (Di Biase, 2019; Waqas, 2020). IFAS are associated with lower excess sludge production (Liang, 2022). Direct emissions result from anoxic and anaerobic zones. Mannina et al. (2017) measured N_2O emission in the range of 1% of the TN influent. Additional prevention measures against sulphuric bacteria and H_2S odour have to be taken (Teichmann, 1997).

The coexistence of suspended and attached biomass in a bioreactor necessitates elevated volumetric oxygen transfer requirements and dissolved oxygen concentration levels, which can entail high α-factors (about 0.8) and the need of spiral roll aeration systems (Grady Jr., 2011; Teichmann, 1997). Additional retention of air bubbles within the biofilm can be potentially achieved, thus augmenting the volumetric oxygen transfer rates and enabling the use of smaller bioreactors (Grady Jr., 2011).

2.1.3.3. Plant root enhanced IFAS (root IFAS)

The approach of the wastewater treatment system under study is to generate more advanced ecosystems then conventional IFAS (Biopolus, 2022). In this study the system is described as plant root enhanced IFAS (root IFAS). In related studies it is often referenced as “metabolic network reactor” according to the designing company Biopolus (2022), but also occurs under the names “food chain reactor” or “purifying greenhouse” (STOWA, 2017). Purifying greenhouses with plants for sewage treatment refer to the idea of “living machines” that goes back to the American John Todd (1996) in the 1970th, where wastewater passes through different compartments for treatment, each with its own ecosystem.

Figure 9 (p. 14) illustrates the root IFAS, which can be viewed as a combination of a modular fixed bed biofilm system with sludge carriers, along with the integration of the attached microbiome on the rhizosphere of higher plants and an activated sludge system. The suspended biomass can be circulated in a strong cascaded activated sludge system, and a subsequent solid-liquid separation phase is carried out (STOWA, 2017). Different zones with distinct ecosystems are created by anoxic and aerated tanks and submerged carriers like roots and fibre. The influent passes through the different zones generating attached and suspended biomass, followed by recirculation and a solid-liquid separation.

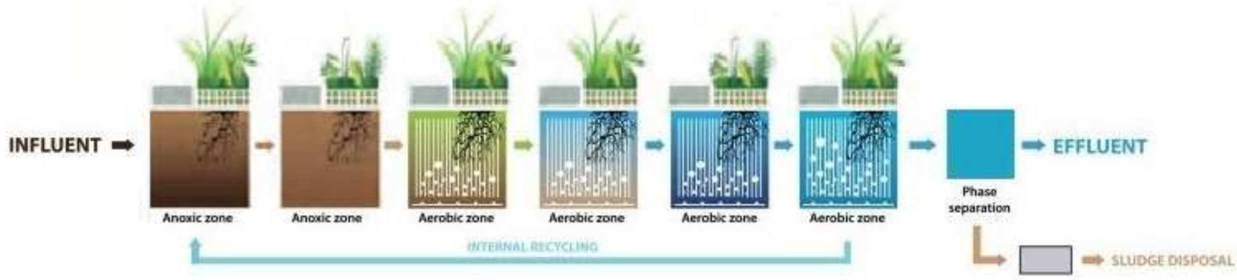


Figure 9: Scheme of a plant root enhanced IFAS (STOWA, 2017).

Multistage systems offer several advantages, such as the ability to impose different environments on different stages, enabling the accomplishment of multiple objectives. In this regard, connecting several Continuous Stirred Tank Reactors (CSTRs) in series provides additional flexibility since feed distribution may be regulated, and biomass recycle be employed throughout the entire chain or certain parts of the process. Tank-in-series has been found to offer benefits for reaction kinetics compared to one single CSTR, leading to zones with higher reaction rates and lower effluent loads (Grady Jr., 2011). Additionally, this configuration provides an opportunity to generate more distinct and defined habitats (Todd, 1996). The complexity of the system is due to the changing physiological state of the biomass as it passes from one bioreactor to another. Split influents and recycle streams can be used for biological nutrient removal processes (Grady Jr., 2011).

Root IFAS often use a combination of anoxic and aerated tanks to achieve higher nitrogen removal rates (e.g. 98.1%), next to COD elimination (about 92.8%), while establishing a diversity of ecosystems (Szilágyi, 2011). The inclusion of higher plants for wastewater treatment is connected to higher phosphorus uptake rates than in CAS without advanced phosphorus removal processes (STOWA, 2017). The roots of the plants have been proven to attract significant more biomass than conventional sludge-on-carrier systems (Todd, 1996) with biofilms up to 25- or 30-mm thickness (STOWA, 2017). The increased biodiversity is meant to result in higher process stability for the WWTP (Todd, 1996). However, biofilm systems have a danger of plugging (Teichmann, 1997), which can lead to defects, lowered WWT efficiencies, and the need of additional air flushing operations. Two thirds of the biomass is attached on biofilm carriers of natural roots or additional extensions made of polypropylene fibres. Typically, tanks with plants are kept in greenhouses to allow for operation in colder seasons and climate regions (USEPA, 2002b).

Relatively high α-factors of 0,85 to 0,90 can be achieved for the oxygen transfer, since aquatic plants ensure micro-aerobic conditions inside and around their roots (STOWA, 2017). However, natural roots protrude only about 2 m into the tanks, while higher volumetric oxygen consumption can be assumed according to the high biomass density (STOWA, 2017). In low-profile tank designs, the efficiency of oxygen transfer may be compromised due to the dependence of efficient pressure aeration systems on the aerated tank's depth (Teichmann, 1997). This factor can lead to reduced performance and must be taken into consideration. In contrast, the European NextGen Research Project expected a relatively low electricity consumption, based on information of technology provider (Poór-Pócsi, 2021).

Similar to conventional IFAS systems, reduced space requirements and excess sludge production (0.6–0.9 kg DM per kg COD-eliminated) is assumed related to a relatively high density of biomass (7–12 kg · m^3), a high SRT and the increased number of protozoa and metazoan in more complex ecosystems (STOWA, 2017; Zheng, 2014). Figure 10 shows the expected decrease of sludge production and undissolved solids in a series of tanks for the root IFAS.

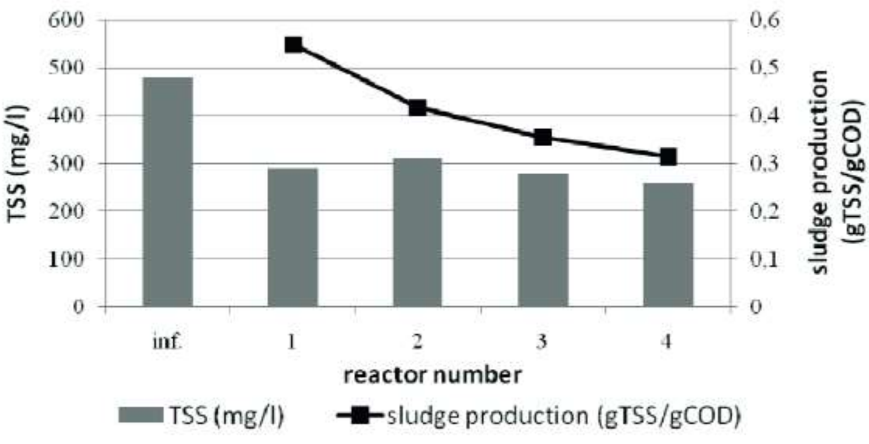


Figure 10: Decrease of the sludge production per reactor and the amount of TSS related to the biomass in the reactors (STOWA, 2017).

In the initial reactors, the bacterial mass increases faster than the number of predators, leading to high MLSS production. In subsequent reactors, protozoa and metazoan reduce the overall bacterial mass as less degradable substrate is available (STOWA, 2017). Sludge mass is lost due to maintenance processes and cell decay with each tank in the series. Despite this, contradicting reports and knowledge gaps raise doubts about the success of implementing predator sludge reduction strategies in full-scale plants (Nelson, 2018). Furthermore, oscillations and anomalies, such as "worm blooms," may occur in practice (Canale, 1973).

2.3.2. Anaerobic treatment by sludge bed reactors

Anaerobic digestion has emerged as a promising approach for the treatment of brewery effluent (EBC, 2003). Through a first process step involving hydrolysis, and acetogenesis, complex organic matter in the effluent is converted into simpler volatile fatty acids (VFA), along with carbon dioxide (CO_2) and hydrogen (H_2). In a subsequent further fermentation step, the VFA are metabolized into acetic acid, which is then transformed by the process of methanogenesis into the final products methane (CH_4), carbon dioxide (CO_2), and water (H_2O) (Metcalf & Eddy, 2013). The recommended influent pH for good performance and stability is in the range of 6.5–7.5 (Stronach, 1986).

A comparison of aerobic and anaerobic treatment is shown in Table 4. An important consideration for the environmental impact of anaerobic wastewater treatment is the biogas production and reduced electricity demand resulting from the absence of aeration (Grady Jr., 2011). The process is further characterised by relatively low excess sludge production, low space requirements, and the ability to reduce COD and TSS at low hydraulic retention times (HRT) (Fillaudeau, 2006). In order to comply with direct discharge regulations, an aerobic post treatment is required to ensure higher removal rates for COD, TN, TP, and TSS (DWA, 2010), that also includes the treatment of potential off-gases and sulphide (EBC, 2003).

Table 4: Main differences of aerobic and anaerobic WWT (Driessen, 2003).

Ability	Aerobic systems	Anaerobic systems
Electricity consumption	high	low
Biogas production	no	yes
Biosolid production	high	low
COD removal	90–98%	70–85%
Nutrient (TN, TP) removal	high	low
Discontinuous operation	difficult	easy

The implementation of anaerobic technology requires corrosion free design materials like stainless steel or glass fibre reinforced polyester, additional gas treatment stages for drying, desulfurization, and storing, as well as additional safety measurements related to the danger of explosion (Bischhoffsberger, 2005).

Granular sludge bed reactors, such as Upflow Anaerobic Sludge Blanket (UASB) and high load variants, also called Expanded Granular Sludge Bed (EGSB) reactors, are the most common technologies used for anaerobic treatment of brewery wastewater (Bischofsberger, 2005). The UASB was developed in the Netherlands during the late 1970s and first high-rate anaerobic WWT for industrial brewery wastewater were built in 1984 (Driessen, 2003). It is a robust system, that doesn't require moving parts like mixers (Bischhofsberger, 2005). can be seen in Figure 11, in the standard UASB reactor, the raw wastewater flows upward through a dense blanket of anaerobic sludge with superior settling characteristics (> 50 m · h^{-1}), typically consisting of granular sludge (1–4 mm) (Driessen, 2003). A three-phase separator located at the top of the reactor separates the biogas, wastewater, and biomass (Lettinga, 1991).

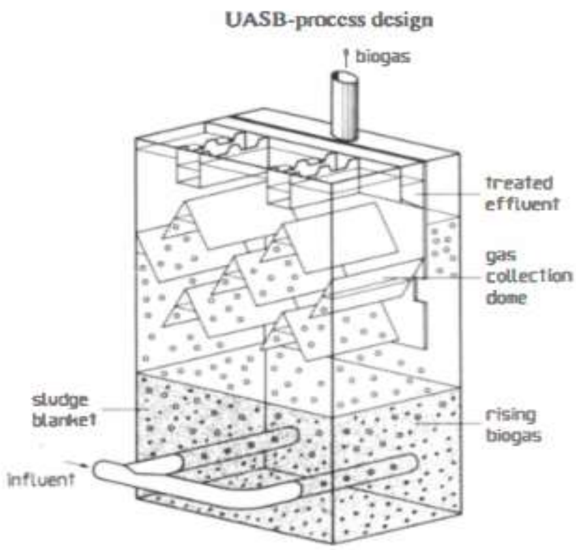


Figure 11: Scheme of a UASB-reactor (Lettinga, 1991).

Unacceptable sludge losses in the blanket zone are prevented by a low upward velocity (0.3–1.0 m · h^{-1}) (Bischhofsberger, 2005). Therefore, the sludge blanket reactor is not suitable for wastewaters with high TSS content (limited to 500 mg · l^{-1}) (Stronach, 1986). Anaerboic digestion can also be inherited by toxic materials (Urban, 2009).

UASB reactors typically have a tank height of 4.5–7.0 m, while EGSB reactors are tower variants with up to 27 m hight (Bischhofsberger, 2005). In the last decades the UASB gets replaced by high tower variants (Bischhofsberger, 2005) such as Fluidised Bed (FB) reactor, the EGSB, and the Internal Circulation (IC) variant, that are illustrated in Figure 12. While the FB uses fluidised carrier materials instead of granular sludge, the EGSB represents a stretched version of the UASB with internal recirculation, an upflow velocity of 1.0 m · h^{-1}, and a modified version of the 3-phase-seperator. These taller designs result in a reduced footprint for the reactor (DWA, 2010). The IC version separates the biogas by two steps, that are built on top of each other, while the biogas in the first stage lets gas rise in the middle of the reactor what leads to circulation (EBC, 2003).

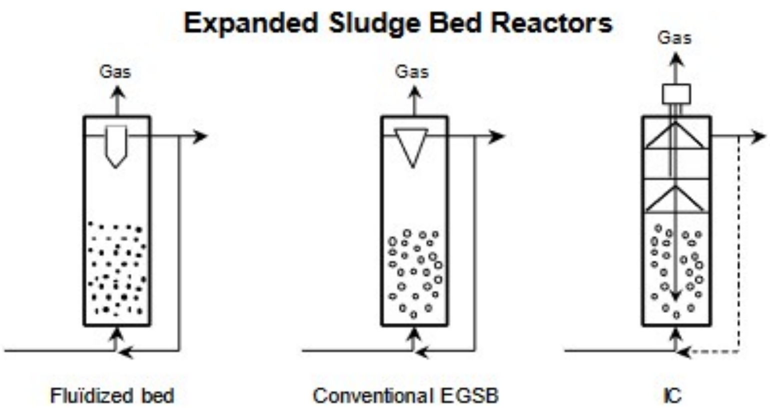


Figure 12: Versions of Extended Sludge Bed Reactors (Driessen, 2003).

As can be seen in Figure 13 (Assumptions: HRT = 4h, $Q_{d,max}$ = 250 m^3 · d^{-1}, B_V = 15 kg COD · m^{-3} · d^{-1}), the reactor size is usually designed in reference to the organic loading rate (B_V) for COD mass loading rates above 2.5 kg COD · m^{-3}.

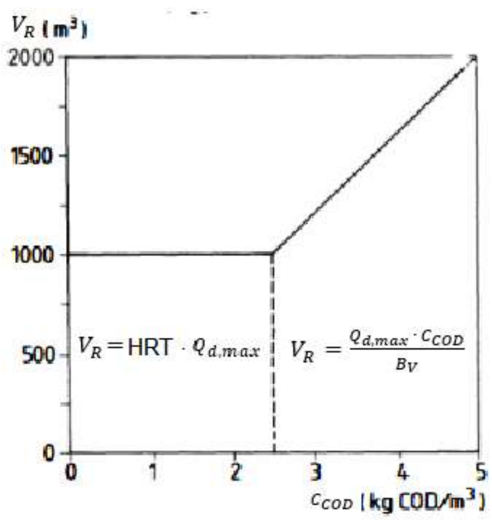


Figure 13: Relationship between COD concentration and reactor volume. (Lettinga, 1991).

Table 5 illustrates typical organic loading rates of sludge blanket reactors for the treating brewery effluent. While Mesophilic bacteria (25–45°C) are preferred (Stronach, 1986), the treatment of brewery wastewater is also stable at lower temperatures (EBC, 2003).

Table 5: Volumetric organic loading rates for UASB, EGSB, and IC to treat brewery effluent (EBC, 2003; Austermann-Haun, 2009).

T [°C]	**Volumetric organic load rate [kg COD · m^{-3} · d^{-1}]**			**Source of information**
	UASB	EGSB	IC	
15–20	3–5	4–7	4–7	EBC (2003)
20–30	6–12	7–14	7–18	EBC (2003)
30–38	12–15	14–18	18–30	EBC (2003)
(other source)	5–10	13–27		Austermann-Haun (2009)

The temperature also effects the losses of the generated CH_4, that get released as dissolved fraction out of the reactor. CH_4 is about 1.5 times more soluble at 15 °C compared to 35 °C (Smith, 2012). Additional techniques for recovery of the dissolved fraction have been proposed, such as membrane separation or air stripping, however their implementation have not fully been evaluated in terms of process safety and economic feasibility (Liu, 2014).

2.3.3. Advanced Treatment with Nanofiltration

The term "nanofiltration" is derived from the relationship between the molecular weight of 200 g · mol^{-1} and a molecule size of approximately 1 nanometer (Melin, 2007). Nanofiltration membrane systems are in particular efficient for the separation ("cut-off") of molecules 300 g · mol^{-1} and work similar than a filter (Götz, 2013). The permeation process involves the selective transport of water with a fraction of smaller molecules across or through the membrane into the permeate, while retaining larger molecules in the retentate (Figure 14).

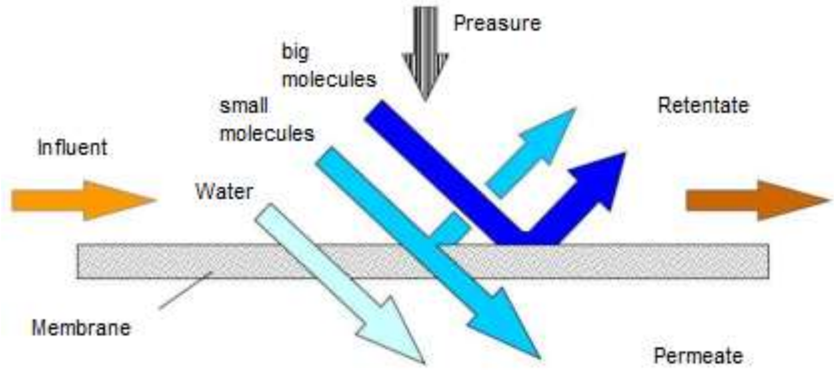


Figure 14: Sheme of a membrane seperatoin (Götz, 2013).

As depicted in Figure 15, the optimal molecular cut-off for a nanofiltration system, and the requisite pressure differential, fall within the range intermediate to those required for reverse osmosis and ultrafiltration processes. The pressure is equitant to the energy loss, the sum of transmembrane pressure and osmotic pressure (STOWA, 2020).

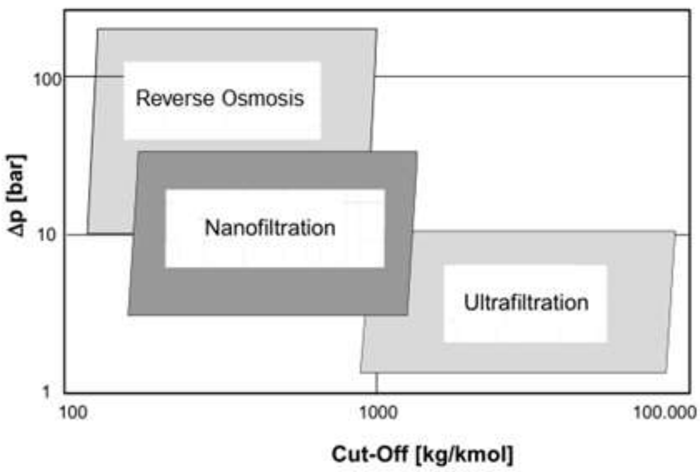


Figure 15: Working ranges of Reverse Osmoses, Nanofiltration, and Ultrafiltration (Merlin, 2007).

If the molecules that have to be (physically) separated are relatively large, a more 'open' membrane is possible, what offers the advantage that the same yield of permeate can be achieved at a lower transmembrane pressure (STOWA, 2020). The cut-off resulting in different retention efficiencies for ions. The retention of anions increases in the order of NO_3^-, Cl^-, OH^-, SO_4^{2-}, CO_3^{2-}, while for cations, the retention increases in the order of H^+, Na^+, K^+, Ca^{2+}, Mg^{2+}, Cu^{2+} (Merlin, 2007).

One reason for the implementation of nanofiltration systems in WWTPs is the goal of water reuse (Grady Jr., 2011). In theory, up to 30% of a brewery's water demand can be replaced by recycled water of nanofiltration systems (Kunzmann, 2008). Membrane technologies can be used inside breweries to reduce losses during the brewing process or as alternative to conventional secondary clarifiers in WWTP (Fillaudeau, 2006). In contrast, downstream nanofiltration membrane modules provide a tertiary treatment stage and the reusable water (as permeate) is gained from the final wastewater stream (retentate). These systems require for pre-treatment a micro sieve (100–200 µm) to prevent clogging (STOWA, 2020). The practice of water recycling proves to be highly cost-effective, in particular for situations where the availability of fresh water is constrained (Götz, 2013).

Nanofiltration technology can be classified as either 'spiral wound' or 'hollow fibre' (capillary). Spiral wound modules are not recommended for wastewater treatment due to susceptibility to contamination and significant pre-treatment requirements (STOWA, 2020). Hollow fibre membranes operate within a lower range of 1–6 bar, making them suitable for wastewater temperatures in the Netherlands, compared to the 'standard' NF systems that operate at pressures between 3–20 bar. Additionally, a recirculation pump is necessary, which generates up to 0.5 bar per module pass (STOWA, 2020). The diameter of the hollow fibre nanofiltration membranes utilized in treating effluent from wastewater treatment plants ranges from 0.5 to 2 mm, with a lifespan of 3–10 years (STOWA, 2020). Table 6 shows the electricity and chemical demand for a 54,891 m^2 modular membrane system.

Table 6: Electricity and chemical demand, 54,891 m^2 nanofiltration system (STOWA, 2020).

Parameter	Unit	Amount
Electricity	$kWh \cdot a^{-1}$	1,670,481
NaOH	$kg \cdot a^{-1}$	3,923
NaOCl	$kg \cdot a^{-1}$	2,556
Citric Acid	$kg \cdot a^{-1}$	7,644

2.4. The Method of Life Cycle Assessment (LCA)

The methodological framework of this study is based on the LCA method. In the following the approach is clarified, contrasted, and explained by its specific background, structure and evaluation steps.

2.4.1. Background of the LCA method

Life Cycle Assessment (LCA) has a history dating back to the 1960´s, initially focusing on the reduction of raw material and energy consumption (Guinée, 2011). The field of WWT began applying LCA in the 1990s to address a broader perspective of environmental impacts beyond direct water pollution (Corominas, 2013). The use of LCA can be related to research needs, design and operational improvements, or to provide information in order to facilitate decision making and stakeholder communication (Colominas, 2020). General principles for conducting LCAs are international standardized by ISO 14040 (2006) and specific guidelines in ISO 14044 (2006). Further guides and approaches on the method are supported by the European Commission (2010), and the United Nations (UNEP, 2018).

While there are several different methodologies available to evaluate the effects of WWTPs on the environment, such as the Environmental Impact Assessment (EIA), Material Flow Analysis (MFA), and various other concepts of environmental management techniques, the LCA method is beneficial for research on the broader environmental impact of new and emerging WWT technologies and to identify hotspots and unintended impacts, as well as shifting burdens between different impact categories and life-cycle stages (Corominas, 2020). A methodological comparison between LCA, EIA, and MFA can be seen in Table 7.

Table 7: Comparison of evaluation methods for the environmental impact

Assessment factors	LCA	EIA	MFA
Environmental impact data	yes	yes	no
Technology comparison	yes	partially	yes
Life-cycle & indirect effects	yes	partially	yes
Global or local impact focus	more global	more local	-
Hotspots & shifting burdens	yes	partially	yes
Legal impact	no	yes	no
Source: **(**of information**)**	(ISO 14044, 2006; Finnveden, 2009)	(Larrey-Lassalle, 2017)	(Birat, 2020)

A major weakness of the LCA method can be seen in the complex data demand and its extension in space and time by aggregated data values with a global focus, what leads to a rather unspecific and inappropriate consideration of the local ecosystem in comparison to the EIA or an Environmental Risk Assessment (ERA), that can set legal standards (Teodosiu, 2016). Results of an LCA reflect relative information and have no relevance to determine the actual impact, security margins, threshold value exceedances, and risks (ISO 14040, 2006). MFA has influenced LCA approaches and can include upstream and downstream processes of the supply chain and the end-of-life stage, however it mostly tracks physical mass and energy flows and does not directly address the corresponding impacts on the environment (Birat, 2020).

2.4.2. LCA Structure and Principles according to ISO 14040 and ISO 14044

Life cycle assessment (LCA) is a technique to address environmental aspects and to quantify potential environmental impacts associated with all the stages of a product, service or process from cradle-to-grave (Life cycle) (ISO 14040, 2006). The structure of a LCA consists four main steps (see Figure 16): The goal and scope definition, the inventory analysis (LCI), the impact assessment (LCIA), and the interpretation in reference to the other phases and the LCA results.

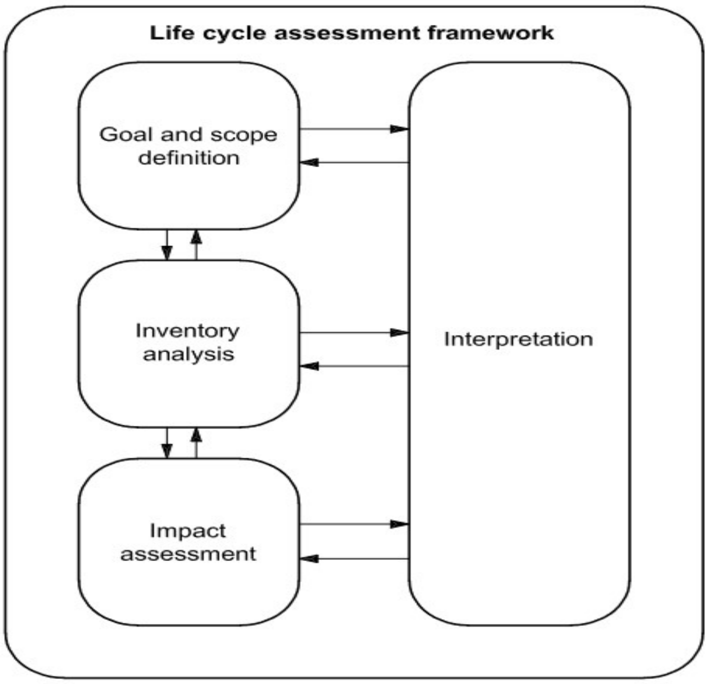


Figure 16: LCA framework according to DIN EN ISO 14040 (2006), modified

The work on the different phases is interconected (Iterative approach). The method is guided by seven main principles listed in Table 8.

Table 8: Seven principles of the LCA according to DIN EN ISO 14040 (2006).

Key principle	**Description**
Life cycle perspective	The life cycle stages consider raw material extraction and acquisition, production, use, and disposal as well as related upstream and downstream processes (including auxiliary materials, energy provision, and transport) (Förtsch, 2015).
Environmental focus	LCA addresses impacts on the natural environment, human health, and resources, but doesn't cover economic and social aspects (ISO 14040, 2006).
Relative approach and functional unit	All input (e.g. resources) and output (e.g. emissions) flows are calculated relative to a reference flow measured in a functional unit, that characterises the function the product system is providing (Finnveden, 2009).
Iterative approach	The work on LCAs follows an iterative approach, which means that new information about the product system is generated during the processing, and may require adjustment of previous steps (Finkbeiner, 2006).
Transparency	In order to ensure the proper interpretation of the LCA results, the communication of an LCA should include information about its boundary conditions, clients and audience, and should conform to the requirements outlined in ISO 14044 (2006).
Comprehensiveness	The LCA method aims to cover all aspects and impacts on the natural environment, human health, and resources in order to consider potential trade-offs by its cross-media perspective (Finkbeiner, 2006).
Priority of scientific approach	According to ISO 14040 (2006), decisions that have to be made within a LCA should strive to adhere to scientific approaches and international conventions, with a preference for those based in natural science. If there are no scientific basis possible value decisions may be utilized within the LCA (Finkbeiner, 2006).

Comparative assertions, require higher standards to achieve a formal conformity with ISO 14040 (2006) if they are disclosed to the public, including a “critical review” process by an external panel of experts. Since most LCAs in the field of wastewater treatment are carried out for research or reporting, a formal recognition by the ISO-standard is often not required (Remy 2020).

2.4.3. Goal and Scope Definition

The initial phase, the methodological framework is determined and transparency is ensured related to subjective decisions made about the goal and scope of the study (Finkbeiner, 2006). At the beginning of this phase, information is provided about the intended application, purpose, and audience of the LCA (ISO 14044, 2006). It has to be mentioned if the LCAs is intended for internal or external use and if comparative assertion is planned to be disclosed to the public (European Commission, 2010). The Scope of the LCA describes the deliverables of the LCA, the product systems and its processes, reference flows, and system boundaries, the completeness requirements, the cut-off-criteria, the modelling inventory framework, the handling of multifunctional processes and products, the chosen methods of environmental impact categories, and the data quality requirements (European Commission, 2010).

Functional unit and reference flow

The comparison between two product systems can only be justified if they have the same function (Guinée, 2002). The essential qualitative and quantitative aspects of the systems function are described as functional unit in the scope of the LCA and provide the basis of the further scaling of the data flows (ISO 14040, 2006).

Common functional units for the WWTP operation are related to the metric volume of the treated wastewater or to the composition of mass and flow produced equivalent by one person per day (PE) in the municipal WWT sector (Byrne, 2017; Tiwari, 2022). It is recommended that legal requirements or treatment objectives (Teodosiu, 2016) and a temporal and spatial dimension have to be addressed (Corominas, 2020).

Product system and system boundaries

Product systems are described as models of all relevant key elements of the physical system to provide the analysed product function through its entire life cycle (ISO 14040, 2006). Depending on the goal definition, a description of the product systems and the analysed scenarios can be given with particular focus to cover all life cycle stages (European Commission, 2010). Guinée et al. (2002) defines three types of boundaries of LCA product systems:

- Between the technical system and the environment
- Between the technical system and other technical systems
- Between significant and irrelevant processes

The boundaries also imply a temporal and graphical dimension and require equivalent definitions for comparability (Tillmann, 1994). According to Lundin et al. (2000) and Corominas et al. (2020) the system boundaries of a WWT system are often set in one of the ranges that can be seen in Figure 17, sorted by an alphabetical order.

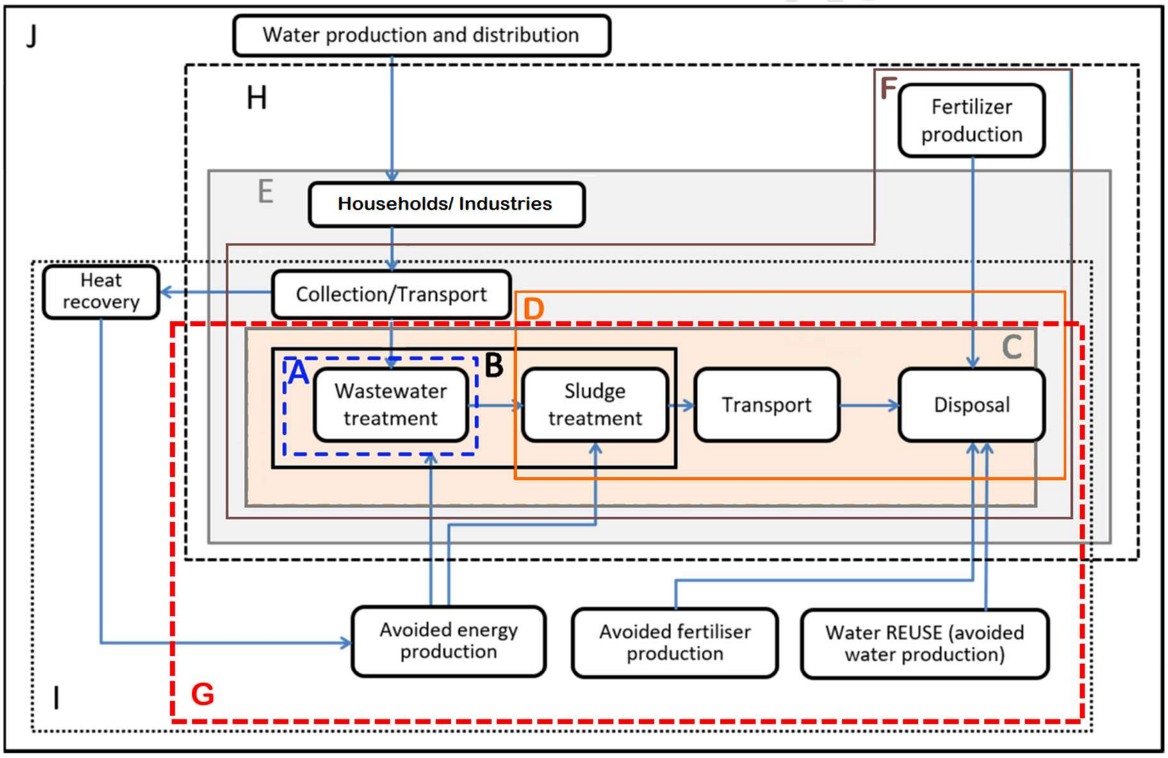


Figure 17: Different concepts for system boundaries for LCAs in the wastewater industries (Corominas, 2020)

Approach A only analyses the wastewater treatment, while approach B also considers the sludge treatment at the plant. Approach C and D also include transport and sludge disposal. Approach E considers the contributor of the wastewater flow, and its collection and transport systems. It is preferred that the system boundaries also include the fate of sludge generated during the wastewater treatment operation (Tiwari, 2022), like in the approaches F and H. In addition, heat recovery from the raw wastewater and the concept of avoided burden in terms of energy, fertilizer and water resources can be implemented (I, J). A similar approach can be taken (G), that excludes internal processes of the brewery and potentials for heat exchange, due to limitations of the scope of this LCA.

Furthermore, upstream and downstream processes concerning chemical supply, construction material, and electricity production are recommended to be included in the system (Corominas, 2020). In contrast, maintenance operations like cleaning and oiling engines as well as the entire deconstruction end-of-life stage of a WWTP can be considered neglectable (Corominas, 2020). Depending on the goal of the LCA, it may be sufficient to exclude certain parts of the life cycle. It should be noted that such partial analyses do not represent ISO standard (Kaltschmitt, 2015).

The complexity of product systems makes it impractical and impossible to consider all material and energy flows within the system boundaries and to predict their significance. As such, the implementation of additional assumptions and defined 'cut-off criteria' helps to prevent the collection of an exorbitant amount of insignificantly impactful data by keeping a symmetric and transparent approach in the LCA (ISO 14044, 2006). The criteria can be defined as a predetermined percentage to which material and energy flows must conform, either in terms of mass or their environmental effects on the system (Kaltschmitt, 2015). Processes that contribute at least 5% of the overall impact have to considered, while the completeness of the data sets has to be compromised with the quality and precision of the LCA data (European Commission, 2010). In the event that relevant data cannot be obtained during research, for example due to unavailability or inaccessibility, it is essential to explicitly mention this in order to maintain maximum transparency (Kaltschmitt, 2015).

Attributional or Consequential inventory framework

The main modelling principles of LCI methods can be distinguished between attributional and consequential LCA approaches (European Commission, 2010). The first one describes a system and the environmental impact attributed to it in a static technosphere with an average set of existing, calculated, estimated, or forecasted data (European Commission, 2010). In contrast, consequential LCAs aims to consider potential changes in several environmentally relevant flows that may occur in response to decisions in presence of a dynamic market or longer time spans what requires marginal data and expert knowledge (Finnveden, 2009).

Allocation and Offsetting

It has to be specified if allocation is needed to divide inputs and outputs, that are not direct proportional, to different parts of the system (ISO 14040, 2006). Three situations can create the need for allocation (Finnveden, 2009):

- Multi-Input: Several input products refer to a common output.
- Multi-Output: The inputs are divided among several valuable products.
- Open-loop Recycling: A waste product is recycled to a secondary product.

In general allocation is preferred to avoid, if possible, by a subdivision of the multifunctional process (ISO 14040, 2006). The Situation of Multi-Output can be solved by the implementation of an additional reference product that leads to a system expansion or a substitution that can be systematically subtracted as offsetting credits related to the avoided burden (European Commission, 2010). The later can create a dependency on the chosen subtracted replacement product and is a common method in the field of WWT where co-products like biogas can be generated by anaerobic digestion or biosolids of the sludge production be used for agricultural applications (Heimersson, 2017).

Data quality

A description and evaluation of the data quality is taken out in order to interpret the reliability of the results (ISO 14040, 2006). The data quality can be evaluated by representativeness (composed of technological, geographical, and time-related), completeness (regarding impact category coverage in the inventory), precision or uncertainty (of the collected or modelled inventory data), and methodological appropriateness and consistency (European Commission, 2010). It is important to ensure the credibility of these sources by verifying their accuracy and reliability. In addition, establishing personal contact with participating companies can allow for more detailed and accurate data to be obtained (Corominas, 2020). Assumptions and missing data are recommended to be mentioned (Guinée, 2002).

2.4.4. Analysis of the Life Cycle Inventory (LCI)

Aim of the Inventory analysis is to comprehensively consider material and energy flows within the defined system boundaries and product system (ISO, 14040). It is necessary to identify relevant processes within the scope of the investigation and to quantify them by collecting, analysing and reconciling data (Corominas, 2020). During this phase, data availability and data quality have to be compromised. This may result in inconsistent and mixed LCI modelling approaches in terms of data gaps, geographic averaging, and product resolution (Corominas, 2020). The resulting material balance can be conceptualized as “inventory tables”, comprising process-related inputs and outputs (Finnveden, 2009).

In the field of WWT additional data for the inventory is usually generated by extrapolating existing data, analogies to other WWTP of similar size and related wastewater quality characteristics, laboratory or pilot facilities, construction guidelines, expert estimates, relevant literature, and design software (Corominas, 2020). Background information, such as information on electricity generation systems and production processes for chemical supply and construction material, can be obtained from LCI databases like “EcoInvent” (Corominas, 2013). It is recommended to include inventory tables for construction, chemical consumption, electricity demand, and direct emissions in the field of WWT (Corominas, 2020).

Construction

The smaller the WWTP, the more relevant is the impact of construction and the required level of detail (Corominas, 2020). Tanks and large equipment parts like blowers, pumps, and dewatering belt filters, should be accounted by the mass of the primary materials like concrete, steel or copper (Corominas, 2020). According to Table 9, based on the United States Environmental Protection Agency (USEPA, 2002a) data, the following lifespan can be estimated for WWTP infrastructure:

Table 9: Lifespan of WWTP infrastructure (USEPA, 2002a)

Infrastructure Type	Lifespan
Concrete structures	50 years
Equipment and installations	15–25 years
Pipes	15–100 years

The calculation of the infrastructure in LCA studies on WWTP depend and differ strongly on the chosen lifespan of the infrastructure (Morea, 2017). Some equipment and installation parts can have significant lower lifespans like modular filtration systems with lifespans of 5–7 years (Tawari, 2022).

Electricity

The chosen electricity mix impacts strongly the overall LCA result and has to be documented (Corominas, 2020). In the first iteration of the electricity consumption bills and literature can be suitable, while for the comparison of different scenarios with a calculated model the modelling data of the consumption related to inputs and outputs shall be taken (Corominas, 2020).

Chemical consumption

Data on dosing chemicals can be obtained through the WWTP operator, direct reports, modelling, and literature, with preference from the first to the last respectively (Corominas, 2020). LCA databanks don’t provide a full coverage of all chemicals of the WWT industry (Corominas, 2020). Current LCA guides recommend to calculate with analogue substances (European Commission, 2010).

Direct emissions

Different and inconsistent estimations of emissions exist in the literature due to different modelling approaches (Heimersson, 2017). Emissions to the air are often not considered in LCAs (Corominas, 2013). They are difficult to measure, predicted with high uncertainty, and highly variable even within a WWTP (Corominas, 2020).

Since COD is mostly of biogenic origin, the microbial CO_2 production of the WWTP is not accounted for greenhouse gas emissions (Corominas, 2020). However, this does not apply for the release of carbon as different greenhouse gases like CH_4. CH_4 and H_2S emissions are related to the process of anaerobic digestion, while N_20 emissions are associated with the process of nitrification (DWA, 2022a).

2.4.5. Life Cycle Impact Assessment (LCIA)

The impact assessment (LCIA) is a key stage of LCA that translates raw LCI data to a set of specific environmental impacts. As can be seen in Figure 18 a series of binding and optional steps are assessed according to ISO 14040 (2006).

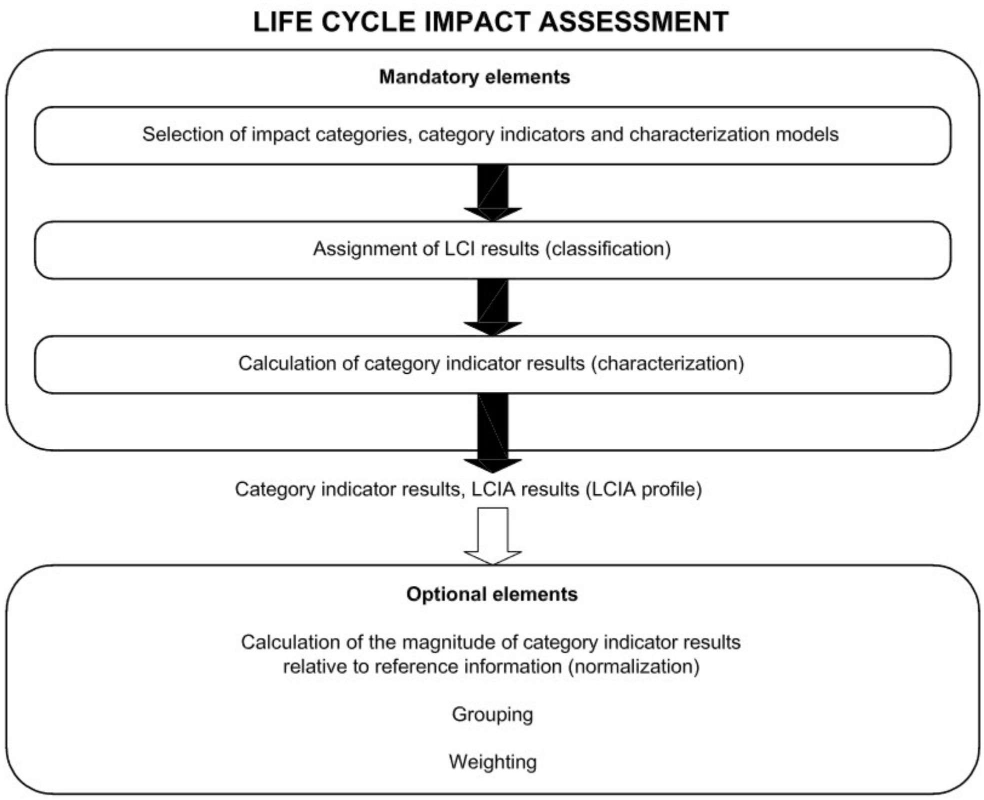


Figure 18: Mandatory and optional parts of the Life cycle Impact Assessment (LCIA) according to DIN EN ISO 14040 (2006).

After relevant impact categories and models have been selected, a “classification” of the LCI results is carried out that assigns the LCI results to the chosen environmental impact indicators. In addition, a “characterization” is performed, where the LCI results are quantified by corresponding impact indicator values (e.g. GWP as indicator for climate change is calculated). Indicators refer to systematic characterisation models, and represent specific concerns about three areas of protection (environment, the human health, resource depletion (ISO 14044, 2010).

Optional stages of the LCIA include “normalization” – the scaling of results to a reference value; “grouping” – a hierarchisation, sorting or ranking of impact categories; and “weighting” – aggregation of graded results (European Commission, 2010). According to the DIN EN ISO 14044 (2006) grouping and weighting are not part of comparative LCAs disclosed to the public, since they depend on value choices and not on a scientific base (ISO 14040, 2006).

The type of LCAI can be divided into midpoint and endpoint level approaches (Huijbregts, 2017). Figure 19 shows the classification of the LCI results to midpoint level environmental impact categories (e.g. the classification of CH_4 emissions as relevant for climate change and summer smog) and an optional further assignment of the midpoint level indicator result to endpoint categories (e.g. climate change is measured in the GWP and can be assigned to damage related to the human health as well as the diversity of the ecosystem).

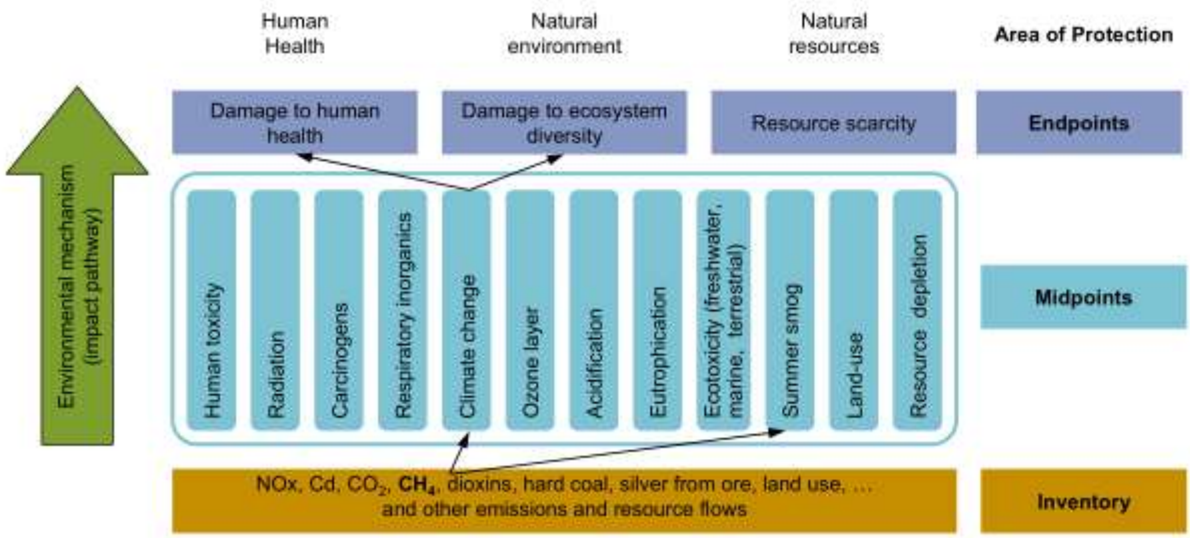


Figure 19: Midpoint and endpoint classification (European Commission, 2010).

LCAs that focus on the midpoint level results differentiate a large number of impacts and generate more accurate results (European Commission, 2010). Midpoint approaches are more common for WWTPs and often conducted by characterisation methodologies of CML or midpoint ReCiPe (Byrne, 2017). Key midpoint categories in the field of WWT refer to climate change, eutrophication, and ecotoxicity (Corominas, 2020). Furthermore, human toxicity, terrestrial acidification, ozone layer depletion, photochemical oxidation and the cumulative energy demand are associated with LCAs within the field of WWT (Tiwari, 2022; Remy, 2012). The use of common LCIA categories, databanks, and models with default characterisation factors substantially improves the comparability of LCAs (European Commission, 2010). In contrast, endpoint level approaches create categories closer to the end of the causality chain that represent the ultimate effect or damage to the aeras of protection, but conflict with a lower level of certainty in the result (Kaltschmitt, 2015).

2.4.6. Interpretation Phase of the LCA

The fourth phase of the LCA serves as a final critical examination of the study. This is where a summary, conclusions, and explanations of the LCA results take place and recommendations and limitations have to be considered (Finkbeiner, 2006). The results of the LCI and LCIA are interpreted in reference to the goal and scope of the study and analysed in terms of the accuracy, completeness, and precision of the data used and the assumptions made throughout the study (European Commission, 2010). According to the DIN EN ISO 14044 (2006) standard the interpretation proceeds through three activities:

1. The identification of significant issues based on the LCI and LCAI results. This may include to emphasize main contributors, the difference between Life Cycle stages and choices related to the precision of the final result (European Commission, 2010).

2. An evaluation of the reliability and robustness of the results is carried out by considering completeness, sensitivity, and consistency checks. Further steps can be supported by uncertainty and scenario analysis. Sensitivity analysis test LCIA results by varying certain parameters of the life cycle inventory, and are recommended for LCAs related to the field of WWT (Corominas, 2020). The choice of impact categories as well as the exclusion of relevant impacts shall be considered in the interpretation of the results, potentially limiting conclusions and interpretation of the study (European Commission, 2010).

3. Finally, conclusions are drawn based on the analysis of the data collected. The limitations of the study have to be considered and recommendations for future research and practical application can be provided (ISO 14044, 2006).

3. Methods and Material

The primary method used in this study is the LCA. In comparison to other evaluation methods like MFA and EIA, LCA is particularly suitable for the investigation of the research questions in this study, as it allows for the identification of focal points of the environmental burden with a comprehensive and holistic approach, along with the establishment of comparative analyses between technologies or wastewater treatment concepts. Furthermore, it allows the consideration of indirect effects of the entire process chain as a research approach (DWA, 2022b).

This chapter is divided into three parts (Figure 20). According to RQ2 a comparison for the root IFAS application in Koningshoeven has to be conducted and evaluated by the LCA method. The related reference technologies are selected in the first step. This is followed by the goals and scope definition of this LCA study. In the third part of this chapter, the input data are presented or calculated and the modelling of the two reference benchmarking scenarios data is performed.

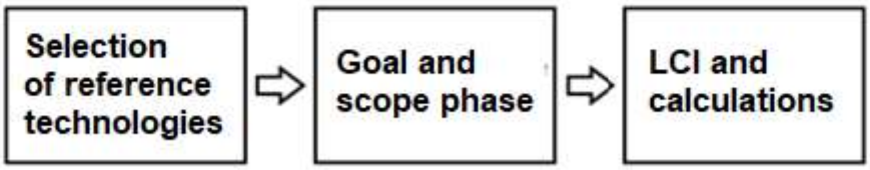


Figure 20: Overview of Chapter 3.

3.1. Selection of the WWT technologies for scenario definition

The comparison shall provide different treatment options as reference scenarios, fitting to the boundary conditions of the brewery at the abbey of Koningshoeven.

The aim of the wastewater treatment in Konigshoeven is to meet legal requirements for direct discharge into a local river. The current root IFAS process has been emphasized by the literature for its adaptability to shifting wastewater characteristics, low space requirements, and to provide good sludge settling abilities (STOWA, 2017). The ecosystems create a resilient biodiversity of over 3,000 different species and provide improved process stability and adaptability to the fluctuation of the organic loading rate as well as the further ability to efficiently break down even hardly degradable components (STOWA, 2017). Low investment and operational cost have been expected by the European Nextgen research project (Poór-Pócsi, 2021). A Dissolved Air Flotation (DAF) unit has been installed in order to reduce further space requirements, while two 200m^3 tanks for equalization, chemical dosing to stabilise the pH-value, and hydraulic balancing are provided to work with the discontinuous production cycles and raw wastewater effluent flows of the brewery.

Both aerobic and anaerobic biological processes can be used to treat brewery wastewater (DWA, 2010). The most common reference in LCA studies for aerobic treatment of industrial wastewater is the CAS (Corominas, 2020). Among aerobic processes, besides CAS, discontinuous processes with suspended biomass, i.e. SBRs, or biofilm processes such as tickle filters or packed bed bioreactors can be recommended to treat industrial brewery wastewater in a technical biological stage (DWA, 2010). Similar advantages for the root IFAS have been reported by other biofilm systems for the treatment of brewery wastewater, including relatively low operational costs and space requirements as well as a fast recovery from shock loads (DWA, 2010). Since modelling biochemical processes in biofilms is rather complicated due to substrate gradients (Horn, 2014), a simpler approach or a different technology has to be taken. In comparison to CAS, SBR systems are considered particularly suitable for industrial wastewaters with high content of carbohydrates (Wilderer, 1986). In consideration of the strong variation in loading rates and wastewater production cycles in small and medium-sized breweries (EBC, 2003), the SBR system was chosen as a reasonable alternative scenario due to its adaptability to the fluctuations, that can regulate the reaction time and the wastewater inlet or exchange rate (DWA, 2009). The research study of Sharda et al. (2013) expected the implementation of SBRs to be more beneficial related to operational costs and space requirements than CAS for the treatment of brewery wastewater. Reduced space requirements can be achieved by a SBR, since no DAF or secondary settler is required (Metcalf & Eddy, 2013). The SBR process is characterised by good sludge settling abilities, and process stability (DWA, 2009; Wilderer, 1986). Another benefit is the biological adaptation to prevent sludge bulking (Wilderer, 1986), that can be challenging for brewery WWT (DWA, 2010). The competition between facultative anaerobic bacteria and denitrifying bacteria, as well as phase shifted water realisation can result in elevated effluent rates and the release of ecotoxic nitrate (Wilderer, 1986). The effect of this does not seem crucial, due to the low nutrient content of brewery wastewater.

Wastewaters from breweries are well suitable for anaerobic digestion, due to their relatively high organic loading rates, their high BOD/COD ratio, and no further need to heat up for high temperatures (Bischhofsberger, 2005; EBC, 2003). However, anaerobic treatment stages require high investment cost around 500.000–700.000 million Euro for a maximum flow rate around 200 m^3 per day (EnviroChemie GmbH, personal communication, 10th august 2022; Craveiro, 1987), that can be considered challenging for small and medium-sized breweries, but usually result in lower operational costs, due to low energy requirements and the use of biogas (Bischofsberger, 2005). The EGSB variant is often preferred by new investments over the standard UASB version due to higher volumetric COD loading rates, related to low space requirements (DWA, 2010; EBC, 2003). Anaerobic sludge bed reactors require relatively low TSS loading rates ($<500\ mg \cdot l^{-1}$) and pH buffering to a neutral setting (Stronach, 1986). While an equalizing tank is already installed in Koningshoeven, an additional pre-treatment step to reduce TSS loads has to be implemented for anaerobic biological treatment by sludge bed reactors. Related to the higher upflow velocity, EGSB are considered less vulnerable to TSS influent loads than standard USAB reactors (Bischofsberger, 2005). Anaerobic WWT can be characterised by relatively low sludge production, but also require an additional aerobic treatment stage to meet legal discharge limits (DWA, 2010). In a resulting scenario, an anaerobic treatment stage can be compared to the root IFAS. However, such a scenario has to implement additional biogas infrastructure, and to be followed by an aerobic treatment stage.

The overall result for the choice of the alternative scenarios is presented in Table 10. Some benefits of the root enhanced IFAS, like its visual attractiveness (Hetem, 2016) might be its unique selling proposition and cannot be implemented in the comparison.

Table 10: Comparison of treatment technologies

Assessment factors	root IFAS	CAS	SBR	Biofilm	EGSB
Low space requirements	(+) / ?	0	+	+	+ +
Sludge settling ability	+	0	+	+	+ +
Robust to shock loads	+	0	+ +	0	0
Low investment costs	+	+	+	+	− −
Low sludge production	(+) / ?	0	0	+	+ +
Visual attractive	+ +	−	−	−	−
Difference from root IFAS	(− −)	+	+	−	+ +
Less complex modelling	− −	+	+	− −	+
Choice for reference			**SBR**		**EGSB-SBR**

3.2. Goal and scope (LCA)

In this chapter the goal and scope phase of the LCA is presented in order to ensure transparency and to meet the methodical requirements.

Goal definition

The LCA study has the goal to model and analyse the environmental impact of the root IFAS for brewery WWT at the Koningshoeven brewery and to conduct a comparative assertion with three reference scenarios – the SBR scenario, the EGSB-SBR scenario, and the root IFAS-NF. The root IFAS-NF consist of an additional sub-sequential nanofiltration membrane unit designed by cooperation partners for the goal of water reuse. The focus of the four scenarios are potential environmental impacts related to the WWT operation and the sludge handling. Shifts, burdens, and main contributors related to different environmental impact categories are investigated for the compared scenarios.

Background

The reason for the study is to provide scientific information on the breweries new WWT system as an example of the application on small and medium-sized breweries. The scenarios also illustrate various distinct offsetting possibilities in a representative and instructive manner. For the general comparison the recycled water of the root IFAS-NF scenario is credited by the avoided burden of the freshwater supply of the brewery. However, the difference between the root IFAS and the root IFAS-NF scenario is also tested by a second crediting scenario, with a reference to Dutch tap water supply. The targeted audience are university students, scientists, researchers, plant operators, and engineering companies related to the field of industrial WWT, LCAs and sustainability. The study cannot reach confirmation with the ISO standard, but may be disclosed to the public.

System function and functional unit

The function of the investigated product systems is the treatment of brewery wastewater with specified water quality and quantity standards. All systems relate to the same influent flow rate of brewery wastewater and have the same reference.

According to the WWTP operator the daily flow rate in 2022 of the operation is about 150 $m^3 \cdot d^{-1}$ and the operation days account 350 $d \cdot a^{-1}$.

The functional unit is described as: treatment of 52,500 m^3 industrial brewery wastewater in a timespan of one year with the attributed wastewater influent quality characteristics of the Koningshoeven Brewery in 2022 (3725 mg COD $\cdot$ l^{-1}, 438 mg TSS $\cdot$ l^{-1}, 46 mg TN $\cdot$ l^{-1}, 5.1 mg TP $\cdot$ l^{-1}) by achieving legal treatment objectives (125 mg COD $\cdot$ l^{-1}, 10 mg TSS $\cdot$ l^{-1}, 10 mg TN $\cdot$ l^{-1}, 1 mg TP $\cdot$ l^{-1}).

System boundaries

The analysed systems include all relevant processes for the brewery wastewater and sludge treatment. This includes the operations at the treatment plant, the external chemical production and sludge treatment, the transport of the sludge and the chemicals, the electricity generation in the Netherlands, and the production of construction and infrastructure installation materials. The construction is considered in terms of the needed materials. Construction works, provided energy for the buildings, maintenance like oiling engines and cleaning rooms, and the entire deconstruction as end-of-life phase have been cut-off or considered as irrelevant for the comparison. The brewery contains an internal mechanical sieve and an additional balancing tank that are not considered part of the subsequential WWTP and therefore excluded in this study. Major direct emissions of the system into the environment are considered in this LCA as elementary flows such as direct wastewater discharges into the rural surface river water and gaseous emissions into the air, due to the anaerobic digestion and the external sludge disposal. The illustration of the system boundaries and the systems key process steps are found in the scenario description of this chapter. Co-products of the wastewater and sludge treatment process are implemented into the system by crediting the avoided burden of the production of the conventional product they replace, as described in the following part.

Description of the Scenarios

Four scenarios are compared in this study, each scaled to size for an average flow rate of 150 $m^3 \cdot d^{-1}$ and a maximum flow rate of 200 $m^3 \cdot d^{-1}$. The following four scenarios are:

- The current waste-water treatment of the brewery purifying greenhouse technology (root IFAS) as a baseline is compared with the following alternative scenarios
- the baseline system is extended with a capillary nanofiltration membrane for water reuse (root IFAS-NF),
- a common aerobic alternative of two phase-shifted sequencing batch reactors (SBR), and
- an anaerobic scenario with an extended granular sludge blanket reactor followed by an aerobe sequencing batch reactor (EGSB-SBR).

The scenarios, as illustrated in Figure 21 (p.41), Figure 22 (p.42), Figure 23 (p.43), and Figure 24 (p.44), are divided in the operations in Koningshoeven (like WWT, sludge dewatering, and chemical dosing), important upstream and downstream processes (external sludge treatment, chemical supply), and background processes (electricity production, construction materials). The blue line in the figures represents the water path, where the brewery wastewater follows different treatment steps and gets discharged into a local river channel used for irrigation of farmland. The dashed blue line refers to the return flow. The orange path refers to the sludge treatment and the dashed orange line represents the offsetting credits. The green path relates to the chemical supply chain, the yellow path to electricity and fuels, the brown path to construction materials and the grey path to air emissions.

The root IFAS scenario is illustrated in Figure 21 (p. 41). After equalising and pH control by chemical dosing in an influent buffer tank, the raw wastewater gets treated in the biological treatment step in a greenhouse. The root IFAS consists of a series of 16 tanks with four possible air blowers and mixers. 14 tanks are protruded by plants and supported by inlay structures of propylene. There are two different located effluent inlet points. The wastewater gets pumped into the solild-liquid separation phase, implemented by a DAF, followed by a micro sieve.

Additional polishing is guaranteed by an effluent buffer tank and a pond. The sludge runs through an aerated storage tank to the dewatering step by a discontinuous working belt filter press. The sludge gets transported to an external treatment anaerobic digestion for sludge and wastes from the food and beverage industries. In this step electricity is generated by a CHP and biosolids to replace mineral fertilizers are produced. Coagulant and flocculants are provided in the solid-liquid separation phase and additional polymer for the internal and external sludge handling processes.

In the root IFAS-NF scenario (Figure 22, p. 42) the first scenario gets extended by a capillary nanofiltration membrane as a post-treatment step. The permeate can be returned to the brewery as processed water and credited as substituted freshwater of the brewery. A further analysis is taken into account for the technology if the brewery has limited water supply, what can potentially make the NF more feasible (cf. Götz, 2013). The retentate gets released as wastewater and still complies with the discharge limits into the river channel. The membrane leads to a higher consumption of chemicals, electricity, and construction materials.

The SBR scenario (Figure 23, p.43) refers as a benchmark for a common alternative for small and medium-sized breweries. Compared to the IFAS scenario, the innovative biological treatment in the greenhouse and the DAF unit are replaced by a batch process with to two parallel SBR tanks.

The EGSB-SBR scenario (Figure 24, p.44) refers to a scenario with an advanced anaerobic treatment stage (EGSB), as a second benchmark. An additional pre-treatment step with a micro strainer was implemented in order to create lower TSS values to protect the EGSB operation. Biogas gets produced by the anaerobic stage, that can be stored and provided as heating agent inside the brewery. Gas infrastructure and gas losses due to the gas flare and the unintended release of CH_4 in the biological treatment phase have been considered. The EGSB is followed by an aerobic SBR and the same micro sieve, sequential polishing and sludge handling steps than in the other scenarios.

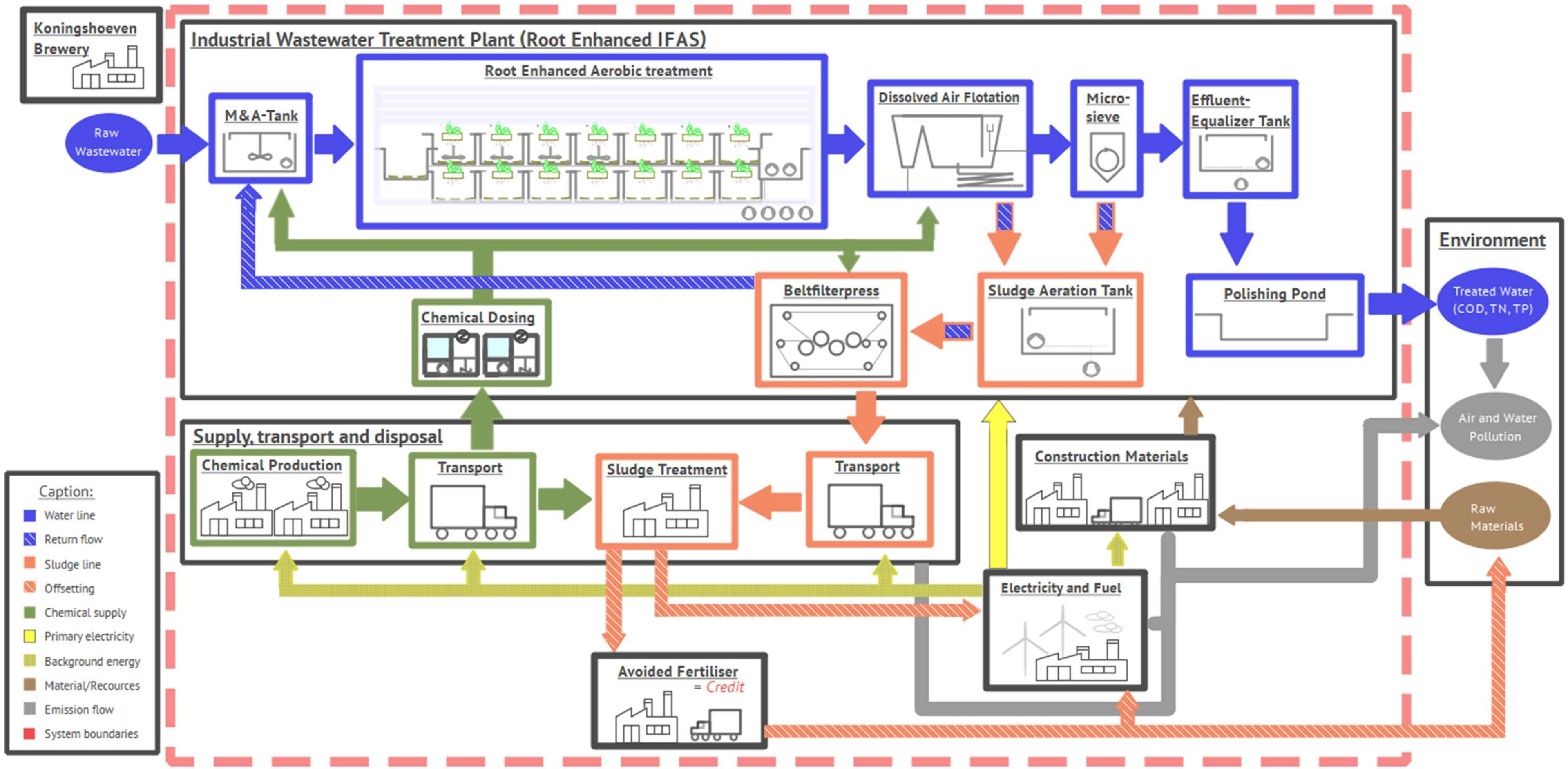


Figure 21: Root IFAS scenario LCA sheet with system boundaries

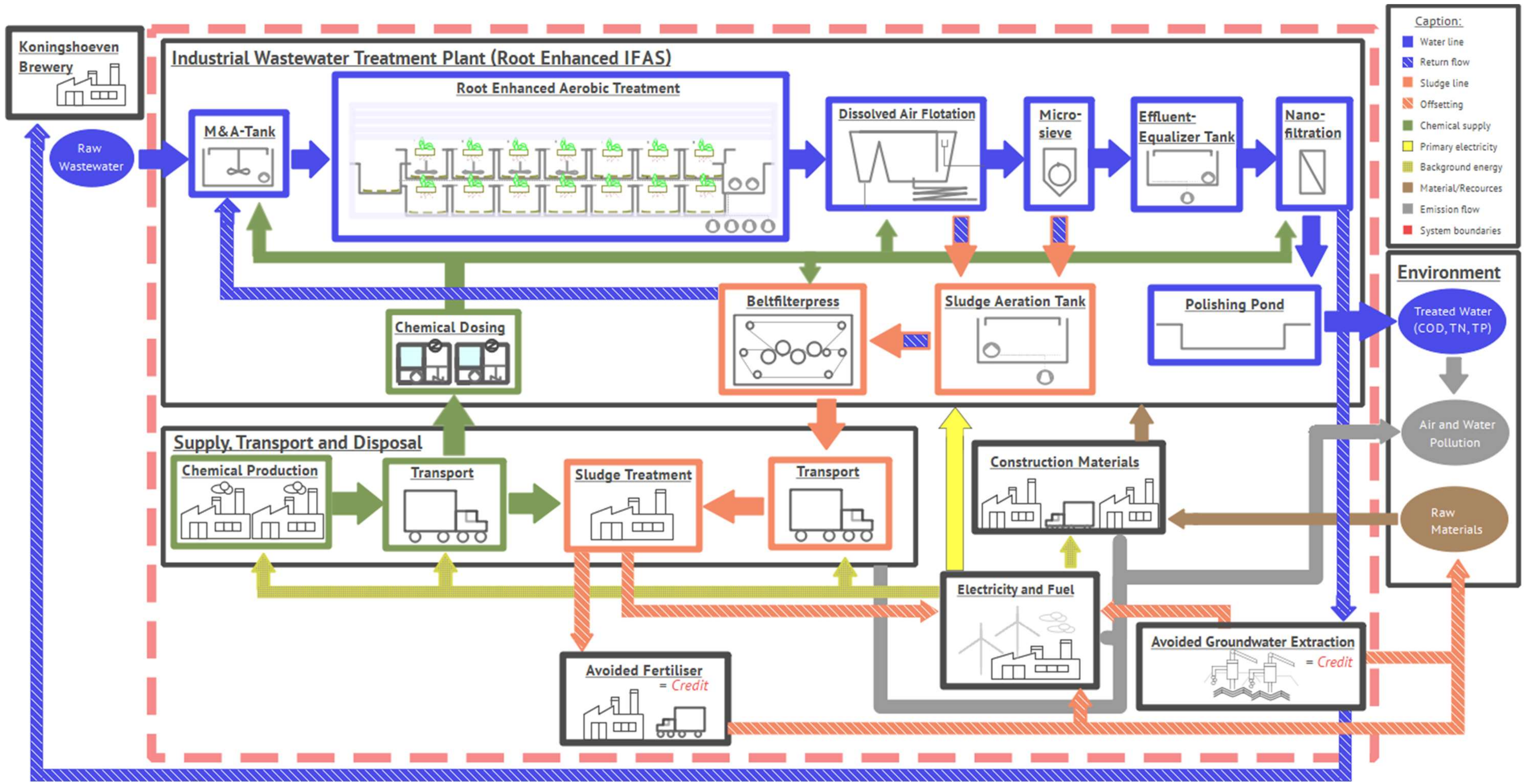


Figure 22: Root IFAS-NF scenario LCA sheet with system boundaries

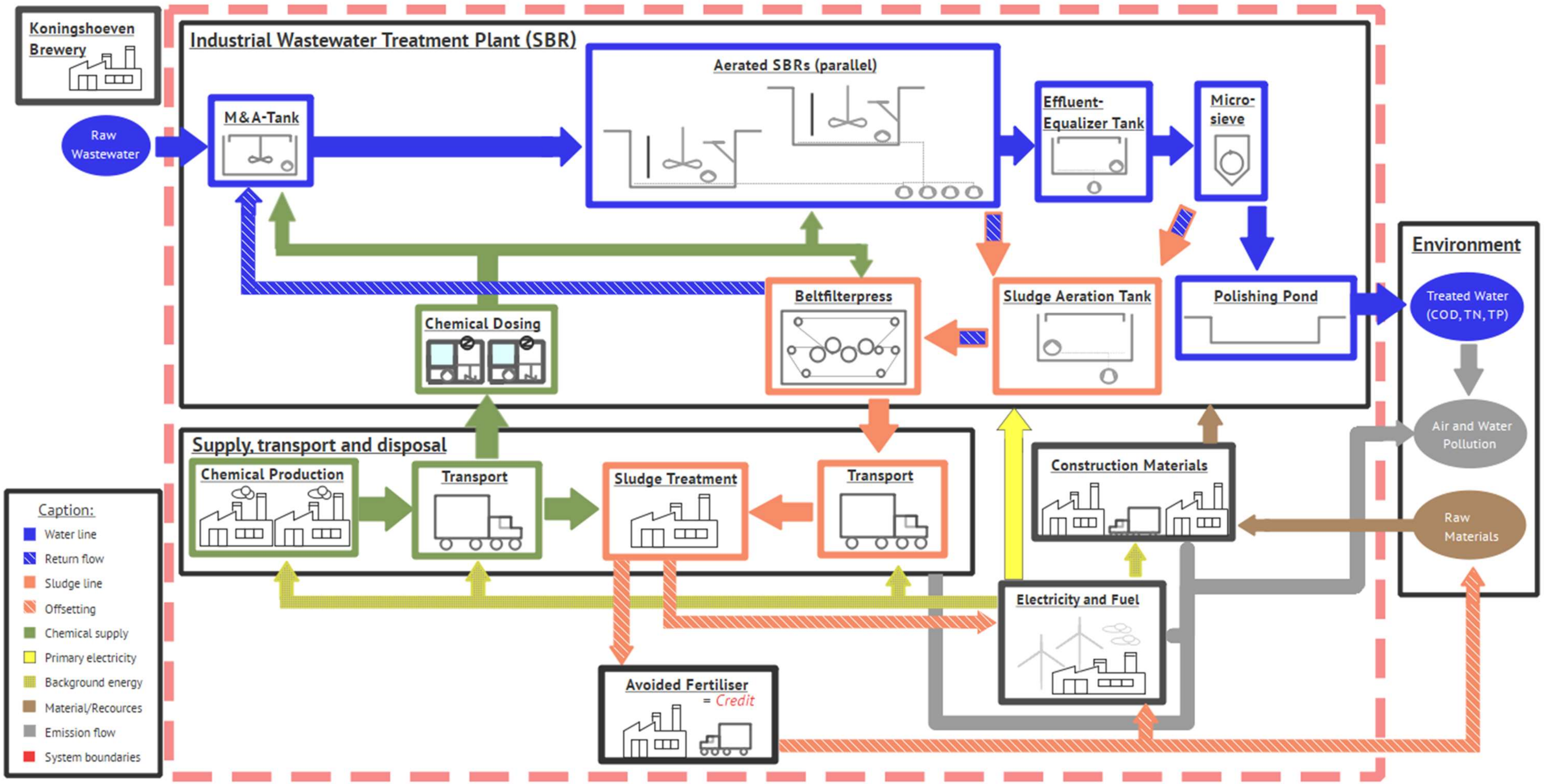


Figure 23: SBR scenario LCA sheet with system boundaries

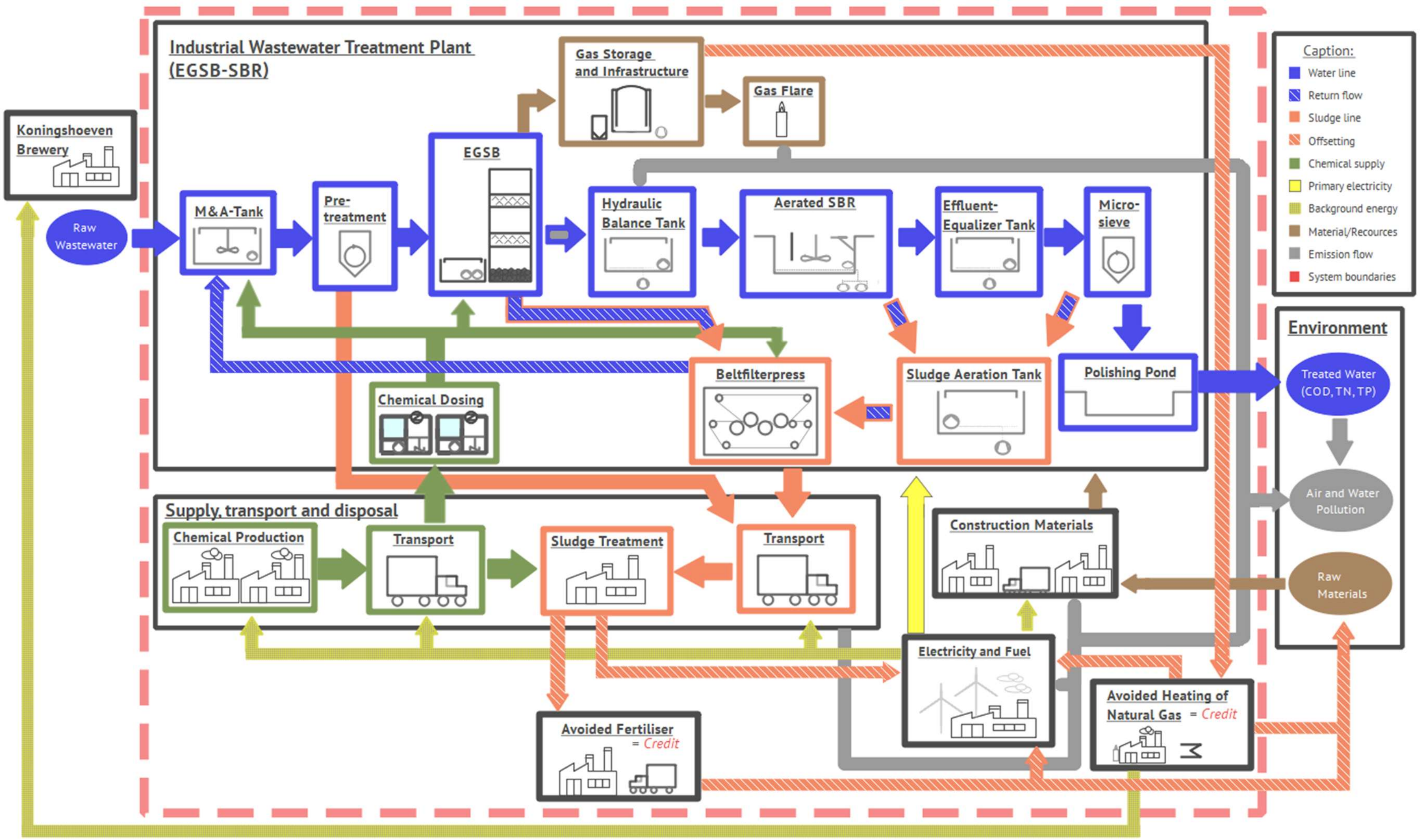


Figure 24: EGSB-SBR scenario LCA sheet with system boundaries

Allocation and offsetting

All environmental impacts of the system are related to the systems operation and functional unit in a one-dimensional way. Biosolids, biogas, and recycled water occur as by-products in this study. The allocation of these refined substances has been avoided in this study by the subtraction of the conventional product they substitute and its associated production processes. Biosolids replace mineral fertilizers. Biogas use replaces the heating of natural gas if generated near the brewery or provides electricity by an external sludge digestion that can has been credited by the same amount of the Dutch electricity grid mix. The recycled water of the subsequential nanomembrane replaces the ground water used by the brewery. In addition, alternative crediting options are investigated for the discussion and sensitivity checks.

Assumptions

The construction is accounted for the WWTP in Koningshoeven based on the primary materials related to the mass of tanks and large equipment installations, including the belt filter press, the DAF, pumps, dosing pumps, pipes, or the potential nanofiltration infrastructure. The Greenhouse and the inlays of the biological treatment stages are considered, because they refer to the main differences between the scenarios. For the EGSB-SBR scenario, gas treatment, security measurements and gas storage are included. A lifespan of 50 years has been chosen for the tanks and housings, while installations have been accounted by a lifespan of 15 years. However, construction works, including the required energy, have been considered neglectable for this lifespan. A sensitivity analysis is carried out for a lower lifespan of 30 years for the main construction. The end-of-life phase of the WWTP related to the demolition has been excluded in the LCA and considered irrelevant based on the assumptions in literature (Corominas, 2020). The construction for external structures outside of Koningshoeven have not been taken into account and considered irrelevant, if not included by estimations in the ecoinvent (2021) process data sets (compare Appendix A, p. 124).

The root IFAS system was originally designed for the treatment of an average flow rate of 320 $m^3 \cdot d^{-1}$ and a maximum capacity of 420 $m^3 \cdot d^{-1}$ (Poór-Pócsi, 2021). Together with the WWTP operator and experts of the KWB (2022) Berlin GgmbH it was decided on a stakeholder meeting to calculate the systems for the actual flow rate of 150 $m^3 \cdot d^{-1}$ in the year 2022 to be more consistent with the available data and of the current situation. To guarantee a fair comparison between the scenarios, relevant process data for the over-sized IFAS was adjusted to reflect an optimised operation of equipment (e.g. blowers, DAF) according to the original load design.

No direct measurements of gaseous emissions were taken. The methane loss of the EGSB-SBR-Scenario into the air has been estimated from the dissolved fraction with an analogy between UASB and EGSB reactors based on literature close to the theoretical result related to Henry's Law (Bandana, 2011). 50% of the dissolved CH_4 effluent of the EGSB were estimated to be released in the environment while the remaining part is assumed to be converted by methanotrophic bacteria. This assumption is tested by sensitivity analysis. Data for direct emissions related to the incineration of the biogas is taken from literature and cooperation partners. Direct N_2O emissions are difficult to estimate. The low COD to TN ratio of brewery wastewater (here around 80) results into nutrient dosing for assimilation of biomass rather than denitrification processes. Therefore, direct gaseous emissions from the biological phase like N_2O and NH_3 have been neglected. The CO_2 emissions of brewery wastewater have been considered biogenic and therefore indifferent to the impact on global warming. Other significant direct gaseous emissions like H_2S have not been possible to analysis due to large insecurities and data gaps for the plant root enhanced IFAS.

The effect of the polishing pond is assumed negligible. Direct emissions into the local river are calculated by the COD, TN and TP loads after the micro sieve or the nanofiltration phase.

Incomplete information of all chemical components of the polymer ST-FLOC 75 LCH of the STOCKMEIER Chemie GmbH & Co. KG, resulted in the replacement of the active matter by polyacrylamide, while the admixture of adipic acid remained according to the original data sheet. The transport data of the chemicals were calculated as product of weight and distance, according to the driving distance to the matching supplier by lorries. As a result, the transport of materials has been estimated for chemicals (150 km), for sludge disposal to the local digestor (10 km) and disposal to farmland (10 km), and materials for infrastructure (100 km).

Data sources and quality

The data quality of the different data sets is discussed in relation to the data sources of major input parameters for the four scenarios.

- General infrastructure (Koningshoeven): Original design assumptions have been checked and estimations are taken by visiting the full-scale plant in Koningshoeven.
- Root IFAS: Data sets related to water quality, nutrient dosing, and sludge production, are based on process modelling (SUMO software) by the technology provider Biopolus (2020). Electricity is based on design assumptions of Biopolus (2022). Data of chemical dosing for pH control was taken from the operator (De Dommel, 2022). Data of the coagulant and flocculant are supported by supplier data.
- NF: Operational data for recovery and water quality has been extrapolated from pilot trials. Electricity and chemical demand have been estimated by SEMiLLA (2022) based on literature of capillary NF designs.
- SBR: Operation and infrastructure data has been generated by DWA guidelines and literature for CAS and SBR systems and brewery wastewater in particular. The water quality and nutrient dosing were determined through mass balancing, while the electricity demand was estimated based on design specifications for individual process components and steps.
- EGSB: Data related to water quality, electricity consumption, chemical demand, biogas yield and infrastructure for the EGSB operation and the pre-treatment is based on supplier information and literature. Direct gaseous emissions are estimated related to literature.
- Sludge disposal: Data of polymer consumption and offsetting potentials due to biogas production and agricultural sludge use related to the external sludge digestion are estimated by KWB based on previous studies.
- Background information: Data for electricity, chemicals, transport, fertilizers, and construction materials are taken from the LCA database Ecoinvent (2021) v3.8. The sets are evaluated related to the spatial and temporal dimension. More details of the data can be found in Appendix A (p.124).

The evaluation criteria are good, medium, and bad. They relate to precision, completeness, and representativeness (technological, geographical, and time-related), and the methodological appropriateness and consistency. An evaluation of the data quality can be taken from Table 11 (p. 48).

Table 11: Evaluated data sets by data quality.

Data sets	Data quality	Explanation
General infrastructure (Koningshoeven)		
Infrastructure materials	Good	Design data, estimations by visiting
Electricity demand	Medium to good	Design data (Biopolus, 2022)
Chemicals demand	Medium to good	Supplier data (De Dommel, 2022)
Root IFAS		
WWT parameters	Medium to good	Simulated (Biopolus, 2022)
Electricity demand	Medium to good	Design data (Biopolus, 2022)
Chemicals demand	Medium to good	Supplier data (De Dommel, 2022)
Infrastructure materials	Good	Original design, estimations by visiting
NF		
WWT parameters	Good	Pilot trials (SEMiLLa, 2022)
Electricity/ chemical demand	Medium	Literature (STOWA, 2020)
Infrastructure materials	Medium	Literature, estimations (KWB, 2022)
SBR		
WWT parameters	Medium to good	DWA guidelines and Literature
Electricity/ chemical demand	Medium to good	Supplier, literature, estimates (KWB, 2022)
Infrastructure materials	Medium to good	Estimates based on modelled design
EGSB		
WWT parameters	Medium to good	Simulation based on Literature
Electricity/ chemical demand	Medium to good	Supplier data and Literature
Biogas yield and emissions	Medium	Literature
Infrastructure materials	Medium to good	Supplier, and estimates (KWB, 2022)
External Sludge disposal		
Electricity/ chemical demand	Medium to good	Literature, estimations (KWB, 2022)
Credits (electricity/ fertilizer)	Medium to good	Literature, estimations (KWB, 2022)
Background data		
Ecoinvent database (v3.8)	Medium to good	update 2021
Electricity	Good	NL power mix
Chemicals, materials	Medium to good	Europe or world market
Fertilizer production	Good	NL market mix
Transport	Good	EU Lorry (+distance by De Dommel)

Environmental impact indicators

The analysed midpoint environmental impact indicators include the CED, GWP, POFP, ODP, TAP, FEP and MEP. The environmental impact categories and underlying models in Table 12 have been used together with the software UMBERTO® LCA+ (IFU, 2018) and the ecoinvent (2021) databank (Version v3.8) for classification and characterisation. Ecotoxicity and Toxicity are not considered in this LCA, due to limitations of the data availability on micropollutants and metals in the influent, associated with toxic contaminants in the effluent and the leaching of heavy metals to soil.

Table 12: Environmental impact categoires

Impact category	Focus	Used LCAI model	Unit
Primary energy demand	Electricity, infrastructure, chemicals	Cumulative energy demand (CED) of fossil and nuclear resources, according to the VDI (2012) method.	MJ required
Climate change	Electricity, chemicals, infrastructure, direct emissions	Global warming potential (GWP) for a time horizon of 100 a (IPCC, 2007).	kg CO2-eq to air
Summer smog	Electricity, infrastructure	Photochemical oxidant formation potential (POFP) (Huijbregts, 2017).	kg NOx-eq to air
Ozone layer depletion	Chemicals	Ozone depletion potential (ODP) ReCiPe v1.13 based on WMO (2011) model (Huijbregts, 2017).	kg CFC-11-eq to air
Terrestrial Acidification	Nutrient dosing, sludge disposal	Terrestrial acidification potential (TAP) for a time horizon of 100 a. (Huijbregts, 2017).	kg SO2-eq to air
Eutrophication	Effluent quality (TN, TP)	Freshwater eutrophication potential (FEP) and Marine eutrophication potential (MEP). ReCiPe v1.13, hierarchist perspective, without long-term emissions (Huijbregts, 2017).	kg P-eq to freshwater; kg N-eq to marine water

3.3. Life Cycle Inventory (LCI)

In the part of the study, the relevant LCI data is generated. First, data related to the situation in Koningshoeven, including the raw wastewater characteristics and the current root IFAS installation is given. Second data for the NF is provided. Then key parameters of the reference scenarios are calculated, that can be used for LCI. This includes electricity demand via aeration and endogenic respiration, effluent quality parameters, reactor volumes, and the quantification of biogas and sludge production. Background data used from the Ecoinvent (2021) databank (Version v3.8) can found in the appendix A (p.124).

3.3.1. General Data for all Scenarios

All scenarios share the first step an equalisation by a buffer tank, afterwards the wastewater is treated by the scenario specific treatment stages, followed by the same micro sieve unit and a pond for additional polishing. The sludge is dewatered by a belt filter press to a Dry Matter (DM) of 22%. The sludge is treated by an external digester for wastes of the food and beverage industry. The treated water gets into a polishing pond and discharged directly into a local river.

3.1.3.1. pH Buffering

The effluent of the brewery arrives in two 200 m^3 balancing tanks in sequence. The first one of them belongs to the brewery, and is not part of this study. The latter belongs to the WWTP operator and is used is used for pH control. The pH control of incoming raw wastewater is achieved through the use of caustic and sulfuric acid (Table 13), as directed by the operator De Dommel (2022).

Table 13: Chemical dosing for pH control in the influent buffer tank.

chemical demand	NaOH (50%)	H_2SO_4 (96%)
kg · a^{-1}	4,590	3,350

The implementation of anaerobic technologies as in the EGSB-SBR scenario can reduce the chemical demand for neutralisation, especially if internal recirculation is provided (DWA, 2010). However, all scenarios are calculated by the same chemical dosing for pH control for the following reasons:

1) All scenarios share the same influent buffer tank (not aerated) and there is no recirculation to the influent buffer tank. Therefore, similar hydrolysation and pre-fermentation processes are expected in all scenarios.

2) The effect on the acid capacity of the wastewater is primarily influenced by the transformation of organic nitrogen (Bischhofsberger, 2005). However, the wastewater under consideration has a relatively low nitrogen content.
3) The influence on the pH level of 7 is recommended for both, aerobic and anaerobic treatment technologies (Bischhofsberger, 2005; Henze, 2000).

3.1.3.2. Sludge Treatment and Disposal

The sludge of the process gets treated with a belt filter press unit and is carried 10 kilometres to an external sludge fermentation for food industry wastes, where biogas and biosolids are produced. The final dry matter (DM) of 22% for the sludge at the WWTP in all scenarios relates to efficiency of the belt filter press, with the exception of the pre-treatment stage in the EGSB-SBR scenario, where the final DM is taken from the data sheet of a potential manufacturer (Huber SE, 2022). While the nitrogen content before dewatering is calculated by 8% of the DM in the SBR and EGSB-SBR scenario design (cf. Henze, 2000), the further treatment based on KWB (2022) estimations is recommended to calculate the TN by 5% of the DM.

The use of polymers for sludge dewatering in a belt filter press and the final dewatering at the external digestion stage is accounted with 4 kg (De Dommel, 2022) and 8 kg (KWB, 2022) of active matter per ton of DM in the sludge or solid waste being used, respectively. The belt filter press requires water for spoiling (about 1 $m^3 \cdot d^{-1}$), that was calculated by tap water. Previous studies conducted by KWB (Remy, 2021) have estimated that the electricity credits generated from sludge digestion amount to 400 kWh per ton DM in sludge. This figure reflects the net electricity output, taking into account the electricity produced from the CHP plant using biogas and the electricity requirements for operating the digestor and final dewatering. The phosphorus data is taken from a closed TP mass balance (TP input = TP output = TP effluent + TP sludge). Credits from sludge disposal are based on the mass of DM, the N, and P content and on estimations for the readily plan-available fraction. According to KWB (2022) efficiencies of 50% and 80% can be seen realistic in relation to substituted mineral fertilizers in the Netherlands for N, and P, respectively.

3.1.3.3. Influent Data of the Raw Wastewater

Table 14 shows raw wastewater data of the brewery measured by Watershap de Dommel (2022), including the modelled additional influent from sludge dewatering. The temperature of the brewery wastewater is usually 20–30°C (Biopolus, 2022).

Table 14: Wastewater characteristics for all scenarios

Parameter	Unit	Raw wastewater	Return flow from sludge dewatering and spoiling		
Scenario		All scenarios	Root IFAS & Root IFAS-NF	SBR	EGSB-SBR
$Q_{d,av}$	[m³ · d⁻¹]	150.0	7.5	18.1	6.5
$Q_{d,max}$	[m³ · d⁻¹]	200.0	10.0	24.1	8.7
$C_{COD,in}$	[mg · l⁻¹]	3,725	1,507	623	687
$\dot{m}_{COD,in}$	[kg · d⁻¹]	558.8	11.3	11.2	4.5
$C_{TSS,in}$	[mg · l⁻¹]	438	1107	494	386
$\dot{m}_{TSS,in}$	[kg · d⁻¹]	65.7	8.3	8.9	2.5
$C_{TN,in}$	[mg · l⁻¹]	46	84	46	44
$\dot{m}_{TN,in}$	[kg · d⁻¹]	6.84	0.63	0.84	0.29
$C_{TP,in}$	[mg · l⁻¹]	5.1	13.3	8.3	6.8
$\dot{m}_{TP,in}$	[kg · d⁻¹]	0.77	0.10	0.15	0.04
Data type		sampling (De Dommel, 2022)	calculated by Sumo (Biopolus, 2022)	assumptions with internal loop	assumptions with internal loop

No representative data for the maximum hourly volume flow is available. During a four weeks measurement campaign in April 2022, the maximum daily flow was 200 m³ · d⁻¹ and minimum flow rate of 20 m³ · d⁻¹ was seen on Sundays. Holidays are excluding a break of 15 days per year (De Dommel, 2022). Accordingly, a year is calculated for 350 working days of operation at the WWTP (De Dommel, 2022).

The wastewater data can be classified in typical ranges for brewery wastewater with a the relatively high TSS and COD load. The phosphorus (TP) and nitrogen values (TN) are rather low compared to the theory. Therefore, nitrification, denitrification, or advanced phosphorus removal processes are not required. Additional nutrient dosing has to be taken into account for some scenarios and no internal recirculation besides a return flow from the sludge dewatering have to be considered. The lower TP values can be explained by a shift in cleaning agents used by the brewery (EBC, 2003). The original root enhanced IFAS was designed for three times higher TP influents (Poór-Pócsi, 2021).

According to previous measurements of the industrial raw wastewater in Koningshoeven, 80% of the COD is dissolved (45µm filtration) (Dommel, 2022). Most of the nitrogen is organically bound, while NH_4^+, NO_2^- and NO_3^- make only 0.5, 0.2, and 4.2 % of the TN load, respectively. 50% of the nitrogen has been estimated to be particular (KWB, 2022). 90% of the TP is considered soluble and related cleaning agents. Specific information to micropollutants and metals have not been available.

3.3.2. Performance Parameters of the root IFAS

The WWTP in Koningshoeven includes a series of 16 cascaded tanks protruded by plants in a greenhouse and supported by inlay structures of propylene and two different located effluent inlet points. The purpose of this construction is to create several distinct dense ecosystems, including anoxic and aerobic zones as well as predators to reduce the amount of sludge volume (STOWA, 2017). The ecosystems create a resilient biodiversity of over 3,000 different species and provide improved process stability and adaptability to the fluctuation of the organic loading rate as well as the further ability to efficiently break down even hardly degradable organic micropollutants (STOWA, 2017). The DAF operation requires the dosing of coagulant ($FeCl_3$) and polymer, as indicated by the supplier, with additional safety factors included.

The effluent quality of the plant root enhanced IFAS and the daily amount of sludge, including assumptions of the dry matter and the nutrient content, are calculated and provided by Biopolus (2022) and the modelling software SUMO (2022). The results of the provided data for COD, TSS and TS, TN, and TP and additional chemical dosing requirements can be seen in the flow sheet of Figure 25 (p. 54). The dosing of nutrients (Urea as N = + 5.80 kg $\cdot$ d^{-1}, mineral P = + 4.20 kg $\cdot$ d^{-1}) has been modelled by Biopolus (2022) due to potential nutrient limitations for the microorganisms in the root IFAS biological treatment stage. The operator (De Dommel, 2022) provided additional information related to the estimated polymer dosing rate for the DAF unit (AM = 1.80 kg $\cdot$ d^{-1}) and the belt filter press (AM = 0.73 kg $\cdot$ d^{-1}) in the year 2022, as well as information on the used FeCl3 (40%) dosing (+ 14.45 kg $\cdot$ d^{-1}).

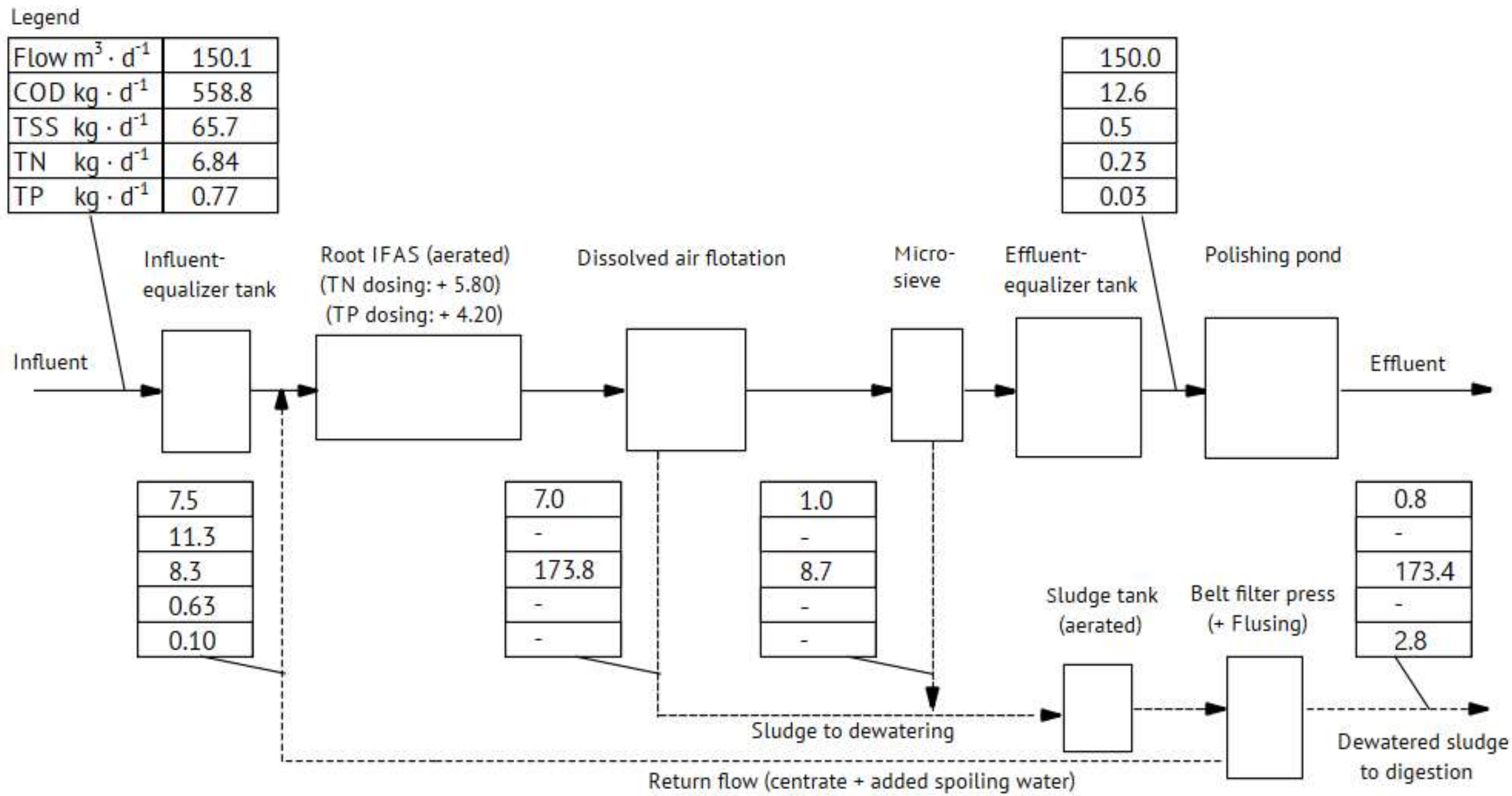


Figure 25: Mass balance of the root IFAS scenario (own illustration). Data are based on Biopolus (2022).

3.3.3. Nanofiltration Modules

The root IFAS-NF scenario is based on the root IFAS scenario, with the difference that an additional Nanofiltration (NF) treatment stage, based on pilot trials of SEMiLLA (2022), and further infrastructure for water reuse has to be installed.

The effluent of the capillary NF is calculated based on a 25% recovery rate into permeate. The permeate (13,125 $m^3 \cdot a^{-1}$) can be reused as process water credited for the brewery. The avoided burden of water production is accounted using an Ecoinvent (2021) dataset (Appendix A, p.129) as "groundwater without further treatment" and is compared to tap water from the general water supply in the analysis. The remaining retentate of the capillary nanofiltration system is discharged with lower flow rates than in the root IFAS scenario (Table 15).

Table 15: Comparison of the effluent between root IFAS and root IFAS-NF.

Parameter	Unit	Root IFAS effluent	Root IFAS-NF effluent
Flow rate	$m^3 \cdot a^{-1}$	52,500	39,375
COD	$mg \cdot l^{-1}$	84	111
TSS	$mg \cdot l^{-1}$	3.1	4.1
Total N	$mg \cdot l^{-1}$	1.5	1.7
Total P	$mg \cdot l^{-1}$	0.2	0.3

The NF consists of 6 modules, each with 50m^2 surface area. The electricity demand and required chemicals for the upkeep, have been taken from Table 6 (p. 21) and dived by 166, as estimated by SEMiLLA (2022). The additional electricity demand of the root IFAS-NF scenario is presented in Table 16, including an assumption of 15 kWh per day for water reuse (KWB, 2022). **Fehler! Verweisquelle konnte nicht gefunden werden.** 17 shows the chemical demand of the membrane, diluted to concentration of market products.

Table 16: Electricity demand for the NF and the water reuse concept.

Electricity demand	NF	Pumping (+mixing) to brewery
$kWh \cdot a^{-1}$	10,063	5,250

Table 17 Chemical demand for membrane cleaning:

Chemical demand	NaOH (50%)	NaOCl (10%)	Citric acid (60%)
$kg \cdot a^{-1}$	48	150	77

3.3.4. Design of the anaerobic Treatment Stage

Since both alternative scenarios contain an aerobic treatment stage, the additional upstream features of the EGSB-SBR-Scenario are designed first. Goal of this chapter is to calculate key data for treatment efficiency, construction, sludge and biogas production, CH_4 losses. While the TSS concentration is recommended to be lower than 500 mg · l^{-1} (Stronach, 1986), an average value of 448 mg · l^{-1} can be considered critical and an additional pre-treatment stage for TSS reduction has to be implemented.

3.4.3.1. Pre-Treatment

Five different companies have been analysed in terms of information towards their products (drum sieves, micro strainers, lamella separators). As a result, a micro strainer (0.5 and 1.0 mm) of stainless steel was chosen, because of data completeness on performance parameters, electricity consumption, weight, type of construction material and previous experiences for industrial WW. No data of the TSS particular size distribution in the WW was available. The company estimated a TSS removal efficiency of 60% (Huber SE, personal communication, 4th august 2022). The operation includes dewatering and a DM of 15% can be achieved. The removed TSS can be calculated according to (2):

$$\dot{m}_{TSS,ele} = \dot{m}_{TSS,in} \cdot 0.6 \tag{2}$$

$$= 68.2 \cdot 0.6 = 40.9 \text{ kg} \cdot \text{d}^{-1}$$

The wastewater parameter related to the pre-treatment are summarized in Table 18. The values in Table 18 are calculated by the raw wastewater influent, the return from sludge dewatering, and the losses related to the pre-treatment.

Table 18: Wastewater characteristics of the EGSB-SBR Scenario related to the mechanical pre-treatment

EGSB-SBR Influent	$Q_{d,av}$	COD	TSS	TN	TP
unit	[m³ · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]
before Pre-treatment	156.5	563.3	68.2	7.13	0.81
after Pre-treatment	156.2	495.7	28.1	4.99	0.76

In result the micro strainer lowers the TSS approximately to 178 mg · l^{-1}. With the loss of TSS the particulate COD (X_{COD}), TN (X_{TN}), and TP (X_{TP}) is reduced. 80% of the COD is soluble, while 50% of the TN, and 10% of the TP was estimated to be particular (chapter 3.1, p. X). The removed loading rates for X_{COD} (3), X_{TN} (4), and X_{TP} (5) can be calculated:

$$\dot{m}_{COD_{ele}} = \dot{m}_{X_{COD,in}} \cdot 0.6 = (\dot{m}_{COD_{in}} - \dot{m}_{S_{COD,in}}) \cdot 0.6 \quad (3)$$

$$= (563.3 - (0.8 \cdot 563.3)) \cdot 0.6 = 67.6 \text{ kg COD} \cdot d^{-1}$$

$$\dot{m}_{TN_{ele}} = \dot{m}_{X_{TN,in}} \cdot 0.6 = \dot{m}_{TN_{in}} \cdot 0.1 \cdot 0.6 \quad (4)$$

$$= 7.13 \cdot 0.5 \cdot 0.6 = 2.14 \text{ kg TN} \cdot d^{-1}$$

$$\dot{m}_{TP_{ele}} = \dot{m}_{P,in} \cdot 0.6 = \dot{m}_{TP_{in}} \cdot 0.1 \cdot 0.6 \quad (5)$$

$$= 0.81 \cdot 0.1 \cdot 0.6 = 0.05 \text{ kg TP} \cdot d^{-1}$$

The screening material gets dewatered to a dry matter content x_{DM} of 15% and its mass ($m_{loss,SP}$) can be calculated by equation (6). In the approach of this study the flow rate is considered and gets lowered according to equation (7) because of the losses due to the pre-treatment. The substitution of (6) in (7) creates equation (8). A density of 1 kg · m^{-3}, similar to sludge (AVT, 1996), is used.

$$m_{loss,SP} = \frac{\dot{m}_{TSS,ele}}{x_{DM}} \quad (6)$$

$$V_{loss} = \frac{m_{loss,SP}}{\rho_{H_2O}} \quad (7)$$

$$V_{loss} = \frac{\dot{m}_{TSS,ele}}{x_{DM} \cdot \rho_{H_2O}} \quad (8)$$

$$= 40.9 \cdot (0.15)^{-1} \cdot 10^{-3} = 0.3 \text{ m}^3 \cdot d^{-1}$$

3.4.3.2. EGSB Reactor

In the EGSB-SBR-scenario the pre-treatment is followed by the EGSB reactor.

Design

According to the good practice guideline of the European Brewery Convention (EBC, 2003) the volume of the anaerobic reactor (V_R) for industrial wastewater is designed by the possible organic loading rate (B_V) if COD levels are above 2,500 mg · l^{-1}. In order to be within the range of two recommended standards of the European brewery convention and the German DWA (2009), an average sludge loading rate of 13 kg COD per m^3 has been taken (compare theory part; Table 5, p. 19). Due to data gaps, no hourly maximum COD load or flow rate was given. The reactor volume is defined by (9a) and solved by (9b):

$$V_R = \frac{Q_{d,max} \cdot C_{COD,in}}{B_V} \quad (9a)$$

$$V_R = \dot{m}_{COD,in} \cdot \frac{Q_{d,max}}{Q_{d,av}} \cdot (B_V)^{-1} \quad (9b)$$

$$= 495.7 \cdot 1.33 \cdot (13)^{-1} = 47 \text{ m}^3 \approx 50 \text{ m}^3$$

A reactor of 50 m^3 is required. Construction material for the EGSB-SBR-Scenario is listed in appendix B4, p. X.

Water parameters

In this study an effective COD elimination of 80% is assumed by the EGSB reactor. An anaerobic treatment stage in the Netherlands, associated with the same brewery company than in this study, achieved an 80% COD removal rate, independent of the temperature (EBC, 2003). The correlating sludge production (11) of the COD elimination (10) can be estimated with a yield (y_{real}) of 0.06 kg TS · (kg COD_{ele})$^{-1}$ (Henze, 2000):

$$\dot{m}_{COD_{ele}} = \dot{m}_{COD_{in}} \cdot 0.8 \quad (10)$$

$= 495.7 \cdot 0.8 = 396.6$ kg COD · d^{-1}

$$F_{SP,EGSB} = y_{real} \cdot \dot{m}_{COD_{ele}} \quad (11)$$

$= 396.6 \cdot 0.06 = 23.8$ kg TS · d^{-1}

With an estimated DM content (x_{DM}) of 1% (U. Austermann-Haun, personal communication, 24th august 2022), the excess sludge volume can be evaluated (12) analogue to equation (8, p. 54):

$$V_{SP,EGSB} = \frac{F_{SP,EGSB}}{x_{DM} \cdot \rho_{H_2O}} \quad (12)$$

$= 23.8 \cdot (0.01)^{-1} \cdot 10^{-3} = 2.4$ m^3 · d^{-1}

The loss of mass on TN and TP is calculated in (13) and (14) via mass balance. According to equation (6) and (7) (p. 54) the sludge contains a mass of 2379 kg and a volume of 2.4 m^3 per day. The content of TN (molar mass of N: 14u) and TP in the sludge can be estimated around 8.0% and 1.4%, respectively (Henze 2000; Stronach, 1986). Nitrate (molar mass: 64u) in the influent amounts 0.29 kg · d^{-1} (De Dommel, 2022) and is converted by anaerobic conditions (denitrification).

$$\dot{m}_{TN_{ele}} = \dot{m}_{TN_{ele,SP}} + \dot{m}_{TN_{ele,d}} = F_{SP,EGSB} \cdot 0.08 + \dot{m}_{NO_{3,in}^-} \cdot \frac{M_N}{M_{NO_3^-}} \quad (13)$$

$= 23.8 \cdot 0.08 + 0.29 \cdot 14 \cdot 64^{-1} = 1.97$ kg TN · d^{-1}

$$\dot{m}_{TP_{ele}} = F_{SP,EGSB} \cdot 0.014 \quad (14)$$

$= 23.8 \cdot 0.014 = 0.33$ kg TP · d^{-1}

Due to data limitations for the TSS effluent an analogy from literature is taken by UASB reactors for brewery wastewater with TSS removal efficiencies about 80%. The effluent concentrations of TSS have been measured between 34 and 41 mg · l^{-1} for TSS influent concentrations below 250 mg · l^{-1} (Sharda, 2013). In this study the average TSS of 37.5 mg · l^{-1} is taken, related to a TSS loading rate of 3.02.

Table 15 summarizes the parameters of the EGSB influent and the effluent quality after subtraction of the eliminated COD and the TSS, TN, and TP gone by the excess sludge of the EGSB.

Table 19: Wastewater characteristics of the EGSB-SBR Scenario related to the EGSB-Reactor.

EGSB-SBR Influent	$Q_{d,av}$	COD	TSS	TN	TP
unit	[m^3 · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]
before EGSB	156.2	495.7	28.1	4.99	0.76
after EGSB	153.8	99.1	5.8	3.02	0.42

Total methane production

The specific CH_4 production of the anaerobic digestion of brewery wastewater is estimated to be y_{CH_4} = 0.285 Nm^3 · kg^{-1} COD in the influent (ATV/VKS Technical Committee 3.8, 1994) by equation for the COD load (15):

$$V_{total,CH_4} = \dot{m}_{COD_{in}} \cdot y_{CH_4} \qquad (15)$$

$$= 495.7 \cdot 0.285 = 141 \text{ m}^3 \cdot \text{d}^{-1}$$

Dissolved methane fraction

Since no data was found about EGSB reactors in the brewery sector related to the dissolved methane fraction, estimations have to be made. Assumptions based on the thermodynamic equilibrium of the interfacial mass transport or comparisons and analogies with literature from different reactor type can be made. The first option does not consider that most studies report an oversaturation of the water that can be explained by the limited surface area and time span related to the operation where the equilibrium cannot be reached (Smith, 2012). Furthermore, the solubility can be assumed increased by the salinity (Liu, 2014) and further wastewater characteristics. The second option struggles with contradicting findings in the literature and the analysis of a different reaction tank format.

Research literature was found related to similar temperature (25°C) and CH_4 content in the biogas (80%) for municipal and standard UASB reactors (Crone 2016; Urban, 2009). The CH_4 losses of low strength wastewater in municipal UASB reactors can be characterised by a relatively high percentage (11–88%) (Stazi, 2021) due to the low COD loading rates of domestic wastewater compared to brewery wastewater. As a solution to consider the analysed system as well as the oversaturation, the saturation is calculated by the theoretical mass transfer and scaled by literature values on the potential oversaturation. However, this approach may not consider specifics of brewery wastewater appropriately.

Next to the dimensionless Henry's law constant, the Henry's law solubility (H_S^{cp}) is the most convenient way to describe the interfacial mass transport of a chemical species (Sander, 2022). According to the IUPAC recommendation (Sander, 2022) it can be defined as ratio between the saturated aqueous-phase concentration (C_S) and its partial pressure (P_g) in the gas phase at the equilibrium (16).

$$H_S^{cp} = \frac{C_S}{P_g} \tag{16}$$

The P_g of a species can be described as product of the gas pressure (P_T) and the volumetric fraction content of the species in the gas phase (p_g) (17).

$$P_g = P_T \cdot p_g \tag{17}$$

Equation (16) reconfigured according to the concentration parameter in the water phase the equation can be described by implementing (17) as (18).

$$C_S = H_S^{cp} \cdot P_g = H_S^{cp} \cdot P_T \cdot p_g \tag{18}$$

The relation of C_S to the assumed dissolved concentration (C) (19) is the factor of oversaturation (f_{OS}). Since the H_S^{cp} is defined in molar quantity, a molar mass term (M) is added in the equation to generate a mass-based concentration.

$$C = C_S \cdot M \cdot f_{OS} = H_S^{cp} \cdot P_T \cdot p_g \cdot M \cdot f_{OS} \tag{19}$$

According to Kosse (2018) different values can be found for H_S^{cp} at 298.15 K for CH_4, resulting in different values for potential CH_4 losses (Table 15). The molar mass (M) of CH_4 equals 16.04 u. The assumption of an ideal P_T is usually 101,325 Pa (Metcalf & Eddy, 2013). 80% CH_4 in the biogas (p_g) is taken for brewery wastewater (ATV/VKS Technical Committee 3.8, 1994). In a study review by Crone et al. (2016) the CH_4 loss of UASB reactors at 298.15 K has been reported between 1.3 and 1.64 times the equilibrium and up to 6.9 times for psychrophilic temperatures (10–30°C).

Table 20: Calcination of the dissolved methane concentration in reference to the solubility.

parameter	H_S^{cp}	c			
f_{OS}		1.0	1.3	1.5	1.7
unit	[mol · m^{-3} · Pa^{-1}]	[mg · l^{-1}]	[mg · l^{-1}]	[mg · l^{-1}]	[mg · l^{-1}]
	$1.2 \cdot 10^{-5}$	15.60	20.28	23.40	26.52
	$1.3 \cdot 10^{-5}$	16.90	21.97	25.35	28.73
	$1.4 \cdot 10^{-5}$	18.20	23.66	27.30	30.94

Besides UASB reactors, one EGSB reactor has been evaluated at 283.15 K with an f_{OS} of 1.57 (Crone, 2016). However, the proportion of the dissolved fraction is higher at colder temperatures (Urban, 2009).

The median of 25.35 mg · l^{-1} with oversaturation (f_{OS}) of 1.5 is taken. The volumetric CH_4 losses are calculated (20, p. 59) with $Q_{d,av}$ = 153.8 $m^3 \cdot d^{-1}$, the density (ρ) of 0.657 kg · m^{-3} for methane and a unit correction factor.

$$V_{loss,CH_4} = \frac{Q_{d,av} \cdot c_{CH_4}}{\rho_{CH_4}} \cdot 10^{-3} \frac{kg \cdot l}{mg \cdot m^3} \tag{20}$$

$$= 153.8 \cdot 25.35 \cdot (0.657)^{-1} \cdot 10^{-3} = 6 \; Nm^3 \cdot d^{-1}$$

In result, the dissolved CH_4 fraction amounts to 4.3% of 141 $Nm^3 \cdot d^{-1}$ produced CH_4. In contrast, equations by Urban (2009) (21) result in only half of the CH_4 losses at 25°C with a CH_4 content between 75% and 85% in the biogas.

$$V_{loss,CH_4} = 0.019 \frac{Nm^3}{m^3} \cdot Q_{d,av} \tag{21}$$

$$= 0{,}019 \cdot 153.8 = 3 \; Nm^3 \cdot d^{-1}$$

An explanation for divagating results can be related to data deviations, measurement techniques, environmental conditions, reactor design, or wastewater characteristics. However, a removal rate of 50% dissolved CH_4 by the biomass is assumed in this LCA study.

Biogas volume

The net usable CH_4 volume (V_{gas,CH_4}) is the difference of production and loss (22).

$$V_{gas,CH_4} = V_{total,CH_4} - V_{loss,CH_4} \tag{22}$$

= 141 – 6 = 135 $Nm^3 \cdot d^{-1}$

This equates the following biogas volume (23), according to an average 80% methane ratio in biogas of anaerobic technology in breweries (ATV/VKS Technical Committee 3.8, 1994).

$$V_{biogas,av} = \frac{V_{gas,CH_4}}{0.8} \tag{23}$$

= 135 · $(0.8)^{-1}$ = 169 $Nm^3 \cdot d^{-1}$

With (22) in (23) with a 1.33 times higher COD load at maximum capacity the daily maximum produced biogas can be considered for the infrastructure (24):

$$V_{biogas,max} = \frac{(1.33 \cdot V_{total,CH_4}) - V_{loss,CH_4}}{0.8} \tag{24}$$

= ((141 · 1.33) – 6) · $(0.8)^{-1}$ = 227 $Nm^3 \cdot d^{-1}$

Iron dosing

$FeCl_2$ (30%) upstream of the EGSB is used to minimize the H_2S content. The dosing rate is calculated by Urban (2009) with 2.71 g Fe per Nm^3 biogas to reduce the H_2S from estimated 1000ppm to 100ppm. With 135 $Nm^3 \cdot d^{-1}$, the dosing of $FeCl_2$ with an Fe content 30% is 1.22 $kg \cdot d^{-1}$.

Heat credit

The CH_4 can be translated according to equation (25) into a heating credit (E_{heat}). CH_4 has a Net Calorific Value (NCV) of 10 $kWh \cdot Nm^{-3}$ (Kiattisak, 2014), while the boilers efficiency (n_{boiler}) can was estimated about 90% by Nextgen (Bischhofsberger, 2005). 0.5% of the biogas are estimated to get burned by the gas flare and therefore don't provide a heating credit (KWB, 2022).

$$E_{H,Credit} = (-1) \cdot V_{gas,CH_4} \cdot NCV_{CH_4} \cdot n_{boiler} \cdot (1 - x_{GF}) \qquad (25)$$

$$= (-1) \cdot 135 \cdot 10 \cdot 0.9 \cdot (1 - 0.5) = -1209 \text{ kWh} \cdot d^{-1}$$

Direct gaseous Emissions

Gaseous emissions occur in the EGSB-SBR-scenario related to biogas use for heating and to the CH_4 losses of the dissolved fraction (Table 21, p.65). Emissions for incineration at the boiler and at the gas flare are calculated in relation to the provided heating energy, based on a literature comparison (Edelmann 2001; Ronchetti, 2002), while the additional assumption of an estimated methane slip of 0.5% is taken (KWB, 2022). Based on estimations of KWB (2022), 50% of the methane of the dissolved fractions gets released into the atmosphere, while the remaining part is consumed by methanotrophic microorganisms.

Table 21: Direct gas emissions

Emission to air	Unit	EGSB-SBR-scenario	Source
Methane loss			
CH_4 stripped	$Nm \cdot d^{-1}$	$V_{loss,CH_4} \cdot 0.50$	Estimation (KWB, 2022)
Biogas incineration			
CH_4 (slide)	$Nm \cdot d^{-1}$	$V_{biogas,av} \cdot 0.005$	Estimation (KWB, 2022)
CH_4	$kg \cdot MJ^{-1}$	$2.5 \cdot 10^{-6}$	Ronchetti (2002)
CO	$kg \cdot MJ^{-1}$	$5.1 \cdot 10^{-5}$	Ronchetti (2002)
NO_X	$kg \cdot MJ^{-1}$	$3.8 \cdot 10^{-5}$	Ronchetti (2002)
N_2O	$kg \cdot MJ^{-1}$	$1.6 \cdot 10^{-6}$	Ronchetti (2002)
SO_2	$kg \cdot MJ^{-1}$	$5.1 \cdot 10^{-5}$	estimation (KWB, 2022); assumption of 1000 ppmV H_2S in biogas

3.3.5. Design of the aerobic SBR Treatment Stage

In the SBR-Scenario, as well as in the EGSB-SBR scenario, two (n =2) phase shifted SBRs replace the root IFAS scenario's aerobic treatment stage and subsequential phase separation. In order improve LCA comparability, similar design assumptions have been made for both scenarios. The sludge wasting during the reaction time and the related Solids Retention Time (SRT) of a SBR is not comparable to a continuous-flow CAS (Metcalf & Eddy, 2013). However, the German DWA (2009) guiding reference M210, for SBR systems, is based on assumptions related to CAS. Many examples of the guide imply nitrification and dentification processes and require a SRT about 25 days for sludge stabilisation (Bever, 2002; DWA, 2009). A SRT equivalent to CAS with 5 days was chosen sufficient, recommended for biological treatment plants without the need of nitrification (DWA, 2016). The average cycles (Table 22, p.66) are similar to the example of DWA-M210 (DWA, 2009).

Table 22: SBR cycles

parameter	time in [h] per cycle	meaning
m_C = 3		cycles per day
t_c	8	hole period
t_{sed}	1	sedimentation
t_{ex}	1	decant and exchange time
t_r	6	reaction time

The SBR design is customized according to the incoming flow rate and WW quality parameters illustrated in Table 23.

Table 23: Influent wastewater parameters of the SBR treatment stage

Parameter	$Q_{d,av}$	COD	TSS	TN	TP
unit	$[m^3 \cdot d^{-1}]$	$[kg \cdot d^{-1}]$	$[kg \cdot d^{-1}]$	$[kg \cdot d^{-1}]$	$[kg \cdot d^{-1}]$
SBR scenario	168.1	570.0	74.6	7.68	0.92
EGSB scenario	153.8	99.1	5.8	3.02	0.42

3.5.3.1. COD Fractions

The COD is the key parameter for the design of aerobic biological WWTP and can be divided into dissolved and particulate fractions, as well as inert and biodegradable components (DWA, 2016).

For the EGSB-SBR scenario the particulate COD is unknown because it had been reduced by the pre-treatment and partially converted by the EGSB operation. It can be approximated by equation (26) with an ignition residue of 20% (f_{IR}) and 1.6 g COD per g oTS if the parameter is unknown and the operation is not the first treatment step (DWA, 2016).

$$\dot{m}_{X_{COD,in}} = \dot{m}_{TSS,in} \cdot 1.6 \cdot (1 - f_{IR}) \qquad (26)$$

$$= 5.8 \cdot 1.6 \cdot (1 - 0.2) = 2.8 \text{ kg COD} \cdot d^{-1} \qquad \text{(EGSB-SBR-Scenario)}$$

The inert part of the dissolved COD in the WWTP influent can be seen close to its effluent concentration (DWA, 2016). The average effluent COD and TSS loading rates of the plant root enhanced IFAS were 12.6 and 0.5 kg · d^{-1}, respectively (Biopolus, 2022). The dissolved COD fraction is calculated by

(27) as difference between the total effluent COD and its approximated particular fraction.

$$\dot{m}_{S_{COD,I,in}} = \dot{m}_{COD_{out}} - \dot{m}_{TSS,in} \cdot 1.6 \cdot (1 - f_{IR}) \qquad (27)$$

$$= 12.6 - 0.5 \cdot 1.6 \cdot (1 - 0.2) = 12.0 \text{ kg COD} \cdot \text{d}^{-1}$$

The inert part of the particulate COD is dependent on the type of wastewater and its pre-treatment in ranges of 20–35% of the particulate COD (DWA, 2016). Brewery wastewater is known to be readily biodegradable with many organic solids related to yeast, turb, or grain. The lower value of f_a = 20% points is taken. In the raw wastewater 80% of the COD is soluble (Biopolus, 2022). The remaining part (20%) can be considered particulate for the SBR-Scenario. The inert particulate COD is calculated by equation (28):

$$\dot{m}_{X_{COD,I,in}} = \dot{m}_{X_{COD,in}} \cdot f_a \qquad (28)$$

$$\dot{m}_{X_{COD,I,in}} = (570.0 \cdot 0.2) \cdot 0.2 = 22.8 \text{ kg COD} \cdot \text{d}^{-1} \qquad \text{(SBR-Scenario)}$$

$$\dot{m}_{X_{COD,I,in}} = 2.8 \cdot 0.2 = 0.6 \text{ kg COD} \cdot \text{d}^{-1} \qquad \text{(EGSB-SBR-Scenario)}$$

The readily biodegradable COD in the WWTP influent is calculated (29) as difference between the total COD and the inert fractions (DWA, 2016).

$$\dot{m}_{S_{COD,in}} = \dot{m}_{COD_{in}} - \dot{m}_{S_{COD,I,in}} - \dot{m}_{X_{COD,I,in}} \qquad (29)$$

$$\dot{m}_{S_{COD,in}} = 570.0 - 12.0 - 22.8 = 535.2 \text{ kg COD} \cdot \text{d}^{-1} \qquad \text{(SBR-Scenario)}$$

$$\dot{m}_{S_{COD,in}} = 99.1 - 12.0 - 0.6 = 86.5 \text{ kg COD} \cdot \text{d}^{-1} \qquad \text{(EGSB-SBR-Scenario)}$$

3.5.3.2. Biomass and Sludge Production

In order to adjust equations from domestic wastewater to the higher temperatures of brewery wastewater, a factor (F_T) is considered and calculated according to Henze (2000) (30):

$$F_T = 1.072^{(T - 288.15K)} \tag{30}$$

In the German DWA-A 131 standard (DWA, 2016) the biomass of a CAS for domestic wastewater can be calculated as follows (31):

$$X_{COD,BM} = \left(C_{COD_S} \cdot Y_{hetero} + C_{COD,dos} \cdot Y_{COD,dos}\right) \cdot \frac{1}{1 + b \cdot SRT \cdot F_T} \tag{31}$$

The biomass yield of the heterotrophic bacteria (Y_{hetero}) at 15°C (T = 288.15 K) in the aerobic process can be estimated by 0.67 kg COD_{BM} per kg COD_{ele} (Henze, 2000) and the decay rate about 0.17 d^{-1} (DWA, 2016). The equation (28) can be simplified since no external C dosing ($C_{COD,dos}$) and no denitrification is required. Multiplied by the average daily flow rate, the following mass balance (32) results:

$$\dot{m}_{X_{COD,BM}} = \dot{m}_{COD_{S,in}} \cdot Y_{heter} \cdot \frac{1}{1 + b \cdot SRT \cdot F_T} \tag{32}$$

The results for (32) in both scenarios and the average temperature and minimum temperature for design, can be seen in Table 24.

Table 24: Calculation of the net biomass production.

parameter	unit	Scenario, T [K]	input for equation (32)	Result
$\dot{m}_{X_{COD,BM}}$	kg COD · d^{-1}	SBR, 298.15	$535.2 \cdot 0.67 \cdot (1 + 0.17 \cdot 5 \cdot 1.072^{(298.15 - 288.15)})^{-1}$	= 132.6
		SBR, 293.15	$535.2 \cdot 0.67 \cdot (1 + 0.17 \cdot 5 \cdot 1.072^{(293.15 - 288.15)})^{-1}$	= 162.7
		EGSB-SBR, 298.15	$86.5 \cdot 0.67 \cdot (1 + 0.17 \cdot 5 \cdot 1.072^{(298.15 - 288.15)})^{-1}$	= 21.4
		EGSB-SBR, 293.15	$82.9 \cdot 0.67 \cdot (1 + 0.17 \cdot 5 \cdot 1.072^{(293.15 - 288.15)})^{-1}$	= 26.3

Remaining inert particulates of the endogenic decay can be estimated as 20% of the biomass decay (33) (DWA, 2016). Results can be seen in Table 25.

$$\dot{m}_{X_{COD,I,BM}} = 0.2 \cdot \dot{m}_{X_{COD,BM}} \cdot SRT \cdot b \cdot F_T \tag{33}$$

Table 25: Calculation of the inert particulates from biomass decay.

parameter	unit	Scenario, T [K]	input for equation (33)	Result
$\dot{m}_{X_{COD,I,BM}}$	kg COD · d^{-1}	SBR, 298.15	$0.2 \cdot 132.6 \cdot 5 \cdot 0.17 \cdot 1.072^{(298.15 - 288.15)}$	= 45.1
		SBR, 293.15	$0.2 \cdot 162.7 \cdot 5 \cdot 0.17 \cdot 1.072^{(293.15 - 288.15)}$	= 39.2
		EGSB-SBR, 298.15	$0.2 \cdot 21.4 \cdot 5 \cdot 0.17 \cdot 1.072^{(298.15 - 288.15)}$	= 7.3
		EGSB-SBR, 293.15	$0.2 \cdot 26.3 \cdot 5 \cdot 0.17 \cdot 1.072^{(293.15 - 288.15)}$	= 6.3

The organic part of produced excess sludge relates (34) to inert particulates, generated biomass, and their inert decay residues (DWA, 2016).

$$\dot{m}_{COD,SP} = \dot{m}_{X_{COD,I,in}} + \dot{m}_{X_{COD,BM}} + \dot{m}_{X_{COD,I,BM}} \tag{34}$$

Based on the assumption that the relevance of the particulate loading rate in the effluent is negligible, the excess sludge equates production equals the overall sludge production (DWA, 2016). The value of 1.33 kg COD per kg organic DM for the particulate inert COD, a ratio of 92% organic parts in the biomass, a biomass conversation of 1.42 kg COD_{BM} per kg organic DM, and an additional inorganic content (f_b) related to 20% of the influent TSS, can be used to calculated the following equation (35) for the excess sludge production (DWA, 2016):

$$F_{SP,M} = \frac{\dot{m}_{X_{COD,I,in}}}{1.33} + \frac{\dot{m}_{X_{COD,BM}} + \dot{m}_{X_{COD,I,BM}}}{0.92 \cdot 1.42} + f_b \cdot \dot{m}_{TSS,in} \tag{35}$$

Implementing the biomass terms of Table 25 and Table 24 (p.68), the inert particulate COD by (28, p.67), and the TSS influent mass loading rate (Table 23, p.66) the following results can be generated by equation (35) for the sludge production (Table 26, p.70):

Table 26: Sludge production of the SBR.

parameter	unit	Scenario, T [K]	input for equation (35)	Result
$F_{SP,M}$	kg MLSS · d^{-1}	SBR, 298.15	$\frac{22.8}{1.33} + \frac{132.6 + 45.1}{0.92 \cdot 1.42} + 0.2 \cdot 74.6$	= 168.1
		SBR, 293.15	$\frac{22.8}{1.33} + \frac{162.7 + 39.2}{0.92 \cdot 1.42} + 0.2 \cdot 74.6$	= 186.6
		EGSB-SBR, 298.15	$\frac{0.6}{1.33} + \frac{21.4 + 7.3}{0.92 \cdot 1.42} + 0.2 \cdot 5.8$	= 23.6
		EGSB-SBR, 293.15	$\frac{0.6}{1.33} + \frac{26.3 + 6.3}{0.92 \cdot 1.42} + 0.2 \cdot 5.8$	= 26.6

A sludge volume index (SVI) of 100 l · kg^{-1} and a related dry matter of DM of 1% was assumed for the sludge, generated by the SBR (U. Austermann-Haun, personal communication, 24th august 2022). The volume of the SBR excess sludge at 298.15 K is calculated analogue to in (36):

$$V_{SP,M} = \frac{F_{SP,M}}{x_{DM} \cdot \rho_{H_2O}} \tag{36}$$

$$= 168.1 \cdot (0.01)^{-1} \cdot 10^{-3} = 16.8 \text{ m}^3 \cdot \text{d}^{-1} \qquad \text{(SBR-Scenario)}$$

$$= 23.6 \cdot (0.01)^{-1} \cdot 10^{-3} = 2.4 \text{ m}^3 \cdot \text{d}^{-1} \qquad \text{(EGSB-SBR-Scenario)}$$

The required amount of sludge for biomass conversion inside an aerobic tank is calculated by the product of sludge production ($F_{SP,CAS}$) and SRT (DWA, 2016) (37).

$$m_{TS,CAS} = F_{SP,M} \cdot SRT \tag{37}$$

For discontinuous reactor types, additional sludge is required to cope with the limitations of the reaction time. According to the DWA (2009) information M 210 for fill and draw systems, the calculated sludge has to be scaled up to be consistent with the limited reaction time (t_r) of the overall time per cycle (t_r) (38):

$$m_{TS,SBR} = m_{TS,CAS} \cdot \frac{t_c}{t_r} \tag{38}$$

Substituting (37) in (38), we get (39, p.71) for the TS of sludge in the SBR. The scenario specific results can be taken from Table 27 (p.71).

$$m_{TS,SBR} = F_{SP,M} \cdot SRT \cdot \frac{t_c}{t_r} \tag{39}$$

Table 27: Amount of sludge required in the SBR for the treatment

parameter	unit	Scenario, T [K]	input for equation (38)	Result
$m_{TS,SBR}$	kg MLSS	SBR, 298.15	$168.1 \cdot 5 \cdot 8 \cdot 6^{-1}$	= 1121
		SBR, 293.15	$186.6 \cdot 5 \cdot 8 \cdot 6^{-1}$	= 1244
		EGSB-SBR, 298.15	$23.6 \cdot 5 \cdot 8 \cdot 6^{-1}$	= 157
		EGSB-SBR, 293.15	$26.6 \cdot 5 \cdot 8 \cdot 6^{-1}$	= 177

The generation of biomass influences the composition of the TSS effluent in the reactor (Metcalf & Eddy, 2013). In this study, the values are taken from TSS effluent concentrations on literature. The value for the SBR is taken from a study review from Mace & Mata-Alvarez (2002), that illustrates that TSS effluent concentration with an average value of 70.5 mg · l^{-1} were found by laboratory scale SBR studies with brewery wastewater for up to 3 SBR operation cycles per day for COD effluent concentration between 50 and 184 mg · l^{-1}. For the EGSB-SBR scenario an analogy to the effluent of a CAS after an anaerobic treatment stage for industrial brewery wastewater was taken. TSS concentrations of 21.0 mg · l^{-1} have been measured (Austermann-Haun, 1998).

3.5.3.3. Construction Design Data

The reactor volume of the SBR has to fulfil two conditions, to be sufficient for the required biomass and the hydraulics conditions (Bever, 2000).

The first requires a minimum volume (V_{min}) to keep the required sludge and an exchange volume (ΔV_{max}) for the fill and draw process (41) (DWA, 2009). The minimum volume is defined (42, p.72) by the biomass in the reactor, divided by its recommended concentration (TS_R) (DWA, 2009).

$$V_R = V_{min} + \Delta V_{max} \tag{41}$$

$$V_{min} = \frac{m_{TS,SBR}}{n \cdot TS_R} \quad (42)$$

The exchange volume relates to the average flow rate per cycle, scaled by the SBR time limitations (43) (Bever, 2000). For several phase shifted tanks, the volume of each is generated by the division by the number of reactors (n).

$$\Delta V_{max} = Q_{d,max} \cdot \frac{1\,d \cdot t_C}{24\,h \cdot n} \quad (43).$$

Replacing equations (42) and (43) in (41) results in (44). The lower temperature is used for construction design (DWA, 2016). A MLSS concentration (TS_R) between 3 and 5 kg TS · m^{-3} can be found in CAS for brewery wastewater (DWA, 2010), while for SBR systems a value of 5 kg TS · m^{-3} is mostly assumed (Bever, 2000; DWA, 2009). In the discontinuous SBR process the influent flow rate is the similar to the effluent flow rate (Metcalf & Eddy, 2013). Several reactor tanks are recommended in order to improve process control (DWA, 2009). Therefore, each system provides two reactor tanks.

$$V_R = \frac{M_{TS,SBR}}{n \cdot SRT} + Q_{d,max} \cdot \frac{t_C}{24 \cdot n} \quad (44)$$

$$= \frac{1244}{2 \cdot 5} + 1.33 \cdot 168.1 \cdot \frac{8}{24 \cdot 2} = 162\ m^3 \quad \text{(SBR-Scenario)}$$

$$= \frac{177}{2 \cdot 5} + 1.33 \cdot 153.8 \cdot \frac{8}{24 \cdot 2} = 52\ m^3 \quad \text{(EGSB-SBR-Scenario)}$$

The hydraulic requirements for the reactor volume can be calculated by (45) with a volume exchange rate (f_{ex}) of 40%, that is manageable by the taken SVI of 100 ml · g^{-1} and the solid concentration of 5 kg TS · m^{-3} in the reactor (Bever, 2000):

$$V_R = \frac{Q_{d,max}}{f_{ex}} \cdot \frac{t_C}{24 \cdot n} \quad (45)$$

$$= \frac{1.33 \cdot 168.1}{0.4} \cdot \frac{8}{24 \cdot 2} = 93\ m^3 \quad \text{(SBR-Scenario)}$$

$$= \frac{1.33 \cdot 153.8}{0.4} \cdot \frac{8}{24 \cdot 2} = 85\ m^3 \quad \text{(EGSB-SBR-Scenario)}$$

For the SBR reactor design, the higher values have to be taken (Bever, 2000). As a result, two SBR reactors, each with a volume of at least 162 m^3 for the SBR-Scenario, and a volume of 85 m^3 for the EGSB-SBR-Scenario are implemented. Further corrections and design optimisations according to DWA (2009) M210 did not result in significant changes for LCA related parameters. The final volume can be seen in Table 28, with the height taken by recommendations of companies (ATB-Water, 2022).

Table 28: SBR reactor size and amount

parameter	n	V	h	A
unit	(amount)	m^3	m	m^2
SBR-Scenario	2	165	4.5	6.8
EGSB-SBR-Scenario	2	87.5	4.0	5.3

3.5.3.4. Aeration

Case specific Calculations according to German DWA rules, including related AVT guides, are done, in order to stay close to the actual design. In DWA (2016) A131 the oxygen requirement is related to the COD elimination (OV_C) and can be calculated by equation (46):

$$OV_C = C_{CSB,S,in} + C_{CSB,dos} - X_{CSB,BM} - X_{CSB,I,BM} \qquad (46)$$

Since nitrification and denitrification stages are not required in this study, only the carbon balance and the endogenous respiration are considered relevant for the aeration of the SBR. The carbon dosing term in (46), related to denitrification, can be neglected. Equation (46) can be simplified and translated into mass loading rates by multiplication with the relevant flow rate as shown in (47). Relevant for yearly average data and process comparisons is the daily average flow rate, that is often scaled down to hourly data (DWA, 2016). The hourly maximum peak consumption of O_2 for machinery capacity is increased for SBR systems compared to CAS (DWA, 2009), but cannot be provided due to data gaps in this study.

$$OV_{d,C} = \dot{m}_{COD_{S,in}} - \dot{m}_{X_{COD,BM}} - \dot{m}_{X_{COD,I,BM}} \qquad (47)$$

In the aerated tank a concentration of 2 mg · l^{-1} is recommended (DWA, 2016). The required O_2 intake (OC) for pure water to achieve a fixed concentration (C_X) in the biological treatment stage can be calculated by equation (48) (Teichmann, 1997). Apart from the O_2 requirements, the intake depends on the reactor volume (V), the saturation concentration for solubility (C_S), and the divergence to pure water (α -coefficiant).

$$\alpha OC = (OV_{d,C} + e \cdot m_{TS,SBR}) \cdot \frac{C_S}{C_S - C_X} \tag{48}$$

The concentration at the solubility equilibrium is dependent on the temperature and the height of the reactor for pressure-based aeration systems (Teichmann, 1997). According to EN 25814, the solubility of O_2 at 25° (C_{SS}) is 8.26 mg · l^{-1} (Teichmann, 1997). The solubility can be adjusted related to the aerated hight (h_a), and a pressure correction (49). The aerated hight (h_a) is 4.5 and 4.0 m in the SBR and SBR-EGSB scenario, respectively. The pressure quotient is assumed neglectable ($\frac{P}{P_0} \approx 1$) in standard examples (Bever, 2000).

$$C_S = \frac{P}{P_0} \cdot C_{SS} \cdot (1 + \frac{h_a}{20.7}) \tag{49}$$

$$= 1 \cdot 8.26 \cdot (1 + \frac{4.5}{20.7}) = 10.06 \text{ mg} \cdot l^{-1} \qquad \text{(SBR-Scenario)}$$

$$= 1 \cdot 8.26 \cdot (1 + \frac{4.0}{20.7}) = 9.86 \text{ mg} \cdot l^{-1} \qquad \text{(EGSB-SBR-Scenario)}$$

The electricity demand can be calculated from the OC by division of the SAE value. Typical values for WWTP are SAE values of 2 kg O2 per kWh for municipal wastewater with an α -coefficient (α_{SAE}) of 0.6 (OTT, 2022). The following correction can be made to calculate the electricity demand (E_a) (50):

$$E_a = \frac{OC}{SAE} \cdot \alpha_{SAE} \tag{50}$$

Substituting the values of equations (47), (48), and (49), in (50), the electricity demand can be calculated as shown in (51, p. 75). The reactor volume of both phase-shifted reactors combined (V_R) amounts 330 m^3 for the SBR scenario and 175 m^3 for the SBR in the EGSB-SBR scenario. The α -factor of brewery wastewater is on average 0.5 (DWA, 2010).

$$E_a = (\dot{m}_{COD_{S,in}} - \dot{m}_{X_{COD,BM}} - \dot{m}_{X_{COD,I,BM}}) \cdot \frac{C_S}{C_S - C_X} \cdot (SAE^{-1}) \cdot \frac{\alpha_{SAE}}{\alpha} \quad (51)$$

$$= (535.2 - 132.6 - 45.1) \cdot \frac{10.06}{10.06 - 2} \cdot (2^{-1}) \cdot \frac{0.6}{0.5} = 267.7 \text{ kWh} \cdot \text{d}^{-1} \quad \text{(SBR-Sc.)}$$

$$= (86.5 - 21.4 - 7.3) \cdot \frac{9.86}{9.86 - 2} \cdot (2^{-1}) \cdot \frac{0.6}{0.5} = 43.5 \text{ kWh} \cdot \text{d}^{-1} \quad \text{(EGSB-SBR-Sc.)}$$

While some guides consider an additional term for endogenous respiration next to BOD elimination with 0.1 kg per kg DM in the reactor (Teichmann, 1997), newer guides (DWA, 2016) and SBR examples (Bever, 2000) with focus on the COD fractions do not address this term.

3.5.3.5. Nutrient Dosing

The exact amount of nutrients needed for WWT operations varies on many factors related to the environmental conditions and parameters like the SRT (Metcalf & Eddy, 2013).

According to Henze (2000) the ratio of TN to VSS ($f_{VSS,N}$) can be estimated as 10% for heterotrophic-organisms in aerobic processes (on average), resulting in the following nitrogen requirements (52). About 80% of the DM in sludge (Metcalf & Eddy, 2013) can be considered VSS (cell fraction).

$$\dot{m}_{TN,BM} = f_{VSS,N} \cdot 0.8 \cdot F_{SP,M} \quad (52)$$

$$\dot{m}_{TN,BM} = 0.1 \cdot 0.8 \cdot 168.1 = 13.45 \text{ kg TN} \cdot \text{d}^{-1} \quad \text{(SBR-Scenario)}$$

$$\dot{m}_{TN,BM} = 0.1 \cdot 0.8 \cdot 23.6 = 1.89 \text{ kg TN} \cdot \text{d}^{-1} \quad \text{(EGSB-SBR-Scenario)}$$

The calculation for phosphorus (53, p. 76) is analogous to the one of nitrogen (52). The ratio of TP to COD ($f_{VSS,P}$) is 2.14% for brewery wastewater (Henze, 2000). Anaerobic digestion changes the composition of brewery wastewater, associated with lower values of $f_{VSS,P}$ (1.75%) in the anaerobic biomass (Stronach, 1986).

$$\dot{m}_{TP,BM} = f_{VSS,P} \cdot 0.8 \cdot F_{SP,M} \quad (53)$$

$\dot{m}_{TP,BM}$ = 0.0214 · 0.8 · 168.1 = 2.88 kg TP · d^{-1} (SBR-Scenario)

$\dot{m}_{TP,BM}$ = 0.0175 · 0.8 · 23.6 = 0.33 kg TP · d^{-1} (EGSB-SBR-Scenario)

For the EGSB-SBR-Scenario the calculated nutrient values are close to the influent data. The EGSB-SBR-scenario requires neither nutrient dosing nor nitrification, as the anaerobic process has a lower biomass yield connected to the nutrient demand. The downstream SBR process after the anaerobic stage operates with a suitable COD/N/P ratio, as most of the COD has been removed in the anaerobic EGSB.

The dosing of nutrients upstream of the biological treatment is necessary to balance an unfavourable COD/N/P ratio and support the growth of the biomass in the SBR-scenario.Ideally, the dosing of nutrient can be calculated (54) (55) as difference between nutrient demand (52, p. 75) (53) and mass loading rates (7.68 kg TN · d^{-1}, 0.95 kg TP · d^{-1} in the SBR-Scenario).

$$\dot{m}_{TN,Dos,min} = \dot{m}_{TN,BM} - \dot{m}_{TN,in} \quad (54)$$

= 13.45 − 7.68 = 5.77 kg TN · d^{-1} (SBR-Scenario)

$$\dot{m}_{TP,Dos,min} = \dot{m}_{TP,BM} - \dot{m}_{TP,in} \quad (55)$$

= 2.88 − 0.92 = 1.96 kg TP · d^{-1} (SBR-Scenario)

Since not all TN and TP of the influent are considered available for the bacteria (Metcalf & Eddy, 2013), a slightly higher dosing rate is chosen for the SBR-scenario, based on more reasonable minimum mass values to release reasonable effluent numbers in the modelling, after removing of additional TSS content with N and P content by a micro sieve. For the SBR-scenario, 6.90 kg · d^{-1} for TN and 2.20 kg · d^{-1} for TP is dosed as urea and mineral phosphate, respectively.

3.3.6. Polishing and Effluent of the Reference Scenarios

The same polishing treatment stage then for the root-IFAS scenario is applied to the SBR and EGSB-SBR scenarios. The removed mass loading rates for the micro sieve are calculated by equation (56), (57), (58), (59), and (60) similar to the mechanical pre-treatment stage of the EGSB-SBR-scenario in equations (2, p.56), (3, p.57), (4, p.57), (5, p.57), (8, p.57). The micro sieve has a TSS removal efficiency of 95% (Biopolus, 2022). The equation for the COD elimination (57) is further modified by distracting the inert soluble COD approximated by equation (27, p.67). The remaining part is estimated to be mostly particulate (95%) (KWB, 2022). For the particulate nitrogen and phosphorus in equation (58) and (59) a 80% rate of VSS to the solid part is considered similar to assumption for equation (52, p.75), (53, p.76). Equation (60) solves the equation (8, p.57) for the dry matter content with volume loss about 1 m^3 per day (Biopolus, 2020).

$$\dot{m}_{TSS,ele} = \dot{m}_{TSS,in} \cdot 0.95 \tag{56}$$

$$\dot{m}_{COD_{ele}} = \dot{m}_{X_{COD,in}} \cdot 0.95 \approx (\dot{m}_{COD_{in}} - \dot{m}_{S_{COD,I,in}}) \cdot 0.95 \cdot 0.95 \tag{57}$$

$$\dot{m}_{TN_{ele}} = \dot{m}_{X_{TN,in}} \cdot 0.95 \approx \dot{m}_{TSS_{ele}} \cdot 0.8 \cdot f_{VSS,N} \tag{58}$$

$$\dot{m}_{TP_{ele}} = \dot{m}_{P,in} \cdot 0.95 \approx \dot{m}_{TSS_{ele}} \cdot 0.8 \cdot f_{VSS,P} \tag{59}$$

$$x_{DM} = \frac{\dot{m}_{TSS,ele}}{V_{loss} \cdot \rho_{H_2O}} \tag{60}$$

The results of the equations (56), (57), (58), (59), (60) are presented in Table 29 by implementing mass loading rates of Table 30 (p.78), a sludge density of 1000 kg · m^3 (Metcalf & Eddy, 2013) and nutrient contents ($f_{VSS,N}$ = 10%; $f_{VSS,P}$, = 2.14%).

Table 29: Removal by micro sieve

Parameter	Scenario	x_{DM}	$\dot{m}_{COD_{ele}}$	$\dot{m}_{TSS,ele}$	$\dot{m}_{TN_{ele}}$	$\dot{m}_{TP_{ele}}$
unit		[%]	[kg · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]	[kg · d^{-1}]
	SBR	1.0	20.6	10.2	0.82	0.17
	EBSB-SBR	0.3	0.5	3.0	0.24	0.05

Additional subsequent stages include an effluent buffer tank, and a polishing pond in the root IFAS-scenario. Their effects on the wastewater effluent characteristics are not part of this study and previous modelling (Biopolus, 2022). For hydraulic control, the effluent buffer was changed before the micro sieve in designed scenarios. An overview of resulting water quality parameters of the SBR and can be seen in Table 30.

Table 30: Treatment with micro sieve

Parameter	Scenario	$Q_{d,av}$	COD	TSS	TN	TP
unit		$[m^3 \cdot d^{-1}]$	$[kg \cdot d^{-1}]$	$[kg \cdot d^{-1}]$	$[kg \cdot d^{-1}]$	$[kg \cdot d^{-1}]$
Raw wastewater		150.0	558.8	65.7	6.84	0.77
before SBR	SBR	168.1	570.0	74.6	7.68	0.92
	EBSB-SBR	153.8	99.1	5.8	3.02	0.42
after SBR	SBR	151.3	34.8	10.7	1.13	0.24
	EGSB-SBR	151.3	12.6	3.2	1.13	0.09
after micro sieve	SBR	150.3	14.2	0.5	0.31	0.07
	EGSB-SBR	150.3	12.1	0.2	0.89	0.04

A complete flow sheet with a mass balance of the SBR-scenario and the SBR-EGSB-scenario is provided in Appendix B (pp. 125–126).

3.3.7. Sludge Dewatering and Return Flow

Based on calculations and assumptions in the previous in chapters, the generated sludge is summarized in Table 31 (p.79). The excess sludge of the EGSB, SBR, and the tertiary micro sieve polishing is dewatered in the belt filter press in Koningshoven with an efficiency of 95% and a result of 22% DM (De Dommel, 2022). Polymer dosing for belt filter press units have been taken from literature with 0.003 to 0.005 kg AM per kg TS (AVT, 1994). The polymer consumption is calculated by equation (61), considering 350 operation days per year.

$$\dot{m}_{AM_{press}} = 0.4 \cdot \frac{350\ d}{a} \cdot \dot{m}_{TS_{SP}} \tag{61}$$

$$= 0.004 \cdot 350 \cdot 178.3 = 249.6 \text{ kg AM } \cdot a^{-1} \qquad \text{(SBR-scenario)}$$

$$= 0.004 \cdot 350 \cdot 50.4 = 70.6 \text{ kg AM} \cdot a^{-1} \qquad \text{(EBSB-SBR-scenario)}$$

Table 31: Sludge balance of the reference scenarios for the WWTP in Koningshoeven.

Parameter	Scenario	$\dot{m}_{TS_{SP}}$	x_{DM}	V_{SP}	$\dot{m}_{TP_{ele}}$
unit		[kg TS · d^{-1}]	[%]	[m^3 · d^{-1}]	[kg · d^{-1}]
Pre-treatment	EBSB-SBR	40.9	15.0	0.3	0.05
EGSB	EBSB-SBR	23.8	1.0	2.4	0.33
SBR	SBR	168.1	1.0	16.8	2.88
	EBSB-SBR	23.6	1.0	2.4	0.33
Micro sieve	SBR	10.2	1.0	1.0	0.17
	EGSB-SBR	3.0	0.3	1.0	0.05
Sum before dewatering	SBR	178.3	1.0	17.8	3.05
	EGSB-SBR (without pre-treatment)	50.4	0.9	5.8	0.71
	EGSB-SBR pre-treatment	40.9	15,0	0.3	0.05
Sum after dewatering	SBR	169.4	22.0	0.8	2.90
	EGSB-SBR (without pre-treatment)	47.9	22.0	0.2	0.67
	EGSB-SBR pre-treatment	40.9	15,0	0.3	0.05

The remaining 5% waste of the belt filter process are returned to the inlet of the WWTP next to the buffer tank. Flushing water is added to the return flow. The return flow is calculated by an internal loop with the following assumptions:

- Flow rate: Sum of the loss of water in the sludge (1% DM to 22% DM) and additional spoiling water (≈ 1 m^3 per day).
- TSS: 5% of the incoming TS content in the sludge, due to a 95% efficiency of the belt filter press
- TN: The TSS mass of the return flow is considered to contain 8% TN. On top, the water flow was calculated to imply TN loads correlating with effluent concentrations of the processes where the sludge has been generated.
- TP: The TP is calculated by a closed TP mass balance (Appendix B, pp. 125-126)

The loop started by the return flow of the root IFAS scenario and has been calculated for the SBR-scenario and the EGSB-SBR-scenario by excel (50 times) and further adjusted manually by the simulation in UMBERTO® LCA+ (IFU, 2018).

3.3.8. Electricity Data Sets

Table 32 shows that certain components such as auxiliaries, tanks, and stages are shared across all scenarios, such as balancing tanks, screw pumps, the belt filter press, or the micro sieve. However, pump can be designed for different run times and higher power requirements for SBRs (DWA, 2009). The electricity demand for the sludge tank is assumed to be halved for the EGSB-SBR, because less sludge has to be aerated. In addition, the aeration and reactor-specific pumping and mixing requirements may differ.

Table 32: Power installations in all scenarios according to data of Biopolus (2020).

Installation	Installed Power [kW]	Runtime [h per d]	Working capacity	Source	Amount per scenario		
					root IFAS	SBR	EGSB-SBR
Mixer in equalizer	1.90	24.00	50%	Biopolus, 2022	1	1	1
Pump for dosing	0.37	2.00	100%	Biopolus, 2022	3	3	3
Vertical mixer for dosing	0.5	2.00	100%	Biopolus, 2022	3	3	3
Screw pump	0.75	1.50 or 2.00	100%	Biopolus, 2022; KWB, 2022 (DAF, 2h) (SBR, 1.50h)	1	1	1
Micro sieve	17.40 kWh			Biopolus, 2022	1	1	1
Aerated tank	43.54 kWh			Biopolus, 2022 aeration + pump	1	1	1
Sludge tank	13.20 kWh		100% (EGSB-SBR only 50%)	Biopolus, 2022 aeration + pump	1	1	1
Belt filter press	0.75	2.00	100%	Biopolus, 2022	1	2	1

The electricity for plant root enhanced IFAS, provided by Biopolus (2022) for the flow rate of 150 $m^3 \cdot d^{-1}$, is determining by WW parameters and the daily operating time for each piece of equipment on-site, and the load factor for each unit. The electricity demand for the aeration and the DAF operation have been adjusted by one third of the efficiency of the system at the maximum capacity of 450 $m^3 \cdot d^{-1}$. The aeration of the root IFAS has been calculated about 311.44 $kWh \cdot d^{-1}$.

Some installations, like reactor specific pumping and mixing, differ between the scenarios, as can be seen in Table 33.

Table 33: Power installations divergent in the scenarios.

Installation	Installed Power [kW]	runtime [h per d]	working capacity	Amount	source
Root IFAS only					
Submersible centrifugal pump	1.7	24	42%	2	Biopolus, 2022
DAF	4.25	24	33%	1	Biopolus, 2022
Polyelectolyte dosing	0.75	0.5	100%		Biopolus, 2022
SBR for reference scenarios (* = SBR in EGSB-SBR different)					
Filling pump	3.00 (3.96*)	2.25	100% (50%*)	4 (2*)	KWB, 2022; Parameshwaran, 2003
Decant system	0.37	2.25	100% (50%*)	2	GVA, 2022
Mixing	1.15 (0.7*)	16.5	100%	1	DWA, 2017
EGSB-SBR scenario only					
Micro strainer	1.5	24	80%	1	Huber SE, 2022
Submerged centrifugal pump	3	24	50%	1	KWB, 2022 estimate (influent)
Recirculation pump EGSB	2	24	70%	1	Veolia, 2022
Screw system	0.15	2	100%	1	Veolia, 2022
Side channel blower (of the gas storage)	0.2	12	100%	1	Sjerp & Jongeneel B.V.; 2022

The SBR fill and draw operation requires more electricity for pumping and decanting than the root IFAS needs. In addition, SBRs contains an internal withdrawal of excess sludge and no secondary clarifier, like the DAF operation, with less chemical dosing requirements.

The EGSB reactor is characterised by an additional recirculation pump, and less electricity demand for the downstream aerobic treatment and the sludge handling.

The electricity demand of the WWTP operation and credits due to biogas use are listed in Table 34 for all four scenarios. The additional electricity demand for the water reuse scenario can be taken from chapter 3.3.3 (p.55).

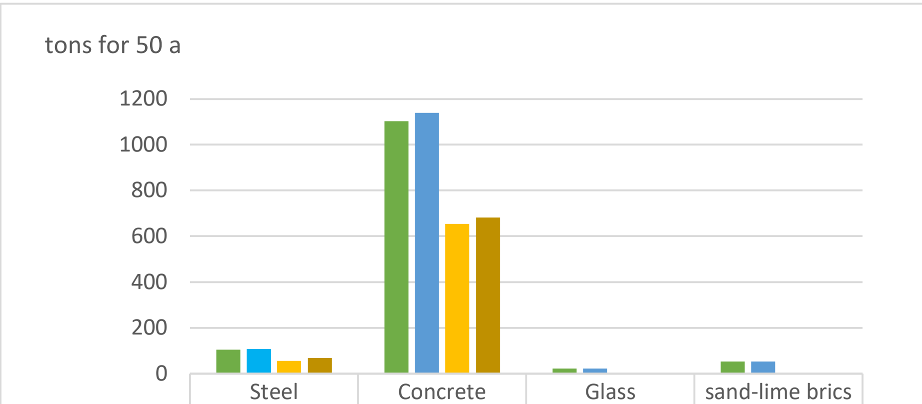
Table 34: Electricity demand oft the four different scenarios

Electricity demand [MWh · a^{-1}]	**root IFAS**	**root IFAS-NF**	**SBR**	**EGSB-SBR**
Aeration	109.0	109.0	93.5	15.2
Pumping and mixing	26.0	31.2	31.5	44.7
Phase separation & sludge handling	24.3	24.3	11.6	8.9
Chemical dosing	11.7	11.7	11.1	11.1
Membrane	-	10.1	-	-
Gas treatment	-	-	-	0.8
Pre-treatment	-	-	-	10.1
Total	171.0	186.3	147.7	90.8
Credits (not included)				
Sludge digestion	−24.2	−24.2	−23.9	−12.2
Heat credit				−423.2

The LCA method aims to cover the average and normal operation of processes and the product system and can therefore not consider the risk of abnormal operations related to accidents, leakages, and unintended instabilities (European Commission, 2010). The EGSB operation is considered stable at the given temperatures (EBC, 2003). Since the LCA method calculates a normal operation phase, no additional energy demand for heat exchanger units to stabilise the biological process, the anaerobic EGSB in particular, is added. However, a heat exchanger is considered as part of the construction part.

3.3.9. Infrastructure Data Sets

This part is a summary of the infrastructure material used for each scenario. For the evaluation, estimations have been made by visiting the full-scale WWTP in Koningshoeven and analysing the size and weight of constructions and materials. Additional data were gathered from supplies of installations, related reports, and literature data. The lifetime is calculated 15 a for the machinery, and 50 a for buildings and tanks. As can be seen in Figure 26, the construction refers mostly to concrete and steel. Glass and sand-lime bricks have been used for the greenhouse of the root-IFAS. Minor construction materials include, besides plastics, materials that are resistant to corrosion like stainless steel and Glass fibre Reenforced Polyester (GRP), that used mostly in EGSB-SBR scenario (Figure 27).



	Steel	Concrete	Glass	sand-lime brics
root IFAS	104,7	1102,2	22,0	52,8
root IFAS-NF	107,7	1138,4	22,0	52,8
SBR	56,2	654,3	0,0	0,0
EGSB-SBR	67,7	682,1	0,0	0,0

Figure 26: Mass of major construction materials of the scenarios.

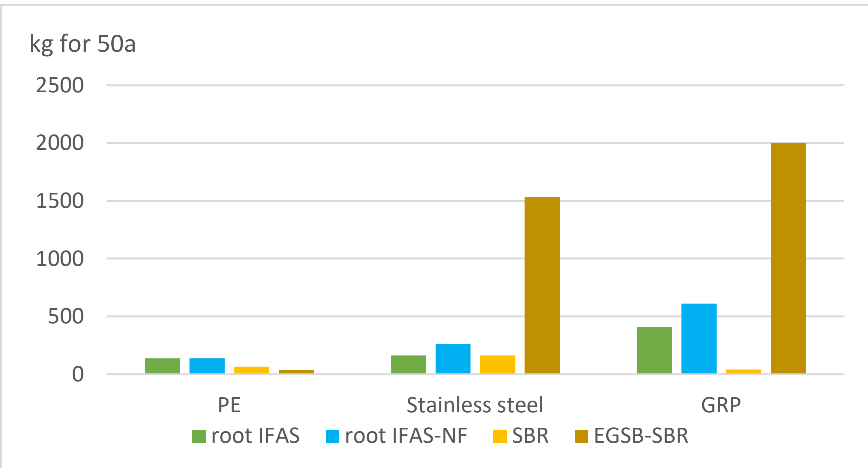


Figure 27: Mass of minor construction materials of the scenarios.

A variety of materials have been identified for the installations, that are considered

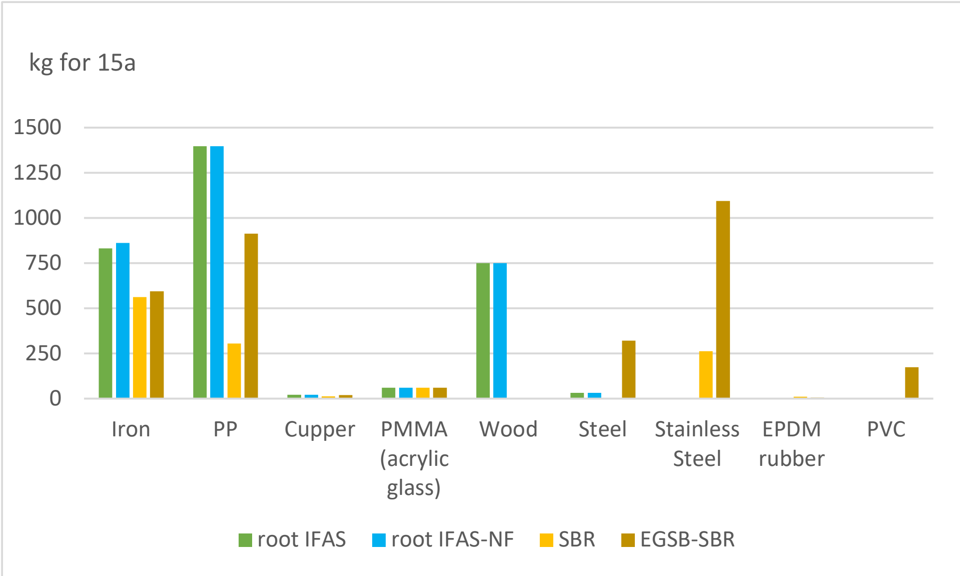


Figure 28: Mass of material for the installations

to replace after 15 years (Figure 28).

Pips have been transferred by their density to 200 diameters, that was available in the Ecoinvent (2021) databank. It can be seen in Figure 29 that the root IFAS-NF scenario has the most pipes to implement a water reuse concept.

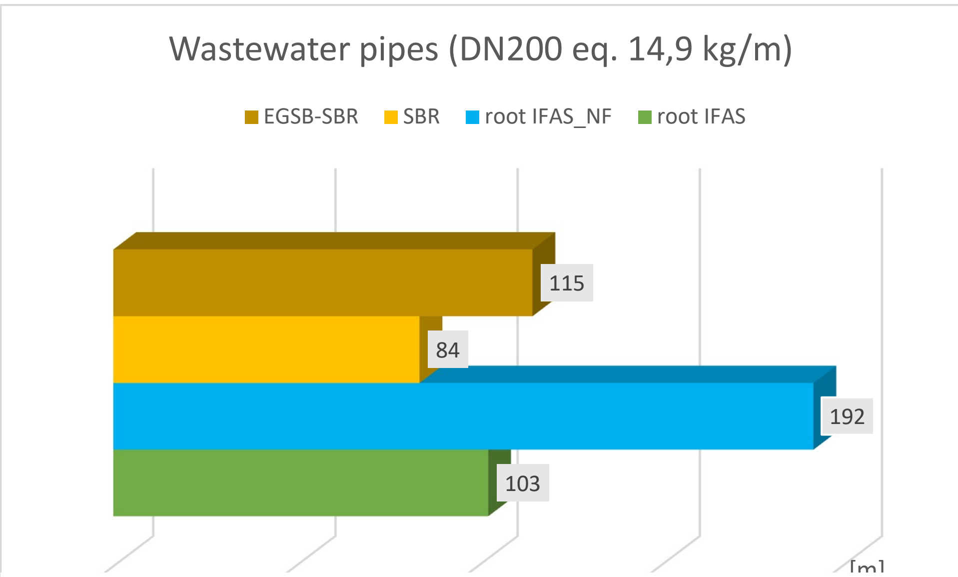


Figure 29: Pipes used, transferred in [m] (adjusted by size and density).

4. Results and Interpretation of the Life-Cycle Assessment (LCA)

In this chapter the LCIA results are presented and interpreted. First the three biological processes (root IFAS, SBR, EGSB-SBR) are discussed. Second, the difference between the root IFAS and the root IFAS-NF scenario is illustrated. Third an overview of the results is given and limitations are discussed.

4.1. Comparison of the biological Treatment Stages

In the following each of the seven environmental impact indicators is presented separately with emphases to major contributors. In

4.1.1. Cumulative Energy Demand (CED)

The cumulative energy demand (CED) CED of non-renewable energy resources for the three scenario approaches is show in Figure 30 (p.86). A total of 2018 GJ are needed for the root IFAS scenario, while 24% of it can be balanced out due to offsetting. This adds up to −1533 GJ $\cdot$ a^{-1} net primary energy requirements, which subdivides into −1379 GJ $\cdot$ a^{-1} from fossil and −154 GJ $\cdot$ a^{-1} from nuclear energy resources.

The root IFAS is mainly driven by the direct electricity consumption for WWT (72%), followed by the dosing of additional nutrients (16%), and chemicals (9%), that refers to polymer (5%) and other chemicals (4%), such as NaOH, H_2SO_4 and $FeCl_3$. Infrastructure has only a minor impact (3%). The offsetting is related to agricultural use of solids as sludge disposal and the recovery of electricity by anaerobic digestion, via external sludge treatment at food industries.

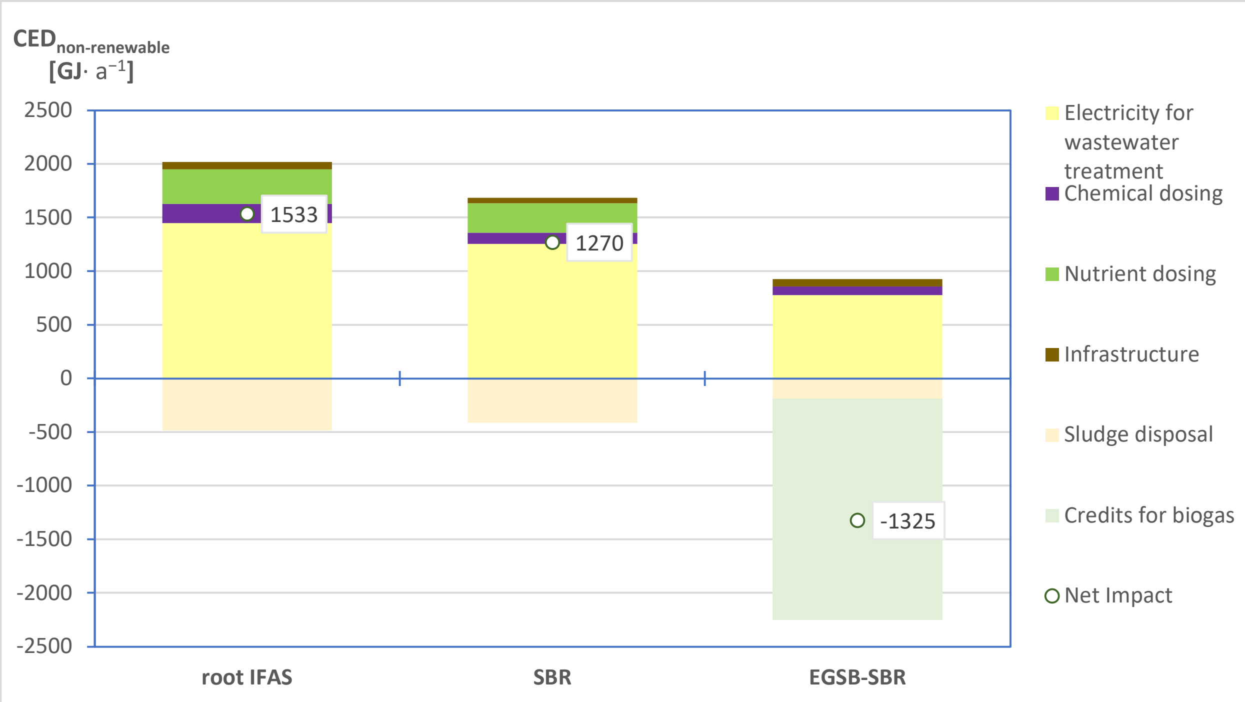


Figure 30: CED of non-renewable energy resources of the biological WWT scenarios.

Figure 30 also shows the CED for the alternative scenarios of the current WWT system.

The net CED of the aerobic alternative scenario (SBR) is 17% lower than the baseline (root IFAS) and amounts −1270 GJ · a^{-1}. 10 of the 17 percentage points difference result from the 14% lower electricity consumption of the SBR. The different factors that contribute to the CED of the SBR scenario are split similarly as for the baseline of the root IFAS with electricity use making 74% of the CED of the SBR, followed by nutrient dosing (16%), chemical dosing (6%) (half of it polymer dosing) and infrastructure (3%) with an off-setting of 25% through efficient sludge disposal.

Savings in the EGSB-SBR scenario are remarkable, related to the heating of biogas (143%) and sludge disposal (21%). The offsetting exceeds its relatively low contribution (926 GJ · a^{-1}) compared to the other scenarios, and result in a net energy-positive value of the CED of + 1325 GJ · a^{-1}. The electricity, directly used at the WWTP, of the anaerobic technology results in a 46% lower amount of contribution as the direct electricity demand of the root IFAS scenario (or a 38% lower value than in the SBR scenario), while still ranking as dominant main contributor (84%) of the EGSB-SBR scenario. The total contribution of the infrastructure is higher than in the other scenarios (−71 GJ · a^{-1} compared to the root IFAS), while the chemicals for pH control remain similar and no nutrient dosing is required.

4.1.2. Global Warnung Potential (GWP)

Figure 31 shows the results and main contributors for the global warming potential (GWP) of the three scenarios. The current WWT with the root IFAS installation in Koningshoeven accounts 101 tons CO_2-eq. per year (130 tons contribution and 29 tons offsetting). This equates net 1.92 kg CO_2-eq. per m^3 wastewater treated.

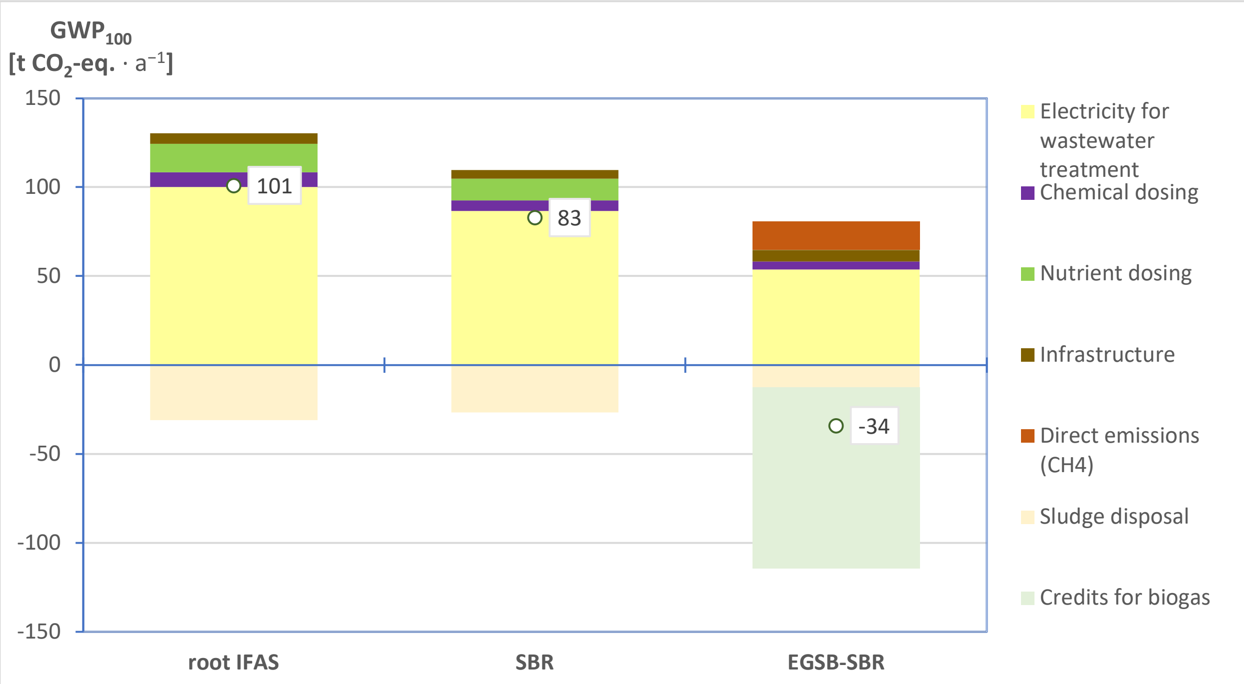


Figure 31: Global warming potential of the biological WWT scenarios.

Similarly, to the non-renewable CED results, the three WWTP conceptions (root IFAS, SBR, EGSB-SBR) are dominated by the electricity demand of the WWTP operation (76%, 79%, 66%), and influenced by nutrient dosing (12%, 11%, 0%) and the off-settings due to the sludge digestion, and disposal (−24%, −24%, −15%), chemical dosing (6%, 5%, 6%) and the construction (5%, 4%, 8%).

Compared to the root IFAS, the aerobic SBR scenario potentially emits 18% fewer greenhouse gases, resulting in a total net GWP of 83 t CO2-eq. per year. The majority (75%) of the difference is due to the total electricity demand. The rest of the difference is connected to nutrient (20%) and chemical (13%) dosing. The lowered amount of phosphorus recovery and biogas gain in the external sludge treatment process reduces the difference by 24%. A simpler construction of the

SBR scenario compared to the root IFAS technology accounts for the remaining 6% of the difference.

In contrast, the scenario with anaerobic technology (EGSB-SBR) benefits from a 46 % lower electricity consumption, and the avoidance of nutrient dosing compared to the root IFAS scenario. 16 t CO_2-eq. · a^{-1} (20% of the GWP without offsetting calculation) refer to direct emissions from the CH_4 losses by the dissolved fraction in the anaerobic treatment. The anaerobe biogas production of the scenario replaces natural gas in the boiler of the brewery and credits 26% more emissions then the entire WWT operation contributes. The scenario with anaerobe technology results to a negative emission value of −34 t CO_2-eq. · a^{-1}. Negative net emissions refer to off-setting credits that fully compensate the overall emissions of the system (Chen, 2013), what indicates the EGSB-SBR treatment as the most climate friendly option of the analysed scenarios. The intensive infrastructure of the anaerobe scenario is higher than in other scenarios, and less benefit can be achieved from sludge utilisation, correlating with the overall lower sludge production in anaerobic systems.

4.1.3. Photochemical Oxidant Formation Potential (POFP)

Figure 32 compares the Photochemical Oxidant Formation Potential (POFP) of the three biological WWTP scenarios. The root IFAS scenario has the highest POFP of 250 kg NO_x-eq. · a^{-1}, with 27% offsetting (net value of 181 kg NO_x-eq. · a^{-1}).

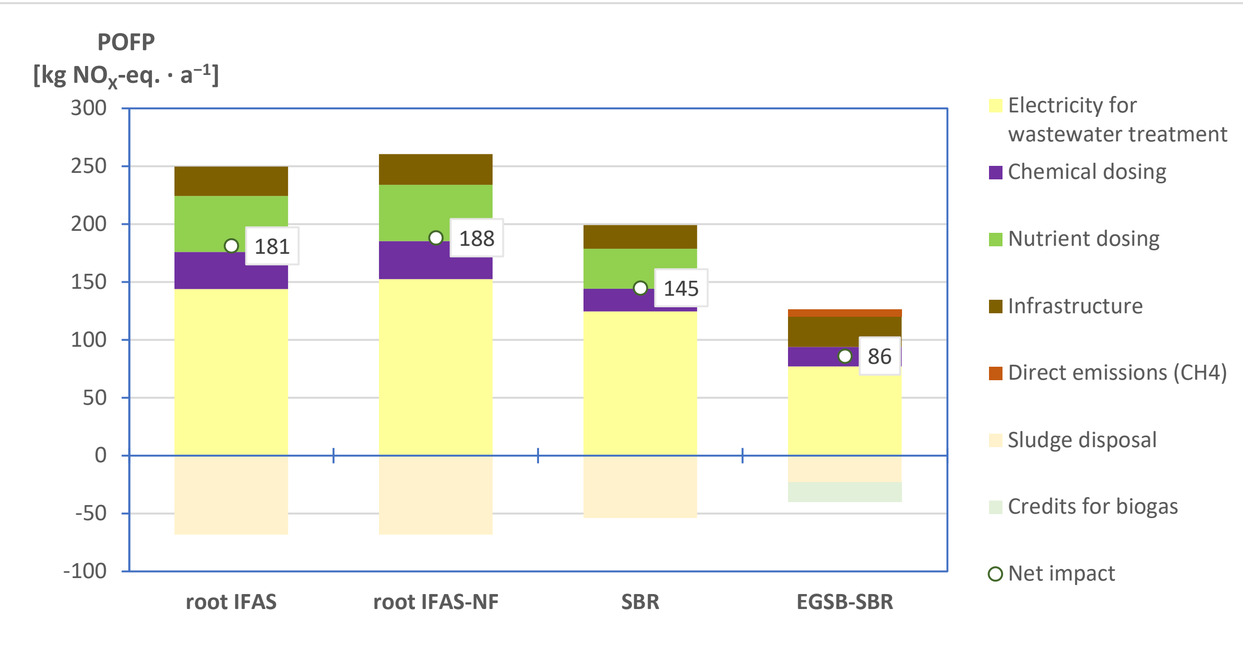


Figure 32: Photochemical oxidant formation potential of the biological WWT scenarios.

The formation of reactive chemical compounds such as ozone as a result of exposure to photochemical oxidants can lead to various respiratory health problems, which can vary depending on the presence of OH-reactive hydrocarbons, nitrogen oxides (NOx), and carbon monoxide (CO) (Zang, 2014). The POFP in the IFAS scenario is mainly driven by the electricity demand for the WWTP through fossil electricity creation (58%). The second largest driver is nutrient dosing (19%). In relation to other impact categories a relatively higher contribution of infrastructure (10%) and chemicals dosing (13%) can be seen in the results. The POFP is partly offset by sludge disposal (27%) for external electricity generation and as a replacement for chemical fertilizer in agriculture.

After offsetting 27% of the POFP with sludge deposal, the SBR scenario makes 145 kg NO_x-eq. · a^{-1}. While the proportion for the main contributors are similar, higher total values are generated by the root IFAS scenario related to its higher electricity consumption (+ 19 NO_x-eq. · a^{-1},), the phosphorus dosing (+ 17 NO_x-eq. · a^{-1},), and chemicals for the DAF operation (Fe_3Cl and Polymer) (+12 NO_x-eq. · a^{-1},) compared to the SBR scenario. The infrastructure makes about 10% of the POFP of the SBR Scenario. In contrast, the root IFAS archived more credits (21%) for sludge utilisation.

The EGSB-SBR scenario archieves a 52% lower POFP values of 86 kg NO_x-eq. · a^{-1}, including the consideration of the offsetting abilities. Savings can be achieved by the lower electricity demand (− 67 NO_x-eq. · a^{-1},), the absence of phosphorus dosing (− 35 NO_x-eq. · a^{-1},), and chemicals for the DAF operation (Fe_3Cl and Polymer) (− 16 NO_x-eq. · a^{-1}). However, lower offsetting (− 30 NO_x-eq. · a^{-1}) can be archived by the with biogas use (13%) and sludge treatment (18%) for the EGSB. Direct emissions due to methane losses result in a minor impact of + 6 NO_x-eq. · a^{-1}.

4.1.4. Ozone Depletion Potential (ODP)

Figure 33 (p. 90) shows the ozone depletion potential (ODP) of the three scenarios. The current WWTP (root IFAS) has the highest ODP with 7.1 g CFC-11-eq. · a^{-1}, that can be partially offset to a net value of 6.8 g CFC-11-eq. · a^{-1}.

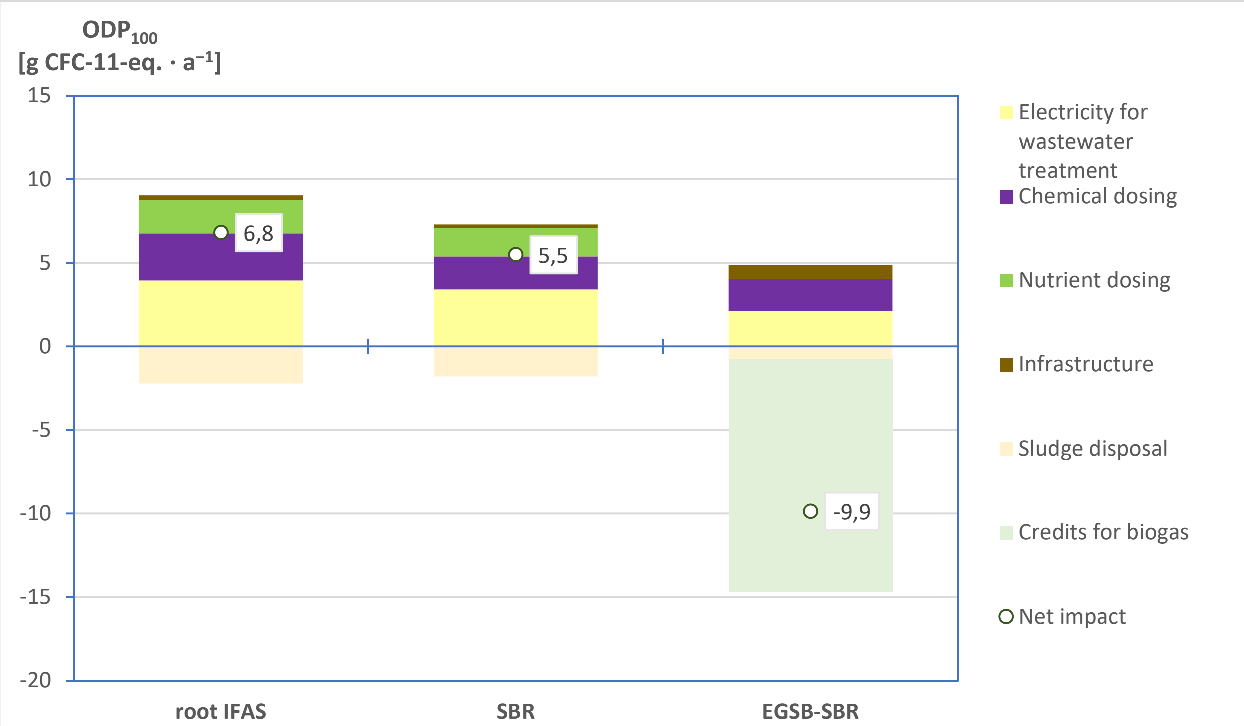


Figure 33: Ozone depletion potential of the biological WWT scenarios.

The largest driver for ODP is in all scenarios the electricity demand. However, its proportional impact (44–47%) is lower than for other metrics, such as the GWP. The ODP is also highly driven by caustic soda (1,8 g CFC-11-eq. · a^{-1}) as biggest contributor of chemical dosing (65%, 93%, 95%) and, for all scenarios except the anaerobic alternative (EGSB-SBR), by nutrient dosing (22–23%). The higher total OPD caused by chemical dosing (+ 0.8 g CFC-11-eq. · a^{-1}) of the root IFAS is related to iron dosing for the DAF operation. For the IFAS scenario, the contribution of P and N dosing is even, while the SBR scenario is mostly (69%) corelated to urea dosing. This shift corelates with the different N and P dosing ratios between the two calculations.ODP is offset by sludge disposal in all scenarios and largely offset by credits for biogas for the anaerobic alternative scenario (EGSB-SBR). These high offsetting numbers of 24%, 25%, and 16% for the root IFAS, SBR, and EGSB-SBR, respectively, are correlating with the effect of nutrient dosing, and electricity consumption that they offset.

The aerobic alternative (SBR) has a substantially lower ODP of 5.5 g CFC-11-eq. · a^{-1} and the anaerobic alternative (EGSB+SBR) has even a negative ODP with a credit of -9.9 g CFC-11-eq. · a^{-1}. Similar than for the GWP insecurities of the result are associated with unknown direct air emissions like N_2O.

4.1.5. Terrestrial Acidification Potential (TAP)

Figure 34 shows the Terrestrial acidification potential (TAP) in 100 years equivalents for the three scenarios. The root IFAS scenario has the highest TAP of 396 kg SO_2-eq. · a^{-1} with 48% offsetting (net: 208 kg SO_2-eq. · a^{-1}).

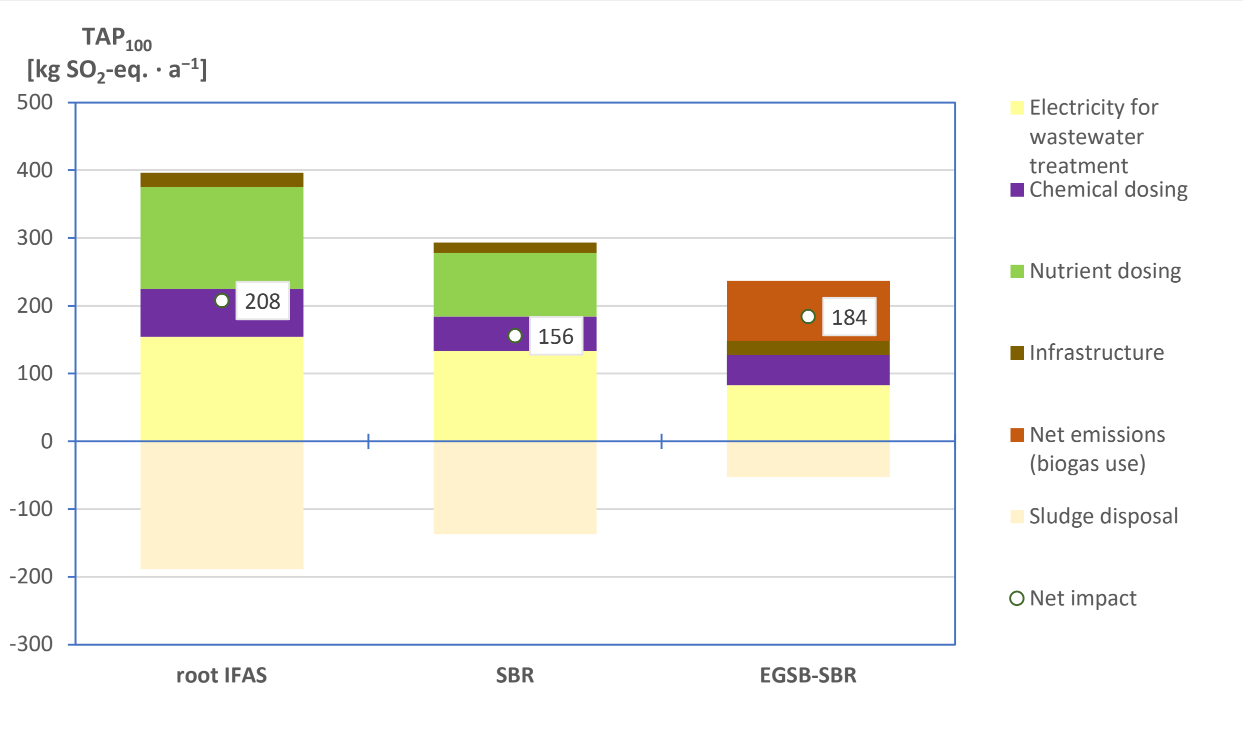


Figure 34: Terrestrial acidification results of the biological WWT scenarios.

Large drivers for TAP in the scenarios (root IFAS, SBR, EGSB-SBR) are electricity use (39%, 37%, 32%) and, except for the anaerobic alternative, nutrient dosing (38%, 31%, 0%). Chemicals dosing represents 17–18% of the TAP in the three scenarios. Important chemicals are the similar pH-buffering with sulfuric acid (26 kg SO_2-eq. · a^{-1}) and the difference of the IFAS and the other scenarios related to polymer and iron dosing from the DAF operation (+ 31 kg SO_2-eq. · a^{-1}).

The SBR has a TAP net value of 156 kg SO_2-eq. · a^{-1}. While the causes for the TAP show a similar distribution, lower impacts can be explained by the scenario's lower electricity use, chemical and nutrient dosing demand, and amount of phosphorus in the sludge, based on the design assumptions in this study.

The anaerobic alternative (EGSB-SBR) TAP impact is calculated between the two other compared scenarios with a net value of 184 kg SO_2-eq. · a^{-1}. Despite a lower impact of electricity use in relation to the other scenarios (about 50% of the root IFAS), the biogas use creates an additional burden for the EGSB-SBR scenario, due to direct emissions, that result to a net value of 88 kg SO_2-eq. · a^{-1} for the entire biogas use (including subtracted credits for the replacement of natural gas).

4.1.6. Freshwater Eutrophication Potential (FEP)

Figure 35 shows the Freshwater eutrophication potential (FEP) for the three scenarios. The current WWTP (root IFAS) has the lowest FEP with 20 kg P-eq. · a^{-1}, that can be partially offset to a net value of 18 kg P-eq. · a^{-1}.

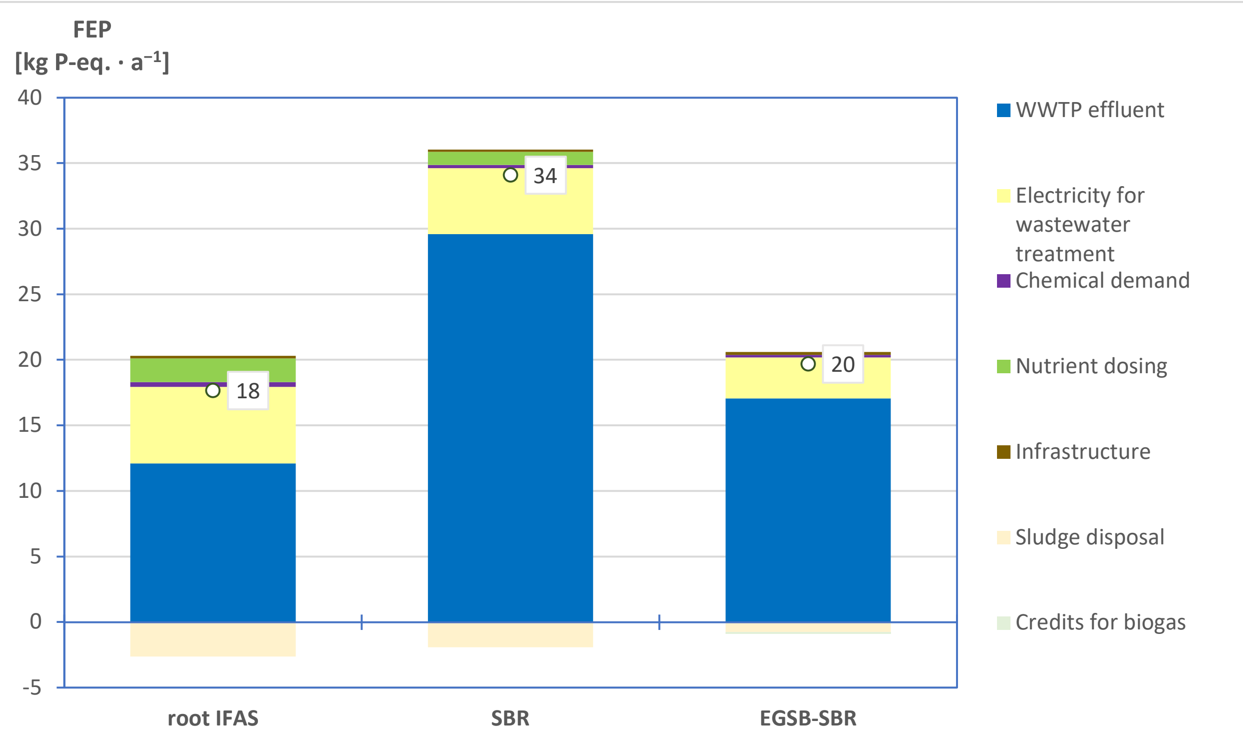


Figure 35: Freshwater eutrophication potential of the biological WWT scenarios.

For all three scenarios (root IFAS, SBR, EGSB-SBR), FEP is mainly driven by WWTP effluent (60%, 82%, 83%), followed by electricity use (29%, 14%, 15%) and to a small amount offset by sludge disposal (13%, 5%, 4%). Nutrient dosing contributes to the root IFAS by 9% and to the SBR by 3% of the total impact. Infrastructure contributes to FEP in a small amount (around 1%).

The aerobic alternative (SBR) has the highest FEP with 34 kg P-eq. · a^{-1}, whereas the other scenarios have similar values of 18 (root IFAS), 20 EGSB-SBR). kg P-eq. · a^{-1}. The main reason lies in the improved effluent water quality, in particular for phosphorus, that can be seen related to the uptake of the biofilm (Jabari, 2014), and iron dosing in the root IFAS compared to the SBR, that leads to a difference of 18 kg P-eq. · a^{-1}.

The EGSB-SBR scenario archives an 11% higher value as the root IFAS. Despite a significant lower impact of the electricity demand (3 kg P-eq. · a^{-1} instead of 6 kg P-eq. · a^{-1} for the root IFAS), the EGSB-SBR has a stronger impact related to the wastewater effluent (17 kg P-eq. · a^{-1} instead of 12 kg P-eq. · a^{-1} for the root IFAS). Impacts of infrastructure, biogas credit, and direct emissions show no significant impact (below 1%). The results can be treated with caution since the effluent data for direct discharge into the river channel highly depend on reactor the design assumptions of this study, based on excess sludge production and its nutrient content.

4.1.7. Marine Eutrophication Potential (MEP)

Figure 36 shows the Marine eutrophication potential (MEP) for the three scenarios. The root IFAS scenario has an MEP of 107 kg N-eq. · a^{-1} with 9% offsetting (net: 98 kg N-eq. · a^{-1}). While the FEP is mostly related to P emissions, the MEP emphasis the impact of nitrogen on the marine environment.

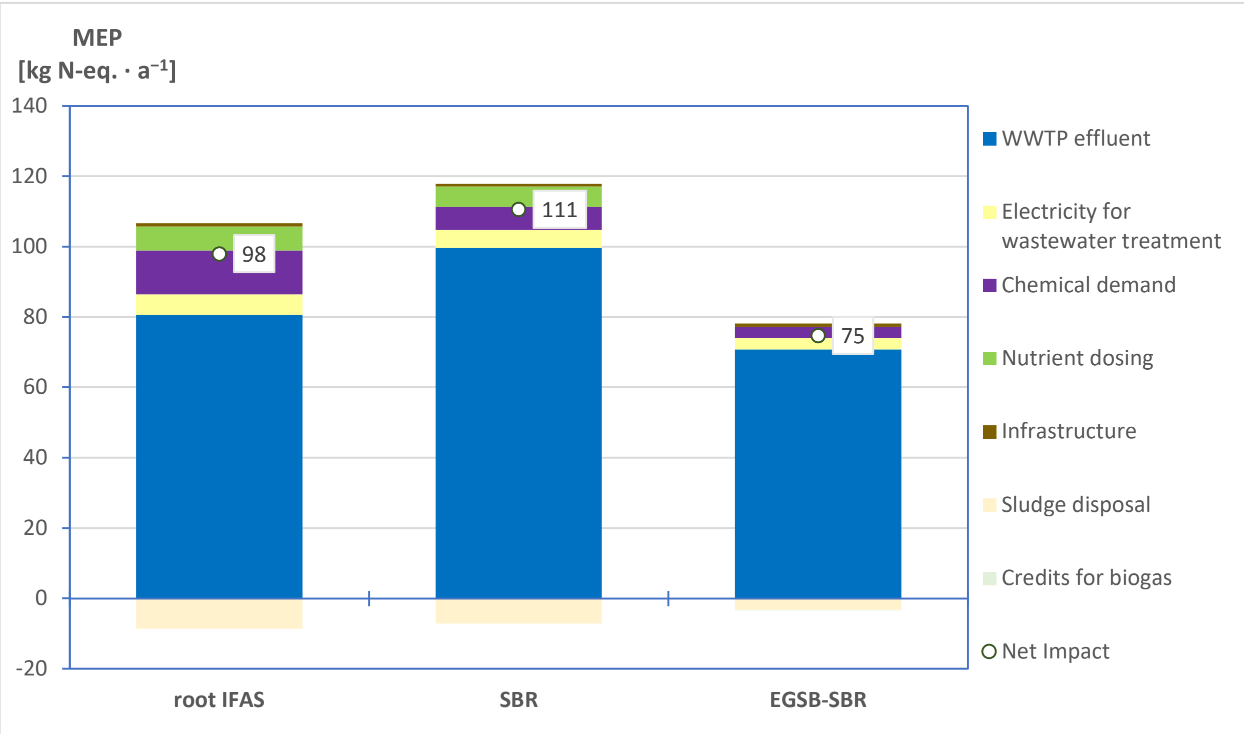


Figure 36: Marine eutrophication potential of the biological WWT scenarios.

MEP is mainly driven by WWTP effluent (76%, 85%, 91%), followed with a substantially smaller contribution by the chemical demand (12%, 6%, 4%), mainly (82–92%) polymer dosing, and by nutrient dosing (6%, 5%, 0%) for the analysed scenarios (root IFAS, SBR, EGSB-SBR). Credits from sludge disposal can offset 8%, 6%, and 4% of the total impact of the root IFAS, SBR, and EGSB, respectively. Electricity use (4–5%) and infrastructure (1%) have only a minor impact.

The SBR scenario has a net value of 111 kg N-eq. · a^{-1} in the MEP category, that is about 13% higher than for the root IFAS scenario. Despite a 50% smaller impact caused by the polymer consumption, the SBR scenario surpasses the burden related to the MEP category of root IFAS scenario due to the higher effluent values, correlating with TN in particular.

The lowest MEP is found for the anaerobic alternative (EGSB-SBR) with a net value of 75 kg N-eq. · a^{-1}. The result is highly dependent on the water quality parameters calculated by this study for the anaerobic-aerobic treatment scenario and have to be treated with caution. In this study, the direct discharge of the EGSB-SBR shows only 88% of the MEP impact of the root IFAS. The use of biogas and related emissions and potential credits show no significant impact in the MEP category.

4.2. Impact of the Nanofiltration and Water Reuse Concept

The water reuse scenarios with the hollow-fibre membrane system are analysed in this part. The impact of the freshwater supply of the brewery is unknown. Two possible crediting options are compared in this part, with

- the situation related to groundwater supply of the brewery by its own well, that is substituted by tap water production from the underground, and

- the replacement of external supply by Dutch water providers. This case is related to a more general market supply, for example if the brewery works above the well’s supply limit, or if the membrane system has to evaluated more independent from the current scenario hypothesis.

Figure 37 shows the difference in LCAI results for the root IFAS-NF scenario to the current root IFAS system for all analysed environmental impact categories.

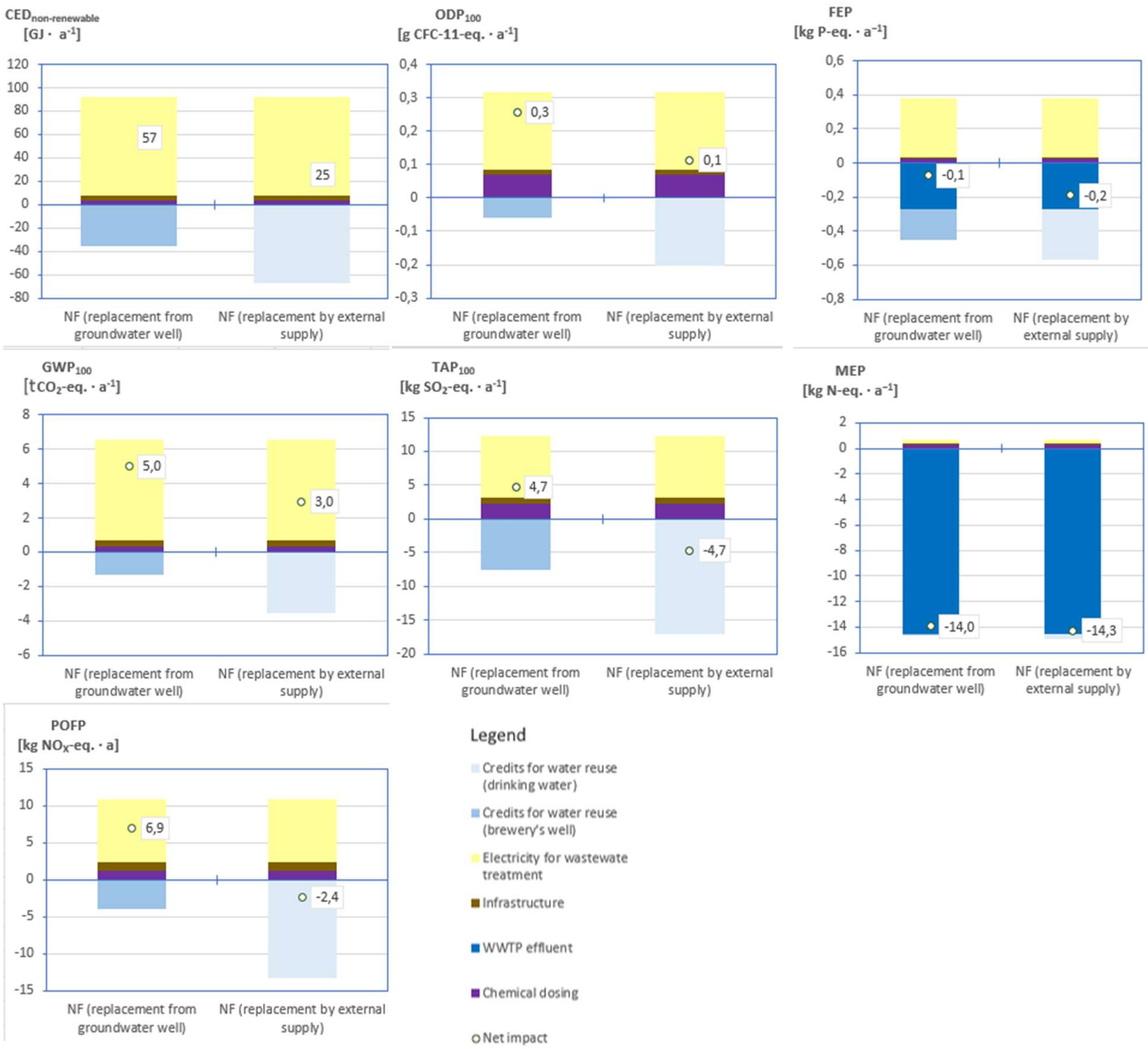


Figure 37: Differences in the results between the root IFAS-NF and the root IFAS scenario.

The nanofiltration system casts an additional burden to the CED (+ 93 GJ · a^{-1}) and the GWP (+ 65 t CO_2-eq. · a^{-1}) that is about 92% driven by the additional electricity demand. The reuse of water is more energy demanding than additional water production from natural resources and surpasses by the factor 2.6 or 1.3 for CED (and GWP by the factor 4.7 or 1.8) in terms of a replacement by the groundwater well or Dutch tap water respectively. About two thirds of the electricity demand refer directly to the membrane operation, while the rest reflects mostly the pumping of the recovered water to the brewery.

The credits for water reuse through the brewery's well (+ 3.9) are able to offset part of the total burden for the POFP (+ 10.8) to a value of + 6.9, while the credits for water reuse through the drinking water supply (+13.2) exceed the burden and contribute to a net positive impact of 2.4 kg NOx-eq*a-1. Main contributor remains the electricity demand (78%). The high impact of electricity can be explained by the release of pollutants like NOx due to burning processes in coal power plants. The impact of the infrastructure makes 11% of the POFP burden, while in all other categories the impact is relatively low between 1% and 5%.

For the ODP impact category, the main contributor to the net burden (+ 0.3) is also the electricity demand (73%), and in relation to the other categories the chemical demand is remarkable for ODP with citric acid (10%), NaOCL (7%), and NaOH (6%) for the differences. The water reuse can only mitigate the additional burden of the membrane operation by up to 19%. However, if the use of tap water can be avoided, a reduction of the burden to +0.1 is possible.

The membrane installation causes a TAP of 12.3, that can be mitigated by the benefit of water reuse to 4.7 kg SO2-eq./a. Electricity consumption makes 73% and chemical consumption 18% of the TAP burden. However, in case of water shortage from the well, a net gain of 4.7 SO2-eq./a can be archived by saving the Dutch tap water.

Nanofiltration has been demonstrated to yield advantageous outcomes in the context of water reuse, serving to mitigate the potential risk of eutrophication. Both FEP and MEP contribute towards a reduction in net impact by decreasing well water replacement by -0.1 and -14.3, respectively. Although the replacement of freshwater can yield greater benefits, the difference in impact between the two parameters is negligible, particularly for MEP. This is attributed to the fact that the majority of benefits stem from improved effluent loading rates, as the membrane serves as an additional cleaning stage in wastewater treatment.

4.3. Interpretation

In the following the LCA study is evaluated and discussed

4.3.1. Key Findings

An overview of the final results of the comparative Life Cycle Assessment between four WWT systems is presented by Figure 38 in seven environmental impact categories that are discussed in detail later in this chapter: CED, GWP, POFP, ODP, TAP, FEP, and MEP. For comparison, the results are scaled to the performance of the root IFAS scenario (100%).

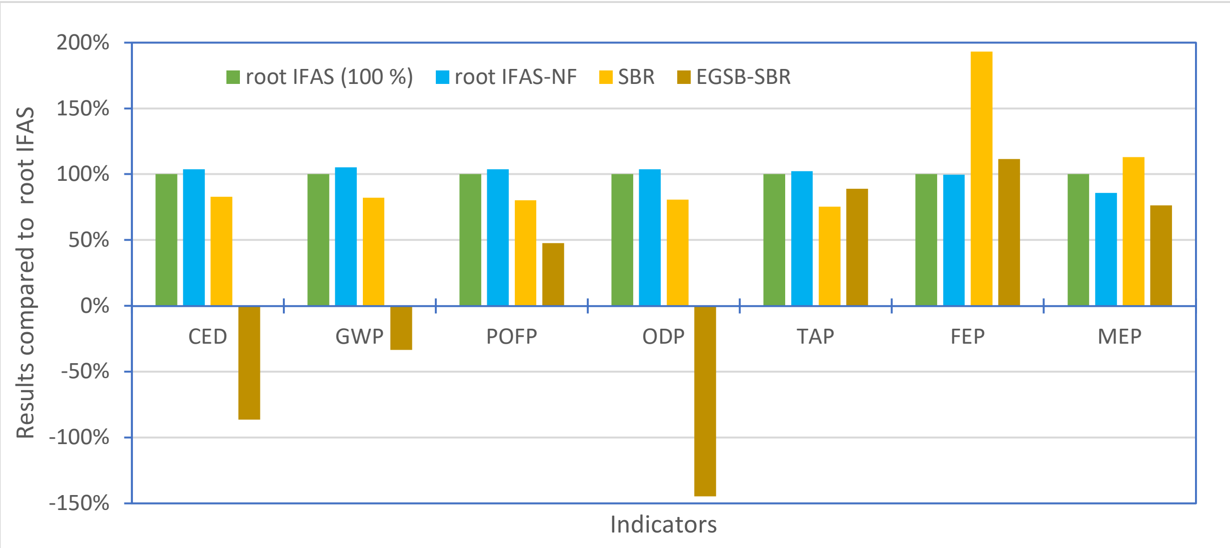


Figure 38: Relative results of the LCA study for all analysed environmental impact indicators

Comparing the aerobe alternative scenario (SBR) to the root-IFAS, Figure 38 shows that this alternative scenario is lower in CED, GWP POFP, ODP, TAP in comparison to the baseline, by 17, 18, 20, 19 and 25 percentage points, respectively. Similar to the literature the electricity consumption represents the main contributor to most environmental impact categories due to aeration (Longo, 2016). First assumptions of a lower energy demand of the root enhanced IFAS by the Nextgen research project (Poór-Pócsi, 2021) were contradicting the used input data of this study and could therefore not be verified or executed by this comparison. The low energy requirements of the SBR scenario can be attributed to the higher size of the designed reactor tank and the absence of the DAF unit, as well es a lower demand, polymer, and no iron dosing. In contrast, the root IFAS scenario suffers from the additional electricity and chemical demand due to the use of a DAF unit instead of a common settler (Metcalf & Eddy, 2016). However, SBR is higher in FEP by 93% and the MEP by 13% than the root IFAS scenario. The increase in eutrophication potential (FEP and MEP) of the SBR scenario is connected to the beneficial wastewater effluent qualities of the root enhanced IFAS, since the technique shows stronger phosphorus removal abilities (STOWA, 2017) and additional iron was dosed into the water at the DAF unit for improved liquid-solid separation. Furthermore, the DAF operation of the IFAS scenario requires additional polymer dosing resulting in minor contributions in all seven impact categories, especially CED, GWP, POFP, TAP and MEP

In the alternative scenario with anaerobic technology (ESGB-SBR), shows negative values for CED, GWP and ODP, that refer to a complete offsetting of their total environmental impact and large reductions compared to the root IFAS by 186, 133, 245 percentage points, respectively. This remarkable benefit compared to the other two scenarios is related to the use of biogas from anaerobic digestion in the boiler of the brewery, while less offsetting was achieved from sludge utilisation compared to the other scenarios. CH_4 has 25 times higher GWP than CO_2 in a 100 years timespan (IPCC AR4, 2007). However, direct emissions from biogas loss influence the GWP and POFP result, while a shift was evaluated to emissions from the incineration of biogas to impact disavowable the TAP results. For POFP, TAP, and MEP, this alternative scenario is lower by 52%, 11%, and 14% compared to the root IFAS, respectively. The second major benefit of the EGSB Scenario is related to its lower electricity requirements, that strongly affects all scenarios in each environmental category, with the exception of eutrophication. As a third benefit, no nutrient dosing was required for the EGSB-SBR scenario, while by contrast the calculated additional nutrient uptake of the root IFAS, in support of the advanced microbiological diversity, also results in unfavourable higher nutrient dosing rates for low nutrient content wastewater. Therefore, the IFAS shows an additional burden in all seven categories of the comparison related to the characteristics of the brewery wastewater, which contains, among other wastewaters of the food and beverage industries, lower nutrient contents. The FEP of EGSB-Scenario is higher by 12% in contrast to the root IFAS.

The difference is rather low, compared to changes of the reactor technology. In comparison to the 100% baseline (root IFAS), the root IFAS-NF shows a higher fossil and nuclear CED, GWP, POFP, ODP and TAP, with a difference of 4, 5, 4, 4 and 2 percentage points, respectively. In these five impact categories a dominating burden has been identified associated with the potential installation of the capillary nanofiltration membrane system, related to the high electricity demand of the membrane system itself, but also from the pumping to the brewery.

In the case of POFP and TAP the impact of water reuse would outbid the burden, if the access to water direct water supply of the brewery gets limited. While only a minor change in the overall net value of the FEP can be found, the MEP of the root IFAS-NF scenario is reduced by 14%. This effect relates to the improved effluent wastewater characteristics with a reduction of the COD ($-10\ kg \cdot a^{-1}$), TN ($-11.9\ kg \cdot a^{-1}$), and TP ($-1.3\ kg \cdot a^{-1}$) mass loading rates. The effect of water reuse crediting is below 3% for the MEP.

4.3.2. Consistency, Completeness, and Sensitivity Analysis

This paragraph emphasizes the significance of maintaining consistency, completeness, and sensitivity of the LCA analysis. In order to generate valid and reliable results, it is imperative to interpret the outcomes as comparative and relative statements, contingent on the underlying assumptions and comprehensiveness of the analysed product system or WWT scenario. LCIA results are relative and indicate potential environmental effects but cannot predict actual impacts, exceeding of thresholds, safety margins or risks. LCIA results are context-dependent and reflect potential environmental effects, but they cannot accurately predict actual impacts, exceeding of thresholds, safety margins, or risks. Therefore, it's crucial to interpret within their intended scope.

Reactor design

Common among LCAs in the field of WWT are the different data qualities and design assumptions (Corominas, 2013). The initial data set of the industrial-scale root IFAS was limited and the operation process was not considered stable in 2022. Inconsistencies relate to the different design approaches of the scenarios (Table 35). For example, the root IFAS and the SBR depend different modelling concepts for aeration parameters and the nutrient demand of the scenario.

Table 35: Design approaches.

WWT Operation	**root IFAS**	**NF**	**SBR**	**EGSB**
Approach	kinetic modelling	upscaling pilot trials	Design guidelines	Literature data
Design	SUMO, 2022	SEMiLLA, 2022	DWA, 2009; 2010; 2016	EBC, 2003

Assumptions based on analogies to characteristics of municipal wastewater have been implied in the DWA guideline approach and the SUMO modelling. The overall treatment efficiency for brewery wastewater can be considered higher than for the readily biodegradable fraction according to guidelines for municipal wastewater at lower temperatures (EBC, 2003). DWA (2016) A 131 recommends to compare with literature data reactor design with temperatures above 20 degrees. According to a review article by Mace et al. (2002), brewery wastewater treated by SBR systems was reported in COD removal ranges between 93% and 98% (94% was assumed for the SBR-Scenario by COD fractioning). The lower COD removal efficiency of the SBR reactors in the EGSB-SBR scenario (86%), can be explained by the difference between the raw wastewater and digested wastewater. Literature on examples of aerobic stages subsequent to anaerobic UASBs in breweries indicated removal efficiencies of 85% (Bischofsberger, 2005). In contrast to existing literature (STOWA, 2017), provided data indicated no evidence of low sludge production for the root IFAS.

Water quality

In order to fulfil the system function, the legal discharge goal has to be met. As can be seen in Table 36 all scenarios meet the local discharge limits and contribute different levels of water pollution.

Table 36: Overview of effluent water characteristics in comparison to the discharge limit.

Parameter	Unit	Discharge limit	Root IFAS effluent	Root IFAS-NF effluent	SBR effluent	EGSB-SBR effluent
Flow rate	$m^3 \cdot a^{-1}$		52,500	39,375	52,500	52,500
COD	$mg \cdot l^{-1}$	125	84	111	87	83
TSS	$mg \cdot l^{-1}$	10	3.1	4.1	3.4	1.3
TN	$mg \cdot l^{-1}$	10	1.5	1.7	2.1	5.1
TP	$mg \cdot l^{-1}$	1	0.2	0.3	0.5	0.1
Source		De Dommel, 2022	Biopolus, 2022	SEMiLLA, 2022	Appendix B, p.126	Appendix B, p.127

For the effluent water quality different assumptions are made. While the SBR and EGSB-SBR scenario data is calculated by literature data and guidelines, the plant root enhanced IFAS was modelled by Biopolus (2022) with the software SUMO (2022). The low phosphorus content in root IFAS scenario has resulted in low eutrophication values, and can be considered a main advantage of the new technology. However, in this study it remains unclear whether the effect was caused by the additional uptake by the biofilm and the plants or by the dosing of iron(III). Furthermore, the additional dosing of phosphorus had a negative impact on most other analysed environmental impact categories. The effluent quality and greenhouse gas emissions of an SBR system can be considered comparable with CAS (DWA, 2009). The final effluent of the root IFAS-NF scenario was calculated by SEMiLLA (2022) based on a 25% recovery rate into permeate. The remaining retentate of the capillary nanofiltration system is characterised as effluent with lower flow rates, associated with higher COD, TSS, TN and TP concentrations.

Electricity data

The present study has identified a significant contribution of electricity consumption, particularly aeration, towards various environmental impact categories. This finding is consistent with the observations made in the broader literature on LCA of WWTPs (Longo, 2016).

The overall range for the electricity demand of the aerobic reactors for the treatment of industrial brewery ranges between 0.7 and 1.0 kWh · (kg CSB_{ele})$^{-1}$, while anaerobic reactors have significantly lower requirements of around 10 times (Van Geest, 2010). Figure 39 shows that the root IFAS and SBR configurations align with the expected range for aerobic treatment, while the EGSB-SBR scenario, which involves both anaerobic and aerobic treatment steps, demonstrates lower energy consumption. Aeration plays a major factor of the electricity required for aerobic wastewater treatment systems. The integration of anaerobic and aerobic treatment steps in wastewater treatment systems, as in the

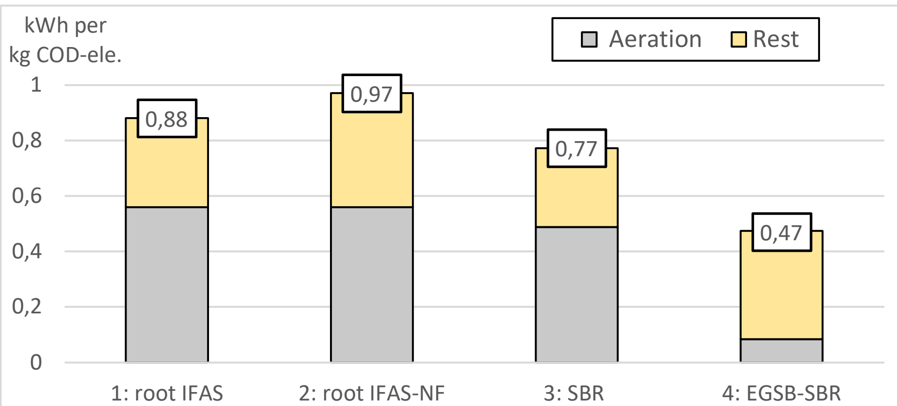
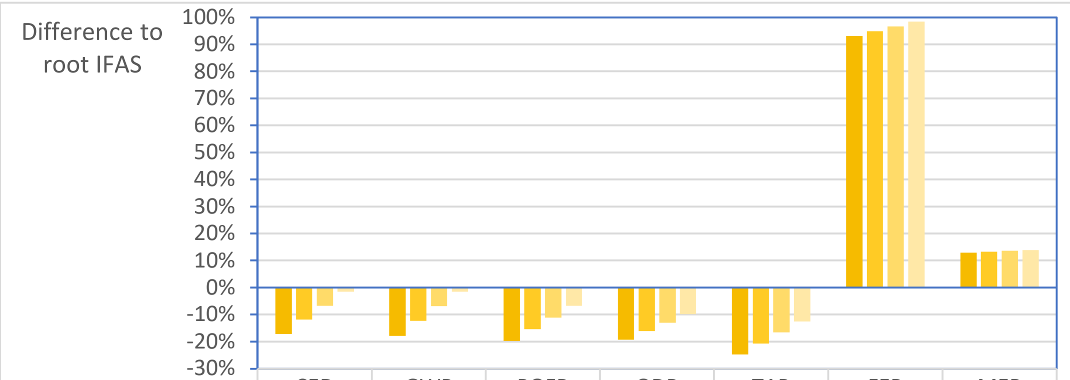


Figure 39: Specific electricity demand of the 4 scenarios

EGSB-SBR scenario, may lead to more energy-efficient systems, particularly in industrial settings (Bischhofsberger, 2005).

In this study, we conducted a sensitivity analysis to compare the root IFAS and SBR scenarios, as aeration is influenced by reactor design. Our results, as presented in Figure 40 demonstrate that a 30% increase in aeration electricity demand may impact the magnitude of differences between scenarios, but not their ranking. These findings highlight the robustness of the comparison between the root IFAS and SBR scenarios and suggest that their relative performance is relatively insensitive to changes in aeration requirements.

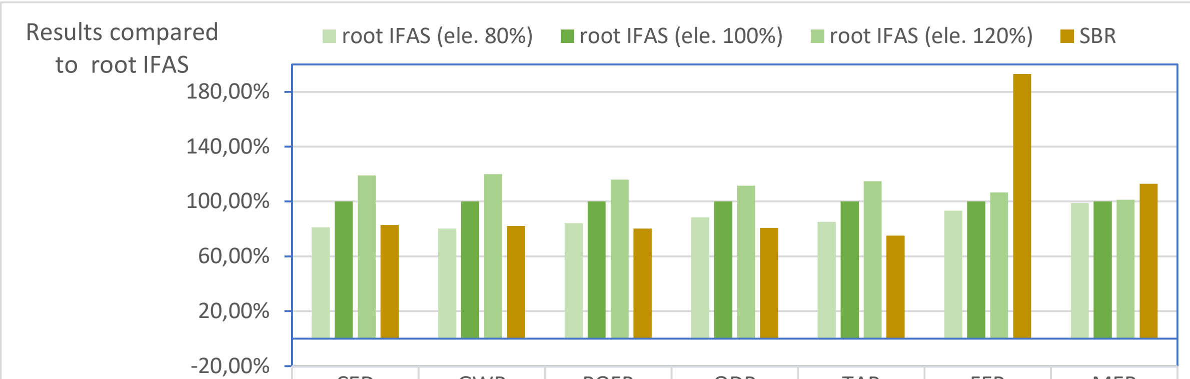


Indicator category	CED	GWP	POFP	ODP	TAP	FEP	MEP
SBR	-17%	-18%	-20%	-19%	-25%	93%	13%
SBR + 10% Aeration	-12%	-12%	-15%	-16%	-21%	95%	13%
SBR + 20% Aeration	-7%	-7%	-11%	-13%	-17%	97%	14%
SBR + 30% Aeration	-2%	-2%	-7%	-10%	-13%	98%	14%

Figure 40: Variation of the calculated electricity demand for aeration of the SBR scenario in relation to the root IFAS.

In this study, we encountered difficulties in collecting reliable data on the electricity demand for the plant root enhanced IFAS and DAF operation from the actual wastewater treatment plant due to operational corrections with intensified aeration. Therefore, we obtained relevant data from the technology provider and made analogous assumptions for scenarios involving additional comparisons and divergent design approaches. A sensitivity analysis of the overall electricity demand for the root IFAS is presented in Figure 41, p. 103. Our results suggest that a 20% reduction in the electricity demand of the root IFAS would be necessary to change the ranking in the CED and GWP between the SBR and root IFAS scenarios.

Despite the potential for improved aeration through the rhizosphere of higher plants, uncertainties regarding the overall energy-efficiency of the plant root enhanced IFAS exist in the literature (STOWA, 2017). The technology may face a potential limitation due to the water level of 1.5 m, which results in an inefficient reactor height for aeration (Teichmann, 1997). Future research could investigate improvements to the technology through testing on larger reactor sizes or the replacement of the DAF unit with a more energy-efficient secondary clarifier that has lower polymer and iron dosing demands (Teichmann, 1997; Metcalf & Eddy, 2013).



	CED	GWP	POFP	ODP	TAP	FEP	MEP
root IFAS (ele. 80%)	81%	80%	84%	88%	85%	93%	99%
root IFAS (ele. 100%)	100%	100%	100%	100%	100%	100%	100%
root IFAS (ele. 120%)	119%	120%	116%	112%	115%	107%	101%
SBR	83%	82%	80%	81%	75%	193%	113%

Figure 41: Changes of the environmental impact of the root IFAS according to variations of the overall electricity demand in reference to the SBR.

The environmental burden of the electricity demand was evaluated using the Dutch electricity mix of the Ecoinvent (2021) v3.8. The proportion of renewable energy resources could potentially lower the system's overall impact in the future, given the constant changes in the electricity mix related to political goals and the overall economic situation. However, a further analysis of the future energy market is not part of the attributional approach in this LCA.

Construction data

Infrastructure is expected to be a minor contributor while its often based on estimations due to its complexity and existing data gaps (Corominas, 2020). The root IFAS design may have been intended for higher flow rates (Poór-Pócsi, 2021), but modifying the construction would not have significant implications for the analysed categories. Morera et al. (2017) questions the validity of LCA studies on WWTP with primary aerobic biological treatment operations that report a construction phase impact of less than 5% for the GWP, ODP, or FEP, while considering a lifespan of 30 years or less. They suggest that the actual impact of construction might be underestimated, emphasizing the need for more accurate assessments. As shown in Table 37, the construction phase of the root IFAS and the SBR scenarios are within the theoretical range of 5–10% for GWP and OPD impact categories, as reported by Morera et al. (2017) if a lifespan of 30 years was taken. However, the FEP impact category shows a deviation from the theoretical range (Table 37) and thus warrants further scrutiny. Caution should be exercised regarding the FEP category, because evaluations related to eutrophication categories can be divergent, depending on the targeted ecosystem, the underlying analytical model of the LCIA indicators, and the relevance of specific substances such as phosphorus emissions (Hospido, 2012; Renou, 2008).

Table 37: Relative impact of the construction phase for the root IFAS and the SBR.

Scenario	**construction lifespan [a]**	**GWP**	**ODP**	**FEP**
root IFAS	30	9%	7%	2%
root IFAS	50 (this study)	6%	4%	1%
SBR	30	9%	6%	1%
SBR	50 (this study)	6%	4%	0%

Direct emissions

The consideration of emissions to water, air and soil depends on the selected impact categories. As can be seen in Table 38, some impact categories are significantly affected by certain pollutants of the WWT operation:

Table 38: Impact of direct emissions from a WWTP (Corominas, 2020).

Substance family	**GWP**	**FEP & MEP**	**TAP**	**ODP**	**Human toxicity**	**Eco-toxicity**
Carbon (COD)	CH_4; CO_2 (fossil)	BOD				
Nitrogen (N)	N_20	NH_4^+, NH_3	NH_3	N_20		
Phosphorus (P)		PO_4^{3-}				
Sulphur (S)			H_2S		H_2S	
Heavy metals						x
Micropollutants					x	x

In this study, it should be noted that direct emissions such as H_2S and volatile organic compounds (VOCs) from anaerobic digestion and biofilm ecosystems have not been evaluated or estimated. Similarly, data limitations and low nitrogen contents in the raw wastewater have excluded the impact of N_2O. It is important to consider that direct N_2O emissions can have a significant impact on the GWP of WWTPs, as highlighted by Corominas et al. (2020). However, N_20 emissions are associated with the process of nitrification (DWA, 2022a). A unsignificant low impact has been attributed, as no nitrification step was required in the WWT process. In contrast, the direct air emissions from controlled biogas combustion and CH_4 losses have been estimated, based on literature comparisons, in this study and impacted the GWP and POFP in particular. As can be seen in Figure 42 and Figure 43, no significant change can be made for the EGSB-SBR scenario, with the different literature assumptions of the calculation in the construction part.

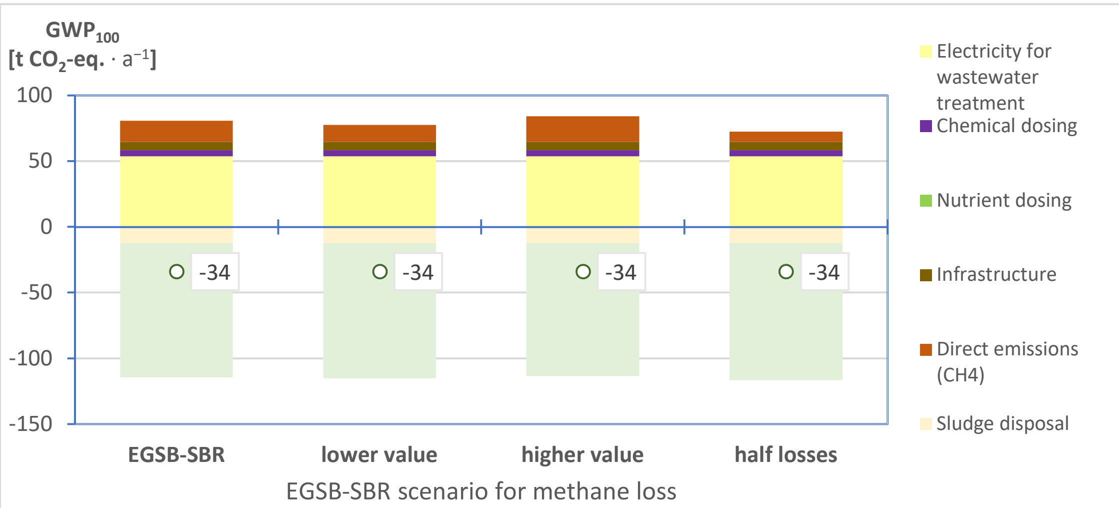


Figure 43: Sensitivity analysis of the methane loss for the GWP

Figure 42: Sensitivity analysis of the methane loss for the POFP.

The higher and lower values in Figure 42 (p.105) and Figure 43 (p.105) represent assumptions made about on the potential CH_4 oversaturation of the wastewater. While the EGSB-SBR scenario is calculated with a dissolved fraction of 25.35 mg · l^{-1}, a lower (20.28 mg · l^{-1}) and higher value (30.94 mg · l^{-1}) have been analysed. In addition, the overall value was calculated to be halved, representing even less losses and further conversion of methane by bacteria.

Limitation of comprehensiveness by indicator coverage

A wide range of environmental impact categories have been investigated in this study. It should be noted, that the inclusion of all relevant categories aligns with the guiding principle of comprehensiveness in LCA studies (Finkbeiner 2006). Some categories like toxicity and ecotoxicity have not been evaluated in this study, due to data limitations. However, ecotoxicity is considered a key factor to evaluate the overall environmental impact in LCA studies on WWTPs (Coromoninas, 2020). Specifically, the further identification and modelling of the fate of different micropollutants, including heavy metals, pesticides and pharmaceutical substances, in the wastewater and the excess sludge was not applied in this study. The topic remains challenging in current LCAs in the field of wastewater treatment and could provide valuable insights into potential impacts on human health and the environment (Corominas 2013; Chen, 2013). In the case of brewery wastewater, the presence of zinc and nickel may be of particular concern (EBC, 2003). Furthermore, the incorporation of the plant root enhanced IFAS systems may potentially offer additional benefits, such as the removal of micropollutants (STOWA, 2017). Therefore, we suggest that future research integrate toxicology and ecotoxicology LCA results to improve the comprehensiveness to evaluate the environmental impact of WWTPs.

5. Conclusion

The aim of this study was to examine the potential environmental impact of a novel treatment technology (root IFAS) for industrial brewery wastewater in Koningshoeven by the utilisation of the LCA methodology for (1.) the identification of key contributing factors, (2.) a comparative assertion to reference scenarios with conventional WWT technologies, and (3.) the consideration of a water reuse scenario with nanofiltration membranes. Therefore, four different scenarios, including SBR and EGSB technologies for benchmark approaches, have been simulated and evaluated by seven different midpoint indicator categories. The scenarios were designed by various modelling approaches, guidelines, and literature data, incorporating LCI key parameters such as effluent quality, sludge disposal, electricity and chemical consumption, or construction materials. Furthermore, offsetting opportunities related to the use of biosolids, biogas, and recycled water were taken into account. In some instances, nutrient addition was necessary to support the growth of microorganisms during the biological treatment stage.

In all analysed environmental impact categories, besides eutrophication, the electricity demand of the root IFAS was identified as main contributor, responsible for around 75% of the total GWP and the CED in particular. POFP, ODP, and TAP are also strongly related to the dosing of phosphorus, nitrogen, caustic soda and polymers. The sludge disposal credited electricity and nutrient use and had therefore a significant impact on the same categories, with a maximum credit of almost 50% for the TAP category. In contrast, the FEP and MEP were mostly related to the water quality of the effluent and its correlating phosphorus and nitrogen mass loading rates.

The present study has demonstrated that a scenario based on aerated SBRs exhibits superior performance in comparison to the root IFAS in various environmental impact categories, lowering the impact of CED, GWP, POFP, ODP, and TAP, by 17%, 18%, 20%, 19%, and 25%, respectively. Primary reasons for this are the (12.5%) lower electricity and the ($2\ kg \cdot d^{-1}$) lower phosphorus demand of the SBR. The SBR profits from a higher tank size, associated with improved oxygen transfer rates of pressure aeration systems, and the absence of a secondary clarifier, such as the DAF unit for the root IFAS scenario. The integration of anaerobic technology (EGSB-SBR) further improved the system's performance, requiring less electricity and provided a more favourable COD/N/P ratio, related to less sludge production and biogas generation, that prevents additional N and P dosing. Moreover, the credits for generated biogas from anaerobic treatment completely offset impacts in CED, GWP, and ODP. Compared to the reference scenarios, the IFAS has lower N and P effluent concentrations, correlating with the implementation of the DAF operation with high iron dosing and a potential strong phosphorus uptake ability of the root IFAS system, resulting in beneficial low impacts on freshwater and marine eutrophication.

Compared to the different approaches for the biological treatment stage, implementing a downstream NF for water reuse resulted only in minor additional impacts (2-4%) on CED, GWP, POFP, ODP, and TAP. The added electricity demand outweighed the expected benefit from water reuse in the brewery. brewery. In case of shortage of water supply by the well of the brewery, a net positive result by the technology for POFP and TAP can be contrasted if compared to conventional tap water use. The membrane also improved the final effluent wastewater characteristics, resulting in a 14% improvement in the MEP category.

The outcomes of the sensitivity analysis endorse the stability of the overall findings, although recent literature on LCA theory suggests an even greater impact of the construction phase on the FEP. It should be noted that direct emissions of N_2O, H_2S, have not been fully accounted for in the overall comparison. Caution should be exercised due to variations in modelling approaches among the scenarios.

This study presents contributions towards LCA results of an emerging technology, with practical implications for sustainable wastewater management in the context of industrial brewery WWT, and highlights the trade-offs and shifts that exist between different types of environmental impacts and their mitigation and offsetting opportunities. Anaerobic EGSB and conventional SBR reactors offer significant advantages over the current root IFAS scenario in terms of many relevant environmental impacts and the potential for resource recovery. Specifically, the anaerobic EGSB method was found to outperform the aerobic treatment approach in most analysed categories. Optimizing reactor tank size and reducing chemical and nutrient dosing requirements could potentially enhance the root IFAS environmental performance. Further investigation and experimentation could reveal the optimal conditions for this approach. To enhance the methodological rigor of future studies, direct data from full-scale plants and more standardized modelling techniques should be utilized.

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7. Appendix

7.1. Appendix A: Ecoinvent data set

Table 39: Background data taken from Ecoinvent v3.8 (2021).

Related process of:	Used in the LCA for:	Name in Ecoinvent v3.8 (2021)
Electricity	all scenarios as electricity	market for electricity, medium voltage [NL]
Polymer	belt filter press, DAF unit, external sludge treatment	market for polyacrylamide [GLO]
$FeCl_3$	DAF unit	market for iron (III) chloride, without water, in 40% solution state [GLO]
$FeCl_2$	sulphide binding in EGSB	market for iron (II) chloride [GLO]
NaOH	pH control, membrane cleaning	market for sodium hydroxide, without water, in 50% solution state [GLO]
H_2SO_4	pH control	market for sulfuric acid [RER]
Urea	N dosing	market for urea [RER]
Citric acid	membrane cleaning	market for citric acid [GLO]
NaOCl	membrane cleaning	market for sodium hypochlorite, without water, in 15% solution state [RER]
Concrete	infrastructure material	market for concrete block [DE]
Reinforcing steel	infrastructure material	market for reinforcing steel [GLO]
Stainless steel	infrastructure material	market for steel, chromium steel 18/8 [GLO]
Iron	infrastructure material	market for cast iron [GLO]
HDPE	infrastructure material	market for polyethylene, high density, granulate, recycled [Europe without Switzerland]
PP	infrastructure material	market for polypropylene, granulate [GLO]
PVC	infrastructure material	market for polyvinylidenchloride, granulate [RER]
GRP	infrastructure material	market for glass fibre reinforced plastic, polyester resin, hand lay-up [GLO]
Glass	infrastructure material	market for flat glass, uncoated [RER]
Sand-lime bricks	infrastructure material	market for sand-lime brick [GLO]
Transport truck	transport (chemicals, sludge, materials)	transport, freight, lorry 16-32 metric ton, EURO5 [RER]
Mineral N fertilizer	biosolids credits (sludge)	market for inorganic nitrogen fertiliser, as N [NL]
Mineral P fertilizer	P dosing, biosolids credits (sludge)	market for inorganic phosphorus fertiliser, as P2O5 [NL]
Heat	credits from biogas (EGSB)	heat production, natural gas, at industrial furnace >100kW [Europe without Switzerland]
Process water	credits from water reuse (root IFAS-NF)	tap water production, underground water without treatment [Europe without Switzerland]
Tap water	spoiling water, alternative credits from water reuse, chemicals	market for tap water [Europe without Switzerland]
adipic acid	secondary ingredient of the polymer	market for adipic acid [GLO]

7.2. Appendix B: Mass balance of reference scenarios

Flow sheet and mass balance of the reference scenarios. Illustrated by Figure 44 (SBR-scenario) and Figure 45 (p.126) (EGSB-SBR-scenario).

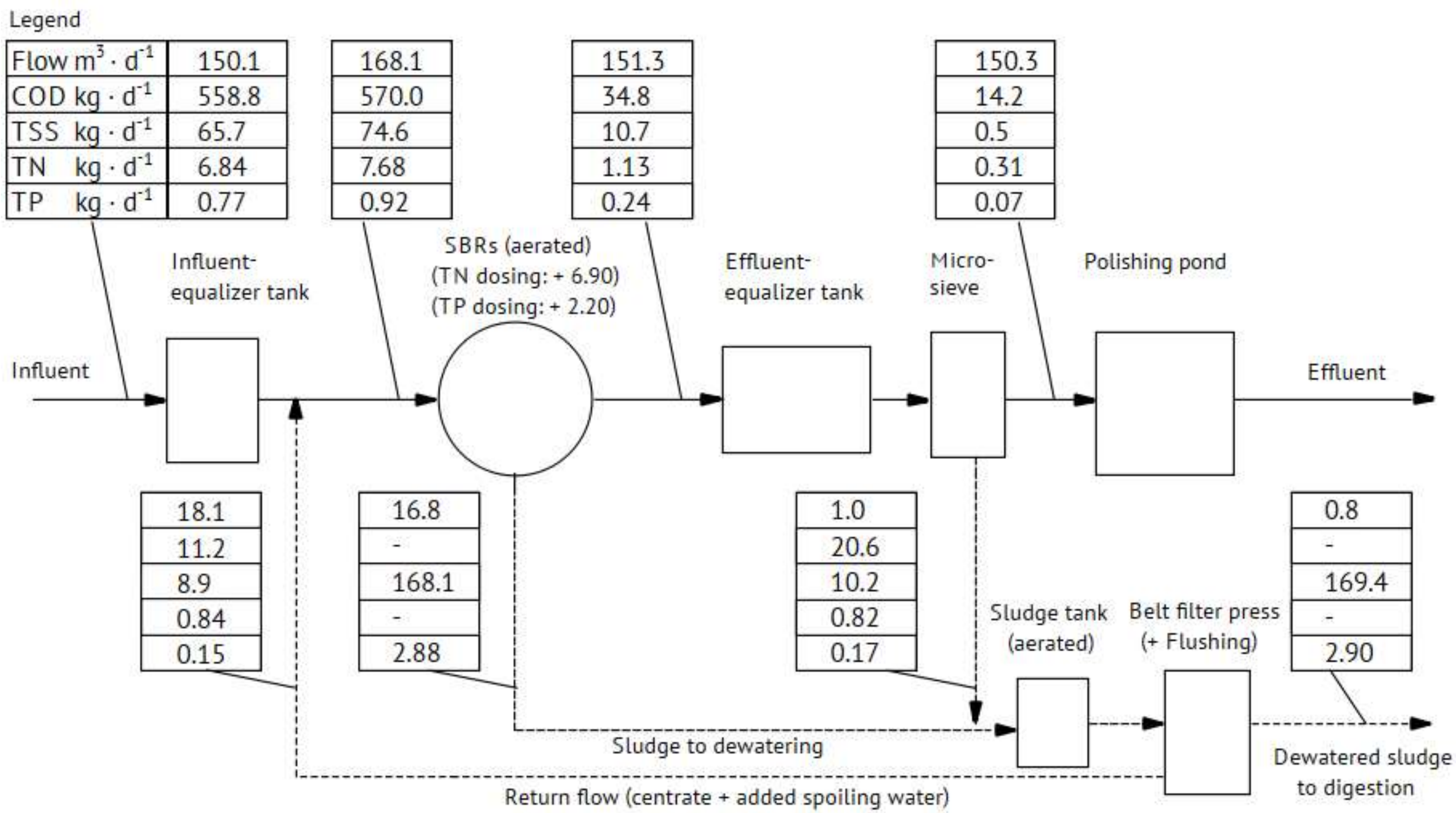

Figure 44: Mass balance of the SBR-scenario (own illustration).

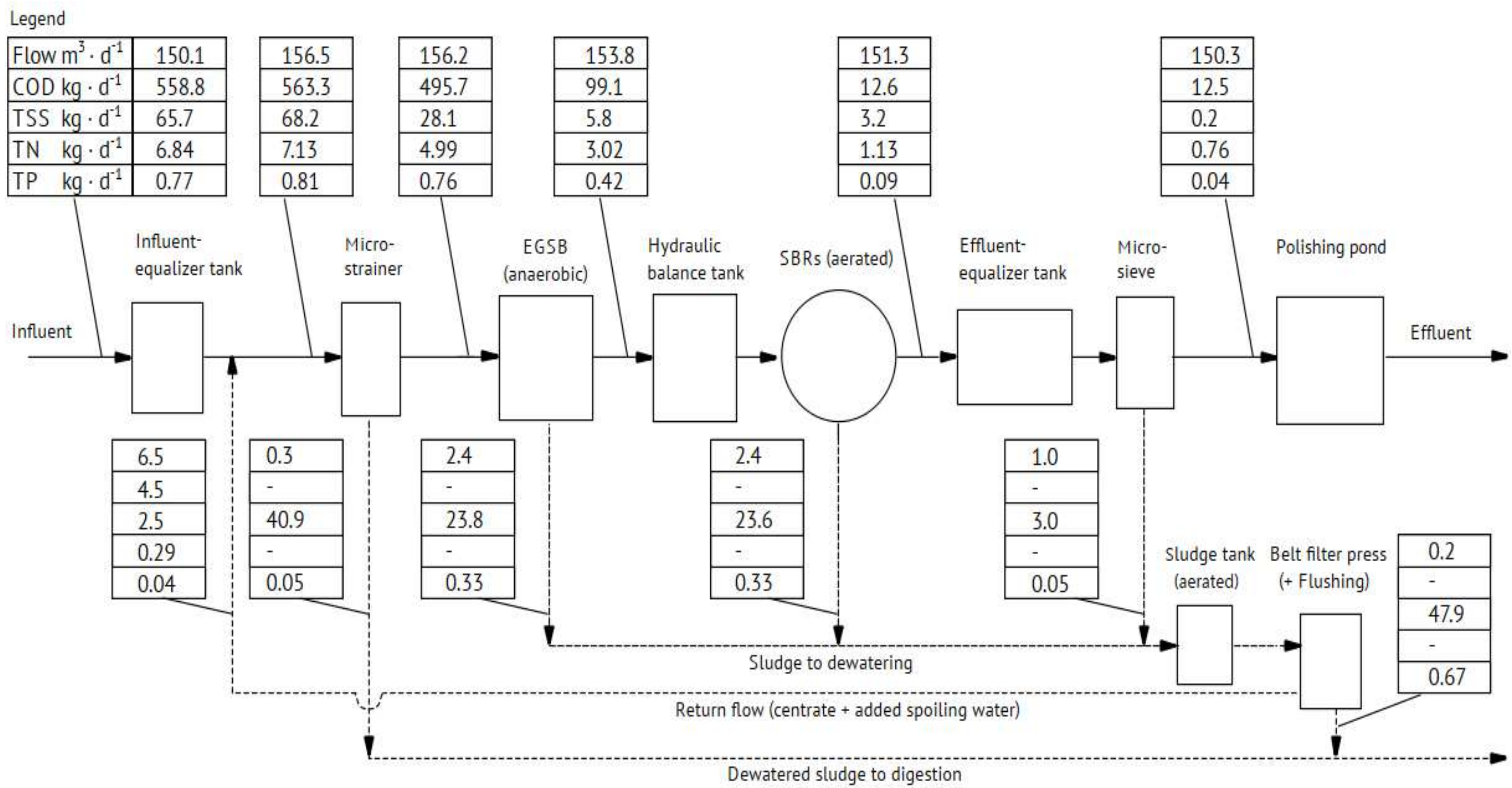

Figure 45: Mass balance of the EGSB-SBR scenario (own illustration).