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Comparative Environmental and Economic Assessment of the Zahle Wastewater Treatment Plant in Lebanon under Different Scenarios

By

Michelle Leba Ghosn

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THESIS APPROVAL FORM

Student Name: Michelle Ghosn I.D. #: 201407844

Thesis Title : Comparative Environmental and Economic Assessment of the Zahle Wastewater Treatment Plant in Lebanon under Different Scenarios

Program: Master of Science in Civil and Environmental Engineering

Department: Civil Engineering

School: Engineering

The undersigned certify that they have examined the final electronic copy of this thesis and approved it in Partial Fulfillment of the requirements for the degree of:

Master of Science in the major of Civil and Environmental Engineering

Thesis Advisor's Name Jean G. Chatila	Signature [redacted]	DATE: 27 / 09 / 2022 (Day / Month / Year)
Committee Member's Name Mahmoud Wazne	Signature	DATE: 27 / 09 / 2022 (Day / Month / Year)
Committee Member's Name Moustapha Harb	Signature	DATE: 27 / 9 / 2022 (Day / Month / Year)

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DEDICATION

To my loving family

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This work would not have been possible without the support of many people. Many thanks to my advisor, Dr. Jean Chatila, for providing guidance and feedback throughout this research. I want also to express my gratitude to Dr. Mahmoud Wazne and Dr. Moustapha Harb for serving on my thesis committee and sharing their valuable knowledge.

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Michelle Leba Ghosn

ABSTRACT

Assessing the environmental impacts of wastewater treatment plants (WWTPs) is a growing concern that needs to be addressed. Although several studies have been published on the environmental performance of WWTPs, few have been reported from developing countries, especially in Lebanon. In this study, the environmental and economical performance of an oxidation ditch-based secondary treatment technology in Zahle, Lebanon, was evaluated using life cycle assessment (LCA) and life cycle cost (LCC) in conjunction with computer modeling. The current plant was calibrated and validated using GPS-X v. 8.0 software and three hypothetical scenarios were suggested and compared with its current state. The first scenario (S_1) included adding anaerobic sludge digestion. The second scenario (S_2) included using extended aeration. The third scenario (S_3) included using a five-stage Bardenpho (FSB) process. For this purpose, a series of LCAs were performed using SimaPro 9.3.0.3 together with the ecoinvent 3.8 database, and the ReCiPe midpoint (H) and endpoint (H/A) methodologies. A functional unit of 1 m^3 of treated wastewater was used as a basis. The WWTPs costs were estimated using the CapdetWorks v4.0 software. Further, LCA was monetized using external costs. The analysis revealed that the environmental categories were primarily influenced by energy consumption. The current plant had a global warming effect of $0.678\ \mathrm{kgCO_2\ eq/m^3}$. Normalized results showed that the first scenario was the most environmentally friendly alternative in the human carcinogenic toxicity impact category quantified as follows: $\mathrm{S_2} < \mathrm{S_0} < \mathrm{S_3} < \mathrm{S_1}$. Normalized results also showed that freshwater ecotoxicity and human carcinogenic toxicity were the main impact categories. The economic evaluation revealed that the addition of anaerobic digestion led to more expenses, while $\mathrm{S_0}$ process was found to be the most cost-effective. The descending order concerning environmental costs was as follows: $\mathrm{S_2}(0.171\$/\mathrm{m^3}) > \mathrm{S_0}(0.159\$/\mathrm{m^3}) > \mathrm{S_3}\ (0.147\$/\mathrm{m^3}) > \mathrm{S_1}\ (0.117\$/\mathrm{m^3})$. Considering the collective evaluation, scenario $\mathrm{S_1}$ was chosen as a potential enhancement for the Zahle WWTP (ZWWTP). This study emphasized the importance of the environmental impact assessment in the wastewater treatment (WWT) sector and contributed to the growing body of literature related to WWT in Lebanon. Further research is needed to provide a clear data collection process that will increase the modeling precision of an existing plant and accurately reflect the outcomes.

Keywords: Life Cycle Assessment, Wastewater Treatment, Life Cycle Cost, SimaPro, Environmental cost, GPS-X, Model calibration and validation, CapdetWorks.

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LIST OF ABBREVIATIONS AND SYMBOLS

$	United States Dollar
%	Percentage
b_H	Maximum endogenous respiration rate of heterotrophic biomass
$K_{NH,A}$	Half-saturation constant for ammonium fraction of soluble total Kjeldahl nitrogen
L_0	Methane generation potential
m^2	Squared meter
m^3	Cubic meter
m^3/d	Cubic meter per day
$m^3 year/Kg$	Cubic meter-years per kilogram
N_2O	Nitrous oxide
O_2	Oxygen
X_S/X_I	Slowly biodegradable organic compounds to inert particulate organic compounds ratio
$Y_{low\ PP}$	Fraction of phosphorus stored in releasable poly-phosphorus form
$Y_{P,acetic}$	Amount of phosphorus released for 1 milligram of acetate sequestered in the form of poly-β-hydroxyalkonates
μ_A	Maximum aerobic growth rate of autotrophic biomass
€	Euro
°C	Degree Celcius
$1,4-DCB$	1,4-Dichlorobenzene
A^2O	Anaerobic-anoxic-oxic
AD	Anaerobic digestion
AE	Aquatic ecosystems
Al	Aluminum
ASM	Activated sludge model
Avg	Average
BOD_5	Five-day biochemical oxygen demand
BOD	Biochemical oxygen demand
BT	Biological treatment
CH_4	Methane
CO_2	Carbon dioxide
CAS	Conventional activated sludge
CAS-N	Conventional activated sludge with pre denitrification
CC	Chemical consumption
CDR	Council for Development and Reconstruction
CFC	Chlorofluorocarbons
CFR	Code of Federal Regulations
COD	Chemical oxygen demand
CRF	Capital recovery factor
CS	Contact stabilization
CTUe	Comparative toxic unit for ecosystems
Cu	Copper

d	Day
DALY	Disability-adjusted life year
DO	Dissolved oxygen
DOC	Degradable organic carbon
EA	Extended aeration
ED	Effluent discharge
EDL	Electricity of Lebanon
EMSEL	Environmental management and systems engineering lab
EPA	Environmental protection agency
eq	Equivalent
F/M	Food to Microorganism Ratio
FE	Freshwater eutrophication
FEC	Freshwater ecotoxicity
FECO	Freshwater ecosystems
FPMF	Fine particulate matter formation
FRS	Fossil resources scarcity
FSB	5-stage Bardenpho
FU	Functional unit
g	Gram
GHG	Greenhouse gas
GLO	Global
GW	Global warming
GWP	Global warming potential
h	Hour
HCT	Human carcinogenic toxicity
HH	Human health
HNCT	Human non-carcinogenic toxicity
hrs	Hours
HRT	Hydraulic retention time
i	Interest rate
IFAS	Integrated fixed-film activated sludge
ILCD	International Reference Life Cycle Data System
IPCC	Intergovernmental Panel on Climate Change
IR	Ionizing radiation
ISO	International Organization for Standardization
k	Decay rate
$\mathrm{kBq\ Co-60\ eq}$	Kilobecquerel Cobalt-60
kg	Kilogram
kWh	Kilowatt-hour
L	Liter
LandGEM	Landfill gas emissions model
LB	Lebanon
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory

LCIA	Life cycle impact assessment
LU	Land use
m	Meter
m^2a crop eq	Square meter-year annual crop equivalents, wherein “a” is an abbreviation of ‘annum’
MBBR	Moving bed bioreactor
MBR	Membrane bioreactor
ME	Marine eutrophication
MEC	Marine ecotoxicity
mg/L	Milligram per liter
MLSS	Mixed liquor suspended solids
MPN/100 mL	Most probable number per 100 milliliters
MRS	Mineral resource scarcity
n	Number of years of operation
NH_3	Ammonia
NH_4^+	Ammonium
$NO_2 - N$	Nitrite nitrogen
$NO_3 - N$	Nitrate nitrogen
NO_x	Nitrogen oxides
NCF	Annual net cash flow
NPV	Net present value
O&M	Operation and maintenance
OF-HH	Ozone formation human health
OF-TE	Ozone formation terrestrial ecosystems
OUR	Oxygen uptake rate
P	Phosphorus
PAF	Potentially affected fraction
PAOs	Phosphorus accumulating organisms
PC	Power consumption
pH	Potential of hydrogen
PM2.5	Particulate matter with diameters of 2.5 micrometers or less
ppb	Parts per billion
Ppm. h	Part per million-hours
PV	Photovoltaic
r	Interest rate
RAS	Returned activated sludge
RoW	Rest-of-World
RSM	Response surface methodology
RTI	Research triangle international
SO_2	Sulfur Dioxide
SO_x	Sulfur oxides
S0	Scenario 0/ Baseline scenario
S1	Scenario 1
S2	Scenario 2
S3	Scenario 3

SA	Step aeration
SETAC	Society of environmental toxicology and chemistry
SL	Sludge landfilling
SOD	Stratospheric ozone depletion
SRT	Solids retention time
t	The year in the NPV equation
TA	Terrestrial acidification
TC	Total cost
TE	Terrestrial ecotoxicity
TECO	Terrestrial ecosystems
TIC	Theil inequality coefficient
TKN	Total Kjeldahl Nitrogen
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
TVM	Time value of money
US	United States
USA	United States of America
USD	United States dollar
USD2013	United States dollar reference year 2013
UV	Ultraviolet
v	Version
VSL	Volumetric solids loading
VSR	Volatile solids reduction
VSS/TSS	Volatile suspended solids to total suspended solids ratio
WAS	Waste activated sludge
WC	water consumption
WWT	Wastewater treatment
WWTP	Wastewater treatment plant
yr	Year
ZWWTP	Zahle wastewater treatment plant

Chapter One

Introduction

The increase in public awareness about environmental issues, particularly climate change and global warming, has driven global interest in sustainable practices. Numerous aspects of human activity are causing damage to the environment every day including resource depletion, destruction of ecosystems, and overpopulation. According to recent figures, the total worldwide greenhouse gas (GHG) emissions for 2018 was $55.6\ GtCO_2$ eq, when accounting for emissions related to land-use change (Olivier & Peters, 2020). Water pollution, which is mostly brought on by the discharge of industrial waste and sewage from cities, is undoubtedly one of the most urgent worldwide environmental concerns. Therefore, WWT processes are essential to purify wastewater prior to discharge and turn it into usable water, reducing the risks to human health and ecology. In principle, WWT is crucial to prevent eutrophication and ecotoxicity by removing anthropogenic accumulation from water such as nitrogen, phosphorus, heavy metals, and complex organic compounds (Z. Yang et al., 2021). However, the significance of wastewater treatment plants is no longer restricted to the effluent quality but also to their overall impacts. Although in theory WWT prevents environmental pollutants, however, it might also generate negative impacts on the environment (Moussavi et al., 2021) resulting from energy consumption, sludge landfilling, and emissions to the atmosphere. In particular, WWTPs use a lot of energy (Zou et al., 2018), and release direct GHG emissions including CH_4 and N_2O which have a significant impact on global warming (Baresel et al., 2022). According to estimates, the global contribution of sewage treatment facilities to GHG emissions is around 3%

(Mika, 2022). With stricter discharge criteria being imposed, it may be more accurate to see WWTPs as a source of pollution rather than a barrier to emissions (Hao et al., 2019). As the selection of process designs and operating procedures is critical in reducing the amount of GHG released, many efforts have been made to optimize these technologies. As a result, thorough environmental studies are necessary to include the complete life cycle and all potential effects brought on. Several evaluation methodologies have been proposed to assess the overall environmental implications of WWT. In particular, life cycle assessment techniques are used to quantify and analyze the possible environmental effects of WWTPs throughout their full life cycle, covering all inputs and outputs from cradle to grave. LCA is a system of environmental impact assessment methods in which resource consumption and data emissions are identified, quantified, and analyzed, to assess the environmental impacts of a product over its lifetime. Additionally, LCA studies are typically coupled with economic factors accounted for across a system's life cycle using the life cycle costing (LCC) approach that estimates the cost of the plant and its operation. To evaluate and understand the mechanisms occurring in treatment trains, mathematical modeling of WWT processes are established. Computer model simulation software packages are utilized to produce data, such as GHG emissions, that is challenging to get from actual WWTPs (Daskiran et al., 2022). In the past years, the assessment of WWT facilities has been widely done to examine their environmental effects using LCA methodology and assist in their development and optimization. However, this is not the case in developing countries, where WWT performance is rarely taken into account. Some of the encounters that WWT faces in developing countries include inadequate design and operation of facilities, low effluent quality, a lack of supportive policies, and a lack of continuous power for operation. The decisions about wastewater projects in developing countries are normally more affected by economic factors without considering their environmental impacts, given that the design meets local regulations (Awad et al., 2019). Despite the fact that current literature has given some insight into the environmental outlines of WWT facilities (Moussavi et al., 2021), plants in Lebanon were not explicitly studied. This study aimed to evaluate the performance and ecological impact of the constructed WWTP facility in Zahle, Lebanon, using environmental and economical assessments and compare it to three hypothetical scenarios with different WWT processes. To accomplish this, LCA and LCC analyses were carried out and the total impacts and costs were studied. The findings of this research will provide insights into the environmental hotspots that may be used to design improved WWT systems, assist decision-makers in making more informed decisions on their design, and deliver better environmental perceptions.

Chapter Two

Literature Review

2.1 Computer simulation of WWTPs

Computer simulations have become an essential component of the design and operation of WWTPs. Their use allows for the design and analysis of new WWTPs as well as the reproduction of the performance of existing ones without interfering with the actual operation. Numerous scenarios and a wide range of technological solutions can be tested in a brief amount of time and at a low cost under different conditions, assisting in the discovery of the best solution in the design or operational processes. There are several mathematical models available now that represent WWTPs in various ways. Most of the modeling software now in use comprises activated sludge models (ASMs) or models based on ASMs, including BioWin, GPS-X, and others (Mu’azu et al., 2020). In particular, GPS-X is one of the most common simulation software containing a comprehensive model library that includes biological, chemical, and physical equations (Daskiran et al., 2022). When assessing the performance of already-existing plants, it must be verified that the developed computer model appropriately reflects the physical plant by calibrating and validating the created model using actual on-site data. Model calibration process involves setting up a (roughly) comparable model on the computer of the WWTP under investigation by fitting a set of information into the software layout. Different calibration techniques may be used, which mostly depend on the modeling objectives and the study's overall purpose, requiring either more or fewer steps (Petersen et al., 2002). Analyzing the literature, it is seen that there is no standardized published calibration procedure, however, many studies provided guidelines that will give a familiar transfer of knowledge. For instance, Petersen et al., (2002), Langergraber et al., (2004), and Fall et al., (2011) have all developed various calibration procedures. But at the same time, these procedures were not explicitly explained, making it difficult to find a thorough calibration case study in the open literature that can be simply replicated. Specifically, Petersen et al., (2002) defined a systematic model calibration procedure that was commonly referred to in recent studies. They evaluated the process for a municipal-industrial WWTP and assessed the influence of the model parameters. They established a set of information and steps needed to calibrate the model. The information included capacity volumes, physical properties, flow rates, etc. Their scheme of the model calibration steps was divided into two categories: one related to the procurement of the data characterization, and the other related to the different levels of calibration. Despite this, to the author's knowledge, none of the procedures described in the literature were clear to be strictly followed. However, several studies have raised new calibration patterns while referencing the previous ones. For instance, Makinia et al., (2005) developed a new procedure that synthesized components of the various guidelines. They aimed to obtain a validated model to estimate the maximum peak flow for WWTPs during stormwater conditions. Specifically, to calibrate the solids retention time (SRT) and nitrification process, they selected two groups of relevant variables: the first group included the slowly biodegradable organic compounds to inert particulate organic compounds ratio ($\frac{X_S}{X_I}$) and the maximum endogenous respiration rate of heterotrophic biomass (b_H), while the second group contained the maximum aerobic growth rate of autotrophic biomass (μ_A) and the half-saturation constant for the ammonium fraction of soluble total Kjeldahl nitrogen ($K_{NH,A}$). In this context, determining parameter subsets and selecting the right ones for a specific system proved to be not only critical but challenging due to the complexity of the models and the number and interdependency of integrated model parameters (Mu'azu et al., 2020). Accordingly, the selection of the parameters with the greatest influence on the process is possible through sensitivity analysis. To best describe the behavior of the systems, authors studied the identification of the most influential parameters for model calibration using sensitivity analysis. Liwarska-Bizukojc & Biernacki, (2010) analyzed the most significant kinetic and stoichiometric variables to validate the predictability of the ASM used in the BioWin software. They found that 17 kinetic and stoichiometric parameters are regarded as significant and half of them were associated with growth and decay of phosphorus-accumulating organisms (PAOs) such as the amount of phosphorus released for 1 milligram of acetate sequestered in the form of poly-β-hydroxyalkonates ($Y_{P/acetic}$) and the fraction of phosphorus stored in releasable poly-phosphorus form (Y_{lowPP}). Additionally, Mu'azu et al., (2020) determined the most important and relevant parameters for the treatment performance and capacity analysis of an activated sludge process using GPS-X. They found that the waste activated sludge (WAS) has more impact on the effluent quality compared to the returned activated sludge (RAS). They also found that higher hydraulic retention time (HRT) improved the chemical oxygen demand (COD) and the total nitrogen (TN) removal while a lower ratio of volatile suspended solids to total suspended solids (VSS/TSS) was not beneficial for the removal of nitrogen and carbonaceous matter. Particularly, GPS-X software has been frequently used in WWTPs studies to simulate different scenarios and enhance operating performances. For example, Cao et al., (2021) used GPS-X integrated with response surface methodology (RSM) to optimize the operational performance of a full-scale WWTP and specifically enhance nitrogen removal. They examined various TN removal scenarios and conducted the sensitivity of 53 kinetic and 8 stoichiometric parameters in the GPS-X dynamic simulations. They assessed the effect of different aeration conditions, SRT, and internal recycle ratio, which were considered critical variables in the denitrification rate and nitrogen removal efficacy. Their analysis revealed that the denitrification rate was associated with the SRT and the dissolved oxygen (DO) concentrations in several biological compartments. Precisely, nitrogen removal was significantly enhanced by lowering the DO while slightly raising the SRT. In light of this, intermittent aeration with a DO aeration controller was adopted in many WWTP designs to serve as an aeration time for nitrification to occur and low to no oxygen time for denitrification to occur. Hanhan et al., (2011) provided a comprehensive evaluation of the intermittent aeration-activated sludge process's mechanism and design for nitrogen removal. The total cycle time, aerated fraction, and the cycle time ratio were determined as the main design parameters. Their simulation results showed that by lowering the dissolved oxygen set point to $0.5\ mg/L$, which is a level that encourages simultaneous nitrification and denitrification, the effluent total nitrogen could be reduced. The concept of simultaneous nitrification was studied by Rittmann & Langeland, (1985), who documented the results of the process in single-channel oxidation ditches. They reported successful one-reactor denitrification with nitrification in an oxidation ditch which was easy to run, and oxygen transfer rate as the key controlling factor with average values of 0.1 to 0.5 mg O_2/L being effective.

On the other hand, Ahn et al., (2014) assessed the optimal operating parameters of a modified four-stage Bardenpho process using the GPS-X ASM1 model and multiple RSMs for the modeling approach. They calibrated the parameters for microbial respiration and effluent concentration using the Environmental Management and Systems Engineering Lab (EMSEL) technique. They found that using this protocol and under the optimized conditions, the differences in COD and TN concentrations between the calibrated and experimental data were reduced, however, the concentrations were still slightly different. Furthermore, Drewnowski et al., (2018) used GPS-X to predict energy output considering three different scenarios in which different variables were modified. The first scenario revealed that a 40% increase in suspended solids removal led to a 25% increase in biogas volume. The second scenario showed that increasing the RAS led to more energy consumption. Specifically, an increase of 400% in the RAS led to a 300% increase in energy usage compared to the reference level. And the third scenario revealed that oxygen concentrations play an important role in energy usage as low DO concentrations are associated with low amounts of blower energy usage while high DO concentrations require more energy to be delivered. As a result, it can be inferred that proper calibration and validation are important for the prediction and optimization of the model's effluent quality, energy consumption, and ultimately environmental impacts.

2.2 Life Cycle Assessment

In general, the interaction between the system and the environment is measured through an LCA. It is a comprehensive methodology that evaluates the environmental impacts of a process or activity throughout the course of its entire life cycle. It is a four-step procedure that includes the aim and scope, inventory, impact assessment, and interpretation, which are based on a detailed analysis of the system's whole input and output (ISO, 2006). Each of the four main processes in LCA is described here using the International Organization for Standardization (ISO) 14040 standards. The LCA stages are related to one another, making LCA an iterative evaluation tool. Given that LCA is not an absolute tool, it may be supplemented with many other impact assessment techniques to provide a thorough environmental study. Thus, methods and modes are flexible and employed according to the functional unit of the plant, the goal, and the scope of the research.

2.2.1 Goal and scope definition: In this stage, the main framework of the study is determined. It can include the goal of the research, the information needed, impact categories, the functional unit, the level of detail, and the system boundary. A system boundary defines the units of the plant that needs to be included in the analysis. For instance, the biological treatment process may be included while excluding electricity consumption. LCA can consider any phase of the WWTP's life, which can be divided into three: constructional phase, operational phase, and end-life phase (Corominas et al., 2020).

2.2.2 Inventory analysis: The life cycle inventory (LCI) stage entails compiling corresponding data relevant to the intended study's objectives. These data depend on the impact categories selected, serving as a foundation for the analysis's input.

2.2.3 Impact assessment: The life cycle impact assessment (LCIA) stage associates each parameter in the LCI analysis with the corresponding environmental impact categories (Corominas et al., 2020). The relevant impact categories are first identified before relating them to raw data which will be transformed later on into impact scale values. The parameters will be converted into an indicator that represents an index or a certain value indicating how much it has an impact on that specific category. Additionally, each pollutant can be attributed to one or more different impact categories. For instance, according to Corominas et al., (2020), phosphorus emissions solely have an impact on eutrophication but nitrogen emissions have an impact on climate change, eutrophication, acidification, and others. Midpoint and endpoint indicators are two categories utilized to measure a substance's environmental impact. Midpoint indicators, which refer to particular environmental issues like ozone depletion and global warming, are often more accurate in their estimations. Endpoint indicators' results are more uncertain since they are more standardized and generalized (ISO, 2006). ReCiPe, TRACI, and CML are three common LCIA methodologies that have been studied in the literature. ReCiPe is typically utilized for midpoint and endpoint indicators while TRACI is solely used for midpoint analysis. CML is mostly used to compare simulated results with those that have already been published (Corominas et al., 2020).

2.2.4 Interpretation: In this stage, data from the impact assessment and inventory stages are combined to give conclusions and recommendations based on the stated purpose and scope. The system is analyzed over time, and conclusions are formed on how to lessen the impacts and use it more effectively (ISO, 2006).

Software for LCA are accessible in several forms such as openLCA, SimaPro, and GaBi. In particular, SimaPro is commonly used in the literature and it is managed by PRé-Consultants in the Netherlands. According to Herrmann & Moltesen, (2015), SimaPro is a world's leading LCA software tool. The SimaPro software includes access to various databases, the most popular of which is the ecoinvent database containing over 18,000 life cycle inventory datasets. Among the several assessment approaches available, ReCiPe is a mutual midpoint and endpoint levels methodology that converts LCI data into environmental impact scores and presents results in 18 midpoint categories and three damage categories (Huijbregts et al., 2017). Huijbregts et al., (2017) and PRé Sustainability, (2020) describe each impact category and its associated indicators as summarized in Table 2-1. A flowchart of the damage pathways of the impact categories considered in ReCiPe is provided in Figure 2-1.

Table 2-1: A description of the midpoint impact categories and their associated indicators

Midpoint impact category	Indicator	Description
Global warming	Increase in infrared radiative forcing	Measures the integrated infrared radiative forcing increase of greenhouse gas (GHG)
Stratospheric ozone depletion	Decrease of stratospheric ozone	Refer to a time-integrated decrease in stratospheric ozone concentration over an infinite time horizon
Ionizing radiation	Increase in absorbed dose	Determined from the collective dose arising from the emission of a radionuclide
Ozone formation, Human health	Increase in tropospheric ozone population intake	Refer to the change in ambient ozone concentration following the emission of a precursor
Fine particulate matter formation	PM2.5 population intake	Estimates the change in ambient particulate matter 2.5 (PM2.5) concentration following the emission of a precursor
Ozone formation, Terrestrial ecosystems	Increase in tropospheric ozone	Relates to the sum of the variations between the hourly mean ozone concentration and 40 ppb during daylight hours over the relevant growing season in $ppm \cdot h$
Terrestrial acidification	Increase of proton in natural soils	Measures the change in acid deposition following changes in air emissions and changes in soil acidity
Freshwater eutrophication	Increase of phosphorus in freshwater	Measures phosphorus emissions to freshwater and agricultural soils

Table 2-1: A description of the midpoint impact categories and their associated indicators *(continued)*

Marine eutrophication	Increase in emissions of nitrogen-containing nutrients	Indicates how much of the released nitrogen-containing nutrients make it to the marine end compartment
Terrestrial ecotoxicity	Hazard-weighted increase in natural soils	Refers to impacts of toxic substances on terrestrial ecosystems
Freshwater ecotoxicity	Hazard-weighted increase in freshwaters	Refers to the comparative toxic unit for ecosystems (CTUe) expressing an estimate of the potentially affected fraction of species (PAF) integrated over time and volume per unit mass of a chemical emitted (PAF $m^3 year/kg$)
Marine ecotoxicity	Hazard-weighted increase in marine water	Refers to impacts of toxic substances on marine ecosystems
Human carcinogenic toxicity	Risk increase of cancer disease incidence	Concerns effects of carcinogenic toxic substances impact on the human environment
Human non-carcinogenic toxicity	Risk increase of non-cancer disease incidence	Concerns effects of non-carcinogenic toxic substances impact on the human environment
Land use	Occupation and time-integrated land transformation	Refers to the relative species loss caused by a specific land use type
Mineral resource scarcity	Increase of ore extracted	Refers to the primary extraction of a mineral resource
Fossil resource scarcity	Upper heating value	Refers to fossil resource consumption
Water consumption	An increase in water consumed	Refers to the amount of freshwater consumed

Note: Adapted from "ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level", by M. A. J. Huijbregts, Z. J. N. Steinmann, P. M. F. Elshout, G. Stam, F. Verones, M. Vieira, M. Zijp, A. Hollander, R. van Zelm, 2017, International Journal of Life Cycle Assessment, 22(2), p. 141 (https://doi.org/10.1007/s11367-016-1246-y). Copyright 2016 by Springer-Verlag Berlin Heidelberg.

LCA is widely used to quantify the environmental impacts of WWT facilities and related processes and to serve as a useful decision-support tool to assess the environmental viability of new technologies and capture trade-offs across different categories of environmental concern (Corominas et al., 2020). LCA is employed for different purposes and assists researchers in making decisions about design optimization and environmental sustainability. Since many emissions are released from the WWT and associated operations that influence the environment's health in a way, reducing these detrimental impacts is the focus and primary objective of many studies.

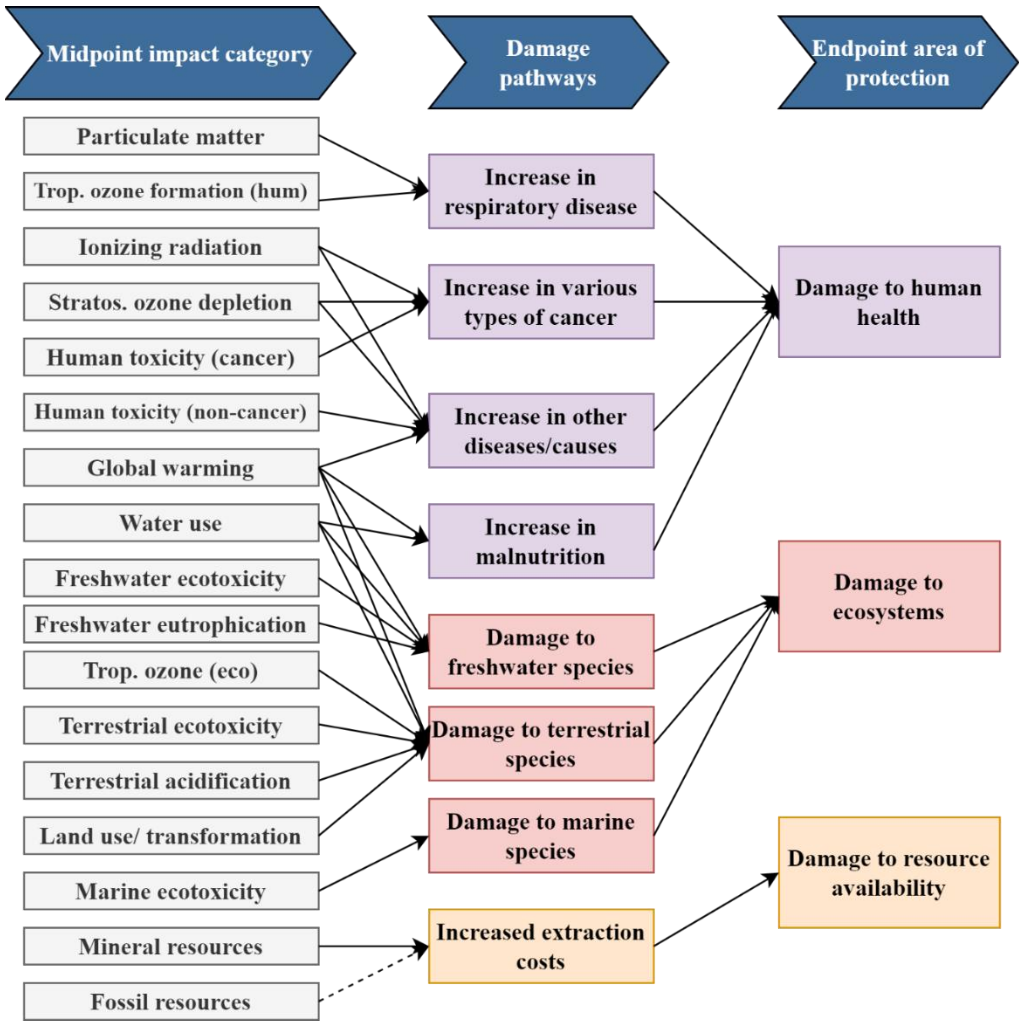


Figure 2-1: Impact categories covered in the ReCiPe 2016 method and their relation to the areas of protection. The dotted line indicates there is no constant mid-to-endpoint factor for fossil resources

Note: Reprinted from "ReCiPe2016: a harmonized life cycle impact assessment method at midpoint and endpoint level", by M. A. J. Huijbregts, Z. J. N. Steinmann, P. M. F. Elshout, G. Stam, F. Verones, M. Vieira, M. Zijp, A. Hollander, R. van Zelm, 2017, International Journal of Life Cycle Assessment, 22(2), p. 140 (https://doi.org/10.1007/s11367-016-1246-y). Copyright 2016 by Springer-Verlag Berlin Heidelberg.

For instance, Pradel & Aissani, (2019) applied LCA to determine whether recovering dissipated phosphorus by producing sludge-based phosphate fertilizer can be an effective way to reduce its depletion. In their study, they intended to find a way of supply of the non-renewable element of phosphorus that is necessary for life by comparing four scenarios of phosphorus recovery from wastewater through producing sludge-based phosphate fertilizer from a French WWTP to a mineral phosphate fertilizer used as a reference scenario. Each scenario applied a different phosphorus recovery method, such as biological acidification, phosphorus crystallization, and others. They assessed the scenarios using GaBi software and the CML-IA midpoint characterization method approach, which examined many impact categories, including acidification, photochemical oxidation, and mineral resource depletion. Their analysis indicated that sludge-based phosphate fertilizers had greater impacts on the environment than mineral phosphate due to the low quantity of phosphorus recovered and low phosphorus content of the sludge which took substantial amounts of energy and reactants for recovery. On the other hand, Flores et al., (2019) carried out an LCA to evaluate the environmental performance of constructed wetlands for winery wastewater treatment by comparing six scenarios including constructed wetlands and the common treatment methods implemented in different wineries located in South-Western Europe namely third-party management and activated sludge systems. They used SimaPro software and the ReCiPe midpoint method. Some of the impact categories that were addressed comprise climate change, marine eutrophication, and terrestrial ecotoxicity. Their results revealed that the constructed wetland scenarios were the most environmentally friendly alternatives. Another function of the LCA method was predicting the feasibility of established technologies as demonstrated by Roman & Brennan, (2021). In their study, they evaluated the concept of Eco-Machines™, or ecologically designed wastewater systems intended to have a similar treatment efficiency as conventional treatment, however, with less energy and chemical consumption. Roman & Brennan, (2021) utilized a pilot-scale Eco-Machine™ at the Pennsylvania state university campus as a base model of the study. They carried out a series of LCA to evaluate the environmental impacts of different scenarios. The LCIA method used was Impact2002+ in SimaPro software associated with the ecoinvent 3.6 databases. Fifteen impact categories were considered including climate change, human health, and respiratory organics. Among the conclusions drawn, their analysis revealed that when placed in a warm climate, Eco-Machines™ that don't need a greenhouse or additional heating, used about one-third of the energy needed and released less GHG emissions compared to conventional treatment, as well as benefiting resources, ecosystem, and human health.

A common thing among LCA research on environmental burdens related to WWT is that most of the studies identified energy consumption as the greatest contributor to environmental impacts such as in Phelan et al., (2014), Moussavi et al., (2021), and Daskiran et al., (2022). In an effort to reduce energy usage for the sustainability of WWT systems, Daskiran et al., (2022) evaluated a biological nutrient removal facility using mathematical modeling, LCA, and three scenarios that would lower the facility's net power usage. They created three scenarios by combining various values for DO concentration, SRT, and internal recirculation parameters. Contrary to the typical perception of reduction, their study revealed that while the quantity of energy saved was constant, changes in SRT, DO concentration, and internal recirculation had different effects on the plant's environmental behavior. For instance, their analysis showed that low DO concentration had a negative influence on global warming due to the rise of N_2O emissions.

DO concentrations were found to have a critical role in effluent quality and energy consumption (Drewnowski et al., 2018; Cao et al., 2021; Daskiran et al., 2022). In particular, Cao et al., (2021) found that nitrogen removal was significantly increased by lowering the DO. Additionally, as cited in Daskiran et al., (2022), the study conducted by Flores-Alsina et al., (2010) revealed that there were few positive environmental effects from increasing nitrification to lowering the quantity of nitrogen in the effluent. Abiotic depletion, climate change, photochemical oxidation, and acidification potential environmental impacts have been seen to rise in the study since high aeration energy was needed to induce nitrification. Besides, it was discovered that adding a DO or oxygen uptake rate (OUR) controller to the treatment system helped in reducing the negative effects of aeration on the environment. In this context, and since energy consumption had been frequently reported as the greatest contributor to environmental impacts, studies refocused their environmental assessments on suggesting methods and technologies for producing biogas and using renewable energy sources in WWT. For instance, Bravo & Ferrer, (2011) focused on climate change and depletion of abiotic resources categories since they relate to energy-related issues. They assessed the environmental performance of a large WWTP in the Barcelona Metropolitan Area and identified the processes having the most negative effects on the environment by carrying out an LCA and using the CML 2 baseline method. Their analysis found that the WWTP's environmental performance can be enhanced by incorporating sludge treatment solutions that could increase biogas generation.

Thus, LCA technique has been often used in WWT assessments and has aided in their understanding and advancement. However, the use of the LCA tool in developing countries has just recently begun examining the environmental effects of WWT technologies (Gallego-Schmid & Tarpani, 2019). In this regard, Awad et al., (2019) studied an already-existing WWTP located in Gamasa, Egypt, and employed LCA to evaluate whether to add tertiary treatment, anaerobic digestion, or both, using the CML2000 baseline method for the determination of seven different impact categories. They found that adding both tertiary treatment and anaerobic digestion had the largest environmental advantages across all categories due to energy savings and the possibility of water reuse. Furthermore, Mamathoni & Harding, (2021) analyzed the environmental effects of the sequential batch reactor and extended activated sludge process treatment methods in South Africa. They employed the ReCiPe midpoint (H) technique in LCA and used the ecoinvent 3.6 database in SimaPro software. According to their standardized results, the extended activated sludge procedure had the greatest impact on all categories. Additionally, they found that the sequential batch reactor was the preferable option and that South Africa's energy mix significantly affects both processes and the majority of the life cycle effect was brought on by the need for power production.

In contrast, when it comes to decision-making, using LCA alone to assess a system is not sufficient. Life cycle impacts are typically coupled with economic analyses to determine the most economical and environmentally effective product. Additionally, some studies incorporated social costs referred to as environmental costs. Social costs of carbon evaluate the expenses borne by society from the total carbon dioxide equivalent footprint (Harclerode et al., 2020). One way of determining environmental costs is by monetizing LCA data by translating the physical environmental impacts into monetary values (Canaj et al., 2021). Normally, literature findings on social costs support the conclusion that improvements to WWT technologies can result in operations that are less environmentally harmful and thus have lesser external costs (Harclerode et al., 2020). On the other hand, studies that included economic analyses, such as the one conducted by Foglia et al., (2021), generally found that improving environmental performance resulted in higher economic costs. This is not always the case, though; according to Awad et al., (2019), adding a tertiary treatment in WWTPs in developing countries resulted in increased economic profits.

2.3 Life Cycle Cost

From an economic point of view, cost estimation is critical during the evaluation of treatment processes as it may affect their viability. WWTPs' costs are comprised of costs incurred during plant construction referred to as capital costs and operation and maintenance (O & M) costs required to keep the facility running (Arif et al., 2020). Cost estimation commonly applied in WWT literature is known as life cycle costing, designed to help decision-makers evaluate the economic profits and losses related to the system. Three types of LCC were distinguished by The Society of Environmental Toxicology and Chemistry (SETAC): conventional, environmental, and societal LCC. Generally, LCC is termed conventional LCC when it only considers internal costs and includes the time value of money (TVM). This is done by deducting the upcoming costs when assessing the current cost. In contrast, environmental LCC is when cost estimation is associated with LCA, and it considers evident costs and environmental assessment costs of the system on the agenda. Societal LCC is when societal factors are considered, and the financial value is estimated on a social basis. A survey of the literature showed that 83% of the case studies used conventional LCC, 14.6% used environmental LCC, and 2.4 % used societal LCC (Ilyas et al., 2021).

Many WWTPs cost estimation methods were found in the literature. Some analyses used equations while others used software estimation tools. Specifically, CapdetWorks software is usually adopted to estimate WWTPs costs as it is easy to use and provides results divided between project cost, maintenance, operation, materials, etc. It uses the influent characteristics and process design parameters for a quick and accurate WWTPs cost estimation (Nowrouzi et al., 2021).

For instance, Arif et al., (2020) identified the most cost-effective treatment among three WWT facilities using CapdetWorks software. They compared conventional activated sludge (CAS) without denitrification, CAS with pre-denitrification (CAS-N), and membrane bioreactor (MBR). Their economical evaluation included capital, operation, maintenance, material, chemical, and energy costs. In the case of the framework, their results revealed that over 39 years, the CAS had the lowest removal efficiency and the cheapest cost while the MBR has the highest removal efficiency and highest cost. As a result, it can be inferred that lower costs are associated with lower removal efficiency. In another comparable economical study, Abbasi et al., (2021) compared the effluent quality and cost of three facilities using CAS, contact stabilization (CS), and step aeration (SA). They used the CapdetWorks software to calculate the total project costs, including those for implementation, maintenance, and energy use, and found that contact stabilization was the most economical.

Similarly, Nowrouzi et al., (2021) used CapdetWorks to analyze the economic performance of four WWTPs designs consisting of anaerobic/anoxic/oxic (A^2O), membrane bioreactor (MBR), moving bed bioreactor (MBBR), and integrated fixed-film activated sludge (IFAS). They noted that the biological units account for the majority of expenses in the design of WWTPs. For each alternative, energy and material consumption costs changed. The flow rate, COD, and biochemical oxygen demand (BOD) were found to have a significant effect on the total cost (TC). The present worth and TC showed the greatest sensitivity to changes in construction costs. Their results revealed that concerning the total cost, MBBR was the most cost-effective configuration while in terms of energy and material use, the A^2O and MBR designs were rated as the most and least cost-effective systems, respectively. On the other hand, Lin et al., (2016) performed the cost analysis by combining equations of net present worth and cost with the CapdetWorks database and literature data to emphasize the design of nitrogen recovery systems. They assessed three different nitrogen removal and recovery methods integrated into WWT systems: nitrification-denitrification, anammox, and the anaerobic ion exchange route. Their results suggested that ion exchange has a great deal of potential to attain high nitrogen removal and recovery efficiency and deliver economically and environmentally optimal performance.

2.4 Justification for the current study

Although literature has provided some understanding of the environmental and economic performance of WWTPs and related operations, developing countries still lack this knowledge. Despite the recent growing interest, there hasn't been in-depth research analyzing the environmental impacts of WWTPs in Lebanon. In fact, environmental concerns in Lebanon are almost non-existent and their framework has been mostly neglected. This study aims to investigate the environmental and economic profiles of the

WWTP in Zahle, Lebanon, and suggest potential alternatives for it. It contributes to filling the gap of limited knowledge about the environmental and economic performance of operating WWTPs in Lebanon. To achieve this purpose, LCA and LCC methods are used. The objective is to provide a calibrated model of the ZWWTP as a basis to assess its costs and environmental impacts and compare it to three hypothetical treatment scenarios. The outcomes of this study can help future applications with potential system modifications that may reduce environmental damage.

Chapter Three

Materials and Methods

3.1 Description of the ZWWTP

The assessed WWTP is located in Zahle, Lebanon, and serves a population of more than 200,000 people with a site area of 17.5 acres. The inlet structure building consisted of an inlet channel, coarse screen, pumping station, fine screen, and grit and grease removal. Following preliminary treatment, wastewater passes through an anaerobic tank and intermittently aerated ditch. Chemical precipitation of phosphorus was required to reach the concentration of 1 mg P/L in the effluent. The precipitant is added to the mixed liquor in the intermittent aeration tank and flocs are settled out in the clarifier. A Filtrazur™ type filter is then utilized with a final ultraviolet (UV) disinfection tank for tertiary treatment. In contrast, surplus sludge is routed separately to a thickening unit, after which it is dewatered and subsequently disposed of. The plant head receives the filtrate that has accumulated in the sludge treatment section. Table 3-1 lists the physical characteristics of the biological treatment tanks at the plant. The assumption for the pumped flows between thickening and dewatering was made on the basis that mechanically dewatered sludge included $12-25\%$ solids (Guangyin & Youcai, 2017).

Table 3-1: The physical properties of the biological WWT process

Parameter	Unit	Anaerobic	Oxidation ditch	Secondary clarifier
Surface area	m^2	−	2798	3902
Diameter	m	−	−	46
Depth	m	7.7	6.97	4
Volume	m^3	3400	19,500	−

Channel length	m	−	46.8	−
Width	m	−	37	−
Length	m	−	83.8	−

The average inflow of wastewater was $25{,}081\ \mathrm{m^3/d}$ whereas the maximum flow reached $27{,}540\ \mathrm{m^3/d}$. The total wastewater to be treated during this month was $752{,}440\ \mathrm{m^3}$and the total water cleared as effluent was $667{,}950\ \mathrm{m^3}$ directly discharged to the Litani river. The pH varied within permissible ranges with 6.2 as the lowest value and 7.29 as the high. The inflow water's temperature ranged from 14°C to 23°C. The effluent water discharge standards are provided in Table 3-2.

Table 3-2: Effluent water discharge standards

Parameter	Unit	Concentration limit
COD	mg/L	50
BOD_5	mg/L	15
TSS	mg/L	10
TN	mg/L	10
TP	mg/L	1
Total coliforms	MPN/100 mL	< 100

3.2 GPS-X Modeling Approach

To model and simulate the WWTP, GPS-X v.8.0 simulation software of Hydromantis Inc. was employed. GPS-X helps in improving design quality and operational effectiveness when designing a new facility or simulating an existing one. GPS-X Mantis3 library was used as it has a GHG emission estimation tool (Daskiran et al., 2022). In particular, GHG emissions were divided into scopes 1, 2, and 3. Scope 1 included process emissions such as CO_2 released from anaerobic, anoxic, and aerobic biological processes, N_2O released from nitrification/ denitrification, and CH_4 released from anaerobic processes. Scope 2 comprised energy emissions from power used by pumps and air blowers while scope 3 included material emissions from chemical usage, transportation, and other materials. Only the direct GHG emissions of CO_2, N_2O, and CH_4 from the biological treatment operations were collected from GPS-X; all other emissions were accounted for by other means. Figure 3-1 depicts the ZWWTP model's simulated layout on GPS-X. After the model was built, the physical properties of each tank unit were inputted. The influent was characterized using the bodbased model. The plant was running partially due to the reduced inlet flow rate during the studied month; thus, one treatment train was considered.

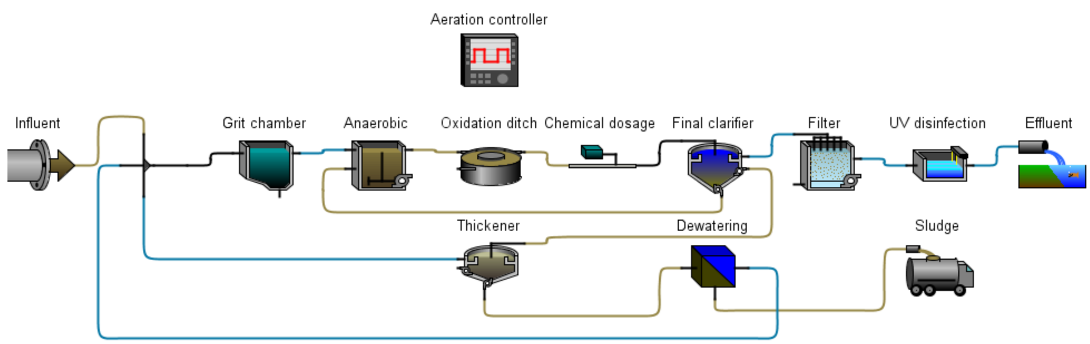


Figure 3-1:Schematic layout of the ZWWTP created in GPS-X software

3.3 Calibration and validation data

The input data for the simulations were obtained from the Council for Development and Reconstruction (CDR) and comprised the months of June 2020 and October 2020. A summary of the average wastewater characterization values used for model calibration and validation is presented in Table 3-3 and Table 3-4, respectively. The input data measurements included daily data for certain parameters and monthly data for others. Both the model calibration and model validation simulations were run under steady-state settings. Data from June 2020 were utilized for calibration whereas data from October 2020 were used for validation. It is preferable to utilize validation data from a time that is obviously different from the calibration one (Hulsbeek et al., 2002). Total COD and BOD were included in the comparison of daily simulation data, while the remaining effluent concentrations were compared with average values.

Table 3-3: The average concentrations in the influent and effluent in June 2020

Parameter	Unit	Influent	Effluent
pH	$-$	7.7	7.2
COD	mg/L	468	17.9
BOD_5	mg/L	164	6.5
TSS	mg/L	307	3.4
TKN	mg/L	33.1	2.1
NH_4^+ & NH_3	mg/L	17.1	0.4
$NO_3 - N$	mg/L	0.5	1.12
$NO_2 - N$	mg/L	0.062	0.015
TN	mg/L	33.7	3.24
TP	mg/L	4.18	0.2

Table 3-4: The parameter concentrations in the influent and effluent in October 2020

Parameter	Unit	Influent	Effluent
pH	$-$	7.5	7.15
COD	mg/L	907	26.2
BOD_5	mg/L	307	10.5
TSS	mg/L	799	4.5
TKN	mg/L	53.7	2.84
NH_4^+ & NH_3	mg/L	40.4	0.48
$NO_3 - N$	mg/L	0.55	1.1
$NO_2 - N$	mg/L	0.153	0.04
TN	mg/L	54.4	3.95
TP	mg/L	8.76	0.86

The influent flow was characterized by inputting the plant influent quality data, such as the VSS/TSS ratio, BOD/COD, pH, etc. into the GPS-X influent advisor. The default model set of parameters was employed for unknown values. In order to balance the model's predictability for all factors, it was necessary to identify the most significant parameters by carrying out a sensitivity analysis on some parameters for adjustment. Sensitive parameters and operating factors that impact the effluent concentration were identified by monitoring the changes of each parameter one at a time while leaving the others constant. The entire procedure is depicted in the flowchart in Figure 3-2. In terms of validation, data from October 2020 had an influent flow average of $15{,}027\ \mathrm{m^3/d}$ and a maximum flow of $16{,}940\ \mathrm{m^3/d}$. During this month the plant was running partially as well. The theil inequality coefficients (TICs) produced by GPS-X were taken into account to determine how well-fitted operational conditions of the model match.

3.4 Treatment scenarios and design parameters

Three scenarios were considered and assessed based on their effluent quality and environmental and economic performance. Each of these three alternatives was analyzed for 30 days and compared with the main model or baseline scenario S0. All scenarios were intended to treat the same wastewater influent and to fulfill the standards imposed for release into the Litani River.

The first scenario (S1) consisted of adding an anaerobic digestion (AD) tank to S0. Anaerobic digestion serves as a means for energy recovery, pollution reduction, and fertilizers production. It is used to biologically decompose organic waste and turn it into biogas. It also generates useful sludge that may be recycled for agricultural applications. The biogas generated is made up of $48-65\%$ of methane and is utilized to generate electricity. The sewage sludge resulting will have improved stability, low pathogens, low odor emissions, and small sludge dry matter, ensuing in a considerable reduction in final sludge volume (Hanum et al., 2019). The type of anaerobic digestion chosen was the high-rate digestion process designed according to the equations and calculations in Appendix B. The process design performance included the physical design of the anaerobic tank, the heat required for functioning, and the calculation of biogas production.

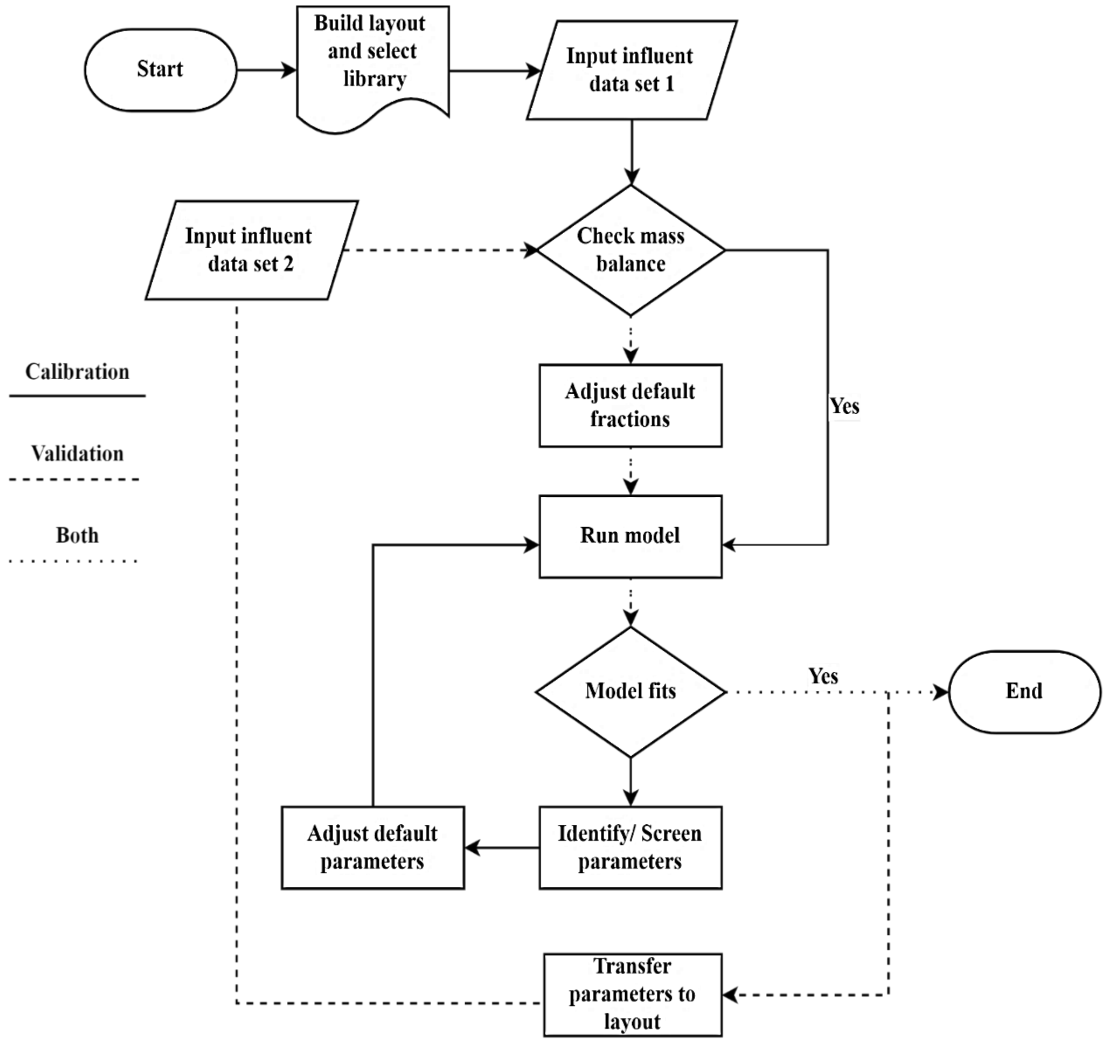


Figure 3-2: Process flowchart for calibrating and validating the GPS-X model

Note: Reprinted from "Systematic Modeling of Municipal Wastewater Activated Sludge Process and Treatment Plant Capacity Analysis Using GPS-X" by N.D. Mu'azu, O. Alagha, & I. Anil, 2020, *Sustainability*, *12*(19), p. 8 (https://doi.org/10.3390/su12198182).Copyright 2020 by the authors.

The basic performance factors included volatile solids reduction (VSR), volumetric solids loading (VSL), and SRT. The digestion period equal to SRT was assumed to be 20 days with the typical range of $15 - 20$ days for the high-rate digestion process (Qasim & Zhu, 2018). The digester conditions were to maintain the mesophilic operating temperature of $35°C$. It was assumed that the volume of the sludge entering the anaerobic digestion is equal to $150\ m^3/d$ and its COD content was calculated using mass balance. The amounts of heat required and biogas generated were calculated based on the equations in Appendix B (Qasim & Zhu, 2018). The biogas produced was used as a fuel source and converted to kWh. The potential estimation assumed an electrical conversion efficiency of 35% and that $1\ m^3$ of biogas will yield 2.14 kWh of electricity (Suhartini et al., 2019). It was assumed that all biogas generated is used in energy recovery and no leakage was happening. It is also assumed that all CH_4 in the biogas is transformed into CO_2 emitted into the air. The total biogas production was taken as a conservative estimate with excess biogas that may increase in warm weather, to use in other activities at the plant. Figure 3-3 shows the layout of the model on GPS-X.

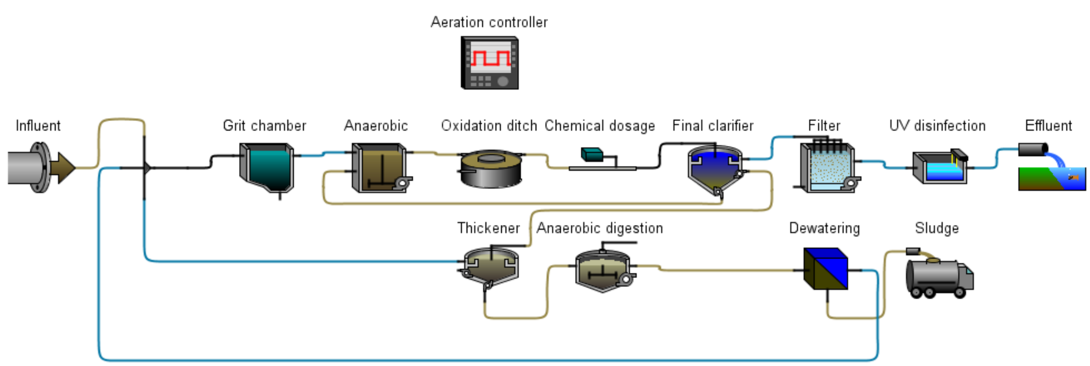


Figure 3-3: ZWWTP + anaerobic digestion (Scenario 1)

In the second scenario (S2), the biological treatment included anoxic and extended aeration (EA) tanks (Figure 3-4). In the EA process, the aeration is carried out over an extended period of time using a long HRT $(18\ \text{to}\ 24\ \text{hours})$ and a low food-to-microorganism (F/M) ratio (high SRT) which results in little sludge but higher oxygen uptake per kg of BOD removed. This method was most commonly used where ease of operation and low sludge generation are needed (Eckenfelder & Grau, 1998). The volume of the extended aeration tank was calculated to be 23,000 m^3 using the average monthly influent flow and an HRT of 22 hrs. The average sludge calculation criteria were according to the equations in Appendix C (Arceivala & Asolekar, 2012). The volume of the anoxic tank was 4180 m^3 computed using an HRT of 4 hrs (Brown et al., 2011; Qasim & Zhu, 2017). It was assumed that dewatering removes 95% of the water from the sludge to be landfilled.

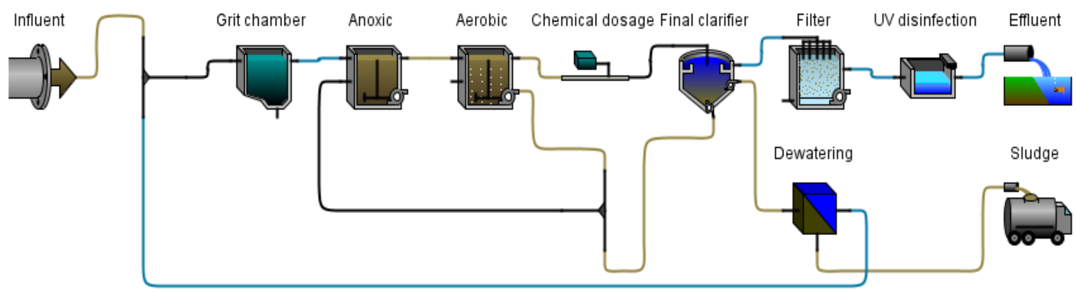


Figure 3-4: Extended aeration process (Scenario 2)

In the third scenario (S3), the biological treatment was substituted with the 5-stage Bardenpho process (FSB) (Figure 3-5). The modified Bardenpho process or FSB adds two reactors (anoxic and aerobic) to the A^2O process. The second anoxic reactor reduces nitrite and nitrate from the first aerobic reactor's effluent and their concentration recycled to the anaerobic reactor with RAS resulting in enhanced nitrogen and phosphorus removal. A second aerobic reactor is also used because the concentration of near zero in the effluent of the second anoxic reactor is likely damaging the sludge settling qualities. This second aerobic reactor's function was to supply oxygen, preventing the sludge from thickening up. The FSB also had an advantage by lowering TSS, COD, and BOD contents, however, a major drawback is the extremely large reactor volumes. The HRTs of the tanks in series were taken as 1.5 hrs, 3 hrs, 6 hrs, 3 hrs, and 1 hr, respectively (Demir, 2020). The tanks' volumes were orderly as follows: 1570 m^3, 3135 m^3, 6270 m^3, 3135 m^3, and 1000 m^3. In this scenario, there was no need for aluminum sulfate precipitation. Figure 3-5 shows the S3 process layout on GPS-X.

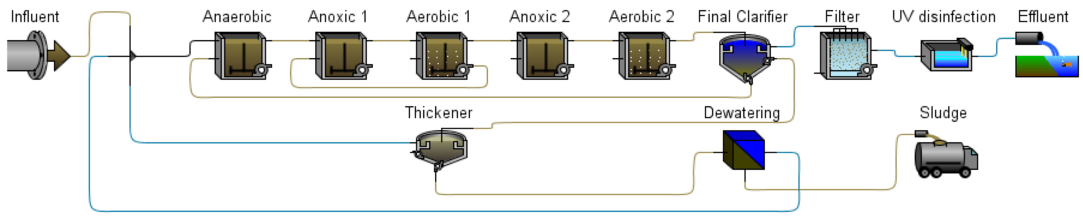


Figure 3-5: FSB process (Scenario 3)

3.5 LCA methodology

The LCA methodology was used to examine the environmental impacts of the WWTP systems following the standardized LCA scheme defined by ISO14040 (ISO, 2006). The steps of the LCA framework included the study's goal and scope, LCI, LCIA, and interpretation.

3.5.1 Goal and scope definition

The goal of the current study was to identify, assess, and compare the potential environmental impacts of the investigated ZWWTP throughout its operational phase to those generated by three suggested hypothetical scenarios.

3.5.1.1 Functional unit

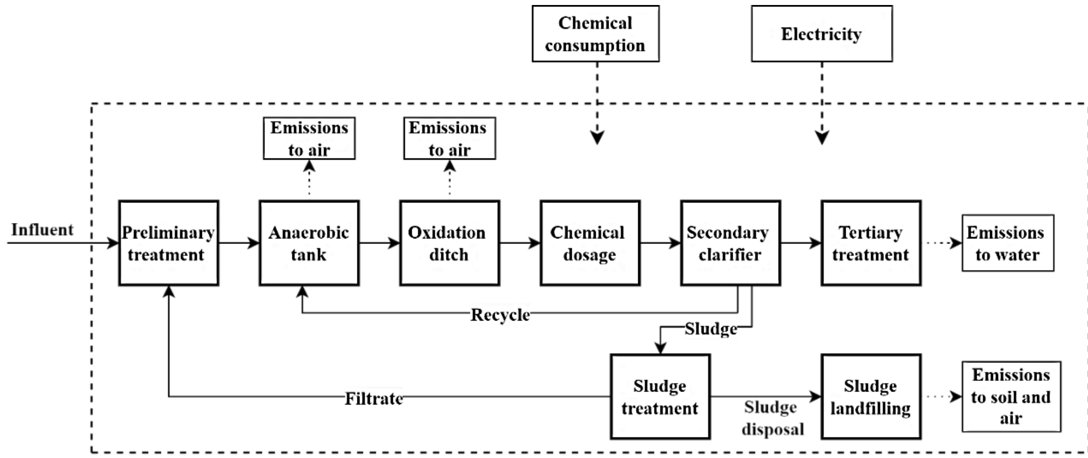
The functional unit (FU) applied to all scenarios was defined as 1 m^3 of treated wastewater to meet the study's objective. It served as the FU for comparing the effects of various situations and it is the most widely used FU in wastewater treatment LCA studies (Corominas et al., 2020).

3.5.1.2 System boundaries

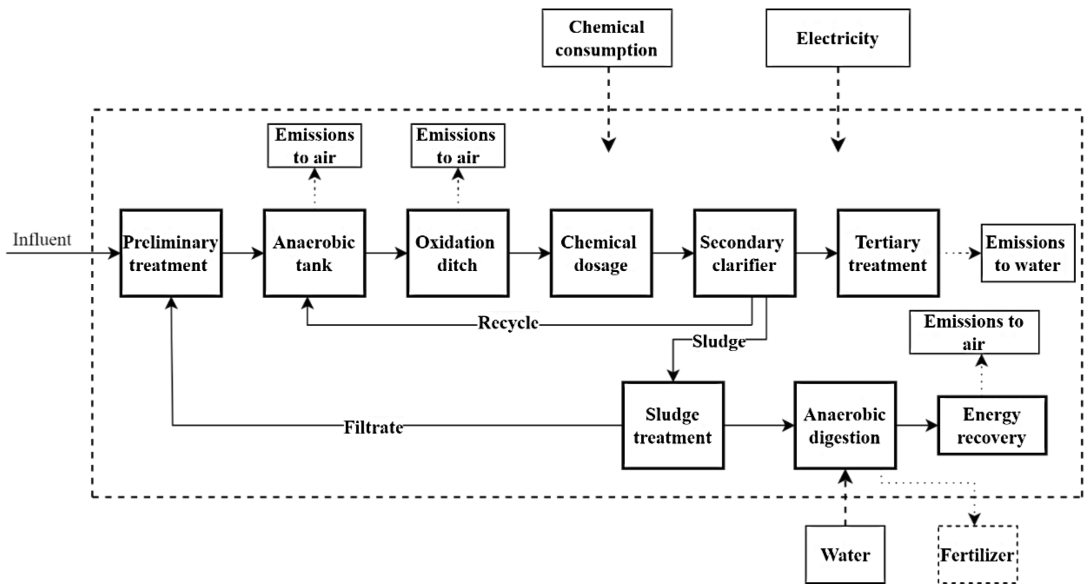
System boundaries included the WWTP's operation stage for the LCA analysis. For each scenario, the input flows of chemicals and energy resources and recovery were systematically studied. System boundaries also included direct emissions to the air such as GHG emissions from biological WWT and landfilling, direct water emissions, and direct soil emissions from heavy metals. The ZWWTP facility had three sources of energy: diesel generators, the electricity of Lebanon (EDL), and a Photovoltaic (PV) solar system. The power consumed was mostly supplied by EDL and it was considered the only electricity source since the PV system contributed only about 10% of the monthly consumption and no generator was used. The emissions from sludge landfilling and biogas power generation were also included. The transportation and spreading procedure of digested sludge (fertilizer) were disregarded, and the construction and demolishment phases were not considered. The employed system boundaries are shown in Figure 3-6.

3.5.2 Life cycle inventory

Inventory data for the scenarios under investigation are depicted in Table 3-5. The inventory input and output data were based on the Ecoinvent 3.8 databases for all scenarios. The system's input included the energy usage in kWh and materials consumed, while the outputs comprised the direct emissions to air, water, and soil. There were three types of emissions: scope 1 direct emissions, scope 2 indirect emissions, and scope 3 indirect emissions resulting from activities that the reporting company does not own or control (EPA, 2020).

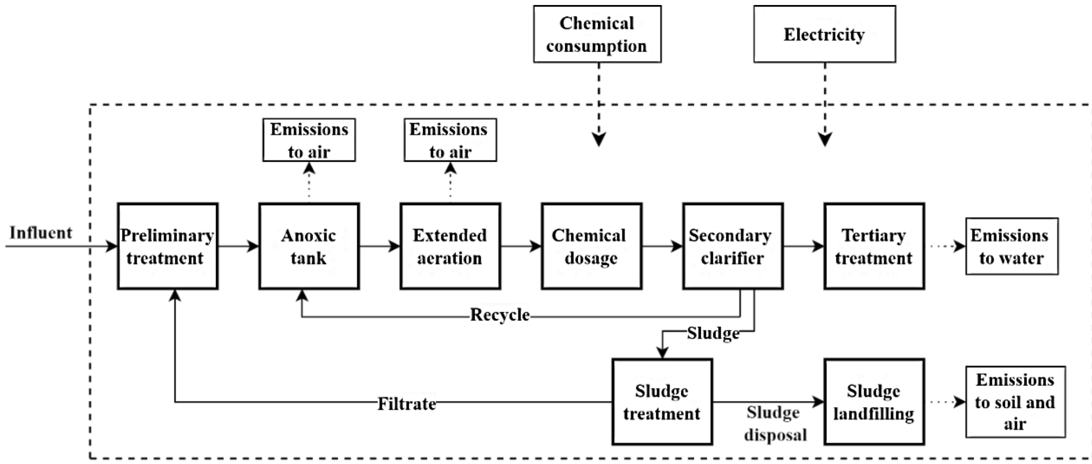


(a) Baseline scenario

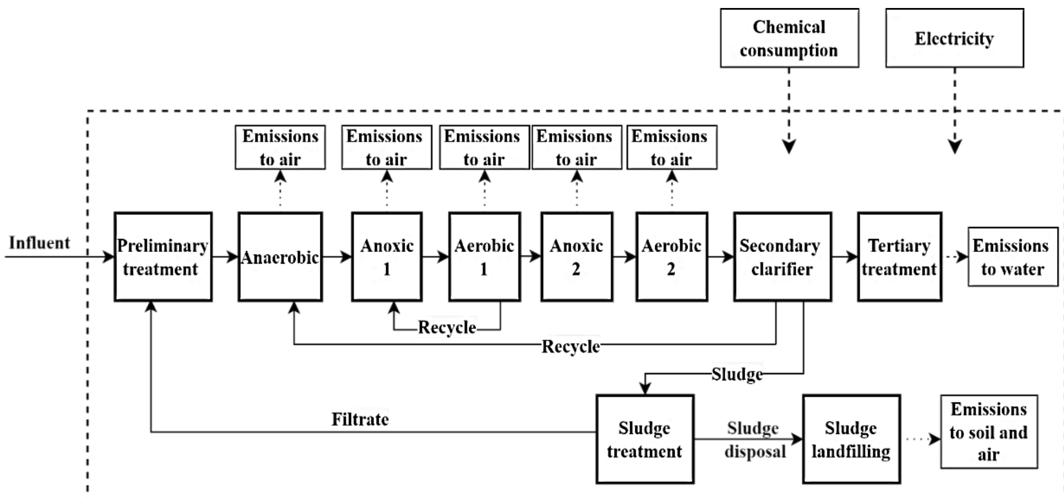


(b) Scenario 1

Figure 3-6: System boundaries of: a) Main model: ZWWTP, b) Scenario 1: ZWWTP + anaerobic digestion, c) Scenario 2: Extended aeration, d) Scenario 3: FSB



(c) Scenario 2



(d) Scenario 3

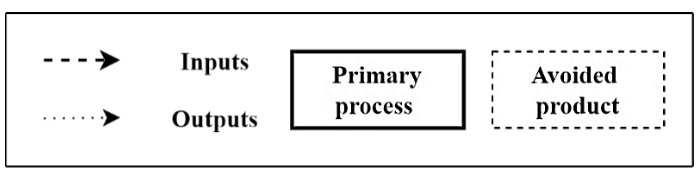


Figure 3-6: System boundaries of: a) Main model: ZWWTP, b) Scenario 1: ZWWTP + anaerobic digestion, c) Scenario 2: Extended aeration, d) Scenario 3: FSB *(continued)*

The WWTP's scope 1 impacts referred to wastewater discharge, WWT air pollutants, and sludge disposal emissions. Scope 2 emissions were associated with its power production consumption, and scope 3 accounted for the chemical demand. The Mantis3 carbon footprint library was employed to estimate the CO_2, CH_4, and N_2O greenhouse gas emissions emerging from the biological treatment. The N_2O model emission took into consideration the nitrification and denitrification activities of heterotrophic and autotrophic bacteria (Daskiran et al., 2022). The net GHG emissions were calculated by subtracting the GHG offsets that lie in the system from the total GHG emissions. The offsets of scope 1 considered were based on some of the organic material that is degrading and were of biogenic nature, not coming from fossil fuels. In the case of municipal WWTPs, 100% of the carbon coming into the plant was biogenic so all the CO_2 emissions that are coming out of the WWTP could be counted as an offset, not regarded as a GHG emission (IPCC, 2006). It is important to note that the CH_4 and N_2O emissions during the WWT were non-biogenic (Corominas et al., 2020). The 100-year global warming potentials (GWPs) used to derive the mass of methane and nitrous oxide emitted are shown in Appendix A, Table A-1 (RTI, 2010). All scenarios' effluents were eventually discharged to the Litani River, where they contributed to a variety of environmental problems such as freshwater eutrophication and marine eutrophication. With regards to eutrophication, the pollutants' direct water emissions involved BOD, COD, TSS, TN, and TP. Direct GHG emissions from sludge landfilling were evaluated for all scenarios using the Landfill gas emissions model (LandGEM) v3.03, 2020 Microsoft Excel™ (RTI, 2010). LandGEM computed both CH_4 and CO_2 emissions directly, simulating one form of waste at a time. The input needed included the start and closure year of the landfill, the CH_4 generation potential (L_0), the amount of sludge per year, the degradable organic carbon DOC, and the decay rate k. Detailed calculations are provided in Appendix A. Moreover, sludge landfilling emissions included the composting process impacts on the ecosystem from the heavy metals present in the sludge that pose a high risk of toxicity to the environment (Zhang et al., 2017). The CO_2 and CH_4 emissions released from landfilling in addition to the CO_2 emissions released from biogas combustion were considered biogenic (Bogner et al., 1997; IPCC, 2007; RTI, 2010; Paolini et al., 2018; Zhao et al., 2019). Since the sludge waste was probably landfilled nearby, the transportation fuel consumption was not accounted for. The overall anticipated electricity requirements were identified using mathematical simulation on GPS-X. The polymer was used as a flocculant for solid separation processes in thickening and dewatering (B. A. Bolto et al., 1996; Lichtfouse et al., 2019) while lime is for sludge stabilization (Jr et al., 2015; Jasim, 2020). The polymer was known to be cationic; polyacrylamide was assumed to be the type used (B. Bolto & Xie, 2019). It was assumed that the quantity of chemicals employed for sludge stabilization and flocculation had not changed as a result of the scenarios except in scenario 1 since sludge resulting from AD was stable and thus did not need lime addition (Radaideh et al., 2010; Hanum et al., 2019). Data were gathered as a raw quantity, then converted to mass, and finally to the FU. Particularly, in scenarios S0, S2, and S3, the electrical grid was used, whereas in scenario S1 the plant was operated using both the electrical grid and power generated from the anaerobic digestion of sludge. In this scenario (S1), the amount of kWh generated from recovery was subtracted from the total power consumption to calculate the plant's net energy consumption.

Table 3-5: Inventory data for the operation stage per FU: 1 m^3 of treated wastewater

Ecoinvent v3.8 database item	Unit	Scenarios			
		S0	S1	S2	S3
Inputs					
Aluminum sulfate, powder {RoW[a]}	kg/m^3	$5.00E-03$	$5.00E-03$	$5.00E-03$	$-$
Quicklime, milled, packed {RoW}	kg/m^3	$5.32E-03$	$-$	$5.32E-03$	$5.32E-03$
Polyacrylamide {GLO[b]}	kg/m^3	$4.65E-04$	$4.65E-04$	$4.65E-04$	$4.65E-04$
Electricity consumption					
Electricity, medium voltage, {LB[c]}	kWh/m^3	$2.46E-01$	$1.60E-01$	$2.50E-01$	$2.10E-01$
Outputs					
Emissions to water					
COD, LB	kg/m^3	$1.59E-02$	$1.59E-02$	$1.35E-02$	$1.9E-02$
BOD_5, LB	kg/m^3	$5.77E-03$	$5.77E-03$	$2.4E-03$	$2.7E-03$
Suspended solids, unspecified	kg/m^3	$3.00E-03$	$3.00E-03$	$2.13E-03$	$1.41E-03$
Nitrogen, LB	kg/m^3	$2.88E-03$	$2.88E-03$	$4.51E-03$	$2.8E-03$
Phosphorus, LB	kg/m^3	$1.77E-04$	$1.77E-04$	$1.27E-04$	$2.6E-04$
Emissions to air (biological WWT)					
Methane	kg/m^3	$3.93E-03$	$3.93E-03$	$0.00E+00$	$1.67E-09$
Dinitrogen monoxide	kg/m^3	$7.68E-04$	$7.68E-04$	$1.5E-03$	$1.26E-03$
Emissions to air (sludge landfilling)					
Carbon dioxide, biogenic	kg/m^3	$6.81E-03$	$-$	$4.14E-03$	$6.81E-03$
Methane, biogenic	kg/m^3	$2.00E-03$	$-$	$1.23E-03$	$2.00E-03$
Emissions to soil (sludge landfilling)					
Aluminium	kg/m^3	$2.54E-03$	$-$	$2.54E-03$	$2.54E-03$
Copper	kg/m^3	$6.00E-04$	$-$	$6.00E-04$	$6.00E-04$
Chromium	kg/m^3	$1.87E-04$	$-$	$1.87E-04$	$1.87E-04$
Lead	kg/m^3	$1.22E-04$	$-$	$1.22E-04$	$1.22E-04$
Nickel	kg/m^3	$4.36E-04$	$-$	$4.36E-04$	$4.36E-04$
Zinc	kg/m^3	$4.44E-04$	$-$	$4.44E-04$	$4.44E-04$
Sulfide	kg/m^3	$2.71E-04$	$-$	$2.71E-04$	$2.71E-04$
Emissions to air (biogas combustion)					
Carbon dioxide, biogenic	kg/m^3	$-$	$2.20E-01$	$-$	$-$
Water consumption					
Water, unspecified natural origin, LB	m^3/m^3[d]	$-$	$1.54E-02$	$-$	$-$

[a] RoW= Rest-of-World
[b] GLO= Global
[c] LB= Lebanon
[d] m^3 of water/ m^3 of treated water

Accordingly, this approach allocated the WWTP's environmental impacts without crediting the system with the avoided impacts. Electricity consumed for heating and stirring in the anaerobic digestion was estimated using the equations provided in

Appendix B (Qasim & Zhu, 2018). The water rate consumed for the hot water recirculation through the external heat exchanger was designed to be $387\ \mathrm{m^3/d}$ (or per FU $0.0154\ \mathrm{m^3 water/m^3 wastewater}$) (Qasim & Zhu, 2018). Direct emissions to air from energy recovery were calculated according to the combustion reaction for methane (Appendix B) and the CO_2 emissions were computed to be equal to $0.22\ \mathrm{kg/m^3}$. The process of biogas combustion has additional emissions such as nitrogen oxides (NO_x) and sulfur oxides (SO_x) (Pradel & Aissani, 2019), however only the CO_2 emissions are considered. In scenario 2 (S2), the sludge amount calculated was fit for direct dewatering, thus thickening was removed (Arceivala & Asolekar, 2012). Concerning S3, the sludge to be landfilled was assumed to be the same as the baseline scenario S0.

3.5.3 Life cycle impact assessment

The LCIA was conducted on SimaPro v.9.3.0.3 using the ReCiPe 2016 assessment technique with a hierarchal perspective using both midpoint (H) and endpoint (H/A) approaches. The ReCiPe assessment method was selected based on the references from the International Reference Life Cycle Data System (ILCD) for integrating midpoint and endpoint approaches in a consistent framework (JRC European Commission, 2010). ReCiPe was the most recent and popular technique used among LCA practitioners (Slorach et al., 2019). The endpoint level impact categories for damage assessment enclosed damage to human health, ecosystems, and resources. Within these endpoint categories, the midpoint level comprised 18 impact categories: global warming (GW) expressed in (kg CO_2eq), stratospheric ozone depletion (SOD) in (kg $\mathrm{CFC11\ eq}$), ionizing radiation (IR) in (kBq $\mathrm{Co-60\ eq}$), ozone formation human health (OF-HH) in (kg NO_x eq), ozone formation terrestrial ecosystems (OF-TE) in (kg NO_x eq), terrestrial acidification (TA) in (kg SO_2 eq), fine particulate matter formation (FPMF) in (kg PM2.5 eq), freshwater eutrophication (FE) in (kg P eq), marine eutrophication (ME) in (kg N eq), terrestrial ecotoxicity (TE) in (kg $1,4-\mathrm{DCB}$), freshwater ecotoxicity (FEC) in (kg $1,4-\mathrm{DCB}$), marine ecotoxicity (MEC) in (kg $1,4-\mathrm{DCB}$), human carcinogenic toxicity (HCT) in (kg $1,4-\mathrm{DCB}$), human non-carcinogenic toxicity (HNCT) in (kg $1,4-\mathrm{DCB}$), land use (LU) in ($\mathrm{m^2a\ crop\ eq}$), mineral resource scarcity (MRS) in (kg $\mathrm{Cu\ eq}$), fossil resource scarcity (FRS) in (kg oil eq), and water consumption (WC) in ($\mathrm{m^3}$). The human health endpoint category was expressed in "Disability-Adjusted Life Years" (DALY), assessed the danger of the illness, taking into consideration both morbidity and death. It is mostly influenced by respiratory effects brought on by inorganic pollutants released into the atmosphere. The influence on the ecosystem was denoted under the ecosystems damage category expressed in ($\mathrm{species.yr}$), while the total of non-renewable energy and mineral extraction midpoint categories is represented in the resources damage category measured in ($\mathrm{USD2013}$) (Roman & Brennan, 2021). This method allowed to give an inclusive analysis of the potential environmental damages because of the large number of impacts included (Mainardis et al., 2021). There was no cut-off of impacts and all contributions were reported. According to ISO, the impact categories are considered for the classification and characterization as mandatory steps, while the normalization and weighing procedures were optional due to potential bias and value choices they are associated with (Pizzol et al., 2017). In this study, both characterized and normalized impacts were generated at midpoint and endpoint levels to identify important impact categories and understand the meaning of the results.

3.6 LCC methodology

3.6.1 CapdetWorks

CapdetWorks v.4.0 simulation software package (Hydromantis Inc., Ontario, Canada) was used to model the conceptual design of WWTPs and estimate costs according to the Hydromantis 2014, (USA Avg) equipment costing database. The default costs were adjusted to reflect actual costs which were acquired based on personal communication with the ZWWTP consulting engineers. The approximated adjusted costs input parameters are shown in Table D-21. The software was run for each scenario with an average flow of $25{,}081\ \mathrm{m^3/d}$. The quantified design unit parameters were modified and the parameters that were not given or computed were kept as default. All scenarios were calibrated based on the units utilized in GPS-X. Costs generated for all scenarios included the present worth, project cost, operation, maintenance, material, chemical, energy, and amortization. The economic possibility calculations were conducted considering a 40-year operating life period (Nakatsuka et al., 2020). For the scenarios that include energy recovery, the net total costs were calculated by considering produced biogas as a profit. The construction cost of the biogas power plant was established based on the values from Mensah et al., (2021). The power plant was assumed to be designed with a maximum pipeline length of $500\ \mathrm{m}$ and the gasometer to hold 6.7% of the monthly biogas volume produced (with a period of two days for gas discharge) (Santos et al., 2016). The study considered one incinerator. Table 3-6 summarizes the considered equipment costs used in the biogas power plant. The entire annualized project and O&M costs are divided by the average yearly flow to get the unit $\mathrm{cost/m^3}$. To determine the annualized project cost, the project cost is multiplied by the capital recovery factor

(CRF) to convert the project cost to a series of equal annual cashflows. By definition, a CRF is a factor that divides the cost of the current project into a number of equal payments over the contemplated term (n) at an interest rate (i) (Waleed, 2007; Arif et al., 2020). The CRF and the net present value (NPV) were calculated using the following equations (Waleed, 2007; Carlini et al., 2017):

$$\mathrm{CRF} = \frac{i(1+i)^n}{(1+i)^n-1} \qquad \mathrm{NPV} = \sum_{t=0}^{n} \frac{\mathrm{NCF}_t}{(1+r)^t}$$

where NCF ($/year) is the annual net cash flow, n is the number of years of operation, t is the year, and r & i are the interest rate (%). An interest rate of 6% was used. As CapdetWorks did not have algorithms that provide the flexibility to estimate the effect of intermittent aeration on oxidation ditch performance, energy costs for all scenarios were obtained from GPS-X at a rate of 0.1$/kWh. Additionally, because the anaerobic tank cannot be displayed separately on CapdetWorks, this unit was left out of all scenarios for comparison purposes. Costs for the profit of energy recovery from biogas were computed with the 0.1$/kWh rate for preliminary costing. As only one line is operational in all scenarios, the costs were estimated with one line considered for simplification. The effects of price changes and inflation were not considered.

Table 3-6: Price of equipment used in the biogas power plant

Equipment	Cost in US $	Unit
Generator: Otto	510.8	USD/kW
Incinerator	102,159.27	USD/unit
Pipeline	127.70	USD/m
Gasometer	45.97	USD/m^3
Compressor	255.4	USD/m^3/h

Note: Reprinted from "Assessment of electricity generation from biogas in Benin from energy and economic viability perspectives" by J.H.R. Mensah, A.T.Y.L. Silva, I.F.S. Santos, N.S. Ribeiro, M.J. Gbedjinou, V.G. Nago, G.L. Filho, & R.M. Barros, 2021, *Renewable Energy, 163*, p. 618 (https://doi.org/10.1016/j renene.2020.09.014).Copyright 2020 by Elsevier Ltd.

3.6.2 Environmental costs

Environmental prices were indices that indicated the social marginal cost of avoiding environmental emissions. In other words, they showed how much people were willing to pay to avoid pollution and other negative impacts. As a result, environmental prices reflected the loss of welfare associated with the pollution released into the environment (De Bruyn et al., 2018). The pollution cost was calculated by putting a monetary value on the imposed environmental footprint for each impact category to provide a financial valuation to society of how much damage is being done, as a more intuitive and practical measure than (DALY) or $(\mathrm{species.yr})$. These costs were calculated for all scenarios using the environmental prices by Ponsioen et al., (2020). The monetization method was based on the ReCiPe 2016 impact categories which enclosed a list of factors for all substances covered by ReCiPe. The estimate for human health was $72{,}000\ €/\mathrm{DALY}$, while the estimate for ecosystems was $11.5 \times 10^6\ €/\mathrm{species.year}$, based on a land use value of $0.10\ €/\mathrm{m}^2.\mathrm{year}$ (Ponsioen et al.,2020; Canaj et al., 2021). The monetization factors derived based on these estimates are shown in Table 3-7. It is important to remember that these prices were not country-specific characterization factors and this study serves as a preliminary estimation. However, since the ReCiPe method was based on European averages, the usage of these costs in specific countries assumed that the pollutant's impact in relation to its midpoint environmental price was the same as in Europe (Ponsioen et al., 2020; De Bruyn et al., 2018). The final comparison was made by combining both CapdetWorks and environmental costs.

Table 3-7: Monetization values for the LCA-ReCiPe 2016 impact categories

Midpoint impact category	Unit	Monetization value (€)
Global warming	kg CO_2 eq	0.15
Stratospheric ozone depletion	kg CFC11 eq	38
Ionizing radiation	kBq Co − 60 eq	0.00061
Ozone formation, Human health	kg NOx eq	0.066
Fine particulate matter formation	kg PM2.5 eq	14
Ozone formation, Terrestrial ecosystems	kg NOx eq	0.0093
Terrestrial acidification	kg SO_2 eq	2.73
Freshwater eutrophication	kg P eq	2
Marine eutrophication	kg N eq	3.1
Terrestrial ecotoxicity	kg 1,4 − DCB	0.00013
Freshwater ecotoxicity	kg 1,4 − DCB	0.0080
Marine ecotoxicity	kg 1,4 − DCB	0.0012
Human carcinogenic toxicity	kg 1,4 − DCB	0.24
Human non-carcinogenic toxicity	kg 1,4 − DCB	0.016
Land use	m^2a crop eq	0.10
Mineral resource scarcity	kg Cu eq	0.20
Fossil resource scarcity	kg oil eq	0.39
Water consumption	m^3	0.045

Note: Adapted from "Life cycle-based evaluation of environmental impacts and external costs of treated wastewater reuse for irrigation: A case study in southern Italy" by K. Canaj, A. Mehmeti, D. Morrone, P. Toma, M. Todorović, 2021, *Journal of Cleaner Production*, *293*, p. 5 (https://doi.org/10.1016/j.jclepro.2021.126142).Copyright 2020 by Elsevier Ltd.

Chapter Four

Results and Discussion

4.1 Mathematical modeling

4.1.1 Model calibration and validation

The GPS-X model was calibrated using the dynamic data from the studied ZWWTP collected in June 2020. The main objective of the calibration was to obtain a GPS-X representation of the investigated ZWWTP that is generally equivalent in order to calculate the GHG emissions emitted during the biological treatment. Input data was verified in the influent advisor to enable the dynamic simulation. For an initial calibration, GPS-X default parameter settings were utilized, and the model was run on a 30-day simulation period. There were no notable fluctuations in the influent flow characteristics throughout this month, however, the initial calibration findings failed to capture acceptable effluent quality.

An aeration controller is introduced as intermittent aeration was implemented in the ZWWTP. Operating at a DO concentration of 2 mg O_2/L with intermittent aeration resulted in successful adjustment. When aeration was on, the dissolved oxygen setpoint in the oxidation ditch was considered to hold 2 mgO_2/L. It was important to keep in mind that DO concentrations beyond this value may prevent effective denitrification. Simulations showed that an increase of DO to 5 mgO_2/L results in an increase in nitrogen in the effluent. Literature reports that excessive oxygen levels could inhibit the anoxic organisms and prevent effective denitrification (Dey et al., 2011; Hanhan et al., 2011; Gogina & Gulshin, 2021).

Nitrogen removal appeared to be achieved with a 150-minute cycle time and 75 minutes on time in one cycle. In other words, 50% of the time anoxic conditions were present. In particular, nitrate removal occurred during non-aerated phases. The denitrification process began when aeration was paused because the dissolved oxygen content dropped quickly. It was reported that the non-aerated phase's design is determined by the overall nitrogen effluent requirement and the quantity of nitrates that were produced during the aerated phase.

Additionally, according to Mu'azu et al., (2020), RAS and WAS are two crucial parameters that must be established in order to maintain the active microbial community in the activated sludge process. The simulations showed that these factors have a considerable impact on the model's performance. In particular, RAS percentage influenced MLSS values and effluent concentrations. For instance, simulations indicated a reduction in the nitrogen effluent concentration when the RAS fraction was increased. Previous research also revealed the importance of the WAS and RAS values in model calibration, particularly for nitrogen removal (Elawwad et al., 2019). RAS was also discovered to be a major factor in energy consumption as higher RAS resulted in higher energy requirements. This complies with the findings of Drewnowski et al., (2018). In contrast, aluminum sulfate was precipitated to complement the anaerobic tank in phosphorus removal. The amount of aluminum sulfate used was set to 5 gAl/m^3. These parameters were modified as they were found to be sensitive to the effluent concentrations required. Acceptable calibration was achieved with no further changes. The mantis3 default settings had been shown to work effectively with the actual data, and no further adjustments were required. The output variables that characterized the effluent were as follows: COD, BOD, TSS, TN, and TP. The BOD and COD parameters output model simulated concentrations were compared with real daily data whereas the remaining variables were compared with average values. Figure 4-1 shows the simulation results for BOD and COD and Figure 4-2 shows the predicted and actual average values of the output variables defining the effluent quality.

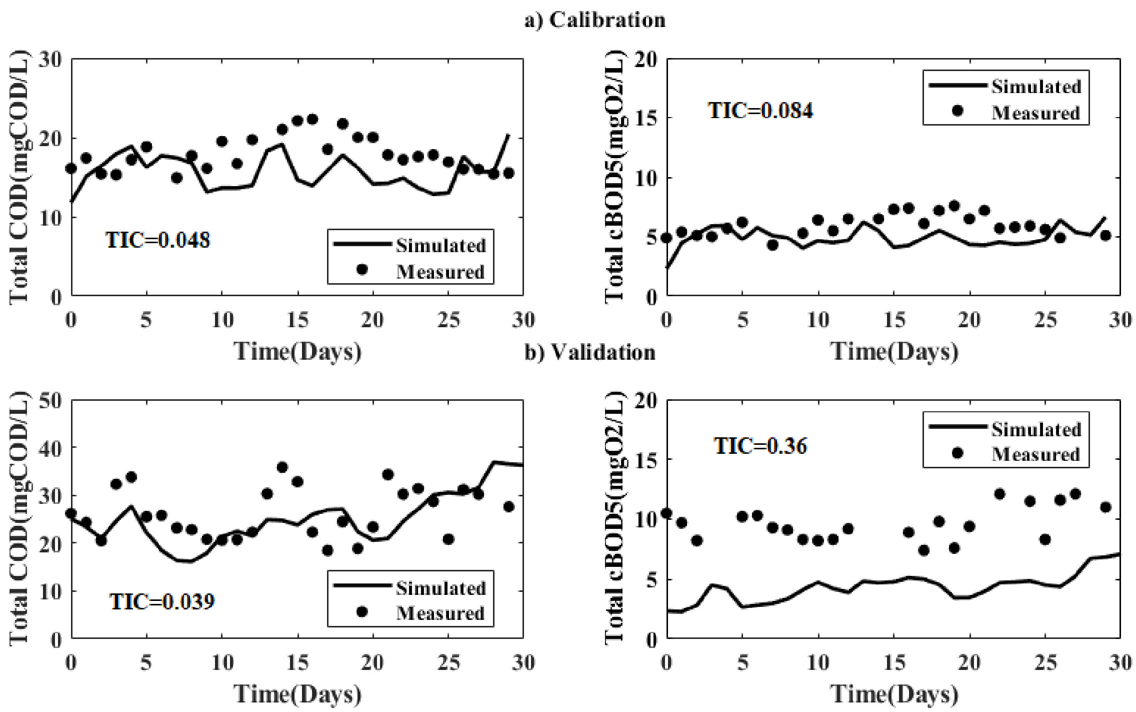


Figure 4-1: Model simulation based on the measured and simulated total BOD and COD using the data for a) calibration and b) validation

For the validation study, the October 2020 dynamic data was used. The validation data had different influent characterization for data validation on GPS-X. The model was successfully validated with the default parameter settings. Similar DO concentrations were implemented however RAS was modified. Similarly, the validation simulated results lined up well with the measured data within acceptable limits.

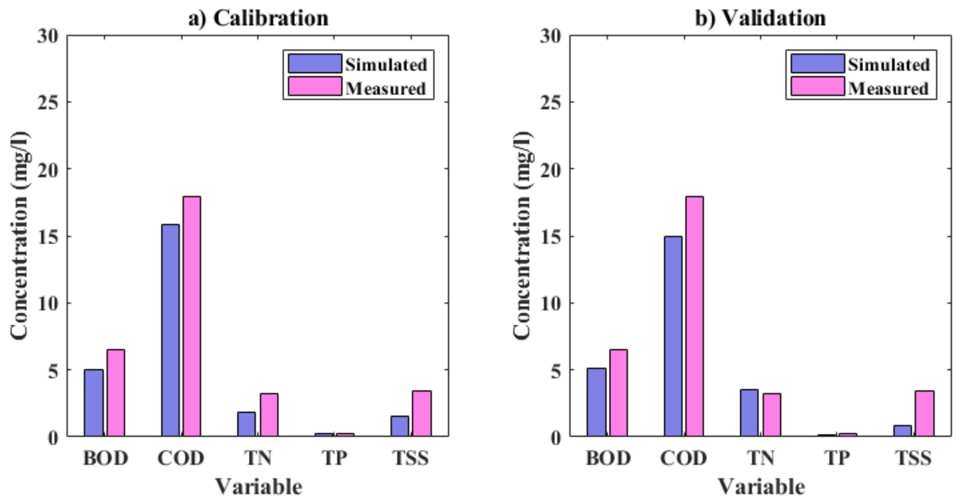


Figure 4-2: Average actual and predicted effluent concentration for a) calibration and b) validation

TIC values generated by GPS-X for BOD and COD served as a measure of how well the measured data and simulated values match together. According to Acosta-Cordero et al., (2020), TIC ranges between 0 and 1, with values closer to 0 indicating more model validity. When TIC is less than 0.3, it may be said that the measured values and the simulated values match well together. The BOD and COD have a TIC value of 0.084 and 0.048 in the calibration model and 0.36 and 0.039 in the validation model, respectively. These numbers were considered acceptable and matching with measured data.

4.1.2 Tested scenarios analysis

All scenarios had an effluent quality that met the discharge standard and was suitable for release. Table 4-1 lists the derived effluent components. The scenario with the anaerobic digestion (S1), led to the production of biogas and energy recovery. The total biogas produced was taken as a conservative estimate of $2800\ \mathrm{m^3 biogas/d}$. This amount generated was sufficient to meet the heat requirement of $1806\ \mathrm{m^3 biogas/d}$.

Table 4-1: Operational parameters and effluent qualities obtained in each scenario

Parameter	Unit	S0	S1	S2	S3
COD	mg/L	17.9	17.9	13.8	19.83
BOD	mg/L	6.5	6.5	2.44	2.83
TSS	mg/L	3.4	3.4	2.18	1.47
TN	mg/L	3.24	3.24	4.62	2.92
TP	mg/L	0.2	0.2	0.13	0.27
DO	mg/L	2	2	1	2
RAS	−	0.8	0.8	0.8	0.8
Internal recycle	%	−	−	300	300
Variation in WWT electricity consumption	%	−	−35	+1.84	−15.2

The biogas generated provided electricity usage to drop by 35% in S1. This was in compliance with other studies that found the AD process to save energy (Drewnowski et al., 2018). RAS and DO were the same as the baseline scenario with 0.8 and 2 mgO_2/L, respectively. Regarding Scenario 2, electricity usage increased by 1.84% due to longer aeration delivery time which consumed more electricity. The minor electricity increase difference between S0 and S1 was due to the lower DO concentration applied in S2 which might be equivalent to the reduction induced by lower aeration periods. According to Drewnowski et al., (2018), low DO concentrations were linked to low blower energy use and high DO concentrations required more energy to be given. SRT and mixed liquor suspended solids (MLSS) were designed in the range for an extended aeration system. In particular, a design of 20 days SRT was enough time for solids degradation which yielded lower sludge quantities than the baseline scenario since wastewater would be subject to extended aeration instead of intermittent aeration as is the case of S0. In this scenario, simulations showed that a DO concentration of 1 mg/L appeared to achieve a better nitrogen removal rate. In contrast, S3 consumed 15.2% less electrical energy than the baseline model. This could be due to the decreased mean cell residence time which caused less net energy consumption to stabilize aerobic solids in the reactor. A similar supposition was reached by Sarpong et al., (2020). An internal recycle brought nitrates and mixed liquor from the nitrification process in the aerobic zone to the anoxic tank for denitrification. Simulations showed that high internal and returned recycle rates were associated with higher nitrogen removal efficiencies and energy used for pumping. A 300% internal recycling was considered for both S2 and S3 and a recycle ratio of 80% of the wastewater stream from the secondary clarifier sent back to the anaerobic reactor was applied in all scenarios.

4.2 Life cycle assessment

4.2.1 Environmental performance of baseline scenario

The ZWWTP processes considered included biological treatment, chemical consumption, sludge landfilling, effluent discharge, and power consumption. The results shown in Figure 4-3 illustrate the midpoint impact characterization and contribution analysis associated with the baseline scenario (S0). As illustrated, the key factor in the plant's environmental impacts was electricity consumption, which had an influence on most midpoint indicators and yielded the most effect on 9 out of 18 midpoint indicators. It was responsible for the largest contribution of ionizing radiation (86.4%), ozone formation (97.4%), fine particulate matter formation (96.3%), terrestrial acidification (97%), terrestrial ecotoxicity (90%), land use (91.7%), fossil resource scarcity (96.5%), and water consumption (88.5%). This finding was consistent with previous LCA studies that observed electricity usage as the main contributor to the impacts of conventional WWTPs (Awad et al., 2019; Moussavi et al., 2021; Daskiran et al., 2022). Since the ZWWTP's PV solar panels only make a minor contribution to the overall monthly energy usage, expanding the local renewable energy source would help to reduce the negative effects of electricity generation on the environment.

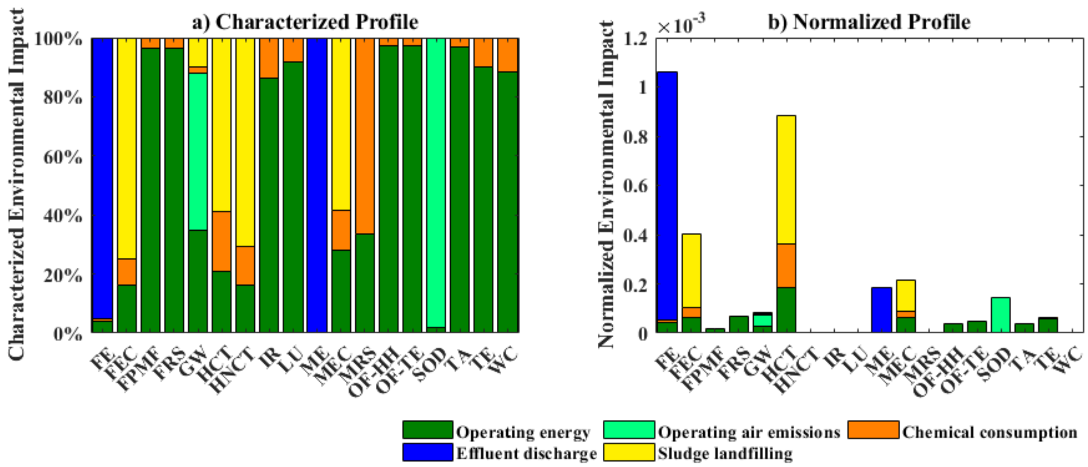


Figure 4-3: a) Characterized and b) normalized environmental impacts contributions for S0

As shown in Figure 4-3a, the emissions produced by biological treatment and energy use had a significant impact on global warming (53.5%) and stratospheric ozone depletion (97.9%). The operation of WWT facilities produced direct GHG emissions such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) through biological activities. In particular, CH_4, and N_2O emissions of biological treatment were frequently only taken into consideration as CO_2 emissions were typically obtained from biogenic organic matter and should not be included in total emissions as confirmed by Gruber et al., (2021). On the other hand, nitrous oxides N_2O emissions were released during the nitrification and denitrification processes used to remove nitrogen from wastewater. Studies reported that these emissions of N_2O played a role in both climate change and stratospheric ozone depletion (Gruber et al., 2021). CH_4 emissions were attributed to the incoming COD (around 1%) emitted as methane (Campos et al., 2016). All processes considered except the effluent discharged were contributors to the GWP indicator

(0.678 kg CO_2 eq). The impact on GW was mainly attributable to the gaseous emissions (i.e. CH_4, CO_2, and N_2O) released from the WWT, sludge landfilling, and energy use (Awad et al., 2019). Installing an energy capture system to use these CH_4 emissions as a power source could be one way to lessen these effects. Another common factor was chemical intake which included aluminum sulfate, polyacrylamide, and quicklime. Phosphates and aluminum phosphates were in general precipitated out of the solution using aluminum sulfate. Ionizing radiation (13.6%), terrestrial ecotoxicity (10%), freshwater ecotoxicity (9.24%), marine ecotoxicity (13.2%), human carcinogenic toxicity (20.4%), human non-carcinogenic toxicity (13.2%), land use (8.32%), mineral resource scarcity (66.6%), and water consumption (11.5%) were all impacted by chemical use (Figure 4-3a). The remaining contributions of chemical consumption were relatively small. This process only governed the mineral resource scarcity indicator as it represented the supply chain of chemicals from mineral resources. Similarly, the fossil resource scarcity indicator is nearly completely driven by power consumption (96.5%). Figure 4-3a further demonstrated that discharged effluent was the dominant contributor to freshwater eutrophication (95.1%) and marine eutrophication (99.8%). Eutrophication was caused by the nutrient content such as nitrogen and phosphorus of the released treated wastewater effluent to the surface water body (X. E. Yang et al., 2008). In particular, phosphorus content in discharged water accounted for FE (Foglia et al., 2021). In contrast, results demonstrated that the sludge landfilling process impacts GW (10%), FEC (74.7%), MEC (58.6%), HCT (59%), and HNCT (70.7%) (Figure 4-3a). Besides GW, sludge landfilling affects the other categories due to its direct emissions of heavy metals to the soil. These inorganic chemical substances were primarily toxic and had the potential to cause cancer (Agoro et al., 2020). According to studies, these hazardous substances were found in landfill leachates and constituted a severe public health concern (Boateng et al., 2019). Prior LCA research had reported on the detrimental effects of heavy metals on human health and ecosystem toxicity (Hospido et al., 2010; Foglia et al., 2021). On the other hand, sludge landfilling contributed to GW because of the GHGs it released into the atmosphere. Particularly, methane and carbon dioxide were emitted from the sludge decomposition in landfills (Wang et al., 2013). A strategy to reduce impacts would be adding a collection system that attempts to capture and utilize landfill biogas or what is known as a modern landfill (Themelis & Ulloa, 2007). The findings of normalized results (Figure 4-3b) revealed that the most important environmental impact categories were FE, HCT, and FEC. This was consistent with previous studies that found that freshwater eutrophication has the most significant impact on the ecosystem followed by human toxicity (Mamathoni & Harding, 2021). HCT was influenced by power consumption, chemical consumption, and sludge landfilling. Its high significance to the total impacts was mostly due to Lebanon’s production of energy. Figure 4-4 displays the normalized outcomes of power generation in Lebanon so that this influence may be examined. The primary environmental consequence of producing energy in Lebanon was human carcinogenic toxicity, as seen in Figure 4-4. This may be attributed to the fact that EDL's increased air pollution was caused by the burning of heavy fuel oil and diesel, which emitted harmful chemicals like NO_x and CO_2 that impair human health and contribute to air pollution (Julian et al., 2020).

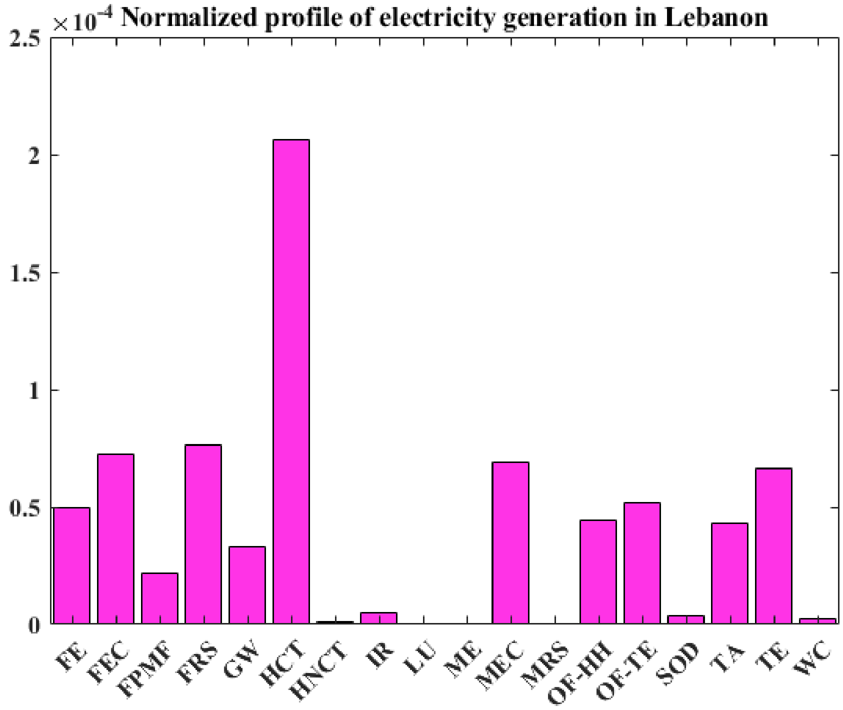


Figure 4-4: Normalized results for electricity generation in Lebanon

In terms of endpoint damage assessment, Figure 4-5a displays the percent contributions to endpoint impact indicators. The total impact on human health for $1\ \mathrm{m}^3$ of treated wastewater was $1.02\mathrm{E}-06$ DALY. All processes, aside from discharged effluent, had an influence on human health. Power consumption (53%) was the biggest factor affecting human health, followed by biological treatment (33.4%), sludge landfilling (10.3%), and chemical consumption (3.32%). All processes had an impact on the ecosystem indicator which totaled $2.86\mathrm{E}-09$ species. yr. The largest impact on ecosystems came from power consumption (40.4%) and biological treatment (35.4%). According to previous studies, WWT impacts on human health and ecosystems were associated with WWT aeration (Roman & Brennan, 2021).

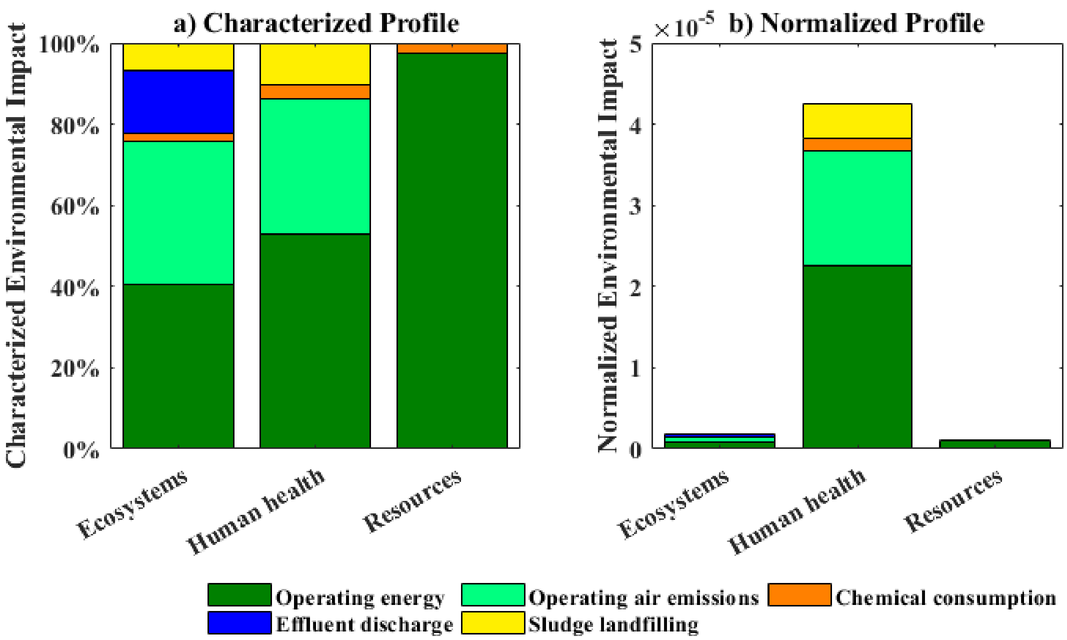


Figure 4-5: a) Characterized and b) normalized damage assessment contributions for S0

The sludge landfilling emissions contributed to the ecosystem by $1.96\mathrm{E}-10$ species. yr and to human health by $1.05\mathrm{E}-07$ DALY. In contrast, impacts on resources added up to $3.07\mathrm{E}-02$ USD2013 where chemical consumption contributed by $8.00\mathrm{E}-04$ and power consumption contributed by $2.99\mathrm{E}-02$ USD2013. From this endpoint analysis, power consumption appeared to have the largest impact on the endpoint categories and was the most concerning. Even though there was a big portion of impacts coming from chemical consumption and sludge landfilling to the midpoint categories, they did not appear to have the largest contribution to endpoint indicators as different allocations were given. Despite the fact that emissions from operational tanks seemed significant to human health damage, they were inevitable when considering the current plant. In regard to the normalized results for the endpoint categories in Figure 4-5b, it was identified that the impact on human health is the largest ($4.26\mathrm{E}-05$), followed by ecosystems ($1.94\mathrm{E}-06$), and resources ($1.10\mathrm{E}-06$). As a result of this examination, it could be concluded that as the current ZWWTP is operating, the detrimental impacts come majorly in its power consumption.

4.2.2 Scenario analysis

4.2.2.1 Scenario 1

Scenario 1 was the upgraded version of the baseline scenario as it included the addition of anaerobic digestion. This process could cut landfill emissions, remove energy consumed for deodorization, and provide biogas to power the plant. Figure 4-6 compares S1's environmental performance to that of the baseline scenario and depicts S1's percent contribution to the midpoint impact categories. There had been no modifications to the biological treatment and emissions remain unchanged. As shown in Figure 4-6a and Figure 4-6b, AD impacts on impact categories were almost nonexistent as they only influenced the system's water consumption. The increase in water consumption was attributed to the hot water recirculation needed for heating in the AD tank (Qasim & Zhu, 2018). AD consisted of 97.1% of the total impacts on water consumption. Carbon dioxide emissions from biogas combustion were not considered since they are biogenic. As expected, the addition of an anaerobic digestion tank reduced the environmental impacts in all categories except water consumption. The reduction of these impacts aligned with previous LCA studies in which an AD was added (Awad et al., 2019). The amount of power used was reduced from 0.246 kWh to 0.16 kWh per FU. In addition, global warming was lowered from 0.678 to 0.521 kg CO_2 eq. This decrease was a result of the biogas production utilized to generate a part of the plant's power requirements, consistent with Awad et al., (2019). As previously acknowledged, power consumption was accounted for in most categories, thus all reductions were anticipated as a result of the energy recovery process that was put forward.

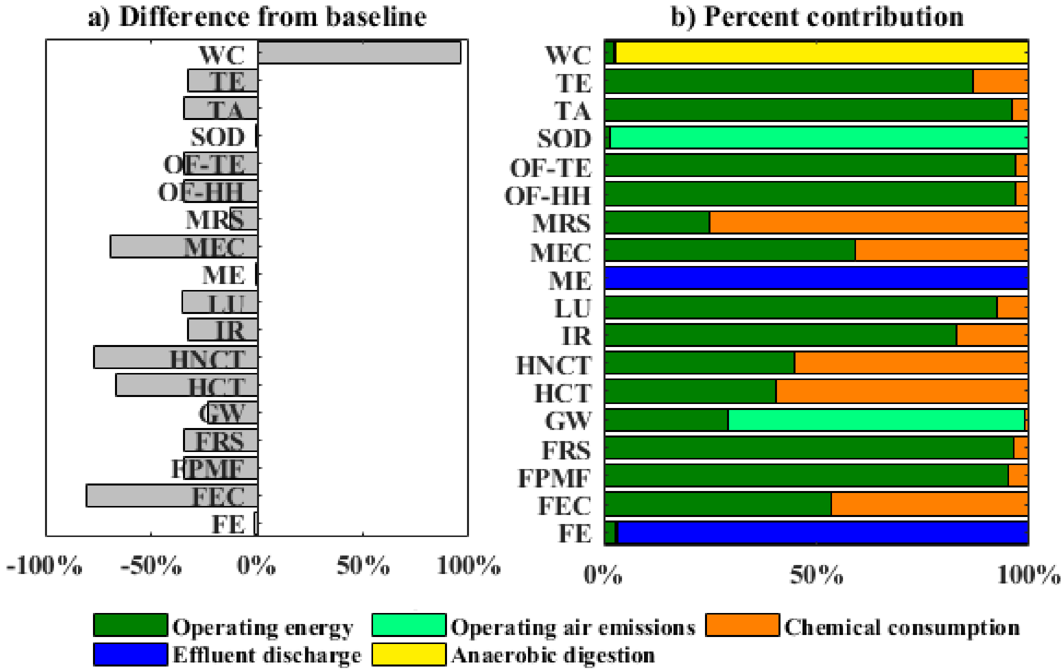


Figure 4-6: Scenario 1 results: a) Difference from baseline and b) percent contribution

On the other hand, the sludge resulting from the AD process referred to as a digestate could be directly land applied as a fertilizer product (Costa et al., 2015). The reuse of this sludge avoided the production of fertilizer to use in agricultural land applications (Hospido et al., 2010). However, due to the presence of heavy metals in the digestate for agricultural use, it was essential to ensure that the freshwater ecotoxicity did not exceed the limits anticipated for thermal operations as stated by Tarpani et al., (2020). While marine eutrophication remained unchanged, freshwater eutrophication and mineral resource scarcity had both somewhat decreased. Eutrophication was unchanged since it was mostly governed by the effluent quality which had not altered (Awad et al., 2019). The mineral resource scarcity had diminished as a result of the decline in electricity demand. The possibility for terrestrial acidification and ecotoxicity had been lowered by 34.3% and 32.7% respectively. It was crucial to note that the inclusion of AD could have both good and negative impacts occurred on the toxicity of the terrestrial environment. While it could reduce electricity consumption, the production of biosolids and digestate liquid could also have a detrimental effect. The toxicity categories yielded the greatest reduction in this scenario because these categories were predominantly dominated by power consumption and sludge landfilling emissions.

4.2.2.2 Scenario 2

In scenario 2, the biological treatment was substituted by an anoxic tank and an extended aeration tank. Extended aeration is a modified activated sludge procedure where aeration tank residence time is prolonged, allowing for adequate solids degradation. Figure 4-7 displays S2's percentage contribution and environmental performance relative to the baseline scenario. The overall environmental performance of this scenario was mostly influenced by power consumption similar to the baseline model. This was compatible with other LCA studies on extended aeration (Lopsik, 2013). Energy utilization had impacts that were essentially equivalent to S0. The majority of the environmental impacts of this scenario were similar, and 3 out of 18 impact categories were altered. In this scenario, aeration was substantially higher since supply was spread out over a longer time, which increased the operational energy consumption of the process. However, as discussed earlier, this increase was relatively small due to the lower DO concentration given over the extended aeration period.

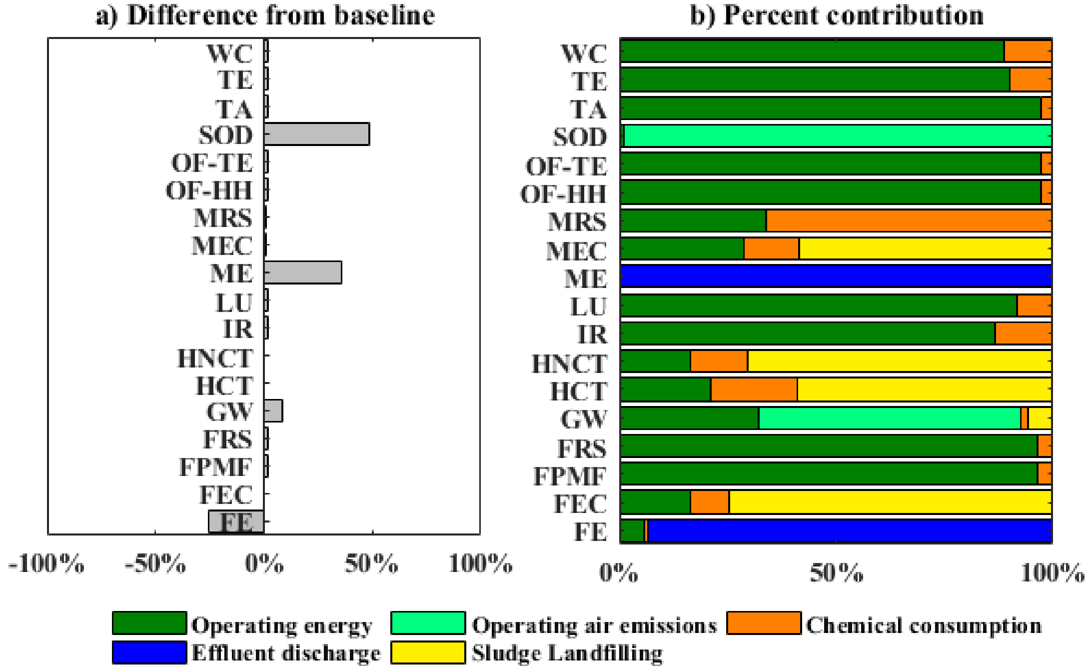


Figure 4-7: Scenario 2 results: a) Difference from baseline and b) percent contribution

Studies reported that the extended aeration process needed a higher electricity consumption due to higher aeration (Drewnowski et al., 2018; Sean et al., 2020; Daskiran et al., 2022). As electricity use was linked to most categories, this scenario led to comparable impacts on most categories. Figure 4-7a shows a very slight increase in fossil resource scarcity (1.5%), land use (1.5%), water consumption (1.4%), and others. The impact on GW increased by 8.4%. This was attributed to the higher GHG emissions from biological treatment particularly N_2O. As less nitrogen was removed from wastewater, higher N_2O was emitted. According to Law et al., (2012), plants that removed a lot of nitrogen generally emit less N_2O. This was also apparent in the increase in stratospheric ozone depletion (48.3%) due to the increased N_2O emissions from the EA process as well as the increase in electricity consumption. In particular, N_2O emissions were identified as the main contributor to the stratospheric ozone depletion category (Ravishankara et al., 2009). Accordingly, marine eutrophication appeared to increase by 36.1% due to the greater nitrogen concentrations in effluent water whereas freshwater eutrophication was reduced by 25.7% due to lower phosphorus concentration in the effluent. According to the literature, nitrogen emissions had been identified as the primary factor contributing to marine eutrophication (Daskiran et al., 2022) whereas phosphorus emissions contributed to freshwater eutrophication (Foglia et al., 2021). On the other hand, there was a reduction in sludge landfilling impact compared to the baseline model. Although this reduction was not vividly apparent in Figure 4-7a, it was determined by the reduced amount of sludge produced resulting in lower landfilling emissions. In this type of treatment, sludge production was kept at a minimum as sludge mass and volume decreased resulting in relatively low sludge yield (Mccarty & Brodersen, 1962; EPA, 2000; Nikmanesh et al., 2018). In actuality, the amount of sludge that would be landfilled was around 37% lower than the amount produced by the existing ZWWTP. Three factors could account for this: HRT, SRT, and extended aeration delivery. As the plant's hydraulic capacity was being exceeded, and stormwater infiltration may be to blame for this, the time activated sludge bacteria are retained in the secondary clarifiers and the detention time in the aeration tank was shortened (Ramanadham et al., 2013). Additionally, the SRT controlled how much bacteria were present throughout the treatment system; when it was raised, more bacteria grew in the reactor, resulting in reduced sludge production (Wong et al., 2003). Moreover, as the mixture was aerated for a longer period of time, it enabled the organisms that were breaking down the sludge to continue growing and feeding. As a result, the system generated less total sludge that was suitable for direct dewatering as in this study. This reduction affected the GW and the contribution to it fell by 35%. Given that the reduced emissions concerned CO_2 and CH_4, they did not have an impact on the categories where soil emissions are a factor. As follows, the change of impacts was mostly attributable to the change in energy demand for the plant's operation, which is the main cause of CO_2 emissions (Sharvini et al., 2022).

4.2.2.3 Scenario 3

Scenario 3 replaced the biological treatment with a five-stage Bardenpho process and thus eliminated the need for chemical dosage. The contributions to environmental impacts of scenario 3 in comparison to the reference scenario are depicted in Figure 4-8. As shown in Figure 4-8a, S3 showed higher benefits compared to the baseline scenario in terrestrial ecotoxicity $(0.782 \text{ kg } 1{,}4 \text{ DCB})$, human carcinogenic toxicity $(0.00708 \text{ kg } 1{,}4 \text{ DCB})$, and ionizing radiation $(0.00184 \text{ kBq Co} - 60 \text{ eq})$. Since energy consumption was the greatest contributor to this scenario's environmental impacts and since it was reduced, fewer impacts were caused in 15 out of 18 categories. The majority of impacts were reduced by $10 - 30\%$. In terms of air emissions impacts, the biological treatment in this scenario emitted greater amounts of N_2O. Specifically, on a time scale of 100 years, N_2O had a 300-times greater global warming potential than carbon dioxide and plays a significant role in stratospheric ozone depletion (Ravishankara et al., 2009; Griffis et al., 2017). This increase could be attributed to the greater number of tanks involved to ensure greater removals of carbonaceous matter and nutrients (Kyung et al., 2015). Particularly, higher N_2O emissions increased the impact on stratospheric ozone depletion by 38.4%, and on global warming which was counterbalanced by the eliminated effect of chemical dosage on GW.

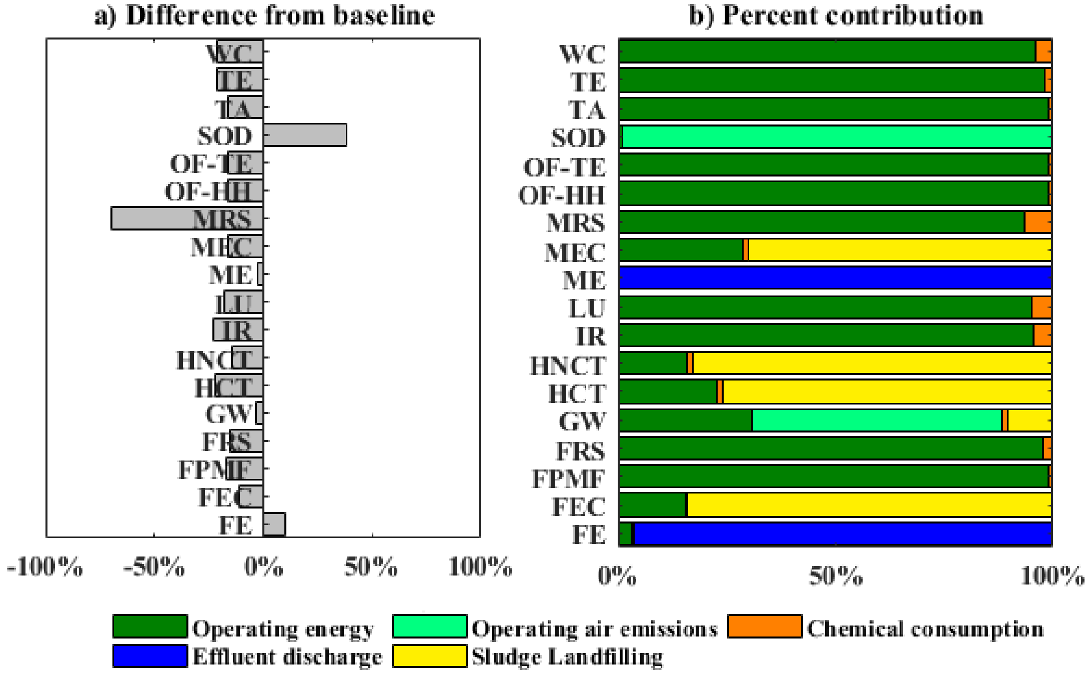


Figure 4-8: Scenario 3 results: a) Difference from baseline and b) percent contribution

Additionally, this scenario showed a larger impact on freshwater eutrophication (10%) due to the increased nutrient concentrations in the effluent from the FSB process (Foglia et al., 2021). Specifically, the final effluent was richer in phosphorus content compared to the baseline scenario. According to Daskiran et al., (2022), the most significant factor contributing to freshwater eutrophication was phosphorus emissions in wastewater. On the other hand, this scenario eliminated the need for aluminum phosphate precipitation which reduced the contribution of chemical usage to the mineral resource depletion category to 6.79%. Referring to Figure 4-8b, chemical consumption's contribution to the remaining environmental impacts was minimal. This reduced amount of chemicals and energy required produced the most positive effect on impact reduction. Similar to S0, the direct emissions to soil (heavy metals) and air from sludge landfilling had the same contributions. It was apparent that this alternative performs better regarding most environmental indicators.

4.2.3 Overall results

In all scenarios, direct GHG emissions from biological treatment accounted for more than half of the total impacts on GW. In scenarios where sludge landfilling was considered, heavy metals emissions to the soil accounted for $30 - 70\%$ of the overall impact on ecotoxicity. In contrast, sludge reuse (i.e. fertilizer) in S1 reduced these environmental impacts. The better environmental performance of a scenario was mainly attributed to the amount of energy consumption. It had been also demonstrated that biological treatment emissions have a high contribution to stratospheric ozone depletion. Most impacts were to some extent influenced by chemical usage apart from the mineral resource scarcity where chemical consumption had the most influence. The findings from the midpoint impact assessment across the several scenarios tested with each category in its unit of reference are shown in Table 4-2. An overall comparison of the characterized analysis results for all scenarios is shown in Figure 4-9. Supplementary results are found in Appendix D. Overall, S1 proved to be the most environmentally friendly alternative. This was primarily due to the reduced energy requirements when the AD was added. The system had the lowest impact in all of the impact categories except in water consumption as the AD unit consumed water for heating. This was in accordance with previous studies which observed that adding an AD significantly reduced the environmental impact of the plant (Awad et al., 2019). As expected, water depletion was the most apparent impact category between the baseline and proposed scenarios.

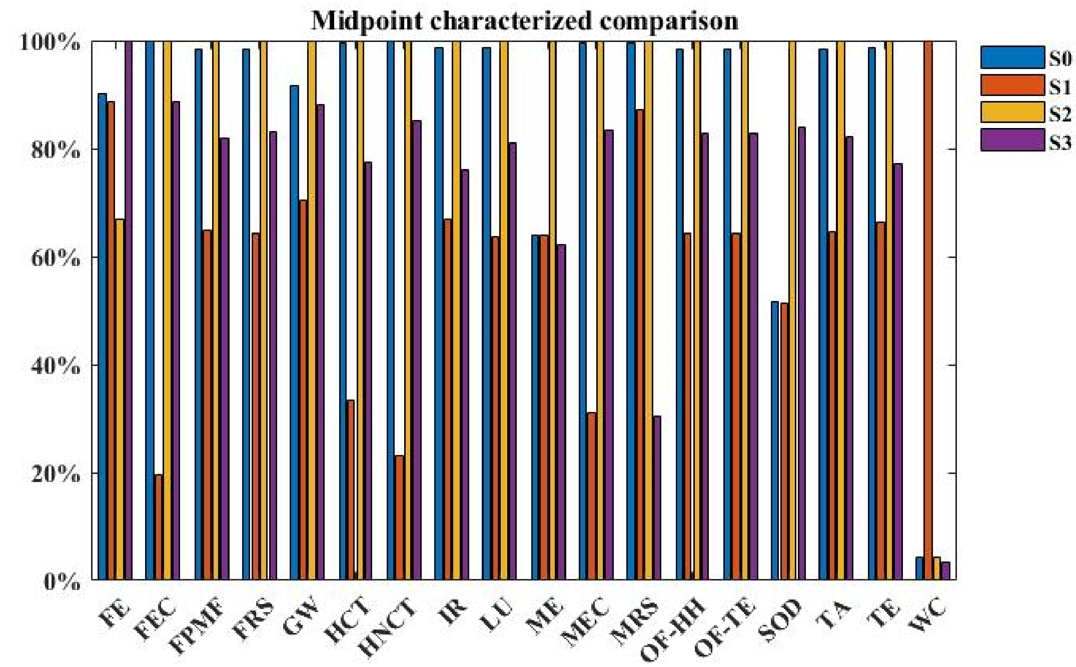


Figure 4-9: Comparison of the characterized midpoint impact categories for all scenarios

Table 4-2: Characterized midpoint assessment results per m^3 of treated wastewater

Impact category	Unit	Scenarios			
		S0	S1	S2	S3
Global warming	kg CO2 eq	$6.78E-01$	$5.21E-01$	$7.40E-01$	$6.53E-01$
Stratospheric ozone depletion	kg CFC11 eq	$8.63E-08$	$8.57E-06$	$1.67E-05$	$1.4E-05$
Ionizing radiation	$kBq\ C-60$ eq	$2.38E-03$	$1.61E-03$	$2.41E-03$	$1.84E-03$
Ozone formation, Human health	kg NO_x eq	$8.24E-04$	$5.39E-04$	$8.37E-04$	$6.92E-04$
Fine particulate matter formation	kg PM2.5 eq	$5.11E-04$	$3.37E-04$	$5.19E-04$	$4.25E-04$
Ozone formation, Terrestrial ecosystems	kg NO_x eq	$8.32E-04$	$5.44E-04$	$8.45E-04$	$7.00E-04$
Terrestrial acidification	kg SO_2 eq	$1.62E-03$	$1.10E-03$	$1.64E-03$	$1.35E-03$
Freshwater eutrophication	kg P eq	$6.87E-04$	$6.77E-04$	$5.11E-04$	$7.64E-04$
Marine eutrophication	kg N eq	$8.57E-04$	$8.57E-04$	$1.34E-03$	$8.33E-04$
Terrestrial ecotoxicity	$kg\ 1,4-DCB$	$9.97E-01$	$6.71E-01$	$1.01E+00$	$7.82E-01$
Freshwater ecotoxicity	$kg\ 1,4-DCB$	$1.00E-02$	$1.97E-03$	$1.00E-02$	$8.97E-03$
Marine ecotoxicity	$kg\ 1,4-DCB$	$9.42E-03$	$2.94E-03$	$9.46E-03$	$7.88E-03$
Human carcinogenic toxicity	$kg\ 1,4-DCB$	$9.11E-03$	$3.00E-03$	$9.14E-03$	$7.10E-03$
Human non-carcinogenic toxicity	$kg\ 1,4-DCB$	$1.48E-01$	$3.50E-02$	$1.49E-01$	$1.27E-01$
Land use	m^2a crop eq	$3.12E-03$	$2.00E-03$	$3.16E-03$	$2.56E-03$
Mineral resource scarcity	kg Cu eq	$2.28E-04$	$2.00E-04$	$2.29E-04$	$6.96E-05$
Fossil resource scarcity	kg oil eq	$6.89E-02$	$4.50E-02$	$7.00E-02$	$5.81E-02$
Water consumption	m^3	$6.74E-04$	$1.60E-02$	$6.84E-04$	$5.31E-04$

The best environmental performance in terms of global warming was S1 which was anticipated given the reduced energy use. S0 and S2 had almost the same performance applied in 14 out of 18 environmental impact categories. The baseline model and S2 appeared to have the greatest environmental impact potentials compared to the other scenarios except for global warming where S2 is higher by 8.4%. This could be due to the higher emissions of air pollutants and full reliance on the electrical grid as well as the substantial intake. In contrast, S2 showed the highest impact on stratospheric ozone depletion due to higher N_2O emissions. This was consistent with the findings of Castro-Barros et al., (2015) who reported that most of N_2O emissions produced in WWT were highly affected by aeration. Indeed, in S2, a higher aeration time was taking place in the opposite of S0, and S1 where intermittent aeration was adopted. Mainly, intermittent aeration had been shown to be a successful method for reducing N_2O emissions while still maintaining high nitrogen removal efficiency in WWTPs (Batch et al., 2021). As mentioned earlier, S2 generated less sludge. Since heavy metals soil emissions did not change, S2 saw a small positive impact from reduced sludge on global warming where the sludge landfilling process impact had decreased. This reduction was offset by the increased nitrous oxides emissions. In addition, the same amount of sludge was landfilled in S0 and S3 therefore air emissions arising were the same. As heavy metals emissions to soil were the same, the sludge landfilling process' impact on ecotoxicity categories remained constant in all scenarios. This was compatible with recent LCA studies that had shown that the main factors considered in freshwater and marine ecotoxicity were heavy metal emissions (Daskiran et al., 2022). According to Thomsen et al., (2018), the effluent nitrogen was connected with the impact category for marine eutrophication, thus S2 had the highest environmental impact due to its highest nitrogen emission. For the mineral resource scarcity, Figure 4-9 shows that the Bardenpho scenario (S3) had the least impact (69.6%) among all scenarios as it was the only scenario where aluminum precipitation was not applied. S3 gave relatively higher environmental benefits than S0 and S2 except for FE and SOD. It was seen earlier that S3 effluent had the greatest phosphorus and lowest nitrogen contents, which accounted for the worst and best results of the freshwater and marine eutrophication effect categories. In terms of terrestrial acidification, S0 is slightly better than S2. This was explained by the nearly identical net energy contributions in S0 and S2 which led to the same performance as well in fossil resource scarcity. This was explained by the fact that TA and the scarcity of fossil fuels are energy-related categories and that their respective net energy contributions were essentially equal (Daskiran et al., 2022). With respect to the land use category, S1 appeared to be the best performing followed by S3 while the performance of S2 was more or less the same as S0.

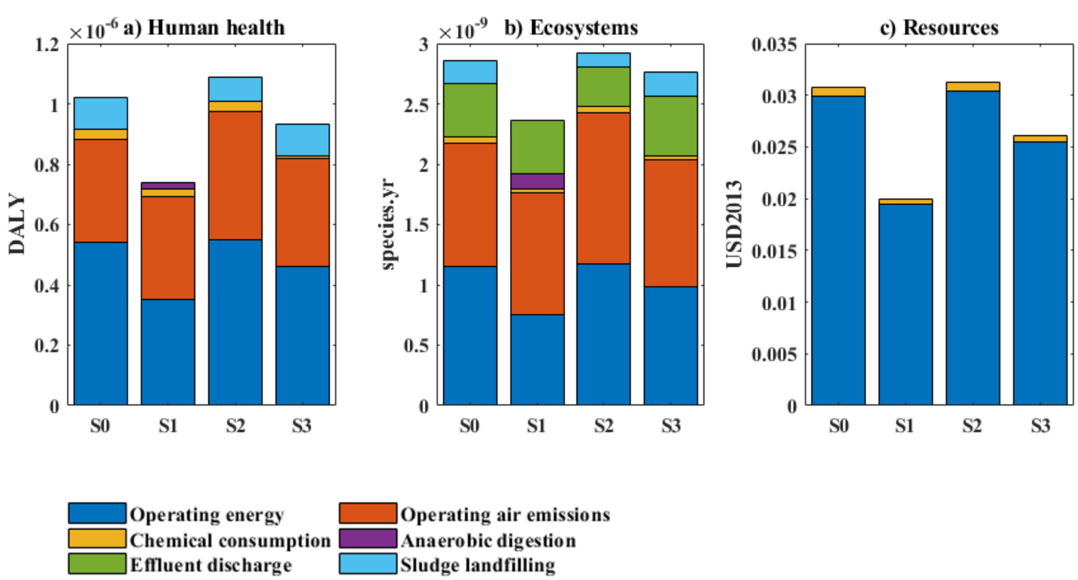


Figure 4-10: Comparison of the characterized endpoint impact categories for all scenarios

As currently operated, the endpoint categories impacts are shown in Figure 4-10. As demonstrated in Figure 4-10, S1 represented the most environmentally friendly option in all damage categories. It was less by 32% in human health, 19.2% in ecosystems, and 35.9% in resources. S0 and S3 had comparable impacts on resources while S2 had a greater impact on each of human health by 6.2% and ecosystems by 2.2%. Impacts on ecosystems emerged significantly from GW, OF, TA, and FE.

S0 performed slightly better than S2 in all three indicators. S2 delivered higher impacts compared to other options due to high CO_2 equivalent emissions and zero resource recovery, which would have a higher impact on related categories. Regarding the analysis of the contribution, it was seen that treated water affects solely the ecosystem through nutrients in discharged water while biological treatment and sludge landfilling affect human health via the GHG emissions released. Energy consumption dominated and chemical use was low-lying. The only difference that arose among scenarios is the contributions of each. For instance, as S3 consumed fewer chemicals, its consumption contribution to human health and ecosystems was less. As a result, the endpoint results supported the previously reported midpoint results and endpoint contribution pattern which showed that electricity is the main contributor to the impacts on human health, ecosystems, and resource depletion.

To compare impact categories on the same basis, the normalized environmental profiles of all scenarios are presented in Figure 4-11. According to the normalization findings, FE and HCT were the most important impact categories in all scenarios.

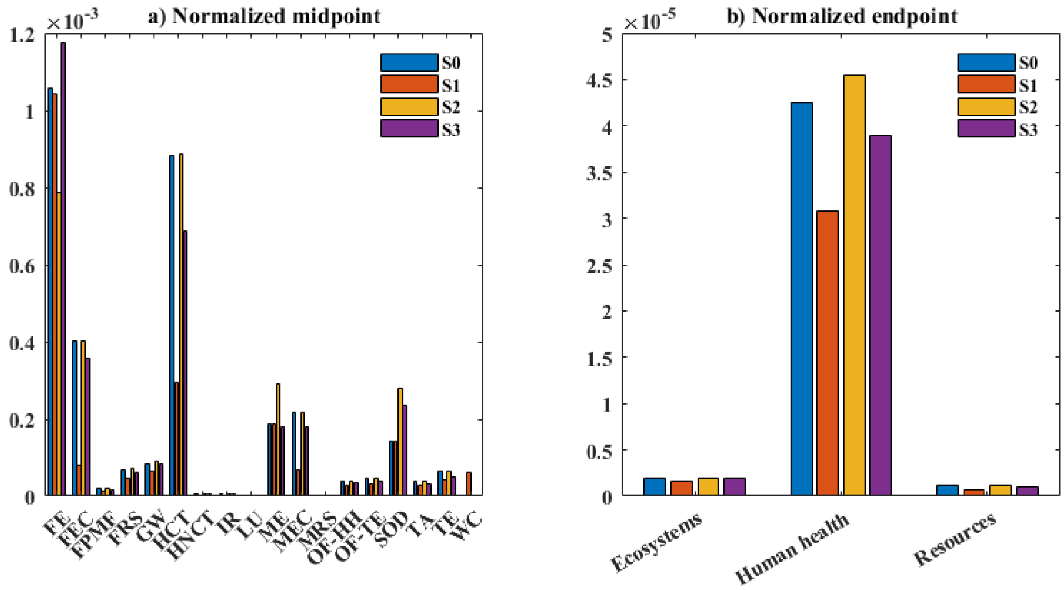


Figure 4-11: Results obtained using the normalization mode

This work demonstrated that the combination of an anaerobic digester with the current plant seems to be the most environmentally friendly alternative as carbon footprints are minimized. Additionally, it could provide a financial profit as fewer fertilizers will be supplied, creating a situation where both the economy and the environment win. Finally, it could be concluded that the biggest factor affecting the environment is energy consumption. However, it should be noted that when the electric grid switches to renewable resources, the relative operating energy input of some impact categories may decline.

4.3 Life cycle cost

4.3.1 CapdetWorks

CapdetWorks v.4.0 generated costs in 7 categories: project, operation, maintenance, material, chemical, energy, and amortization. In particular, operation and maintenance (O&M) costs were divided into five areas under the CapdetWorks model: operation labor, maintenance labor, power, chemicals, and materials (Arif et al., 2020). All categories were expressed in \$/yr except the project cost expressed in \$. The yearly costs could be translated into present worth using the net present value equation while project costs could be presented in a series of equal annual payments using the CRF. Table 4-3 and Figure 4-12 show a summary cost comparison of the four alternatives.

Table 4-3: Cost summary of WWTPs' different scenarios

Layout	Present Worth (\$)	Project $\times 10^3$ (\$)	Operation (\$/yr)	Maintenance (\$/yr)	Material (\$/yr)	Chemical (\$/yr)	Energy (\$/yr)	Amortization $\times 10^3$ (\$/yr)
S0	35,101,506	23,800	123,000	23,900	166,000	174,000	226,000	1,620
S1	37,563,195	26,300	135,000	31,800	183,000	150,000	186,708	1,810
S2	38,784,840	26,300	113,000	30,700	234,000	180,000	230,000	1,790
S3	35,257,988	25,400	114,000	37,900	199,000	71,400	191,000	1,730

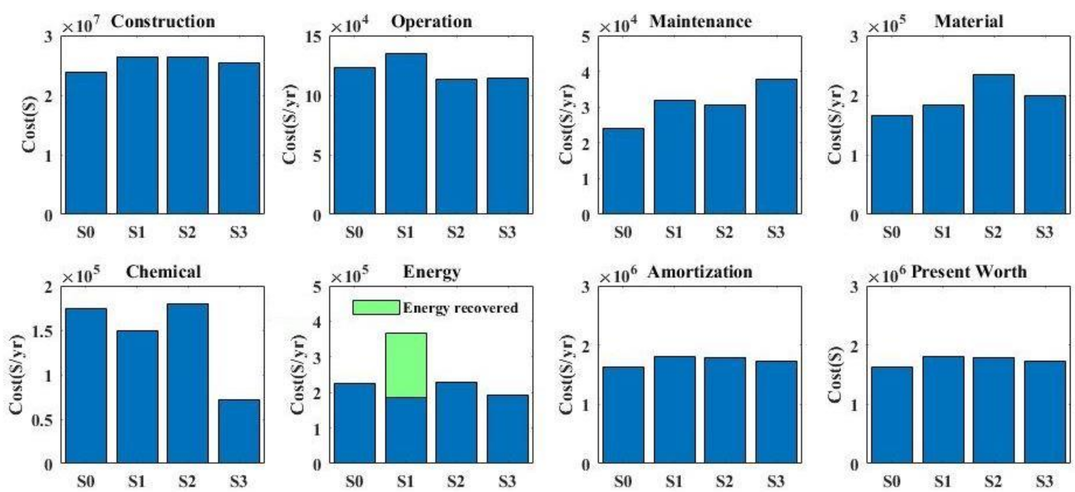


Figure 4-12: Cost summary of WWTPs' different scenarios

According to detailed cost data, the baseline model had a present worth of \$35 million, and a project cost of \$24 million. Comparing the different scenarios, results revealed that S2 has the highest present worth while S0 had the lowest. It was shown that the present worth cost of S2 was around 10%, 3%, and 10% more than that of S0, S1, and S3, respectively. Higher operation and maintenance costs in S1 originated from the additional unit process added whereas lower energy costs were attributed to the energy recovered from the AD system. The biogas power plant expenses were added as recovered energy is deducted. As shown in Figure 4-12, S1 ranked top from an energy demand perspective. This outcome was in line with Awad et al., (2019) who reported that energy consumption expenses were reduced when AD is added. The data collected showed that energy recovery had a substantial part in energy cost reduction as the biogas generated met around 50% of the needed energy at the WWTP. These results were completely consistent with literature reporting that biogas recovery lowers total costs (Nowrouzi et al., 2021).

In contrast, the applied aeration systems in the S2 resulted in higher energy consumption which ultimately drove up energy costs. According to Arceivala & Asolekar, (2012), by increasing denitrification in the aeration tank concurrently with nitrification, the price of extended aeration systems could be reduced. However, compared to activated sludge systems, the power consumption was higher and thus the system's operating expenses were higher. Additionally, the expenses of air supply systems in S2 were to some extent higher than S0's energy costs as lower DO concentrations were implemented. This complied with literature reporting that low DO concentrations were associated with low amounts of blower energy usage. Additionally, S1 and S2 had the highest construction costs whereas the main model (S0) had the lowest. This was explained by the more expensive equipment needed. According to Nasr et al., (2019), the construction section made up the majority of the costs associated with implementing a WWTP. In fact,

Figure 4-12 shows that the construction process played a significant part in the overall assessed costs as most of the contribution to total cost came from these expenses alone. Concerning amortization costs, results revealed that S1 has the highest while S0 had the lowest.

In regard to maintenance costs, S3 plant held the highest place because of the higher tank number involved in the process. Maintenance of scenario 3 included the maintenance of 5 reactors. It is important to point out that S1 plant ranked second highest place in terms of maintenance costs which was explained by the additional AD tank. On the other hand, chemical costs were the lowest in S3 where they had decreased by around 60% compared to the baseline model. The cost of chemicals ranked the highest for S2 ($180,000) and the lowest for S3 ($71,400) as chemical precipitation was eliminated in the Bardenpho process. The difference on the other hand in chemical costs between S0, S1, and S2, was the usage of polymers and other chemicals in the system which depended on the process. The Bardenpho technology (S3) was more expensive than S0 in terms of materials, maintenance, amortization, and construction expenses. This was due to the higher number of tanks involved. However, on the other hand, S3 held lower energy costs than the baseline model as the energy required was reduced.

According to the results, S1 was the most economically advantageous technique in terms of energy usage, S2 in terms of operation and maintenance, S3 in terms of chemical usage, and S0 in terms of construction, materials, and amortization. The cost of the plant for each m^3 of treated wastewater were determined using the equations in Table 4-4 (Arif et al., 2020; Nowrouzi et al., 2021). The annualized project costs were determined by multiplying the project cost by CRF to divide the total cost into annual equal payments at an interest rate $i = 6\%$ and life span $n = 40$ years. Costs per unit volume of wastewater flow were calculated referring to the computations done by Arif et al., (2020) and Waleed, (2007). Table 4-4 shows the summary of the costs of treated wastewater per m^3.

Table 4-4: Cost summary of treated wastewater per m^3

Cost item	Unit	Value			
		S0	S1	S2	S3
Total project costs	$	23,800,000	26,300,000	26,300,000	25,400,000
Total operation and maintenance costs	$/yr	712,900	686,508	787,700	613,300
Annualized project costs[a]	$/yr	1,570,800	1,735,800	1,735,800	1,676,400
Annualized project costs + annual O&M costs	$/yr	2,283,700	2,422,308	2,523,500	2,289,700
Cost /m^3[b]	$/$m^3$	0.249	0.265	0.276	0.25

[a] $\text{Annualized project cost} = \text{project cost} \times \text{CRF}, i = 6\%, \text{period} = 40 \text{ yrs}, \text{CRF} = 0.066$
[b] $\text{Cost/m}^3 = (\text{annualized project cost} + \text{annual O\&M cost}) \frac{/\text{yr}}{(\text{average design flow} \times 365)\ \text{m}^3/\text{yr}}$

Adapted from "Cost analysis of activated sludge and membrane bioreactor WWTPs using CapdetWorks simulation program: Case study of Tikrit WWTP (middle Iraq)", by A. U. A. Arif, M. T. Sorour, S. A. Aly, 2020, *Alexandria Engineering Journal*, *59*(6), p. 4665 (https://doi.org/10.1016/j.aej.2020.08.023). Copyright 2020 by The Authors.

The most economical scenario was the wastewater treatment facility with the lowest unit cost per cubic meter. According to Table 4-4 and using the average flow of $25{,}081\ m^3/d$, S0 was the least expensive scenario and costs $\$0.249/m^3$ compared to S1 ($\$0.265/m^3$), S2 ($\$0.276/m^3$), and S3 ($\$0.25/m^3$). The comparison of the total costs for the four scenarios showed that costs were almost similar. S0 process was regarded as the most cost-effective when comparing the results of the cost estimations for the four processes since it had the lowest $cost/m^3$.

4.3.1.1 Removal efficiency

In terms of efficiency, Table 4-5 lists the removal efficiency of each COD, BOD, TN, TSS, and TP from each WWTP scenario. The highest removal percentages for all parameters taken into consideration were either achieved by S2 or by S3. The baseline scenario and S1 had the same removal efficiencies as no modifications were done to the biological treatment. S2 and S3 showed higher efficiencies in BOD removal with 98.5% and 98.3% respectively. For the simulated Bardenpho configuration, the average TP (93.5%) and BOD(98.3%) removal efficiencies for S3 were lower by 3.4% and 0.2%, from S2 respectively. However, the modified Bardenpho scenario (S3) demonstrated high effluent quality and a superior nitrogen removal performance compared to the remaining scenarios. This was in accordance with studies reporting that the FSB design had high removal efficiencies and achieved high nitrogen removal efficiencies (Banayan Esfahani et al., 2019; Demir, 2020). These results were also somewhat similar to literature findings by Bashar et al., (2018) who found removal efficiency for a modified Bardenpho to be of 94%, 99.5%, and 83.5%, for COD, TSS, and TP, respectively.

It could also be noted from Table 4-5 that S3 had exceptional effluent quality apart from phosphorus concentration which was the highest in this scenario mainly due to the exclusion of aluminum phosphate precipitation. In this regard, TSS and COD removal efficiencies were found to be nearly equivalent in all scenarios whereas TN showed a bigger difference. Particularly, S2 had the lowest and the highest TN and TP removal. The overall highest removal efficiencies were found in the EA design. The factor that led to S2 superiority was the highest removal rate of each COD (97%), BOD (98.5%), and TP (96.9%). As a result, S2 was found to be the most expensive alternative with the best overall removal efficiencies. This finding about the association of bigger operating costs with higher removal efficiencies complied with other literature results (Arif et al., 2020).

Table 4-5: Removal percentages of water pollutants obtained for the different scenarios

Pollutant parameters	Scenario 0	Scenario 1	Scenario 2	Scenario 3
COD	96.2	96.2	97	95.8
BOD	96	96	98.5	98.3
TSS	98.9	98.9	99.3	99.5
TN	90.4	90.4	86.3	91.3
TP	95.2	95.2	96.9	93.5

4.3.2 Environmental costs

LCA results were economically monetized to determine the external environmental costs. With ReCiPe 2016 indicators, the financial assessment outcomes for S0, S1, S2, and S3 were computed to be $0.159\ \$/m^3$, $0.117\ \$/m^3$, $0.171\ \$/m^3$, and $0.147\ \$/m^3$, respectively. Figure 4-13 shows the a) Monetized environmental impacts derived from ReCiPe2016 impact categories and b) combined costs both expressed in \$ per 1 m^3 of treated wastewater. The expenses incurred by society for all treatment scenarios shown in Figure 4-13a validated that utilizing an energy recovery replacement greatly decreased the financial consequences of the CO_2eq footprint as S1 resulted in a lower societal carbon cost than S0, S2, and S3. In addition, S1 could generate high-quality effluent that satisfied the effluent regulations.

Generally, scenarios that had observed reduced energy consumption had lower overall societal costs of carbon whereas scenarios that saw higher energy consumption than the main model had a higher societal cost. These findings supported the conclusion drawn earlier about the dominant contribution of energy to the environmental impacts of WWTPs. In Figure 4-13b, economic and environmental costs were integrated. This was to give a final cost that would account for the economic and environmental costs of a process.

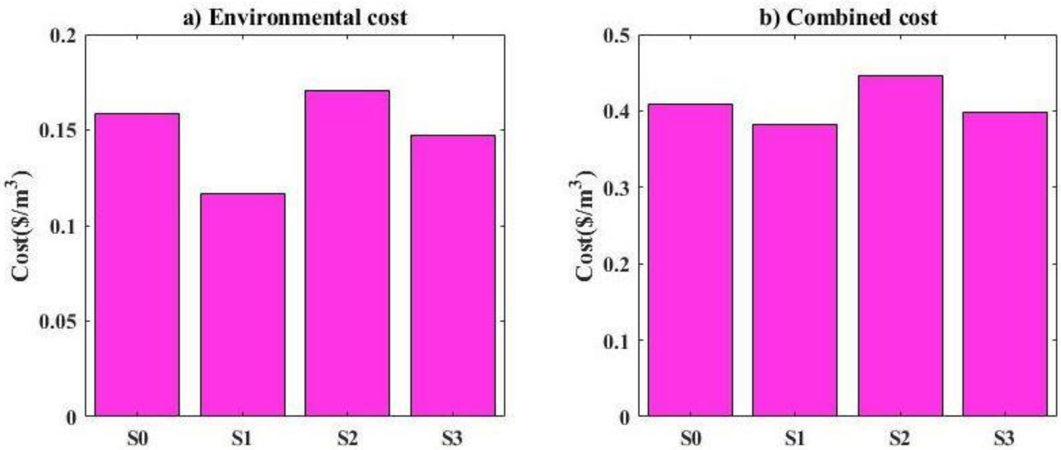


Figure 4-13: a) Process external cost values and b) combined economic and environmental costs in $\$/\mathrm{m}^3$ for all scenarios

Results showed that S0, S1, and S3 have almost comparable combined costs detailed as follows: $\mathrm{S2}(0.447\ \$/\mathrm{m}^3)$ > $\mathrm{S0}(0.408\ \$/\mathrm{m}^3)$ > $\mathrm{S3}\ (0.398\ \$/\mathrm{m}^3)$ > $\mathrm{S1}(0.381\ \$/\mathrm{m}^3)$. In addition to having a smaller footprint, S1 system had a low combined running cost. With the net expenses of anaerobic digestion, the profits of scenario 1 were higher than the costs added which made it the cheapest scenario at the same time. If only environmental costs were taken into account, S1 would be the best option. On the other hand, if only financial advantages were taken into account, S0 could be the best one for decision-makers. If both were considered, S1 would be regarded as the most cost-effective solution based on the findings as it had the lowest value of combined economic and environmental $\mathrm{cost}/\mathrm{m}^3$. Ultimately, due to its favorable biogas yields and energy savings, S1 provided the most competitive alternative.

4.4 Limitations and future work

A notable limitation of this study was that the construction and demolition phases of the plant were not considered in the LCA boundaries in addition to other processes that were not considered. As a result, LCA underestimated the system's life cycle emissions. It was critical to keep in mind that LCA had limitations as it depends on estimations. In addition, as estimated NPVs were based on general cost assumptions, they were not absolutely accurate. However, they helped in finding the most cost-effective approach by comparison. It was also vital to keep in mind that, with better and more accurate data, all model parameters may be further optimized. Finally, any computer-based research might be prone to inaccuracy due to the limited ability to perfectly reproduce the behavior of the real model.

Future studies need to address the construction and end-of-life phases of the plant. Future work also needs to further validate the model’s performance by improving the level of detail of the input data in order to reflect the unique situation in Lebanon. Moreover, future studies should focus on developing and implementing country-specific monetization factors for external costs.

Chapter Five

Conclusion

This study assessed the environmental and economic performance of the ZWWT facility in Lebanon and three distinct scenarios were proposed. A successful calibration and validation of the plant were performed on GPS-X. The model was dynamically calibrated and validated using the data acquired for two different months. A series of LCAs were performed to evaluate the environmental impacts of the scenarios considered using SimaPro. The LCA approach was based on the ReCiPe 2016 midpoint (H) and endpoint (H/A) approaches. WWTPs were evaluated based on 1 m^3 of treated wastewater functional unit. The findings of this study indicated that energy use is a common characteristic of environmental impacts in all scenarios. When the electricity provided by normal processes was replaced by the power generated by the anaerobic digestion process, the system demonstrated lower environmental impacts. According to the normalized results, freshwater eutrophication and human carcinogenic toxicity had the greatest environmental importance. According to endpoint characterized results, S1 showed to be the most environmentally friendly alternative having an impact on GW of 0.521 kg CO_2 eq. Regarding the total economical $cost/m^3$, all scenarios had nearly comparable numbers, however, S2 was found to be the most expensive ($0.276\ \$/m^3$) while S0 was the most cost-effective ($0.249\ \$/m^3$). Additionally, it was discovered that high removal efficiencies induce higher economic expenses. Results for external expenses showed that S1 was expectedly in the first place ($0.117\ \$/m^3$). In addition, the FSB had better environmental costs compared to EA and the baseline model. It has been also shown that if the two costs were considered together, S1 may be a more cost-effective and ecologically friendly alternative. Based on these findings, the WWTP's environmental performance could be enhanced by reducing energy consumption. The results of this study also highlighted the significance of using a life cycle perspective as better effluent obtained did not ensure better environmental performance. Finally, this study validated the significant value of environmental and economical assessment in discovering other possibilities that had traditionally been disregarded in developing countries. Despite the unknowns, monetizing LCA results could provide valuable extra data and insights for environmentally responsible WWT. This case study could be used in the future for the more environmentally friendly WWTPs and better environmental judgments. It could also help decision-makers to have a comprehensive knowledge of the economic and environmental conditions of different WWTPs designs.

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Appendices

Appendix A: LandGEM calculations

According to RTI, (2010), the first-order decay model for CH_4 generation is as follows (adapted from IPCC, 2006 and U.S. EPA, 2008):

$$A = \left[\sum_{x=S}^{T-1}\left\{W_x L_x'\left(e^{-k(T-x-1)} - e^{-k(T-x)}\right)\right\}\right]$$

Where:

$A = CH_4$ generation (Mg/yr)

$x =$ Year in which waste was disposed

$S =$ Start year of inventory calculation

$T =$ Inventory year for which emissions are calculated

$W_x =$ the quantity of waste disposed of at the solid waste disposal site (Mg)

$L' = CH_4$ generation potential (Mg CH_4/Mg waste)

$= MCF \times DOC \times DOC_F \times F \times \frac{16}{12}$ [IPCC nomenclature]

$= L_0 \times \frac{16}{0.02367} \times 10^{-6}$

$L_0 = CH_4$ generation potential ($m^3 CH_4$/Mg waste) [AP-42 nomenclature]

$MCF = CH_4$ correction factor (fraction), typically 1 for managed landfills

$DOC =$ degradable organic carbon [fraction (Mg C in waste/Mg waste)]

$DOC_F =$ fraction of DOC decomposed (fraction), generally assumed to be 0.5

$F =$ fraction by volume of CH_4 in landfill gas, generally assumed to be 0.5

$k =$ decay rate constant (yr^{-1})

Table A-1 and Table A-2 provide default values that comply with the GHG Reporting Rule for municipal solid waste (MSW) landfills (40 CFR part 98, subpart HH) and industrial waste landfills (40 CFR part 98, subpart TT).

L_0 can be calculated from the DOC using the equation below; this calculation assumes that the default values for MCF, DOC_F, and F apply.

$$L_0 = 493 \times DOC$$

Where: $L_0 = CH_4$ generation potential ($m^3 CH_4/Mg$ waste)

$DOC =$ Degradable organic carbon [fraction (Mg C in waste/Mg waste)]

This calculation procedure only accounts for CH_4 and CO_2 generation without soil oxidation. Therefore, it is commonly estimated that 10% of the produced CH_4 is oxidized to CO_2 near the landfill's surface (U.S. EPA, 2010a), implying that CH_4 emissions account for 90% of CH_4 generation with no gas collection (RTI, 2010).

Table A-1: Global Warming Potentials for 100-Year Time Horizon[a]

Pollutant Name	Chemical Formula	CAS No.	Global Warming Potential
Carbon dioxide	CO_2	124-38-9	1
Methane	CH_4	74-82-8	21
Nitrous oxide	N_2O	10024-97-2	310

[a] Source: 40 CFR part 98 subpart A, Table A-1. Note the GWP values presented in this table are subject to change if changes occur by rulemaking or notice to Table A-1 in the Reporting Rule. GWP values can be updated in the future; however, only a rulemaking on the Reporting Rule would supersede the values in this table.

Note: Reprinted from "Greenhouse Gas Emissions Estimation Methodologies for Biogenic Emissions from Selected Source Categories: Solid Waste Disposal Wastewater Treatment Ethanol Fermentation" by RTI, 2010, *U.S. Environmental Protection Agency Sector Policies and Programs Division Measurement Policy Group*, p. 1-3 (https://www3.epa.gov/ttnchie1/efpac/ghg/GHG_Biogenic_Report_draft_Dec1410.pdf). In the public domain.

For landfills without gas collection systems, CO_2 emissions can be calculated from the CH_4 generation as follows:

$$B = A \times \left(\frac{1 - F}{F} + OX\right) \times \frac{44}{16}$$

Where:

$B = CO_2$ emissions $(Mg\ CO_2/yr)$

$A = CH_4$ generation for $(Mg\ CH_4/yr)$

$F =$ Fraction by volume of CH_4 in landfill gas, generally assumed to be 0.5

$OX =$ Soil oxidation fraction, typically 0.1 (fraction)

$44 =$ Molecular weight of CO_2 $(Kg/Kg - mol)$

$16 =$ Molecular weight of CH_4 $(Kg/Kg - mol)$

Table A-2: Additional Landfill Model Defaults

Parameter	Parameter Description	Parameter Value
MCF	Methane correction factor (dimensionless)	1
DOC_F	Fraction of DOC degraded	0.5
F	Fraction CH_4 in generated gas	0.5
OX	Soil oxidation factor (dimensionless) [IPCC model only, in the Recovery_OX tab]	0.10
Delay Time	Time (in months) prior to the start of anaerobic decay [IPCC model only]	6

Note: Reprinted from "Greenhouse Gas Emissions Estimation Methodologies for Biogenic Emissions from Selected Source Categories: Solid Waste Disposal Wastewater Treatment Ethanol Fermentation" by RTI, 2010, *U.S. Environmental Protection Agency Sector Policies and Programs Division Measurement Policy Group*, p. 2-4 (https://www3.epa.gov/ttnchie1/efpac/ghg/GHG_Biogenic_Report_draft_Dec1410.pdf). In the public domain.

Table A-3: Recommended DOC and Decay Rate Values for Landfills[a]

Waste Model/Waste Type	DOC (weight fraction, wet basis)	k [dry climate[b]] (yr^{-1})	k [moderate climate[b]] (yr^{-1})	k [wet climate[b]] (yr^{-1})
MSW Landfills-Bulk Waste Option				
All waste materials	0.2028	0.02	0.038	0.057
MSW Landfills-Bulk MSW Option				
Bulk MSW	0.30	0.02	0.038	0.057
Construction and demolition waste	0.08	0.02	0.03	0.04
Inert waste (glass, metal, plastic)	0.0	0.0	0.0	0.0
MSW Landfills-Waste-Specific Option				
Food waste	0.15	0.06[c]	–[c]	0.185[c]
Garden waste	0.20	0.05[c]	–[c]	0.10[c]
Paper waste	0.40	0.04[c]	–[c]	0.06[c]
Wood and straw waste	0.43	0.02[c]	–[c]	0.03[c]
Textile waste	0.24	0.04[c]	–[c]	0.06[c]
Diapers	0.24	0.05[c]	–[c]	0.10[c]
Sewage sludge	0.05	0.06[c]	–[c]	0.185[c]
Inert waste (glass, metal, plastic)	0.0	0.0	0.0	0.0
Industrial Waste Landfills				
Food processing industry	0.22	0.06	0.12	0.18
Pulp and paper industry	0.20	0.02	0.03	0.04
Wood and wood products	0.43	0.02	0.03	0.04
Construction and demolition waste	0.08	0.02	0.03	0.04
Inert waste (glass, metal, plastic)	0	0	0	0
Other industrial solid waste (not otherwise listed)	0.20	0.02	0.04	0.06

[a] Taken from 40CFR part 98, subparts HH and TT (with expected corrections for DOC for construction and demolition waste in subpart TT).
[b] The applicable climate classification is determined based on the annual rainfall plus the recirculated leachate application rate. Recirculated leachate application rate (in inches/year) is the total volume of leachate recirculated and applied to the landfill divided by the area of the portion of the landfill containing waste (with appropriate unit conversions). Unless otherwise specified, the classifications are as follows:
– Dry climate = precipitation plus recirculated leachate less than 20 inches/year
– Moderate climate = precipitation plus recirculated leachate from 20 to 40 inches/year (inclusive)
– Wet climate = precipitation plus recirculated leachate greater than 40 inches/year.
[c] The climate is considered dry when the potential evapotranspiration rate exceeds the mean annual precipitation rate plus recirculated leachate. The climate is considered wet when the potential evapotranspiration rate does not exceed the mean annual precipitation rate plus recirculated leachate.

Note: Reprinted from "Greenhouse Gas Emissions Estimation Methodologies for Biogenic Emissions from Selected Source Categories: Solid Waste Disposal Wastewater Treatment Ethanol Fermentation" by RTI, 2010, *U.S. Environmental Protection Agency Sector Policies and Programs Division Measurement Policy Group*, p. 2-3 (https://www3.epa.gov/ttnchie1/efpac/ghg/GHG_Biogenic_Report_draft_Dec1410.pdf). In the public domain.

Table A-4: Sludge landfilling emissions calculation results

Variable	Unit	Value
S	year	2020
T	year	2021
W_x	ton/year	21,468
DOC	[fraction (Mg C in waste/Mg waste)]	0.05
K	$year^{-1}$	0.06
L_0	$m^3 CH_4$/Mg waste	24.65
A	Mg CH_4/yr	20.62
CH_4 emissions	Mg CH_4/yr	18.56
F	−	0.5
OX	−	0.1
B	Mg CO_2/yr	62.38

Appendix B: Anaerobic digestion design

B.1 Mass balance

$\text{Mass TSS}_{\text{in}} = \text{Mass TSS}_{\text{out}} + \text{Mass TSS}_{\text{disposed of}}$

B.2 Anaerobic digestion design

The anaerobic digestion was designed according to Qasim & Zhu, (2018) as follows:

A) Digester capacity from different methods:

1. $V = \theta Q$

   Where V = the required digester volume without recycle (m^3)

   θ = hydraulic retention time (days)

   Q = flow entering anaerobic digestion (m^3/d)

2. $V = \frac{W_{VSS}}{VSL}$; $W_{VSS} = f_{VSS,FS} \times W_{TSS}$

   Where V = the required digester volume without recycling (m^3)

   W_{VSS} = total volatile solids reaching the digester (kg VSS/d)

   $f_{VSS,FS}$ = VSS/TSS ratio or weight fraction of volatile suspended solids in the digester feed on a dry weight basis, kg VSS/kg TSS or dimensionless

   W_{TSS} = weight of total solids (kg TSS/d)

   VSL = volumetric solids loading ($\text{kg VSS/m}^3.\text{d}$)

B) Dimensions of an anaerobic digester:

1. Develop the digester geometry.

   a. Provide a cylindrical digester with a bottom cone. The floor of the digester is sloped at 1 vertical (V) to 3 horizontal (H).

   b. Provide depth (H_{grit}) for the grit accumulation in the bottom of the cone.

c. Provide a water depth for the scum blanket (H_{scum}) below the maximum sludge surface level.

d. Provide an active side water depth of $H_{act,swd}$ below the scum layer.

e. Provide an additional clearance space (H_{space}) between the floating cover and the maximum sludge surface level.

f. Side water depth of the digester, $H_{swd} = H_{act,swd} + H_{scum}$ below the maximum sludge surface level.

g. Total height of the sidewall of the digester cylindrical portion, $H_{sw} = H_{swd} + H_{space}$.

2. $A = \frac{V}{H_{act,swd}}$; $D = \sqrt{\frac{4}{\pi} \times A}$

   Where A = the surface area of the digester (m^2)

   V = volume of digester (m^3)

   D = required digester diameter (m)

   $H_{act,swd}$ = active side water depth (SWD) (m)

3. $H_{cone} = \frac{V}{H} \times \frac{D}{2}$; $H_{total} = H_{sw} + H_{cone}$

   Where H_{cone} = the depth of the bottom cone (m)

   V = vertical slope

   D = required digester diameter (m)

   H = horizontal slope (m)

   H_{total} = total height of the digester (m)

   H_{sw} = height of sidewall (m)

4. $D_{grit} = 2 \times \frac{H}{V} \times H_{grit}$; $V_{grit} = \frac{\pi}{12}\left(D_{grit}\right)^2 H_{grit}$

$V_{cone} = \frac{\pi}{12}(D)^2 H_{cone}$; $V_{act,cone} = V_{cone} - V_{grit}$

$V_{act,cyl} = \frac{\pi}{4}(D)^2 H_{act,swd}$; $V_{act} = V_{act,cyl} + V_{act,cone}$

Where D_{grit} = cone-diameter for grit accumulation (m)

H_{cone} = the depth of the bottom cone (m)

V = vertical slope

D = required digester diameter (m)

H = horizontal slope

H_{grit} = depth of grit (m)

V_{grit} = volume provided for grit accumulation (m^3)

V_{cone} = total volume of the cone (m^3)

$V_{act,cone}$ = active volume available in the cone (m^3)

$V_{act,cyl}$ = active volume available in the cylindrical portion (m^3)

V_{act} = total active volume available in the digester (m^3)

C) Heating requirements for digester feed and radiation losses:

1. $W_S = \rho_s \times Q_s$; $H_{r,s} = C_{sh,s} W_S (T_d - T_s)$; $H_{r,s,adj} = 1.45\ H_{r,s}$

Where W_s = mass of feed wet sludge (kg/d)

ρ_s = density of sludge flow entering anaerobic digestion (kg/m^3)

Q_s = sludge flow entering anaerobic digestion (m^3/d)

$H_{r,s}$ = heat requirement for heating the sludge (J/d)

$C_{sh,s}$ = specific heat of the sludge, $(J/kg.°C)$. It is a common practice to assume that the specific heats of sludge and water are the same, $C_{sh,s} = C_{sh,w} = 4200\ J/kg.°C$

T_d = average digester operating temperature $(°C)$

T_s = average temperature of the feed sludge (°C)

$H_{r,s,adj}$ = adjusted heat requirement for entire incoming feed sludge (J/d)

2. The heat loss occurs from the roof, sidewalls exposed to air (sw), buried side wall (bur), and bottom slab as follows:

$$A_{roof} = \pi \times \left(\left(\frac{D}{2}\right)^2 + (H_{roof})^2\right); \; A_{sw,exp} = \pi D H_{sw,exp}; \; A_{sw,bur} = \pi D H_{sw,bur}$$

$$L_{slab} = \sqrt{\left(\frac{D}{2}\right)^2 + (H_{slab})^2}; \; A_{slab} = \pi \times \frac{D}{2} \times L_{slab}$$

Where A_{roof} = surface area of the domed roof (m^2)

D = diameter of the digester (m)

H_{roof} = height of roof (m)

$A_{sw,exp}$ = surface area of the exposed sidewall (m^2)

$H_{sw,exp}$ = height of the exposed sidewall (m)

$A_{sw,bur}$ = surface area of the buried side wall (m^2)

$H_{sw,bur}$ = height of buried side wall (m)

A_{slab} = Surface area of the bottom slab (m^2)

L_{slab} = Lateral length of the slab (m)

H_{slab} = height of the cone (m)

3. $H_{l,sf,roof} = U_{sf,roof} A_{roof} (T_d - T_{air}); \; H_{l,sf,sw,exp} = U_{sf,sw,exp} A_{sw,exp} (T_d - T_{air})$

$$H_{l,sf,sw,bur} = U_{sf,sw,bur} A_{sw,bur} (T_d - T_{earth,sw})$$

$$H_{l,sf,slab} = U_{sf,slab} A_{sw,slab} (T_d - T_{earth,slab})$$

$$H_{l,sf,d} = H_{l,sf,roof} + H_{l,sf,sw,exp} + H_{l,sf,sw,bur} + H_{l,sf,slab}; \; H_{l,sf,adj} = 1.45 \times H_{l,sf,d}$$

Where $H_{l,sf}$ = heat loss from through the surface (J/d)

U_{sf} = overall heat transfer coefficient, ($J/s.m^2.°C$)

A = surface area through which the heat loss occurs (m^2)

T_d = average digester operating temperature (°C)

T_{air} = average ambient temperature outside of the surface (°C)

T_{earth} = average earth temperature of the buried surface (°C)

$H_{l,sf,d}$ = total heat loss from digester (J/d)

$H_{l,sf,adj}$ = adjusted total heat loss from digester (J/d)

4. $H_r = H_{r,s,adj} + H_{l,sf,adj}$

   Where H_r = total heat required to heat the incoming sludge and compensate for the heat losses (J/d)

   $H_{r,s,adj}$ = adjusted total heat requirement for heating sludge (J/d)

   $H_{l,sf,adj}$ = adjusted total heat loss from digester (J/d)

5. $Q_{r,biogas} = \frac{H_r}{\text{Heat value of biogas} \times \text{Heating efficiency}}$

   Where $Q_{r,biogas}$ = the digester gas required to meet the heating demand (m^3/d)

   H_r = total heat required to heat the incoming sludge and compensate for heat losses (J/d)

D) Design of heat exchanger for digester heating:

1. The hot water recirculation rate through the external heat exchanger:

   $h_{hw} = C_{sh,w}(T_{hw,in} - T_{hw,out})$; $H_{hex} = H_r$; $Q_{hw} = \frac{H_{hex}}{h_{hw}E_{hex}}$

   Where h_{hw} = total heat supplied by 1 kg recirculated hot water via the jacket (J/kg water)

   $C_{sh,w}$ = specific heat of water (J/kg °C)

   $T_{hw,in}$ = temperature of hot water entering the jacket (°C)

$T_{hw,out}$ = temperature of hot water leaving the jacket (°C)

H_{hex} = heat output requirement for heat exchanger (J/d)

H_r = total heat required to heat the incoming sludge and compensate for heat losses (J/d)

E_{hex} = heat transfer efficiency

Q_{hw} = hot water recirculation rate through the heat exchanger (m^3/d)

E) Digester biogas generation from different methods:

1. $W_{VSSbd,FS} = f_{VSSbd,FS} \times W_{TSS,FS}$; $\Delta S_0 = \frac{COD}{VSS_{bd}}$ ratio $\times W_{VSSbd,FS}$

Where $W_{VSSbd,FS}$ = mass flow of biodegradable VSS (VSS_{bd}) in the feed sludge (kg VSS/d)

$f_{VSSbd,FS} = VSS_{bd}/TSS$ ratio or weight fraction of biodegradable volatile suspended solids in the digester feed on a dry weight basis, (kg VSS_{bd}/kg TSS) or dimensionless

$W_{TSS,FS}$ = mass flow of TSS in the digester feed sludge, (kg TSS/d)

ΔS_0 = COD exerted in feed sludge (kg COD/d)

$\frac{COD}{VSS_{bd}}$ ratio = weight fraction, (kg COD/kg VSS_{bd})

$Y_{obs} = \frac{Y}{1+k_d\theta_c}$; $P_x = Y_{obs}E\Delta S_0$; $\Delta S_M = E\Delta S_0 - 1.42P_x$

$Q_M = f_v\Delta S_M$; $Q_{biogas} = \frac{Q_M}{f_{CH_4}}$

Where Y_{obs} = observed biomass yield (kg VSS/ kg COD) utilized

Y = biomass yield coefficient (mg VSS/ mg COD) utilized

k_d = specific endogenous decay coefficient (d^{-1})

θ_c = solids retention time (SRT) (d)

P_x = net mass of biosolids (VSS) produced (kg VSS/ d)

E = COD utilization efficiency

ΔS_0 = COD exerted in feed sludge (kg COD/d)

ΔS_M = COD consumption or stabilization rate (kg COD/ d)

Q_M = methane production ($m^3 CH_4$/ d)

f_v = volumetric conversion factor for the amount of methane produced from the COD consumption ($m^3 CH_4$/ kg COD)

f_{CH_4} = CH_4 content by volume in the biogas (%)

Q_{biogas} = biogas production (m^3biogas/ d)

2. $W_{VSS,FS} = \frac{VSS}{TSS}\text{ratio} \times W_{TSS,FS};\ \ Q_{biogas} = R_L \times W_{VSS,FS}$

Where $W_{VSS,FS}$ = mass of VSS in feed sludge (kg VSS/d)

$\frac{VSS}{TSS}$ ratio = weight fraction, (kg VSS/kg TSS)

$W_{TSS,FS}$ = mass flow of TSS in the digester feed sludge, (kg TSS/d)

Q_{biogas} = biogas production (m^3biogas/ d)

R_L = digester gas production rate (m^3biogas/ kg VSS loading)

3. $VSR = 13.7\ln(\theta_c) + 18.9;\ \ \Delta W_{VSS,reduced} = \frac{VSR}{100\%} \times W_{VSS,FS}$

$Q_{biogas} = R_R \times \Delta W_{VSS,reduced}$

Where VSR = volatile solids reduction (%)

θ_c = solids retention time (SRT) (d)

R_R = digester gas production rate (m^3biogas/ kg VSS reduced)

$\Delta W_{VSS,reduced}$ = mass of VSS destroyed in the digester (kg VSS/d)

Q_{biogas} = biogas production (m^3biogas/ d)

Table B-1: Anaerobic digestion design calculation results

Variable	Unit	Value
Digester capacity from different methods		
1. Q	m^3/d	150
θ	d	15
Volume V	m^3	2250
2. VSL	$kg\ VSS/m^3.d$	4.6
W_{TSS}	kg TSS/d	7624
$f_{VSS,FS}$	kg VSS/kg TSS	0.75
W_{VSS}	kg VSS/d	5718
Volume V	m^3	1250
Dimensions of an anaerobic digester		
Vertical slope V	−	1
Horizontal slope H	−	3
H_{grit}	m	1
H_{scum}	m	0.6
$H_{act,swd}$	m	8.5
H_{space}	m	0.6
H_{swd}	m	9.1
H_{sw}	m	9.7
A	m^2	147
D	m	13.7
H_{cone}	m	2.3
H_{total}	m	12
D_{grit}	m	6
V_{grit}	m^3	9.4
V_{cone}	m^3	113
$V_{act,cone}$	m^3	104
$V_{act,cyl}$	m^3	1250
V_{act}	m^3	1354
Heating requirements:		
Digester feed & radiation losses		
ρ_s	kg/m^3	1010
Q_s	m^3/d	150
W_s	kg/d	151,500
T_d	°C	35
T_s	°C	12
$C_{sh,s}$	J/kg.°C	4200
$H_{r,s}$	J/d	1.46E + 10
$H_{r,s,adj}$	J/d	2.12E + 10
H_{roof}	m	0.5
A_{roof}	m^2	148
$H_{sw,exp}$	m	4.7
$A_{sw,exp}$	m^2	202
$H_{sw,bur}$	m	5
$A_{sw,bur}$	m^2	215

Table B-1: Anaerobic digestion design calculation results *(continued)*

H_{slab}	m	2.3
L_{slab}	m	7.23
A_{slab}	m^2	156
$U_{sf,roof}$	$J/s. m^2. °C$	2.84
T_{air}	$°C$	-2
$H_{l,sf,roof}$	J/d	$1.17E + 09$
$U_{sf,sw,exp}$	$J/s. m^2. °C$	1.99
$H_{l,sf,sw,exp}$	J/d	$1.29E + 09$
$U_{sf,sw,bur}$	$J/s. m^2. °C$	1.02
$T_{earth,sw}$	$°C$	3
$H_{l,sf,sw,bur}$	J/d	$6.06E + 08$
$U_{sf,slab}$	$J/s. m^2. °C$	0.68
$H_{l,sf,slab}$	J/d	$2.66E + 08$
$H_{l,sf,d}$	J/d	$3.33E + 09$
$H_{l,sf,adj}$	J/d	$4.83E + 09$
H_r	J/d	$2.6E + 10$
H_r	kW	301
Heat value of biogas	MJ/m^3	24
Heating efficiency	$-$	0.6
$Q_{r,biogas}$	m^3/d	1806
Design of heat exchanger for digester heating		
$T_{hw,in}$	$°C$	65
$T_{hw,out}$	$°C$	45
h_{hw}	J/kg water	$8.4E + 04$
H_{hex}	J/d	$2.6E + 10$
E_{hex}	$-$	0.8
Q_{hw}	m^3/d	387
Digester biogas generation from different methods		
1. f_{VSSbd}	kg VSS_{bd}/kg TSS	0.56
$W_{TSS,FS}$	kg TSS/d	7624
$W_{VSSbd.FS}$	kg VSS/d	4270
$\frac{COD}{VSS_{bd}}$ ratio	kg COD/ kg VSS_{bd}	1.42
ΔS_0	kg COD/d	6063
Y	mg VSS/mg COD	0.08
k_d	d^{-1}	0.03
θ_c	d	15
Y_{obs}	mg VSS/mg COD	0.055
E	$-$	0.75
P_x	kg VSS/d	250
ΔS_M	kg COD/d	4192
f_v	$m^3 CH_4$/kg COD	0.35
Q_M	$m^3 CH_4/d$	1467
f_{CH_4}	%	65
Q_{biogas}	m^3biogas/d	2257

Table B-1: Anaerobic digestion design calculation results *(continued)*

2. $\frac{VSS}{TSS}$ ratio	kg VSS/kg TSS	0.75
$W_{VSS,FS}$	kg VSS/d	5719
R_L	m^3biogas/kg VSS loading	0.55
Q_{biogas}	m^3biogas/d	3145
3. VSR	%	56
$\Delta W_{VSS,reduced}$	kg VSS/d	3203
R_L	m^3biogas/kg VSS reduced	0.85
Q_{biogas}	m^3biogas/d	2722

B.3 The combustion reaction of methane stoichiometry

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O$$

Initial conditions $\quad n_{0_{CH_4}} \quad n_{0_{O_2}} \quad n_{0_{CO_2}}$

Final conditions $\quad 0 \quad n_{f_{O_2}} \quad n_{f_{CO_2}}$

$Q_{biogas} = Q_{CO_2} + Q_{CH_4}$

$n_{biogas} = n_{0_{CH_4}} + n_{0_{CO_2}} = \frac{V_{biogas}}{V_m}$

$n_{f_{CO_2}} = n_{reacted_{CH_4}} + n_{0_{CO_2}} = n_{0_{CH_4}} + n_{0_{CO_2}} = n_{biogas}$ (CH_4 limiting agent)

$m_{f_{CO_2}} = n_{f_{CO_2}} \times M_{CO2}$; where Q = flow (m^3/d)

n = number of moles (mol)

n_0 = number of moles at initial conditions (mol)

$n_{reacted}$ = number of moles that reacted (mol)

n_f = number of moles at final conditions (mol)

V_{biogas} = generated biogas volume (m^3)

V_m = molar volume at standard temperature and pressure (STP)

m_f = final mass of the substance (g)

M = molar mass of the substance (g/mol)

Table B-2: Mass of CO_2 calculation results

Variable	Unit	Value
Q_{biogas}	$m^3 biogas/d$	2800
V_{biogas}	m^3	2800
V_m	L/mol	22.4
n_{biogas}	mol	125,000
$n_{f_{CO2}}$	mol	125,000
M_{CO2}	g/mol	44
$m_{f_{CO_2}}$	kg	5500

Appendix C: Surplus sludge production

C.1 Surplus sludge production

The surplus activated sludge production was calculated according to Arceivala & Asolekar, (2012) as follows:

$t = \frac{V}{Q}$; Net VSS produced $= \frac{xV}{\theta_c}$

In terms of suspended solids (SS):

$$\text{Net SS production} = \frac{\text{Net VSS produced}}{(\text{VSS}/\text{SS})\ \text{ratio}}$$

Where V = volume (m^3)

θ_c = solids retention time (SRT) (days)

x = mixed liquor volatile suspended solids (MLVSS) (mg/L)

Net VSS produced = Net volatile suspended solids produced (kg/d)

Q = average influent flow (m^3/d)

t = hydraulic retention time (HRT) (hours)

Net SS production = Net suspended solids produced (kg/d)

$\frac{VSS}{SS}$ = ratio ranging between $(0.5 - 0.8)$ for extended aeration

If these SS are removed as underflow from the final settling tank with a solid concentration of 1%, and if the specific gravity of sludge is assumed as nearly 1.00:

$$\text{Liquid sludge to be removed} = \text{Net SS production} \times \frac{100}{1}$$

Assuming 95% water removal from dewatering:

$$\text{Water removed from dewatering} = 0.95 \times 0.99 \times \text{liquid sludge to be removed}$$

$$\text{Water remaining after dewatering} = 0.05 \times 0.99 \times \text{liquid sludge to be removed}$$

$$\text{Sludge to be landfilled} = \text{water remaining after dewatering} + \text{net SS production}$$

Table C-1: Mass of sludge to be landfilled calculation results

Variable	Unit	Value
HRT	hours	22
Q	m^3/d	25,081
V	m^3	23,000
SRT	days	20
MLVSS	mg/L	2600
Net VSS production	kg/d	3000
$\frac{VSS}{SS}$ ratio	−	0.5
Net SS production	kg/d	6000
Liquid sludge to be removed	m^3/d	600
Water removed from dewatering	m^3/d	564.3
Water remaining after dewatering	m^3/d	29.7
Sludge to be landfilled	kg/d	35,700

C.2 Sludge landfilling emissions calculations

Sludge landfilling emissions are calculated by referring to Appendix A:

Table C-2: Sludge landfilling emissions calculation results

Variable	Unit	Value
S	year	2020
T	year	2021
W_x	ton/year	3036
DOC	[fraction (Mg C in waste/Mg waste)]	0.05
K	$year^{-1}$	0.06
L_0	$m^3 CH_4$/Mg waste	24.65
A	Mg CH_4/yr	12.52
CH_4 emissions	Mg CH_4/yr	11.27
F	−	0.5
OX	−	0.1
B	Mg CO_2/yr	37.87

Appendix D: Supplementary results

PC: Power consumption; BT: Biological treatment; CC: Chemical consumption; SL: Sludge landfilling; ED: Effluent discharge; AD: Anaerobic digestion; HH: Human health; TECO: terrestrial ecosystems; FECO: Freshwater ecosystems; AE: Aquatic ecosystems.

Table D-1: Characzterized midpoint results of S0

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW	kg CO2 eq	6.78E-01	2.35E-01	3.62E-01	1.22E-02	6.80E-02	0.00E+00
SOD	kg CFC11 eq	8.63E-06	1.81E-07	8.45E-06	2.61E-09	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	2.38E-03	2.05E-03	0.00E+00	3.24E-04	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	8.24E-04	8.02E-04	0.00E+00	2.17E-05	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	5.11E-04	4.92E-04	0.00E+00	1.88E-05	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	8.32E-04	8.10E-04	0.00E+00	2.21E-05	0.00E+00	0.00E+00
TA	kg SO_2 eq	1.62E-03	1.57E-03	0.00E+00	4.85E-05	0.00E+00	0.00E+00
FE	kg P eq	6.87E-04	2.87E-05	0.00E+00	4.90E-06	0.00E+00	6.54E-04
ME	kg N eq	8.57E-04	3.13E-07	0.00E+00	1.27E-06	0.00E+00	8.55E-04
TE	kg 1,4 − DCB	9.97E-01	8.97E-01	0.00E+00	1.00E-01	3.14E-19	0.00E+00
FEC	kg 1,4 − DCB	1.01E-02	1.62E-03	0.00E+00	9.31E-04	7.53E-03	0.00E+00
MEC	kg 1,4 − DCB	9.42E-03	2.66E-03	0.00E+00	1.24E-03	5.52E-03	0.00E+00
HCT	kg 1,4 − DCB	9.11E-03	1.88E-03	0.00E+00	1.85E-03	5.37E-03	0.00E+00
HNCT	kg 1,4 − DCB	1.48E-01	2.39E-02	0.00E+00	1.96E-02	1.05E-01	0.00E+00
LU	m^2a crop eq	3.12E-03	2.86E-03	0.00E+00	2.59E-04	0.00E+00	0.00E+00
MRS	kg Cu eq	2.28E-04	7.60E-05	0.00E+00	1.52E-04	0.00E+00	0.00E+00
FRS	kg oil eq	6.89E-02	6.65E-02	0.00E+00	2.40E-03	0.00E+00	0.00E+00
WC	m^3	6.74E-04	5.97E-04	0.00E+00	7.73E-05	0.00E+00	0.00E+00

Table D-2: Characterized midpoint results of S1

Impact category	Unit	Total	PC	BT	CC	ED	AD
GW	kg CO2 eq	5.21E-01	1.53E-01	3.62E-01	5.72E-03	0.00E+00	0.00E+00
SOD	kg CFC11 eq	8.57E-06	1.17E-07	8.45E-06	1.84E-09	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	1.61E-03	1.34E-03	0.00E+00	2.75E-04	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	5.39E-04	5.22E-04	0.00E+00	1.68E-05	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	3.37E-04	3.20E-04	0.00E+00	1.63E-05	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	5.44E-04	5.27E-04	0.00E+00	1.70E-05	0.00E+00	0.00E+00
TA	kg SO_2 eq	1.06E-03	1.02E-03	0.00E+00	4.26E-05	0.00E+00	0.00E+00
FE	kg P eq	6.77E-04	1.87E-05	0.00E+00	4.08E-06	6.54E-04	0.00E+00
ME	kg N eq	8.57E-04	2.03E-07	0.00E+00	1.22E-06	8.55E-04	0.00E+00
TE	kg 1,4 − DCB	6.71E-01	5.83E-01	0.00E+00	8.78E-02	0.00E+00	0.00E+00
FEC	kg 1,4 − DCB	1.97E-03	1.05E-03	0.00E+00	9.14E-04	0.00E+00	0.00E+00
MEC	kg 1,4 − DCB	2.94E-03	1.73E-03	0.00E+00	1.21E-03	0.00E+00	0.00E+00
HCT	kg 1,4 − DCB	3.04E-03	1.23E-03	0.00E+00	1.81E-03	0.00E+00	0.00E+00
HNCT	kg 1,4 − DCB	3.45E-02	1.55E-02	0.00E+00	1.90E-02	0.00E+00	0.00E+00
LU	m^2a crop eq	2.01E-03	1.86E-03	0.00E+00	1.55E-04	0.00E+00	0.00E+00
MRS	kg Cu eq	2.00E-04	4.94E-05	0.00E+00	1.51E-04	0.00E+00	0.00E+00
FRS	kg oil eq	4.50E-02	4.33E-02	0.00E+00	1.75E-03	0.00E+00	0.00E+00
WC	m^3	1.59E-02	3.88E-04	0.00E+00	7.31E-05	0.00E+00	1.54E-02

Table D-3: Characterized midpoint results of S2

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW	kg CO2 eq	7.40E-01	2.39E-01	4.47E-01	1.22E-02	4.18E-02	0.00E+00
SOD	kg CFC11 eq	1.67E-05	1.84E-07	1.65E-05	2.61E-09	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	2.41E-03	2.09E-03	0.00E+00	3.24E-04	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	8.37E-04	8.15E-04	0.00E+00	2.17E-05	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	5.19E-04	5.00E-04	0.00E+00	1.88E-05	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	8.45E-04	8.23E-04	0.00E+00	2.21E-05	0.00E+00	0.00E+00
TA	kg SO_2 eq	1.64E-03	1.60E-03	0.00E+00	4.85E-05	0.00E+00	0.00E+00
FE	kg P eq	5.11E-04	2.92E-05	0.00E+00	4.90E-06	0.00E+00	4.77E-04
ME	kg N eq	1.34E-03	3.18E-07	0.00E+00	1.27E-06	0.00E+00	1.34E-03
TE	kg 1,4 − DCB	1.01E+00	9.12E-01	0.00E+00	1.00E-01	3.14E-19	0.00E+00
FEC	kg 1,4 − DCB	1.01E-02	1.65E-03	0.00E+00	9.31E-04	7.53E-03	0.00E+00
MEC	kg 1,4 − DCB	9.46E-03	2.70E-03	0.00E+00	1.24E-03	5.52E-03	0.00E+00
HCT	kg 1,4 − DCB	9.14E-03	1.92E-03	0.00E+00	1.85E-03	5.37E-03	0.00E+00
HNCT	kg 1,4 − DCB	1.49E-01	2.42E-02	0.00E+00	1.96E-02	1.05E-01	0.00E+00
LU	m^2a crop eq	3.16E-03	2.90E-03	0.00E+00	2.59E-04	0.00E+00	0.00E+00
MRS	kg Cu eq	2.29E-04	7.73E-05	0.00E+00	1.52E-04	0.00E+00	0.00E+00
FRS	kg oil eq	7.00E-02	6.76E-02	0.00E+00	2.40E-03	0.00E+00	0.00E+00
WC	m^3	6.84E-04	6.07E-04	0.00E+00	7.73E-05	0.00E+00	0.00E+00

Table D-4: Characterized midpoint results of S3

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW	kg CO2 eq	6.53E-01	2.01E-01	3.75E-01	8.02E-03	6.80E-02	0.00E+00
SOD	kg CFC11 eq	1.40E-05	1.54E-07	1.39E-05	1.01E-09	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	1.84E-03	1.75E-03	0.00E+00	8.33E-05	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	6.92E-04	6.85E-04	0.00E+00	7.53E-06	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	4.25E-04	4.20E-04	0.00E+00	4.25E-06	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	6.99E-04	6.91E-04	0.00E+00	7.77E-06	0.00E+00	0.00E+00
TA	kg SO_2 eq	1.35E-03	1.34E-03	0.00E+00	1.21E-05	0.00E+00	0.00E+00
FE	kg P eq	7.64E-04	2.45E-05	0.00E+00	1.73E-06	0.00E+00	7.37E-04
ME	kg N eq	8.33E-04	2.67E-07	0.00E+00	1.15E-06	0.00E+00	8.32E-04
TE	kg 1,4 − DCB	7.82E-01	7.66E-01	0.00E+00	1.63E-02	3.14E-19	0.00E+00
FEC	kg 1,4 − DCB	8.97E-03	1.38E-03	0.00E+00	5.96E-05	7.53E-03	0.00E+00
MEC	kg 1,4 − DCB	7.88E-03	2.27E-03	0.00E+00	8.77E-05	5.52E-03	0.00E+00
HCT	kg 1,4 − DCB	7.08E-03	1.61E-03	0.00E+00	9.44E-05	5.37E-03	0.00E+00
HNCT	kg 1,4 − DCB	1.27E-01	2.04E-02	0.00E+00	1.56E-03	1.05E-01	0.00E+00
LU	m^2a crop eq	2.56E-03	2.44E-03	0.00E+00	1.24E-04	0.00E+00	0.00E+00
MRS	kg Cu eq	6.96E-05	6.49E-05	0.00E+00	4.72E-06	0.00E+00	0.00E+00
FRS	kg oil eq	5.81E-02	5.68E-02	0.00E+00	1.34E-03	0.00E+00	0.00E+00
WC	m^3	5.31E-04	5.10E-04	0.00E+00	2.17E-05	0.00E+00	0.00E+00

Table D-5: Normalized midpoint results for S0

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW	kg CO2 eq	8.48E-05	2.94E-05	4.53E-05	1.52E-06	8.50E-06	0.00E+00
SOD	kg CFC11 eq	1.44E-04	3.02E-06	1.41E-04	4.36E-08	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	4.95E-06	4.27E-06	0.00E+00	6.75E-07	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	4.01E-05	3.90E-05	0.00E+00	1.06E-06	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	2.00E-05	1.93E-05	0.00E+00	7.33E-07	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	4.68E-05	4.56E-05	0.00E+00	1.24E-06	0.00E+00	0.00E+00
TA	kg SO_2 eq	3.95E-05	3.83E-05	0.00E+00	1.18E-06	0.00E+00	0.00E+00
FE	kg P eq	1.06E-03	4.43E-05	0.00E+00	7.55E-06	0.00E+00	1.01E-03
ME	kg N eq	1.86E-04	6.78E-08	0.00E+00	2.76E-07	0.00E+00	1.86E-04
TE	kg 1,4 − DCB	6.56E-05	5.90E-05	0.00E+00	6.59E-06	2.07E-23	0.00E+00
FEC	kg 1,4 − DCB	4.00E-04	6.43E-05	0.00E+00	3.69E-05	2.99E-04	0.00E+00
MEC	kg 1,4 − DCB	2.17E-04	6.12E-05	0.00E+00	2.85E-05	1.27E-04	0.00E+00
HCT	kg 1,4 − DCB	8.85E-04	1.83E-04	0.00E+00	1.80E-04	5.22E-04	0.00E+00
HNCT	kg 1,4 − DCB	4.75E-06	7.64E-07	0.00E+00	6.28E-07	3.35E-06	0.00E+00
LU	m^2a crop eq	5.05E-07	4.63E-07	0.00E+00	4.20E-08	0.00E+00	0.00E+00
MRS	kg Cu eq	1.90E-09	6.33E-10	0.00E+00	1.26E-09	0.00E+00	0.00E+00
FRS	kg oil eq	7.03E-05	6.79E-05	0.00E+00	2.45E-06	0.00E+00	0.00E+00
WC	m^3	2.53E-06	2.24E-06	0.00E+00	2.90E-07	0.00E+00	0.00E+00

Table D-6: Normalized midpoint results for S1

Impact category	Unit	Total	PC	BT	CC	ED	AD
GW	kg CO2 eq	6.52E-05	1.91E-05	4.53E-05	7.15E-07	0.00E+00	0.00E+00
SOD	kg CFC11 eq	1.43E-04	1.96E-06	1.41E-04	3.07E-08	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	3.35E-06	2.78E-06	0.00E+00	5.72E-07	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	2.62E-05	2.54E-05	0.00E+00	8.15E-07	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	1.32E-05	1.25E-05	0.00E+00	6.39E-07	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	3.06E-05	2.96E-05	0.00E+00	9.59E-07	0.00E+00	0.00E+00
TA	kg SO_2 eq	2.60E-05	2.49E-05	0.00E+00	1.04E-06	0.00E+00	0.00E+00
FE	kg P eq	1.04E-03	2.88E-05	0.00E+00	6.29E-06	1.01E-03	0.00E+00
ME	kg N eq	1.86E-04	4.41E-08	0.00E+00	2.65E-07	1.86E-04	0.00E+00
TE	kg 1,4 − DCB	4.42E-05	3.84E-05	0.00E+00	5.78E-06	0.00E+00	0.00E+00
FEC	kg 1,4 − DCB	7.81E-05	4.18E-05	0.00E+00	3.63E-05	0.00E+00	0.00E+00
MEC	kg 1,4 − DCB	6.76E-05	3.98E-05	0.00E+00	2.78E-05	0.00E+00	0.00E+00
HCT	kg 1,4 − DCB	2.95E-04	1.19E-04	0.00E+00	1.76E-04	0.00E+00	0.00E+00
HNCT	kg 1,4 − DCB	1.10E-06	4.97E-07	0.00E+00	6.07E-07	0.00E+00	0.00E+00
LU	m^2a crop eq	3.26E-07	3.01E-07	0.00E+00	2.52E-08	0.00E+00	0.00E+00
MRS	kg Cu eq	1.67E-09	4.12E-10	0.00E+00	1.25E-09	0.00E+00	0.00E+00
FRS	kg oil eq	4.59E-05	4.41E-05	0.00E+00	1.79E-06	0.00E+00	0.00E+00
WC	m^3	5.95E-05	1.46E-06	0.00E+00	2.74E-07	0.00E+00	5.78E-05

Table D-7: Normalized midpoint results for S2

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW	kg CO2 eq	9.25E-05	2.99E-05	5.59E-05	1.52E-06	5.23E-06	0.00E+00
SOD	kg CFC11 eq	2.79E-04	3.06E-06	2.76E-04	4.36E-08	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	5.02E-06	4.34E-06	0.00E+00	6.75E-07	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	4.07E-05	3.96E-05	0.00E+00	1.06E-06	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	2.03E-05	1.96E-05	0.00E+00	7.33E-07	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	4.76E-05	4.63E-05	0.00E+00	1.24E-06	0.00E+00	0.00E+00
TA	kg SO_2 eq	4.01E-05	3.89E-05	0.00E+00	1.18E-06	0.00E+00	0.00E+00
FE	kg P eq	7.87E-04	4.50E-05	0.00E+00	7.55E-06	0.00E+00	7.34E-04
ME	kg N eq	2.91E-04	6.89E-08	0.00E+00	2.76E-07	0.00E+00	2.91E-04
TE	kg 1,4 − DCB	6.66E-05	6.00E-05	0.00E+00	6.59E-06	2.07E-23	0.00E+00
FEC	kg 1,4 − DCB	4.01E-04	6.54E-05	0.00E+00	3.69E-05	2.99E-04	0.00E+00
MEC	kg 1,4 − DCB	2.18E-04	6.22E-05	0.00E+00	2.85E-05	1.27E-04	0.00E+00
HCT	kg 1,4 − DCB	8.88E-04	1.86E-04	0.00E+00	1.80E-04	5.22E-04	0.00E+00
HNCT	kg 1,4 − DCB	4.76E-06	7.76E-07	0.00E+00	6.28E-07	3.35E-06	0.00E+00
LU	m^2a crop eq	5.12E-07	4.70E-07	0.00E+00	4.20E-08	0.00E+00	0.00E+00
MRS	kg Cu eq	1.91E-09	6.44E-10	0.00E+00	1.26E-09	0.00E+00	0.00E+00
FRS	kg oil eq	7.14E-05	6.90E-05	0.00E+00	2.45E-06	0.00E+00	0.00E+00
WC	m^3	2.57E-06	2.28E-06	0.00E+00	2.90E-07	0.00E+00	0.00E+00

Table D-8: Normalized midpoint results for S3

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW	kg CO2 eq	8.16E-05	2.51E-05	4.69E-05	1.00E-06	8.50E-06	0.00E+00
SOD	kg CFC11 eq	2.34E-04	2.57E-06	2.31E-04	1.69E-08	0.00E+00	0.00E+00
IR	kBq Co − 60 eq	3.82E-06	3.65E-06	0.00E+00	1.73E-07	0.00E+00	0.00E+00
OF-HH	kg NO_x eq	3.37E-05	3.33E-05	0.00E+00	3.66E-07	0.00E+00	0.00E+00
FPMF	kg PM2.5 eq	1.66E-05	1.64E-05	0.00E+00	1.66E-07	0.00E+00	0.00E+00
OF-TE	kg NO_x eq	3.93E-05	3.89E-05	0.00E+00	4.37E-07	0.00E+00	0.00E+00
TA	kg SO_2 eq	3.30E-05	3.27E-05	0.00E+00	2.95E-07	0.00E+00	0.00E+00
FE	kg P eq	1.18E-03	3.78E-05	0.00E+00	2.67E-06	0.00E+00	1.14E-03
ME	kg N eq	1.81E-04	5.79E-08	0.00E+00	2.50E-07	0.00E+00	1.80E-04
TE	kg 1,4 − DCB	5.15E-05	5.04E-05	0.00E+00	1.07E-06	2.07E-23	0.00E+00
FEC	kg 1,4 − DCB	3.56E-04	5.49E-05	0.00E+00	2.37E-06	2.99E-04	0.00E+00
MEC	kg 1,4 − DCB	1.81E-04	5.22E-05	0.00E+00	2.02E-06	1.27E-04	0.00E+00
HCT	kg 1,4 − DCB	6.87E-04	1.56E-04	0.00E+00	9.16E-06	5.22E-04	0.00E+00
HNCT	kg 1,4 − DCB	4.06E-06	6.52E-07	0.00E+00	5.00E-08	3.35E-06	0.00E+00
LU	m^2a crop eq	4.15E-07	3.95E-07	0.00E+00	2.01E-08	0.00E+00	0.00E+00
MRS	kg Cu eq	5.80E-10	5.41E-10	0.00E+00	3.93E-11	0.00E+00	0.00E+00
FRS	kg oil eq	5.93E-05	5.79E-05	0.00E+00	1.37E-06	0.00E+00	0.00E+00
WC	m^3	1.99E-06	1.91E-06	0.00E+00	8.13E-08	0.00E+00	0.00E+00

Table D-9: Characterized endpoint results for S0

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW, HH	DALY	6.30E-07	2.19E-07	3.37E-07	1.13E-08	6.32E-08	0.00E+00
GW, TECO	species.yr	1.90E-09	6.59E-10	1.01E-09	3.41E-11	1.90E-10	0.00E+00
GW, FECO	species.yr	5.19E-14	1.80E-14	2.77E-14	9.32E-16	5.20E-15	0.00E+00
SOD	DALY	4.58E-09	9.58E-11	4.49E-09	1.39E-12	0.00E+00	0.00E+00
IR	DALY	2.02E-11	1.75E-11	0.00E+00	2.75E-12	0.00E+00	0.00E+00
OF, HH	DALY	7.50E-10	7.30E-10	0.00E+00	1.98E-11	0.00E+00	0.00E+00
FPMF	DALY	3.21E-07	3.09E-07	0.00E+00	1.18E-08	0.00E+00	0.00E+00
OF, TECO	species.yr	1.07E-10	1.04E-10	0.00E+00	2.85E-12	0.00E+00	0.00E+00
TA	species.yr	3.43E-10	3.33E-10	0.00E+00	1.03E-11	0.00E+00	0.00E+00
FE	species.yr	4.62E-10	1.93E-11	0.00E+00	3.29E-12	0.00E+00	4.39E-10
ME	species.yr	1.46E-12	5.31E-16	0.00E+00	2.17E-15	0.00E+00	1.45E-12
TE	species.yr	1.14E-11	1.02E-11	0.00E+00	1.14E-12	3.58E-30	0.00E+00
FEC	species.yr	7.00E-12	1.12E-12	0.00E+00	6.44E-13	5.23E-12	0.00E+00
MEC	species.yr	9.89E-13	2.80E-13	0.00E+00	1.30E-13	5.80E-13	0.00E+00
HCT	DALY	3.02E-08	6.26E-09	0.00E+00	6.16E-09	1.78E-08	0.00E+00
HNCT	DALY	3.38E-08	5.44E-09	0.00E+00	4.47E-09	2.39E-08	0.00E+00
LU	species.yr	2.76E-11	2.53E-11	0.00E+00	2.30E-12	0.00E+00	0.00E+00
MRS	USD2013	5.27E-05	1.76E-05	0.00E+00	3.51E-05	0.00E+00	0.00E+00
FRS	USD2013	3.07E-02	2.99E-02	0.00E+00	7.65E-04	0.00E+00	0.00E+00
WC, HH	DALY	4.29E-10	3.00E-10	0.00E+00	1.29E-10	0.00E+00	0.00E+00
WC, TECO	species.yr	3.04E-12	2.21E-12	0.00E+00	8.27E-13	0.00E+00	0.00E+00
WC, AE	species.yr	1.88E-16	1.13E-16	0.00E+00	7.53E-17	0.00E+00	0.00E+00

Table D-10: Characterized endpoint results for S1

Impact category	Unit	Total	PC	BT	CC	ED	AD
GW, HH	DALY	4.84E-07	1.42E-07	3.37E-07	5.30E-09	0.00E+00	0.00E+00
GW, TECO	species.yr	1.46E-09	4.29E-10	1.01E-09	1.60E-11	0.00E+00	0.00E+00
GW, FECO	species.yr	3.99E-14	1.17E-14	2.77E-14	4.37E-16	0.00E+00	0.00E+00
SOD	DALY	4.55E-09	6.23E-11	4.49E-09	9.74E-13	0.00E+00	0.00E+00
IR	DALY	1.37E-11	1.14E-11	0.00E+00	2.33E-12	0.00E+00	0.00E+00
OF, HH	DALY	4.90E-10	4.75E-10	0.00E+00	1.53E-11	0.00E+00	0.00E+00
FPMF	DALY	2.11E-07	2.01E-07	0.00E+00	1.03E-08	0.00E+00	0.00E+00
OF, TECO	species.yr	7.01E-11	6.79E-11	0.00E+00	2.20E-12	0.00E+00	0.00E+00
TA	species.yr	2.25E-10	2.16E-10	0.00E+00	9.03E-12	0.00E+00	0.00E+00
FE	species.yr	4.55E-10	1.26E-11	0.00E+00	2.74E-12	4.39E-10	0.00E+00
ME	species.yr	1.46E-12	3.46E-16	0.00E+00	2.08E-15	1.45E-12	0.00E+00
TE	species.yr	7.65E-12	6.65E-12	0.00E+00	1.00E-12	0.00E+00	0.00E+00
FEC	species.yr	1.36E-12	7.29E-13	0.00E+00	6.33E-13	0.00E+00	0.00E+00
MEC	species.yr	3.09E-13	1.82E-13	0.00E+00	1.27E-13	0.00E+00	0.00E+00
HCT	DALY	1.01E-08	4.07E-09	0.00E+00	6.02E-09	0.00E+00	0.00E+00
HNCT	DALY	7.86E-09	3.54E-09	0.00E+00	4.33E-09	0.00E+00	0.00E+00
LU	species.yr	1.78E-11	1.65E-11	0.00E+00	1.38E-12	0.00E+00	0.00E+00
MRS	USD2013	4.63E-05	1.14E-05	0.00E+00	3.48E-05	0.00E+00	0.00E+00
FRS	USD2013	1.99E-02	1.94E-02	0.00E+00	4.96E-04	0.00E+00	0.00E+00
WC, HH	DALY	2.11E-08	1.95E-10	0.00E+00	1.25E-10	0.00E+00	2.08E-08
WC, TECO	species.yr	1.25E-10	1.44E-12	0.00E+00	7.97E-13	0.00E+00	1.22E-10
WC, AE	species.yr	1.43E-16	7.34E-17	0.00E+00	6.99E-17	0.00E+00	0.00E+00

Table D-11: Characterized endpoint results for S2

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW, HH	DALY	6.88E-07	2.22E-07	4.16E-07	1.13E-08	3.89E-08	0.00E+00
GW, TECO	species. yr	2.07E-09	6.70E-10	1.25E-09	3.41E-11	1.17E-10	0.00E+00
GW, FECO	species. yr	5.66E-14	1.83E-14	3.42E-14	9.32E-16	3.20E-15	0.00E+00
SOD	DALY	8.86E-09	9.74E-11	8.76E-09	1.39E-12	0.00E+00	0.00E+00
IR	DALY	2.05E-11	1.77E-11	0.00E+00	2.75E-12	0.00E+00	0.00E+00
OF, HH	DALY	7.62E-10	7.42E-10	0.00E+00	1.98E-11	0.00E+00	0.00E+00
FPMF	DALY	3.26E-07	3.14E-07	0.00E+00	1.18E-08	0.00E+00	0.00E+00
OF, TECO	species. yr	1.09E-10	1.06E-10	0.00E+00	2.85E-12	0.00E+00	0.00E+00
TA	species. yr	3.48E-10	3.38E-10	0.00E+00	1.03E-11	0.00E+00	0.00E+00
FE	species. yr	3.43E-10	1.96E-11	0.00E+00	3.29E-12	0.00E+00	3.21E-10
ME	species. yr	2.28E-12	5.40E-16	0.00E+00	2.17E-15	0.00E+00	2.28E-12
TE	species. yr	1.15E-11	1.04E-11	0.00E+00	1.14E-12	3.58E-30	0.00E+00
FEC	species. yr	7.01E-12	1.14E-12	0.00E+00	6.44E-13	5.23E-12	0.00E+00
MEC	species. yr	9.94E-13	2.84E-13	0.00E+00	1.30E-13	5.80E-13	0.00E+00
HCT	DALY	3.03E-08	6.36E-09	0.00E+00	6.16E-09	1.78E-08	0.00E+00
HNCT	DALY	3.39E-08	5.53E-09	0.00E+00	4.47E-09	2.39E-08	0.00E+00
LU	species. yr	2.80E-11	2.57E-11	0.00E+00	2.30E-12	0.00E+00	0.00E+00
MRS	USD2013	5.30E-05	1.79E-05	0.00E+00	3.51E-05	0.00E+00	0.00E+00
FRS	USD2013	3.12E-02	3.04E-02	0.00E+00	7.65E-04	0.00E+00	0.00E+00
WC, HH	DALY	4.34E-10	3.04E-10	0.00E+00	1.29E-10	0.00E+00	0.00E+00
WC, TECO	species. yr	3.08E-12	2.25E-12	0.00E+00	8.27E-13	0.00E+00	0.00E+00
WC, AE	species. yr	1.90E-16	1.15E-16	0.00E+00	7.53E-17	0.00E+00	0.00E+00

Table D-12: Characterized endpoint results for S3

Impact category	Unit	Total	PC	BT	CC	SL	ED
GW, HH	DALY	6.06E-07	1.87E-07	3.49E-07	7.45E-09	6.32E-08	0.00E+00
GW, TECO	species. yr	1.83E-09	5.63E-10	1.05E-09	2.25E-11	1.90E-10	0.00E+00
GW, FECO	species. yr	4.99E-14	1.54E-14	2.87E-14	6.14E-16	5.20E-15	0.00E+00
SOD	DALY	7.44E-09	8.18E-11	7.36E-09	5.38E-13	0.00E+00	0.00E+00
IR	DALY	1.56E-11	1.49E-11	0.00E+00	7.07E-13	0.00E+00	0.00E+00
OF, HH	DALY	6.30E-10	6.23E-10	0.00E+00	6.85E-12	0.00E+00	0.00E+00
FPMF	DALY	2.67E-07	2.64E-07	0.00E+00	2.67E-09	0.00E+00	0.00E+00
OF, TECO	species. yr	9.02E-11	8.92E-11	0.00E+00	1.00E-12	0.00E+00	0.00E+00
TA	species. yr	2.87E-10	2.84E-10	0.00E+00	2.56E-12	0.00E+00	0.00E+00
FE	species. yr	5.13E-10	1.65E-11	0.00E+00	1.17E-12	0.00E+00	4.96E-10
ME	species. yr	1.42E-12	4.54E-16	0.00E+00	1.96E-15	0.00E+00	1.41E-12
TE	species. yr	8.91E-12	8.72E-12	0.00E+00	1.86E-13	3.58E-30	0.00E+00
FEC	species. yr	6.23E-12	9.57E-13	0.00E+00	4.13E-14	5.23E-12	0.00E+00
MEC	species. yr	8.28E-13	2.39E-13	0.00E+00	9.22E-15	5.80E-13	0.00E+00
HCT	DALY	2.35E-08	5.34E-09	0.00E+00	3.13E-10	1.78E-08	0.00E+00
HNCT	DALY	2.89E-08	4.64E-09	0.00E+00	3.56E-10	2.39E-08	0.00E+00
LU	species. yr	2.27E-11	2.16E-11	0.00E+00	1.10E-12	0.00E+00	0.00E+00
MRS	USD2013	1.61E-05	1.50E-05	0.00E+00	1.09E-06	0.00E+00	0.00E+00
FRS	USD2013	2.60E-02	2.55E-02	0.00E+00	5.22E-04	0.00E+00	0.00E+00
WC, HH	DALY	2.91E-10	2.56E-10	0.00E+00	3.57E-11	0.00E+00	0.00E+00
WC, TECO	species. yr	2.12E-12	1.89E-12	0.00E+00	2.29E-13	0.00E+00	0.00E+00
WC, AE	species. yr	1.17E-16	9.63E-17	0.00E+00	2.10E-17	0.00E+00	0.00E+00

Table D-13: Damage assessment results of S0

Damage category	Unit	Total	PC	BT	CC	SL	ED
Human health	DALY	1.02E-06	5.41E-07	3.41E-07	3.39E-08	1.05E-07	0.00E+00
Ecosystems	species. yr	2.86E-09	1.16E-09	1.01E-09	5.56E-11	1.96E-10	4.41E-10
Resources	USD2013	3.07E-02	2.99E-02	0.00E+00	8.00E-04	0.00E+00	0.00E+00

Table D-14: Damage assessment results of S1

Damage category	Unit	Total	PC	BT	CC	ED	AD
Human health	DALY	7.40E-07	3.52E-07	3.41E-07	2.61E-08	0.00E+00	2.08E-08
Ecosystems	species. yr	2.36E-09	7.51E-10	1.01E-09	3.39E-11	4.41E-10	1.22E-10
Resources	USD2013	2.00E-02	1.95E-02	0.00E+00	5.30E-04	0.00E+00	0.00E+00

Table D-15: Damage assessment results of S2

Damage category	Unit	Total	PC	BT	CC	SL	ED
Human health	DALY	1.09E-06	5.49E-07	4.24E-07	3.39E-08	8.06E-08	0.00E+00
Ecosystems	species. yr	2.93E-09	1.17E-09	1.25E-09	5.56E-11	1.23E-10	3.23E-10
Resources	USD2013	3.12E-02	3.04E-02	0.00E+00	8.00E-04	0.00E+00	0.00E+00

Table D-16: Damage assessment results of S3

Damage category	Unit	Total	PC	BT	CC	SL	ED
Human health	DALY	9.34E-07	4.61E-07	3.56E-07	1.08E-08	1.05E-07	0.00E+00
Ecosystems	species. yr	2.76E-09	9.86E-10	1.05E-09	2.88E-11	1.96E-10	4.97E-10
Resources	USD2013	2.61E-02	2.55E-02	0.00E+00	5.23E-04	0.00E+00	0.00E+00

Table D-17: Normalized endpoint results of S0

Damage category	Total	PC	BT	CC	SL	ED
Human health	4.26E-05	2.25E-05	1.42E-05	1.41E-06	4.37E-06	0.00E+00
Ecosystems	1.94E-06	7.81E-07	6.86E-07	3.76E-08	1.33E-07	2.98E-07
Resources	1.10E-06	1.07E-06	0.00E+00	2.86E-08	0.00E+00	0.00E+00

Table D-18: Normalized endpoint results of S1

Damage category	Total	PC	BT	CC	ED	AD
Human health	3.09E-05	1.47E-05	1.42E-05	1.09E-06	0.00E+00	8.67E-07
Ecosystems	1.60E-06	5.08E-07	6.86E-07	2.29E-08	2.98E-07	8.28E-08
Resources	7.14E-07	6.95E-07	0.00E+00	1.89E-08	0.00E+00	0.00E+00

Table D-19: Normalized endpoint results of S2

Damage category	Total	PC	BT	CC	SL	ED
Human health	4.54E-05	2.29E-05	1.77E-05	1.41E-06	3.36E-06	0.00E+00
Ecosystems	1.98E-06	7.94E-07	8.46E-07	3.76E-08	8.31E-08	2.18E-07
Resources	1.11E-06	1.09E-06	0.00E+00	2.86E-08	0.00E+00	0.00E+00

Table D-20: Normalized endpoint results of S3

Damage category	Total	PC	BT	CC	SL	ED
Human health	3.89E-05	1.92E-05	1.49E-05	4.52E-07	4.37E-06	0.00E+00
Ecosystems	1.87E-06	6.67E-07	7.10E-07	1.94E-08	1.33E-07	3.36E-07
Resources	9.30E-07	9.12E-07	0.00E+00	1.87E-08	0.00E+00	0.00E+00

Table D-21: Costs input parameters

Parameter	Unit	Default Value	Adjusted Value
Unit Costs			
Building Costs	$\$/m^2$	1184.04	350
Excavation	$\$/m^3$	10.4637	1.76
Wall Concrete	$\$/m^3$	850.173	245
Slab Concrete	$\$/m^3$	457.786	239
Crane Rental	$\$/hr$	250	40
Canopy Roof	$\$/m^2$	215.28	150
Handrail	$\$/m$	246.06	60
Labor Rates			
Construction Labor Rate	$\$/hr$	40	1.9
Operator Labor Rate	$\$/hr$	51.5	9
Administration Labor Rate	$\$/hr$	51.5	5
Laboratory Labor Rate	$\$/hr$	51.5	9
Chemical Costs			
Hydrated Lime-[$Ca(OH)_2$]	$\$/kg$	0.396828	0.078
$Al_2(SO_4)_3 * 14H_2O$	$\$/kg$	0.595242	0.207
Polymer	$\$/kg$	2.86598	2.85
Region Specific			
Land Costs	$\$/m^2$	4.94205	20
Financial			
Interest Rate	%	8	6
Other Costs			
Engineering Design Fee	%	15	3
Administration/Legal	%	2	1