https://doi.org/10.1016/j.resconrec.2020.105196

# A Socio-Eco-Efficiency Analysis of Water and Wastewater Treatment Processes for Refugee Communities in Jordan

Siti Nurhawa Binte Muhammad Anwar[1], Valeria Alvarado[2], and Shu-Chien Hsu[3],*

[1] Research Associate, Nanyang Environment and Water Research Institute, Nanyang Technological University. Email: siti.nurhawa@ntu.edu.sg

[2] Ph.D. Student, Department of Civil and Environmental Engineering, The Hong Kong Polytechnic University. Email: valeria-isabel.alvaradoroman@connect.polyu.hk

[3] Associate Professor; Corresponding Author, Department of Civil and Environmental Engineering, The Hong Kong Polytechnic University. Email: mark.hsu@polyu.edu.hk

## Abstract

As of 2017, the United Nations has estimated that there are 68.5 million displaced people in the world which live in refugee camps (RCs) in 125 host countries. RCs frequently encounter water scarcity issues which lead to a low daily limit of water consumption, as well as face management difficulties such as septic tank overflowing which contribute to the development of health problems. Considering the need for more sustainable water, sanitation, and hygiene system in RCs, a socio-eco-efficiency analysis (SEEA) framework is proposed for the analysis and comparison of different wastewater treatment methods. The SEEA framework consists of the integration of the economic and environmental aspects analysed by an eco-efficiency analysis (EEA) with the social aspects evaluated by a social life cycle assessment (S-LCA) using the analytic hierarchy process. The SEEA framework was applied to compare different wastewater treatment methods in Zaatari Refugee Camp in Jordan. The SEEA results show that, if adopted, an effluent water reuse-based treatment would increase economic efficiency by 75%, decrease environmental impacts by 57%, and increase social sustainability by 57%

compared to the current operation of the camp, where a wastewater system connects groups of seven to nine households to communal septic tanks. A ternary diagram is used to represent the comparison of different wastewater treatment methods for an RC. The diagram shows the degree of socio-eco-efficiency of each wastewater treatment method, in terms of its social impacts, environmental impacts, and cost by normalizing results of the EEA and S-LCA into one score.

Keywords: *Life Cycle Assessment, Life Cycle Cost, Social Life Cycle Assessment, Wastewater Treatment, Analytic Hierarchy Process*.

## 1. Introduction

Refugees include those who have crossed the borders of countries where they previously resided, internally displaced people, asylum seekers, people in refugee-like situations, and stateless people. The causes of displacement include armed conflict, violence, disasters, famine, development, and economic changes (UNESCO, 2017). In 2016, the number of new refugees due to conflict and violence was 11.8 million (38.6%), while those of disasters (termed 'climate refugees') was 18.8 million (61.4%) (IDMC, 2018). These figures are expected to rise in response to the increase in disasters, as current forecasts on climate refugees vary from 25 million to 1 billion in the year 2050 (Kamal, 2017).

The United Nations High Commissioner for Refugees (UNHCR) field operations for water, sanitation, and hygiene (WASH) services (2008) have "fundamental responsibility of providing legal security, physical safety (against natural or man-made threats) and material assistance (necessities of life)" for refugees. Water and adequate sanitation including excreta disposal are among the necessities of life (UNHCR, 2008). Once 2 years elapse from the time of the original emergency, a refugee setting transitions from a 'communal' phase to a ‘household’ phase. Guidelines for wastewater management and treatment processes for a refugee setting during the ‘household’ phase recommend boreholes as a source of water, surface source and treatment, a pipe network, and a sewer network with desludging treatment (UNHCR, 2018). Sludge management practices in refugee camps (RCs) have included ‘lagooning’, directly disposing sludge into a water body, or drying beds before discharging sludge into dumpsites (UNHCR, 1992). These wastewater treatments and sludge management options are widely applied in RCs around the world because of their low cost and simple installation in emergency situations.

WASH services in RCs or similar communities have been studied for their technical, social, economic, and environmental implications through multicriteria analysis (Garfi & Ferrer-Marti, 2011), decision algorithms (Fenner et al., 2007), mental models (Kosonen & Kim, 2018), hydrogeological assessments (Eggen, 2019), surveys (Nyoka et al., 2017), and input-mediator-output models (Kosonen et al., 2018). However, these analyses did not integrate environmental, economic, and social aspects of WASH services with the specific objective of achieving long-term and sustainable WASH services in RCs.

A widely used decision-making aid tool for the quantification of environmental impacts in the water and wastewater treatment field is life-cycle assessment (LCA) (Byrne et al., 2017). To complement the environmental insights provided by LCAs, some studies have included economic components by integrating a life cycle cost (LCC) analysis and combining these results in an eco-efficiency analysis (EEA) (Kicherer et al., 2007; Lam et al., 2017). LCA and data envelopment analysis have been combined in an EEA framework for the study of eco-efficiency in wastewater treatment plants (WWTPs) (Lorenzo-Toja et al., 2014). Even though numerous studies have evaluated the environmental and economic impacts of different wastewater treatment systems (Abdallah et al. 2020; Lam et al., 2015; Shiu et al., 2017), few studies have integrated social considerations into the analysis (Appendix A). The inclusion of social aspects of RCs is vital because refugees are a vulnerable population in need of safe and adequate water and sanitation health. In light of these considerations, new designs for refugee settlements are shifting from being efficiency-oriented to people-oriented, and from temporary to permanent (UNHCR, 2018a). Conducting an LCA that includes an analysis of social impacts can thus inform the implementation of more socially sustainable policies and practices, leading to more beneficial outcomes for stakeholders.

Methodologies for the integration of social factors with environmental and economic analysis of different products or processes are under development (Kloepffer, 2008). For example, the Baden Aniline and Soda Factory (BASF) developed a method called the SEEbalance®, which calculates socio-efficiency using social indicator systems and specific databases such as the EU classification of economic activities (Schmidt et al., 2004). The BASF method was applied to determine the socio-eco-efficiency of crop livestock forestry systems in Brazil (Costa et al., 2018). Opher et al. (2018) combined analytic hierarchy process (AHP) with a life-cycle sustainability assessment framework for the comparison of urban water reuse at different centralization scales. AHP involves drawing from expert judgments when weighting sustainability criteria and producing a composite score of the weighted sum of all criteria.

Studies related to RCs or similar settlements using life-cycle tools have especially focused on housing (Alnsour & Meaton, 2013; Atmaca & Atmaca, 2016; van Kempen et al., 2016). Aside from housing, other necessities must be analysed to ensure the well-being of displaced communities. The objectives of this study are: (i) to develop a socio-eco-efficiency analysis (SEEA) framework as a decision-making aid tool in accordance with the tripartite sustainability model for water and wastewater treatment, and (ii) to compare the environmental, economic, and social implications of different WASH services in Jordan as a case study. In addition to its contributions to methodological development, this study provides practical analysis for science-driven decision-making with particular attention to water reuse as a sustainable solution for water scarcity in refugee settlements.

## 2. Methodology

### *2.1 Case study – Zaatari Refugee Camp*

Since 2014, more sophisticated wastewater treatments have been adopted in some RCs. For example, the Azraq RC in Jordan is the world's first camp to adopt an on-site WWTP that utilizes a modular moving-bed biofilm reactor (MBR) along with pre-treatment and chlorination. The Azraq RC is seen as a 'model refugee camp' as its facilities were designed to overcome problems experienced in the older Zaatari RC in Jordan (Knell, 2014). Located in the Mafraq governorate, Zaatari RC serves approximately 80,000 refugees from Syria (UNHCR, 2019). As of 2017, UNHCR has implemented a long-term master plan for WASH in Zaatari RC. One analysis, comparing the cost-effectiveness of the water supply and treatment network in Zaatari RC to the UNHCR long-term plan, recommended that the camp undergo an integrated transition—including technically, socially, economically, and financially optimized solutions—from the emergency phase to long-term sustainability (van der Helm et al., 2017). The analysis also recognized that a decision-making aid model for the processes involved in active disaster response situations is needed to provide better water management and treatment facilities in refugee communities (Kosonen et al., 2018).

Jordan is one of the most water scarce countries in the world, thus the Ministry of Water and Irrigation's National Water Strategy 2016-2025 has aimed for more resilience in the protection of the nation's WASH sector coordination system and access to safe, affordable, and adequate water supply and sanitation for all citizens (MWI, 2016). The current water supply in Zaatari RC is within the limits of the camp demand. Yet, there are several WASH problems regarding sewage disposal and treatment methods that require improvements. Responsible NGOs in Zaatari RC conducted surveys throughout the camp to a) identify primary household sources of drinking water, b) assess the prevalence and suitability of WASH infrastructure across all households, c) record primary wastewater and solid waste disposal practices across all households and d) gauge refugee community perceptions of the adequacy of WASH repair and maintenance services within the RC (UNICEF & REACH, 2017). Several issues with the WASH infrastructure and service were identified by the surveys, such as blockages in the sewer wastewater network (WWN), overflowing septic tanks, and inefficient communication of WASH infrastructure problems to the primary NGO in each district (UNICEF & REACH, 2017). Thus, the UN prepared a long-term plan to tackle these issues as well as improve community outreach and services (UNICEF & REACH, 2017).

An SEEA framework (Figure 1) was developed in this study to evaluate wastewater treatment options specifically for refugee communities and similar settlements. The framework, which includes EEA and S-LCA, aims to systematically calculate the social, economic, and ecological scores for an array of wastewater treatment scenarios for RCs. The results of each step are to be interpreted on a progressive basis.

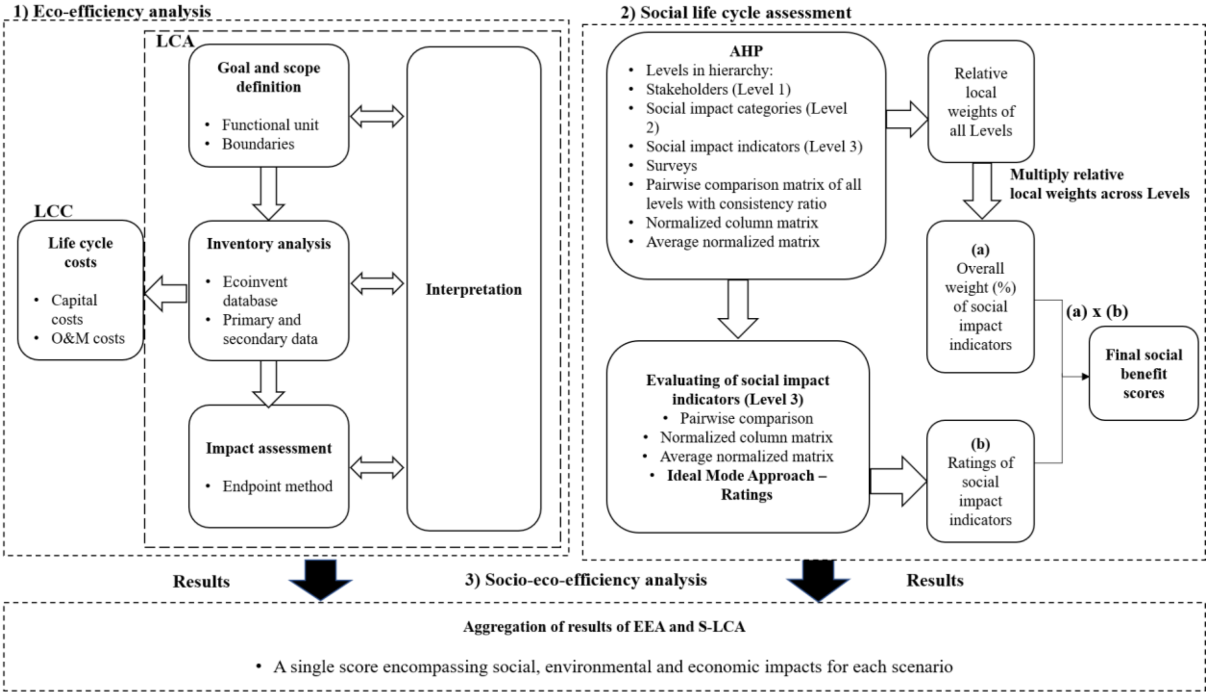


*Figure 1 – SEEA framework, which includes steps in the 1) Eco-efficiency analysis (EEA), 2)*

*Social life cycle assessment (S-LCA), and 3) Socio-eco-efficiency analysis. LCA: Life cycle assessment; LCC: Life cycle costs; O&M: Operation and maintenance; AHP: Analytic hierarchy process.*

### *2.2 Eco-efficiency analysis*

EEA is a management tool for LCA that integrates the analysis of the environmental impact and cost-effectiveness of a product's or service's life cycle (BASF, 2018). In this study, the EEA was based on the modified method presented in Lam et al., (2017), which integrated the BASF, and the Kicherer et al. (2007) normalization approach. The economic aspect is integrated through an LCC, while the environmental aspect with an LCA. The results of an EEA are typically represented in an eco-efficiency portfolio which consists of a graph where the x-coordinate represents the costs, while the y-coordinate represents the environmental impacts (Kicherer et al., 2007). The methodology for the eco-efficiency portfolio calculation can be found in Appendix B. Eco-efficiency is achieved through low costs and low environmental impacts.

#### *2.2.1 Life cycle assessment*

LCA consists of four main steps described by ISO14040 (Finkbeiner et al., 2006). In the LCA, the goal and scope are first defined, a life cycle inventory (LCI) is collated, and a life cycle impact assessment (LCIA) is then conducted. The goal of this study was to quantify the environmental impacts of three wastewater treatment scenarios to improve the current WASH facilities in Zaatari RC for sustainable improvement in living conditions. The functional unit used for this study is 1 $m^3$ of treated wastewater. The system boundary includes the impacts from the operation and maintenance (O&M) of the water treatment system in Zaatari RC. A 20-year time boundary was selected, which has been used in previous studies for similar infrastructure (Guereca et al., 2011; Lopes et al., 2018). The geographical boundary was based on Jordan for foreground information, while background information was taken from the ecoinvent v.3.2. database. The data sources for the LCA include primary and secondary data. Primary data on RCs are available in UN reports, namely from UNHCR and UNICEF. Secondary data was gathered from the literature and life-cycle databases such as the ecoinvent database.

The LCI focused on direct emissions, groundwater consumption, and electricity consumption (Appendix C). Direct emissions include water, air, and soil emissions from the wastewater treatment process. The emissions to water included biological oxygen demand, total organic carbon, dissolved organic carbon, ammonium ($NH_4^+$), nitrate, and chemical oxygen demand (COD) (UNICEF, 2014). The air emissions for each scenario are presented in Appendix D, showing the emissions of greenhouse gases (GHGs), such as nitrous oxide ($N_2O$) and methane directly from untreated $NH_4^+$ and COD respectively, that contribute to climate change. The emissions for GHGs were estimated using primary WWTP data from UNICEF (2014), IPCC emission factors (2007), and recommended GHG emission values for an effluent discharge without treatment (Godin et al., 2012). The soil emissions were based on ecoinvent data for agriculture application in similar systems of wastewater treatment processes i.e. WSP, MBR, and TF.

In the LCIA, the inputs and outputs in terms of materials, fuels, electricity, and heat are accounted for, as well as the emissions to air, water, and soil. Emissions from the construction phase of the case study were not included as the impacts were negligible. Emissions associated with electricity consumption and chemical usage, along with direct emissions from untreated effluent, were included in the LCIA. The ecoinvent v.3.2 database was used for background information applied in SimaPro software. ReCiPe Endpoint (H) v1.12/World ReCiPe H/A was selected as the impact assessment methodology. The LCIA impact categories considered in this method were: terrestrial acidification, marine eutrophication, photochemical oxidant formation, particulate matter formation, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, ionising radiation, water depletion, human toxicity, freshwater eutrophication, agricultural land occupation, urban land occupation, natural land transformation, metal depletion, fossil depletion, climate change, and ozone depletion.

Three life-cycle impact categories were identified as being relevant for wastewater treatment systems in the RC: i) water depletion, ii) human toxicity, and iii) freshwater eutrophication. Water depletion and freshwater eutrophication were chosen due to the critical water scarcity in places like Jordan (Schyns et al., 2015, Abu-Allaban et al., 2014) and the need to improve conditions of water management in refugee settlements (van der Helm et al., 2017). Human toxicity refers to the effects on human health caused by toxic substances in the environment, and accounting for this toxicity is critical to ensuring that the health of refugees and surrounding communities is not compromised.

### *2.2.2 Life cycle costing*

According to the US General Services Administration, LCC refers to an economic analysis used in the selection of alternatives that impact both pending and future costs (GSA, 2017). In environmental life-cycle costing, a framework is provided for evaluating decisions with consistent yet flexible system boundaries as a component of product sustainability assessments (Swarr et al., 2011). Hence, LCC is a tool for the quantification of the costs of a system or product incurred during its lifetime. The system boundary of the LCC of WWTPs generally includes capital costs (CC) and O&M costs. The typical data comprising O&M costs include the cost of electricity, maintenance, transportation, labor, and equipment (Hong et al., 2009; Lam et al., 2015).

The CC for the construction of the three scenarios were equal in terms of the on-site WWTP, and an additional cost for two of the options was from the simplified piped network system. All three options included the usage of boreholes but in different quantities, though the cost difference is minimal as the technology used in borehole construction is simple and inexpensive. In a study focusing on WWN, the CC for installation of boreholes and a simplified piped network in Zaatari RC represented less than 10% of the differences among the scenarios when compared to the overall LCC (ACTED et al., 2014).

The O&M costs were calculated using data from a water network study (ACTED et al., 2014), which considered Jordanian inflation rates and sensitivity analysis of recurrent costs at discount rates in net present cost. Typical discount rates used in these systems range from 2.5% to 9%, throughout a forty-year period (CEIC, 2017). The O&M costs included water trucking services, borehole operation, on-site WWTP, and waste stabilization ponds (WSP). Under transportation costs, the data analyzed included truck capacities and travel distances in present values. For water trucking services and borehole operations, costs were taken from the water network studies for Zaatari RC (ACTED et al., 2014). The costs of the WSP, trickling filter (TF), and MBR operations were based on comparable technologies used in India (Khalil et al., 2008). This cost estimate source was chosen due to the similar GDP (in terms of purchasing power parity) per capita of India (#113) and Jordan (#100) (World Bank, 2016). Cost analysis data for water effluent reuse was estimated from the USEPA Handbook (2016). In this study, economic output was omitted as the main stakeholders are non-profit organizations. A summary of all the data as well as assumptions and sources of data used in the LCC are found in Appendix E.

### *2.3 Social life cycle assessment*

The S-LCA framework presented by Opher et al. (2017) involving AHP was applied and adapted in this study to evaluate wastewater treatment options in RCs. The S-LCA framework consists of three main steps: applying AHP, evaluating social impact indicators, and rating the indicators through the ideal mode approach. Developed by Thomas L. Saaty (1980), AHP arranges the criteria of a specific goal into a hierarchy. The AHP method utilized by Opher et al. (2017) includes making pairwise comparisons of elements within each level of the hierarchy to produce a pairwise comparison matrix, normalizing the matrix, and averaging the normalized matrix to determine the relative local weight of the elements. A final homogenizing of the relative local weights produce a single social benefit score for each alternative scenario.

The AHP approach used in this study included the Goal (Level 0), Stakeholders (Level 1), Categories (Level 2), Indicators (Level 3), and Alternatives (Level 4). The UNEP/SETAC Methodological Sheets for 31 Sub-Categories of Impacts for S-LCA (2013) were used to choose the categories for the S-LCA. The selection of social impact categories should be analyzed on a case-by-case basis because stakeholders constantly vary in contrast to a typical residential area as they include temporary settlers and humanitarian organizations. The UNEP/SETAC (2013) guidelines considered several stakeholders and subcategories as a framework for the S-LCA of products or systems. The stakeholders considered in this study were *community* and *consumer*. Other stakeholders mentioned in the guidelines, such as *society*, *local community*, and *workers*, were not included. The chosen sub-categories were derived from the *community* and *consumer* issues, such as 'safe and healthy living conditions' (*community*), or 'feedback mechanism' (*consumer*). The social impact categories were chosen from the guidelines based on the impact of the wastewater management project on the WASH practices in Zaatari RC and the residents' perceptions of the adequacy of the WASH facilities (UNICEF & REACH, 2017) as explained in Appendix F. The S-LCA proposed in this study emphasizes treated effluent reuse as a sustainable solution for the water scarcity frequently experienced in this type of settlement.

The social impact categories chosen for the S-LCA were *safe and healthy living conditions*, *equity*, *community engagement*, *consumer health & safety*, and *feedback mechanism*. Quantifiable social impact indicators were then chosen for each social impact category. For the category *safe and healthy living conditions,* the two indicators chosen were *adequate ownership of WASH facilities* (AO) and reduction of desludging issues (RD). In the *equity* category the two indicators were *increased population access to improved WASH facilities regardless of the district* (IP), and *higher water supply equivalence* (HE). For the *community engagement* category, the indicators were *increased diligence of residents in reducing damages* (ID), and *management efforts by NGOs to curb damages* (ME). In the *consumer health & safety* category, the indicators were *a lower incidence of water-related illnesses* (LI), and *reduction of chlorine taste in water* (RC). Lastly, in the *consumer feedback mechanism* category, the indicators were *more sustainable septic tanks to reduce the need for repairs* (MS) and *increased awareness of respective districts' NGO services* (IA). The alternatives were three wastewater treatment scenarios in Zaatari RC.

As the S-LCA in this study favors non-objective data, it was necessary to consult experts in the field for their opinions on the social impacts of wastewater systems and WASH facilities in RCs. The experts surveyed for this research assisted in the weighting of social criteria based on their judgments on the importance of several social impacts on the refugee communities when subjected to different water treatment methods. The experts were selected based on their experience in Zaatari RC and/or WASH management in similar temporary settlements. 16 respondents of varied occupations and locations were approached to complete the survey including two pilot surveys, with a final number of 8 experts being chosen for the AHP due to their high relevance in expertise and location. The small number of 8 survey respondents were chosen to provide more knowledgeable judgment in the criteria weighting from experts who were directly involved in Zaatari RC or worked in the camp. Out of the 8 respondents, 6 were based in Jordan while 2 were based overseas. In terms of their occupations, 5 were engineers while the other 3 were either NGO officers or WASH advisors.

The survey was carried out individually, using an online questionnaire to solicit responses. The questions included ranking the importance of the different criteria through pairwise comparisons of the elements in each level of the hierarchy using a scale from 1 to 5, where 1 meant equal importance between the two criteria compared, 2 meant that one is moderately more important than the other, 3 indicated that one is more important, 4 indicated that one is much more important, and 5 meant one option is extremely more important than the other. In the original development of AHP by Saaty (1980), a scale of 1-9 was used for pairwise comparisons. This scale was adjusted to be 1-5 in this study due to certain limitations of the original Saaty scale when conducting pairwise comparisons. A study by Aupetit and Genest (1993) suggested reducing the scale to 1-5, as the number of pairwise comparisons (n*(n-1)/2) may become very large when using the Saaty (1-9) scale. Furthermore, past studies have concluded that users (i.e. individuals surveyed) may not consider their past assigned value when giving new input value; which in turn creates inconsistency (Hossain et al., 2014), especially when the scale of judgment is large as it becomes a lengthy task (Macharis et al., 2004). Hence, a smaller scale was used in this study to reduce inconsistency in the responses.

The pairwise comparison of elements within each level resulted in a pairwise comparison matrix, whose elements are normalised into a normalized column matrix and then averaged to get the local relative weight of each element at each level. An example of the comparison questions for the set of elements in Level 2 (Community) is: “Which of the two (safe and healthy living conditions or equity) has a greater influence on the social implications of a selected sewage treatment method for the camp? By how much more? (Choose 1-5 on the scale)”. As Level 2 (Community) consisted of three elements, three questions were asked for the comparison of the three elements, two at a time. Therefore, for every set of n elements, there were n*(n-1)/2 pairwise comparisons. A sample of the questionnaire can be found in Appendix G. The judgements for each pairwise comparison were collected and the calculations of the respective weights of the elements in each level of the hierarchy were performed for each expert. For sets of comparisons with $n > 2$, a consistency ratio (CR) was calculated using the AHP method (Saaty, 1980) as shown in Appendix H. The threshold for CR is typically set below 0.10. CR is dependent on matrix size (Wedley, 1993) and for a greater matrix size, a higher CR is acceptable. Furthermore, to account for a wide range of responses in group surveys, a higher CR is accepted (Ho et al., 2005). Hence the threshold for CR was set at 0.2, as done in past studies conducted using AHP to account for the wide range of responses from experts in different fields (Ho et al., 2005; Kumar et al., 2009). From the AHP, the overall weights for each social indicator were finally calculated by multiplying the relative local weights for each element in descending order from Level 1 to Level 3 to obtain a single overall weight (%) for each indicator. All overall weights (%) from respondents were then averaged. In the present study, an evaluation of the social impact indicators was done through a separate pairwise comparison to determine the ratings of the different social impact indicators through defining numerical values to non-quantitative data. After pairwise comparison, a normalized column matrix was produced, averaged, and an ideal mode approach was applied to calculate a rating for each social impact corresponding to the different water and wastewater treatment scenarios. All ratings from all respondents were then averaged for each social impact indicator. The averaged overall weights (%) from all respondents were multiplied with the averaged ratings of the social impact indicators to get a final social benefit score for each indicator. The indicator with the highest final social benefit score thus had the largest social benefit in this study.

### *2.4 Socio-eco-efficiency analysis*

The normalized results of the LCA, S-LCA, and LCC were inputted into the OriginPro software to obtain a ternary diagram. Ternary diagrams have been widely used, especially in the field of chemistry, to plot the composition of a mixture of 3 components (Stringfellow & Greene, 1969). The minimum factor chosen was 0.

### *2.5 Scenarios*

To conduct the EEA, three wastewater treatment scenarios were considered for Zaatari RC (Figure 2). Scenario 1 (S1) represents the original WWT operation in Zaatari RC upon its establishment in 2012. In S1, groundwater (approximately 3,600 $m^3$/d) was drawn from 3 boreholes, then chlorinated and distributed via 82 water trucks into the camp (UNICEF & REACH, 2017, and UNHCR, 2016). About 2,100 $m^3$ of wastewater was generated daily, from which 20% was transported by desludging trucks and treated by a municipal WWTP approximately 45 km away that employed WSPs. As of 2016, the municipal WWTP had already approached its capacity (MWI, 2016a). The remaining 80% of the wastewater was treated by an on-site MBR and TF containerized package plants to form potable water. Effluent from the WWTP was used in irrigation of crops (USAID, 2005).

Scenario 2 (S2) is the UNHCR long-term plan and current operation of WWT in Zaatari RC where groundwater (approximately 3,800 $m^3$/d) is drawn from 4 boreholes then chlorinated and distributed via a simplified piped network (van der Helm et al., 2017). The 260km integrated pipe network supplies water at the household level. In addition, the simplified sewer WWN and bathing units are improved along with private WASH infrastructure at the household level, increasing the percentage of households having at least one private toilet from 91% in 2015 to 98.4% in 2017 (UNICEF & REACH, 2017). This is a vast improvement from 2013, where there was only 1 toilet for every 50 people (IMC & UNICEF, 2013). In the present study, it was assumed that about 2,217 $m^3$ of wastewater generated daily is treated at an on-site MBR and TF containerized package plants (UNICEF, 2014). Effluent is used for irrigation of crops.

Scenario 3 (S3) incorporates effluent water reuse into the UNHCR long-term plan. It was assumed that about 2,534 $m^3$ of wastewater is treated daily by an on-site MBR and TF containerized package plants with a reuse option of the effluent water in the camp. The higher wastewater quantity in S3 compared to S2 was deduced from the wastewater production per water supply in S1 multiplied by the assumed water consumption percentage in S3 (part (v) found in Appendix C Table C.3). As the majority of the water supply is used in bathing (29.4%), the assumption that a higher flowrate is diverted to the WWN and bathing units rather than to the simplified piped network was made. The higher water supply is attributed to the increase in wastewater reuse as effluent. Hence, with a higher water supply in S3 compared to S2, the wastewater production increases. Higher water production is considered for S3 as the current daily consumption in Zaatari RC falls significantly below the daily consumption in Jordan. The daily limit of water consumption per capita in Zaatari RC is 35 liters per day (UNHCR, 2020), which is only 29% of the average urban water usage of Jordanian citizens at 120 liters per person per day (Water Authority of Jordan, 2010). Hence, there is currently a discrepancy in the average daily water usage for each resident in Zaatari RC. Effluent for non-potable water use (i.e. toilet flushing or usage in washing) is supplied to the households through the WWN with flush toilets and bathing units (UNHCR, 2018). Groundwater is obtained via a borehole then chlorinated and distributed by the 260km simplified piped water network (removing the need for 3 additional boreholes in S2 and 2 additional boreholes and the water trucks in S1). The enhancement of septic tanks through household plumbing upgrades is also incorporated into S3 as proposed in the UNHCR WASH manual (UNHCR, 2018). This scenario aims to continue the usage of available treatment plants and the UN long-term plan water network with added reuse of effluent treated by the on-site WWTP.

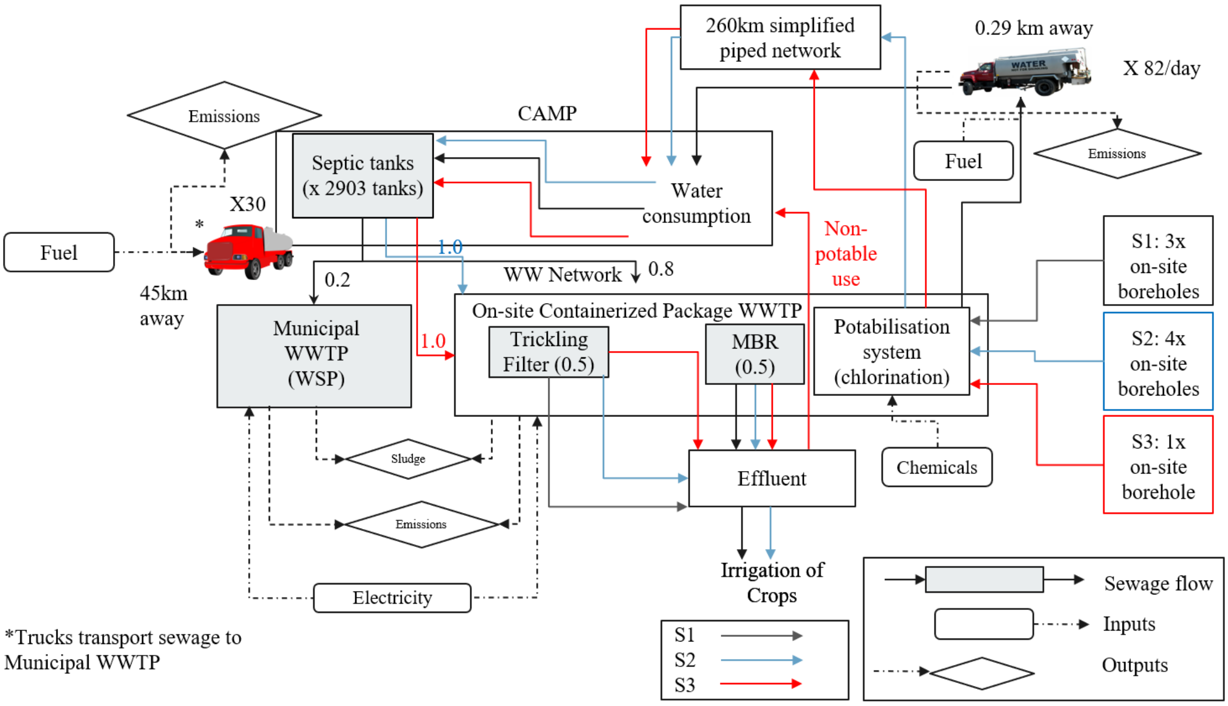


*Figure 2- Process flow and boundaries of scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3). Solid arrows show the sewage flow. Inputs are shown in rectangles followed by dotted arrows. Outputs are shown in diamonds with dotted arrows. MBR: membrane bio-reactor; WWTP: wastewater treatment plant; WW: wastewater; WSP: waste stabilization pond.*

## 3. Results and discussion

### 3.1 Eco-efficiency analysis

#### 3.1.1 Life cycle assessment

Groundwater consumption was particularly relevant for this study due to the severe water scarcity in Jordan. In order to assess the LCI of groundwater consumption, the water extracted from groundwater was calculated based on the water pumped from boreholes in each scenario. To deduce the life cycle impact, the lower the percentage of water consumption attributed to groundwater consumption, the more eco-efficient the scenario is deemed to be. The groundwater consumption ($m^3$/day) from pumped boreholes is 3,600 for S1 (ACTED et al., 2014), 3,800 for S2 (Van der Helm et al., 2017), and 950.4 for S3 (estimated from ACTED et al., 2014). The addition of effluent reuse in S3 also increases the daily total water supply to 4,344.4 $m^3$/d, which increases the water consumption limit to 54.3 liters per person per day. Another assumption made in the LCI is the percentage of water usage in the RC (Cronin et al., 2008) to account for the quantity of potable water needed, which is assumed to be 26.4% of the total water usage attributed to drinking, cooking, and rearing animals (part (v) of Appendix C Table C.3).

For WSP in S1, the electricity consumption was calculated from the electrical consumption rate in Jordan (USD/kWh) and the electrical costs from the water network studies in Zaatari RC (ACTED et al., 2014). For the MBR (in S1, S2, and S3), the average electricity consumption of an MBR was calculated based on the different input flows using electricity consumption data gathered from a 2014 UNICEF report and Shin et al. (2014). For TF (in S1, S2, and S3), the electricity consumption was estimated using primary data on electrical costs (ACTED et al., 2014) and on the average electricity consumption of TF according to the different input flows of each scenario (Young and Koopman, 1991). Lastly, for borehole pumps (in S1, S2, and S3), electricity consumption data was drawn from the water network studies on Zaatari RC (ACTED et al., 2014).

The normalized LCIA of the three scenarios is presented in Figure 3. Of these scenarios, the one with the lowest environmental impact in water depletion is S3. This is due to the reuse option in S3, which offsets the increased water usage of the camp overall by 37.5% (reuse of 950.4 $m^3$/d of 2,534 $m^3$/d of wastewater produced). For human toxicity, S2 and S3 had zero impacts due to the removal of the usage of water trucking services, hence reducing $CO_2$, $NO_x$, CO, and other emissions to air from transportation. Other sources of emissions leading to human toxicity include heavy metals released into the environment. For freshwater eutrophication, S3 does not include the option of irrigating crops, hence removing the possibility of freshwater eutrophication from untreated nitrates and phosphates. Lower agricultural land occupation is also seen in S2 and S3 due to the removal of WSP usage and further removal of borehole (including elevated water tanks) in S3.

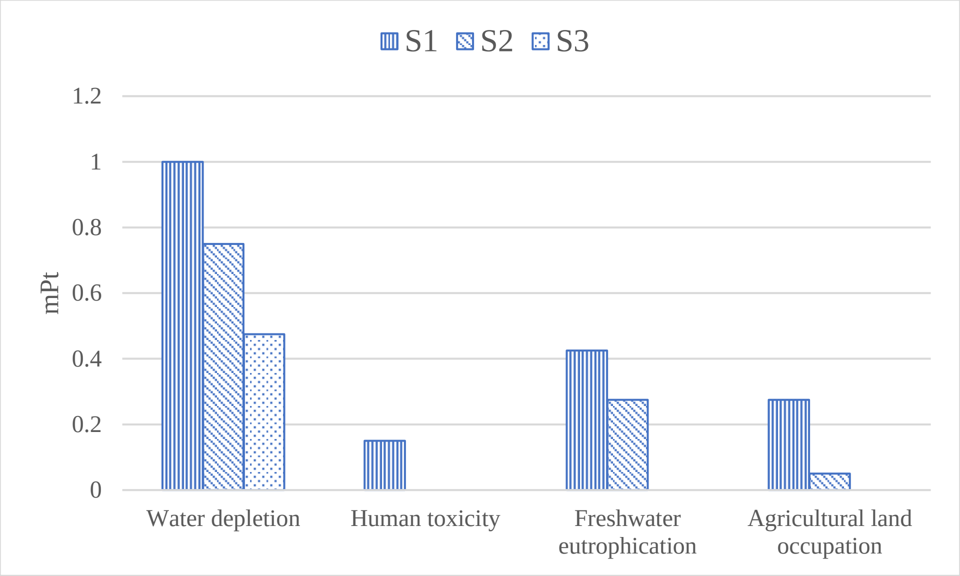


*Figure 3- ReCiPe Endpoint normalized results for scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3).*

An environmental portfolio was derived from the results of the LCIA. The values denote the relative performance of each scenario (0 to 1). The closer to the center the values are on the environmental portfolio, the less impact the scenario has on the environment.

As seen in Figure 4, S2 has the least environmental impacts overall except in 'water depletion' and 'freshwater eutrophication', where S3 instead has the least impact. For S1, the process which contributes the most to direct emissions was the WSP, as the amount of air emissions in the form of $N_2O$ from untreated $NH_4^+$ is much higher than the emissions of $N_2O$ from the on-site WWTP, which was the main treatment option in S2 and S3. For water emissions, the high impact of freshwater eutrophication in S1 is due to the possibility of untreated nitrates and phosphates spreading during the irrigation of crops. In S2 and S3, the emission with the highest impact was COD. However, this result is attributed to the higher wastewater effluent flow of the on-site WWTP in both S2 and S3 as compared to S1. Ecotoxicity was found to be higher in S1 (and attributed to soil emissions due to heavy metals) than in S2 and S3 (also attributed to soil emissions but due to irrigation), while emissions to water and atmosphere contributed to eutrophication, climate change, and acidification, which were all the highest in S1 due to higher $N_2O$ emissions. A limitation of the LCIA results derives from the usage of soil emissions from the ecoinvent database, as primary data was not available. Hence, the results of impact categories affected by emissions to soil due to heavy metals should be interpreted with caution.

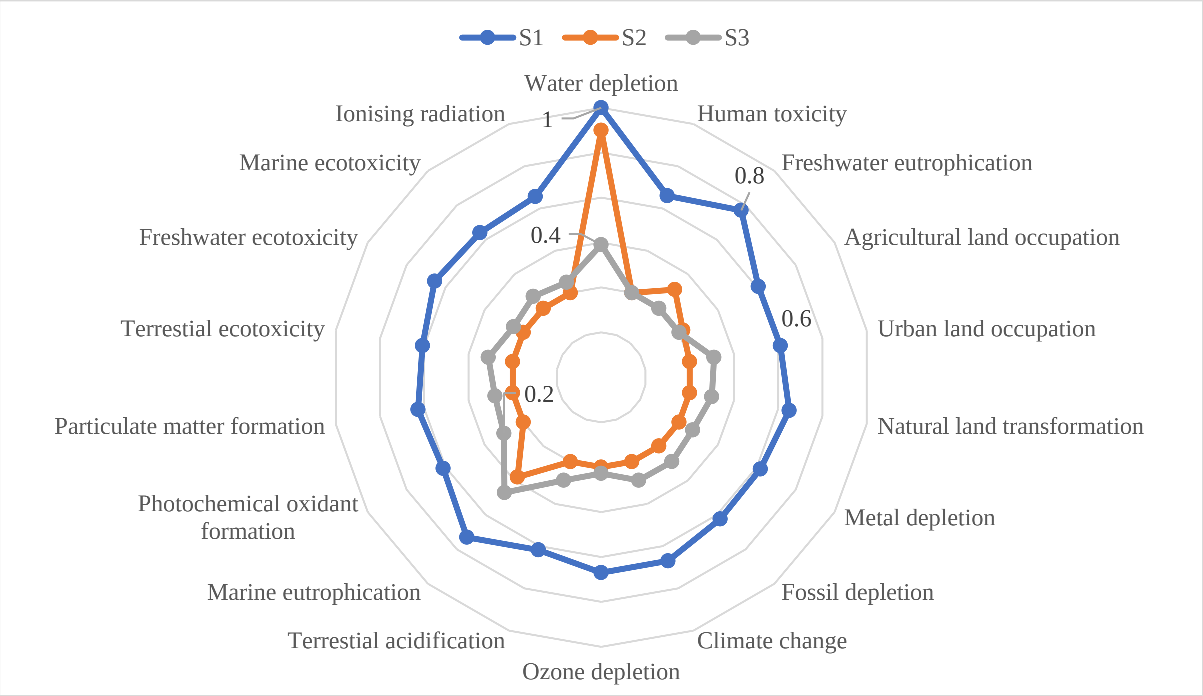


*Figure 4- Environmental portfolio of LCAs for scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3).*

### *3.1.2 Life cycle cost*

The results of the LCC show that S1 accounted for 21.13 million USD/year from which 7.86, 0.19, 12.71, and 0.37 million USD/year were from water trucking, boreholes, on-site WWTP, and WSP, respectively. In S2, only boreholes with 0.25 million USD/year and on-site WWTP with 12.71 million USD/year comprised the total cost of 12.96 million USD/year. At a lower total cost of 11.3 million USD/year, S3 included 0.06 and 11.24 million USD/year for boreholes and on-site WWTP, respectively. Thus, S3 saves 9.83 and 1.66 million USD/year compared to S1 and S2, respectively. Appendix E shows the data and assumptions made for the LCC.

Figure 5 shows that S3 performs most favorably with the lowest LCC, while S1 has the highest costs. The LCC incurred by S2 represents savings of 8.17 million USD/year compared to S1; thus, the UNHCR long-term water cycle plan has significant improvements over S1 while the proposed scenario S3 is the most cost-effective The most significant cost avoided in S2 and S3 is the water trucking service, with the LCC for on-site WWTP in S3 being lower than that for either S2 and S1, due to the reuse of effluent, leading to 1.47 million USD/year in avoided costs (USEPA, 2016).

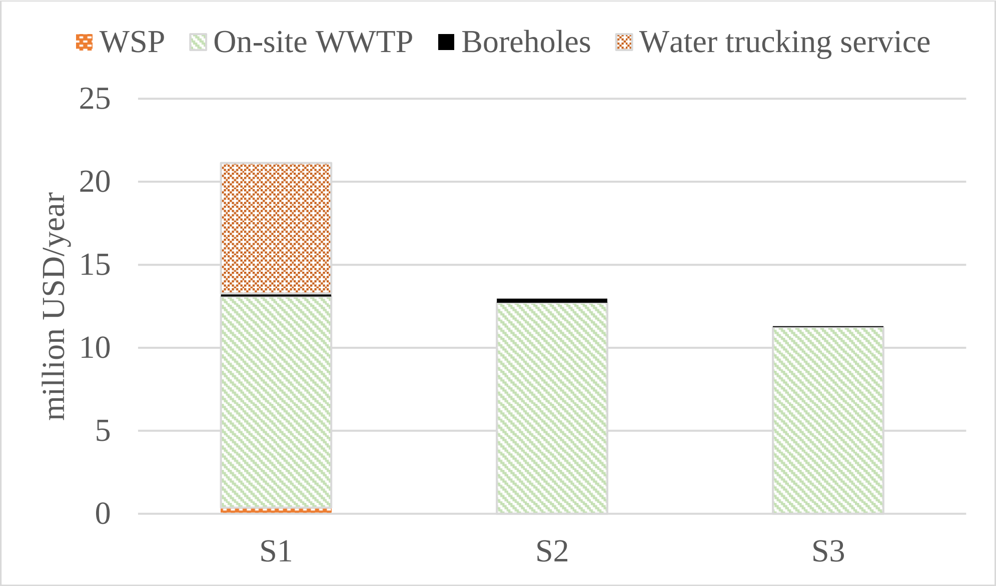


*Figure 5- Life-cycle cost (million USD/year) for scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3). WSP: waste stabilization pond; WWTP: wastewater treatment plant.*

### *3.1.3 Eco-efficiency portfolio*

The eco-efficiency portfolio was calculated using the results of the LCC and LCA to determine the position of each scenario (S1, S2, or S3) in the portfolio as presented in Figure 6. S1 is in the completely eco-inefficient area of the portfolio, S2 is in the completely eco-efficient area of the portfolio, and S3 is in the half eco-efficient area. The EEA results show that S2 is 14%

and 12% more environmentally friendly than S1 and S3, respectively. In terms of costs, S3 had 43% and 7% better performance than S1 and S2, respectively.

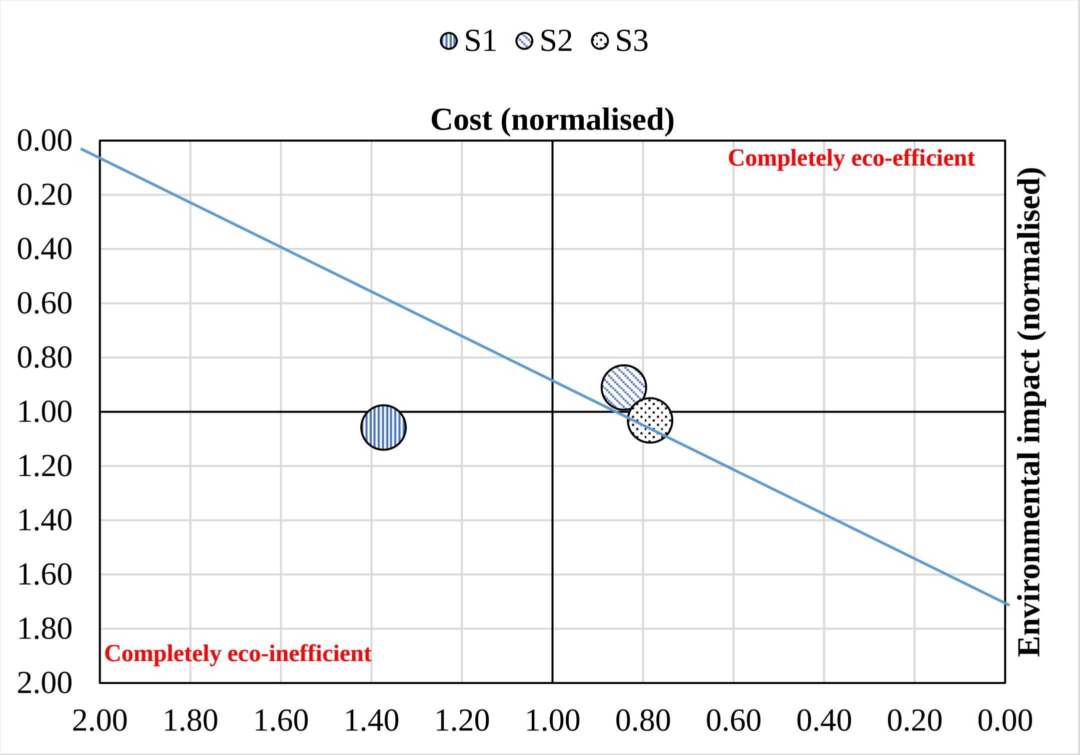
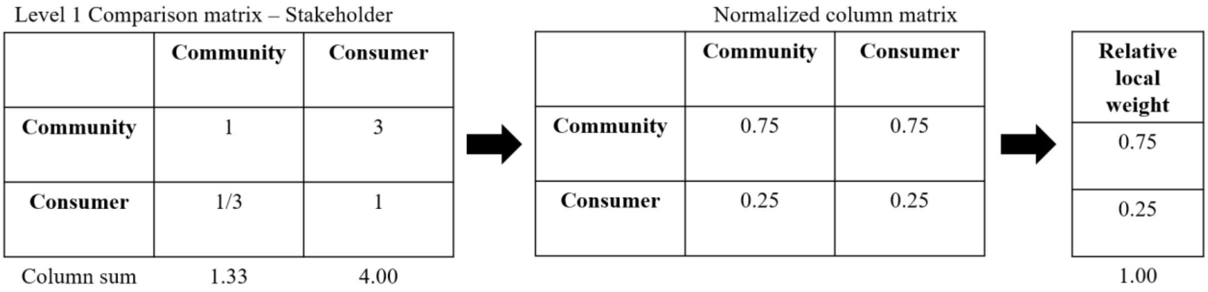


*Figure 6 - Eco-efficiency portfolio for scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3).*

### *3.2 Social life cycle assessment*

Survey responses from individual experts were incorporated into the AHP starting with pairwise comparisons of elements within each level, then a normalized-column matrix, which is then averaged into the relative local weight for each indicator. The results for Level 1 (Community or Consumer) for one respondent is shown in Figure 7. By averaging all the relative local weights of all elements from responses in the survey, an averaged relative local weight can be calculated for each element across all levels. The averaged local relative weights across all levels are shown in the hierarchy in Figure 8.



Level 1 Comparison matrix – Stakeholder

| | Community | Consumer |
|---|---|---|
| Community | 1 | 3 |
| Consumer | 1/3 | 1 |
| Column sum | 1.33 | 4.00 |

Normalized column matrix

| | Community | Consumer |
|---|---|---|
| Community | 0.75 | 0.75 |
| Consumer | 0.25 | 0.25 |

| Relative local weight |
|---|
| 0.75 |
| 0.25 |
| 1.00 |

*Figure 7- Comparison matrix, normalised-column matrix, and relative local weight of each element in Level 1 for one respondent.*

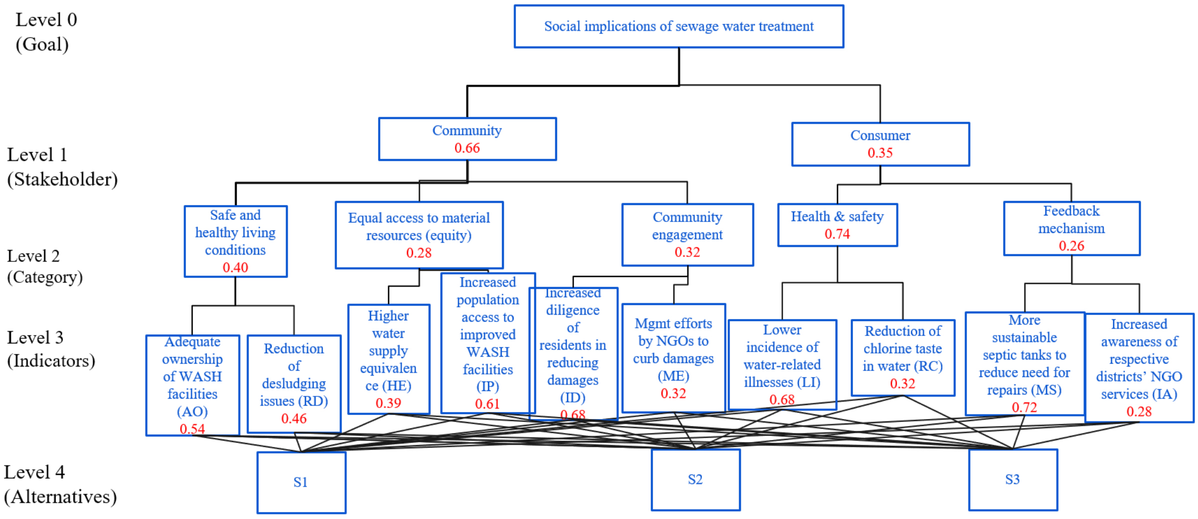


*Figure 8 - AHP hierarchy used in the S-LCA with the relative local weights of each stakeholder, social impact category and indicator.*

### 3.2.1 Overall weight (%) of social impact indicators

The AHP responses were analysed to produce the overall weights of the different social impact indicators. The overall weight (%) of each social impact indicator in Table 2 was derived by multiplying the local relative weight of each element in Level 1 with the local relative weight in Level 2 and finally the local relative weight in Level 3. For example, for Level 3 - IA, the local relative weight for the *consumer* was multiplied by the local relative weight of *feedback mechanism*, and finally multiplied with the local relative weight of IA. Thus, the overall weight was 2.55% for that indicator. According to the experts surveyed, the most important social impact indicators with the highest weights were LI (17.61%), ID (14.36%), and AO (14.26%).

*Table 2. Summarised results from the AHP indicating the overall weight of each social impact indicator averaged from all respondents*

| Stakeholder | Community | | | | | | Consumer | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Category | Safe & healthy living conditions | | Equity | | Community engagement | | Health & safety | | Feedback mechanism | |
| Indicator | **AO** | **RD** | **HE** | **IP** | **ID** | **ME** | **LI** | **RC** | **MS** | **IA** |
| Weight (%) | 14.26 | 12.14 | 7.21 | 11.27 | 14.36 | 6.76 | 17.61 | 8.29 | 6.55 | 2.55 |
| Total sum (%) | 100 | | | | | | | | | |

### *3.2.2 Evaluation of social impact indicators*

For each social impact indicator in Level 3, a separate pairwise comparison was generated through attributing numerical values to a conceptual scale (Opher et al., 2017) and applying an ideal mode approach to obtain a rating for each social impact indicator. The evaluation process for each social impact indicator is described in depth in Appendix I. Pairwise comparison was performed by creating a 1-5 scale of each indicator, and responses were gathered to produce the pairwise comparison matrix, followed by the normalized column matrix, a priority vector, and finally the ratings where the highest priority score in the priority vector was set as 1.00 and the other scores were calculated proportionally. Table 3 was generated by compiling all the ratings of the social impact indicators in Appendix I.

*Table 3. Ratings of social impact indicators*

| | Community | | | | | | Consumer | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Safe & healthy living conditions | | Equity | | Community engagement | | Health & safety | | Feedback mechanism | |
| | **AO** | **RD** | **HE** | **IP** | **ID** | **ME** | **LI** | **RC** | **MS** | **IA** |
| **S1** | 0.2 | 0.2 | 0.16 | 0.16 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 |
| **S2** | 0.2 | 0.2 | 0.41 | 0.41 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 | 0.2 |
| **S3** | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 | 1.00 |

*AO = Adequate ownership of WASH facilities, RD = Reduction of desludging issues, HE = Higher water supply equivalence, IP = Increased population access to improved WASH facilities regardless of district, ID = Increased diligence of residents in reducing damages, ME*

*= Management efforts of NGOs to curb damages, LI = Lower incidence of water-related illnesses, RC = Reduction in chlorine taste in water, MS = More sustainable septic tanks to reduce the need for repairs, IA = Increased awareness of respective districts' NGO services*

### *3.2.3 Final social benefit scores and interpretation*

The final social benefit scores as shown in Figure 9 were deduced by multiplying the overall weight (%) of each indicator as calculated in Table 2 with the rating of each social impact indicator as calculated in Table 3. The scenario with the highest and most beneficial social score based on the expert judgements and social impact indicators ratings is S3.

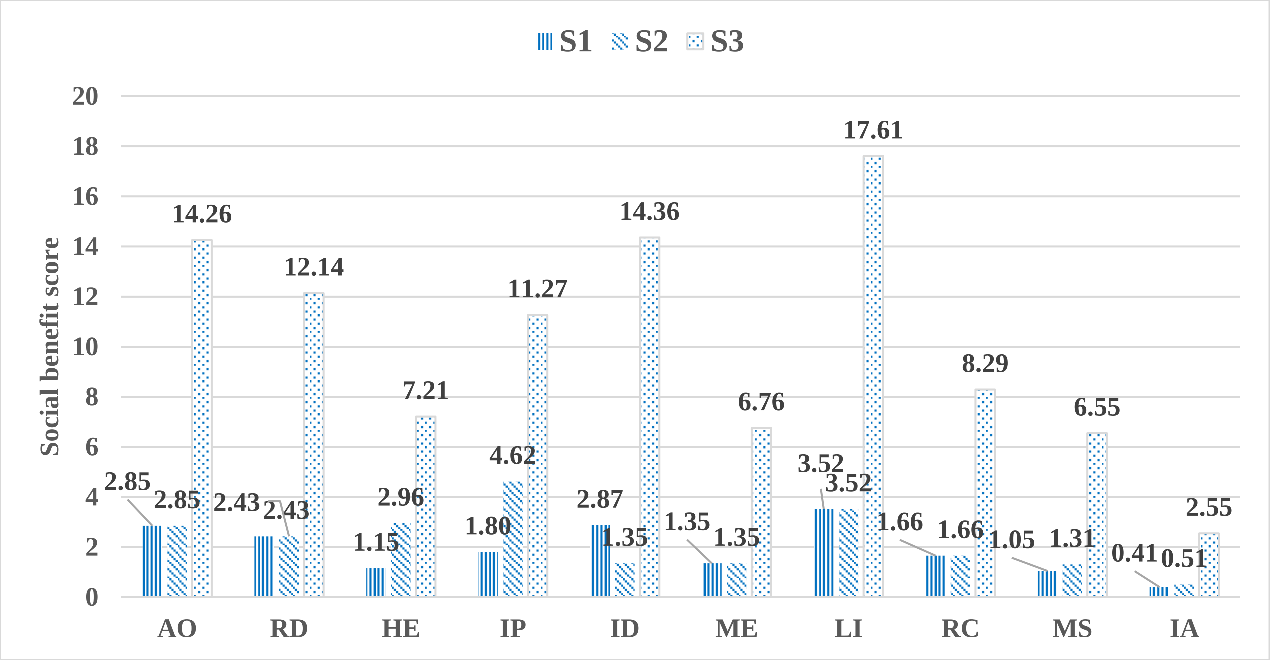


*Figure 9- Final social benefit scores for S-LCA indicators for scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3).*

## *3.3 Socio-eco-efficiency analysis*

According to the UNEP/SETAC (2011), it is not recommended to aggregate the results of the three life cycle methodologies (LCA, LCC, and S-LCA). However, it is our opinion that some degree of aggregation is necessary to provide holistic sustainability information to stakeholders and to further develop multi-objective methodologies. For the SEEA a ternary diagram was generated to integrate the results of the S-LCA and the EEA. The normalized results of the S-LCA, LCC, and LCA were combined in OriginPro, producing a ternary diagram that shows the normalized social impacts, environmental impacts, and costs of S1, S2, and S3. The most socio-eco-efficient scenario is closest to the bottom left side of the diagram (Point B), whereas the least socio-eco-efficient scenario is closest to Point A of the diagram. Figure 10 shows the respective positions of S1, S2, and S3 on the socio-eco-efficiency ternary diagram. The squares represent points normalized from the SEEA results. The summation of the normalized values for EEA and S-LCA scores must add up to 1 for each scenario. The diagram illustrates that S3 is positioned closest to the 'Most socio-eco-efficient' (Point B) end whereas S1 is positioned closest to the 'Least socio-eco-efficient' (Point A) end. Based on the normalized results, S3 is 57% more environmentally friendly than both S1 and S2. In terms of costs, S3 exhibits 75% and 65% better performance than S1 and S2, respectively. Lastly, S3 is 57% and 44% more socially sustainable than S1 and S2, respectively.

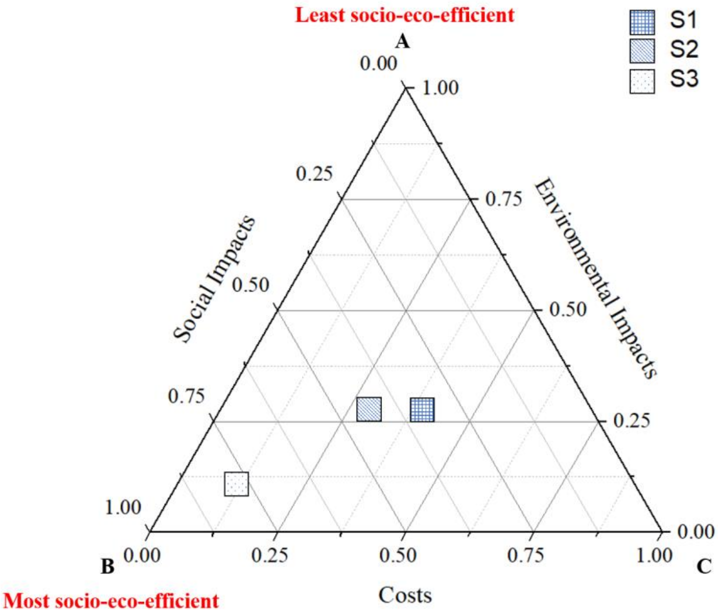


*Figure 10- Socio-eco-efficiency ternary diagram showing the normalized social impacts, environmental impacts, and costs for scenario 1 (S1), scenario 2 (S2), and scenario 3 (S3).*

### *3.4 Limitations and challenges*

The limitations of the study include the exclusion of the construction phase and end-of-life phase in the LCA, i.e. dismantling of the camp. This was explicitly done as there is a lack of certainty in the dismantling impacts of a refugee settlement. Previous LCA studies for urban water systems also show that infrastructure construction and end-of-life phases cause negligible impacts when compared to the operation phase (Friedrich, 2002; Lundie et al., 2004). There are several limitations inherent to the nature of the LCA in connection with the inventory and impact assessment methodologies. Regarding the inventory, the heavy metals considered in the soil emissions and the percentage of water usage in RC were not directly measured in Zaatari RC. Also, emergent contaminants have not been included in the calculations.

In terms of the S-LCA, the data collected could be improved by conducting surveys with the main stakeholder (refugees) to gain refugees' inputs and perceptions. It is important to note that in the survey, the indicators given were relative statements such as 'lower incidence of water-related illnesses' and 'higher water supply equivalence' as compared to the current situation in the RC. Users of this method may find that the use of relative or comparative statements such as the above may promote bias in the responses, thus indicators with independent statements are recommended for future studies. Independent statements can help reduce uncertainty in the responses as different individuals may have different perspectives on the current situation in the RC. Another limitation to note would be the non-inclusion of questions that capture the potential unwillingness of residents in using reclaimed water, expressing the downside of water reuse in the RC. This might bias results when gathering information on expert views on using reclaimed water, as consumer preferences or concerns could not be collected in the S-LCA. Hence, a more balanced approach should be taken when designing the questionnaire in future research.

The SEEA results must be interpreted cautiously as the use of OriginPro adds a normalization step which requires the components to add up to 100%. Using primary data is recommended to improve the overall accuracy of the SEEA when applied to similar refugee community cases. Furthermore, the results of this study must be interpreted cautiously because single score results usually include several assumptions that can lead to increased uncertainty.

## 4. Conclusion

The SEEA framework can be used in a broader context because it provides a means for complementing the efforts of current wastewater treatment research that analyzes just one or two aspects of the tripartite model of sustainability.

As clearly illustrated through a ternary diagram, the SEEA identified that the proposed scenario with non-potable water reuse integrated into the UN long-term plan (S3) is environmentally, economically, and socially advantageous as a wastewater treatment method alternative for Zaatari RC when compared to the original wastewater management approach in Zaatari (S1) and the UN long-term plan consisting of the installation of a simplified piped network (S2). The main characteristics of S3 are the reuse of treated effluent for non-potable activities, reduced need for chlorination, enhancement of the WW network, and a wastewater treatment system that consists of on-site MBR and TF containerized package plants.

As distinct from the results of the EEA, which identified S2 as the most eco-efficient scenario, the holistic approach from the SEEA identified S3 as the most socio-eco-efficient scenario. This demonstrates that the lack of social considerations present in EEA may affect recommendations for decision-making and is therefore an important addition to the overall assessment.

## Acknowledgments

We are very grateful to Dr. Tamar Opher for her consultations on S-LCA and AHP, as well as the experts who took part in the S-LCA survey for contributing their time and expertise.

## References

Abdallah, M., Shanableh, A., Elshazly, D., & Feroz, S. (2020). Techno-economic and environmental assessment of wastewater management systems: Life cycle approach. Environmental Impact Assessment Review, 82, 106378. doi:10.1016/j.eiar.2020.106378

Abu-Allaban, M., El-Naqa, A., Jaber, M., & Hammouri, N. (2014). Water scarcity impact of climate change in semi-arid regions: A case study in Mujib basin, Jordan. *Arab. J. Geosci., 8*(2), 951-959. doi:10.1007/s12517-014-1266-5

ACTED. (2017, March 16). First step completed towards Zaatari's wastewater network - Jordan. Retrieved July 22, 2020, from https://reliefweb.int/report/jordan/first-step-completed-towards-zaatari-s-wastewater-network

ACTED, JEN, OXFAM, UNICEF, & UNHCR. (2014). *Water Network studies for Zaatari Camp*. Lecture presented at Zaatari Water Network Technical Working Group in Jordan. Retrieved from https://data2.unhcr.org/en/documents/details/40598.

Alnsour, J., & Meaton, J. (2014). Housing conditions in Palestinian refugee camps, Jordan. Cities, 36, 65-73. doi:10.1016/j.cities.2013.10.002

Atmaca, A., & Atmaca, N. (2016). Comparative life cycle energy and cost analysis of post-disaster temporary housings. *Appl. Energ.,171,* 429-443. doi:10.1016/j.apenergy.2016.03.058

Aupetit, B., & Genest, C. (1993). On some useful properties of the Perron eigenvalue of a positive reciprocal matrix in the context of the analytic hierarchy process. *Eur. J. Oper. Res.,70*(2), 263-268. doi:10.1016/0377-2217(93)90044-n

BASF. (2018). Eco-Efficiency Analysis. Retrieved from https://www.basf.com/global/en/who-we-are/sustainability/management-and-instruments/quantifying-sustainability/eco-efficiency-analysis.html.

Byrne, D. M., Lohman, H. A., Cook, S. M., Peters, G. M., & Guest, J. S. (2017). Life cycle assessment (LCA) of urban water infrastructure: Emerging approaches to balance objectives and inform comprehensive decision-making. Environmental Science: Water Research &amp; Technology, 3(6), 1002-1014. doi:10.1039/c7ew00175d

CDC. (2014). *Chlorine Residual Testing* (United States, Centers for Disease Control and Prevention).

CEIC. (2017). Jordan: JO: Discount Rate: End of Period: Economic Indicators: CEIC. Retrieved July 23, 2020, from https://www.ceicdata.com/en/jordan/money-market-and-policy-rates-annual/jo-discount-rate-end-of-period

Costa, M. P., Schoeneboom, J. C., Oliveira, S. A., Viñas, R. S., & Medeiros, G. A. (2018). A socio-eco-efficiency analysis of integrated and non-integrated crop-livestock-forestry systems in the Brazilian Cerrado based on LCA. Journal of Cleaner Production, 171, 1460-1471. doi:10.1016/j.jclepro.2017.10.063Cronin, A. A., Shrestha, D., Cornier, N., Abdalla, F., Ezard, N., & Aramburu, C. (2008). A review of water and sanitation provision in refugee camps in association with selected health and nutrition indicators – the need for integrated service provision. *J. Water Health, 6*(1). doi:10.2166/wh.2007.019

De Faria, A. B., Sperandio, M., Ahmadi, A., & Barna, L. T. (2015). Evaluation of new alternatives in wastewater treatment plants based on dynamic modelling and life cycle assessment (DM-LCA). *Water Res., 84*, 99-111. doi.org/10.1016/j.watres.2015.06.048

Eggen, E. W. (2019). Sustainable sanitation – A case study in Yasmine and Awda informal settlements (Lebanon) (Unpublished master's thesis). Norwegian University of Life Sciences. Retrieved September 11, 2020, from https://nmbu.brage.unit.no/nmbu-xmlui/bitstream/handle/11250/2608112/Elisa Winger Eggen_master_2019.pdf?sequence=4&amp;isAllowed=

Fenner, R. A., Guthrie, P. M., & Piano, E. (2007). Process selection for sanitation systems and wastewater treatment in refugee camps during disaster-relief situations. Water and Environment Journal, 21(4), 252-264. doi:10.1111/j.1747-6593.2007.00071.x

Finkbeiner, M., Inaba, A., Tan, R., Christiansen, K., & Klüppel, H. (2006). The New International Standards for Life Cycle Assessment: ISO 14040 and ISO 14044. The International Journal of Life Cycle Assessment, 11(2), 80-85. doi:10.1065/lca2006.02.002

Foley, J., Haas, D. D., Hartley, K., & Lant, P. (2010). Comprehensive life cycle inventories of alternative wastewater treatment systems. *Water Res.*, *44*(5), 1654-1666. doi.org/10.1016/j.watres.2009.11.031

Friedrich, E. (2002). Life-cycle assessment as an environmental management tool in the production of potable water. Water Sci Technol 46:29–36

Garfi, M. & Ferrer-Martí, L. (2011). Decision-making criteria and indicators for water and sanitation projects in developing countries. Water Science and Technology, 64(1), 83-101. doi:10.2166/wst.2011.543

Godin, D., Bouchard, C., Vanrolleghem, P.A. (2012). Net environmental benefit: introducing a new LCA approach on wastewater treatment systems. *Water Sci. Technol.*, 65(9), 1624-1631.

GSA. (2017, August 13). 1.8 Life Cycle Costing. Retrieved from https://www.gsa.gov/node/81412.

Guereca, P., Musharrafie, A., Martinez, E., Padilla, A., Morgan, J., & Noyola Robles, A. (2011). Comparative life cycle assessment of a wastewater treatment technology considering two inflow scales. IDRC-Related Report.

Ho, D., Newell, G., & Walker, A. (2005). The importance of property-specific attributes in assessing CBD office building quality. *J. Prop. Inv. Finan.*,*23*(5), 424-444. doi:10.1108/14635780510616025

Hong J.L., Hong J.M., Otaki M. & Jolliet O. (2009, July 22). Environmental and economic life cycle assessment for sewage sludge treatment processes in Japan. *Waste Manage.*, vol. 29(2), 696-703. doi.org/10.1016/j.wasman.2008.03.026

Hossain, M., Adnan, Z., & Hasin, M. A. (2014). Improvement in Weighting Assignment Process in Analytic Hierarchy Process by Introducing Suggestion Matrix and Likert Scale. *Int. J Sup. Chain. Mgt*,*3*(4), 91-95.

IDMC. (2018, May). *GRID 2018 Global Report on Internal Displacement*(Rep.). Retrieved from http://www.internal-displacement.org/global-report/grid2018/downloads/2018-GRID.pdf.

IMC & UNICEF. (2013). Mental Health/Psychosocial and Child Protection Assessment for Syrian Refugee Adolescents in Za'atari Refugee Camp, Jordan July 2013. Retrieved from https://reliefweb.int/sites/reliefweb.int/files/resources/IMC MHPSS and CP Assessment Zaatari July 2013 final (1).pdf.

IPCC. (2007). 2.10.2 Direct Global Warming Potentials. Retrieved March 16, 2018, from https://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch2s2-10-2.html.

Kamal, B. (2017, August 21). Climate Migrants Might Reach One Billion by 2050 - World. Retrieved from https://reliefweb.int/report/world/climate-migrants-might-reach-one-billion-2050.

Khalil, N., Sinha, R., Raghav, A. K., & Mittal, A. K. (2008). UASB technology for sewage treatment in india: experience, economic evaluation and its potential in other developing countries. *Twelfth International Water Technology Conference*. Retrieved from http://www.iwtc.info/2008_pdf/15-3.PDF.

Kicherer, A., Schaltegger, S., Tschochohei, H., & Pozo, B. F. (2007). Eco-efficiency: combining life cycle assessment and life cycle costs via normalization. *Int. J. Life Cycle Ass.,12*(7), 537-543. doi:10.1065/lca2007.01.305

Kloepffer, W. (2008). Life cycle sustainability assessment of products. *Int. J. Life Cycle Ass.,13*(2), 89-95. doi:10.1065/lca2008.02.376

Knell, Y. (2014, May 24). Syrian refugees finding renewed dignity in Azraq camp. Retrieved from https://www.bbc.com/news/world-middle-east-27545397.

Kosonen, H. K., & Kim, A. A. (2018). Mental Model Approach to Wastewater Treatment Plant Project Delivery during Emergency Response. Journal of Construction Engineering and Management, 144(6), 05018005. doi:10.1061/(asce)co.1943-7862.0001488

Kosonen, H., Kim, A., Gough, H., Mikola, A., & Vahala, R. (2018). A Comparative Study on Rapid Wastewater Treatment Response to Refugee Crises. *Glob. Challeng.,*1800039. doi:10.1002/gch2.201800039

Kumar, S., Parashar, N., & Haleem, A. (2009). Analytical Hierarchy Process Applied to Vendor Selection Problem: Small Scale, Medium Scale and Large Scale Industries. *Bus. Int. J.*, *2*(2), 355-362.

Lam, C., Lee, P., & Hsu, S. (2016). Eco-efficiency analysis of sludge treatment scenarios in urban cities: the case of Hong Kong. *J. Clean. Prod.*, 112, 3028-3039. doi.org/10.1016/j.jclepro.2015.10.125

Lam, C., Leng, L., Chen, P., Lee, P., & Hsu, S. (2017). Eco-efficiency analysis of non-potable water systems in domestic buildings. *Appl. Energ.*, 202, 293-307. doi.org/10.1016/j.apenergy.2017.05.095

Lam, L., Kurisu, K., & Hanaki, K. (2015). Comparative environmental impacts of source-separation systems for domestic wastewater management in rural China. Journal of Cleaner Production, 104, 185-198. doi:10.1016/j.jclepro.2015.04.126

Lopes, T. A., Queiroz, L. M., & Kiperstok, A. (2018). Environmental performance of a full-scale wastewater treatment plant applying Life Cycle Assessment. Ambiente E Agua - An Interdisciplinary Journal of Applied Science, 13(4), 1. doi:10.4136/ambi-agua.2216

Lorenzo-Toja, Y., Rowe, V., Chenel, S., Marin Navarro, D., Moreira, M., & Feijoo, G. (2015). Eco-efficiency analysis of Spanish WWTPs using the LCA+DEA method. Water Res., 68, 651-666. doi.org/10.1016/j.watres.2014.10.040

Lu, B., Du, X., & Huang, S. (2017). The economic and environmental implications of wastewater management policy in China: From the LCA perspective. Journal of Cleaner Production, 142, 3544-3557. doi:10.1016/j.jclepro.2016.10.113

Lundie, S., Peters, G. M., & Beavis, P. C. (2004). Life cycle assessment for sustainable metropolitan water systems planning. Environ. Sci. Technol., 38(13). 3465-3473. doi.org/10.1021/es034206m

Macharis, C., Springael, J., Brucker, K. D., & Verbeke, A. (2004). PROMETHEE and AHP: The design of operational synergies in multicriteria analysis. *Eur. J. Oper. Res.*,*153*(2), 307-317. doi:10.1016/s0377-2217(03)00153-x

MWI. (2016). National Water Strategy 2016 - 2025. Retrieved from http://www.mwi.gov.jo/sites/en-us/Hot Issues/Strategic Documents of The Water Sector/National Water Strategy (2016-2025)-25.2.2016.pdf.

MWI (2016a). Jordan water sector facts and figures 2013. Jordan Ministry of Water and Irrigation. Amman, Jordan.

Nyoka, R., Foote, A. M., Woods, E., Lokey, H., O’Reilly, C. E., Magumba, F., . . . Morris, J. F. (2017). Correction: Sanitation practices and perceptions in Kakuma refugee camp, Kenya: Comparing the status quo with a novel service-based approach. Plos One, 12(12). doi:10.1371/journal.pone.0190129

Opher, T., Shapira, A., & Friedler, E. (2017). A comparative social life cycle assessment of urban domestic water reuse alternatives. The International Journal of Life Cycle Assessment, 23(6), 1315-1330. doi:10.1007/s11367-017-1356-1

Opher, T., Friedler, E., & Shapira, A. (2018). Comparative life cycle sustainability assessment of urban water reuse at various centralization scales. *Int. J. Life Cycle Ass.*, 24(7), 1319-1332. doi:10.1007/s11367-018-1469-1

Risch, E., Loubet, P., Núñez, M., & Roux, P. (2014). How environmentally significant is water consumption during wastewater treatment?: Application of recent developments in LCA to WWT technologies used at 3 contrasted geographical locations. Water Research, 57, 20-30. doi:10.1016/j.watres.2014.03.023

Saaty T.L. (1980). The analytic hierarchy process. McGraw-Hill Book Co, N.Y.

Saaty T.L., Vargas L.G. (2012). International Series in Operations Research & Management Science Models, Methods, Concepts & Applications of the Analytic Hierarchy Process. Springer Science+Business, Second New York.

Schmidt, I., Meurer, M., Saling, P., Reuter, W., Kicherer, A., & Gensch, C.-O. (2004) SEEbalance ® Managing Sustainability of Products and Processes with the Socio-Eco-Efficiency Analysis by BASF. *Greener Manage. Int.*, 45, 79-94.

Schyns, J., Hamaideh, A., Hoekstra, A., Mekonnen, M., & Schyns, M. (2015). Mitigating the Risk of Extreme Water Scarcity and Dependency: The Case of Jordan. *Water, 7*(10), 5705-5730. doi:10.3390/w7105705

Shin, C., Mccarty, P. L., Kim, J., & Bae, J. (2014). Pilot-scale temperate-climate treatment of domestic wastewater with a staged anaerobic fluidized membrane bioreactor (SAF-MBR). *Bioresource Technol.,159*, 95-103. doi:10.1016/j.biortech.2014.02.060

Shiu, H., Lee, M., & Chiueh, P. (2017). Water reclamation and sludge recycling scenarios for sustainable resource management in a wastewater treatment plant in Kinmen islands, Taiwan. Journal of Cleaner Production, 152, 369-378. doi:10.1016/j.jclepro.2017.03.110

Stringfellow, G., & Greene, P. (1969). Calculation of iii–v ternary phase diagrams: In-Ga-As and In-As-Sb. *J. Phys. Chem. Solids, 30*(7), 1779-1791. doi:10.1016/0022-3697(69)90246-7

Swarr T.E, Hunkeler D., Klöpffer W., Pesonen H-L., Ciroth A., Brent A.C. & Pagan R. (2011). Environmental life cycle costing: a code of practice. Society of Environmental Chemistry and Toxicology (SETAC), Pensacola.

UNEP/SETAC. (2013). Methodological Sheets for 31 Sub-Categories of Impact for Social LCA. Life Cycle Initiative, United Nations Environment Programme.

UNESCO. (2017). Displaced Person / Displacement. Retrieved from http://www.unesco.org/new/en/social-and-human-sciences/themes/international-migration/glossary/displaced-person-displacement/.

UNHCR. (1992). UNHCR Water Manual For Refugee Situations. Retrieved from https://www.unhcr.org/publications/operations/3ae6bd100/unhcr-water-manual-refugee-situations.html.

UNHCR. (2008, January). A Guidance For Unhcr Field Operations On Water And Sanitation Services. Retrieved from https://www.unhcr.org/49d080df2.pdf.

UNHCR. (2018, September). UNHCR WASH Manual for Refugee Settings. Retrieved July 22, 2020, from http://wash.unhcr.org/unhcr-wash-manual-for-refugee-settings/.

UNHCR. (2018a, July 02). Investing in People, Not Projects: A look at UNHCR's experimental funding. Retrieved from https://www.unhcr.org/innovation/investing-in-people-not-projects-a-look-at-unhcrs-experimental-funding/.

UNHCR. (2018b, June 25). *Global Trends Forced Displacement In 2017*(Rep.). Retrieved from UNHCR website: http://www.unhcr.org/5b27be547.pdf.

UNHCR. (2019). Zaatari Refugee Camp - Factsheet, June 2019 - Jordan. Retrieved from https://reliefweb.int/sites/reliefweb.int/files/resources/70183.pdf.

UNHCR. (2020). Step 3: Water at Za'atari Camp. Retrieved July 23, 2020, from https://www.unhcr.org/7steps/en/water/.

UNICEF. (2014). Supply & Installation of Containerized Packaged Wastewater Treatment Plants. Retrieved from https://data2.unhcr.org/en/documents/download/41966.

UNICEF & REACH. (2017). Wash Infrastructure & Services Assessment In Zaatari Camp Jordan Assessment Report March 2017. Retrieved from http://www.reachresourcecentre.info/system/files/resource-documents/reach_jor_report_zaatari_wash_infrastructure_assessment_march_2017.pdf.

USAID. (2005). *Assessment of The Upgrading Of The Mafraq Wastewater Treatment Plant* (United States, Jordan, USAID).

USEPA. (2016). *Life Cycle Assessment and Cost Analysis of Water and Wastewater Treatment Options for Sustainability: Influence of Scale on Membrane Bioreactor Systems* (United States, Environmental Protection Agency).

van der Helm, A.W.C., Bhai, A., Coloni, F., Koning, W. J., & Bakker, P. T. (2017). Developing water and sanitation services in refugee settings from emergency to sustainability – the case of Zaatari Camp in Jordan. *J. Water Sanit. Hyg. De.*, 7(3), 521-527. doi.org/10.2166/washdev.2017.107

van Kempen, E. A., Spiliotopoulou, E., Stojanovski, G., & De Leeuw, S. (2016). Using life cycle sustainability assessment to trade off sourcing strategies for humanitarian relief items. *Int. J. Life Cycle Ass.*, *22*(11), 1718-1730. doi:10.1007/s11367-016-1245-z

Water Authority of Jordan. (2010). Water Authority of Jordan - Web Presence. Retrieved March 16, 2018, from http://www.waj.gov.jo/sites/en-us/Lists/FAQs/DispForm.aspx?ID=3&Source=http://www.waj.gov.jo/sites/en-us/Lists/FAQs/AllItems.aspx&RootFolder=/sites/en-us/Lists/FAQs&ContentTypeId=0x0100CA3969498A95804496287985C46F2F24.

Wedley, W. C. (1993). Consistency prediction for incomplete AHP matrices. *Math. Comput. Model.*, *17*(4-5), 151-161. doi:10.1016/0895-7177(93)90183-y

WHO. (1996). Chlorine in Drinking-water Background document for development of WHO Guidelines for Drinking-water Quality. Retrieved from http://www.who.int/water_sanitation_health/dwq/chlorine.pdf.

World Bank. (2016). GDP per capita, PPP (current international $). Retrieved March 16, 2018, from https://data.worldbank.org/indicator/NY.GDP.PCAP.PP.CD?view=chart,.

Young, D. F., & Koopman, B. (1991). Electricity Use in Small Wastewater Treatment Plants. *J. Environ. Eng.*, 117(3), 300-307. https://ascelibrary.org/doi/10.1061/(ASCE)0733-9372(1991)117:3(300).