D3.2 Show case of the environmental benefits and risk assessment of reuse schemes

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Abstract	This report summarizes the results of Life Cycle Assessment, Water footprinting, and quantitative microbial and chemical risk assessment for selected demosites of water reuse in Europe, measuring the potential impacts of different types of water reuse on environment and human health. The case studies show that water reuse is often preferable from an environmental point of view in areas with water scarcity problems if compared to other alternatives such as water import or seawater desalination. Potential risks of water reuse for ecosystems or human health can be adequately managed if suitable processes for reclaimed water treatment are used and operated correctly. However, the study also shows the trade-offs between a higher level of reclaimed water treatment and increased environmental impacts from associated efforts in energy, chemicals and infrastructure. This inherent trade-off requires a site-specific assessment of reuse schemes to choose an adequate treatment scheme for risk management with reasonable global environmental impacts.

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Glossary

AOP	Advanced oxidation process
AT	Average lifetime
BOD	Biological oxygen demand
CED	Cumulative energy demand
COD	Chemical oxygen demand
DALY	Disability adjusted life years
DBP	Disinfection byproduct
DL	Detection limit
DOC	Dissolved organic carbon
DWTP	Drinking water treatment plant
EC	Equivalent concentration
ED	Exposure duration
EF	Exposure frequency
ET	Exposure time
ETP	Ecotoxicity potential
FEP	Freshwater eutrophication potential
GAF	Gastrointestinal absorption factor
GWP	Global warming potential
HHRA	Human health risk assessment
HQ	Hazard quotient
HTP	Human toxicity potential
IPR	Indirect potable reuse
LCA	Life Cycle Assessment
LOD	Limit of detection
LOQ	Limit of quantification
MBR	Membrane bioreactor
MEP	Marine eutrophication potential
OFWRP	Old Ford Water Recycling Plant
QCRA	Quantitative Chemical Risk Assessment
QMRA	Quantitative Microbial Risk Assessment
PPPY	Per person per year
RA	Risk assessment
RAIS	Risk assessment information system
RfD	Reference dose
RO	Reverse osmosis
SAT	Soil aquifer treatment
SF	Slope factor

SWRO	Seawater reverse osmosis
TAP	Terrestrial acidification potential
UF	Ultrafiltration
USEPA	United States Environmental Protection Agency
VOC	Volatile organic compound
WAF	Water availability footprint
WF	Water footprint
WHO	World Health Organisation
WIIX	Water Impact Index
WRP	Water recycling plant
WWTP	Wastewater treatment plant

Executive Summary

This report summarizes the results of Life Cycle Assessment, Water footprinting, and Quantitative Microbial and Chemical Risk Assessment for selected demosites of water reuse in Europe, measuring the potential impacts of different types of water reuse on environment and human health. Case studies include water reuse for agricultural purposes (Braunschweig/GER, Shafdan/ISR), non-potable reuse (El Port de la Selva/ES, Sabadell/ES, London/UK), and indirect potable reuse (El Port de la Selva/ES, Torreele/BE).

The case studies show that water reuse is often preferable from an environmental point of view in areas with water scarcity problems if compared to other alternatives such as water import or seawater desalination. Although tertiary treatment of WWTP effluent needs electricity, chemicals and additional infrastructure, the associated environmental impacts are usually lower than the efforts for water import over long distances or energy-intensive seawater desalination. However, the comparison between different alternatives for augmenting existing water supply is affected by site-specific factors and cannot be generalized for all systems. If natural water resources are available in sufficient quality and quantity, water supply from local freshwater sources can have the lowest environmental impact and should be prefered from an environmental point of view. If local sources are over-exploited or at low quality, water reuse can form a valuable and environmentally preferable alternative to minimise the additional energetic efforts and water footprint of water supply. The study also shows optimisation potential for existing water reuse schemes towards higher energy efficiency and improved management of water and nutrients in agriculture.

Potential risks of water reuse for ecosystems or human health can be adequately managed if suitable processes for reclaimed water treatment are used and operated correctly. Microbial hazards from human exposure to reclaimed water can be minimised by suitable disinfection systems and multi-barrier approaches to remove bacteria, viruses and parasites to acceptable levels. If indirect potable reuse is targeted, removal of residual trace organic substances from WWTP effluent should be realized in tertiary treatment to comply with existing regulatory guidelines for final drinking water quality. Apart from advanced technical systems (ozonation, activated carbon, membrane filtration), natural treatment during soil passage can also form a suitable and effective barrier for microbial and chemical hazards. Here, a detailed understanding of the characteristics of the subsurface passage and a close monitoring of water quality should be established to validate the performance of the natural system and provide an adequate risk management. Overall, a continuous risk assessment and management process should be established for each system of water reuse to understand the critical points of the system and keep the resulting risk below acceptable levels. Here, Bayesian statistics can be used to integrate former knowledge and new data from the system into a conclusive set of parameters for risk assessment with defined uncertainty intervals.

Linking the results of LCA and risk assessment, the study also shows the trade-offs between a higher level of reclaimed water treatment and increased environmental impacts from associated efforts in energy, chemicals and infrastructure. This inherent trade-off requires a site-specific assessment of reuse schemes to choose an adequate treatment scheme for risk management to an acceptable level while using treatment steps with reasonable global environmental impacts.

From a methodological perspective, results of LCA and Water Footprinting are affected by some uncertainty in study definitions (e.g. boundaries, functional unit), but also in underlying input data and impact assessment methods. Hence, application of these tools and interpretation of their results has to be carefully validated and discussed in light of the individual study and its characteristics.

1 Introduction

1.1 Project Background and Content of this Report

Within the DEMOWARE project, the objective of work package 3 (WP3) is the sustainability assessment of water reuse schemes. The focus is on assessing measures for mitigating risks for human health and for minimising negative environmental impacts in water reuse schemes. This task includes risk assessment (RA), Life Cycle Assessment (LCA) and assessment of the water footprint (WF) for 6 out of 10 demo sites of the DEMOWARE project. The demo sites reflect different aspects of water reuse in Europe regarding

- types of water reuse: restricted or unrestricted irrigation for agricultural reuse, reuse of water for urban usage in irrigation or street cleaning, and artificial groundwater recharge for indirect potable reuse and as a saltwater intrusion barrier
- treatment schemes: partial disinfection with UV or performic acid (PFA), filtration, chlorination, advanced oxidation processes and membrane schemes using ultrafiltration and reverse osmosis
- alternative options of water supply: enhanced groundwater pumping, water import via pipeline, or seawater desalination.

In this report, Chapter 1 gives a general overview on the selected demo sites (= case studies), their individual challenge and general information regarding the method framework of this study. Chapter 2-7 represent the different case studies, describing definitions, input data, and results of the assessments. Each of these chapters is divided into a subchapter on LCA including WF and a subchapter on RA (if conducted) together with comprehensive conclusions for each case study. In chapter 8, more general conclusions are drawn from the type of assessment in DEMOWARE and its outcomes, together with a scientific discussion of method specific aspects of water footprinting (Water Impact Index) and merging of the results of risk assessment and LCA. Finally, an attempt was made to combine the assessment approaches of LCA and risk assessment in order to investigate trade-offs between global effects and local effects and questions about environmental justice (see chapter 8.5).

1.2 Overview of Case Studies

Table 1-1 provides an overview on the different case studies investigated in this report, which have been selected in close cooperation with the partners and according to the Description of Action [1]. It should be mentioned here that a generic assessment of technologies or reuse types is not within the scope of this report, but is provided in a separate document (D 3.3 [2]). For the selected case studies, individual challenges for water management are present due to the degree of local water scarcity, the different reuse purposes, and mandatory technologies for risk reduction. These aspects lead to site-specific options of scenarios that are compared for each case study in this assessment, using the methods of RA, LCA, and WF.

In general, the case studies are located in different countries (localization in Figure 1-1) and can be grouped depending on their reuse purpose:

- The demo sites of **El Port de la Selva (ES)** and **Torreele (BE)** are directly located at the coast and face the problem of limited local freshwater resources and high seasonal water demand (touristic activity). Hence, they reuse secondary effluent of the local wastewater treatment plant (WWTP) to complement local groundwater. Artificial groundwater recharge of reused water is realized at both sites to promote **indirect potable reuse** (IPR). Part of the reused water is also used as barrier for saltwater intrusion into the local freshwater aquifer, or for local urban irrigation. Tertiary treatment of WWTP effluent is realized with different technology: whereas water reuse in El Port de la Selva is based on filtration and disinfection prior to infiltration ponds, a two stage membrane process with ultrafiltration and reverse osmosis is applied in Torreele before dune infiltration.
- The demo sites **Braunschweig (DE)** and **Shafdan (IL)** provide reclaimed water from WWTP effluent for **agriculture**. Local water scarcity is very high in Shafdan near Tel-Aviv, whereas Braunschweig exhibits a less severe water stress. Tertiary treatment of WWTP effluent is currently not required in Braunschweig, although partial disinfection can mitigate existing health risks for local workers. At Shafdan, soil-aquifer treatment (SAT) is used to polish secondary effluent, and future options to upgrade the system or enhance its capacity include advanced oxidation/ozonation and membrane treatment.
- The demo sites **OldFord Water (GB)** and **Sabadell (ES)** provide reclaimed water for **non-potable urban purposes** (toilet flushing, park irrigation, street cleaning, etc.). Coincidently, the treatment trains of both reuse sites are similar, as both include secondary treatment with a membrane bioreactor (MBR) and chlorination (see Table 1-1). The scheme at OldFord also applies a granular activated carbon system for removal of colour and trace organics.

Table 1-1: Short overview on case studies investigated in this report

GAC: granular activated carbon, SWRO: seawater reverse osmosis, PFA: performic acid, LCA: Life Cycle Assessment, WF: Water footprint, QMRA: quantitative microbial risk assessment, QCRA: quantitative chemical risk assessment

Chapter	Case Study	Reuse type/usage	Treatment technology	Alternatives assessed	Assessment in this study
2	El Port de la Selva (Spain)	Indirect potable reuse, public and private irrigation	(GAC) filtration, UV disinfection, chlorination	Membranes, water import via network, SWRO	LCA + WF: p17 QMRA + QCRA: p42
3	Braun-schweig (Germany)	Restricted irrigation in agriculture	UV or PFA disinfection	(Decoupling) water and nutrient management	LCA + WF: p66 QMRA: p90
4	OldFord Water (United Kingdom)	Park irrigation, toilet flushing	Membrane bioreactor, GAC filtration, chlorination	Water supply via drinking water system	LCA + WF: p151 QMRA: p162
5	Sabadell (Spain)	Garden and park irrigation, street cleaning	Membrane bioreactor, UV-disinfection, chlorination	-	LCA + WF: p176 QMRA + QCRA: p196
6	Shafdan (Israel)	Unrestricted irrigation in agriculture	Soil-Aquifer-Treatment, filtration and ozonation, membranes	Water supply via potable water mix	LCA + WF: p214
7	Torreele (Belgium)	Indirect potable reuse	Membranes, Soil-Aquifer-Treatment, brine treatment with willows	Water import via network or SWRO	LCA +WF: p235

Figure 1-1: Localization of case studies investigated in this report ©Google

1.3 General Framework and Methods

1.3.1 Life Cycle Assessment and Water Footprint

Review of previous LCA studies in the field of water reuse

Several authors report on carrying out an LCA of water reuse systems, also comparing different options for water treatment in their potential environmental impacts. Studies for LCA of water reuse systems include the following:

- Ortiz et al. (2007) [3]: The authors analysed different technologies for reclaiming secondary WWTP effluent with LCA. Technologies included ultrafiltration (UF) and membrane bioreactors (MBR). Results show that tertiary treatment will only slightly increase environmental impacts of WWTP, but provides new options for reusing this water and mitigate local water stress.
- Munoz et al. (2009) [4]: This paper compares different scenarios for water reuse with and without tertiary treatment of secondary WWTP effluent. The LCA focusses on toxicity-related impact categories and states that it is important to include non-regulated compounds (e.g. pharmaceuticals, personal care products) into the assessment. Water reuse is highly beneficial in environmental impacts compared to seawater desalination.
- Meneses et al. (2010) [5]: The authors compared different alternatives for disinfection of secondary effluent in their environmental impacts, targeting non-potable reuse of the reclaimed water. They concluded that non-potable reuse has particular advantages in its environmental profile when compared to seawater desalination.
- Hancock et al. (2012) [6]: This LCA study compares existing membrane desalination technologies with a novel hybrid system of forward osmosis of reclaimed secondary effluent coupled to seawater reverse osmosis. It concludes that forward osmosis may be a viable alternative to conventional membrane systems, but membrane performance has to be improved.
- Baresel et al. (2015) [7]: The authors assessed eight different treatment trains for three reuse categories with LCA, analysing agricultural reuse, groundwater recharge, and industrial reuse. They concluded that size of the reuse system has a high impact on its environmental profile, shifting the relative comparison between simple and more advanced treatment for water reclamation. Hence, target water quality and plant size should be considered when evaluating different options for water reuse with LCA.
- Pintilie et al. (2016) [8]: This study compares direct discharge of secondary WWTP effluent with tertiary treatment and water reclamation in an industrial area. It concludes that water reuse has higher impacts in most impact categories, but that it can reduce depletion of local water resources substantially.

From this short review of available literature, it can be observed that previous studies have mostly focused on the additional impacts of water reuse systems and the optimisation of their technical performance. A conclusive comparison of different site-specific alternatives for water supply with water reuse has only been attempted in very few studies. In addition, the environmental benefits of water reuse for the local freshwater resources have only been quantified in one study. Both aspects will be addressed throughout the DEMOWARE case studies, always including alternative options for water supply to compare environmental impacts of both water reuse and also other options of water supply. This will help to weight the potential additional efforts for water reclamation against those efforts that are required for alternative sources of water. Moreover, water footprinting will be used to comprehensively address and illustrate the benefits of water reclamation and reuse on the local freshwater resources.

LCA framework in this study

Life Cycle Assessment in DEMOWARE has been performed according to ISO 14040 and 14044 [9, 10]. The assessment primarily focusses on a detailed inventory on direct emissions on-site and indirect emissions from consumptives (electricity and chemicals) and materials for infrastructure. Scope and system boundaries for each case study as well as the investigated scenarios were defined by the authors in close cooperation with the local plant operator to cope with the specific characteristics of each case study. The final inventory was validated by regular exchange with operators and detailed discussion of preliminary results. These iterative cycles created a better understanding of the particular reuse system for the assessment team of KWB and CTM and also a deeper understanding of LCA and the environmental performance of their treatment scheme for the operator. Water footprinting was embedded into the scope and system boundaries of LCA and is recognized as one of different LCA impact categories.

LCA indicators

In total, eight different LCA indicators are reported for each case study. An explicit overview on LCA methods applied in this study is provided in Deliverable D3.1 [11].

LCA indicators for the impact assessment were selected from the ReCiPe methodology [12], using the midpoint approch and the Hierarchist perspective without long-term emissions (ReCiPe midpoint (H) w/o LT). In particular, global warming potential (GWP), freshwater eutrophication potential (FEP), marine eutrophication potential (MEP), and terrestrial acidification potential (TAP) have been selected from ReCiPe. In addition to these four ReCiPe indicators, cumulative energy demand (CED) of non-renewable resources (sum of fossil and nuclear CED) was selected as indicator for limited energetic resources [13].

For assessing toxicity in LCA, the USEtox® model was used with categories human toxicity (total, sum of cancer and non-cancer) and freshwater ecotoxicity, not accounting for long-term emissions (w/o LT) [14]. The assessment of toxicity in LCA is affected by several uncertainties, which demand a careful interpretation of the results and limits the strength of these results for decision making. These short-comings are discussed in detail in chapter 8.4 at the end of this report.

Selected LCA indicators are finally normalised to the average impact per European citizen (EU-27) to reveal the contribution of the investigated systems to the total environmental footprint per person. Weighting and endpoint assessment have been deliberately excluded from this LCA.

Water footprint

For water footprinting, this LCA study applied the Water Impact Index (WIIX) which reflects aspects of water quality and water scarcity [15]. As water reuse addresses explicitly problems of local water scarcity, the WIIX is expected to be useful to show the overall benefits of water reuse to the water footprint for water supply.

Accounting water volumes from/to natural freshwater

In principle, the WIIX method accounts for all water withdrawal from and water release to natural freshwater resources as impact (= withdrawal) or credit (= release). This includes e.g. the intake of groundwater or surface water as raw water for drinking water production or cooling, the discharge of WWTP effluent or cooling water to surface water, or the use of water for irrigation of agricultural fields. For the field of water reuse, it is important to note that withdrawals from or release to the technosphere (e.g. a city) are not accounted in the WIIX. If the system boundaries include e.g. only the tertiary treatment of WWTP effluent and subsequent reuse, the input water comes without WIIX impact, i.e. from the city/technosphere. The same effect can be observed if reclaimed water is used for indirect potable reuse, i.e. for drinking water production. This drinking water is an input into the city/technophere, and is thus not accounted as WIIX release. Similiarly, water withdrawal or release to marine waters is not accounted in the WIIX, e.g. intake of seawater for desalination, or WWTP discharge into ocean. Hence, water coming from ocean has no WIIX impact, and water released to ocean has no WIIX credits.

Water scarcity

For water scarcity information, the newly developed AWARE method was used [16], as it reflects the latest consensus indicator on water scarcity in LCA. AWARE provides annual average or monthly information on available water remaining in the watershed after the demand of humans and aquatic ecosystems has been met (Figure 1-2), bechmarked against the world average of water availability ("m^3 world-eq"). This use of AWARE is an update to the method description in the report D3.1 [17], where water scarcity for the WIIX was still described by the Pfister index [18]. The case studies in this report use either annual average of water scarcity or monthly values if seasonal effects are to be reflected for the scope of the study. In general, "non-agricultural" factors of AWARE are used for this LCA study.

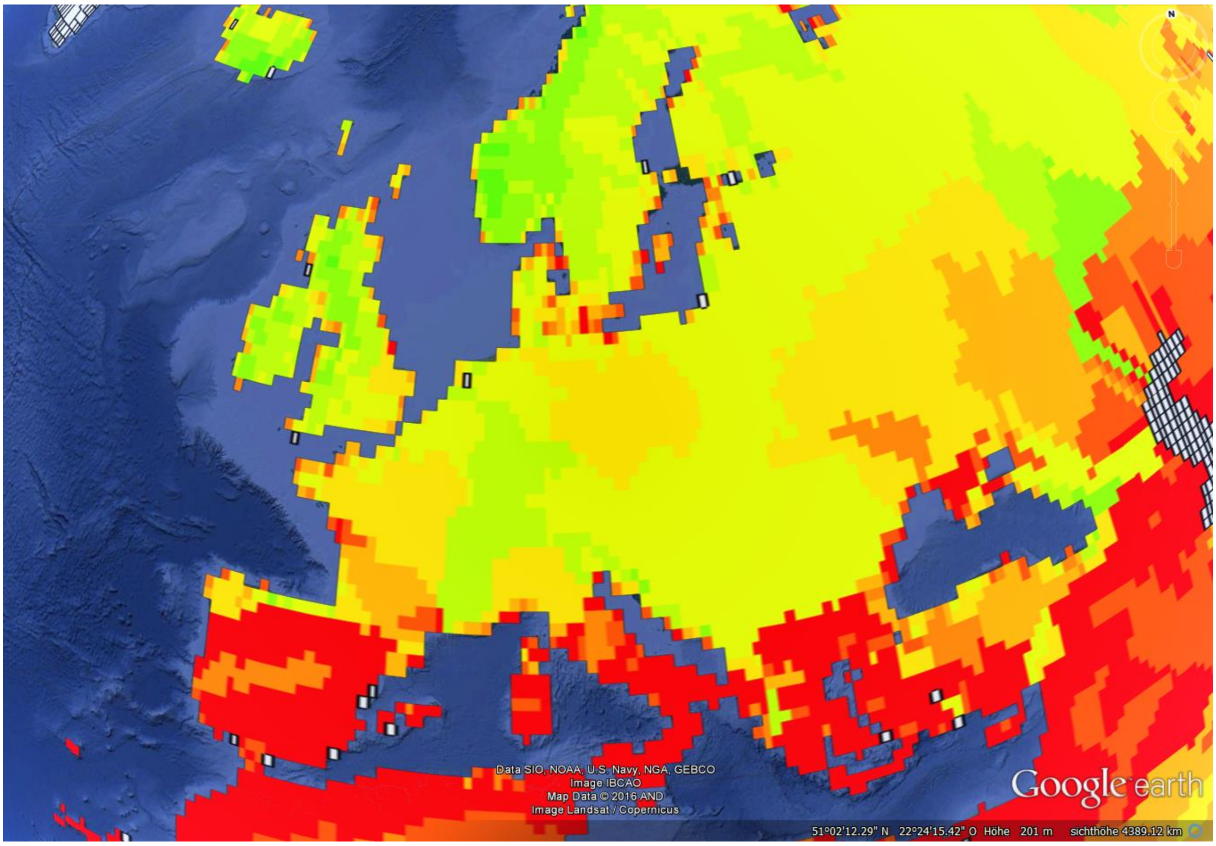


Figure 1-2: Map of annual AWARE index for Europe [16], measuring water scarcity per watershed
Water scarcity decreases with colour range from red to orange to yellow to green, map is © Google Earth

Water quality

Water quality is evaluated in the WIIX by benchmarking selected water quality parameters of the respective withdrawal or release flow against related standards for good environmental quality. These standards refer to the EC or respective national environmental quality standards for good surface water quality (e.g. [19-21]). Calculating the ratio of benchmark divided by actual concentration for each parameter, the lowest coefficient determines the water quality index (WQI) of the flow ("one-beats-all"). Hence, water quality is ranked by that parameter which is highest above its benchmark (WQI < 1), with

WQI = 1 as maximum value. This “one-beats-all” approach is very simple to apply, but has an inherent drawback: if only one substance is close to or higher than the reference concentration, WQI for the water will be determined only by that substance, even though other concentration of pollutants may be far below targets for good water quality. Hence, WQI evaluates water quality not based on the mixture of pollutants, the potential uses of the water or the type of treatment required, but only towards a strict reference for each single substance. It thus can happen that water resources are evaluated with a low WQI and hence a low WIIX in a water-scarce area just because one substance does not hold the benchmark. This may clearly underestimate the value of water resources in water scarce areas, where quantity rather than quality is the major problem.

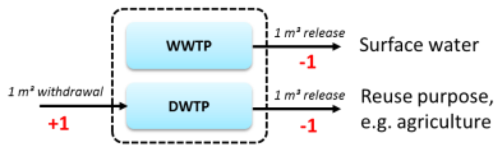
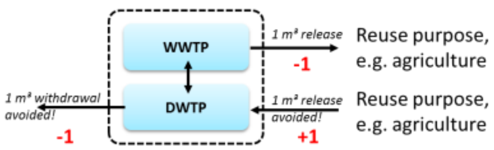
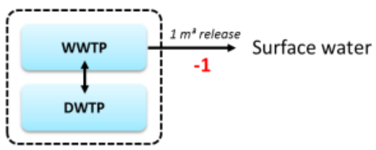
System expansion or avoided burden (credits) to account for reclaimed water

In LCA there are two principal approaches to address secondary functions of a system, such as the production of reclaimed water as a secondary product of wastewater treatment: the “system expansion” approach and the “avoided burden” approach. For a fair comparison between different options, secondary functions have to be somehow reflected in the LCA to enable functional equivalency between scenarios.

A first option to reach this functional equivalency is to expand the systems with alternative processes supplying the same function (“system expansion”). An example would be to expand the model of a reference WWTP without water reuse with another process for water production (e.g. a drinking water plant), so that this expanded system fulfils both functions of wastewater treatment and production of water for other uses (see Table 1-2 for system expansion). Now it can be directly compared to a WWTP with water reuse, as both systems supply equivalent functions. The corresponding water withdrawals and releases for calculating the WF are easy to determine (see Table 1-2 for system expansion). This procedure is recommended if an LCA study with WF assessment is conducted. It will be adopted for the LCA case studies of this report if possible within the scope of the study and depending on available data.

Table 1-2: Accounting for water withdrawal and release in water footprinting for different scenarios with system expansion and substitution calculation

WWTP: wastewater treatment plant, DWTP: drinking water treatment plant

<table>
<tr><th>Approach</th><th>Reuse scenario</th><th>No Reuse scenario</th></tr>
<tr><td>System expansion</td><td>
</td><td>
</td></tr>
<tr><td>Avoided burden</td><td>
</td><td>
</td></tr>
</table>

Another option follows the “avoided burden” approach: the impacts for supplying secondary products are directly subtracted from the bi-functional scenario, crediting the avoided burden of the process which would supply the secondary product in a reference system. This approach is more easy to use, as secondary products can be reflected by subtracting equivalent processes (e.g. drinking water production) from the system without modelling an expanded reference system. However, the avoided burden approach could be misleading when dealing with systems of water reuse: a WWTP with water reuse is credited with the avoided burden of a drinking water treatment plant (see Table 1-2 for avoided burden). Hence, water reuse is credited with the avoided withdrawal for drinking water production in the WF, but it should also be "credited" with the avoided release from the drinking water plant. The latter effect is somewhat counter-intuitive, as avoided production of drinking water should include both avoided withdrawal (leading to a decrease of WF), but also avoided release of freshwater (leading to an increase of WF). Finally, the "avoided burden" approach has to be carefully justified in WF calculation of water reuse systems, and all effects of avoided water production should be reflected properly in the WF inventory.

1.3.2 Risk assessment and application for wastewater reuse safety

Health and environmental risk assessment is a systematic approach to quantify the probability and severity of adverse effects to human health or the environment caused by certain hazards. A hazard is defined as a chemical, biological or physical agent with the potential to cause adverse effect on human health or the environment. The present report aims at quantifying human health risk caused by water reuse under different reuse scenarios.

Four of the DEMOWARE case studies (El Port de la Selva, Braunschweig, Olf Ford and Sabadell) have been investigated for health risks caused by pathogens via quanitative microbial risk assessment (QMRA). Microbial contamination is still the most common and widestread health risk associated with water and wastewater use. Additionally, at two case studies (El Port de la Selva, Sabadell) health risks caused by selected chemical substances have been assessed quantitatively (QCRA).

In the present study a hazard is considered to be "of concern" and thus poses a "risk" if existing health targets are likely to be not achieved, given the available information. This is also the case when the given information is so uncertain that a substantial amount of the probability distribution of the estimated average concentration overlaps with a predefined health target. In such cases the conclusion *"the system is able to achieve the required treatment performance"* cannot be derived with certainty. Then either the quality of information has to be improved by getting more reliable data or additional risk reduction measures have to be considered.

Health targets are either expressed in terms of:

1. Precautionary or toxicologically derived limit values for chemical substances
2. Disease burden expressed as *disability adjusted life years* (DALYs)
3. Required treatment performance water reuse system to achieve one of the above mentional health targets of the (log unit removal for pathogens, substance removal in % for chemcials substances)

Goal and scope

The goal of this risk assessment report is to demonstrate how quantitative risk assessment of microbial and chemical hazards may contribute to a more informed decision making and open communication in the fields of water reuse. It is not considered an exhaustive assessment of all potenial hazards and risks related to water reuse and reclamation. Since risk assessment and management approaches are always considered site specific generalization takes place on a methodological level.

Within the present report microbial risk is addressed much more comprehensively as chemical risk. This has two major reasons. First, pathogens are considered the most relevant health hazards related to water reuse. Second, only limited information was available on the concentraiton of substances present at the investigated WWTPs.

This report is consistent in that way that all chemical substances for which local data have been available or provided by the operator have been assessed quantitatively. From a risk-avers, precautionary perspective, which should be especially applied in cases where potable reuse is considered, substances for which information is not available have to be assessed with “relevant, due to missing information” until more information becomes available. Mixture effects of chemical substances are not considered.

Key elements of quantitative risk assessment

The terminology of the risk assessment literature is not always consistent. However, the principal steps are equivalent in almost all of the existing approaches. Following the terminology of a recent WHO publication [22] on quantitative microbial risk assessment, the following steps have to be conducted:

1. Problem formulation: Definition of the goal and scope of the assessment (Hazard selection, exposure scenarios, health outcomes)
2. Exposure assessment: Modelling of exposure of selected population groups to the selected hazards
3. Health effects assessment: Derivation of dose response relationships of the selected hazards
4. Risk characterization: Risk estimate based on combination of all previous steps (risk quantification, uncertainty assessment, sensitivity analysis)

Addressing uncertainties

In this report, often probability distributions of random variable are used instead of point estimates (mean, max, percentile), which allows for transparently addressing the uncertainty regarding the quantity of interest. Simulating a high number (e.g. 10000) of “samples” from these distibutions allows for considering all possible combinations of uncertain inputs and thus for a more transparent expression of scientifiy knowledge. This methodology is refered to as Monte Carlo simulation and can be considered good practice in QMRA and other risk assessment studies.

This higher transparency regarding uncertain inputs comes at the cost that the reader has to be somehow familiar with reading histograms where the possible values of a random variable are plotted on the x-axis, while the probability density (or sometimes frequency) as a measure of the relative probability of a certain sample is shown on the y-axis. The area under the curve always intergrates to 1, since it describes a probability. Thus, in the case that the range of possible values is very narrow the probability density itself can take values larger than 1.

Bayesian updating for probabilistic risk assessment

Scientific knowledge is always incomplete and often distributed over numerous scientific disciplines [23]. In risk assessment it is the challenge to summerize the present information about the problem under study in order to derive a probability statement of certain adverse effects occurring in the future.

Limited or missing data is a common problem is risk assessment and often literature information or expert knowledge is used to make reasonable assumptions instead [22].

Moreover, although site specific information in the form of data should be given preference, there might be valuable local expert knowledge which might be worth including into the assessment.

In such cases, in which different sources of information, i.e. other than locally measured data, are intended to be included in the assessment Bayesian updating represents and mathematical formalism with which this can be done transparently and reproducibly [23]. In a Bayesian framework this information is expressed in terms of a prior probability distribution which is subsequently updated with new information, which is e.g. included in the data.

In Bayesian data analysis parameters of distributions (like the mean μ, or the variance σ^2 of a normal distribution N (μ, σ^2)) are considered random variables and estimations about these parameters are expressed as a probability distribution (posterior) conditional on the observed data and prior informaiton.

This posterior distribution in turn is the product out of the probability distribution of the data generating process (likelihood) and the probability distribution of the prior information available regarding the plausible values of the parameter to be estimated (prior). An illustrative example is given in Figure 1-3.

In summary, advantages of using Bayesian methods for risk based approaches are:

1. Probability is used as a measure of uncertainty. Thus, Bayesian methods already use the approach and language of risk.
2. Bayesian methods demand the formulation of priors (prior knowledge) for every parameter and allow therefore the inclusion of different types of information (e.g. expert knowledge, literature information).
3. Bayesian approaches can easily be updated when new information becomes available. This characteristic captures the notion of continuous learning and improvement which risk management approaches are aiming at.
4. By allowing the use and mathematical combination of different sources of information in a transparent way instead of having to choose between different possible assumptions, Bayesian approaches make assumptions more credible and easier to communicate.
5. Bayesian approaches use a broader notion of probability, which expresses the current “state of knowledge”. Using this interpretation for risk assessment and management underlines already that those approaches should be seen as “living documents” which should be revised and checked for new knowledge about the assumptions made once they become available.

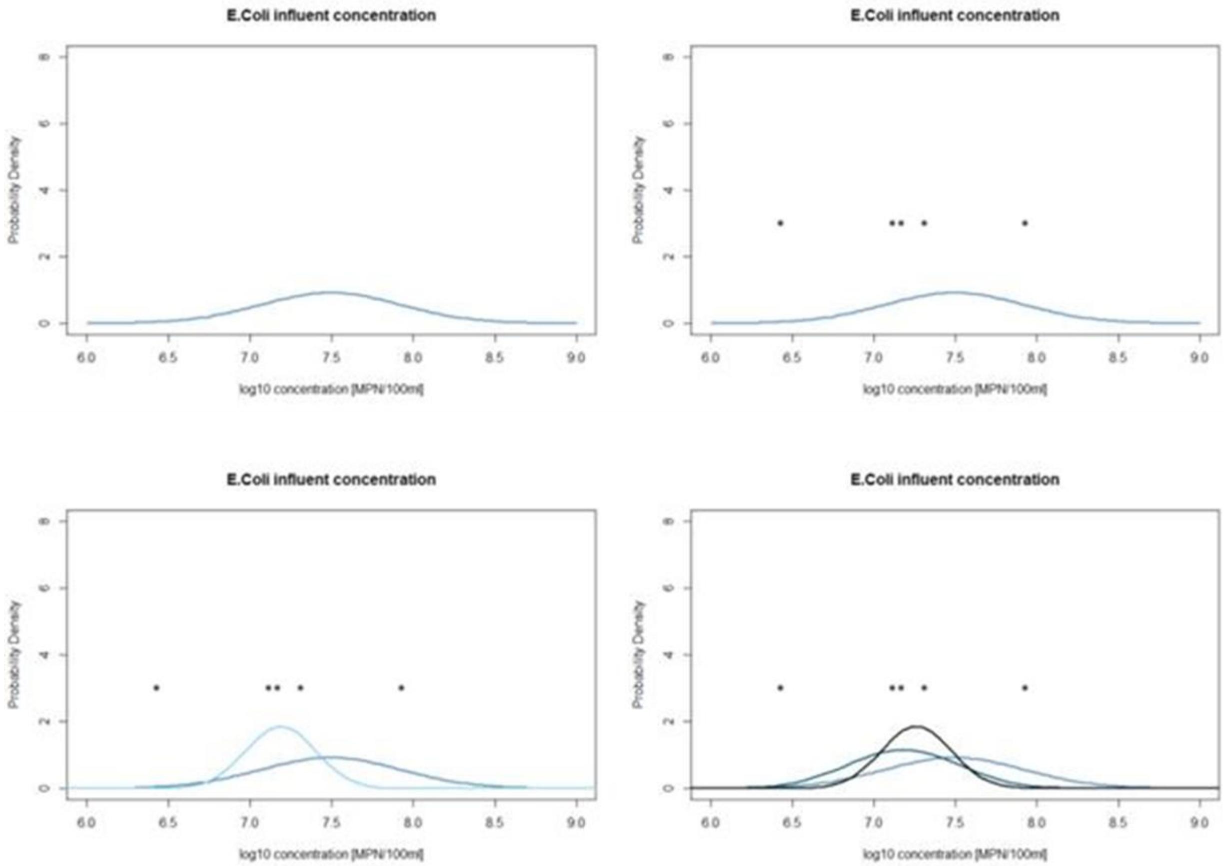


Figure 1-3: Illustration of the steps of Bayesian updating process

E.coli influent concentrations as an example: (1) Upper left: Setting up prior distribution based on literature information, (2) Upper right: arrival of data points (black dots), (3) Lower left: defining likelihood distribution for the data (light blue), (4) Lower right: combining information and calculate posterior distribution (black line shows marginal distribution of the mean)

Other risk assessment approaches

Risk assessment can be done qualitatively, semi-quantitatively or quantitatively with each of the approaches having certain advantages and disadvantages in certain situations. While in this report focus is put on quantitative approaches for risk assessment, there are other methods shall at least be mentioned for completeness.

Other commonly proposed methods include risk matrixes and sanitary inspections which are easy to implement and can lead to significant improvements regarding the overall system understanding and may help to identify and rank areas of further action. Sanitary inspections provide a checklist approach for checking the most relevant risk factors which might potentiall impair water quality [22]. Risk matrices, like presented in Table 1-3 combine the infromation about the severity of the consequences of a certain hazardous event with the probability/frequency of the event occurring. Note that probability is expressed as a number between 0 and 1, whereas frequencies are expressed as events happening in a given time span or within a predefined number of trials (e.g. 5 times per year, 3 failures out of 10 trials). In the semi-quantitative risk matrix like shown below both these quantities as well as the severity factor are transformed into scores between 1 and 6.

Given that risk is addressed in an at least semi- quantitative manner, the rankings of frequency and severity have to be combined in order to come to a final ranking of the present risks (e.g. in Table 1-3) . Most commonly this combination is done by multiplication, although there has been discussion about the sum being the better way of combining the two aspects [24].

Table 1-3: Example of a semi-quantitative risk assessment matrix
Green: low risk, yellow: moderate risk, bright red: high risk, dark red: very high risk

Severity	Frequency/Probability					
	1	2	3	4	5	6
1	1	2	3	4	5	6
2	2	4	6	8	10	12
3	3	6	9	12	15	18
4	4	8	12	16	20	24
5	5	10	15	20	25	30
6	6	12	18	24	30	36

Risk matrices as a semi-quantitative methodology are the way of risk assessment promoted by WHO within the risk management framework manuals of Water Safety Plans and Sanitation Plans. However, despite the fact that scoring systems from 1- 6 can easily be implemented and communicated, care must be taken to precisely define what is meant by a probability score of 5 in comparision to one of 4. If the difference is not clearly defined such ranking still might help to rank decision options internally, but might fail to compare result to other systems. Moreover, uncertainties regarding the assessment are not directly addressed.

Review of similar studies and projects

- Tozo et al. (2010) [25] used static quantitative microbial risk assessment to assess microbial health risk. They combined both in situ data on pathogen removal with literature information to estimate the residual risk of reused water after subsurface passage. The results highlighted the importance of site specific hydrogeological data to account heterogeneity when estimating travel time. The latter is a key variable when predicting pathogen removal by this kind of treatment.
- Mok et al. (2014) [26] investigated the norovirus disease burden from wastewater irrigations of vegetables in Australia. Results indicate that after conventional wastewater treatment and a waste stabilization pond treatment risk from viruses was not reduced below the threshold of 10^{-6} DALY pppy year set by World Health Organization (WHO). Additional disinfection was subsequently successful to guarantee the required performance.
- Beaudequin et al. (2016) [27] and Beaudequin et al. (2015) [28] introduced and utilized Bayesian networks as an conceptual approach for QMRA.
- Regarding chemical risk assessment, Paíga et al. (2016) [29] carried out an environmental risk assessment along the Lys river (Portugal) and in the influents and effluents of two wastewater treamtent plants (WWTP). The paper embraces 33 pharmaceuticals and metabolites and included a total of 91 samples in river and wastewaters. Regarding the risk assessment, three different trophic levels (algae, daphnids and fish) and 20 out of 33 compounds were taken into account. Results obtained in the WWTP effluents showed that seven substances had a risk quotient (RQ) higher than one. In this sense, Sulfamethoxazole, Clarithromycin, Azithromycin, Fluoxetine presented a risk to algae. Furthemore, Acetaminophen and Diclofenac also showed a

RQ higher than one regarding daphnids and fish respectively. In addition, Ibuprofen had also RQ >1 concerning two trophic levels: algae and fish. The risk assessment carried out in surface water showed a decrease in the RQs compared to the outputs from WWTP effluents. However, the majority of the compounds such as Sulfamethoxazole, Clarithromycin, Azithromycin and Ibuprofen remain having a RQ > 1. This performance indicates that low dilution effects and autodepurative or photodegration process may not be enough to avoid the presence and risks arising from these compounds.

- Houtman et al. (2014) [30] performed a human health risk assessment of the mixture of pharmaceuticals in Dutch drinking water. For this purpose 42 pharmaceuticals were monitored at three drinking production plants. The risk assessment was carried out using average concentrations of pharmaceuticals. Results in this work were presented for one of the drinking water plants (Rhine) and indicated that adverse health effects due to lifelong exposure could be considered negligibly for all the substances under study [30]. All risk quotients were below 0.01 (negligible risk).
- The paper by Santos et al. (2006) [31] evaluates influent and effluent samples from four wastewater treatment plants in Seville (Spain). To achieve this task an environmental risk assessment was performed. The study includes a total of six substances allocated to several drug groups such as anti-inflammatory drugs (Diclofenac, Ibuprofen, Ketoprofen and Naproxen), antiepileptic drugs (Carbamazepine) and nervous stimulants (Caffeine). The paper concludes that there are three drugs (Caffeine, Carbamazepine and Ketoprofen) for which no ecological risk is expected to occur neither in the influents nor in the effluents of the WWTPs. Ibuprofen showed a RQ > 1 at both sampling points and Naproxen's RQ exceeded the threshold value in the influent but not in the effluent of the WWTPs.
- A study published by Papadakis et al. (2015) [32] evaluated pesticides pollution caused by agricultural activities in the basin of Lake Vistonis (Greece). Therefore, 302 active substances and their transformation products were monitored in the surface waters of the basin. In addition, an environmental risk assessment was carried out based on the RQ method where three trophic levels were taken into account (fish, aquatic invertebrates, algae). Results showed that 7 insecticides, 3 herbicides, 1 nematicide, 1 fungicide and 1 acaricide were likely to present a risk to the environment (RQ >1).
- Palma et al. (2014) [33] studied the impact of 25 pesticides and some of their degradation products on the aquatic organisms belonging to the Alqueva reservoir (Guadiana basin, southern of Portugal). For this aim, a risk assessment was performed. Results included in this paper indicate that the majority of the pesticides exhibited either low (Cyanazine, DEA, Dimethoate, Isoproturon, Metolachlor, Molinate, Propanil and Simazine) or no risk (Atrazine, Linuron, Alachlor, Fenitrothion, Malathion, 2,4-D, Bentazone, MCPA and Mecocrop. However, three substances (Diazinon, CFP and Terbuthylazine) and two compounds (Chlortoluron and Diuron) were associated with high and medium risk respectively.

Further risk related publications include Salgot et al (2006) [34] who summarized risks related to water reuse, Westrell et al. (2004) [35] and Schönning et al. (2007) [36], who investigated microbial risks related to the handling of human feces and various activities related to wastewater and sludge treatment. The necessity for a risk based derivation of common quality criteria for water reuse is discussed in Paranychianakis et al. (2015) [37].

Acceptable risk and safety in water supply systems

The discussion about a tolerable or acceptable level of risk cannot be answered in an absolute way. The outcomes of a quantitative risk assessment might be “there is a probability of 20 % that the concentration of substance X will be above a certain threshold level”. Such an outcome clearly has to be set into a decision making context, which includes the risk, benefits, and other factors relevant for decision making associated with alternative options. It might be that even if the risk is high it is still the best decision for achieving a certain objective[1]. In contrast, even very low risks can be too high if there are alternatives with even smaller risk and the same benefit. For the specific case of water supply and reuse system the WHO formulates vaguely:

“The judgement of safety — or what is a tolerable burden of disease in particular circumstances — is a matter in which society as a whole has a role to play. The final judgement as to whether the benefit resulting from the adoption of any of the health-based targets justifies the cost is for each country to decide” ([38], p. 36).

It is important to understand that for the scientific evaluation and quantification of the probabilities of certain negative outcomes (e.g. harm, disease, loss), these outcomes have to be linked to other value judgements which should be part of a wider decision making processes managed by the responsible (health or environmental) authorities.

In the present report risk is regarded to be acceptable if the above mentioned health targets are met. The health targets for the individual demonstration sites are presented in the respective chapters of the report.

[1] Given that the target is not to be questioned

2 Case Study of El Port de la Selva

2.1 Introduction and Setting

El Port de la Selva is located near Cap de Creus in the northern part of Catalonia (Spain). In this coastal village the population during the summer months is ten times the 1’000 permanent residents present in winter. The village is not connected to the regional water distribution network and relies on local groundwater as its only water source. The municipal wastewater treatment plant (WWTP) was last refurbished in 1997 and has a capacity of 10’500 p.e. (secondary effluent).

Due to dry periods in the first decade of the 21st century, El Port de la Selva, like other settlements along the Costa Brava, added a tertiary treatment step consisting of double-filtration, UV disinfection and residual chlorination to the WWTP. The additional treatment step has a capacity of 25 m^3/h. A dual-pipe network for reclaimed water was constructed in order to make use of reclaimed water during the summer for urban purposes. From the WWTP, the water is pumped up to an elevated storage tank, from which it can be distributed by gravity-driven flow to the city’s reclaimed water network. A concession for urban use of reclaimed water has however not yet been obtained.

Average annual drinking water abstraction in El Port de la Selva is about 350’000 m^3, while the average volume of wastewater treated annually is in the range of 220’000 m^3/a. Both abstraction and wastewater volumes have large fluctuations between summer and winter (Figure 2-1). Chloride concentrations in drinking water abstracted from the municipal well were above the drinking water limit of 250 mg/L during the autumn months (Figure 2-1). Water reclamation by aquifer recharge was intended to counteract these seasonal salinity peaks and improve groundwater availability. A site approximately 700 m upstream of the local drinking water wells, along the Riera de Rubies (or Riera de Romanyac) was chosen as the recharge site. It was planned to infiltrate 200 m^3/d of tertiary treated within 200-240 d/a, resulting in 40,000 – 48,000 m^3/a (about 10% of the abstracted groundwater).

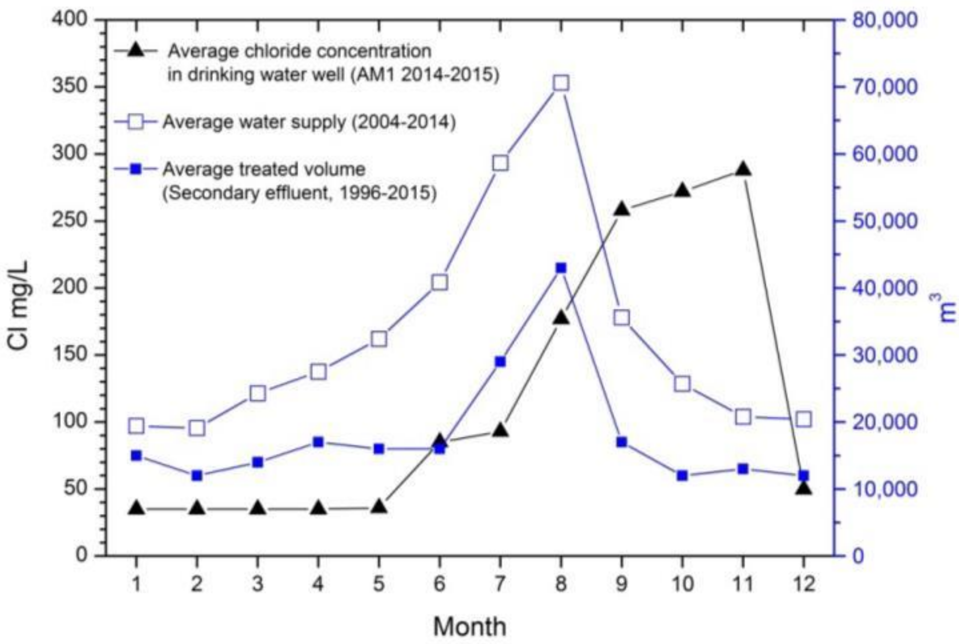


Figure 2-1: Seasonal fluctuations of treated and abstracted water volumes and measured chloride concentration in drinking water well

2.2 Life Cycle Assessment

2.2.1 Goal and scope definition

The goal of this LCA is to analyse and compare different options to increase the water supply in El Port de la Selva by 96 000 m³ per year, which is approximately 30 % of the total supply of drinking water. This LCA can serve as example for sites with seasonal water scarcity situated at the coast and no other sources of water nearby or easily available, quantifying the environmental profile of different alternatives. The target group of this study consist primarily of the local stakeholders such as the water utility (CCB), the treatment plant operators (EMACBSA), the local government and the citizens of El Port de la Selva, but also planers and engineers in the field of wastewater treatment and water supply.

Function/ Functional Unit

The function of the system under study is providing wastewater treatment and drinking water supply for the city of El Port de la Selva, including all processes that are related to this function. Consequently, the functional unit of this LCA is defined by providing this service annually for each inhabitant, measured via person equivalents (pe) regarding the wastewater treatment process (i.e. provision of water services “per pe and year” or $(pe*a)^{-1}$). Although the population of El Port de la Selva accounts for approximately 1 000 residential inhabitants, this population number is not used for the functional unit in this LCA, as intensive touristic activities increase the load to the WWTP to more than 10 000 pe in the peak summer months, which would not be reflected when accounting only the resident population. Referring to the annual COD-load of the WWTP in El Port de la Selva (121 t COD/a) and an expected daily load of 120 g COD/(pe*d) [39], the annual amount of raw wastewater corresponds to an equivalent of 2 700 pe, which is taken as the basis for calculating the functional unit. For reasons of simplification the function of drinking water supply is associated with the same functional unit, i.e. accounting the annual drinking water production (327 000 m³/a) on the equivalent of 2 700 pe.

Alternatively, a second functional unit “per m³ additional water supplied” is defined to illustrate the *additional* environmental impacts that are specifically related to the provision of additional water *compared to the traditional way of water supply*, i.e. on top of the efforts for existing water supply from groundwater. Related impacts are calculated by accounting the changes between the reference scenario (= status quo with additional supply from groundwater) and the scenarios with alternative routes of additional water supply (= water reclamation, import, or seawater desalination) on the total amount of additional water produced (96 000 m³/a).

System boundaries

Due to required changes in the mainstream WWTP process in case of the reuse scenarios, it was decided to include the existing WWTP process (secondary treatment including sludge handling and disposal) as well as the tertiary treatment into the system boundaries. Drinking water treatment and pumping of groundwater is also included, since certain alternatives (pipeline, seawater reverse osmosis) will partly reduce the quantity of supplied drinking water from groundwater. Release of reused water (=reclaimed water) in the environment via irrigation and infiltration in recharge ponds is accounted. Pumping and distribution of different water streams (drinking and reclaimed water) is also taken into account. Finally the background process for production of electricity, chemicals, fuels, materials, the infrastructure and maintenance are considered (Figure 2-2).

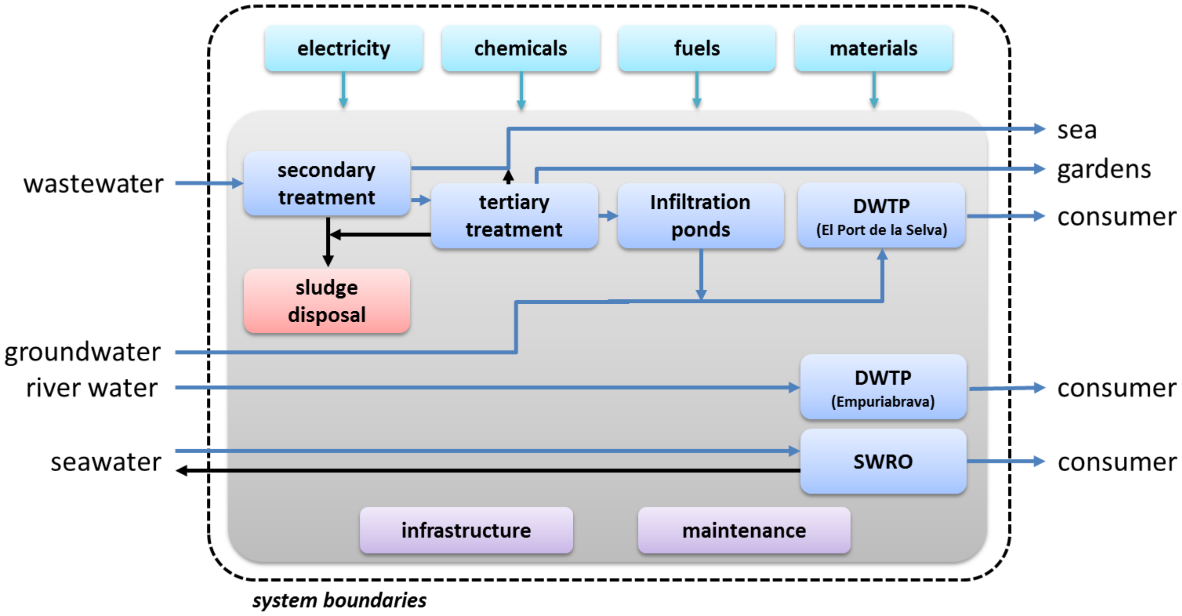


Figure 2-2: System boundaries and scope for LCA study El Port de la Selva
DWTP: drinking water treatment plant; SWRO: seawater reverse osmosis

Allocation

All environmental efforts (e.g. energy consumption in water treatment) and benefits (e.g. substitution of mineral fertilizer by nutrients in sludge compost) are related to the function of wastewater treatment and drinking water supply and its functional unit. Consequently, no allocations have been conducted.

Scenarios

The scenarios are selected to compare different approaches which increase the local water supply in El Port de la Selva. A comparative overview of all scenarios including scenario description and annual water volumes (Figure 2-3) is provided below:

0. Status represents the existing situation until 2015, which consists of regular treatment of wastewater in the WWTP (including sludge disposal) and discharge of secondary effluent into the sea. Drinking water is produced from local groundwater, and this water supply is assumed to be increased by 96 000 m³/a compared to mean volumes of 2012-2015 (231 000 m³/a), providing a total volume of 327 000 m³ per year of drinking water.
1. Reuse A is the partly implemented reuse scheme within DEMOWARE in El Port de la Selva since autumn 2015. Wastewater is treated in secondary treatment operating with enhanced nutrient removal during the winter months (October to May). This includes operational changes to a continuous nitrification-denitrification system compared to the previous batch operation, and the addition of iron(III)-chloride in the activated sludge tank for phosphorus removal via simultaneous precipitation. While a part of the secondary effluent is still directly discharged into the sea, approximately half of the annual volume of secondary effluent is treated in tertiary treatment (pressurized filtration, UV disinfection and chlorination) during the summer months (June to September). This tertiary treated effluent is pumped to the reclaimed water storage tank and then reused for public and private irrigation. Although the related equipment is in place, this mode of operation has not been applied yet due to existing concerns of the local health authority. Within the winter months, approximately 40 % of the annual secondary treatment is treated in tertiary treatment including pressurized filtration, filtration with granular activated carbon (GAC) for removal of trace organics (GAC filter designed for 50 % removal of gabapentin,

to be installed in short-term) and UV disinfection. This tertiary effluent is pumped to the reclaimed water storage tank and then conveyed to three infiltration ponds located upstream of the village for artificial groundwater recharge. Drinking water is still provided by groundwater, but due to direct reuse in summer and artificial groundwater recharge in winter the stress on the aquifer will be reduced, and less native groundwater will be required for water supply.

2. Reuse B is a similar reuse scheme to '1 Reuse A', but using membrane technology in tertiary treatment. In summer operation, the tertiary treatment includes ultrafiltration (UF) and chlorination to provide sufficient quality for irrigation, while in winter ultrafiltration and reverse osmosis (RO) are foreseen to enhance the removal of trace organics and salinity. In this scenario, the upgrade of the secondary treatment with dosage of iron(III)chloride is not required as residual P is removed in tertiary treatment with the RO membrane. The quantity of reclaimed water for irrigation or groundwater recharge is similar to '1 Reuse A'.
3. Connection to the water network of Costa Brava North, which receives water from the surface water treatment plant in Empuriabrava (Muga River) is another option to reduce the stress on the local aquifer in El Port de la Selva. Approximately 30 % of the annual drinking water demand of El Port de la Selva would be supplied via an external drinking water treatment plant, transporting the water via new pipelines (2.65 km) to the village. The scenario includes the materials for construction of an additional pipeline, which is needed to connect El Port de la Selva to Llanca and the water network of Costa Brava North, and the electricity for pumping of water to El Port de la Selva. Wastewater treatment is similar to '0 Status', and 70 % of the drinking water demand of El Port de la Selva is still provided by local groundwater.
4. SWRO (seawater reverse osmosis) is another possibility to mitigate the high pressure on local water resources during summer. A seawater desalination plant (including pre-treatment via ultrafiltration and then reverse osmosis) would provide approximately 30 % of the annual drinking water demand. Wastewater treatment is similar to '0 Status' and 70 % of the drinking water demand of El Port de la Selva is still provided by local groundwater.

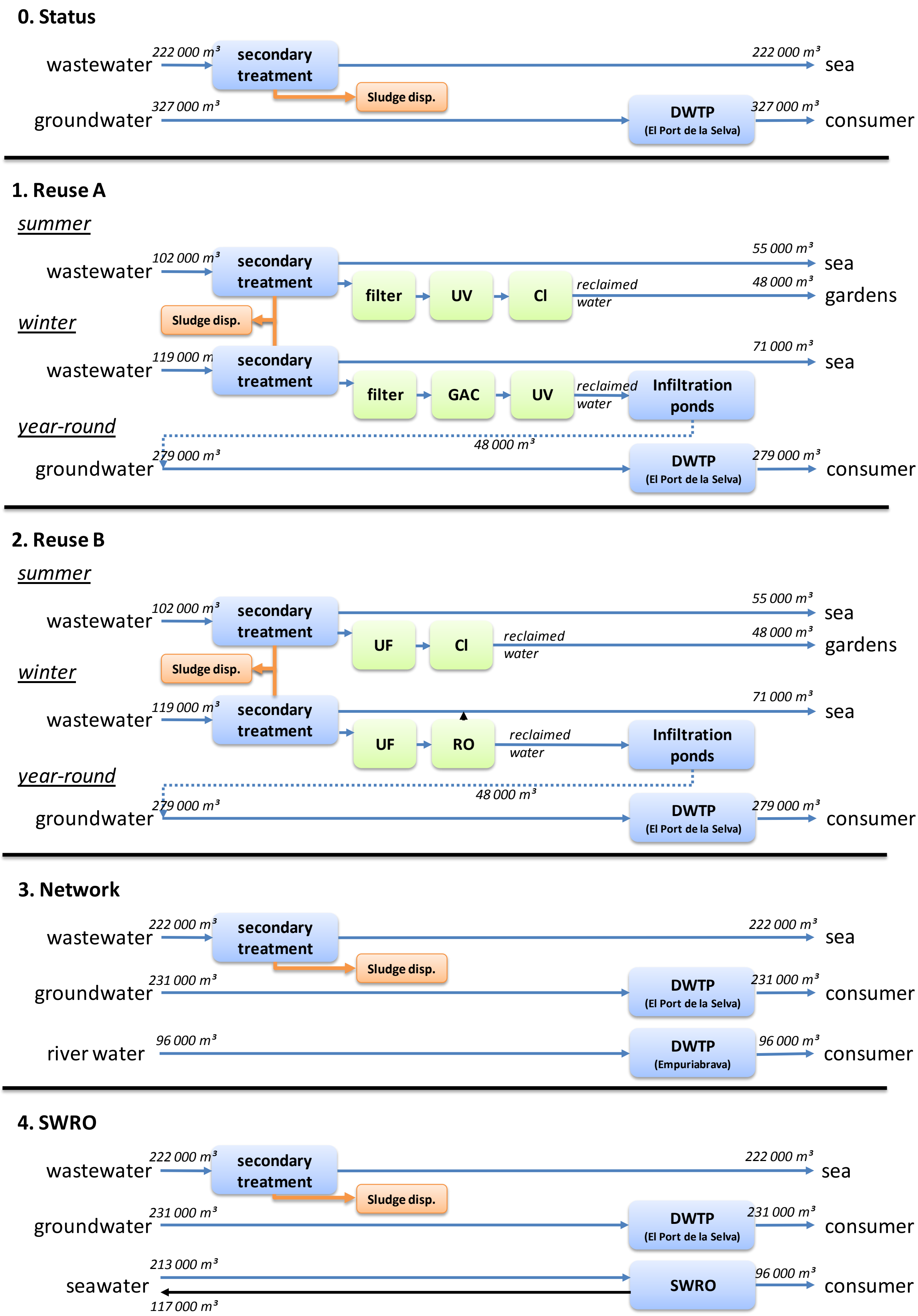


Figure 2-3: Overview of the LCA scenarios for El Port de la Selva and annual water volumes
DWTP: drinking water treatment plant; SWRO: seawater reverse osmosis, UF: ultrafiltration; RO: reverse osmosis; GAC: granular activated carbon

Data quality and limitations of this study

Parameters for the LCA inventory are discussed regarding data quality and uncertainties to clearly point out inherent limitations of this study. An overview of data source and quality is provided in Table 2-1.

- Water quality: CCB communicates wastewater quantities and a lot of relevant water quality data very transparent on their website [40]. The dataset includes water quantities and concentration of the main parameters (solids, COD, TN and TP) on monthly base for the past 10 years. With this extensive dataset, the differences in wastewater quantities and loads of main parameters in summer and winter could be exactly defined, and different wastewater compositions for summer and winter operation were derived for the LCA. However, the information on heavy metal concentration in raw wastewater, secondary and tertiary effluent is very limited, and additional monitoring data within DEMOWARE suffered from low concentration of metals, mostly below the limit of quantification. Finally, best estimates based on available data had to be taken for the LCA. As a consequence, the validity and score in the impact assessment regarding toxicity (eco- and human toxicity) has to be critically reflected.

  In terms of drinking water quality, potential direct effects on human health were not considered in this LCA, as drinking water does not represent an emission into the environment. However, the quality of intake water for drinking water production is mandatory to calculate a quality index for water footprinting. The available information from the water source of DWTP Empuriabrava (scenario Network) is very limited in terms of water quality.
- Water quantities and operation mode: The estimated water volumes (Figure 2-3) can be assumed as plausible and validated due to long-term data available from CCB. However, the actual water quantity used for irrigation in public/private gardens is only based on best estimates of the operators. Unfortunately no direct information on actual water demand for irrigation is available, which is a limitation of this study. In '2 Reuse B' the production of 48 000 m^3 reclaimed water via the UF/RO system is assumed. However in certain winter month with dry weather conditions, this may not be possible due to limited water availability from the WWTP effluent. A second problem could be the mode of operation for the RO under variable flow conditions (e.g. longer shutdown time in summer).
- Energy, chemicals and material consumption: The dataset on energy and chemical consumption for secondary and sludge treatment is based on detailed information of CCB, as they provided detailed information on monthly energy and polymer consumption over a 10 year timeframe. Variations in specific energy or polymer consumption could be attributed to the changing volume of wastewater treated. This provision of data enabled a detailed energy analysis of the WWTP in El Port de la Selva, including the existing tertiary treatment.

  The data for sludge disposal (via composting) and associated emissions have been estimated based on other studies. Nonetheless, the sludge quantity does not change significantly between the scenarios, so this part of the system is rather negligible for the comparison.

  The information on energy for water pumping and drinking water treatment, as well as the chemical consumption for drinking water treatment has been internally validated by the operators. The data quality on consumptives for tertiary treatment in '1 Reuse A' is also very high, since the filters, UV disinfection and chlorination have been operated over 10 years. However, the specific energy consumption for these aggregates was directly measured on-site only for a couple of days, so changes over time could not be reflected. The GAC filter has not been implemented so far, so data is based on estimates of KWB from the experience of previous studies treating secondary effluent with GAC [41]. Overall, existing uncertainties are assumed to be rather small, so data quality for tertiary treatment (Reuse A) is seen as high.

Table 2-1: overview on data quality of input data

Parameter/Process	Data source	Data quality
Reference system		
Water quality (standard parameters)	[40, 42], local operators	very good
Water quality (heavy metals)	DEMOWARE monitoring	low (grab samples)
Energy consumption in secondary and sludge treatment	[40, 42], local operators	very good
On-site chemical consumption	[42], local operators	very good
Sludge composting	[43, 44]	low
Infrastructure WWTP	[45]	medium
Drinking water treatment plants	[42], local operators	good
Scenarios for additional water supply		
Water quality	[42], local operators	medium
Energy & chemical consumption for Reuse A and Network	[42, 46], local operators	good
Energy & chemical consumption for Reuse B and SWRO	[47, 48], estimations	low-medium
Infrastructure for treatment processes	[49]	medium
Infrastructure for pipelines	[42, 46], local operators	very good
Background		
Electricity mix	Mix of Spain 2010	medium
Chemicals and materials	EU or global datasets	medium
Transport	Truck transport (EU)	good

Significantly higher uncertainties regarding consumptives exist for the membrane processes, both in tertiary treatment in '2 Reuse B' and regarding the seawater RO. Data on energy and chemical of the wastewater membranes had been adopted from the Torreele (BE) case study (cf. Chapter 7), which represents an optimized membrane reuse scheme with long-term experience in operation. To account for existing uncertainties regarding the applicability of the same scheme in El Port de la Selva and to reflect the potentially higher specific energy consumption for small-scale systems (El Port de la Selva = 2 700 pe) a higher electricity consumption (+ 50 %) for the wastewater membranes has been assumed in contrast to Torreele (80 000 pe). The energy and chemical consumption as well as the recovery rate of the membrane scheme depends heavily on influent water quality, scale, and implementation and operation strategy. Pilot trials would be required to enhance the precision of the dataset, so the current assumptions represent a major shortcoming of this study.

Similar approaches have been followed for the seawater desalination system '4 SWRO', where process data is based on literature [47] and feasibility studies for a SWRO system for the Vendee (FR) case study (cf. DEMOWARE deliverable 6.5). In contrast to the wastewater membranes no safety factor regarding the system-scale had been taken into account, meaning the energy demand for '4 SWRO' can be interpreted as optimistic.

The demand of materials for major infrastructure has been calculated based on registers of materials for existing systems (e.g. tertiary treatment, water supply network) that have been recently built, providing a good data quality. For the remaining infrastructure of the WWTP treatment plant, material demand has been estimated based on previous studies [45, 49] and adapted to the local boundary conditions, resulting in higher uncertainty for the infrastructure data.

Normalization

Normalisation reveals the contribution of the system under study towards the total environmental footprint of each citizen. Principles for normalization and normalization factors are shown in Annex 9.1.1.

2.2.2 Inventory (Input data)

Primary data

Inventory data for the LCA study were provided by local operators like CCB and complemented with estimates of KWB based on previous studies (Table 2-1). For consumptives, Table 2-2 summarizes the electricity demand and Table 2-3 summarizes chemical demand for all scenarios. The materials for infrastructure are shown in detail in Annex 9.2.1.

Table 2-2: Inventory data for energy demand as electricity (summarized in categories) in LCA El Port de la Selva
Different volumes per categories and scenario (Figure 2-3), primary data mainly provided by CCB [42, 46-48]

	Unit	0. Status	1. Reuse A	2. Reuse B	3. Network	4. SWRO
Wastewater treatment and pumping, total	**kWh/a**	**114 109**	**257 046**	**271 403**	**114 109**	**114 109**
Secondary treatment (including sludge disposal)	kWh/ m^3 wastewater	0.65	0.67	0.67	0.65	0.65
Tertiary treatment (summer)	kWh/ m^3 reclaimed water	-	0.41	0.18	-	-
Tertiary treatment (winter)	kWh/ m^3 reclaimed water	-	0.44	1.10	-	-
Pumping tertiary treatment	kWh/ m^3 reclaimed water	-	0.69	0.69	-	-
Drinking water treatment and pumping, total	**kWh/a**	**168 357**	**143 707**	**143 707**	**311 600**	**548 552**
Treatment of groundwater (El Port de la Selva)	kWh/ m^3 groundwater	0.04	0.04	0.04	0.04	0.04
Treatment of surface water (Empuriabrava)	kWh/m^3 external water	-	-	-	0.11	-
Pumping via water network	kWh/m^3 external water	-	-	-	1.42	-
Treatment via seawater reverse osmosis	kWh/m^3 external water	-	-	-	-	4.00
Pumping of drinking water in El Port de la Selva	kWh/m^3 drinking water	0.47	0.47	0.47	0.47	0.47
Overall electricity demand, total	**kWh/a**	**312 467**	**400 753**	**415 110**	**455 709**	**692 662**

For the scenario 'Reuse A' 13 ppm Fe^{3+} is dosed in secondary treatment for enhanced phosphorus removal. The backwash volume of the rapid filters is approximately 3.5 % related to the influent volume of the filters. For the GAC-filter a volume of 2.8 m^3 (respectively 970 kg GAC) is needed (EBCT = 20 min). To reach at least 50 % removal of Gabapentin a bed-volume of 7 000 is mandatory, meaning GAC has to be regenerated 2-3 times per year. Including regular regeneration a total Lifetime of 30 years had been assumed for GAC. The UV dose had been estimated to 800-820 J/m² based on a transmission of 71 %. The chlorine dose for disinfection is estimated to 8 ppm Cl based on the consumption of sodium hypochlorite.

In scenario 'Reuse B' an UF (Recovery = 85 %) and in winter a RO (Recovery = 75 %) is implemented. The energy consumption is estimated with 0.18 kWh/m³ filtrate for UF and 0.86 kWh/m³ filtrate for RO. The UF concentrate is treated in WWTP, whereby the RO brine is directly discharged into the sea. The consumption regarding chemicals of membranes is shown in Table 2-3.

Table 2-3: Inventory data for materials demand in LCA El Port de la Selva

Related on different volumes and aggregates, all concentrations per feed volume and chemicals in concentrations with water [41, 42, 47, 48]; GW = groundwater in El Port de la Selva; ST = secondary treatment; Cl = wastewater Chlorination; UF = wastewater UF; RO = wastewater RO; RW = river water in Empuriabrava; SW = seawater reverse osmosis per SWUF feed volume; Co = sludge compost, avoided burden for fertiliser production including substitution factors 20 % for Nitrogen and 80 % for Phosphorus [50]

Chemical	Unit	0. Status	1. Reuse A	2. Reuse B	3. Network	4. SWRO
Polyacrylamide	g/kg DS	14.5	14.5	14.5	14.5	14.5
$FeCl_3$ (40 %)	mg/L	-	94.0 (ST)	-	-	7.2 (SW)
Fresh GAC	kg/a	-	32	-	-	-
Regenerated GAC	kg/a	-	2 455	-	-	-
NaOCl (15 %)	mg/L	28.0 (GW)	28.0 (GW) 113 (Cl)	28.0 (GW) 113 (Cl) 40.3 (UF)	28.0 (GW)	28.0 (GW) 28.2 (SW)
NaOH (50 %)	mg/L	-	-	4.5 (RO)	-	1.80 (SW)
Citric Acid (40 %)	mg/L	-	-	0.53 (UF) 0.90 (RO)	-	0.80 (SW)
Antiscalant	mg/L	-	-	2.46 (RO)	-	- [2]
H_2SO_4 (32 %)	mg/L	-	-	52.0 (RO)	-	61.0 (SW)
$NaHSO_3$ (39 %)	mg/L	-	-	1.48 (RO)	-	-
NH_4Cl (50 %)	mg/L	-	-	4.00 (RO)	-	-
Chlorine gas	mg/L	-	-	-	2.02 (RW)	-
PACl (10 %)	mg/L	-	-	-	1.07 (RW)	-
Lime (92 %)	mg/L	-	-	-	-	123 (SW)
HCl (32 %)	mg/L	-	-	-	-	9.50 (SW)
N fertiliser	kg N/a	- 894 (Co)	- 894 (Co)	- 894 (Co)	- 894 (Co)	- 894 (Co)
P fertiliser	kg P_2O_5/a	- 4 418 (Co)	- 4 418 (Co)	- 4 418 (Co)	- 4 418 (Co)	- 4 418 (Co)

The treatment train in the surface water treatment plant in Empuriabrava ('Network') includes pre-chlorination, coagulation (with PACl), flocculation, decantation, filtration and disinfection with chlorine gas. The water is pumped via an existing water network to Llançà and then to El Port de la Selva with a hypothetical new network connection, thereby passing a height difference of 250 m.

[2] Antiscalant for SWRO assumed to be citric acid

The scenario 'SWRO' includes a pre-treatment with pre-chlorination, coagulation (with $FeCl_3$), flocculation and ultrafiltration. After reverse osmosis sulfuric acid and lime are dosed for remineralisation of the desalinated water. The detailed chemical consumption is shown in Table 2-3.

Water inventory

Table 2-4 and Table 2-5 show the water volumes and qualities for the WWTP influent and the related effluents for summer operation (May-September) and winter operation (October – April). Differences between influent and effluent loads are removed by WWTP or tertiary treatment and partially integrated into sludge. Small differences between effluent concentrations result from model calculations based on effects of backwash from tertiary treatment on the WWTP performance.

Background data

Information on background processes are shown in the Annex 9.2.1.

Inventory for Water Impact Index

The WIIX is calculated according to the methodology described in *D3.1* [11].

The accounted volumes of water withdrawal and release from/to the freshwater environment are summarized in Table 2-6 for all scenarios according to Figure 2-3. Withdrawals from groundwater (or river water in DWTP Empuriabrava) are fully accounted, while withdrawals and releases from/to the sea or technosphere (e.g. WWTP influent, drinking water consumed by inhabitants without irrigation water) are not accounted in water footprinting

Table 2-4: Water inventory for summer operation (MAY-SEP)

Measured data by [40]; * estimates or calculated by LCA model; membrane efficiencies in Reuse B according to [48]

Parameter	Unit	WWTP influent	Scenario 0/3/4 Effluent to the sea	1. Reuse A Effluent to the sea	1. Reuse A Reclaimed water	2. Reuse B Effluent to the sea	2. Reuse B Reclaimed water
Volume	m^3/a	102 550	102 429	*54 429 **	*48 000 **	*54 429 **	*48 000 **
SS	mg/L	262.1	5.3	5.3	2.3	*10.1 **	*0.3 **
COD	mg/L	639.9	57.5	56.7	56.7	*53.5 **	*53.5 **
TN	mg/L	77.5	23.9	23.7	23.7	*22.6 **	*22.6 **
TP	mg/L	17.9	4.3	4.3	4.2	*4.1 **	*4.1 **
Cd	µg/L	*0.4 **	0.1	*0.1 **	0.1	*0.1 **	*0.1 **
Cr	µg/L	*8.0 **	2.0	*2.0 **	2.0	*1.8 **	*2.2 **
Cu	µg/L	*45.2 **	11.3	*11.1 **	11.5	*10.5 **	*12.3 **
Hg	µg/L	*0.2 **	0.05	*0.05 **	0.05	*0.05 **	*0.05 **
Ni	µg/L	*27.6 **	6.9	*6.8 **	70	*6.4 **	*7.5 **
Pb	µg/L	*2.0 **	0.5	*0.5 **	0.5	*0.5 **	*0.6 **
Zn	µg/L	*294.0 **	73.6	*72.4 **	75.0	*68.0 **	*80.0 **

Table 2-5: Water inventory for winter operation (OCT-APR)
Measured data by [40]; * estimates or calculated by LCA model; membrane efficiencies in Reuse B according to [48]

Parameter	Unit	WWTP influent	0./3./4. Effluent to the sea	1. Reuse A Effluent to the sea	1. Reuse A Reclaimed water	2. Reuse B Effluent to the sea	2. Reuse B Reclaimed water	2. Reuse B Brine to the sea
Volume	m^3/a	119 396	119 300	*71 301 **	*48 000 **	*71 301 **	*48 000 **	*16 000 **
SS	mg/L	181.1	3.7	3.6	2.1	*8.0 **	-	*0.8 **
COD	mg/L	466.5	56.2	55.4	55.4	*51.8 **	*1.0 **	*204.3 **
TN	mg/L	53.1	9.0	7.7	7.7	*8.3 **	*2.0 **	*27.3 **
TP	mg/L	9.8	2.8	1.2	1.2	*2.6 **	*0.1 **	*10.0 **
Cd	µg/L	*0.4 **	0.1	*0.1 **	0.1	*0.1 **	-	*0.5 **
Cr	µg/L	*8.0 **	2.0	*2.0 **	2.1	*1.8 **	-	*8.6 **
Cu	µg/L	*45.2 **	11.3	*11.2 **	11.6	*10.3 **	-	*48.6 **
Hg	µg/L	*0.2 **	0.04	*0.04 **	0.04	*0.04 **	-	*0.2 **
Ni	µg/L	*27.6 **	6.9	*6.8 **	7.1	*6.3 **	-	*29.7 **
Pb	µg/L	*2.0 **	0.5	*0.5 **	0.5	*0.5 **	-	*2.2 **
Zn	µg/L	*294.0 **	73.6	*72.5 **	75.1	*67.2 **	-	*316.2 **

For release into groundwater, water supplied for public and private irrigation (48 000 m^3/a) is only partially accounted as release. The effective fraction of irrigation water which is reaching the groundwater table (after subtracting evaporation and plant uptake) is assumed with 25 % [51], so an 'effective' release to the environment (= aquifer) of 12 000 m^3/a via irrigation is accounted in the WIIX for each scenario during the summer period. For groundwater recharge in the reuse scenarios in winter, only 5% water loss via evaporation is estimated due to fast infiltration in ponds, so the 'effective' release to the environment for these scenarios is additionally 45 600 m^3/year during the winter period.

The monthly water scarcity index (WSI) according to WULCA AWARE [52] is used for calculation of WIIX (monthly WSI are listed in Annex 9.1.1 Table 9-2). For both sites with freshwater withdrawal for drinking water production (El Port de la Selva and Empuriabrava), the same water scarcity index is reported in the AWARE method. However, actual water stress is reported only in El Port de la Selva in form of over-exploitation of the local aquifer, while no water stress is reported for the area of Empuriabrava which is supplied by a local river coming from the Pyrenees. This discrepancy between projected water scarcity and actual water scarcity is further discussed in sensitivity analysis of the WIIX reseults. Uptake of seawater and release into ocean are not accounted in the WIIX, as saltwater is seen as an infinite water resource. It has to be underlined here that due to the monthly WSI taken for WIIX calculations, water release into the environment in summer (i.e. irrigation) has higher credits than water release in winter (i.e. groundwater recharge via ponds) due to monthly variations in water scarcity. Seasonal water volumes are distributed to monthly values following mean consumption patterns of 2012-2015.

The water quality index (WQI) is calculated based on the intake or effluent water quality parameters [40, 48] which are listed in Table 2-4 and Table 2-5. For WQI calculation, phosphorus determines the final WQI of all releases (see detailed WQI inventory in Annex 9.2.1 Table 9-7), as P is the limiting parameter according to the existing surface water benchmarks in the EU water framework directive. Due to higher P concentration in reclaimed water for irrigation (Reuse A and B), this infiltration water has a significantly lower WQI than irrigation water supplied from groundwater. Since P emissions are already assessed in the LCA impact category ‘freshwater eutrophication potential’, this may be seen as ‘double-counting’ to again consider the impact of P in the WIIX, an aspect which is further discussed in 2.2.4. In terms of intake water quality for the two DWTP, no primary data was available for quality of groundwater or river water, so an optimum water quality (WQI = 1) is assumed for both withdrawal flows (Table 2-6). It has to be noted that high chloride concentration is detected in groundwater for drinking water production in late summer (cf. Figure 2-1). However, there is no environmental benchmark for Cl concentration in groundwater in the relevant directives, so issues of potential high Cl concentration are not reflected in the WIIX calculation regarding WQI of groundwater withdrawal. Assumptions and calculation regarding the indirect water impact index are discussed in Annex 9.1.1.

Table 2-6: Overview on direct withdrawals and releases and water quality indices (WQI) for the different scenarios
GW = groundwater in El Port de la Selva; RW = river water in Empuriabrava; IR = irrigation water; IN = infiltration water

Scenario	0. Status	1. Reuse A	2. Reuse B	3. Network	4. SWRO
Withdrawals [m³/a]	327 000 (GW)	279 000 (GW)	279 000 (GW)	231 000 (GW) 96 000 (RW)	231 000 (GW)
WQI (Withdrawals)	1[3]	1[3]	1[3]	1[3] (GW) 1 (RW)	1[3]
Releases [m³/a]	12 000 (IR)	12 000 (IR) 45 600 (IN)	12 000 (IR) 45 600 (IN)	12 000 (IR)	12 000 (IR)
WQI (Releases)	1	0.05 (IR) 0.17 (IN)	0.06 (IR) 1.00 (IN)	1	1

2.2.3 Impact Assessment (Results)

Environmental impacts were assessed with a set of 8 impact categories (including WIIX), representing different areas of environmental concern. After an overview of all indicators, selected impact categories are discussed more in detail to reveal individual contributions of different processes and aggregates to the total environmental impact

Total environmental impacts and benefits of all scenarios

The environmental profile of all scenarios for all selected impact categories is shown relatively to the gross impact of the existing system ‘0 Status’ (= 100 %) in Figure 2-4 related to the functional unit “per pe*a”. The fossil and nuclear cumulative energy demand (CED), the global warming potential (GWP) and terrestrial acidification potential (TAP) are strongly influenced by the background processes, such as electricity, chemicals or material production. Valorisation of sludge compost gives small credits for nutrient content in CED, GWP and TAP due to substitution of mineral fertilizer production. The main drivers regarding CED for ‘0 Status’ are wastewater treatment and drinking water pumping to the storage tank, whereas the drinking water treatment process itself has only a small contribution. Direct onsite emissions of the WWTP process (i.e. N_2O and NH_3 from the activated sludge tank) or during sludge disposal (composting) are minor contributors regarding the GWP. However, direct ammonia emissions

[3] WQI = 1 assumed for groundwater intake, reflecting optimum water quality. Cl is not included in as environmental benchmark, hence potential high Cl level in groundwater is not represented in the WIIX.

within these process steps reveal a high contribution to the overall TAP. The toxicity indicators are mainly influenced by the valorisation of sludge compost and the associated application of heavy metals into soil.

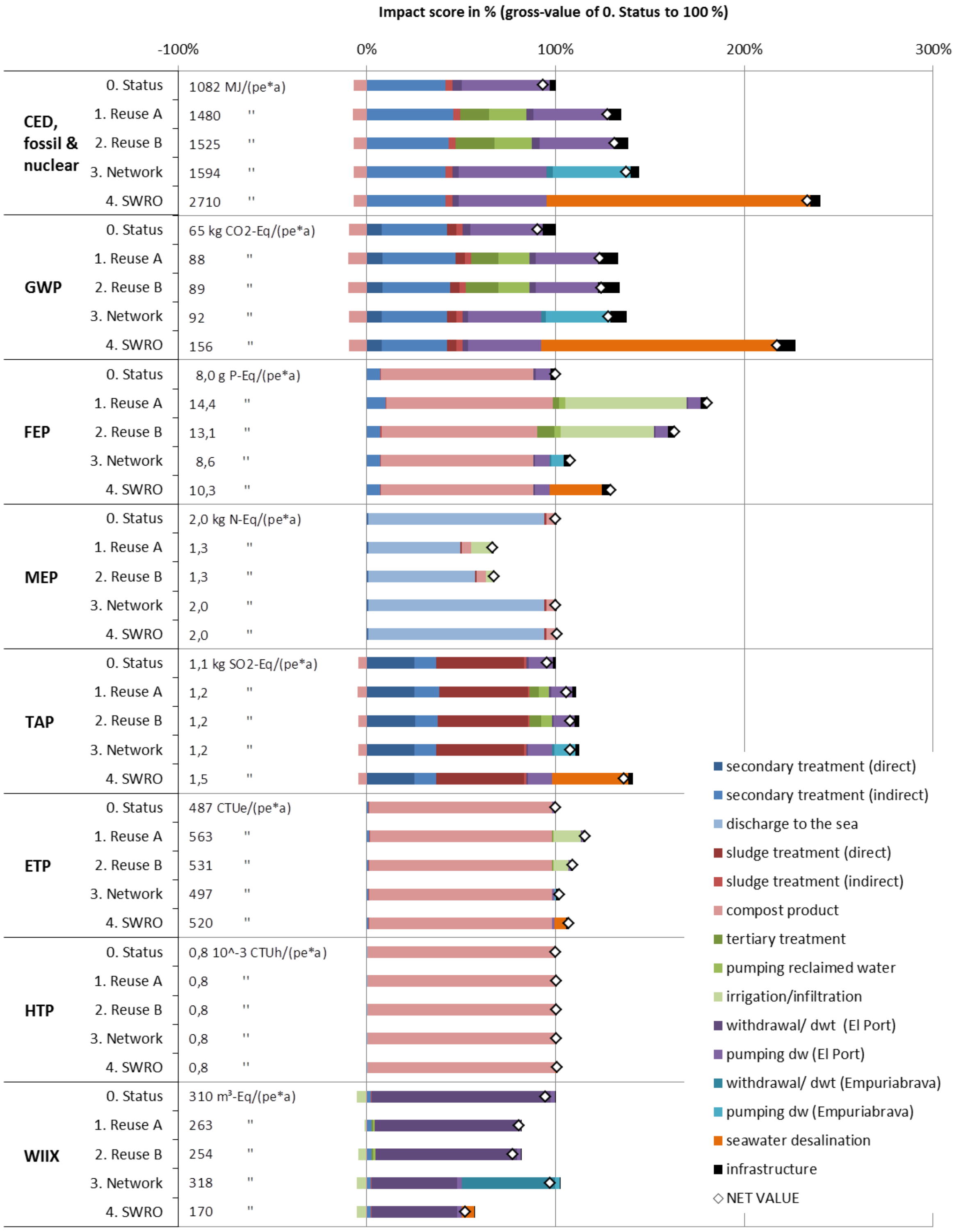


Figure 2-4: Environmental profile for all scenarios related to gross-value of '0 Status' (= 100 %) and total net values per scenario and impact category

CED = cumulative energy demand; GWP = global warming potential; FEP = freshwater eutrophication potential; MEP = marine eutrophication potential; TAP = terrestrial acidification potential, ETP = eco toxicity potential; HTP = human toxicity potential; WIIX = water impact index

Marine eutrophication potential (MEP) is dominated by the volume of water released to the sea and the related nitrogen removal in secondary treatment. The effects of sludge recycling and water reuse on the MEP are minor, as reclaimed water is released to the freshwater environment. However, sludge recycling and water reuse are of higher importance for the freshwater eutrophication potential (FEP) because P in WWTP effluent discharged into the sea is not accounted for FEP, but is effective if released to groundwater in scenarios 'Reuse A and B'. The WIIX is mainly influenced by the volume of freshwater withdrawal, and also by the different water quality of water releases and energy intensive background processes. Water reuse and seawater desalination can both reduce the overall WIIX of the system considerably.

In summary, all alternatives to the existing water supply system increase the environmental impact in most indicators (i.e. CED, GWP, FEP, TAP, ETP), as additional energy and resources are required to produce and transport this water compared to the existing supply from local groundwater. However, the investigated alternatives can supply additional water *without* increasing local water scarcity, as is indicated by a lower WIIX for most alternatives except for water import. In addition, water reuse also decreases significantly MEP, as nitrogen emissions with WWTP into the ocean discharge are reduced. Overall, the existing water supply from groundwater represents the option with the lowest environmental impact due to low energy and chemicals demand and would be preferable if local water stress would not be increased with extended exploitation of the aquifer. As this traditional way of water supply should now be complemented by alternative options due to negative effects of over-exploiting the local groundwater, a more detailed analysis of benefits and drawbacks of the specific alternative scenarios is useful.

Relative changes/ Effects per m³ additional water supply for selected impact categories

To analyse the *additional* impacts that these alternative water sources provoke compared to the existing water supply, the second functional unit is used. This unit relates the total differences between the baseline score (0 Status) and the respective scenario for an alternative water source to the total amount of additional water produced (96 000 m^3/a). Hence, this perspective reveals the additional impact that new water sources require compared to the the existing water supply via local groundwater.

The CED for the existing drinking water supply via groundwater in El Port de la Selva is 4.9 MJ per m^3, mainly for drinking water distribution an intermediate storage tank uphill. Groundwater treatment has only a very low CED (< 0.5 MJ/m^3), as it is only pumped from the aquifer and chlorinated. The changes for the CED per m^3 additional water are shown in Figure 2-5 for each alternative water source. The excess energy demand varies between 11.2 MJ per m^3 additional water for '1 Reuse A' and 45.8 MJ per m^3 additional water for '4 SWRO'.

In the reuse scenarios, the main drivers of higher energy consumption are the tertiary treatment of WWTP effluent and the pumping of reclaimed water to the reclaimed water storage tank for subsequent distribution to irrigation network or infiltration ponds. Comparing the two different technologies for tertiary treatment, the existing train with filtration, GAC and UV ('Reuse A') is comparable in energy demand to the membrane schemes ('Reuse B'). For both reuse scenarios, credits are accounted during summer operation, as reclaimed water for irrigation substitutes regular drinking water and electricity required for its treatment and distribution. The additional energy demand of enhanced secondary treatment in the WWTP for better nutrient removal and the infrastructural efforts for tertiary treatment, reclaimed water network and infiltration ponds are minor compared to the tertiary treatment and distribution.

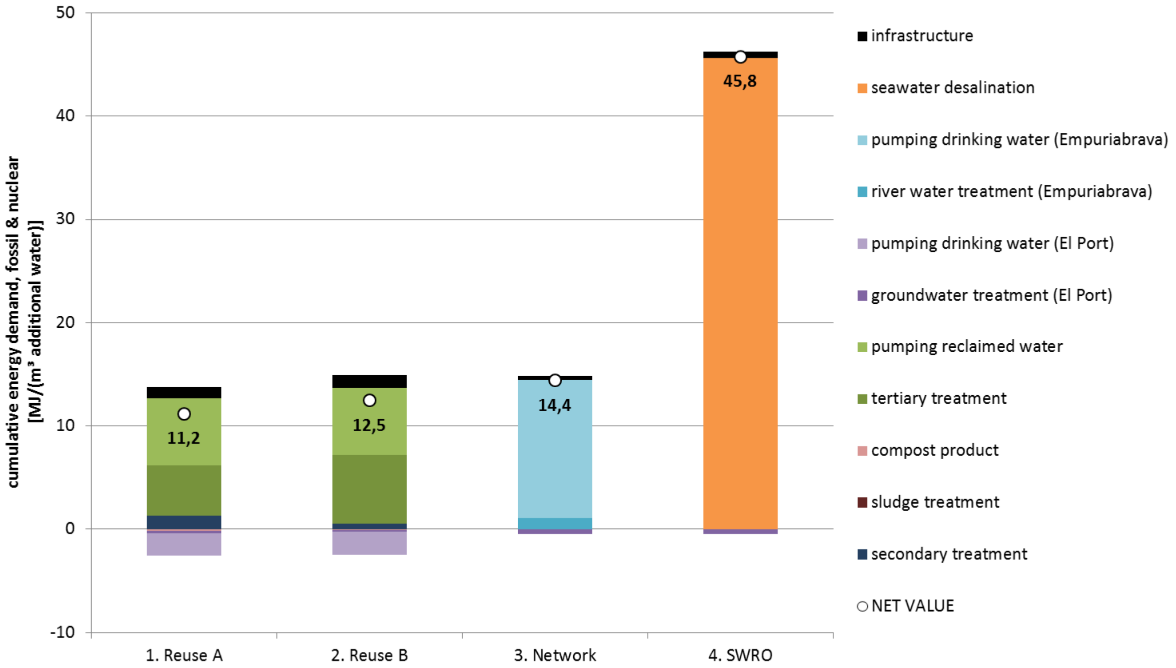


Figure 2-5: Changes in fossil and nuclear cumulative energy demand of the different scenarios compared to '0 Status' per m³ additional water

The higher CED for water import in scenario '3 Network' is mainly determined by the energy required for pumping the water from Empuriabrava to El Port de la Selva, requiring a pumping height of 250 m. The higher demand of chemicals and electricity in the surface water treatment plant of Empuriabrava compared to the simple groundwater treatment in El Port de la Selva has only a minor impact. The additional net energy effort for water import is 14.4 MJ per m³ water and thus slightly higher than for the reuse scheme with membranes ('2 Reuse B'). Seawater desalination has by far the highest CED of all alternative options for water supply, with > 80 % of the additional efforts due to the intensive electricity consumption of seawater reverse osmosis and < 20 % for chemicals, waste treatment and infrastructure.

For a more detailed analysis of the two reuse schemes, a direct comparison of the two trains for tertiary treatment is provided in Figure 2-6 showing the contribution of different aggregates to the overall net CED. Net CED for the tertiary treatment is 4.9 and 6.9 MJ per m³ produced water for '1 Reuse A' and '2 Reuse B', respectively. For scheme A, the operation of pressure filters in the existing tertiary treatment is by far the main driver in energy consumption, producing > 75% of the CED. Comparing schemes A and B, pressure filters consume even higher electricity than ultrafiltration in scheme B, which is somewhat surprising. This indicates a high site-specific electricity consumption of the pressure filters in the existing scheme, representing primary data measured in 2015, and reveals potential for optimisation. Overall, total CED of the two reuse schemes is quite comparable in the annual perspective, keeping in mind that the energy-intensive RO unit is only operated in winter for groundwater recharge (= treating 50% of annual reclaimed water). Minor energy demand is caused by chemicals for membranes, for chlorination, the material for the GAC and electricity for UV disinfection. Finally, the hybrid membrane scheme is almost competitive in total CED to the reuse scheme with filtration/GAC/UV, increasing additional energy demand by less than 35% while producing a higher quality of reclaimed water for indirect potable reuse (cf. Table 2-5).

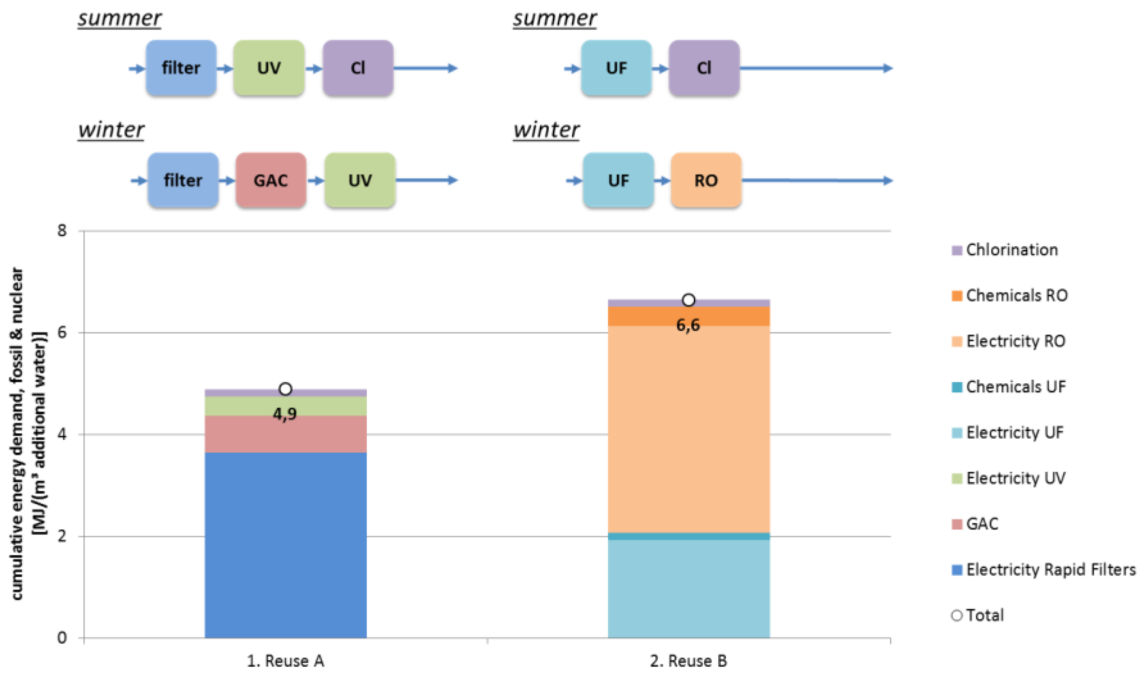


Figure 2-6: Comparison of different tertiary treatment schemes and contribution of different aggregates per m³ produced water (annual mean)

The results for GWP correlate closely to CED, since the share of fossil energy resources in the Spanish power mix is quite significant and electricity production has a high GWP. The net GWP of '0 Status' for drinking water supply in El Port de la Selva is 0.25 kg CO_2-eq/(pe*a). Figure 2-7 shows the excess GWP associated with additional water supply per m³ water in relation to the existing drinking water supply. The additional net GWP ranges from 0.66-0.68 kg CO_2-eq/m³ for the reuse schemes up to 2.56 kg CO_2-eq/m³ for seawater desalination, while water import has a net GWP of 0.76 kg CO_2-eq/m³. In general, individual contributions to the GWP for each scenario are comparable to the analysis of CED above. For the direct comparison of the two reuse schemes, the GWP difference is even smaller than in CED due to the relatively high carbon footprint of the activated carbon applied in scheme A, minimizing the differences between the scenarios.

The changes in FEP are shown in Figure 2-8. As the WWTP releases the effluent directly into the Mediterranean Sea, phosphorus in the WWTP effluent is not accounted in terms of FEP in the existing situation. Consequently, other routes of phosphorus emissions into the environment, e.g. via sludge compost to arable land, have a relatively high contribution on the FEP in '0 Status' (cf. Figure 2-4). In scenarios of water reuse, P emissions with WWTP effluent are reduced, but reclaimed water is directed to irrigation or groundwater recharge, thus increasing P emissions into freshwater environment. In direct comparison of the reuse schemes, '1 Reuse A' has a higher FEP due to respectively lower phosphorus removal in secondary and tertiary treatment compared to '2 Reuse B', applying RO during winter treatment with very low P levels in reclaimed water. In addition, removed P will end up in the sludge in the reuse schemes, causing some FEP in agricultural valorisation (Figure 2-8). Overall, the effect of indirect phosphorus emissions (e.g. via energy consumption for SWRO) has a relatively high contribution in this impact category, indicating that the overall FEP by all scenarios is low (see also results of normalization below). The additional FEP ranges from 17 mg P-Eq/m³ additional water for '3 Network' to 181 mg P-Eq/m³ additional water for '1 Reuse A', again indicating the diversion of effluent P from ocean discharge to groundwater or compost in the reuse schemes.

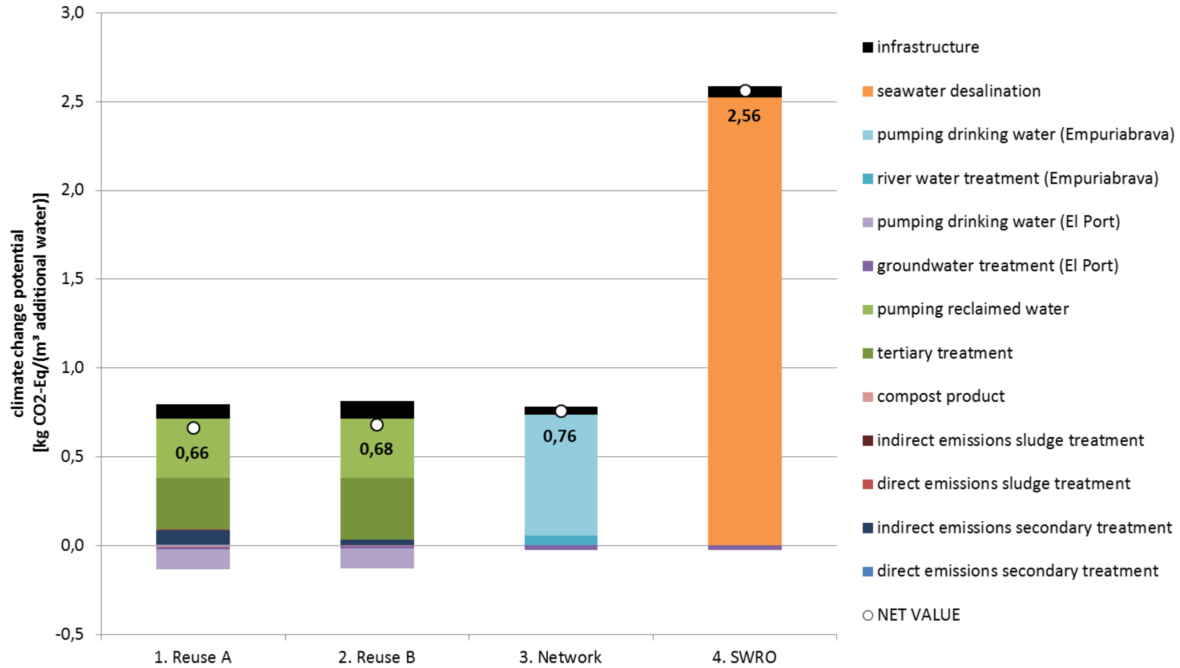


Figure 2-7: Changes in global warming potential of the different scenarios compared to '0 Status' per m³ additional water

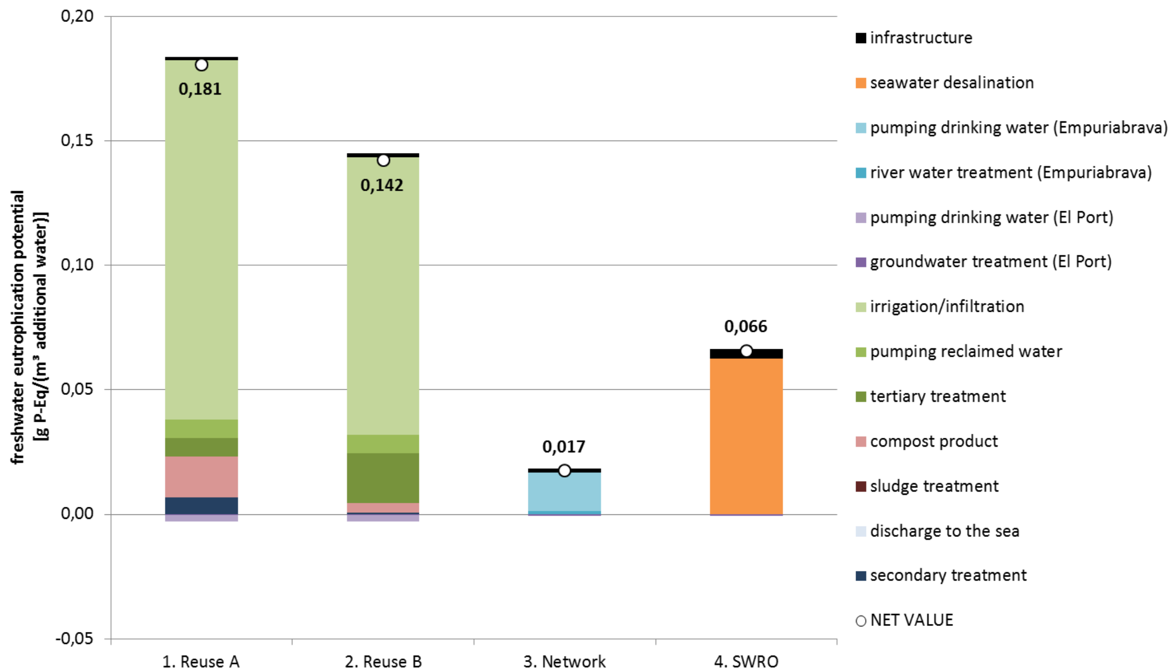


Figure 2-8: Changes in freshwater eutrophication potential of different scenarios compared to '0 Status' per m³ additional water

The MEP is shown in Figure 2-9, reflecting the impact of emissions of nitrogen species into the environment. Compared to '0 Status', the overall changes in MEP for the scenarios 3 and 4 are minor and only due to indirect processes, as the major route of emissions (i.e. direct N emissions with WWTP effluent) is not changed. For the reuse scenarios, MEP is significantly reduced, as (i) the effect of enhanced N removal in the WWTP is accounted and (ii) the water which is reclaimed for reuse is not discharged into the sea and hence cannot cause direct marine eutrophication. The latter effect is lower for the scheme B, because the nitrogen removed via RO will still end up in the ocean due to direct discharge of RO brine into the sea. Nonetheless, the reclaimed water of scheme B also contains less nitrogen than the water produced in scheme A, which reduces N emissions during irrigation and infiltration of reclaimed water. Overall, both reuse schemes have a comparable and high reduction of net MEP compared to the existing system.

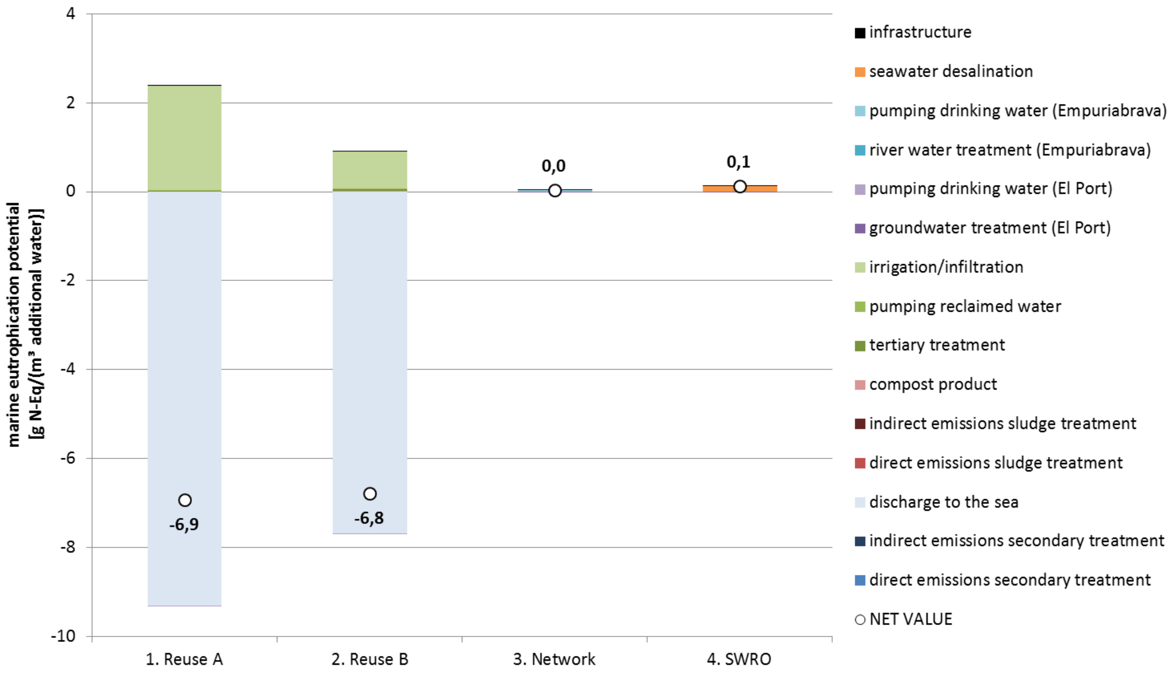


Figure 2-9: Changes in marine eutrophication potential of different scenarios compared to ‘0 Status’ per m³ additional water

For water footprinting, the WIIX shows the effects of additional water supply on the local and global water scarcity for all scenarios in relation to status quo (Figure 2-10). Water reuse and seawater desalination both reduce the net WIIX, taking into account local water stress, water quality and volumes. In contrast, water import from Empuriabrava has the same WIIX than the traditional water supply in El Port de la Selva in this study, as the related water stress index is comparable in both areas. Here, the low spatial resolution of the WSI method (watershed level) leads to the same WSI for both routes of water supply, even though in reality a significantly higher water stress is observed in El Port de la Selva compared to Empuriabrava. In fact, negative impacts of groundwater over-exploitation are already showing up in El Port de la Selva such as seawater intrusion, which was the starting motivation for the search of an alternative water source in the first place. In addition to the same WSI, both water sources are also assumed to have the same water quality index, i.e. optimal quality or WQI = 1 (cf. Table 2-6). With comparable water stress and quality factors, water import from Empuriabrava does not alleviate the water stress in the region in this calculation, having bascially the same WIIX compared to the existing situation.

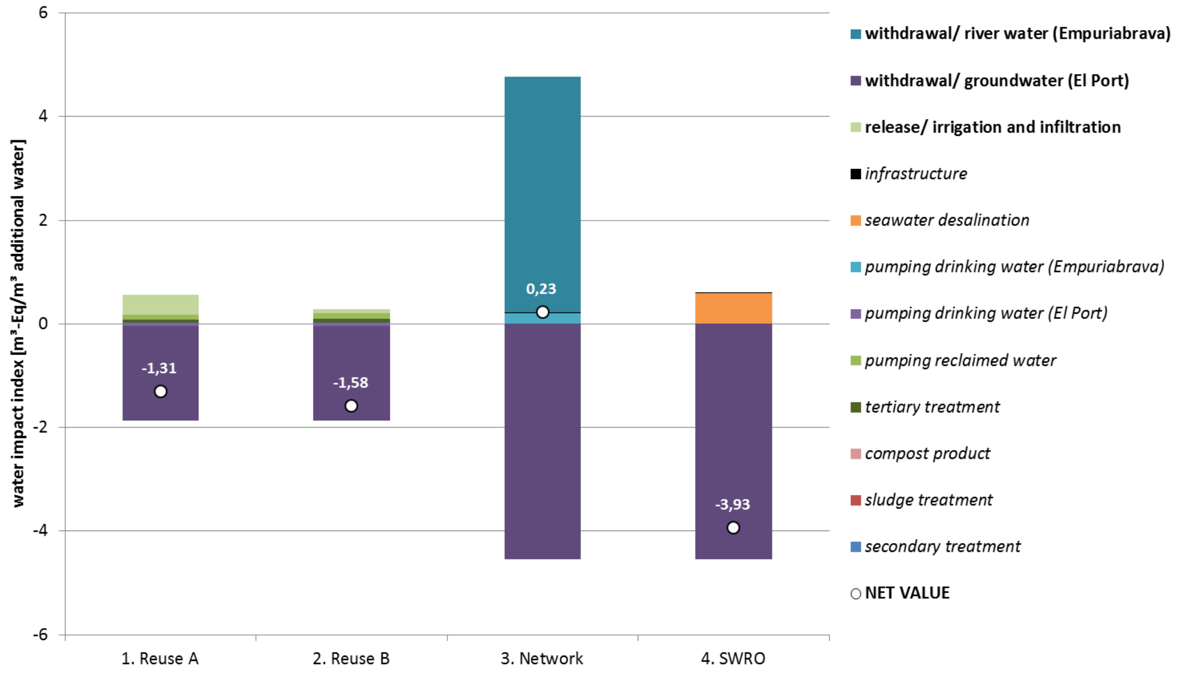


Figure 2-10: Changes in the water impact index of different scenarios compared to '0 Status' per m³ additional water

In terms of water release to the environment, the reuse schemes release higher water volumes to the watershed than the other scenarios due to local groundwater recharge, but all reclaimed water has a lower quality than water import or SWRO (Table 2-6). In addition, half of the reclaimed water is released in winter for groundwater recharge, where a low water stress index is further reducing the credits for this water release. Overall, reclaimed water volume used for irrigation and groundwater recharge is high, but has smaller environmental benefits than drinking water due to the lower water quality and the time of release (50% in winter). Hence, water release credits in WIIX are smaller for the reuse schemes, resulting in an additional WIIX for "release/irrigation and infiltration" (Figure 2-10).

However, the withdrawal of natural freshwater for additional water supply is completely avoided with water reuse, taking instead water from the technosphere (i.e. WWTP effluent). Hence, water reuse scenarios receive a large credit for avoided water withdrawal from local groundwater, resulting in a decrease in net WIIX compared to the existing system. The same effect can be observed for seawater desalination, where additional water is also supplied without any negative effect on local groundwater sources, resulting in a decrease in WIIX. Finally, both water reuse and SWRO can substantially reduce local water stress ("negative WIIX") due to water supply originating from alternative water sources and not from local groundwater.

The indirect effects in the WIIX (e.g. the water footprint of energy consumption) are relatively small for all scenarios, indicating that direct water handling is far more important for water footprinting than indirect water use. A detailed sensitivity analysis for the water footprint of El Port de la Selva is provided in chapter 2.2.4., showing the effects of different methodological choices on the results of the WIIX.

Normalization

Normalised scores for each impact category are shown in Figure 2-11 in relation to the functional unit "per pe and year" (Figure 2-4). Normalisation data per EU-27 citizen is summarized in Table 9-1.

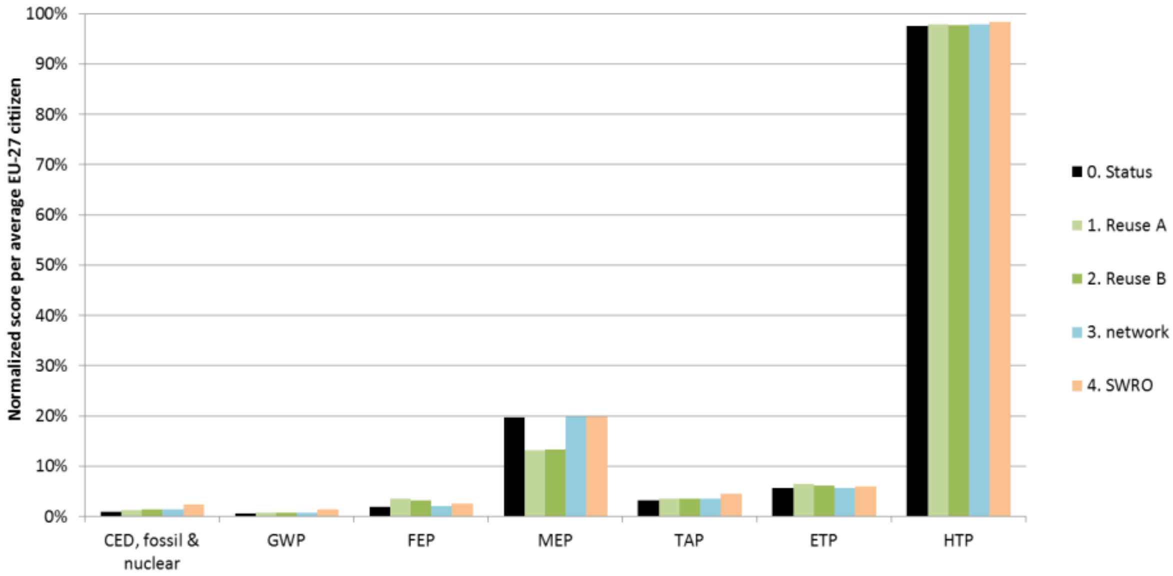


Figure 2-11: Normalized scores for all impact categories per average EU-27 citizen

CED and GWP contribute approximately 1 % to the gross CED or GWP per citizen in the EU-27, showing the low contribution of water and wastewater treatment to the total energy-related impacts of society. The normalized score of FEP is less than 4 % related to the EU-27 average, as the WWTP effluent is directly discharge into the Mediterranean Sea in this case where P emissions are not accounted. In contrast, MEP contributes up to 20 % of the total potential per EU-27 citizen due to the direct discharge of WWTP effluent and corresponding N loads into the sea. Normalised scores for TAP are less than 5% compared to the total TAP per EU-27 citizen, although direct emissions of ammonia to air were accounted in wastewater treatment and sludge composting. While normalised scores for ecotoxicity are < 10%, human toxicity potential of the systems represent >90% of the total HTP in EU-27. The major impact in HTP originates from heavy metals in sludge compost and their transfer to agricultural soil. However, it should be underlined here the available information on metals is limited. The strength and weakness of assessing toxicity in LCA are discussed more in detail in chapter 8.4 for all case studies, since similar results in normalization were achieved for all systems with agricultural valorisation of sewage sludge. At this point, it has to be underlined that the current fate factors and normalization data for the USEtox toxicity indicators have high uncertainties and have to be interpreted with care.

2.2.4 Interpretation and Discussion

Sensitivity analysis on water impact index

Summarizing the results on the WIIX indicator for El Port de la Selva, the following aspects have been elaborated:

1. The overall WIIX score is strongly influenced by freshwater withdrawal; credits for water release in the reuse schemes are less pronounced due to lower water quality (high P concentrations) and resulting lower water quality index.
2. The low spatial resolution of the water stress index grid does not allow a local differentiation of water stress for the two water sources of local groundwater in El Port de la Selva and surface water of the Muga River in Empuriabrava, which does not reflect the local reality of water management.

3. Although higher volumes of water are released to the freshwater environment in reuse scenarios (especially due to groundwater recharge in winter), this fact is not well represented in the overall WIIX score.

To elaborate on these uncertainties connected to the methodology of WIIX, a number of sensitivity analyses are conducted to show the effects of different assumptions and scopes on the results of the water footprint assessment:

1. The dependency of results of the WIIX from freshwater withdrawals results from (i) the high quantity of freshwater withdrawals in El Port de la Selva and (ii) the assumed optimal WQI. The second assumption was made because no accurate dataset was available for the real water quality of source waters in both drinking water treatment plants (El Port de la Selva and Empuriabrava). Using different datasets with different quality parameters leads to a definitive shortcoming of the methodology of WIIX, since the effective WQI might be lower using a larger dataset of water quality parameters. In terms of water releases to the non-marine environment, the use of WQI has to be critically reflected. In the total perspective of LCA with eutrophication and toxicity indicators, nutrient and heavy metal emissions are already accounted in the respective impact categories. If these parameters also determine the WQI, this 'double-counting' would distort the midpoint perspective applied in the impact assessment of LCA. Furthermore, it is also questionable that nutrients can deteriorate the final WQI even if the water is reused for plant irrigation, where nutrients could be seen as valuable ingredients rather than water pollutants.
   Given the high uncertainties in WQI calculation, the WQI is excluded in the first sensitivity analysis of water footprinting. Instead, a **water availability footprint (WAF)** is calculated using only information on the volumetric withdrawal/release and the local water stress (WSI), neglecting water quality information.
2. The existing low resolution in the available WSI grid for water stress calculation results in the same direct WIIX for both local groundwater use (0 Status) and water import from Empuriabrava (3 Network). This does not adequately reflect the situation described by local operators. The experience shows that even in dry years, there has been no threat for domestic supply with water in the region which is receiving water from the drinking water treatment plant of Empuriabrava. For El Port de la Selva, a certain degree of saltwater intrusion into the aquifer has been detected in dry years, since the aquifer level near the groundwater wells is lower than the sea level. [42]
   Reflecting the actual lower water stress at Empuriabrava, the **WSI for Empuriabrava is hypothetically reduced by 50% compared to the WSI of El Port de la Selva** to demonstrate the effect of different water scarcities on the WIIX result.
3. The respective low benefit of groundwater recharge via water reuse in the WIIX is due to the seasonal difference in WSI between times of infiltration in winter (low WSI) and projected times of "use" of this recharged water in summer (high WSI). WIIX accounts for the winter recharge with a low benefit, but still projects a high impact for the summer use and consequently a high water footprint. However, it may effectively be the same water which is infiltrated in winter and used in summer, so that local natural groundwater resources are finally less exploited. Consequently, a reduced WSI should also be accounted for the withdrawal of artificially recharged water in summer, as artifical groundwater recharge should increase groundwater

availability during summer. Since the calculation of a refined WSI is out of the scope of this study, an **expansion of system boundaries** is considered instead in sensitivity analysis. The aquifer could be interpreted as a 'technical system' within the system boundaries which "stores" water from winter into summer times (assuming no losses occur during storage). Consequently, both recharged groundwater and subsequent withdrawal of this water in summer are not considered anymore in the WIIX calculation, with the aquifer acting like a huge storage tank.

The benchmark WIIX and the results of the sensitivity analysis are visualized in Figure 2-12. The following conclusions can be drawn from the sensitivity analysis:

1. The WAF calculation results in higher benefits for the water releases of the reuse schemes, since the low WQI of reclaimed water is no longer taken into account. The fact that higher quantities of water are released to the freshwater environment in the reuse schemes becomes apparent in the WAF, showing clearly the effect of groundwater recharge (Figure 2-12). The differentiation between the reuse schemes 'Reuse A' and 'Reuse B' is now obsolete in the direct WAF, since the different water qualities are not accounted; nonetheless, water quality effects in terms of nutrients or heavy metals are still visualised in the eutrophication and toxicity indicators of LCA.
2. When reducing WSI of water imported from Empuriabrava, the WIIX for that water source is reduced by the same ratio (Figure 2-12). Now, this scenario shows the benefits of water import from a region with lower water stress on the overall situation of water scarcity. Nonetheless, this random assumption (-50% in WSI) does not reflect the actual water scarcity situation in Empuriabrava and El Port de la Selva with sufficient accuracy, but it shows the trend of adapting WSI to the locally reported conditions.
3. Expanding the system boundaries significantly changes the WIIX of the reuse schemes, as recharged groundwater is fully reducing the water stress of summer withdrawals now. This assumption leads to a reduction of 45 % in WIIX compared to '0 Status', so that the reuse schemes are now comparable with the WIIX benefits of seawater desalination. Changes in irrigation water quality between both reuse scenarios have only a minor effect on the WIIX score.

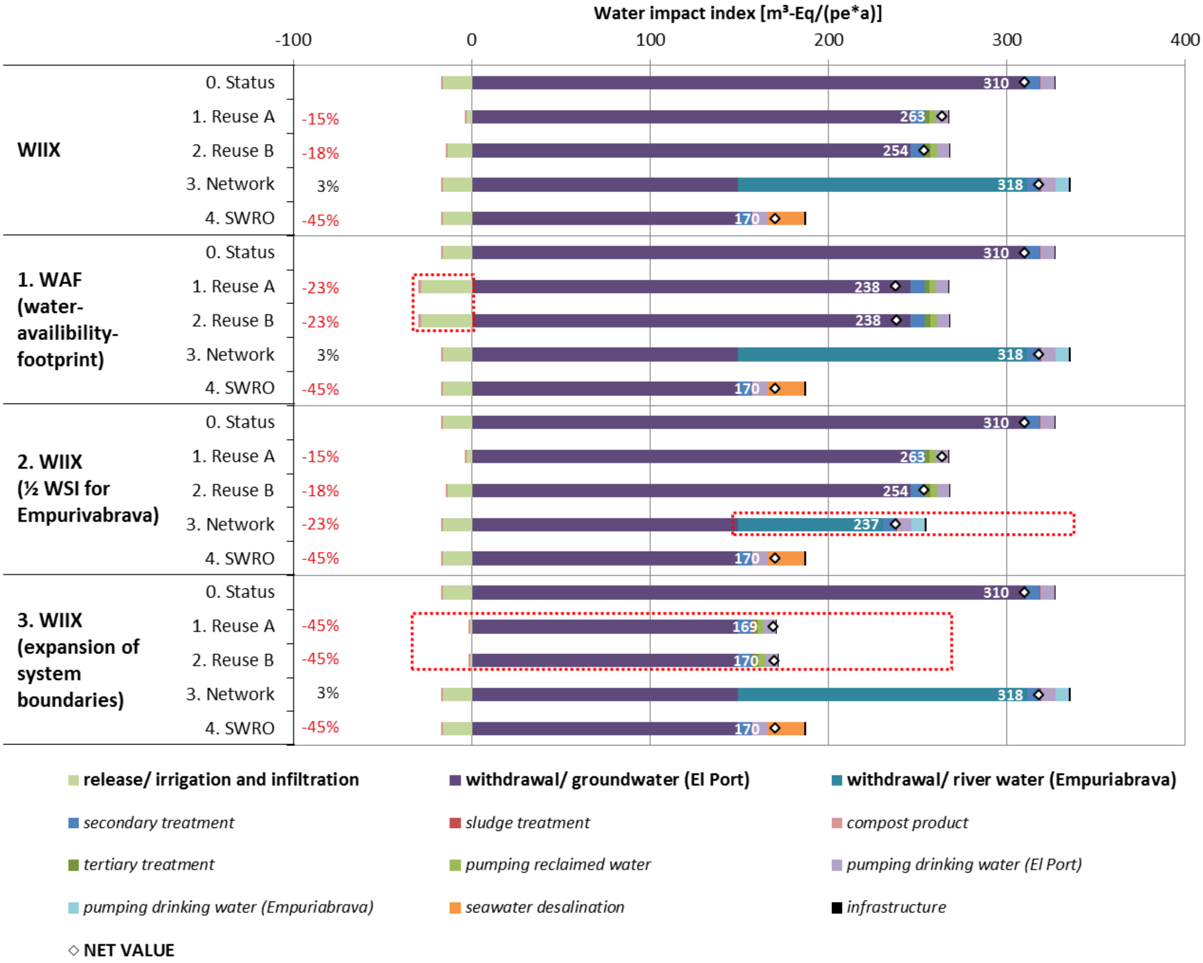


Figure 2-12: Water Impact Index (WIIX) and sensitivity analysis on WIIX

In summary, the different results for the observed scenarios in sensitivity analysis illustrate potential short-comings of the methodology of the WIIX which should be addressed in future studies in this field. A definitive score of water footprint (or favoured mode for calculation) could not be provided in this study. A detailed evaluation of the WIIX as scientific method based on the reflections of this case study is conducted in chapter 8.3.

Summary and Interpretation of results

Table 2-7 gives a summary on the net environmental efforts and benefits of the scenarios for all impact categories, always compared to the existing situation in '0 Status'.

All alternative options to increase the availability of water resources in El Port de la Selva increase both the CED and GWP of the system compared to the existing drinking water supply. The additional water supply accounts for 96 000 m³ per year, thus representing a plus of 30 % of the current total water supply in El Port de la Selva. For this task, water reuse or water import increase the energy consumption and GWP by around 40 % while seawater desalination more than doubles CED and GWP of the existing system. Regarding eutrophication potential, the reuse schemes divert WWTP effluent from the ocean to freshwater environment, which results both in a significant increase in FEP and a comparable reduction of MEP. The normalization of these impact categories reveals a higher score of the MEP, indicating that the reduction of nitrogen discharge into the sea may be seen as more important than the additional load of phosphorus to freshwater. In addition, the actual local risk of eutrophication of freshwater is expected to be low, as no important freshwater bodies are present (which results in freshwater scarcity as a motivation to look for alternative sources). Nonetheless the local boundary conditions have to be considered for weighting these indicators against each other. Indirect impacts on water quality arising in background processes (e.g. electricity production) are neglibile, as are the changes in toxicity indicators between scenarios.

In relation to the potential mitigation of water scarcity with water reuse, the WIIX exemplifies the reduced water stress coming with water reuse (- 15 to 18%) or seawater desalination (-45%). In this study, water import was evaluated with the same WIIX than local drinking water production, but only due to the same WSI (cf. sensitivity analysis) which seems to not properly reflect the local situation.

Table 2-7: Summary of net environmental efforts and benefits of the scenarios for all impact categories for LCA El Port de la Selva, related to '0 Status' as reference

Scenario	0. Status	1. Reuse A	2. Reuse B	3. Network	4. SWRO
CED	1 082 MJ/(pe*a)	+ 37 %	+ 41 %	+ 47 %	+ 150 %
GWP	65 kg CO_2-Eq/(pe*a)	+ 36 %	+ 37 %	+ 42 %	+ 141 %
FEP	8.0 g P-Eq/(pe*a)	+ 80 %	+ 63 %	+ 8 %	+ 29 %
MEP	2.0 kg N-Eq/(pe*a)	- 33 %	- 33 %	± 0 %	+ 1 %
TAP	1.1 kg SO_2-Eq/(pe*a)	+ 11 %	+ 13 %	+13 %	+ 43 %
ETP	478 CTUe/(pe*a)	+ 16 %	+ 9 %	+ 2 %	+ 7 %
HTP	0.8 10^{-3} CTUh/(pe*a)	± 0 %	± 0 %	± 0 %	+ 1
WIIX	26 m^3-Eq/(pe*a)	- 15 %	- 18 %	+ 3 %	- 45 %

Summarizing the results of this LCA, it can be stated:

- The existing reuse scheme '1 Reuse A' has the lowest additional CED and GWP from all options studied and alleviates the local water stress significantly. Water reuse also lowers the negative effects of WWTP effluent discharge to the sea in terms of nitrogen pollution. Apart from the positive environmental effects, is has to be noted that reclaimed water used for irrigation or infiltration may still contain some trace organics, as GAC does not provide complete removal of these substances. In addition, salinity of reused water is higher than natural groundwater which could induce negative effects for irrigation.
- An alternative reuse scheme with membranes ('2 Reuse B') would only marginally increase the CED and GWP compared to the existing reuse scheme. However, the assumed energy demand for membrane treatment has to be validated in full-scale, as the data of this LCA is based on extrapolation from other studies. As RO provides high removal of trace organics and also salinity, these potential drawbacks of an indirect potable reuse scheme (see '1 Reuse A') would not be relevant for this specific reuse scheme. However, higher water quality in salinity and trace organics after RO treatment are not reflected in the indicator results of this LCA, which can be regarded as a limitation of the study.
- Water import from the network of Costa Brava North ('3 Network') has a slightly higher CED and GWP compared to both reuse schemes. Concerning water scarcity, this option does not alleviate the water stress in the catchment when taking regional factors for water stress into account. However, a mitigation of water stress should be possible from a local point of view if water stress is lower in the area of water production than at El Port de la Selva. Finally, the decision between water reuse or water import will not be based directly on environmental benefits or drawbacks, but may rather be seen as a political and economically driven decision.

- Seawater desalination ('4 SWRO') is by far the option with the highest increase in CED and GWP (up to 150 % compared to '0 Status'), but also the maximum reduction of the WIIX due to generation of "new" freshwater of excellent quality in the watershed. Taking into account national and European strategies for energy efficiency and GHG reduction goals, seawater desalination remains as last option for drinking water supply only if all other options are excluded based on other reasons (e.g. potential health risks).

Recommendations to improve the environmental profile of existing water reuse schemes in El Port de la Selva towards higher energy efficiency

In the course of this study, potential optimization strategies emerged to improve the environmental profile of the implemented reuse scheme, reducing the additional energy efforts to a minimum. The summarized potentials for energetic optimization are shown in Figure 2-13 and include the following measures:

- CCB investigated the **installation of solar panels** on the WWTP of El Port de la Selva to partly cover the additional energy needs of water reuse with renewable energy. Although this measure will not reduce the gross energy consumption of the system, the environmental profile of the reuse scheme will be improved in terms of CED and GWP. The solar panels on the area of the WWTP could provide up to 29 000 kWh/a at full capacity. Hence, the panels would reduce the respective efforts of water reuse by **92 MJ/ (pe*a)** in CED and by **4.5 kg CO_2-Eq/ (pe*a)** in GWP. Since the installation of solar panels is not connected to any operational boundaries within the scenarios, this environmental 'optimization' could be achieved for each scenario.
- The infiltration water in winter (48 000 m^3) is produced in the WWTP of El Port de la Selva (2 m above sea level or ASL), is then pumped to the storage tank (115 m ASL) and then flowing by gravity to the infiltration ponds (11 m ASL). This procedure is due to the stepwise development of infrastructure and involves significant elecitricity demand for pumping the water to the storage tank. A **direct pipeline** connection (e.g. connecting the uphill pipeline to the storage tank directly with the downhill pipeline to the infiltration ponds) would effectively reduce the energetic efforts for reclaimed water pumping by 90% (currently 0.69 kWh/ m^3 for pumping to storage tank; approx. 0.06 kWh/ m^3 for direct pumping to ponds). The respective efforts for water reuse would then be reduced by **106 MJ/ (pe*a)** in CED and by **5.4 kg CO_2-Eq/ (pe*a)** in GWP. This optimization potential is related to the distribution of reclaimed water to the ponds and hence could be achieved for both reuse scenarios.
- During data collection for the LCA inventory, high specific energy consumption has been observed for the pressurized filters (0.36 kWh/m^3) currently operated in the tertiary treatment at WWTP El Port de la Selva. However, these compact filter tanks have a similar cleaning efficiency than a larger **dual media filter**, which can be operated with gravity and significantly lower energy demand. Hence, pressurized filters could be exchanged at the next exchange interval, and alternative filtration system with significantly lower energy consumption (ca. 0.06 kWh/ m^3) could be considered. The respective efforts for water reuse could be reduced by **108 MJ/ (pe*a)** in CED and by **5.5 kg CO_2-Eq/ (pe*a)** in GWP with low energy filtration. Naturally, this optimization can only be implemented in '1 Reuse A' scheme.

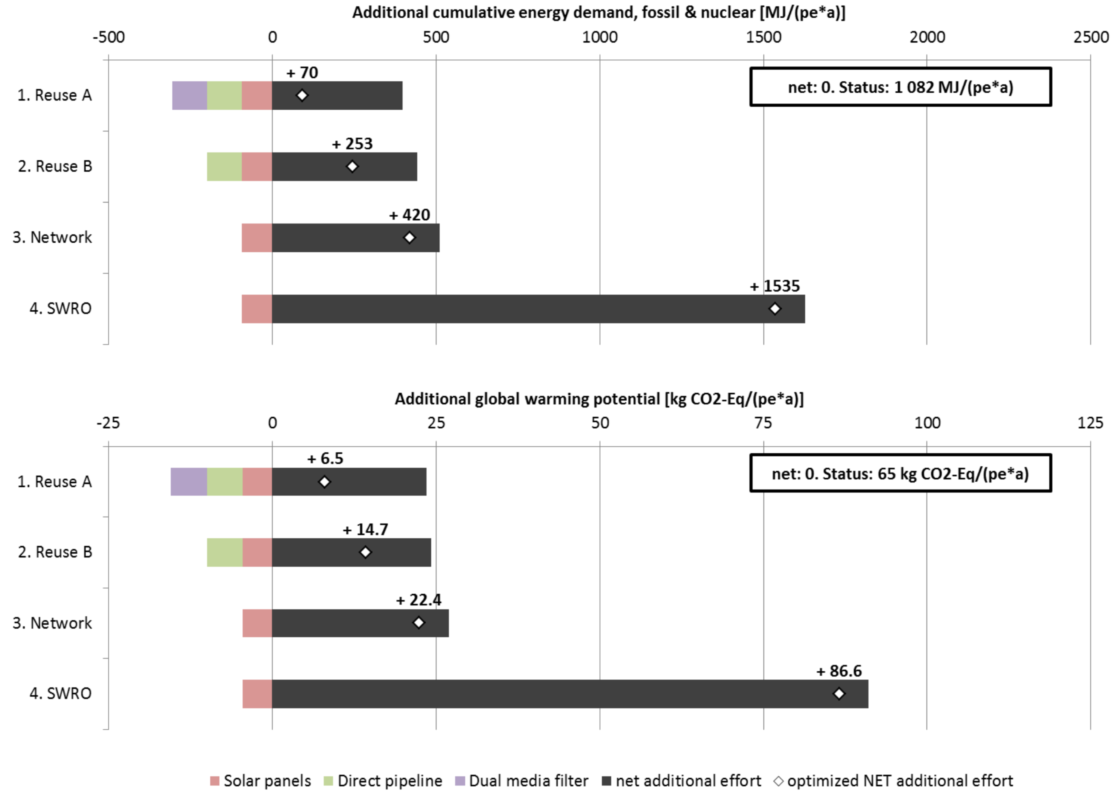


Figure 2-13: Additional net cumulative energy demand and global warming potential of alternative water sources and respective potential for optimisation

Combing all measures for energetic optimisation, the additional efforts of both reuse schemes can be significantly decreased. In particular, additional CED can be reduced by more than 80% for the reuse scheme A, while comparable effects are realized for GWP with -70%. The effect is smaller for reuse scheme B, but still the additional efforts can be reduced by around 40% for CED and GWP. The potential to reduce additional efforts for the options with water import or seawater desalination are restricted to the use of solar panels, as both options are already optimized in energetic terms based on the data used in this LCA study.

2.3 Risk Assessment

2.3.1 Goal and Scope

In El Port de la Selva quantitative microbial risk assessment (QMRA) was conducted in order to quantify the probability that the planned reuse system would be able to meet the WHO health based target (HBT) of 10^{-6} DALYs per person per yea (pppy). In order to do so, first, the WHO target was translated into a log removal performance target for each reference pathogen using the formulas outlined in chapter 2.3.4. Followed by that, the treatment performance of the water reuse system was assessed using available monitoring data as far as available. If no data was available, literature information has been used to estimate pathogen reduction, like e.g. for the assessment of the subsurface passage. Questions the risk assessment aims to address are:

1. Are the existing barriers in winter operation sufficient for drinking water supply, when using the WHO health based target (HBT) of 1 µDALY pppy as a quality benchmark?
2. Are the existing barriers in summer operation sufficient for using reclaimed water for urban and public irrigation, when using the WHO health based target (HBT) of 1 µDALY pppy as a quality benchmark?

The risk assessment focuses on the currently existing systems, which is shown as scenario “Reuse A” in Figure 2-14. The other reuse scenario, using a double membrane system, “Reuse B” which has been assessed in the LCA as an energy optimatization scenario has not been assessed.

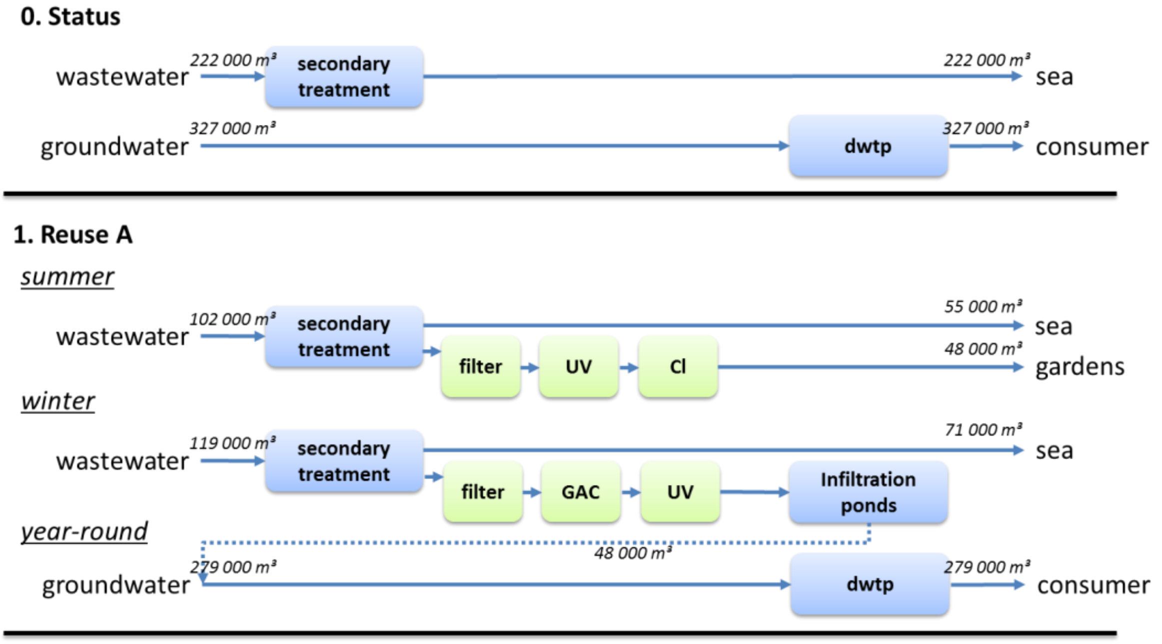


Figure 2-14: Overview of the assessed reuse options in El Port de la Selva

Moreoever, within WP 1 of the DEMOWARE project monitoring campains of organic micropollutants have been conducted in El Port de la Selva. In order to estimate to fate of these substances within the water reuse system the transport of these groups of chemicals has been estimated using the modeling approaches outlined in the Australian Guidelines for Water Recycling - Managed Aquifer Recharge [53].

The calculated concentrations were compared to available limit and precautionary values in order to express risk in term of a risk quotient (RQ). Since the number of measurements of micropollutants is very small especially the chemical risk assessment has to be seen as a screening level assessment. In order to support the future management an assessment tool has been developed which can be used after the DEMOWARE project.

2.3.2 Hazard identification and available information

Pathogens

Wastewater may contain a variety of pathogenic microorganism, depending on the health status of the present population. Since the number of different pathogens is high, the present state of the art of quantifying the risk associated with pathogenic microorganisms is via quantitative microbial risk assessment for so called reference pathogens (see Deliverable 3.1). These pathogens are usually present in high numbers in municipal wastewater, and due to their high prevalence it is assumed that the control of these pathogens suffices to control other pathogens as well. The reference pathogens used in most risk assessment studies are Rotavirus, *Campylobacter jejuni* and *Cryptosporidium parvum* as representatives for viral, bacterial and parasitic pathogens, respectively. Rotavirus has been measured four times in the influent of the WWTP. Analytics have been conducted at Universiity of Barcelona. The concentrations of rotavirus genomic units per liter of raw wastewater obtained during this monitoring campaign are provided in Table 2-12. Concentrations of other pathogens were estimated based on realistic worst case assumption based on literature information.

Organic chemicals

Treated wastewater contains a very high number of different organic substances, which can be grouped into different chemical groups. These groups can be determined by the specific chemical properties or structures of the chemicals (like PAH, furans, dioxins) or by the different purposes the chemicals are used for (e.g. pesticides, pharmaceuticals, personal care products). Organic chemicals which occur in very low concentration in municipal wastewater are most commonly summarizes as micro-pollutants or emerging compounds.

In general, very limited information on the content of micro-pollutants was and still is available in El Port de la Selva. In a first screening level monitoring in 2014 the effluent of the WWTP was analysed for the list of priority compounds outlined in the European regulation. Those compounds contain mostly “classic” organic pollutants among which pesticides are the most common use category. Results are shown in Annex 9.2.2.

Moreover, sampling campaigns were conducted to obtain trace organic compound concentrations indicative for anthropogenic impacts from infiltration to the groundwater composition. Prior to the start of infiltration, samples were obtained from the WWTP effluent, Pz4 (native groundwater) and the drinking water well. Different laboratories measures trace organics with different methods. Therefore, comparisons have to be made with cautions. The laboratories involved were the laboratory of the Berlin Water Utilities (BWB), a Spanish laboratory (CSIC), the laboratory of the European Joint Research Centre (JRC) and the laboratory of Technical University of Berlin (TUB). Given that the analysis was positive in the first screening, the substance was selected for modelling.

In October 2014 Pz4 appeared to be free of anthropogenic impacts (BWB), whereas the drinking water well showed detects for the two pharmaceuticals Metoprolol and Phenazone (May 14, CSIC). This means that these substances have entered the aquifer before infiltration has been started. Other pharmaceuticals or pesticides were (Oct. 14 and Oct. 15) not detected even after repeated sampling.

Until 2016 no further measurements have been conducted, so that no more modelling was conducted. In 2016 Pz2 was sampled during the (discontinuous) infiltration period in February, April, June (2x) and July 2016. As before, the analyses were performed by different laboratories and with different methods.

While CSIC, TU Berlin and BWB analysed water samples, additional samples were taken by JRC using a special filtration device and analysing the filter attachments. Thereby, the different laboratories came to different conclusions. (The selection bases on KWB experiences from similar settings. Please refer to deliverable D1.4 for further monitoring details.)

As these results were contradictory no reliable data on micropollutants was available for this assessment. Because of that and because of new substances of emerging concern are likely to appear in the future a generic assessment tool has been implemented using the programming language "R" which allows for a quick entry level assessment if reliable input data are available or in case that new substance of concern appear. The tool estimates drinking water concentrations after subsurface passage and compares this concentration to predefined risk benchmarks. A default value of 0.1µg/L is used as a risk benchmark in cases where no benchmark is defined by the user. The tool also allows for gaining some general system understanding as it allows to easiliy change boundary conditions and thus to analyze sensitivities.

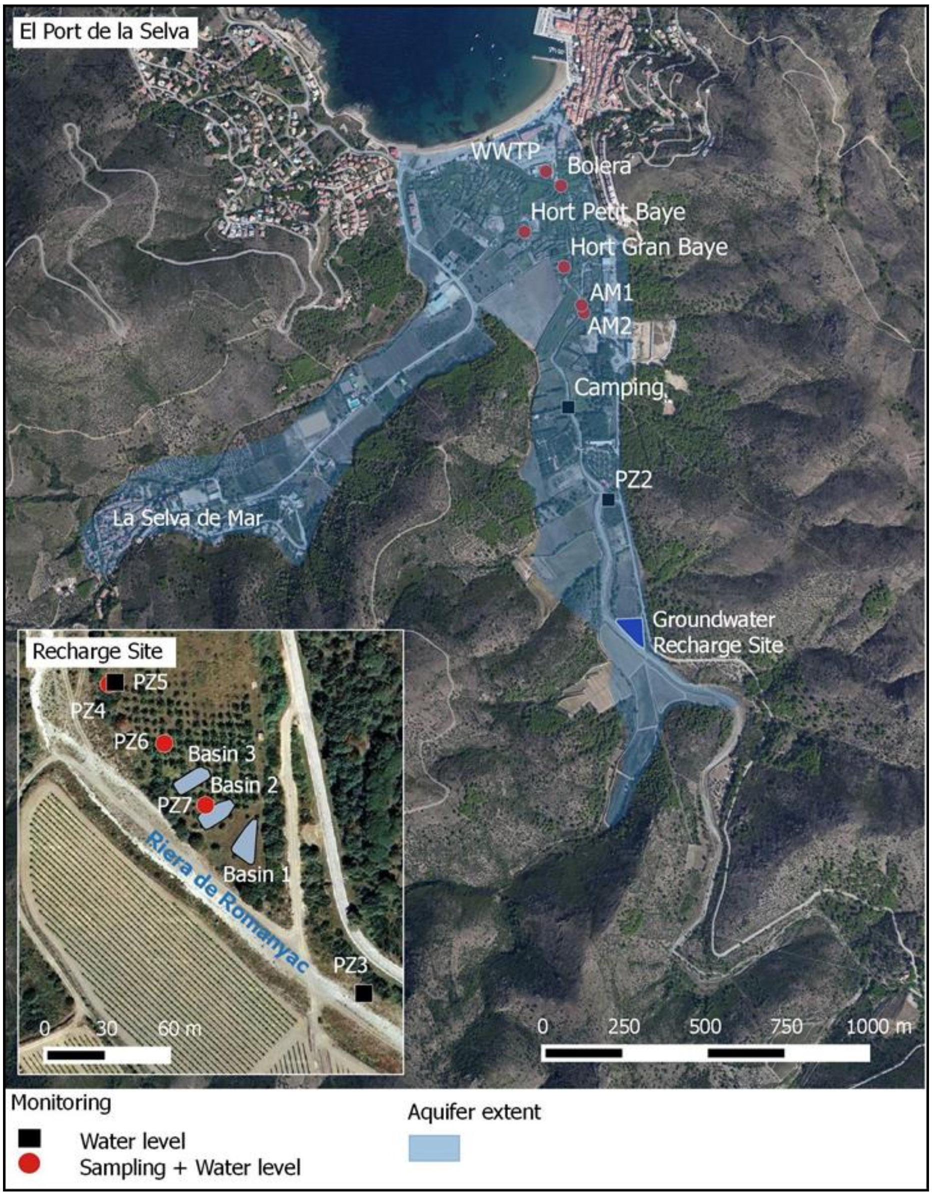


Figure 2-15: Overview of the dome site in El Port de la Selva including the location of groundwater recharge, piezometers and drinking water wells.

2.3.3 Hazard characterization and health based targets

Pathogens

The selected reference pathogens differ in infectivity, physical and biological characteristics as well as in the severity of the kind of disease they cause. In order to make different health outcomes comparable the WHO uses the so called DALY[4] indicator which is calculated by adding the years of life lost (YLL) to the years lived with disability (YLD) due to a specific health outcome. For drinking waters supplies as well as for other water supply systems the WHO defines a tolerable risk level of 10^{-6} DALYs per person per year (pppy). Setting health targets is usually the responsibility of the competent authorities. However, as QMRA is no requirement in the Spanish reuse regulation the WHO guidelines are used.

The most commonly used values for the DALYs caused by the considered reference pathogens were derived by [54]. The average values of DALYs caused per infection as well as the the dose response parameters for the dose response relationship for each pathogen are shown in Table 2-8.

Table 2-8: dose response-models for selected reference pathogens [55], [56], [57]

Pathogen	Dose response model	Pararameters			Probability of infection	Disease per infection ratio	DALYs per case of disease
		k (r=1/k)	N_{50}	α			
Campylobacter	Beta poisson		896	0.145	Probability of infection for campylobacter for low dose approximation	0.7	$4.6*10^{-3}$
Rotavirus	Beta poisson		6.27	0.2531	Probability of infection for Rotavirus	0.05	$1.4*10^{-2}$
Cryptosporidium parvum	Exponential	238			Probability of infection for Cryptosporidium	0.3	$1.5*10^{-3}$

Translation of disease burden into performance targets

The DALY concept was developed in order to make different health outcomes comparable in the domain of public health, e.g. to compare the disease burden from communicable to not communicable types of diseases.

In order to make this health target more applicable for assessing pathogen risk in water reuse systems the health target of 10^{-6} DALYs per person per year is translated into performance targets the system has to achieve. Subsequently, system performance is assessed using indicator organisms. Uncertainty is addressed by using a combination of Bayesian parameter estimation and Monte Carlo Simulation. **In summary, the assessment focuses on the probability that the system is able to achieve the log reduction necessary to be in compliance with a health based target of 10^{-6} DALYs per person per year (pppy).** How the DALY indicator is translated into performance targets in outlined in chapter 2.3.4. An overview of the conceptual approach is outlined in Figure 2-16.

[4] Disabiliy adjusted life years

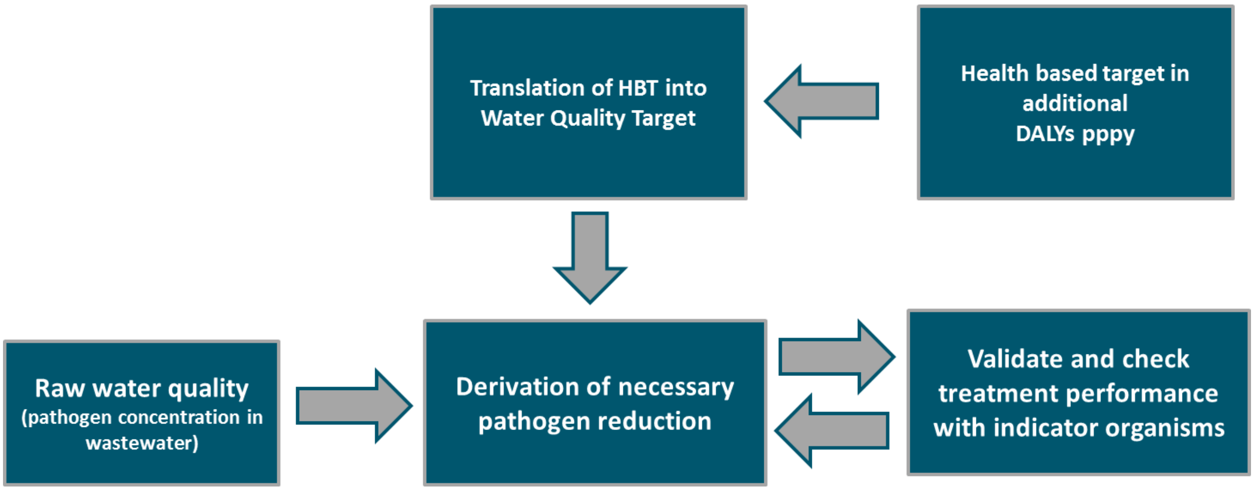


Figure 2-16: Conceptual approach to assess system performance and pathogen risk in El Port de la Selva.
HBT: Health based target

Organic substances

Concerning organic chemcials the European Drinking Water Directive regulates selected single substances as well as certain groups (e.g. pesticides) of organic chemicals. A common European approach which regulates substances of emerging concern like phamaceuticals is currently lacking. Assessment approaches regarding emerging contaminants exist is some European countries, like e.g. in Germany and the Netherlands. In both countries a precautionary value of 0.01 µg / L is applied for highly carcinogenic and genotoxic compounds. Differences exist regarding the guidance values for other organic chemicals. The Dutch Q21 approach applies a value of 0.1µg/L for all other chemicals. Moreover, the sum of organic chemicals should not exceed a value of 1 µg/L. In Germany, a default value of 0.1 µg/L is used as long as no additional toxicological information is available. However, after substance specific toxicological testing this value can rise up to 3µg/L, and partially up to 10µg/L (e.g.EDTA).

During the sampling campaign in 2014, for which risk modelling was conducted four pesticides have been positively detected in the effluent of the WWTP (Diuron, Terbutryn, Dicofol, Cybutryne). For pesticides the limit values from the European Drinking Water Directive of 0.1µg/L for single pesticides and a sum 0.5 µg/L have been applied.

Moreover, five different substances of emerging concern and transpormation productions have been detected (Acesulfam, Carbamazepine (CBZ), Dihydroxycarbamazepine, Sulfametoxazole, Gabapentine). For them the German precautionary health protection value has been applied.

Table 2-9: Overview of used health target values for detected trace organic substances

Substance	Health protection value
Acesulfam	1 µg/L
Carbamazepine	0.3 µg/L
Dihydocarbamazepine	0.3 µg/L
Sulfamethoxazole	0.1 µg/L
Gabapentine	1 µg/L
Pesticides	0.1 µg /L

2.3.4 Exposure scenarios and assessment

For risk assessment three different exposure scenarios have been considered of normal conditions: 1. indirect potable reuse, 2. urban irrigation 3. private irrigation. Table 2-10 gives an overview of the relevant characteristics of the three scenarios. The scenarios for private and public irrigation are summarized in one column of the table.

Table 2-10: Overview of the settings for summer and winter operation in El Port de la Selva with regard to health risk assessment.

Assumptions for exposure assessment for different use categories based on international guidelines ([56],[58])

Settings	Summer (2 Scenarios)	Winter (1 Scenario)
Type of reuse	Private and public irrigation	Indirect potable reuse
Treatment	WWTP → Coagulation + Filtration → UV → Cl_2 → Irrigation	WWTP → Coagulation + Filtration → UV → SAT → Cl_2 → Ingestion
Number of exposure events per year	Exposure event = indirect ingestion via contact with aerosols occurring during watering for private and via contact with plants and lawns for public 90/365 (private) 50/365 (public))	Exposure event = consumption of 2 L per day with 1 L consumed cold every day. 365/365
Volume of water ingested per exposure event [ml]	Route of exposure = Ingestion of sprays 0.1-1(routine) 100 (accidental)	Route of exposure = ingestion of water 1000
Affected population	Users of reclaimed water for both Intended use: garden or park irrigation Unintended use: Ingestion (failure connections in dual pipe system, accidental spills) Children playing	All people in El Port de la Selva via drinking water consumption including: general public (1000 inhabitants) Tourists (10.000 p.)
Hazards	Pathogens (Rotavirus, Campylobacter, Cryptosporidium), toxic anorganic and organic chemical substances	Pathogens (Rotavirus, Campylobacter, Cryptosporidium) toxic anorganic and organic substances
Hazardous events	Misuse, failure connections, failure of wastewater treatment, accidental/illegal chemical spills/ discharges into the sewer system	failure of wastewater treatment, accidental/illegal chemical spills/ discharges into the sewer system, contamination of infiltration ponds by animals
Available information	Monitoring data from secondary and tertiary effluent, aquifer composition, water flows, data from monitoring campaigns for pathogenic viruses, single measurements of organic chemicals	

Pathogens

For the translation of health targets formulated in terms of DALYs into performance targets assumptions have to be made for the number of exposure events per year and the ingested volume of water per exposure event. Moreover, information has to be collected on the expected pathogen concentrations in raw wastewater. Table 2-12 summarizes the made assumptions for the given exposure scenarios as well as the necessary log reductions for the individual reference pathogens.

A tolerable probability of illness can be derived by:

Equation 1 $$P_{tol,ann} = \frac{Tolerable\ disease\ burden}{f_s * db * P_{ill|inf}}$$

$P_{tol,ann}$: Tolerable probability of infection per year

Tolerable disease burden: Tolerable risk in additional DALYs pppy (here 10^{-6})

f_s: suscepitble fraction in the population []

db: disease burden per case of disease [DALYs per case of disease]

$P_{ill|inf}$: probability of illness given infection

Given the tolerable annual probability of infection, for small probabilities the tolerable probability of infection per exposure event can be approximated by:

Equation 2 $$P_{tol,event} = \frac{P_{tol,ann}}{N_{events}}$$

$P_{tol,event}$: tolerable probability of infection per exposure event

$P_{tol,ann}$: tolerable probability of infection per year

N_{events}: Number of exposure events per year [N]

Given the tolerable probability per exposure event the pathogen specific dose-response relationships can be used to calculate the tolerable dose per exposure event. Solving the Beta-Poisson dose-response model, which is most commonly used (e.g. for *Rotavirus, Campylobac*ter) leads to:

Equation 3 $$D_{tol} = \frac{(1 - P_{tol,event})^{-\frac{1}{\alpha}} - 1}{2^{(\frac{1}{\alpha}-1)} * N_{50}}$$

D_{tol}: tolerable pathogens dose per exposure event

$P_{tol,event}$: tolerable probability of infection per exposure event

α, N50: dose response parameters

The dose per exposure event can be translated into a tolerable concentration by dividing by the assumed ingested volume per exposure event by:

Equation 4 $$C_{tol} = \frac{D_{tol}}{V_{event}}$$

D_{tol}: tolerable pathogens dose per exposure event [N/Event]

C_{tol}: tolerable pathogen concentration for the specific exposure scenario [N/100ml]

V_{event}: volume ingested per exposure event [mL]

Finally, the required performance can be formulated in terms of necessary log reductions by:

Equation 5 $$PT = \frac{C_{wastewater}}{C_{tol}}$$

PT: Performance targets in log reduction

C_{tol}: tolerable pathogen concentration for the specific exposure scenario [N/L]

V_{event}: volume ingested per exposure event [L]

Table 2-11: Example for calculating performance targets for public irrigation from disease burden using the formulas above

Symbol	Explanation	Rotavirus	Campylo-bacter	Cryptospori-dium
$DALY_{tol}$	Tolerable risk [additional DALYs pppy]	10^{-6}	10^{-6}	10^{-6}
F_s	Susceptible fraction in the population	0.06	1	1
Db	Disease burden per case of disease	$1.4*10^{-2}$	$4.6*10^{-3}$	$1\text{-}5*10^{-3}$
$P_{ill\mid inf}$	Probability of illness given infection	0.5	0.3	0.7
$P_{tol,ann}$	Tolerable probability of infection per year	$2.4*10^{-3}$	$7.3*10^{-4}$	$4.6*10^{-4}$
$P_{tol,event}$	Tolerable probability of infection per exposure event	$4.8*10^{-5}$	$1.46*10^{-5}$	$9.2*10^{-6}$
N_{events}	Number of exposure events per year [N]	50	50	50
D_{tol}	Tolerable pathogens dose per exposure event [N]	$8.1*10^{-5}$	$7.6*10^{-4}$	$1.6*10^{-4}$
α, r, N_{50}	Dose response parameters (Beta Poisson + exponential model)	$\alpha = 0.2531$ $N_{50} = 6.17$	$\alpha = 0.145$ $N_{50} = 896$	$r = 16.95$
V_{event}	Volume ingested per exposure event [L]	0.001	0.001	0.001
C_{tol}	Tolerable pathogen concentration in reclaimed water [N /L]	$8.1 * 10^{-2}$	$7.6*10^{-1}$	$1.6*10^{-1}$
C_{raw}	Assumed pathogen concentration in raw wastewater (Example) [N/L]	31000	1000000	500
PT	Performance target in required log reduction	5.58	6.12	3.5

Table 2-12: Overview of the assumed influent concentrations of pathogens in El Port de la Selva and the pathogen specific log reduction to achieve the WHO target of 1 µDALY pppy.

Reference pathogens for bacteria, viruses and parasites	Indirect potable reuse (winter operation with UV, SAT and Cl_2)	Urban irrigation (summer operation with UV, and Cl_2	Private irrigation UV, and Cl_2
Assumed Campylobacter concentration	10^6 / L [59]		
Required log reduction Campylobacter	9.98	6.12	5.4 - 6.37
Assumed Rotavirus concentration	31000 genome copies/ L (measured and corrected for recovery rate in El Port de la Selva)		
Required log reduction for Rotavirus	9.45	5.58	5.8
Assumed Cryptosporidium concentration	500 N oocysts/L [60]		
Required log reduction Cryptosporidium	7.37	3.5	3.8

Organic chemcials

For risk assessment of micropollutants, measured micropollutant concentrations have been calculated based on the modelling approaches outlined in the appendices of the Australian Guidelines for Water Recycling. Because there are no drinking water limit values for pharmaceuticals defined in Spanish drinking water regulations, the German approach of health oriented values (HoV) has been used for the assessment. In this approach, a generic limit value of 0.1µg/L is applied for all new and unassessed substances. After further toxicity testing values can become less strict for non-toxic chemicals as well as stricter (0.01µg/L) for e.g. highly potent carcinogenic chemicals. For the selection of relevant substances a screening level monitoring was conducted.

Assumptions

For risk assessment of pharmaceuticals at first, measured concentrations were doubled for making more conservative assumptions. The following boundary conditions were considered:

- first-order exponential decay, distinction between aerobic and anaerobic conditions
- Sorption is expressed by Koc and Kd, L/kg distribution coefficient for linear isotherm
- Volatilisation is neglected
- Just horizontal flow considered: worst case assumption

Table 2-13: Assumptions for aquifer characteristics

Parameter	Values	Unit	Data source/ quality
Organic fraction in soil	0.00002-0.00016	-	Measured
Density of soil	1675	kg/m³	Berlin Tegel
Porosity	0.2	-	Measured
Traveltime	300-500	d	Model Marti Bayer, moderate

The retardation for each substance is calculated based on the following equation:

Equation 6 $$R_f = 1 + \rho_s \cdot K_d / ne$$

R_f: retardation factor [-]

n: porosity [-]

ρ_s: dry bulk density [g/cm³]

K_d: sorption isotherm [ml/g]

Assumptions are shown in Table 2-13. The retarded compound specific flow velocity is calculated by:

Equation 7 $$v_{compound} = \frac{v_{GW}}{R_f}$$

$v_{compound}$: flow velocity of compound [m/d]

R_f: retardation factor [-]

v_{GW}: flow velocity of groundwater [m/d]

The transport time for the compound for the distance between pond and abstraction well is calculated using the equation:

Equation 8 $$t_{compound} = \frac{x}{v_{compound}}$$

$t_{compound}$: flow time for the compound [d]

x: distance between recharge zone and abstraction well [m]

$v_{compound}$: flow velocity of compound [m/d]

The (biological) degradation for each compound during subsurface transport is calculated by first-order degradation term according to:

Equation 9 $$c = c_0 \cdot e^{-\lambda \cdot t_{compound}}$$

c: concentration in abstraction well [µg/L]

c_0: initial concentration in source water [µg/L], gamma distribution of measured concentration in source water

λ: decay constant [1/d]

$t_{compound}$: compound specific transport time [d]

Equation 10
$$\lambda = \frac{\ln 2}{DT_{50}}$$

λ: decay constant [1/d]

DT_{50}: half-life time of the compound [d]

2.3.5 Performance assessment

In order to verify the necessary log reduction for the wastewater treatment plant in El Port de la Selva three sampling campaigns have been conducted for indicator organisms. Samples were taken at 5 points of the system. Additionally historical data have been used to assess the Chlorination step. Figure 2-17 gives an overview of the system and the monitoring locations.

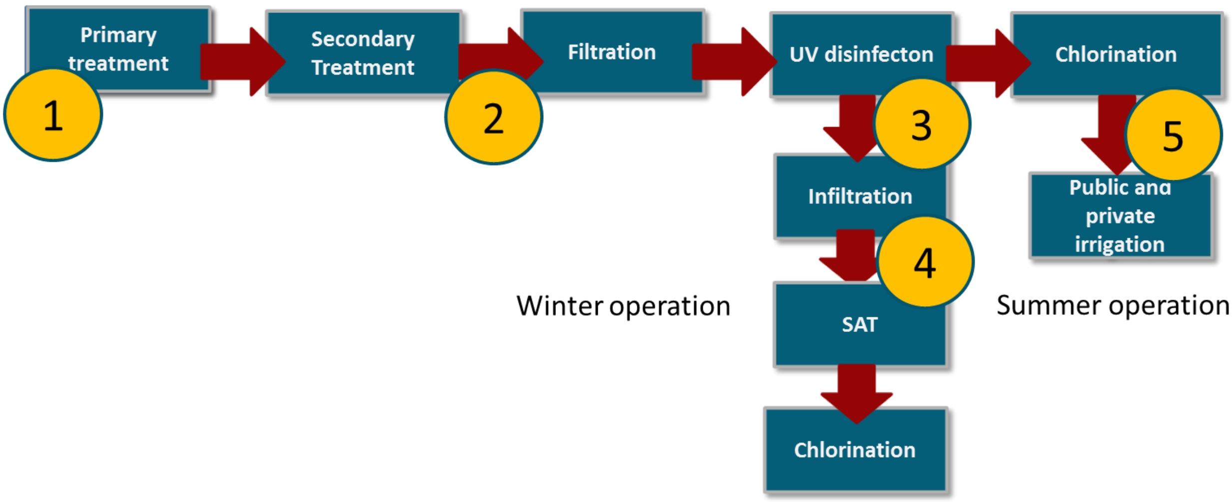


Figure 2-17: Overview of monitoring sport in El Port de la Selva
Given the infiltration scenario wastewater is no longer chlorinated. For assessment of the chlorination step historical data have been used.

The data of monitoring campaigns were continuously updated using Bayesian updating methods for normally distributed data with unknown variance and non-informative prior distribution [61],(cf. annex 9.1.2). Data were transformed by taking the logarithm. By subtracting samples from the effluent from the influent distribution of the mean, the average treatment performance has been quantified for a) the combination out of primary and secondary treatment b) the effluent of the UV disinfection process, and c) the chlorination step. The outlined methods were implemented into the programming language R. Figure 2-18 and Figure 2-19 show the derived distribution for the mean influent, effluent concentrations as well as for the derived treatment performance.

Primary and Secondary treatment

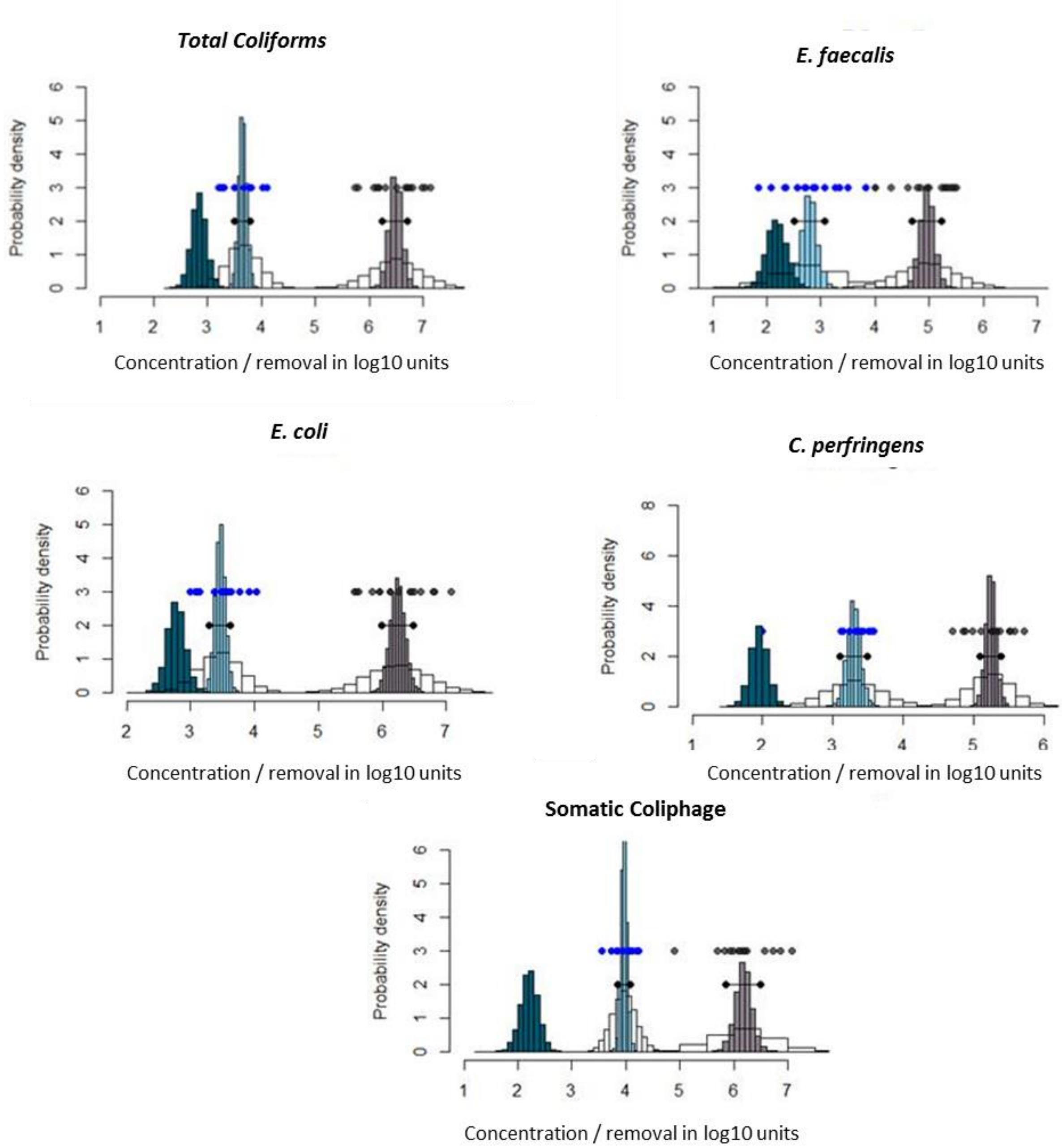


Figure 2-18: Concentrations and removal of indicator organisms by primary and secondary treatment in El Port de la Selva.

The different graphs show the distribution of the mean influent concentration (violet), the distribution of the mean effluent concentration (light blue) and the distribution of the resulting mean log reduction (dark blue) of the different indicator organisms. The underlying measurements (dots) and frequentist confidence intervals are added at arbitrary heights of 3 and 2, respectively. Predictions ($\tilde{y}$ | y) are added as white histograms.

Filtration and UV disinfection

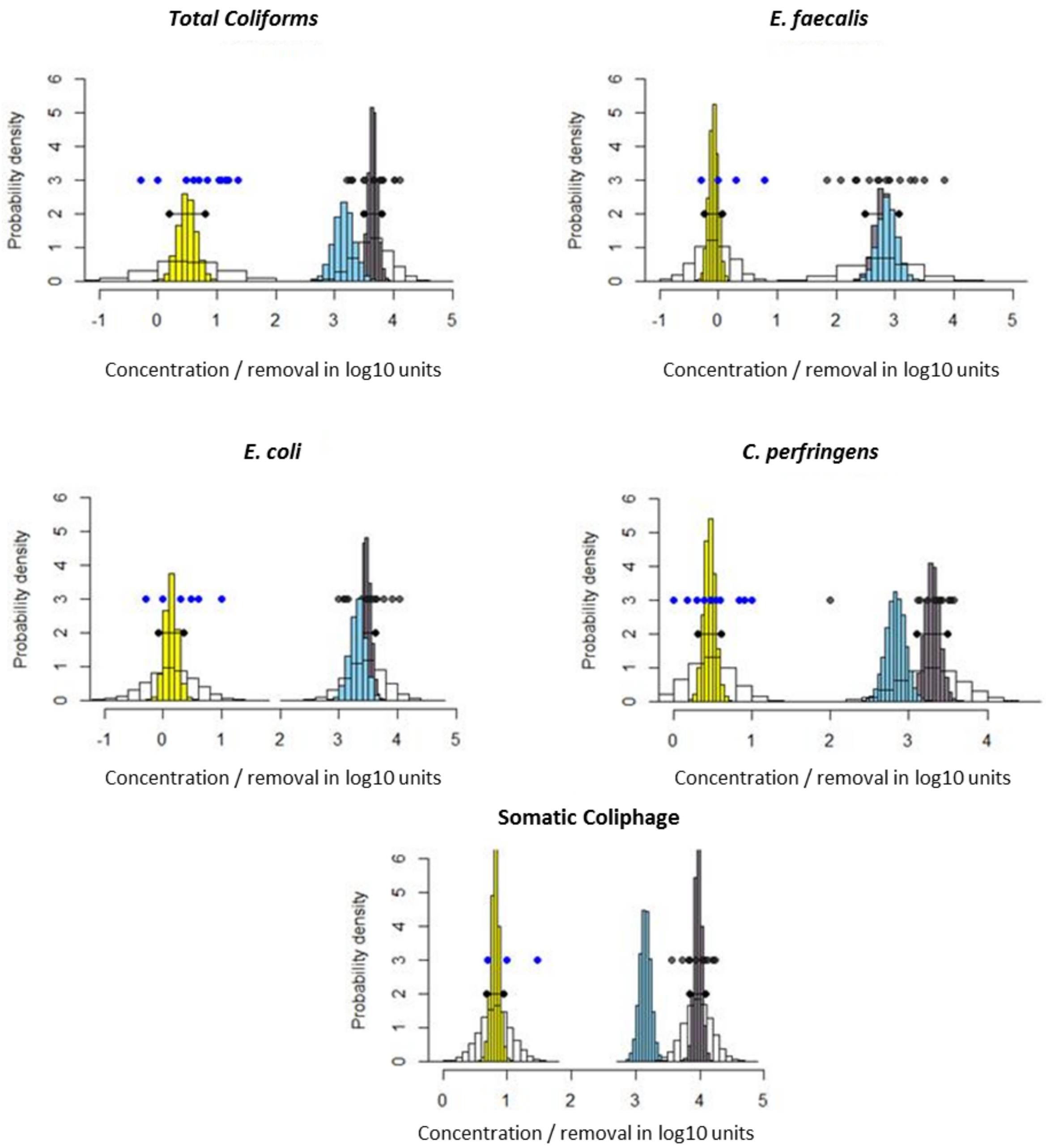


Figure 2-19: Concentrations and removal of indicator organisms by filtration and UV disinfection in El Port de la Selva. The different graphs show the distribution of the mean influent concentration (violet), the distribution of the mean effluent concentration (yellow) and the distribution of the resulting mean log reduction (light blue) of the different indicator organisms. The underlying measurement (dots) and frequentist confidence intervals are added at arbitrary heights 3 and 2, respectively. Predictions ($\tilde{y}$ | y) are added as white histograms.

Chlorination for urban and public irrigation

For the scenarios public and private irrigation the subsequent treatment step after UV disinfection is chlorination. For the assessment of the treatment performance of the chlorination step historical data from El Port del a Selva was analyzed for cases in which still sufficient *E. coli* and other indicators were present after UV disinfection. For *E. coli* 11 data points were identified at which influent concentrations to the chlorination step was above 10 *E. coli* / 100ml. In all cases, *E. coli* concentrations were reduced below the detection limit of 1 / 100 ml. Thus, in the following figure (Figure 2-20) the light blue histogram for the treatment performance completely overlaps with the histogram for the influent concentration.

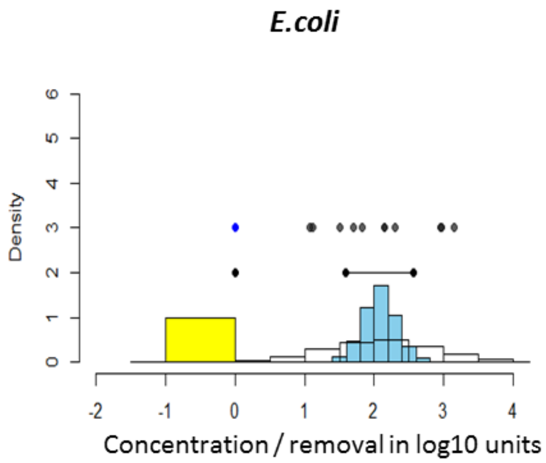


Figure 2-20: Concentrations and removal of E.coli by chorination in El Port de la Selva.
The different histogramms show the distribution of the mean influent concentration (violet), the distribution of the mean effluent concentration (yellow) and the distribution of the resulting mean log reduction (light blue) of E. coli. The underlying measurement (dots) and frequentist confidence intervals are added at arbitrary heights 3 and 2, respectively. Predictions ($\tilde{y}$ | y) of new measurments are added as white histogram.

For other microbial indicators suitable for indicating the removal of viruses and parasites no such cases were found. Thus, literature values are used for estimating the treatment performance (Table 2-14)

Table 2-14: Assumed treatment performance for pathogen reduction by wastewater chlorination.

Source	*Cryptosporidium*	Rotaviruses
WHO (2011)	0 (oocycsts) 2	2
WHO (2006)	0 – 1.5	1 - 3
AUS (2008)	0 – 0.5	1 – 3

Infiltration, subsurface passage and drinking water chlorination

For indirect potable reuse the reclaimed water is infiltrated after UV disinfection. After UV disinfection water quality does not reach drinking water quality requirements. Additional reduction measures consist of the infiltration of the reclaimed water through technical sand, the subsurface passage of 300 days and the final drinking water chlorination. The made assumptions are summarized in Table 2-15. For modelling water quality during subsurface passage the models outlines in the Australian guidelines for Water Recycling have been used. For pathogen removal a first order decay has been assumed (Equation 11).

Equation 11 $$C_t = C_0 * 10^{-time/T90}$$

C_t = pathogen concentration at time t

C_0 = initial pathogen concentration

T90 = time necessary for 1 log reduction [days]

t = time [days]

Table 2-15: Summary of used assumptions for pathogen reduction during managed aquifer recharge (infiltration + subsurface passage) and drinking water chlorination.

Parameter	*Distribution*	*Values*	*Source*
Travel time	Normal	N (μ = 300, sd = 25)	*Model Amphos 21*
Reduction during subsurface passage Campylobacter	Uniform	*T90* *3d -20 days*	Sidhu et al. 2015, from diffusion chamber experiments of 4 different MAR sites
Reduction during subsurface passage Cryptosporidium	Uniform	*T90* *56*-120 days	Sidhu et al. 2015 from diffusion chamber experiments of 4 different MAR sites
Reduction during infiltration for Cryptosporidium	Point estimate	*2*	Literature and expert knowledge Dr. Ulf Miehe
Reduction during subsurface passage Rotavirus	Uniform	*T_{90} = random* *(min = 30, max = 120)*	*Australian Guidelines for Water recycling*
Reduction chlorination (drinking water treatment)	*Point estimate*	*2 log viruses* *2 log bacteria* *0.5 log protozoa*	*WHO Guidelines for Drinking Water Quality*

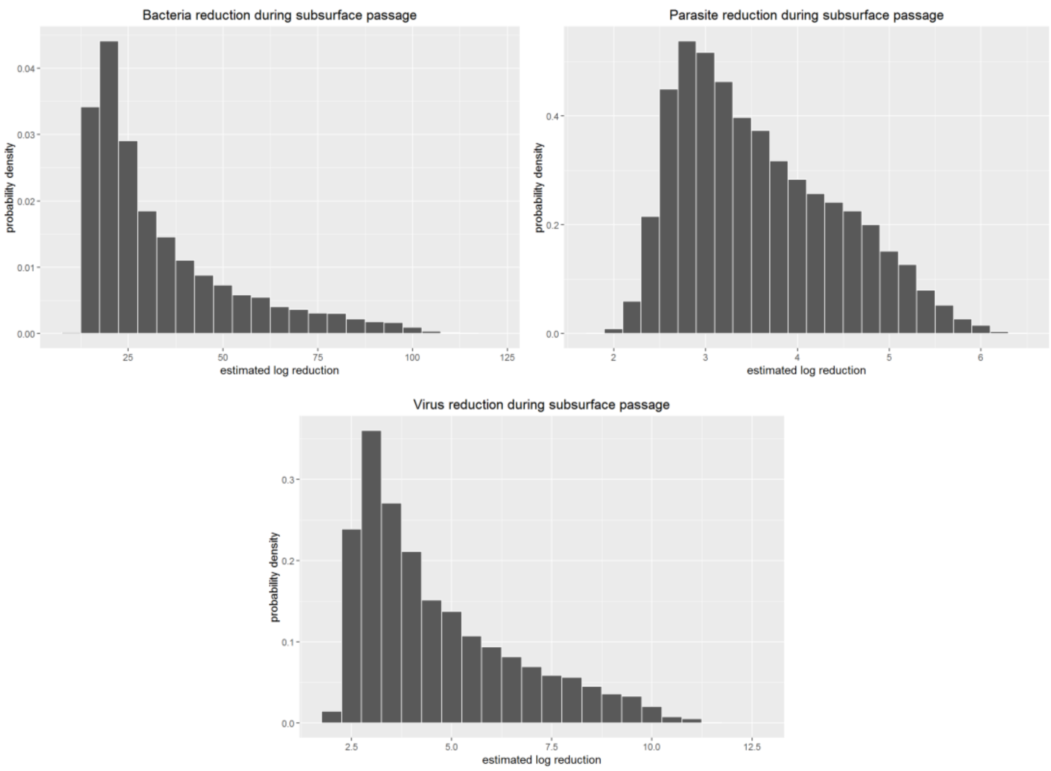


Figure 2-21: Histograms of estimated pathogen reduction during subsurface passage in El Port de la Selva based on the assumptions given in Table 2-15.

2.3.6 Risk characterization

In order to characterize the risk present under normal (i.e. no incident) conditions the performance of the individual treatment steps is added and compared to the initially derived performance targets (Table 2-12). Thereby, the probability of being in compliance with the WHO standard of 1 additional µDALY per person per year is quantified for the three reference pathogens.

Urban and public irrigation

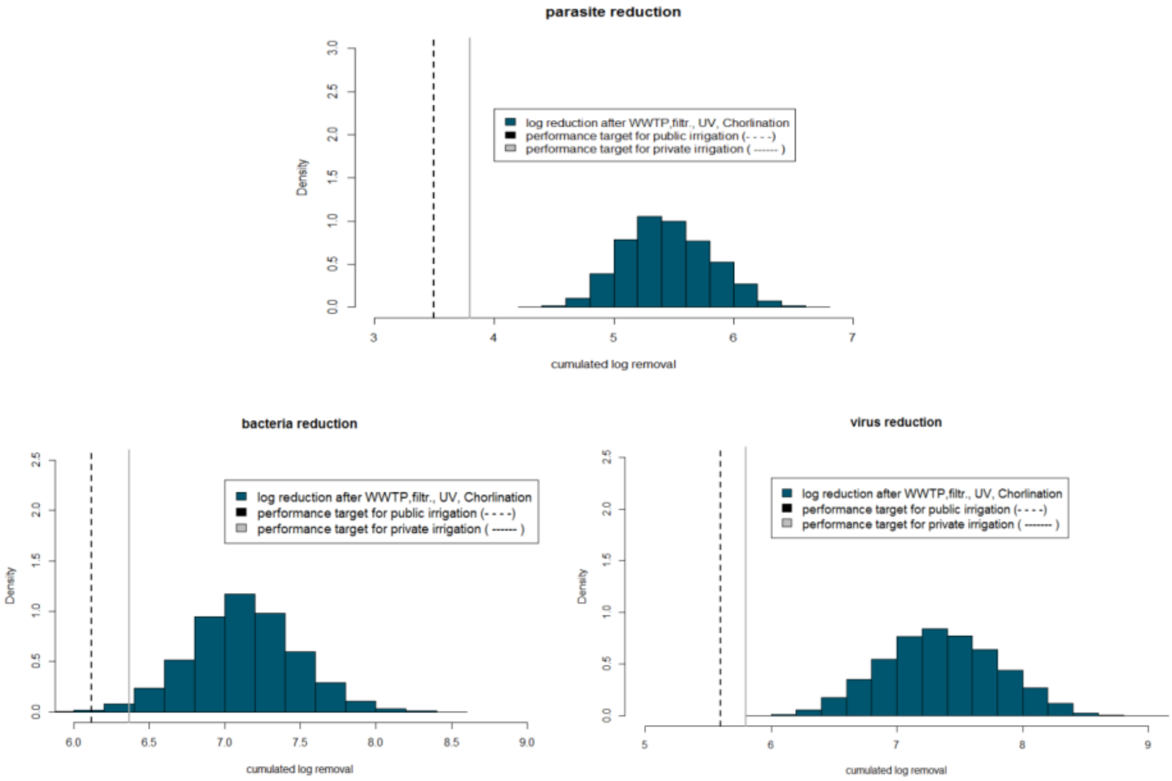


Figure 2-22: The histograms show the expected log removal efficiency of the entire treatment scheme towards bacteria (upper right), parasites (upper left) and virus (lower center).
The vertical lines indicate the performance target for public and private irrigation.

Figure 2-22 show the estimated treatment performance against the set performance targets. For parasites and viruses the estimated distribution lies completely right of the health target indicating almost complete certainty the health target is achieved. For bacteria the histogram slightly overlaps with the set performance target. The area left of the dashed line, which equals the probability of not achieving the required health target however is still < 95% , indicating that for both irrigation scenarios it is also almost certain (i.e. probability > 95%) that the reduction measures in place suffice to reduce the risk below a value of 1 µDALY per person per year (pppy).

However, especially for the private irrigation scenario, it has to be considered that the reclaimed water does not reach drinking water quality and that incident conditions (e.g. failure of the chlorination pump) so far have not been considered. Moreover, the private use of reclaimed water gives a lot of responsibility in the hands of the individual. Especially in tourist regions like El Port de la Selva the population is characterized by high fluctuations. Visitors to the city may not be used to or do not even know that something like dual pipe system might exist. Children may use water from both piping systems for playing. Consequently, a continuous proactive communication about the existence and use of reclaimed water is a necessity. Given the touristic character of the village, communication should be made available not only in national languages but also consider the different nationalities of present tourists.

In comparison, using the reclaimed water for public irrigation may offer additional non-treatment options for risk reduction, like protective cloths for workers or irrigation just during night-time. The direct access of children can be avoided and in case of an incident at the water reclamation plants the communication just has to reach the responsible people at the local municipality instead of the entire village. Thus, if the water demand for public irrigation matches the water supply with reclaimed water preference should be given to the option of public irrigation, which in itself constitutes an additional risk reduction measure.

Indirect potable reuse

As for the scenarios of public and private irrigation risk as characterizes by comparing the initially derived performance target for viruses, parasites and bacteria to the monitored, modelled and estimated log performances of the individual treatment steps. The results before and after drinking water chlorination are illustrated in Figure 2-23, Figure 2-24, and Figure 2-25.

The widths of the histograms illustrate the different degrees of certainty regarding the performance of the individual treatment steps. It becomes evident that although the log performance of the WWTP varies between 2-3 orders of magnitude the major source of uncertainty lies in estimating the log performance of the subsurface passage. For viruses, this uncertainty leads to the results that even after a long subsurface passage of 300 days there is still a probability of approx. 52% (area left of the dotted line) that the required log performance for viruses is not achieved without drinking water chlorination. However, the final chlorination step reduces risk below acceptable limits with probability > 99%.

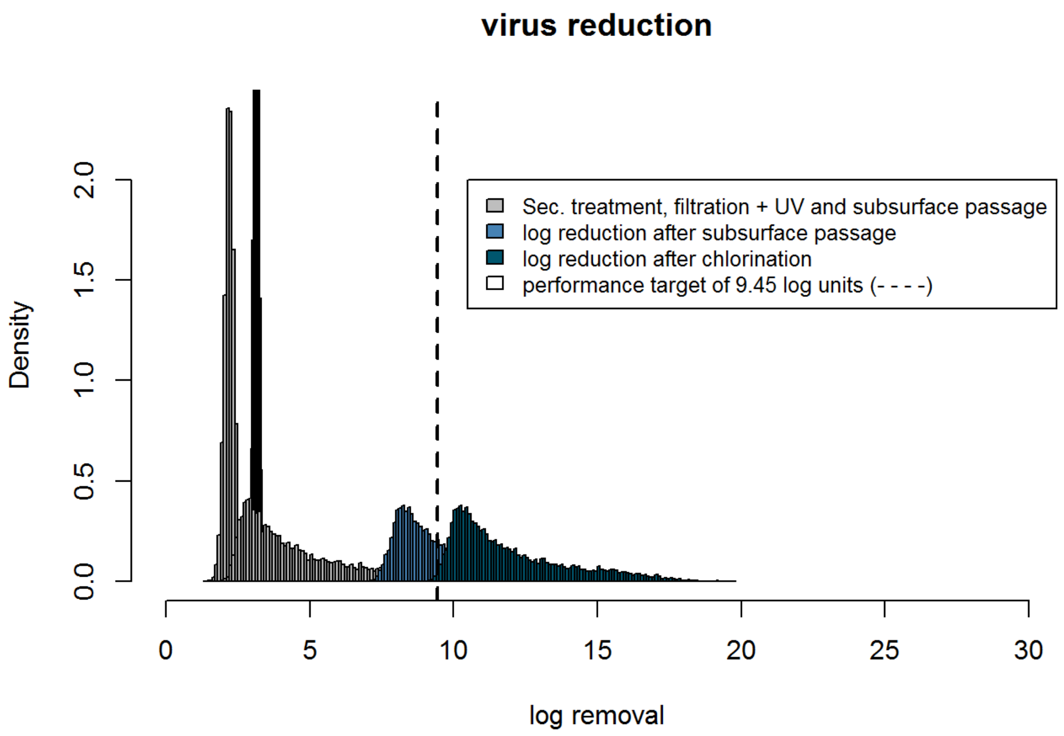


Figure 2-23: Comparison of estimated log performance for viral pathogens against derived performance target.

Due to the high recalcitrance of parasites against chlorination drinking water chlorination just has a minor effect on the overall outcome of the treatment performance (0.5 log units). In this case it is the efficient removal during infiltration and subsurface passage ensures that the probability distribution of being in line with the derived performance target lies completely right of the derived performance target indicating that under the made assumptions the risk of not achieving the required performance target is negligible.

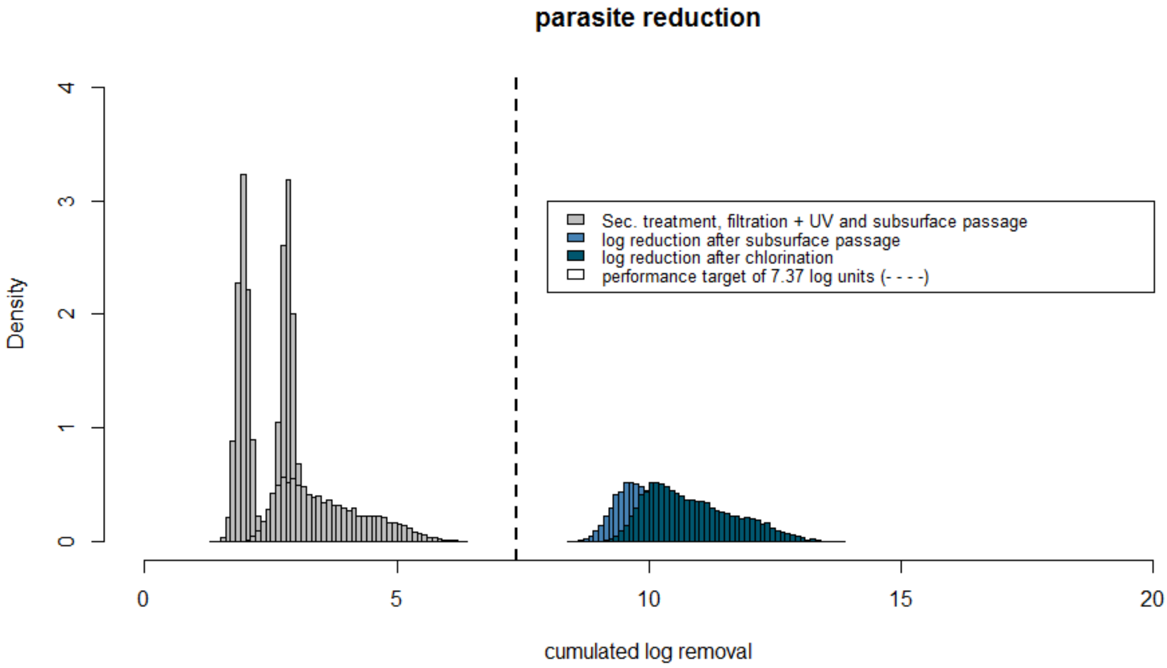


Figure 2-24: Comparison of estimated log performance of parasitic pathogens against derived performance target.

Again, the largest source of uncertainty lies at the treatment performance of the subsurface passage. However although exact predictions of the true removal efficiency of the subsurface passage are quite uncertain, there is high degree of credibility that the required performance of 3 log units after WWTP can be reached. For bacterial pathogens there is a very high degree of credibility that the indirect reuse system in El Port de la Selva will achieve the necessary log reduction with and without the final chlorination step.

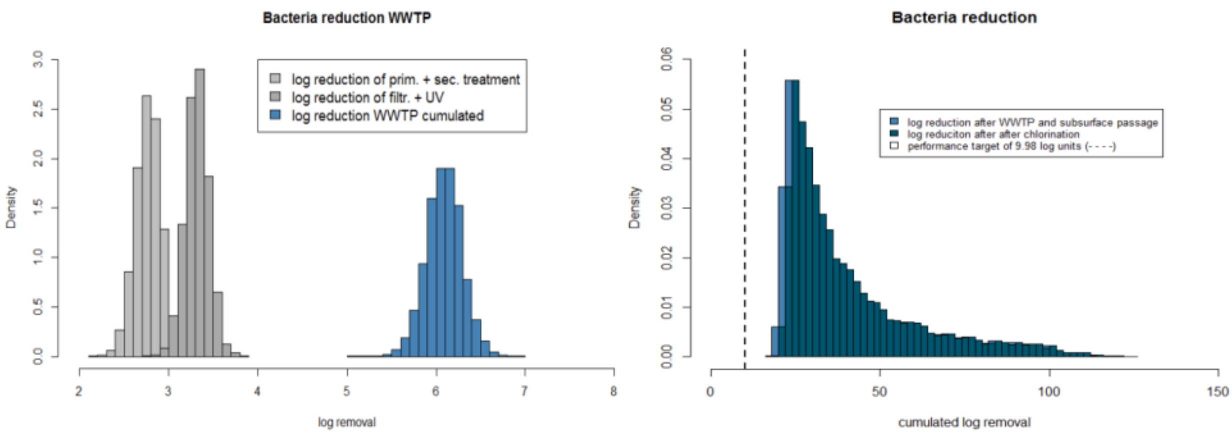


Figure 2-25: Comparison of estimated log performance of bacterial pathogens against required performance target.
Treatment performances of the WWTP (upper right) and the subsurface passage (upper left) were illustrated in separate histograms due to large difference in scale.

Or course, due to the major importance of a safe drinking water supply it is highly recommended to verify the exact flow conditions and pathways, update the information about travel times in the aquifer and monitor groundwater concentrations along these identified pathway on a regular basis.

Organic chemicals

The model results and the substance characteristics are visualized in Figure 2-26 to Figure 2-29 for pesticides and pharmaceuticals, respectively. Sensitivities were checked for dilution (15% and 20%) as well as the expected travel time (300d and 500d) of reclaimed water in the aquifer. For pesticides the model results indicate that Cybutryne is the pesticide of highest relative relevance. Given the measured concentration of 3.2 µg / L in the effluent of the WWTP in combination with a travel time of 300 days and a ratio of reclaimed water of 20% in the drinking water well a risk quotient (RQ) of 1 may be exceeded (Figure 2-26 right, red mark). In all other scenarios the RQ for Cybutryne lied between 0.1 and 1. Up to spring 2016 Cybutryne was a licenced antifouling biocide which was used to protect boats from algae, which explains the measured concentrations in El Port de la Selva. However, in early 2016 the use of Cybutryne has been forbidden by the European Commission for this purpose and concentration should decrease. Since measurements have just taken place in 2014 additional measurements are recommended. For Dicofol and Terbutryn risk quotients were below 0.1 for all scenarios. For Diuron RQ were calculated to lie between 0.1 and 1 for travel times of reclaimed water of 300d and below 0.1 of a travel times of 500d. The increase of travel time had a larger effect than the increase in dilution.

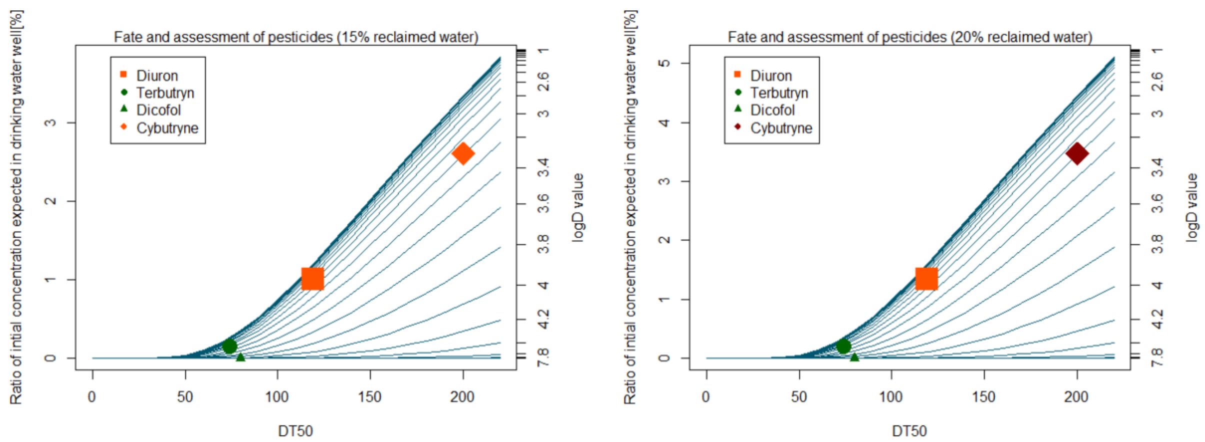


Figure 2-26: Model results as ratio of initial concentration in drinking water for pesticides and a travel time of 300d for the reclaimed water

The colour of the symbols indicates if risk quotients RQ are > 1 (red), between 0.1 and 1 (orange) or below 0.1 (green). The size of the symbols is proportional to the influent concentration of the individual substance.

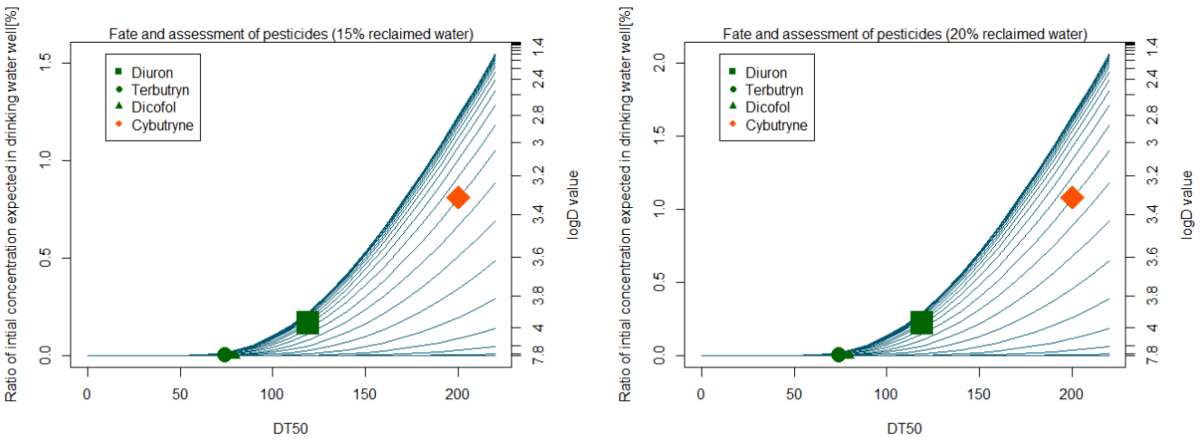


Figure 2-27: Model results as ratio of initial concentration in drinking water for pesticides and a travel time of 500d for the reclaimed water

The colour of the symbols indicates if risk quotients RQ are > 1 (red), between 0.1 and 1 (orange) or below 0.1 (green). The size of the symbols is proportional to the influent concentration of the individual substance

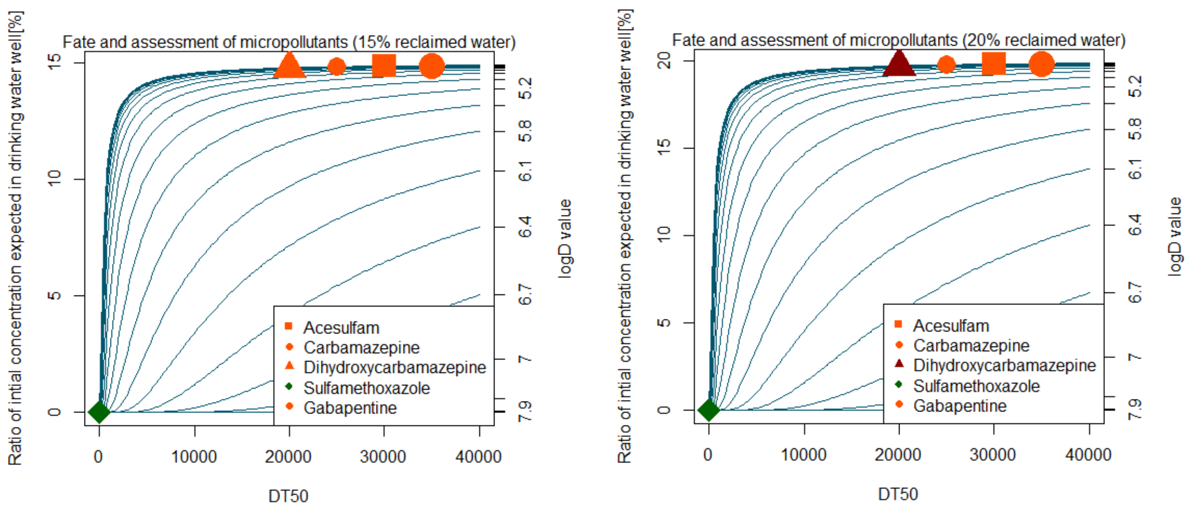


Figure 2-28: Model results as ratio of initial concentration in drinking water for emerging compounds and a travel time of 300d for the reclaimed water

The colour of the symbols indicates if risk quotients RQ are > 1 (red), between 0.1 and 1 (orange) or below 0.1 (green). The size of the symbols is proportional to the influent concentration of the individual substance

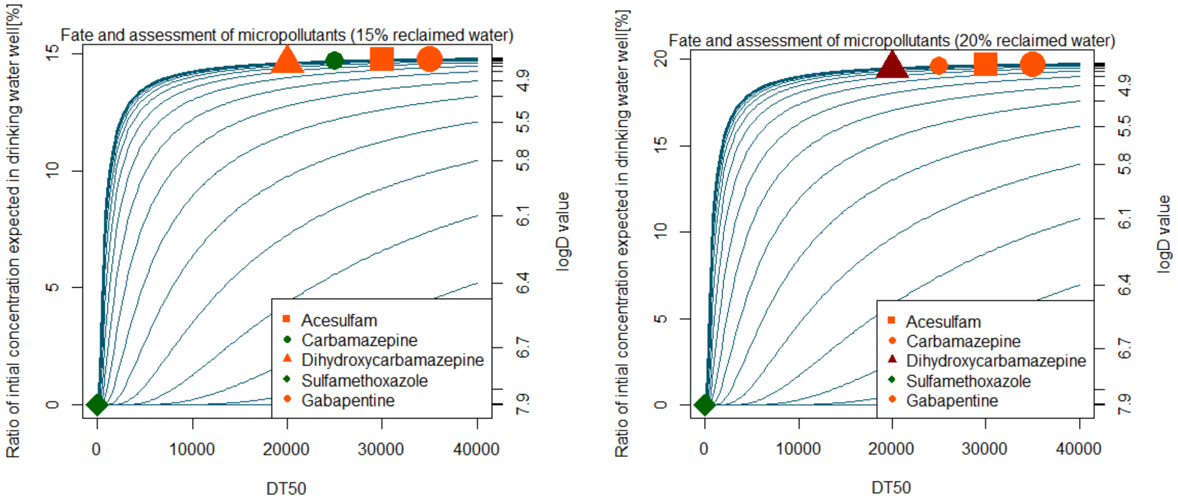


Figure 2-29: Model results as ratio of initial concentration in drinking water for emerging compounds and a travel time of 500d for the reclaimed water

The colour of the symbols indicates if risk quotients RQ are > 1 (red), between 0.1 and 1 (orange) or below 0.1 (green). The size of the symbols is proportional to the influent concentration of the individual substance.

The substances of emerging concern carbamazepine, dihydroxycarbamazepine, gabapentine and acesulfam were considered to be recalcitrant to biological degradation as a realistic worst case scenario. For modelling "persistence" the half-life was set to very high values between 20000 and 35000 days so that biodegradation become negligible in the model. Reductions of concentration are the results of dilution. The figures show that assessment results are very sensitive to the assumption for the dilution rate. While for a dilution of 85% all substances lie below a RQ of 1, a reduction of the dilution rate to 80% (= 20% reclaimed water) changes the assessment of dihydroxycarbamazepine.

The results once more underline the need to increase the amount of reliable information about travel times, dilution rates and aquifer characteristics. The values used for the assessment are precautionary values, which are considered not to pose an additional risk even if the drinking water is consumed during a whole lifetime. The fact, that reclaimed water will not be infiltrated all year long in El Port de la Selva may be an argument for applying a less strict value.

Regarding additional reduction measures, additional investigations for implementing activated carbon treatment is currently considered and additional measurements have been done during early 2016. As results from different laboratories are still getting evaluated they are not included in the present assessment. However, the used model has been included into an easy-to-apply assessment tool programmed in the open source programming language R which will be handed over to Consortia Costa Brava and ACA so that new information can directly be used to update the assessment.

2.4 Conclusions

For the reuse scheme of El Port de la Selva, risk assessment analysed the potential impacts of the reuse of secondary effluent on human health from pathogenic microorganisms. Given the methods and assumptions outlined above, the following conclusions can be drawn:

1. Given the data from 3 sampling campaigns and assuming adequate operation and maintenance of the reuse system, there is high certainty (probability > 99 %) that **risk reduction measures in place in El Port de la Selva are suitable to achieve WHO standards of 1 µDALY per person per year** for urban irrigation, public irrigation, and also the water reclamation scheme for indirect potable reuse (IPR).

2. **Data gaps were identified for indicator organisms indicating viral (coliphages) and parasitic (C. Perfringens) removal by chlorination**, so that data quality should be further improved (e.g. by monitoring) to decrease the uncertainty in the results.

3. **The largest source of uncertainty is the treatment performance of the subsurface passage.** Since the performance of the whole system relies on this treatment performance it is highly recommended to verify flow patterns, update travel time estimates and continuously monitor the hygienic groundwater quality along the transects between the infiltration ponds and the drinking water wells.

4. **In summer, preference should be given to public irrigation (and not private)** from a risk perspective if water demand in public irrigation is high enough so that the reclaimed water can be used sensibly.

5. Given that the reclaimed water is used **for private irrigation clear and easily understandable information signs** should be installed.

6. There is still a **lack of both information and regulation regarding organic micropollutants**. Existing precautionary benchmark from other EU member states might be exceeded, given the current system design.

7. **For verification of the results of risk assessment, the following additional monitoring is recommended**:

    a. Verification of sub-surface travel time by weekly monitoring of suitable organic or inorganic tracer in drinking water well
    b. After infiltrated water has arrived at the well, at least two intensive sampling campaigns over at least 14 days (daily sampling) with high sample volumes to be analysed for viruses and parasites
    c. Installation of an additional piezometer in the aquifer upstream of the drinking water well (> 30 days travel time to well based on verified travel time) equipped with an online probe for conductivity indicating arrival of infiltrated water
    d. If intensive sampling campaigns do not show reason for concern, monitoring frequencies can be reduced and focused on time spans when reclaimed water is supposed to arrive, based on probe measurements (e.g. monthly sampling)

Potential environmental impacts of the reuse scheme were assessed by LCA and WIIX and compared to other alternatives for additional water supply. From the LCA, the following conclusions can be drawn:

- **Water reuse has a lower energy demand and associated environmental impacts (e.g. GHG emissions) than import of water via pipeline or seawater desalination**. In addition, water reuse decreases the additional water footprint of water supply to neutral, posing **no additional stress on the local water balance**. Water reuse also leads to less nutrient emissions into the ocean (e.g. nitrogen causing marine eutrophication), but will introduce some phosphorus into the aquifer. However, local risk for eutrophication of surface waters by P emissions is low.
- The analysis shows that the **existing reuse system could be optimized in energy demand**, reducing electricity consumption for the system significantly by improving pipeline pumping requirements to infiltration ponds (direct intersection without using elevated storage tank) and changing pressurized filtration step of tertiary treatment to a gravity-driven filter. Environmental impacts of energy demand for water reuse can be further minimized by installing solar panels on the WWTP premises.
- LCA results for a potential **membrane system (UF for irrigation, UF + RO for IPR) show that this tertiary treatment would not significantly increase energy demand of water reuse**, based on transfer of data from another reuse site in DEMOWARE (Torreele). This alternative for tertiary treatment is an option for increased risk reduction concerning trace organics and reduction of salinity in the infiltrated water during IPR. However, other aspects should also be taken into account in this decision, e.g. operational and investment costs, and also operational aspects (downtime of membranes?) and maintenance requirements.
- Overall, results of LCA and WIIX show that **water reuse in El Port de la Selva is a recommended option from an environmental point of view compared to water import and seawater desalination**. If further risk reduction is requested by local authorities for IPR, membranes could be considered as alternative tertiary treatment without major drawbacks in environmental profile of water reuse. However, process data for operation of the membrane treatment under the specific conditions in El Port de la Selva should then be further précised and validated against the assumptions in this LCA.
- WIIX results show that water reuse can mitigate local problems of water scarcity. However, **it should be validated if aquifer recharge with reclaimed water in winter does really leads to more available groundwater at drinking water wells in the late summer months**, i.e. help to reduce problems of saltwater intrusion.
- While LCA is well suitable to show environmental impacts of indirect processes for water supply options (i.e. demand of electricity, chemicals, infrastructure), the **potential effects of different water qualities on ecosystems or humans cannot be precisely predicted with existing LCA methods.** Therefore, the combination of LCA/WIIX and risk assessment is seen as a useful tool to analyze both the global and local effects of water reuse systems in such a setting.

3 Case Study of Braunschweig

3.1 Introduction and Setting

In Braunschweig water reuse has a long tradition. The wastewater of the city of Braunschweig and nearby smaller towns is collected centrally and is used for restricted irrigation.

The WWTP in Braunschweig has a design capacity of 275’000 pe, but does currently treat a raw wastewater load equivalent to 350,000 pe with an average annual volume of 21 Mio m³ wastewater. The treatment plant includes primary sedimentation and activated sludge treatment, including enhanced biological removal of nitrogen and phosphorus. Two thirds of the WWTP effluent (ca. 15 Mio m³ per year) is used for the irrigation of 2700 ha of agricultural area of the sewage association Braunschweig (AVB). The remaining third enters a system of ponds and irrigation fields as a final polishing step, before it is re-collected in drainage systems and discharged into the Aue-Oker-Canal.

The sludge from primary sedimentation and the activated sludge process is digested in an anaerobic treatment step in order to reduce its volume as well as to generate methane, which is used for energy production in CHP plants. During the summer period, digested sludge is mixed with the effluent of the WWTP and is used for restricted irrigation of agricultural areas. In winter, digested sludge is dewatered and stored before being used as fertilizer on agricultural fields outside the AVB area. Due to this operating regime, about 50-60% of the annual amount of sludge is applied to the area of AVB [62]. The current practice of wastewater treatment and reuse in Braunschweig is illustrated in Figure 3-1.

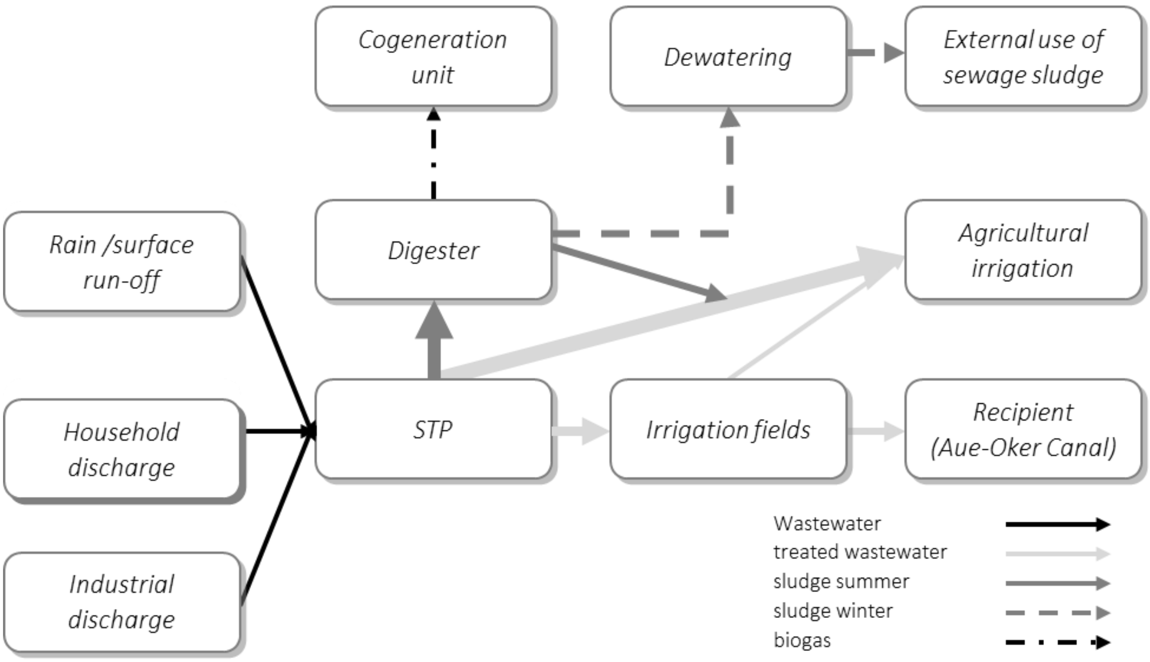


Figure 3-1: Current practice of the wastewater reuse scheme in Braunschweig
STP: sewage treatment plant, thickness of the arrows for water and sludge streams indicates the volume factions of the respective flows (adapted from [63])

3.2 Life Cycle Assessment

3.2.1 Goal and scope definition

The goal of this LCA is to analyse potential environmental impacts of the existing wastewater reuse scheme at WWTP Braunschweig-Steinhof in the current status and for different optimisation measures. In addition, the reuse scheme will be compared with a hypothetical system without water reuse as benchmark to quantify environmental benefits and drawbacks of this particular approach of water reuse.

In detail, the following aspects will be analysed:

1) Comparison of the existing reuse scheme to a system without water reuse to show benefits and drawbacks of water reuse in general
2) Comparison of different options for final water disinfection before reuse in agriculture, which can be introduced in the future to reduce associated health risks from microbial contamination of the WWTP effluent
3) Comparison of optimisation measures to increase energy recovery and nutrient utilization effiency during sludge treatment and disposal for decoupling of water and nutrient management
4) Combining disinfection with optimised sludge and nutrient management to develop an improved "Braunschweig model" which includes measures for risk reduction and decoupling of water and nutrient management [50].

This LCA can serve as an example for reuse sites which are facing rising requirements associated with water and nutrient recycling, either due to the need for risk reduction or the demand for increased efficiency in energy and nutrient use. The target group of this study consists primarily of the treatment plant operator (SEBS), but also planers and engineers in the field of wastewater treatment and technologies for disinfection and nutrient recovery.

Function/ Functional Unit

The function of the system under study is "to provide wastewater treatment according to the legal requirements and to provide related water and nutrient supply to agricultural land of the Abwasserverband Braunschweig (AVB)", including all processes that are related to this function. The functional unit of this LCA is defined via the annual organic load of the WWTP calculated in population equivalents (pe) of the WWTP ("per pe and a" or $[pe*a]^{-1}$), accounting for 120 g chemical oxygen demand (COD) per person and day [39]. The WWTP annually treats wastewater equivalent to 350 000 pe. The additional functions to provide water and nutrients to the agricultural land are also allocated to the function of wastewater treatment.

System boundaries

As this LCA analyses the entire system of water and sludge management in Braunschweig, the system boundaries include the complete WWTP, the wetlands for polishing of the effluent and the distribution system to agriculture, up to the point of water distribution on the fields (Figure 3-2). Water and nutrients delivered to agriculture are accounted as environmental emissions, crediting equivalent avoided products such as pumping of groundwater and mineral fertilizer production in an "avoided burden" approach. Credits are calculated with regard to nutrient and water demand and utilization efficiency [50]. Finally, system boundaries include background processes for production of electricity, chemicals, fuels, materials, infrastructure and maintenance.

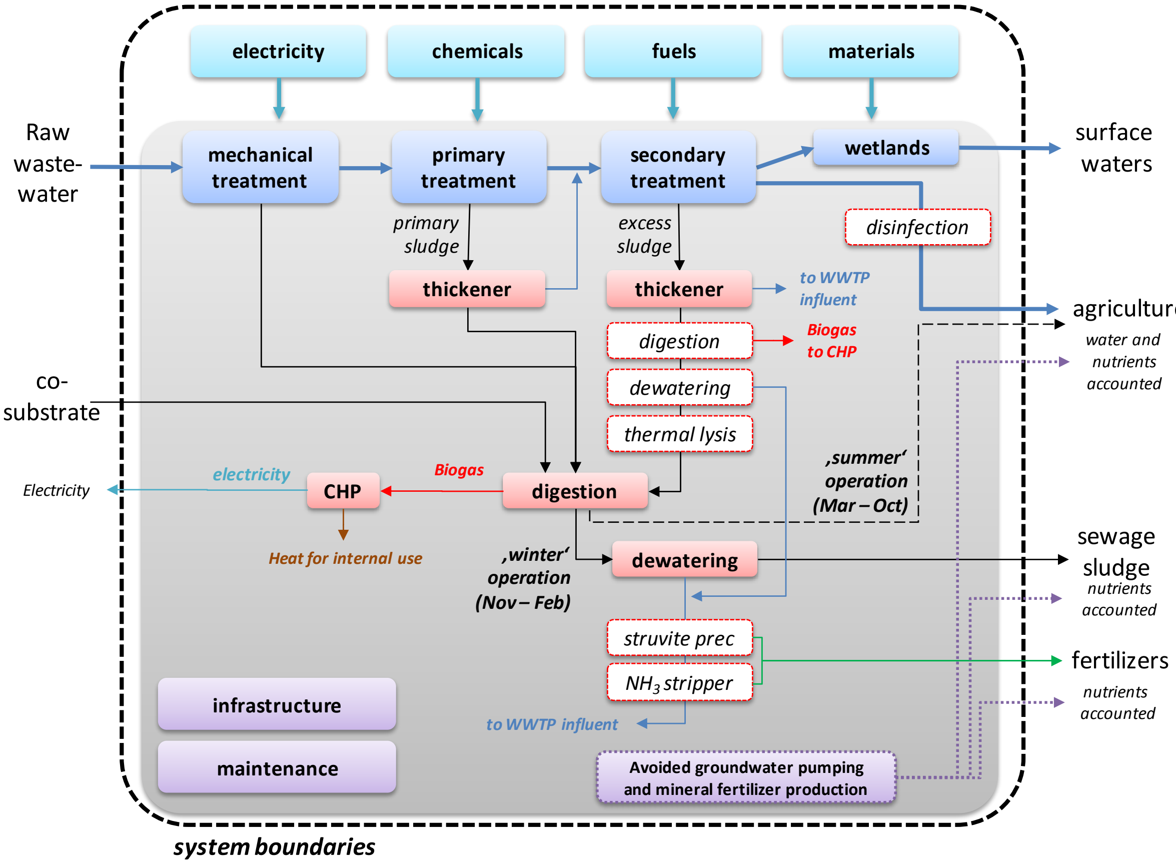


Figure 3-2: System boundaries and scope for LCA study Braunschweig

Allocation

Although the system is multi-functional, all efforts (e.g. energy consumption in wastewater treatment) and benefits (e.g. replacement of mineral fertilizer and water delivery) are related to the function of wastewater treatment and its functional unit. Consequently no allocations were required. Water and nutrients delivered to agriculture are accounted with credits using specific factors with regard to the real substitution of groundwater or mineral fertilizer. For water, groundwater credits are accounted with regards to the effective water demand of the plants (120 mm/a or 3.24 Mio m³/a in total) despite the actual application of higher volumes in the current reuse management scheme. For nitrogen, effective amount of substituted mineral fertilizer is heavily influenced by seasonal demand and continuous supply, so that only < 25% of delivered N in water and sludge can be accounted as credits. For phosphorus, 80 and 100% of P load in sludge and water are accounted, respectively. More details of accounted water volumes and nutrients regarding their effectiveness on arable land are described in the DEMOWARE Deliverable D 1.2 [50].

Scenarios

The scenarios have been selected to show environmental benefits and drawbacks of different technologies and options in the water and nutrient reuse scheme in Braunschweig. A comparative overview of all scenarios including scenario description and annual water volumes and nutrient loads (Figure 3-3) is provided below.

0a. **Reference** is a generic reference scenario, including the WWTP and reuse scheme as it is currently operating, with the exception that sludge dewatering is operated year-round (and not only in winter). This scenario was defined as a benchmark for all other scenarios (1a.-3b.), as continuous sludge dewatering will be obligatory for all these options. Hence, the effect of changing to sludge dewatering also in summer would mask effects of optimisation measures if the current situation (0b) would be taken as benchmark.

0b. Status describes the current status of the WWTP and reuse scheme in the reference year 2014. Around 55% of the WWTP effluent (= 10.7 Mm^3/a) is reused for irrigation of the agricultural area, while the remaining 45% (= 9.0 Mm^3/a) are discharged via the wetlands into the Aue-Oker-Channel as receiving water. Within the vegetation period (March to October), digested sewage sludge is directly mixed with the reused water to enrich it with nutrients for plant supply. In winter months (November to April), digested sludge is dewatered and stored for agricultural application in the next vegetation period.

1a. No Reuse is a hypothetical scenario without water reuse, assuming that the WWTP effluent would be completely discharged to the wetlands and into the Aue-Oker-Channel. Here, the agricultural area would be irrigated using natural groundwater instead, accounting for the actual water demand of the plants with 120 mm per year (= 3.24 M m^3 per year in total) [64]. Sewage sludge is digested and dewatered year-round, and then still delivered to agriculture (but outside the AVB irrigation area).

1b. Irrigation on demand is a reuse scenario with optimised water management. Instead of applying excessive amounts of water beyond the actual water demand of the plants (as in current systems 0a. and 0b.), the volume of reused WWTP effluent is regulated to exactly match the annual demand of the plants (= 3.24 M m^3) [64]. The remaining WWTP effluent is discharged to Aue-Oker-Channel via wetlands, which increases water and nutrient loads to surface waters.

2a. UV-disinfection includes the operation of a UV-disinfection step for the reclaimed water volumes delivered to agriculture. This option for future extension of the scheme will reduce microbial contamination of the WWTP effluent and reduce potential health risks for operators of the water distribution systems (cf. chapter 3.3). The UV system has been tested in the course of DEMOWARE [65]. The capacity of the UV-plant is estimated for the dry weather peak flow of the WWTP.

2b. PFA-disinfection includes the future operation of a chemical disinfection step using performic acid (PFA) for the reclaimed water. Similar to the UV scheme (2a), the PFA system has been tested in pilot scale within DEMOWARE [65]. The capacity of the PFA-installation is also estimated for the dry weather peak flow of the WWTP.

3a. DLD-NR w HR is a future scenario for improved energy recovery and nutrient management. It includes the operation of a two-step digestion with intermediate thermal hydrolysis of the sludge in a DLD (“digestion – lysis – digestion”) setup. Here, excess sludge is digested separately, then dewatered to 15% DS and treated with thermal hydrolysis (160°C) to improve its biodegradability before a second digestion step together with primary sludge. Thermal hydrolysis is equipped with a heat recovery (HR) step to recycle heat of treated sludge back to the input of the process, minimizing the demand for external heat input from CHP off-gas. Finally, sludge is dewatered and valorized in agriculture (as in 0a). Liquor of both dewatering steps is highly loaded with nutrients, enabling the efficient operation of nutrient recovery steps in sidestream treatment. Nutrient recovery (NR) is realized via struvite precipitation and harvesting as crystals (N and P recovery) and ammonia stripping with air and harvesting as diammoniumsulfate (N recovery).

3b. DLD-NR w/o HR is similar to scenario 3a with DLD and nutrient recovery, but without optimized heat management. As heat exchangers can be costly, heat recovery (HR) after thermal hydrolysis may not be economically feasible in full-scale. This scenario requires higher input of external heat into the process, which results in the need of natural gas during times of low excess heat availability from the CHP (i.e. in winter).

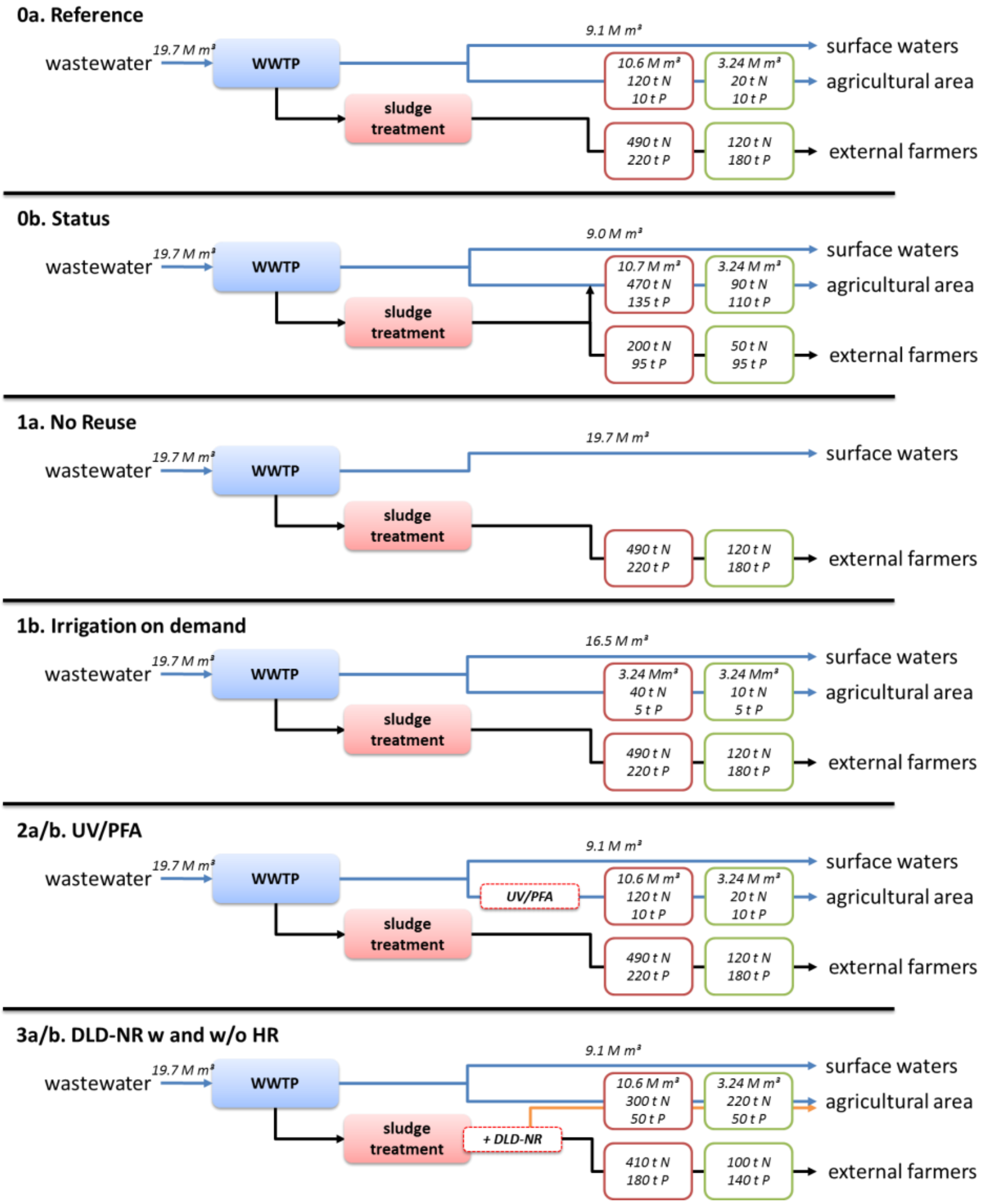


Figure 3-3: Comparative overview of the LCA scenarios for Braunschweig and annual water volumes
M m³ = million m³; nutrient loads: red squares = recycled load, green squares = accounted/effective load

Data quality and limitations of this study

Major input parameters for the LCA inventory are discussed below regarding data quality and uncertainties to clearly point out inherent limitations of this LCA. An overview of data sources and data quality is provided in Table 3-1.

- **Water quality and quantities**: Basic input data on water quality and quantities in the Braunschweig system was provided by the WWTP operator SEBS in 2010 for the CoDiGreen project [66]. This basic dataset was partially updated to reflect the conditions in 2014, with lower influent volumes compared to 2010 due to less rain in 2014 [67, 68]. Assuming constant influent loads for standard parameters and heavy metals, the calculated influent concentrations slightly increased for the 2014 status. Effluent concentrations are kept constant and are based on the inventory of 2010. Since this data is quality controlled and delivered by the WWTP operator, the data quality is assumed to be very good.
- **Quality and mass/volume of sludge and sludge liquor**: Total annual amount of dry substance in different sludge types (primary and excess raw sludge, thickened sludge, digested sludge and dewatered sludge) has been provided by SEBS for 2010 [66] and has been adjusted for 2014 in consultation with the operator [68]. Volume of sludge has been calculated based on the dry substance (DS) content (% DS) for 2014. Volumes of sludge liquor can be calculated from sludge quality before and after dewatering (% DS). Quality data for sludge in terms of standard parameters and metals are measured by the operator. Hence, sludge quality and quantity data are assumed to be good to very good. For the DLD scenarios 3a and 3b, significant changes in the sludge line result in significant changes in sludge quality regarding DS, resulting volumes and distribution of nutrient concentrations between solid and liquid phase. Data for the LCA has been based on lab-tests [69], different digestion modes [68] and detailed calculations by consulting engineers [70], but these estimates only represent expected performance of the full-scale system. Without full-scale data, the reliability of these estimates is limited, meaning that the data quality for these scenarios is only medium.
- **Energy, chemicals and material consumption:** Data on annual energy and chemical consumption for the WWTP operation has been provided by the operator based on experiences for several years of operation [66, 68] based on total and individual demand for different process steps and aggregates. Electricity demand for the activated sludge process is calculated with a dynamic model based on oxygen demand to reflect changes in pollutant load (e.g. via changes in return load) in the total energy demand. In addition to the dynamic part, a constant fraction of energy demand has been considered for the entire WWTP to represent those demands that are independent of pollutant or volume loads. For WWTP effluent, different modes of water transport and discharge into wetlands/Aue-Oker-Channel and agriculture are possible, and this LCA is based on a detailed model of these water flows and respective pumping needs. The data quality of energy and chemical consumption is very good for the existing system and related scenarios 0a, 0b, 1a and 1b. For the disinfection schemes 2a and 2b, pilot-scale trials have been conducted within DEMOWARE, and a detailed up-scaling of costs and related energy efforts has been reported [65]. Hence, data quality for the disinfection scenarios can be assumed as good.

  Estimates for energy and chemicals consumption in scenarios 3a and 3b with energetic optimization of sludge treatment and nutrient recovery are based on different references. Input data for energy consumption and biogas production has been assembled based on the results of lab-tests [69] and calculations by consulting engineers [70] and aligned with expectations of the operator [68]. Taking conservative estimates, positive effects of the new configuration in terms of energy balance reflect the lower range of expected benefits in this study. The data quality regarding electricity and heat consumption of the entire sludge system (DLD including NR) can be assumed as good. For dewatering, new aggregates for excess sludge dewatering (screw extruder) are expected to lower polymer demand and improve DS content compared to existing centrifuges, which have a high polymer demand. Effects of DLD on the final dewatering are difficult to predict and are now based on estimates with medium data quality.

Energy and chemical consumption for the struvite reactor in sidestream is adopted from a previous study [71], where the data has been calculated based on full-scale references of a supplier, leading to a very good data quality. An important limitation of this LCA is related to the chemical demand of the NH3 stripper, namely the required amount of caustic soda for pH adjustment. Caustic soda demand is depending on liquor quality, buffer capacity, temperature of the stripping process, and targeted process efficiency. Chemical demand of NH3 stripping is now based on literature data as best estimate [72], but full-scale implementation may reveal another process optimum and hence increasing or decreasing demand of caustic soda. In constrast, consumption of electricity and sulfuric acid can be predicted with reasonale accuracy.

Infrastructure estimates for the existing WWTP, the water distribution system and the new aggregates (e.g. disinfection units, DLD, struvite reactor and stripping tower) are based on previous studies [73] and cross-checked with operators [74], providing a medium data quality.

Table 3-1: Overview on data quality of input data for LCA Braunschweig

Parameter/Process (scenario)	Data source	Data quality
Existing WWTP with irrigation system (0a, 0b, 1a, 1b)		
Water quality and quantity	[66-68], local operator	very good
Sludge amount (dry substance)	[66, 68], local operator	very good
Sludge quality and volumes (other parameters)	Calculated according to [66, 68]	good
Sludge liquor (volume, COD/N/P loads)	Calculated according to [66, 68]	good
Energy consumption (WWTP and distribution system)	[66, 68], local operator	very good
Chemical consumption (Coagulant, Polymer)	[66, 68], local operator	very good
Infrastructure WWTP	[73], estimations	medium
Infrastructure water distribution system	[74], local operators	good
Disinfection schemes (2a, 2b)		
Energy and chemicals demand	[65], DEMOWARE pilot	good
Infrastructure for disinfection units	[49], estimations	medium
DLD and nutrient recovery schemes (3a, 3b)		
Sludge and sludge liquor quality	[68-70], lab tests, local operators, consulting engineers, estimations	medium
Energy balance DLD (consumption and gas production)	[68-70], lab tests, local operators, consulting engineers, estimations	good
Polymer consumption and performance of dewatering	[66, 68], local operators, estimations	medium
Energy and chemicals for struvite reactor	[71], technology provider	good
Energy and chemical for NH3 stripping (except caustic soda)	[72], estimations	good
Caustic soda demand for NH3 stripping	[72], estimations	low
Infrastructure for DLD and nutrient recovery steps	Estimations	medium

Normalization

Normalisation reveals the contribution of the WWTP/reuse system in relation to the total environmental footprint of each citizen in Europe (EU-27). Principles for normalization and normalization factors are shown in Annex 9.1.1.

3.2.2 Inventory (Input data)

Primary data

Inventory data for the LCA study was provided by the local operator SE|BS [68] and complemented with estimates of KWB based on previous studies [66]. A detailed inventory on electricity, chemical and natural gas consumption for the existing system is provided in Figure 3-4 for the reference scenario 0a. Table 3-2 and Table 3-3summarize electricity and chemicals demand for all scenarios. Material demand for infrastructure is shown in detail in Annex 9.2.1. The following part summarizes most important changes in the inventory for the different scenarios (Table 3-2, Table 3-3).

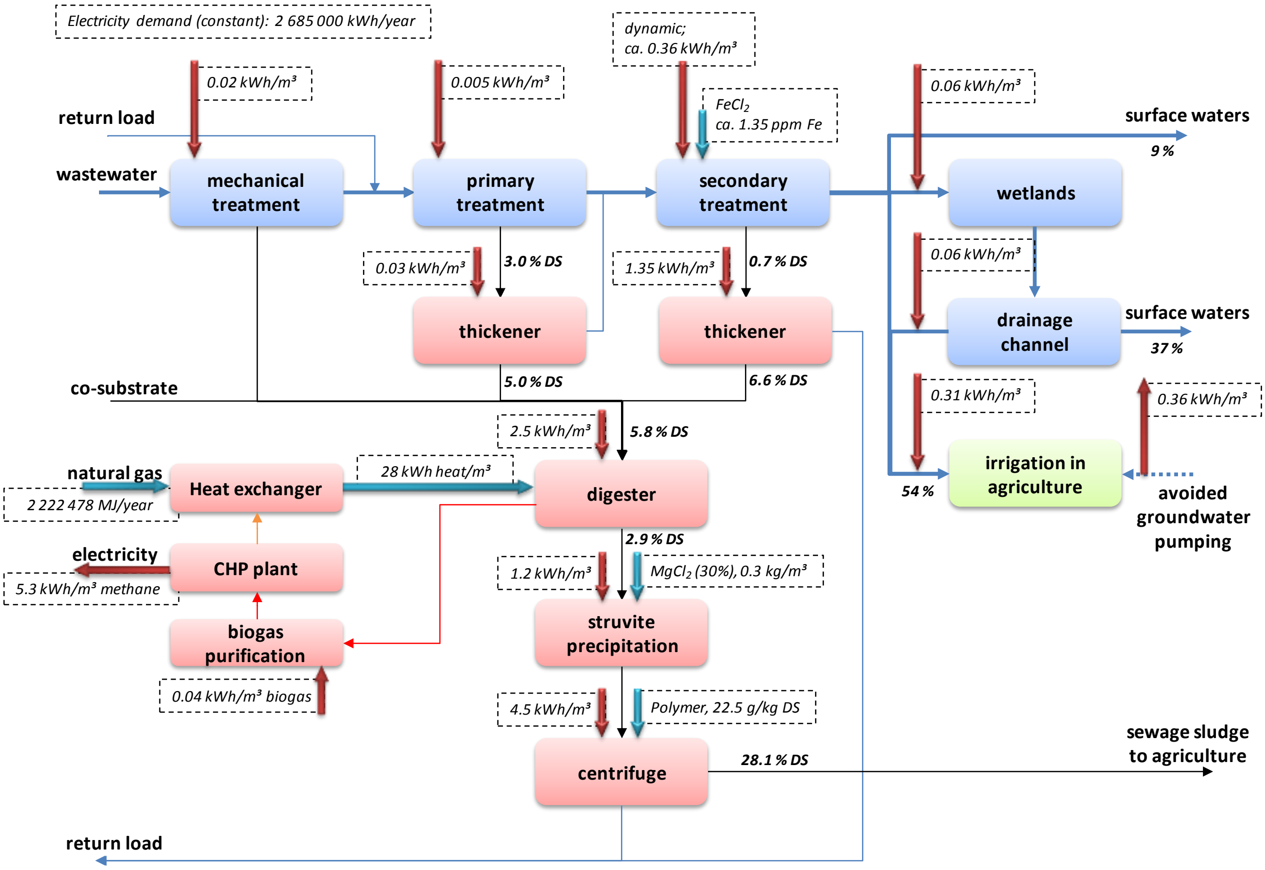


Figure 3-4: Detailed inventory of electricity, chemical and natural gas consumption in the reference scenario (0a)
red arrows: electricity per m³ water, cyan arrows: chemicals/ natural gas per m³ water if not specified

In the current operation ('status'), 55 % of the annual digested sewage sludge is mixed with irrigation water and recycled to agriculture within the vegetation period. As this sludge has not to be dewatered energy and polymer for dewatering can be saved in summer, making it economically attractive. Return load from sludge dewatering is only recycled in winter, reducing the total load to the WWTP and consequently annual electricity and chemical demand compared to continous dewatering ('reference'). Reduced return load in COD also leads to less sludge production and slightly lower biogas yield.

Table 3-2: Inventory data for energy demand as electricity and natural gas (summarized in categories) related to water volumes for LCA Braunschweig.

Volumes per categories and scenario defined as in Figure 3-3. Primary data provided by SE|BS and adjusted as discussed in chapter 3.2.1. Light colors represent no changes to the reference scenario.

	Unit	0a. reference	0b. status	1a. no reuse	1b. irrigation on demand	2a. UV-disinfection	2b. PFA-disinfection	3a. DLD-NR w HR	3b. DLD-NR w/o HR
Total electricity demand	**MWh/a**	**16 803**	**16 157**	**13 434**	**14 460**	**17 176**	**16 835**	**16 742**	**16 770**
primary and secondary treatment	kWh/m³ wastewater	0.522	0.518	0.522	0.522	0.522	0.522	0.520	0.520
sludge treatment (incl. dewatering)	kWh/m³ wastewater	0.134	0.104	0.134	0.134	0.134	0.134	0.133	0.134
water distribution (irrigation and wetlands)	kWh/m³ wastewater	0.196	0.197	0.025	0.077	0.196	0.196	0.196	0.196
disinfection (UV/ PFA)	kWh/m³ effluent	-	-	-	-	0.035	0.003	-	-
Total electricity production	**MWh/a**	**-9 913**	**-9 856**	**-8 747**	**-9 913**	**-9 913**	**-9 913**	**-11 396**	**-11 401**
Electricity from CHP (biogas)	kWh/m³ wastewater	-0.443	-0.441	-0.443	-0.443	-0.443	-0.443	-0.519	-0.519
Credits for avoided groundwater pumping	kWh/m³ ground-water	-0.360	-0.360	-	-0.360	-0.360	-0.360	-0.360	-0.360
Net electricity balance	**MWh/a**	**6 890**	**6 298**	**4 687**	**4 547**	**7 263**	**6 922**	**5 346**	**5 370**
Net electricity demand	kWh/ m³ wastewater	0.349	0.319	0.238	0.230	0.368	0.351	0.271	0.272
Natural gas, external supply	GJ/a	2 222	2 222	2 222	2 222	2 222	2 222	2 107	6 403

In scenarios 'no reuse' and 'irrigation on demand', the reclaimed water volume directed to agriculture changes significantly, directly influencing electricity needs for water distribution. Whereas demand-based irrigation reduces electricity demand for pumping by -60%, stopping reuse altogether will reduce pump energy by 87%, leaving only a minor part for water distribution into wetlands. In the latter case, no credits for avoided groundwater pumping are accounted, as no water is reused in agriculture.

Adding a disinfection step will add electricity for UV/PFA operation, but the increase is only minor compared to the energy demand of the total scheme. UV disinfection consumes 10-fold more electricity than PFA-disinfection, but PFA needs additional chemicals (Table 3-3).

Upgrading the sludge line with DLD and nutrient recovery significantly affects the energy and chemical demand of the system (Table 3-2 and Table 3-3). Net electricity consumption of sludge treatment and dewatering remains constant with DLD and NR, although new aggregates are added to the process.

However, savings in other process steps (e.g. final dewatering) compensate for the new aggregates. Although DLD increases nutrient and COD concentration in the liquor [75], nitrogen recovery from the liquor yields a lower N return load than for the reference scenario, slightly reducing net energy demand of the mainstream process. The model predicts that P return load will be 65% higher with DLD and struvite precipitation (assuming 80% P removal in the MAP process) than before, which slightly increases coagulant dose for P removal in the mainstream. However, higher efficiency of the struvite process may also lead to lower coagulant demand in the mainstream. This has to be checked after full-scale implementation of the DLD-NR configuration.

Table 3-3: Inventory data for materials demand for LCA Braunschweig

Related on different volumes and aggregates, all concentrations per feed volume and chemicals in concentrations with water

Chemical	Unit	0a. reference	0b. status	1a. no reuse	1b. irrigation on demand	2a. UV-disinfection	2b. PFA-disinfection	3a. DLD-NR w HR	3b. DLD-NR w/o HR
Polymer, final dewatering	g/kg DS sludge	22.5							
Polymer, pre-dewatering	g/kg DS sludge	-	-	-	-	-	-	15.0	15.0
$FeCl_2$ (10 %)	g/m³ wastewater	30.6	30.3	30.6	30.6	30.6	30.6	31.1	31.1
Formic acid (100 %)	g/m³ treated wastewater	-	-	-	-	-	10.3	-	-
H_2O_2 (50 %)	g/m³ treated wastewater	-	-	-	-	-	15.9	-	-
$MgCl_2$ (30 %), struvite prec.[5]	kg/m³ sludge water	1.0	1.0	1.0	1.0	1.0	1.0	5.0	5.0
NaOH (40 %), NH_3-stripping	kg/m³ sludge water	-	-	-	-	-	-	8.0	8.0
H_2SO_4 (40 %), NH_3-stripping	kg/kg N in sludge water							9.1	9.1

Most important, biogas production can be increased by 17% with the DLD process, leading to an increase in electricity production at the CHP. This is the major factor in the net electricity balance of the entire system, which can be reduced by 15% with the DLD configuration. Heat demand of the DLD-NR process can be fully met by the excess heat available at the CHP if a heat recovery system is applied for the thermal hydrolysis process, even slightly reducing existing demand for natural gas heating of the reference system. Without heat recovery, significant amounts of natural gas will be required to provide sufficient heat for thermal hydrolysis and NH3 stripper, with an estimate increase of +188% in natural gas demand in the scenario '3b'. A detailed heat balance of these scenarios can be found in a parallel study [75].

[5] Molar dosage current struvite precipitation prior dewatering: 0.4 mol Mg/mol P; Molar dosage future struvite precipitation in sludge liquor: max. 1.5 mol Mg/mol P and additional release of phosphorus in sludge water by DLD

Adding a dewatering step for excess sludge in DLD increases total polymer demand by only 11%, as polymer can be saved in final dewatering due to less DS in digested sludge after DLD. Chemical demand of nutrient recovery steps is significant, requiring MgCl2 for struvite precipitation and caustic soda/sulfuric acid for the NH_3 stripper. Predicting the demand for caustic soda for pH increase is difficult and affected with higher uncertainties (cf. Table 3-1).

Water Inventory

Table 3-4 and Table 3-5 show the water volumes and qualities for the WWTP influent and the related effluents. Differences between influent and effluent are integrated in sludge. Small differences between concentrations are resulting from calculation based on the effects of the return load from sludge dewatering, the variity in charging irrigation fields.

Table 3-4: Water inventory including WWTP influent and discharge to surface waters
Measured data by [66-68] * estimates and model calculations

Parameter	Unit	WWTP influent	Discharge (0a/2a/2b)	Discharge (0b)	Discharge (1a)	Discharge (1b)	Discharge (3a/3b)
Volume	Mm³/a	19.782	*9.076 **	8.990	*19.720 **	*16.480 **	*9.083 **
SS	mg/L	446.2	*11.1 **	11.1	*10.2 **	*10.4 **	*11.1 **
COD	mg/L	777.0	*37.4 **	37.3	*40.0 **	*39.4 **	*37.4 **
TN	mg/L	75.9	*7.4 **	7.2	*9.5 **	*9.0 **	*7.1 **
TP	mg/L	12.1	*0.9 **	0.9	*0.9 **	*0.9 **	*0.9 **
K	mg/L	23.6	*6.8 **	6.7	*14.2 **	*12.6 **	*6.9 **
Cd	µg/L	2.4	*0.3 **	0.3	*0.2 **	*0.3 **	*0.3 **
Cr	µg/L	14.2	*2.2 **	2.2	*2.3 **	*2.3 **	*2.2 **
Cu	µg/L	107.4	*6.5 **	6.5	*7.3 **	*7.1 **	*6.5 **
Hg	µg/L	0.3	*0.2 **	0.2	*0.1 **	*0.1 **	*0.2 **
Ni	µg/L	13.0	*6.3 **	6.3	*5.0 **	*5.3 **	*6.3 **
Pb	µg/L	26.0	*3.7 **	3.7	*3.0 **	*3.2 **	*3.7 **
Zn	µg/L	319.7	*19.5 **	19.6	*18.6 **	*18.8 **	*19.7 **

Table 3-5: Water inventory including WWTP influent and irrigation in agriculture
Measured data by [66-68]; * estimates and model calculations

Parameter	Unit	WWTP influent	Reclaimed water (0a/2a/2b)	Reclaimed water (0b)	Reclaimed water (1a)	Reclaimed water (1b)	Reclaimed water (3a/3b)
Volume	Mm^3/a	19.782	10.644 *	10.644	0 *	3.240 *	10.644 *
SS	mg/L	446.2	9.5 *	9.5	-	9.3 *	9.5 *
COD	mg/L	777.0	42.2 *	42.1	-	42.9 *	42.6 *
TN	mg/L	75.9	11.4 *	10.6	-	11.9 *	10.3 *
TP	mg/L	12.1	1.0 *	1.0	-	1.0 *	1.0 *
K	mg/L	23.6	20.5 *	20.6	-	22.6 *	20.6 *
Cd	µg/L	2.4	0.2 *	0.2	-	0.1 *	0.2 *
Cr	µg/L	14.2	2.4 *	2.4	-	2.4 *	2.5 *
Cu	µg/L	107.4	8.1 *	8.2	-	8.3 *	8.2 *
Hg	µg/L	0.3	0.1 *	3.9	-	0.04 *	0.1 *
Ni	µg/L	13.0	3.8 *	2.5	-	3.4 *	3.9 *
Pb	µg/L	26.0	2.5 *	0.1	-	2.3 *	2.5 *
Zn	µg/L	319.7	17.8 *	18.3	-	17.6 *	18.5 *

Background data

Background data for production of electricity, chemicals, materials, and transport is extracted from ecoinvent database v3.1 [76]. Detailed datasets of background processes are listed in the Annex 9.3.1.

Inventory for Water Impact Index

For water footprinting, the Water Impact Index (WIIX) is calculated according to the methodology described in *D3.1* [11]. Taking all water withdrawals and releases within the system into account, the WIIX also characterizes respective water volumes with scarcity and quality information. For scarcity characterization, the water scarcity index (WSI) according to WULCA AWARE [52] is used for calculation. Monthly or annual WSI for all case studies are shown in the Annex 9.1.1 (Table 9-2), and monthly WSI were chosen for WIIX calculation for the Braunschweig scheme.

Due to the water reuse in Braunschweig, groundwater extraction for agricultural irrigation can be reduced, releaving the local freshwater resources. The avoided water withdrawals as well as the releases in all scenarios are listed in Table 3-6 which is representing volumes of Figure 3-3. For the released water volumes, volume of WWTP effluent discharged into surface waters is fully accounted in the WIIX. In contrast, water going to agricultural irrigation is only accounted with the effective fraction reaching the groundwater table (after subtracting evaporation and plant uptake), assuming 25% of irrigation water replenishing groundwater resources [51]. The avoided withdrawal of groundwater is fully accounted in all scenarios, except for scenario ‘no reuse’ which does not provide irrigation water from the WWTP.

Water quality is characterized by the water quality index (WQI). WQI is calculated based on the intake or effluent water quality parameters [66, 68] compared to a reference benchmark (see Table 9-11 in Annex 9.3.1). In this study, the content of Cu determines the WQI for the WWTP effluent due to the low benchmark for Cu in the EU water framework directive (WFD). This very low benchmark for Cu in the

WFD (1.4 µg/L) may be challenged if compared to other values of environmental quality, as e.g. the minor threshold value for Cu in groundwater by the German *LAWA* is 14 µg/L [77]. In addition, raw sewage contains relatively high loads of Cu and Zn from pipe networks and household inputs, leading to relatively high residual Cu concentrations also in the WWTP effluent despite high specific removal of Cu in the activated sludge process.

Analysing the WQI calculations, the calculated WQI of WWTP effluent is 0.17 if Cu is taken as the benchmark, while the WQI from P is 0.19 (Table 9-11). With respect to the significantly higher concentration of P (factor 100) compared to Cu and its higher significance in surface water quality control, it was finally decided to use a WQI of 0.19 for the WWTP effluent for calculating the WIIX for the Braunschweig system. Hence, all alternatives provide the same quality of reclaimed water or effluent, and no influence on quality is observed for the different scenarios in WIIX.

Table 3-6: Overview on (avoided) withdrawals and releases and water quality indices (WQI) for the different scenarios
GW = avoided groundwater pumping; EF = WWTP effluent; IR = accounted irrigation water (25% of total irrigation volume accounted for groundwater recharge)

Scenario	'reference' and other scenarios (1b, 2a-3b)	'no reuse'	'irrigation on demand'
(avoided) groundwater withdrawals [m³/a]	-3 240 000 (GW)	-	-3 240 000 (GW)
WQI (Withdrawals)	1	-	1
Releases of WWTP effluent [m³/a]	9 659 800 (EF) 2 384 450 (IR)	19 197 600 (EF)	15 957 600 (EF) 810 000 (IR)
WQI (Releases)	0.19	0.19	0.19

3.2.3 Impact Assessment (Results)

Environmental impacts were assessed with a set of 8 impact categories (including WIIX), representing different areas of environmental concern. After an overview of all indicators, selected impact categories are discussed more in detail to reveal individual contributions of different processes and aggregates to the total environmental impact

Total environmental impacts and benefits of all scenarios

The environmental profile of all scenarios for all selected impact categories is shown relatively to the gross impact of the scenario 'reference' (= 100 %) in Figure 3-5 and Figure 3-6. The fossil and nuclear cumulative energy demand (CED) and the global warming potential (GWP) are strongly influenced by the background processes regarding electricity production. CED of the reference system 0a is dominated by the efforts for mainstream WWTP, sludge treatment and water distribution. CED credits are provided by the electricity produced at the CHP and to a lesser extent by substituting mineral fertilizers and groundwater pumping.

GWP correlates strongly with the CED due to emissions from background processes, although direct emissions at the WWTP (e.g. N_2O Emissions from activated sludge tank) also have a relevant contribution here. Credits for avoided N fertilizer production are more relevant in GWP, as the Haber-Bosch process is associated with both high energy demand and high greenhouse gas emissions (e.g. CH_4, N_2O).

In terrestrial acidification potential (TAP), effects of direct emissions and fertilizers are more dominant (Figure 3-6). A major contribution to TAP originates from NH_3 emissions during aeration of raw wastewater in the activated sludge tank. In scenarios 3a/3b with NH_3 stripping, required sulfuric acid also leads to high TAP due to indirect SO_2 emissions during the production of the acid. High demand for sulfuric acid is also relevant for the production of mineral P fertilizer, giving higher credits in TAP for substituting P fertilizer. Similarly, TAP credits for N fertilizer are high due to avoided NH_3 emissions in the Haber-Bosch process.

For the eutrophication indicators, major contributions come from the direct emission of nutrients with the WWTP effluent into the Aue-Oker-Channel. Although P in WWTP effluent accounts for only 9% of total P in raw wastewater, its impact on freshwater eutrophication potential (FEP) is twice as high as from the 91% of P that ends up in sewage sludge and then in agriculture. High P and N loads in sewage sludge applied in agriculture lead to a considerable effect in eutrophication indicators, although this effect is partially offset by avoided application of mineral fertilizers and related effects. P in sewage sludge is accounted to > 80 % for mineral P substitution in all scenarios [50], consequently offsetting >80% of FEP from sewage sludge valorisation in agriculture. For marine eutrophication potential (MEP), N efficiency of sewage sludge application is around 20-25% [50], meaning that less mineral N fertilizer will be avoided which leads to a higher net score in MEP.

Toxicity indicators are mainly influenced by the valorisation of sludge and the associated input of heavy metals into agricultural soil, showing only marginal changes between scenarios. In addition, these indicators are affected with high uncertainties and should be interpreted with care.

In water footprinting, the WIIX shows a negative score (i.e. water savings) for all scenarios, which results from the WWTP effluent or irrigation water which is introduced back into the environment. In detail, the WIIX is mainly influenced by the volume of water released to the Aue-Oker-Channel and by the avoided groundwater use and avoided groundwater release in reuse scenarios. Efficient use of water leads to the lowest WIIX in the scenario ‘irrigation on demand’, as water is reused with high efficiency or discharged into surface water if not needed in agriculture. Background processes have only smaller contributions in the WIIX for all scenarios.

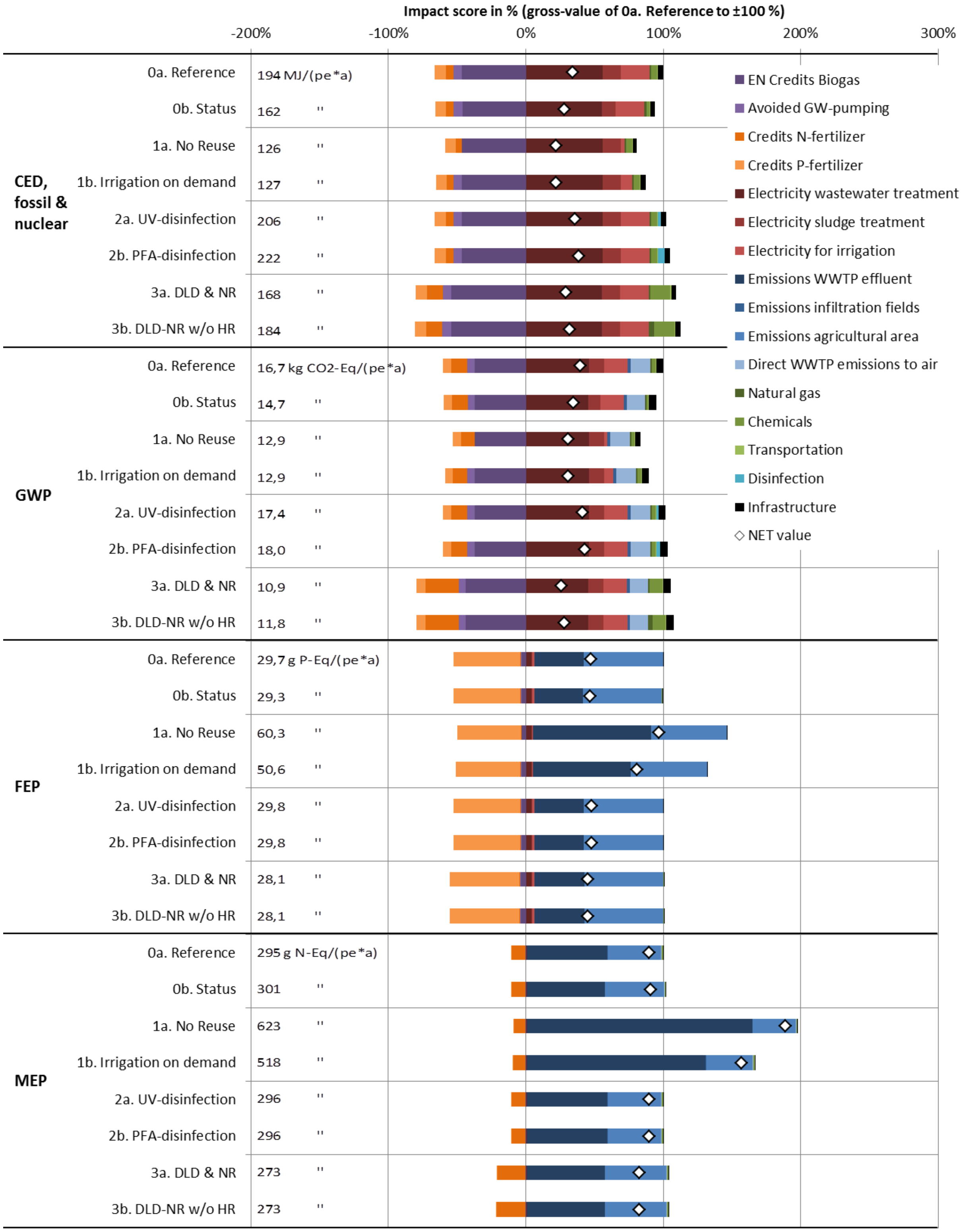


Figure 3-5: Environmental profile for all scenarios of LCA Braunschweig related to gross-value of 'reference' scenario (= 100 %) and total net values per scenario and impact category (Part 1)

CED = cumulative energy demand; GWP = global warming potential; FEP = freshwater eutrophication potential; MEP = marine eutrophication potential

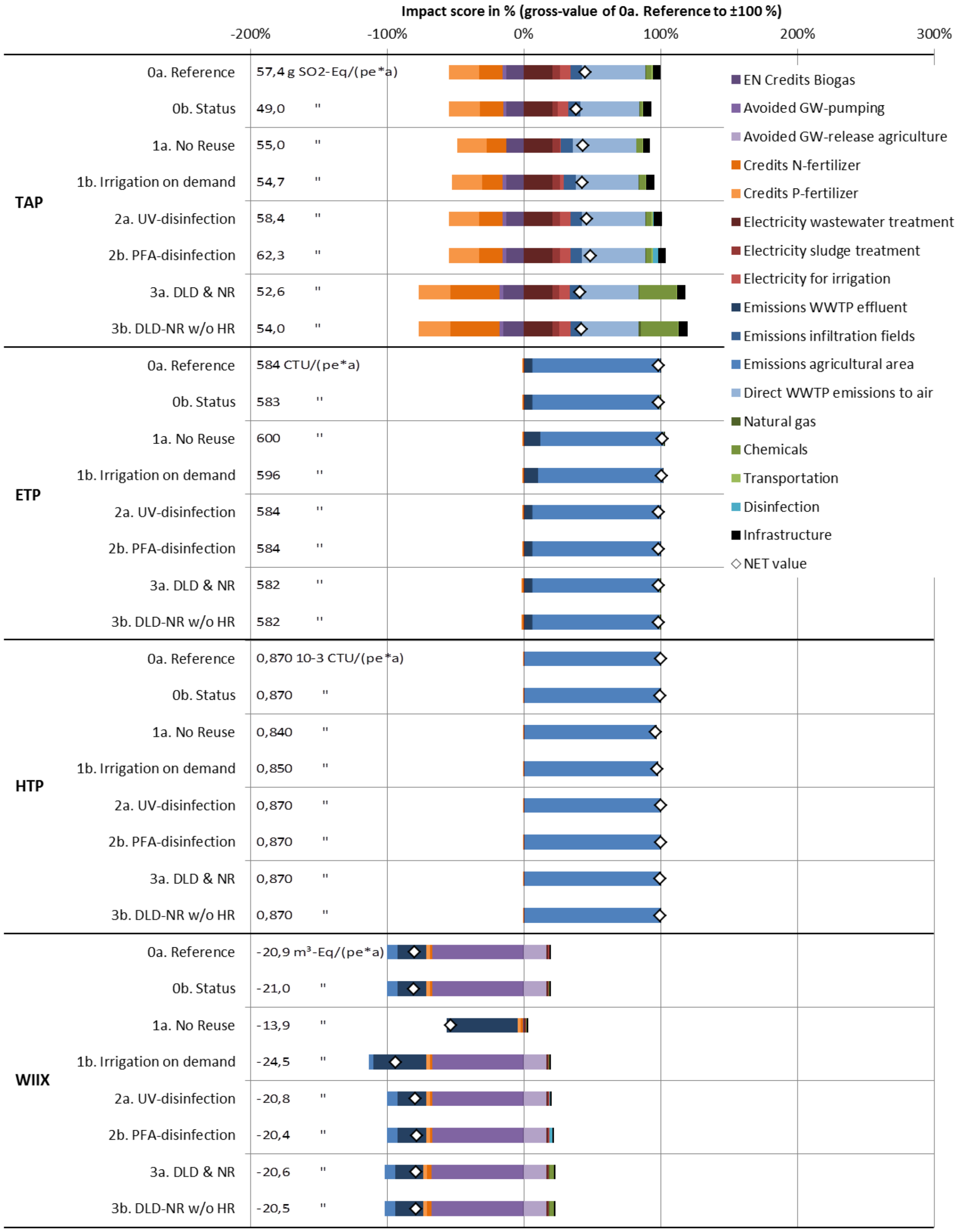


Figure 3-6: Environmental profile for all scenarios of LCA Braunschweig related to gross-value of 'reference' scenario (= 100 %) and total net values per scenario and impact category (Part 2)

TAP = terrestrial acidification potential, ETP = ecotoxicity potential; HTP = human toxicity potential; WIIX = water impact index

Relative changes for selected impact categories

For selected indicators, relative changes of environmental impacts for different scenarios are shown in relation to the 'reference' scenario as a benchmark to illustrate specific effects of system optimisation on the environmental profile. For this perspective, total impacts of 'reference' are deducted from the respective scenario, and the result is divided by the total number of pe served (350'000 pe).

Changes for CED are shown in Figure 3-7, taking the net CED of the 'reference' scenario (194 MJ/(pe*a)) as benchmark. For the current operation ('status'), savings in polymer and electricity for avoided sludge dewatering during the vegetation period lead to a reduction of -32 MJ/(pe*a) or -16% compared to continuous dewatering in the reference scenario.

The highest net savings in CED are indicated for the scenarios 'no reuse' and 'irrigation on demand': both scenarios significantly reduce the electricity required for pumping water to the irrigation system. The net effect is comparable between both options, as both scenarios represent the delivery of an optimum amount of water to the fields, without pumping "useless" water. Some fertilizer credits are lost in both scenarios due to less water reuse, which is offset in the scenario without reuse by saving on the disctribution system infrastructure. In total, both scenarios lead to a signficant reduction of net CED of -68 MJ/(pe*a) or -35 % compared to the 'reference' scenario, showing the energy benefit for an optimum use of water.

For disinfection scenarios, the UV process adds +12 MJ/(pe*a) or 6% to the net CED of the system, whereas PFA will increase CED by +28 MJ/(pe*a) or +14%. Although electricity consumption is significantly lower for the PFA process, the high chemicals demand (especially formic acid) lead to a higher CED in total (Figure 3-8).

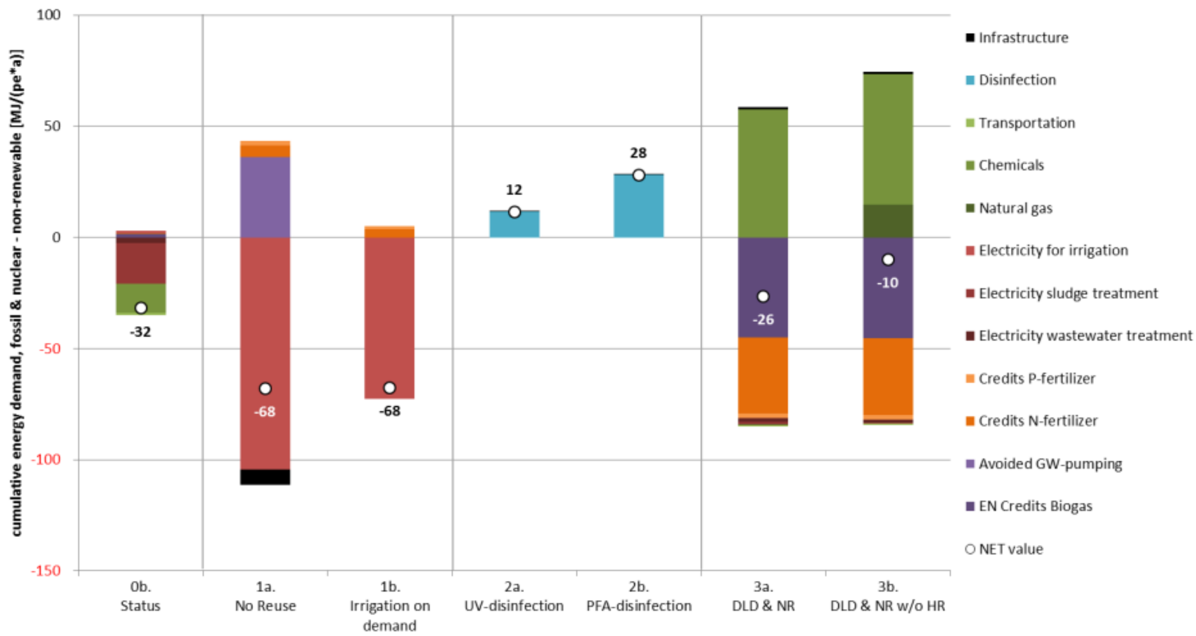
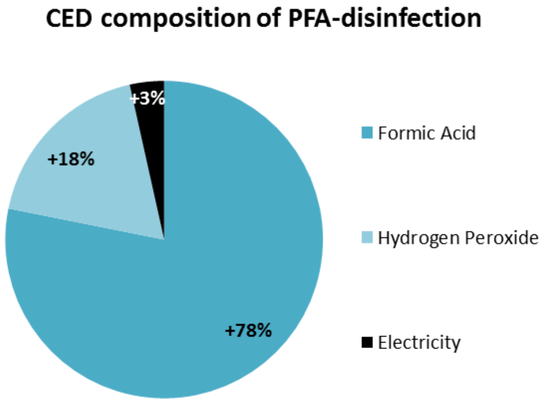


Figure 3-7: Changes in fossil and nuclear cumulative energy demand of the different scenarios in relation to the 'reference'



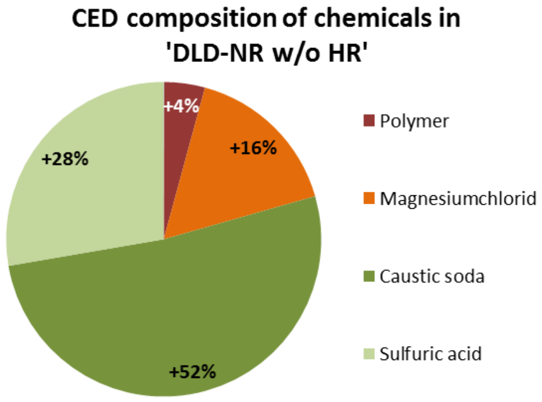


Figure 3-8: Composition of changes in fossil and nuclear cumulative energy consumption for PFA-disinfection and chemicals in the scenario 'DLD-NR w/o HR'

DLD scenarios 3a and 3b also reduce net CED, mainly due to increased biogas production (+17%) with related electricity credits and more efficient recycling of N in sludge. Benefits of this configuration are partially offset by high chemicals demand of NH3 stripping, which originate from caustic soda and sulfuric acid (Figure 3-8). If only nutrient recovery steps in sidestream were analysed (without DLD), energy demand of the system would even increase as fertilizer credits are more than offset by chemicals demand. The latter is valid also for the struvite process, as high Mg dosing leads to high chemical demand for P recovery giving a neutral CED to this process step, which is in contrast to previous studies where struvite recovery led to net energy credits [71].

In total, the introduction of DLD-NR leads to a reduction of -26 MJ/(pe*a) or -13% for the entire system in scenario 3a. If DLD is operated without heat recycling (3b), energy benefits of this configuration will be reduced by -60% due to the high demand of natural gas for heating in winter (see Figure 3-7), leaving only -10 MJ/(pe*a) or -5% of the benefits for this configuration. This underlines the importance of optimised heat management when going for thermal processes in sludge treatment, as energy benefits of increased biogas production may quickly vanish if additional heat has to be produced for the thermal hydrolysis.

In general, GWP shows similar effects as CED for all scenarios (Figure 3-9). Due to the major share of fossil energy in the German electricity mix, GWP correlates with CED regarding the effects on electricity demand and production. In addition, GWP illustrates effects on direct emission profiles in the scenarios, e.g. reduced N_2O emissions in denitrification in those scenarios that lead to a reduction in N return load ('status' and 'DLD-NR'). Credits for mineral N fertilizer substitution are more pronounced in GWP due to direct emission of greenhouse gases (GHG) in the Haber-Bosch process.

Compared to the 'reference' scenario with a net GWP of 16.7 kg CO_2-eq/(pe*a), the current 'status' reduces GHG emissions by -13 % due to less efforts for sludge dewatering. Irrigation on demand decreases GWP by -22 % (1b), which is in the same range if water reuse is abandoned altogether (1a). For disinfection, adding UV and PFA treatment will increase net GWP by +4% and +8%, respectively. Optimised energy and nutrient management with DLD-NR reduces GWP by -35% (-29% without heat recovery in DLD).

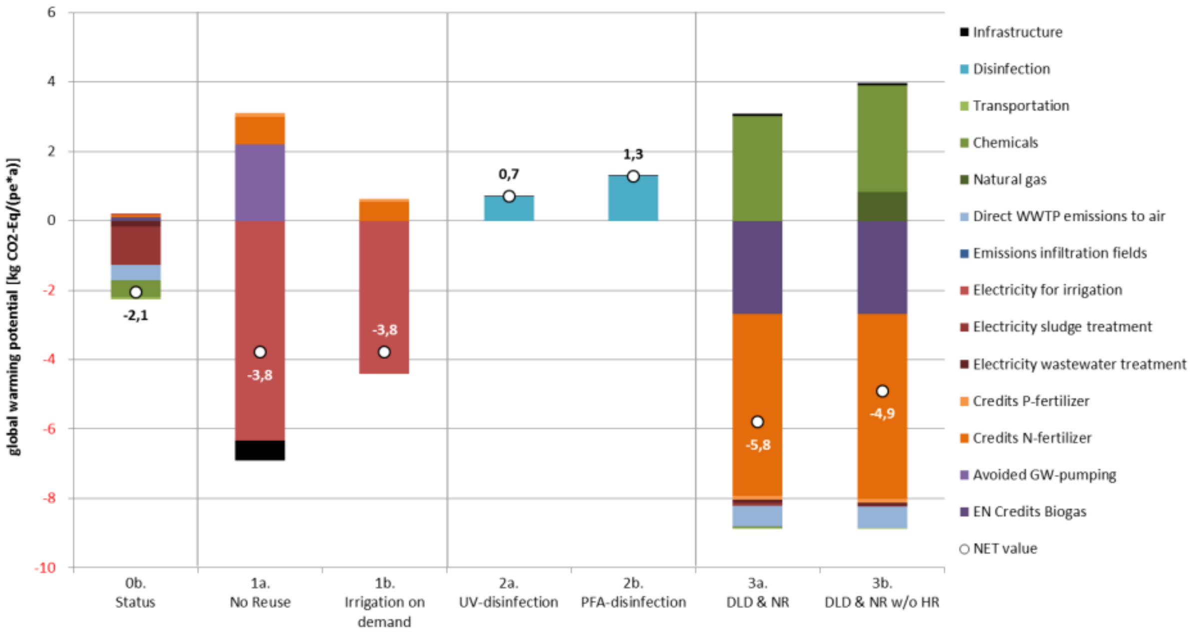


Figure 3-9: Changes in global warming potential of the different scenarios in relation to the 'reference'

As indicated in Figure 3-10, changes in freshwater eutrophication are dominated by P loads in the WWTP effluent discharged into the Aue-Oker-Channel. For scenarios with reduced or no water reuse, discharge into Aue-Oker-Channel increases significantly, leading to a major increase in FEP for scenarios 1a and 1b. Whereas 'no reuse' increases FEP by +30.6 g P/(pe*a) or +203 %, 'irrigation on demand' still leads to +20.9 g P/(pe*a) or +170 % in FEP.

The same effect is visible for the nitrogen-based marine eutrophication, where increased discharge of N into surface water leads to higher MEP for the same scenarios (Figure 3-11): again, 'no reuse' increases net MEP by +328 g N/(pe*a) or + 211 %, whereas 'irrigation on demand' has a net MEP of +223 g N/(pe*a) or +176 % compared to the reference.

Both eutrophication indicators illustrate a major benefit of water reuse: residual nutrient loads in WWTP effluent are diverted away from surface waters into agriculture, which relieves the surface water from nutrient input and reduces the risk of eutrophication. However, nutrient input into agriculture should fit the demand of the plants to prevent inefficient use of the nutrients and potential transfer via groundwater into surface water again. The best scenarios in both FEP and MEP are therefore the DLD-NR scenarios with dedicated recovery of N/P fertilizer products which can be applied precisely at the time of demand to reach maximum efficiency in nutrient valorisation.

For water footprinting, WIIX already showed the benefits of water reuse in reducing water stress to local groundwater resources, with a negative WIIX of -21.0 m³-eq/(pe*a) for the current system. If water reuse is stopped ('no reuse'), WIIX increases considerably due to required groundwater withdrawal (**and release**) for agricultural irrigation (Figure 3-12). In contrast, current WIIX can be further reduced by demand-oriented irrigation, as less water is "lost" by over-irrigation and evaporation. Again, indirect effects of water used for background processes are marginal compared to the direct effects of water reuse on WIIX. Overall, water reuse leads to a significant decrease of the water footprint of agriculture, which is not offset by marginal effects of water treatment and distribution that may be required to provide reused water to the farmers.

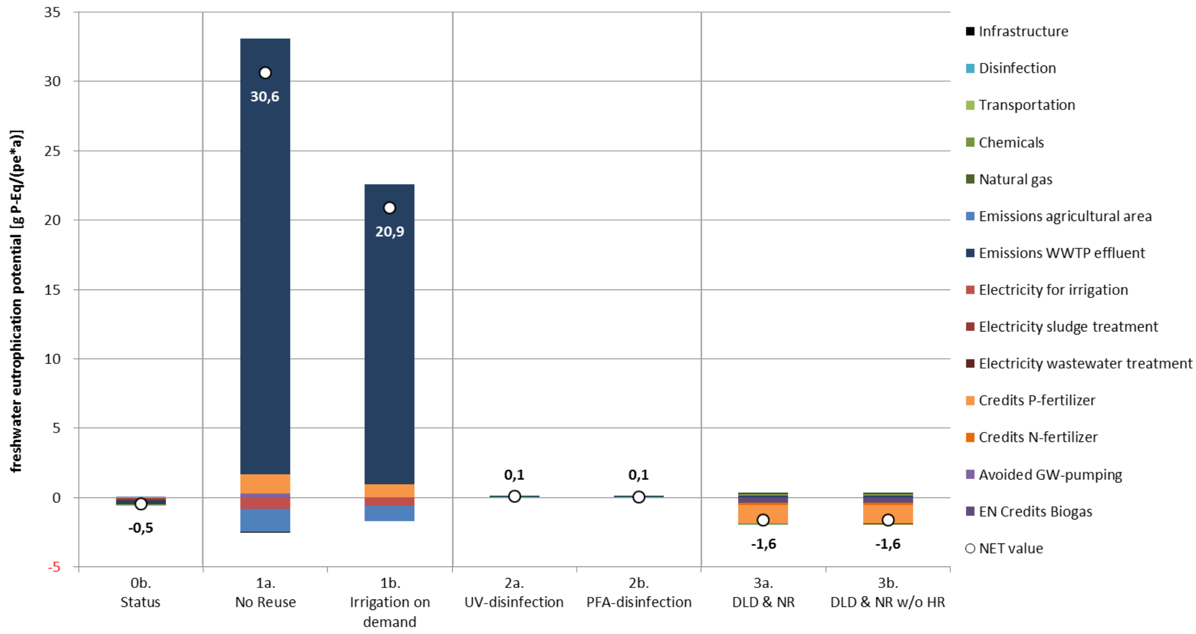


Figure 3-10: Changes in freshwater eutrophication potential of the different scenarios compared to the 'reference'

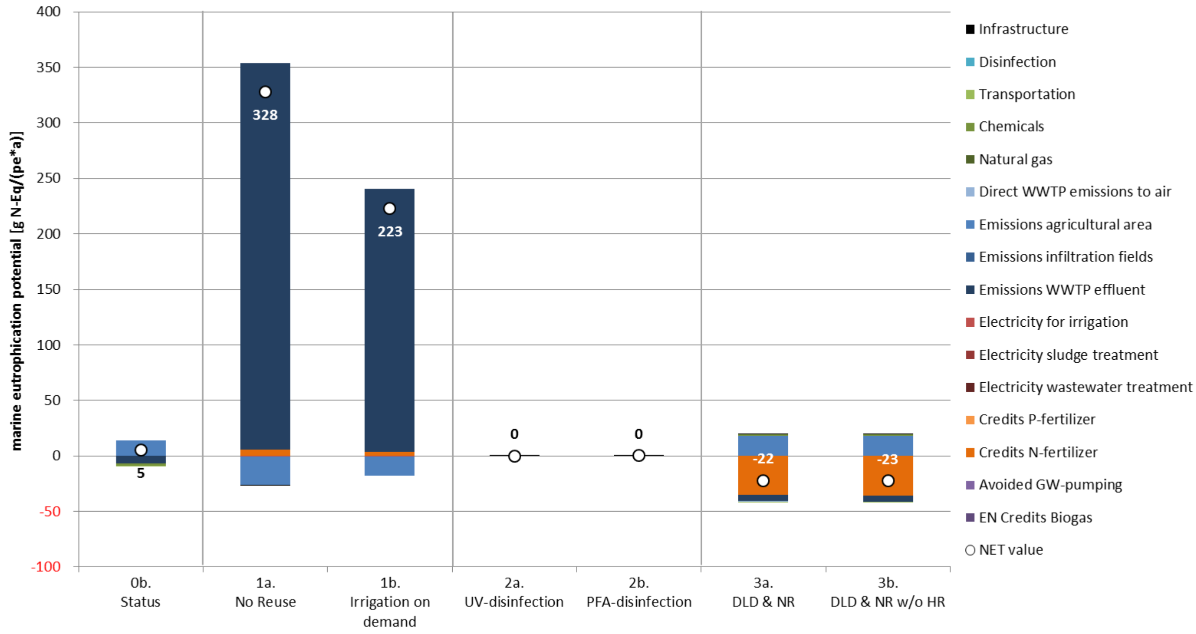


Figure 3-11: Changes in marine eutrophication potential of the different scenarios compared to the 'reference'

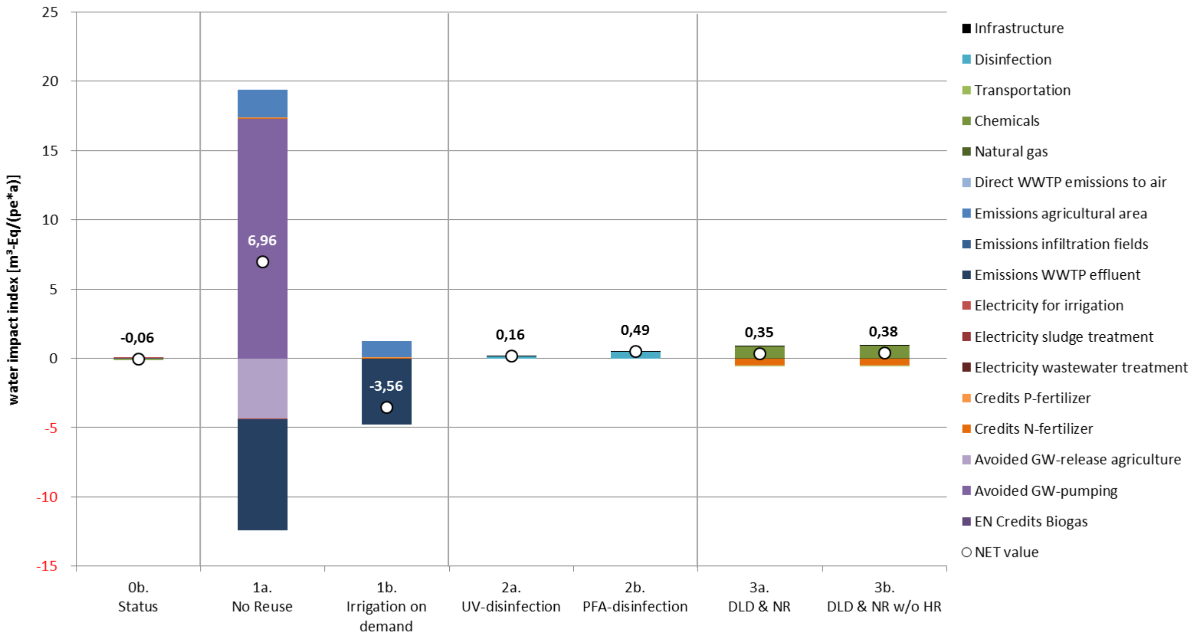


Figure 3-12: Changes in water impact index of the different scenarios compared to the 'reference'

Normalization

The net score for each impact category per pe (Figure 3-5 and Figure 3-6) is related to the normalization data (Table 9-1) per EU-27 citizen, showing the relative contribution of the existing system to the total environmental impacts per person (Figure 3-13).

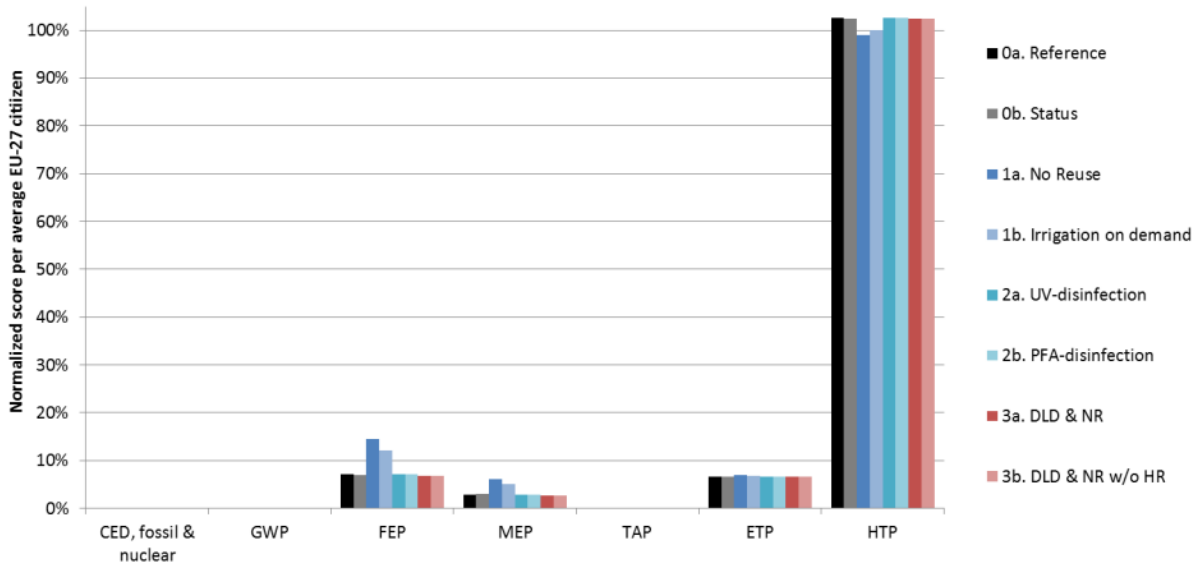


Figure 3-13: Normalized scores for all impact categories per average EU-27 citizen

Energy-related indicators CED, GWP and TAP contribute approximately 0.1-0.2 % to the gross CED, GWP or TAP per citizen in the EU-27, showing that the system of wastewater treatment and reuse has only a marginal contribution to the total environmental impact here. The normalized score of FEP is between 6 and 15% related to the EU-27 average, while MEP contributes 2-7 %. Hence, water quality aspects have higher normalized scores than energy-related aspects, which is no surprise as the system of WWTP is dedicated to the mitigation of these effects.

Normalized scores for the toxicity indicators (especially for human toxicity) are higher (approx. 100 % in score), but are affected with very high uncertainty and have to be interpreted with care. These impact categories are mainly influenced by the effects of heavy metal input into agricultural soil due to sewage sludge valorisation. The strength and weakness of these impact categories are discussed in detail in Chapter 8.4 for all case studies, since similar results have been achieved in those cases when sludge is recycled.

3.2.4 Interpretation and Discussion

Summary and Interpretation of results

Table 3-7 gives a summary on the net environmental efforts and benefits of the scenarios for all impact categories in relation to the net impacts of the ‘reference’ scenario set at 100%. It can serve as an overview of the results of this LCA and illustrates major effects of all scenarios on the environmental profile of the Braunschweig reuse system.

Table 3-7: Summary of net environmental impacts and benefits of the scenarios for all impact categories in the LCA Braunschweig, related to the ‘reference’ scenario (= 100%)

Scenario	0a. reference	0b. status	1a. no reuse	1b. irrigation on demand	2a. UV-disinfection	2b. PFA-disinfection	3a. DLD-NR w HR	3b. DLD-NR w/o HR
CED	194 MJ/(pe*a)	- 16 %	- 35 %	- 35 %	+ 6 %	+ 14 %	- 14 %	- 5%
GWP	16.7 kg CO_2-Eq/(pe*a)	- 12 %	- 23 %	- 23 %	+ 4 %	+ 8 %	- 35 %	- 29 %
FEP	29.7 g P-Eq/(pe*a)	- 2 %	+ 103 %	+ 70 %	± 0 %	± 0 %	- 5 %	- 5 %
MEP	295 g N-Eq/(pe*a)	- 2 %	+ 111 %	+ 75 %	± 0 %	± 0 %	- 7 %	- 8 %
TAP	57.4 g SO_2-Eq/(pe*a)	- 15 %	- 4 %	- 5 %	+ 2 %	+ 9 %	- 8 %	- 6 %
ETP	584 CTUe/(pe*a)	± 0 %	+ 3 %	+ 2 %	± 0 %	± 0 %	± 0 %	± 0 %
HTP	0,87 $\cdot 10^{-3}$ CTUh/(pe*a)	± 0 %	- 3 %	- 2 %	± 0 %	± 0 %	± 0 %	± 0 %
WIIX	- 6.79 m^3-Eq/(pe*a)	± 0 %	+ 33%	- 17 %	+ 1 %	+ 2 %	+ 2 %	+ 2 %

The following aspects can be summarized:

- The current ‘status’ with seasonal dewatering is beneficial from an energetic point of view, compared to the continuous dewatering in the ‘reference’. From this LCA, no environmental benefits can be identified which lead to a better environmental profile when stopping the dewatering in summer.
- Stopping the approach of water reuse altogether (‘no reuse’) decreases energy demand and reduces associated GHG emissions as less pumping is required, but would lead to a major increase in nutrient loads to surface water (Aue-Oker-Channel). In addition, local water resources would be more stressed without reuse as groundwater has to be exploited for supplying agriculture, increasing the water footprint significantly.
- Demand-oriented irrigation will also reduce energy and GHG footprints, but similarly lead to higher N/P loads into the Aue-Oker-Channel. However, water footprint can still be decreased with this “optimised water management”, as no water is wasted “useless” on the agricultural

fields. The latter point needs to be discussed, as excess irrigation could also be seen as groundwater recharge strategy, although not all of the excess water will eventually reach the aquifer.

- Adding a disinfection scheme for the reused water to mitigate health risks for operators will increase energy demand and related impacts of the scheme by max. 10%. UV disinfection needs less energy and associated impacts than PFA, because PFA requires a high amount of chemicals. Benefits of disinfection are not assessed in this LCA, but can be analysed with risk assessment (Chapter 3.3).
- Upgrading the sludge treatment with DLD and nutrient recovery reduces energy demand (-14%) and associated GHG emissions (-35%), although nutrient recovery itself is near energy-neutral due to high chemical demand. Energy benefits of DLD would be significantly reduced if no heat recovery was implemented for the thermal hydrolysis, illustrating the need for optimised heat management in thermal processes. Nitrogen recovery in NH3 stripping increases N utilisation efficiency in the system significantly, resulting in major benefits in GHG emission reduction due to the substitution of the Haber-Bosch process and in less eutrophication potential due to targeted application of nutrients.

Combinations of different scenarios towards a future 'Braunschweig' model fostering risk migration and decoupling of water and nutrient management

As mentioned in the introduction (chapter 3.2.1), this LCA focussed on three different goals and the respective technical changes to reach these goals. However, a combination of these single targets is of course possible, leading to an integrated "future Braunschweig system" which addresses the future challenges of the water reuse system. In particular, two major points for optimisation are identified here:

- Fostering risk mitigation
- Fostering decoupling of water and nutrient management and improvement of energy efficiency

Both strategies can be realized by combining different scenarios of this LCA. To illustrate the potential impact of this integrated strategies, CED and GWP of 'combined scenarios' are calculated below (Figure 3-14 and Figure 3-15).

Fostering risk mitigation

Adding a disinfection step for reclaimed water is only one option to mitigate microbial health risks for operators of the water reuse system. In addition, irrigation on demand can further reduce the potential risk of infection by field workers etc. (see chapter 3.3). This strategy will restrict irrigation to the seasonal water demand of the plants in summer (May to September [50]), reducing the amount of days that field workers are exposed to reclaimed water. In addition, pathogen concentrations in raw wastewater are lower in summer than in winter, further reducing the potential health risks of water reuse (see chapter 3.3). Hence, health risks for the workers and residents are substantially reduced by targeted irrigation-on-demand in summer, a strategy which also saves on disinfection efforts as a lower volume of water has to be treated annually. Combining disinfection and irrigation on demand, CED and GWP of the reuse scheme can be significantly reduced: while energy demand is decreased by 17-20% with this strategy, GWP is reduced by -10%.

Fostering decoupling of water and nutrient management and improvement of energy efficiency

In DEMOWARE D 1.2, options for the decoupling of water and nutrient management were discussed in detail for the Braunschweig scheme [50]. The combination of demand-oriented irrigation and upgraded sludge line with DLD and nutrient recovery leads to a maximum decoupling of water and nutrient management, allowing for an optimised supply of water and nutrients at the time of plant demand. As already mentioned in D 1.2, this setup would have positive side effects on the overall energy consumption [50]. In fact, CED of the current scheme can be reduced by up to 40% with this “decoupling” strategy, providing higher efficiencies in both water and nutrient supply and avoiding the delivery of “useless” water and nutrients in times of low demand. For GWP, the benefits of this combined strategy are even higher, with up to 50% savings in GWP compared to the reference system. This combined strategy illustrates the major potentials which are still to be exploited in the Braunschweig scheme, moving to a more sustainable operation in the future while addressing challenges ahead.

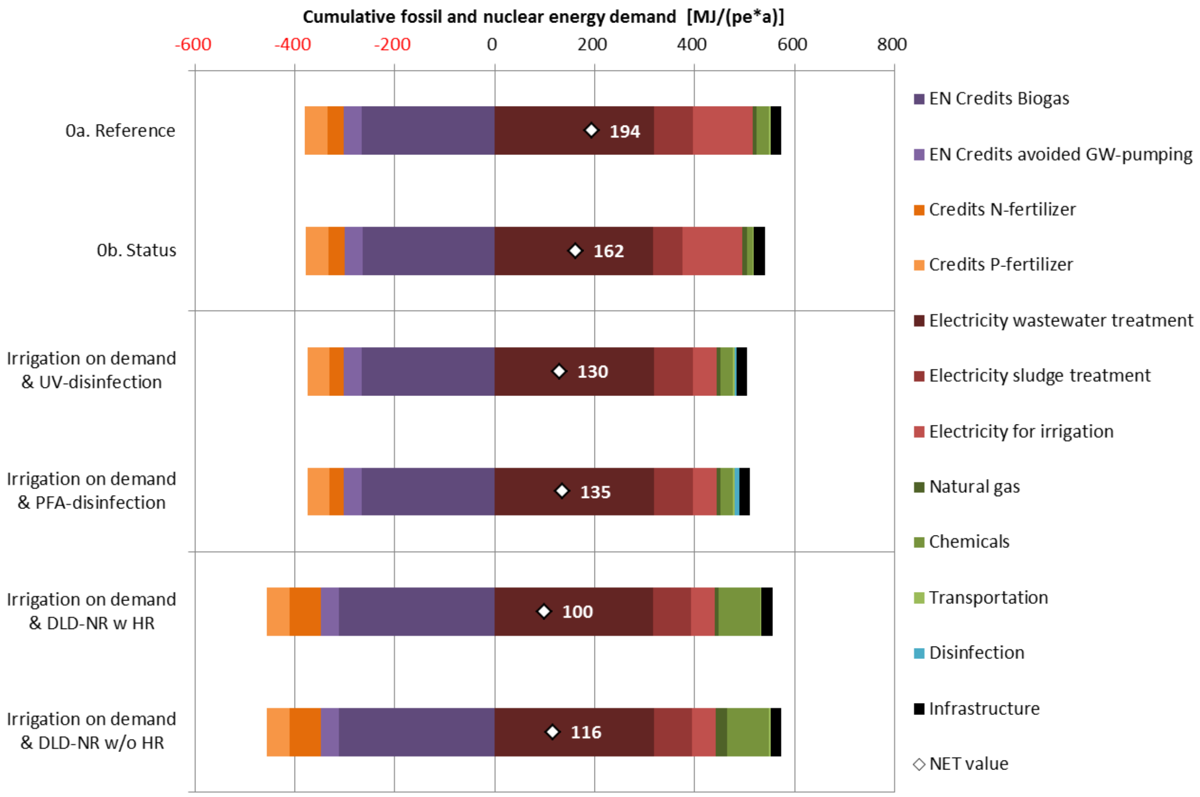


Figure 3-14: Fossil and nuclear cumulative energy demand of the baseline scenarios and possible combinations for a future ‘Braunschweig’ model

In summary, there are different options available to upgrade the current system into a future ‘Braunschweig’ model which addresses the challenges of tomorrow. Focussing on the goals of this LCA, the reduction of health risks as well as the decoupling of water and nutrient management for a demand-oriented supply of water and nutrients is possible. In fact, these goals can be reached while simultaneously reducing energy and GHG footprints of the system, and without compromising the positive effect of water reuse on the water footprint. However, reducing the amount of reused water to an optimum will also lead to a higher hydraulic and nutrient load to the receiving surface water, which has to be carefully addressed in an overall assessment.

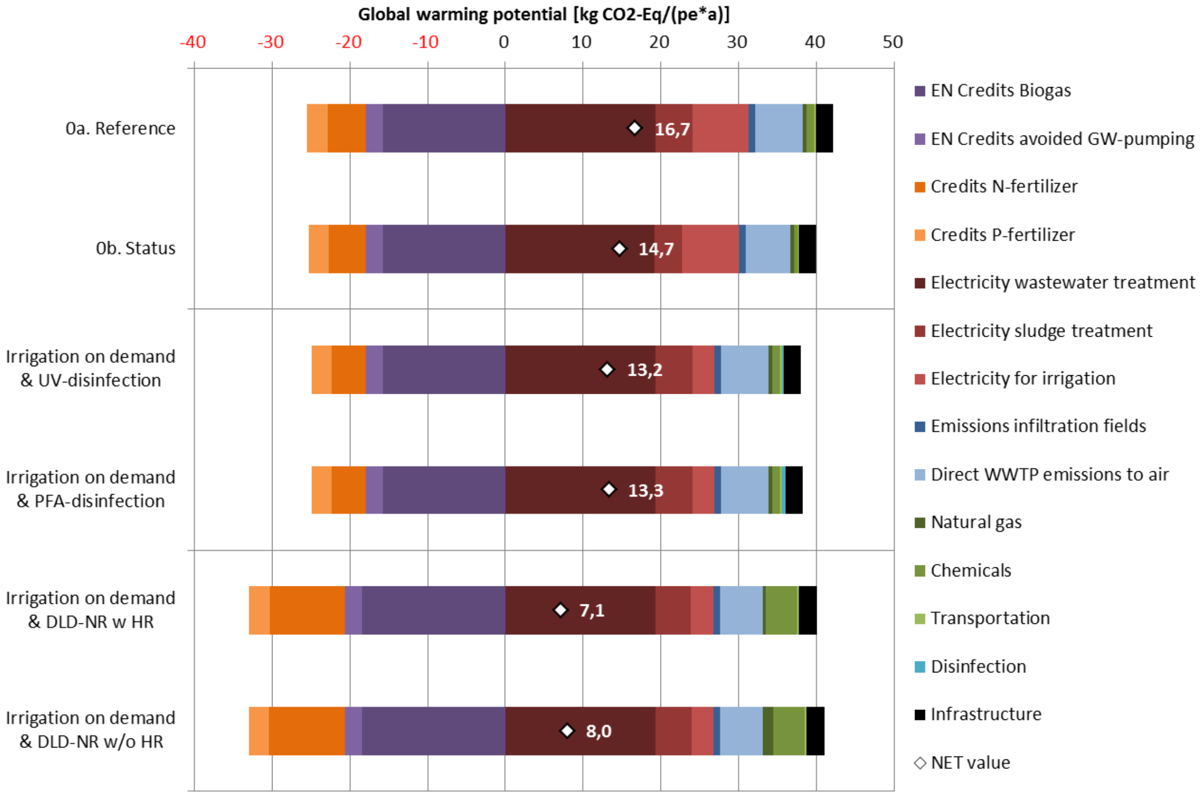


Figure 3-15: Global warming potential of the baseline scenarios and possible combinations for a future 'Braunschweig' model

Naturally, economic and legal conditions also play a major role in the future development of the Braunschweig scheme, and should be analysed in parallel to the environmental impacts. However, this LCA indicates significant potentials for optimisation from an environmental point of view, which can support the constant development of the Braunschweig reuse system towards meeting the challenges of tomorrow.

3.3 Risk Assessment

3.3.1 Goal and Scope

In Braunschweig quantitative microbial risk assessment (QMRA) was conducted in order to quantify the probability that the planned reuse system would be able to meet the WHO health based target (HBT) of 10^{-6} DALYs per person per yea (pppy).

In order to do so, risk assessment was conducted in a two tier process. Within a first quantitative desktop study on microbial risk a thorough literature review was conducted [78]. The main result of the study raised concerns that the risk of the current practice of using secondary effluent for irrigation in agriculture exceeds the WHO guidance value of 10^{-6} additional DALYs per person per year (pppy). Thus, the DEMOWARE project focused on:

1. Verifying and refining the assumptions made during the first risk assessment stage
2. Investigating two different disinfection methods (UV, performic acid) as potential options for risk reduction [78]
3. Investigation of the risk reduction potential by changing the irrigation regime from "excess irrigation" to an "irrigation on demand" system.

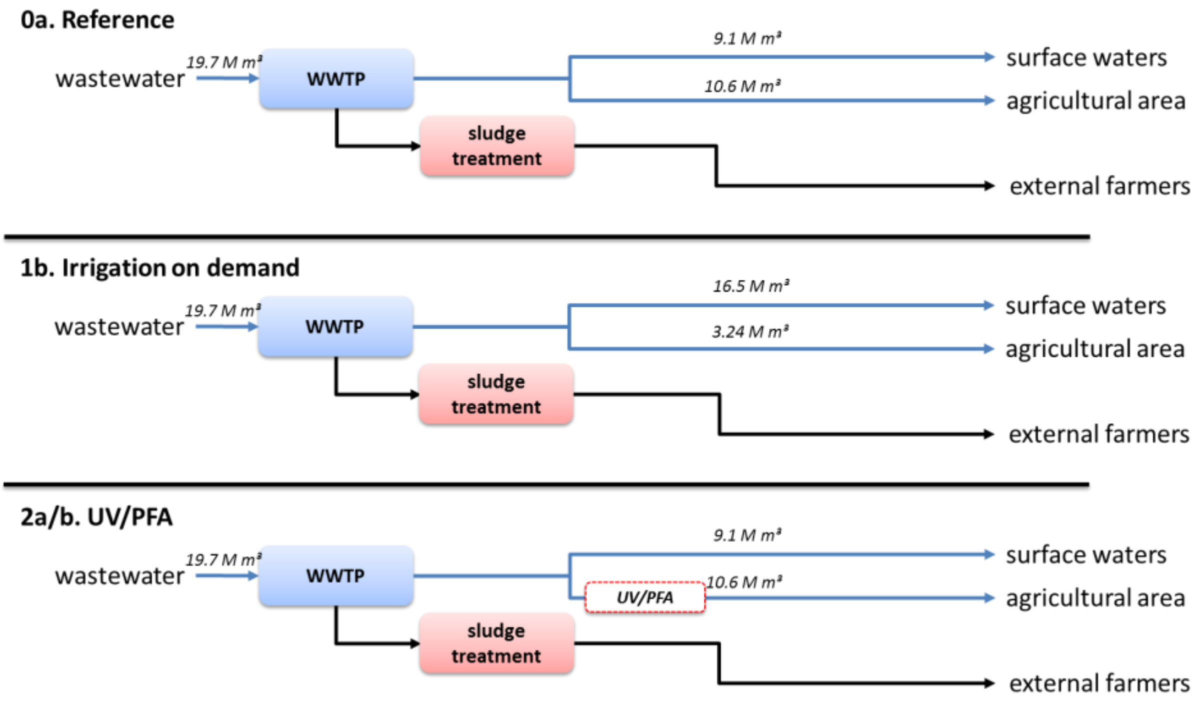


Figure 3-16: Competitive overview of the LCA Scenarios for Braunschweig and annual water volumes (in M m³ = million m³) and nutrient loads

red squares: recycled load, green squares: accounted/effective load

3.3.2 Hazard identification

For risk assessment Rotavirus, Norovirus (GGI and GGII) *Campylobacter* and *Giardia* have been selected. These pathogens represent the most frequently causes of reported gastrointestinal disease in the Federal State of Lower Saxony. Given the information in Miehe et al 2015, neither *Salmonella* nor EHEC was considered as reference pathogens anymore.

3.3.3 Available data and use of information

For the preliminary first risk assessment a literature review was conducted to estimate indicator and pathogen concentration as well as removal efficiency of the wastewater treatment plant. For the first risk calculations triangular distributions were most commonly used for Monte Carlo Simulation and uncertainty analysis.

In DEMOWARE, disinfection trials were conducted during calendar week 42 and 49 in 2014. Monitoring focused on the treatment performance of UV and PFA disinfection against microbial indicator organisms under different configurations Figure 3-19. Additionally to the disinfection trials, samples from the influent of the wastewater treatment plant have been taken. After a first evaluation phase an fluence of 650 J/m^2 and a PFA dose of 2.0 ppm have been chosen to be investigate more closely [65].

In the present study, the information collected from the first risk assessment was intended to be included and updated with the information of the monitoring data. For the implementation of the updating process the same assumption, which were expressed as triangular probability density functions were formulated using normal distributions and used as prior information for parameter estimation. Parameter estimation was conducted using Bayesian inference using the open source software tools “R” and “Stan” (http://mc-stan.org/, www.r-project.org/). Figure 3-17and Figure 3-18 show the updating process for the mean influent concentration of *E.coli* and Norovirus in the influent of the WWTP Braunschweig. Due to the relatively flat priors the information in the data clearly dominates the location and variance of the resulting marginal posterior distribution of the mean although there have been only a few data points.

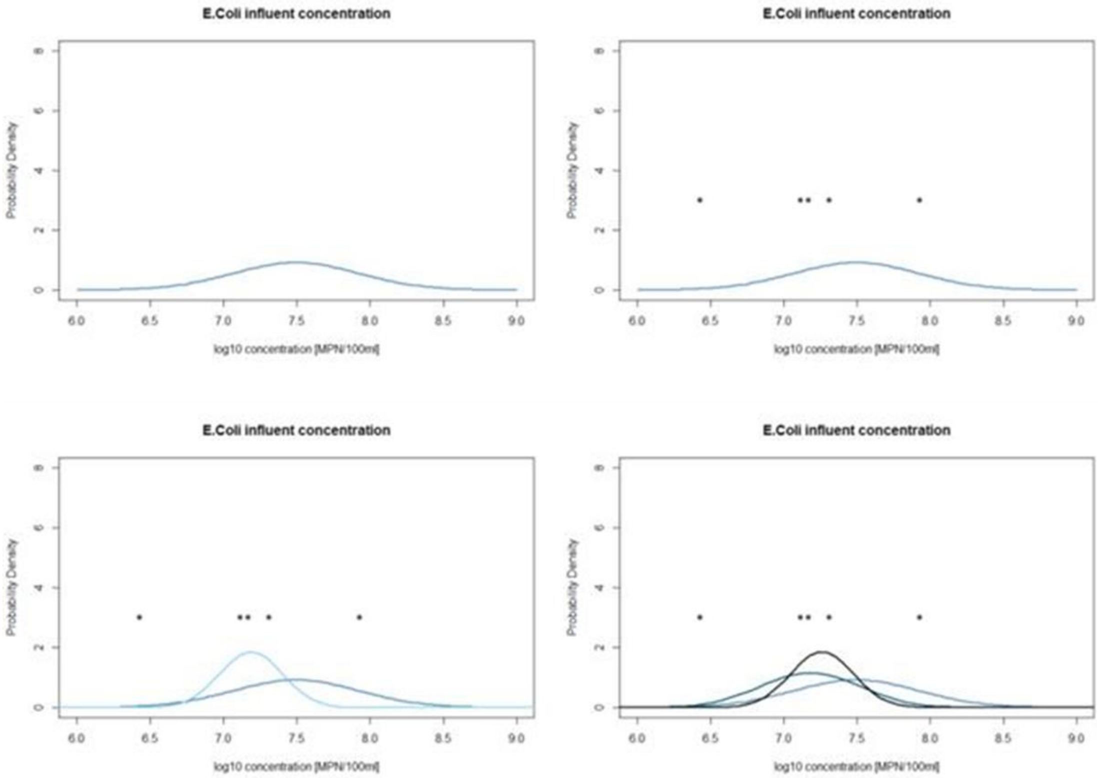


Figure 3-17: Illustration of the steps of Bayesian updating process

E.coli influent concentrations as an example: (1) Upper left: Setting up prior distribution based on literature information, (2) Upper right: arrival of data points (black dots), (3) Lower left: defining likelihood distribution for the data (light blue), (4) Lower right: combining information and calculate posterior distribution (black line shows marginal distribution of the mean)

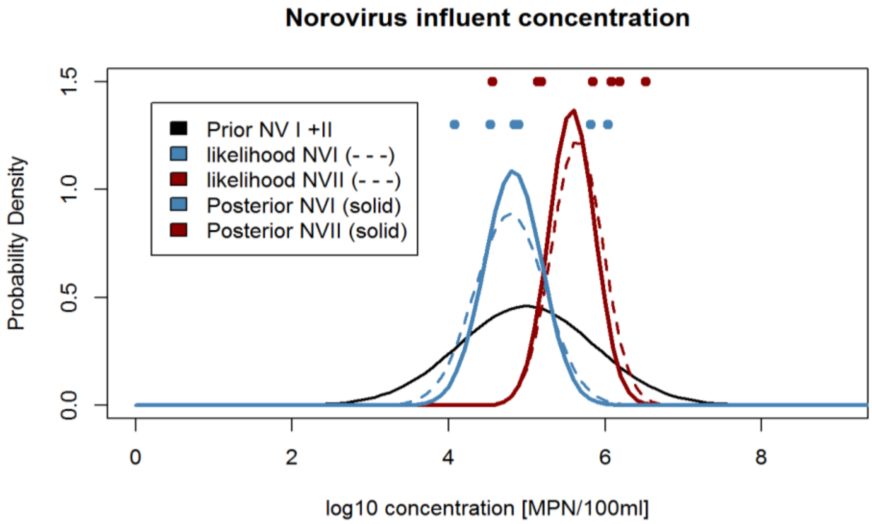


Figure 3-18: Bayesian updating of prior information on Norovirus concentrations in raw wastewater with measured data in the influent of the WWTP Steinhof. NV I and II stand for Norovirus GG I and GGII respectively.

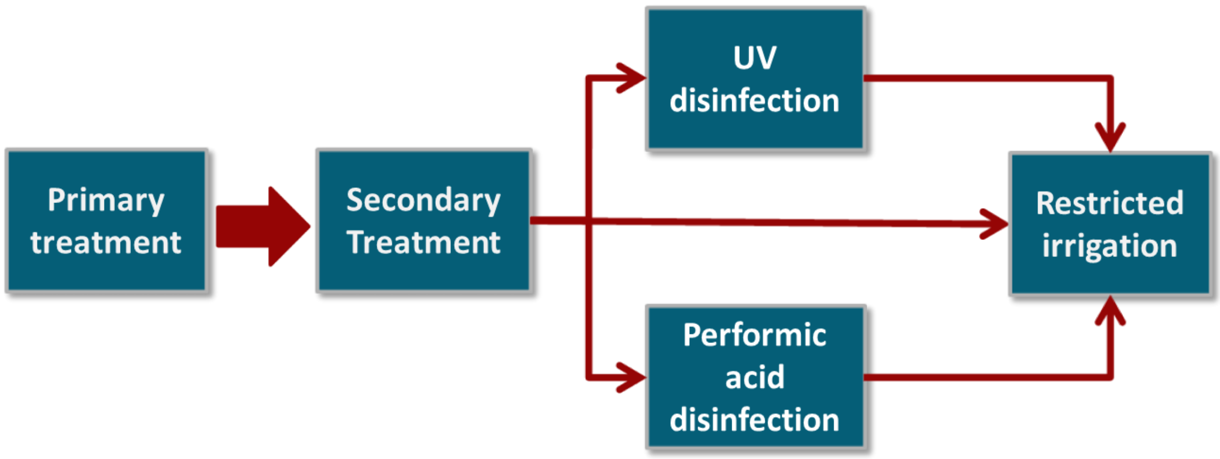


Figure 3-19: Overview of the assessed system in Baunschweig.

3.3.4 Hazard characterization

This section provides a short general characterization of the selected reference pathogens as well as the respective dose-response parameters and models.

Campylobacter

Pathogens of the genus Campylobacter (C.) are gram-negative rod shaped bacteria. Until now there are over 20 species known. The most important pathogenic species are *C.jenuni, C.coli* and *C.lari.*

C.jenuni and *C.coli* are globally distributed. The bacteria colonize a broad spectrum of animals including dogs, cats and pigs. *Campylobacter* bacteria are able to survive in the environment for a certain time but are not capable of multiplying outside the host.

The main route of human infection is via food consumption, especially due to the consumption of poultry. Infections also occur due to the consumption of and bathing in contaminated water. Human to human infections are rather rare.

Campylobacter infectious are currently the second most frequently reported bacterial infections in

Germany causing gastroenteritis. [79].

Norovirus

Noroviruses are known since 1972 and belong to the family of the Caliciviridae. They are globally distributed and account for the majority of global non-bacterial gastroenteritis cases. Up to 30% of all cases in children and 50% in adults are the result of Norovirus infection. Humans are the only known reservoir of Noroviruses.

Norovirus particles are excreted via feces and vomit and are highly infective. Hand contact of contaminated surfaces or inhalation of aerosol particles suffice for infection. Human-to-human infection is the main cause for the viruses' high prevalence. The virus may also be transmitted via food and contaminated water. Throughout infection the infected person is highly infective. Weeks after recovery, virus particles can still be found in feces [80].

Rotavirus

Rotaviruses belong to the family of the Reoviridae and are the main cause of severe gastroenteritis in children. The main routes of infection are human-to-human transmissions as well as the consumption of contaminated food and water [81].

Giardia

The protozoan pathogen *Giardia intestinalis* causes Giardiasis, an acute gastroenteritis. The typical way of transmission is the fecal contamination of water. Moreover, food-borne transmission and transmission via sexual contact have been reported. As a resting stage Giardia form so-called cyst which are highly resistant. Therefore, Giardia can survive for a long time in the environment outside its host. Symptoms of Giardia infection is an *"explosive, foul-smelling, watery diarrhea, intestinal cramps, flatulence, nausea, weight loss and malaise"*([82]).

Dose-response models

The following dose repose models have been used for the different pathogens. For the used dose response parameters, please see Deliverable 3.1. The simplest dose-response model is formulated by an exponential relationship and is used for Giardia.

Equation 12
$$P_I(d) = 1 - e^{-r*d}$$

Equation 13
$$-r = \frac{\ln(0.5)}{N_{50}}$$

$P_I(d)$ = *probability of infection*

d = *dose [N]*

r = *infectivity constant*

N_{50} = *median infectious dose*

The exponential model assumes that the probability of infection is constant for all pathogens of the same kind as well as for all people exposed to that kind of pathogen [55].

In reality not all pathogens of the same species are equally infective. Moreover, not all human show the same response on the exposure of the same amount of a certain pathogens. Old people as well as children may have a less strong immune system than adults. Consequently, they will be more easily become infected than an adult person. In order to consider such variations other functional relations are used. Most frequently the Beta-Poisson-model finds application (Campylobacter, Rotavirus, partly Norovirus).

Equation 14
$$_1F_1(\alpha, \alpha+\beta, d) \; = \frac{\Gamma(\alpha+\beta)}{\Gamma(\alpha)} \sum_{j=1}^{\infty} \left(\frac{\Gamma(\alpha+j)}{\Gamma(\alpha+\beta+j)} * \frac{(-1)^{j-1} * (d)^j}{j!} \right)$$

α, β = Beta-Poisson model parameters

d = dose [N]

[83] approximated the above equation to:

Equation 15 $$P_I(d) = 1 - \left(1 - \frac{d}{\beta}\right)^{-\alpha}$$

The approximation holds true for β ≥ 1 and α ≤ β [84] and low pathogen exposure. The approximation can be rewritten as:

Equation 16 $$P_I(d) = 1 - [1 + \frac{d}{N50}\left(2^{\frac{1}{\alpha}} - 1\right)]^{-\alpha}$$

A more complicated formulation for the dose-response relation was published by [85] for the infectivity of aggregated Norovirus particles. The additional parameter in the equation accounts for virus aggregation.

Equation 17 $$P_I(d, a, \alpha, \beta) = {}_2F_1\left(\alpha; \frac{-d}{\log(1-\alpha)}, \alpha + \beta, \frac{-a}{1-a}\right)$$

d = dose

α, β = model parameters

a = constant for the aggregation of virus particles

3.3.5 Exposure scenarios and assessment

Figure 3-20 gives an overview on relevant exposure pathways regarding water reuse in agriculture.

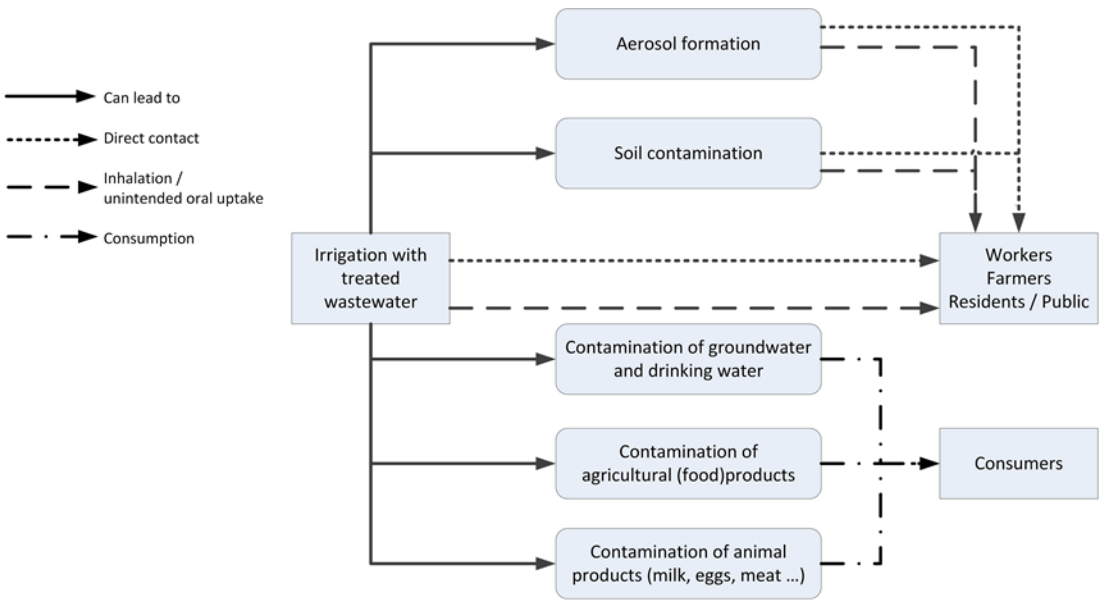


Figure 3-20: Overview of relevant exposure pathways regarding water reuse in agriculture [86]

In Brausnchweig no crops are grown which are comsumed without prior processing. Therefore, hygienic aspects related the produced food products are considered negligible. Regarding the inputs of chemical substance immissions are considered to be dominated by the use of manure, pestices and fertilizers, which are directly applied by conventional agricultural practice. Exposure to pathogens was assessed in three different scenarios: a. exposure of local farmers and fieldworkers, b. exposure of local/nearby residents and c. children ingesting soil irrigated with reclaimed water. For irrigation on demand an irrigation period from calendar week 18 to 36 is assumed, which reduces the amount of exposure events. In the following the scenarios are described shortly. Table 3-8 summarizes the major assumptions.

Reference (Scenarios 0a)

Fieldworkers

Field workers and farmers is the population group most directly exposed to treated wastewater and sewage sludge as they work directly on the irrigation area. A QMRA for wastewater irrigation was conducted by [87], published by the WHO [88]. For highly mechanized agriculture, a daily intake of 1-10mg contaminated soil is assumed. No die-off is considered. In the present study the assumption from the Australian Guidelines for Water recycling are used, which use exposure volumes of 0.1-1 ml/ event

Nearby residents

For exposure assessment of nearby residents, the dose of solid and liquid aerosol particles people are exposed to has to be estimated. Viau et al. conducted a QMRA study in 2011, where particle exposure due to biosolid application was modelled. Depending on the wind speed and distance from the site of application they published a range of inhaled PM10 particles from biosolid land application from 0.05µg per application event at a wind speed of 20m/s and a distance of 1000m to 25.3µg per application event at a wind speed of 1.5 m/s and a distance of 5m.

The legal permission for wastewater reuse in Braunschweig defines minimum distances between the irrigation machine and the landed properties of local residents depending of the size of the nozzle outlet of the irrigation machine the minimum distance varies between 60 and 150m. The average wind speed for the region of Braunschweig is set to 3m/s . For this wind speed and distance range Viau et al. published an inhalation dose of PM10 particles produces by biosolid land application form 4.5-6.9 µg per application event. The study they conducted focused on the application of dewatered sewage sludge, whereas in Braunschweig liquid sludge is mixed into treated wastewater. Within their publication [89] year of publication state that "*land-applying dewatered biosolids [...] produces an aerosol emission rate approximately 80 greater than emission rates observed for liquid sludge spray application*" ([89], p.5466, ll. 17-20). Thus, an additional exposure reduction of a factor of 80 was applied for the calculation of aerosol exposure of the local communities in Braunschweig.

Soil ingestion by children

Children may play on agricultural areas or may accompany adult while they go for a walk. Especially young children tend to ingest higher amounts of soil. To account for this kind of risk an annual number of exposure events of 10 is applied. The amount of soil ingested is set to 20-100mg per exposure event [90].

Table 3-8: Overview of exposure scenarios for status quo and irrigation on demand

Scenario	Volume/amount of soil per exposure event	Number of exposure events per year (Status quo)	Number of exposure events (irrigation on demand)	Number of people exposed
Fieldworker	0.1 – 1ml	180	80	50
Residents	4.5 – 6.9 µg soil	300	126	500
Children	20-100 mg soil	5	2	1

Changes in concentration due to irrigation on demand (Scenario 1b)

Irrigation on demand not only leads to a reduction of the number of exposure events but also reduces the concentration of pathogens, which show high seasonality in their incidence, like e.g. *Norovirus* and *Rotavirus.*

In order to quantify the effect from changing the irrigation regime in Braunschweig epidemiological data from the German Robert Koch Institute about the reported number of Norovirus disease has been has been evaluated [91] (Figure 3-21). The data has to be used with caution, since the numbers are expected not to cover the full number of disease incidence due to underreporting. However, assuming a constant rate for underreporting the data are reliable enough underline the tendency of elevated incidence rates during winter. Moreover, it becomes evident that due to the spread in the data during winter, uncertainty effects additionally increase the expected variations between seasonal incidence rates.

These variations have been used to roughly to estimate the log reduction potential of a change in irrigation management in Braunschweig, since currently irrigation takes place from February to the end of November, even though the irrigation water is not needed. Changing the irrigation patterns to an “irrigation on demand” system, which solely irrigates during dry summer periods, might be one option to reduce the exposure to wastewater when concentrations are expected to be the highest.

In order to estimate changing virus concentrations in the influent of the WWTP a constant rate for underreporting has been used. In England and Wales underreporting rates of *Norovirus* disease of 12.7 have been published. In Germany, a minimum underreporting rate was estimated based on epidemiological data after the EHEC outbreak in 2011. The authors investigated the increased reporting rate in the three calendar weeks between the first public information about a present outbreak with still unknown origin. In this time the authors expect both people and physicians to be more aware and cautions leading to an increase in reporting rates. In Figure 3-21 the effect of this increased awareness becomes visible. The minimum underreporting rate was highest among the age group from 20 -29 with a factor between 2 and 3. Using the information of both, the study of England and Wales and of Germany the assumed underreporting rate has been set to a constant value of 6.

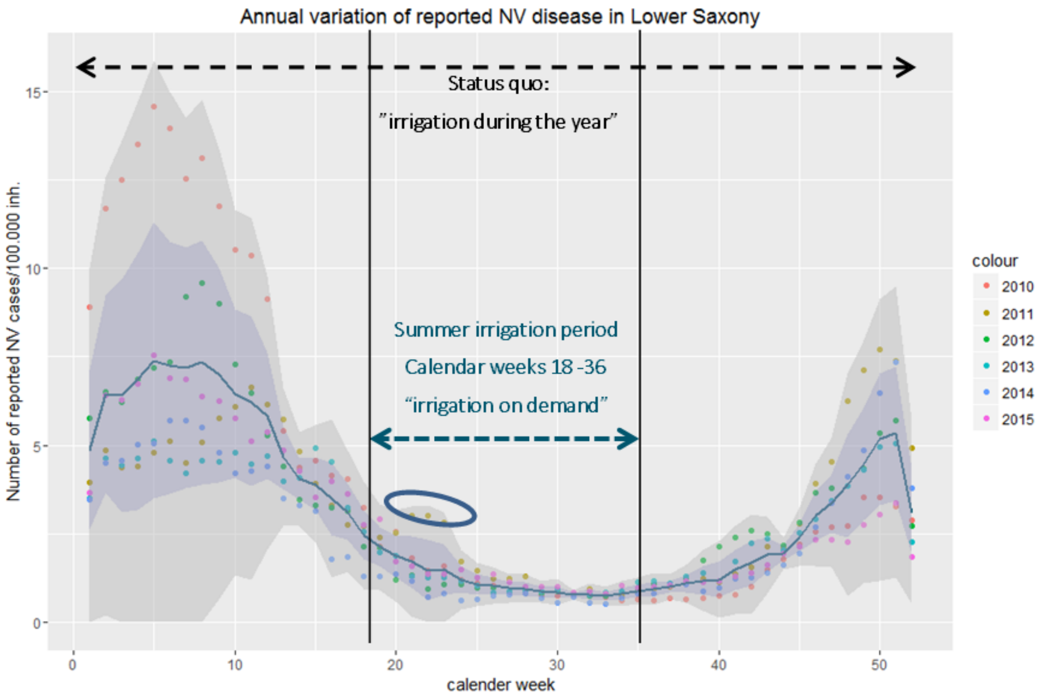


Figure 3-21: Incidence of Norovirus disease per 100.000 inh. per calender week from 2010 to 2015 in the federal sate of Lower Saxony [91]

Solid line: Posterior mean, Dark shading: Credible interval of the posterior mean, light shading: Prediction interval of the joint posterior distribution. The ellipse shows the time after the EHEC outbreak and the increase in reporting rate.

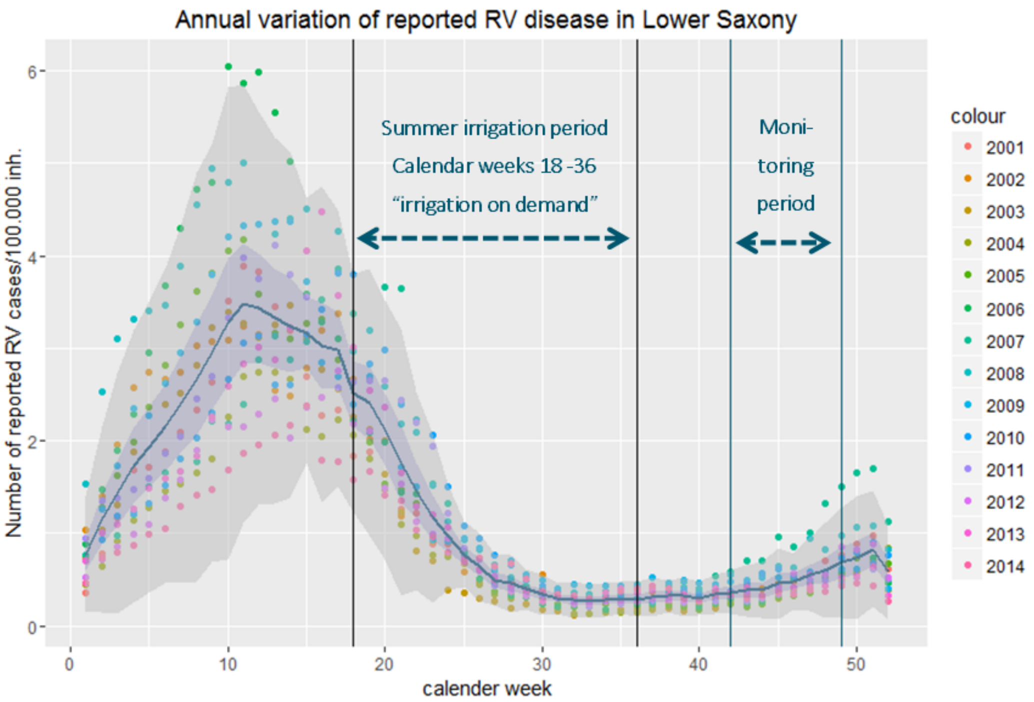


Figure 3-22: Reported incidence of Rotavirus disease in Lower Saxony from 2001 – 2014

Solid line: Posterior mean, Dark shading: Credible interval of the posterior mean, light shading: Prediction interval of the joint posterior distribution.

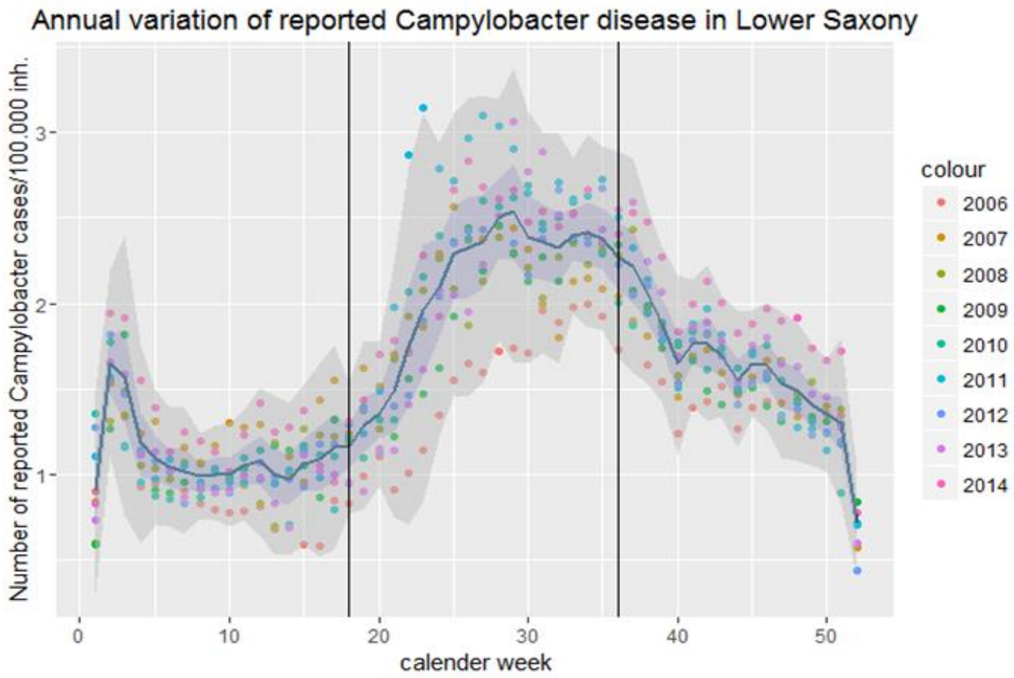


Figure 3-23: Reported incidence of Campylobacter disease in Lower Saxony from 2006 – 2014

Solid line: Posterior mean, Dark shading: Credible interval of the posterior mean, light shading: Prediction interval of the joint posterior distribution.

For estimating seasonal variations, first, the joint posterior distribution for the incidence rate for each calendar week has been derived using the data of several years (see Figure 3-21 to Figure 3-23) and the methods outlined in [61]. First the variance has been sampled from a scaled inverse chi squared distribution. The sampled variance was used to sample from the conditional posterior $\mu \mid \sigma^2, y \sim N\,(\bar{y}, \sigma^2/n)$ to get a sample from the joint posterior distribution. The incidence per 100.000 inhabitants was multiplied with the factor for underreporting and the connected population equivalents of 350.000 PE of the wastewater treatment plant to get an estimate of the number of infected people in the catchment for each calendar week.

Influent concentration is subsequently calculated by:

Equation 18
$$C_{Pathogen}[N/L] = \frac{N_{cases} * m_{faeces}\left[\frac{g}{d}\right] * c_{faeces}\left[\frac{N}{g}\right]}{10^3 * V_{wastewater}\left[\frac{m^3}{d}\right]}$$

Calculations were done for the mean number of cases as well as the upper prediction interval. Table 3-9 summarizes the used information for the estimation of influent concentration.

Given the made assumptions a maximum difference of 0.65 log units is estimated between summer and winter concentration of *Norovirus.* For estimating the effect of "irrigation of demand" summer predictions are compared to the current annual average scenario. This reduces the effect to a value of 0.55 log units. For Rotavirus a reduction 0.45 log units was calculated. No improvement could be achieved for Campylobacter. For Giardia no seasonality is expected [92].

Table 3-9: Information used for the estimation of Norovirus concentrations in municipal wastewater in Braunschweig

Factor	Distribution	Parameter/values	Reference
Daily amount of wastewater	Triangular	57000 ± 25% m^3	Abwasserverband Braunschweig
Number of Norovirus infections	Normal	Sampled for each calendar weeks separately	Data from RKI + literature for underreporting
Pathogens per gram feces	Lognormal	LN (21.6, 1.3)	[93], [94]
Excreted feces per day [g/d]	Triangular	Tri (min=150,max = 400, mode =300)	[95]
PT(Steinhof)	Point estimate	350000 PT	Abwasserverband Braunschweig

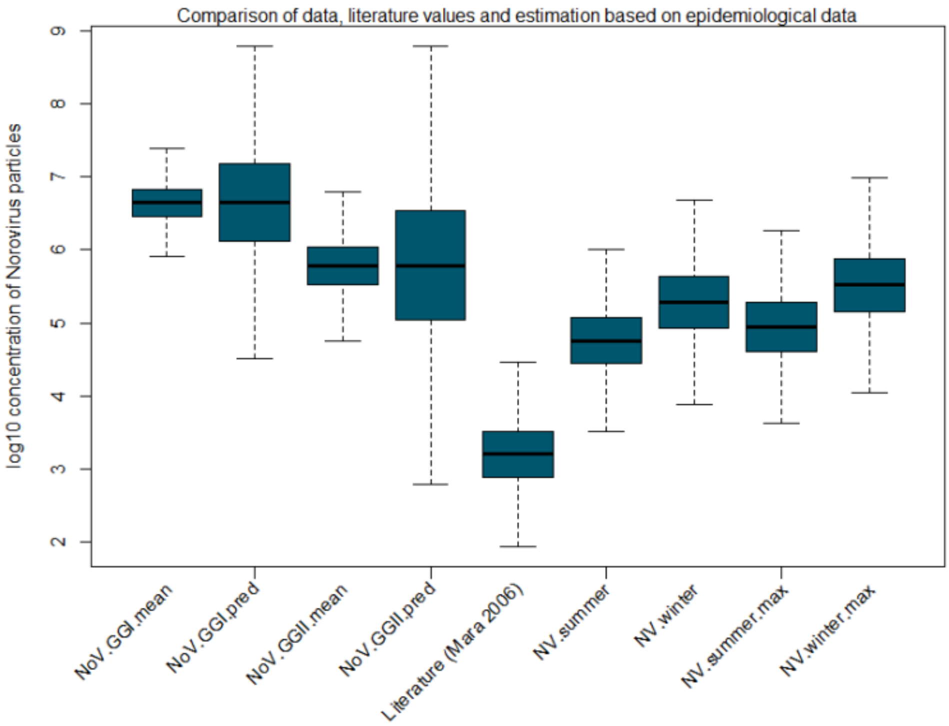


Figure 3-24: Comparison of measured and modelled Norovirus influent concentration in comparison to assumptions found in literature regarding the ratio between E.coli and Norovirus in municipal wastewater.

Disinfection with UV and PFA (Scenarios 2a and 2b)

In the present study, additional the change of the irrigation management, the risk reduction potential of the disinfection with UV radiation and PFA dosage has been assessed based on the data and information of the conducted disinfection trials [78]. Thus, the present results should not be considered as a generic assessment of the performance of these treatment technologies, but rather as an assessment of the setup used for the pilot trials.

Somatic coliphages have been used for the assessment of virus reduction, *E.coli* for the reduction of *Campylobacter* and *Clostridium Perfringens* for the removal of cycst and oocyst of *Giardia* and *Cryptosporidium*, respectively.

Assumption on the volume and frequencies of exposure stay the same of the scenarios without disinfection.

3.3.6 Risk characterization

The following figures show the calculated risk in additional DALYs (disability adjusted life years) per person per year (see Deliverable D1.3 for DALY calculation). Results are shown for the three considered population groups, five reference pathogens, three different water qualities each for irrigation on demand and irrigation as it is today, leading to 90 different risk estimations.

Scenarios for status quo irrigation

Figure 3-25 shows the risk of disease for people working on the agricultural area in Braunschweig given today´s irrigation regime. Given the made assumptions, risk is highest for *Campylobacter*, *Noroviruses* and *Rotavirus* if secondary effluent is used for irrigation (no disinfection). In contrast, due to the low disinfection performance of UV (without prior filtration) against *Clostridium Perfringens* as an indicator for parasites during the disinfection trials, *Giardia* was calculated to be the pathogen of highest concern after this disinfection option. The treatment performance of UV was generally better than the one of PFA disinfection, which leads to a lower risk for all pathogens. Expect from *Giardia* UV disinfection was able to reduce the risk close to a health based target of 10^{-6} DALYs per person per year. High uncertainty remains especially for risk estimates of *Rotavirus* ranging from slightly below 10^{-6} to $10^{-4.5}$ additional DALYs per person per year. PFA disinfection showed results of up to two orders of magnitude higher than risk calculation for UV disinfection.

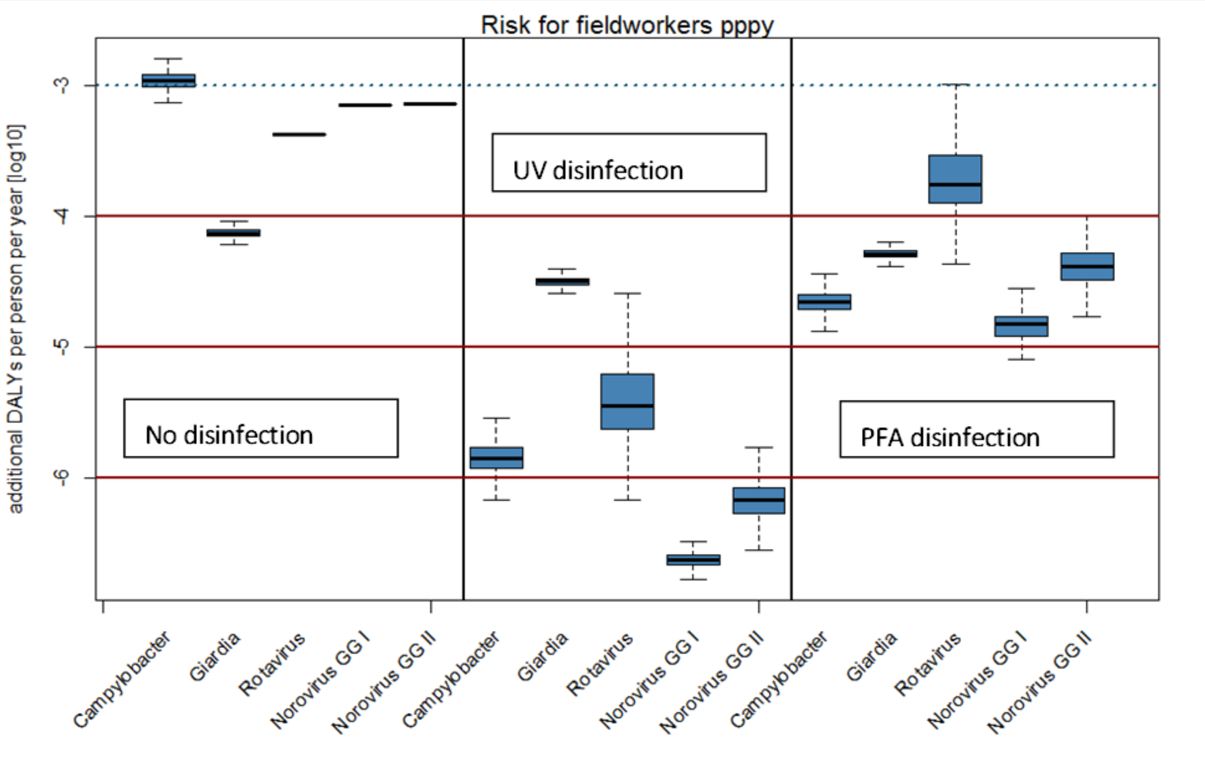


Figure 3-25: Risk expressed in additional DALYs per person per year for fieldworkers
Solid red lines indicate health based targets of 10^{-6}, 10^{-5}, and 10^{-4} additional DALYs per person per year.

For the children scenario (Figure 3-26) the high amount of soil ingested per exposure event (20 -100mg) leads to a risk of disease between 10^{-9} to $10^{-3.8}$ additional DALYs per person per year, with only Rotaviruses excedding the WHO benchmark in about 50 % of the realised simulations. Since no pathogen die off is considered, this estimate counts only for freshly irrigated areas. Thus, on the one hand, risk might be overestimated given that exposure event are expected to occur very rarely. On the other hand, dose response curves relations of children might be steeper, as most of these relations were derived from healthy adults. This counts especially for Rotaviruses, which predominantly infect chidlren. In this case risk might be underestimated. After disinfection none of the realisations for PFA and UV disinfection indicated values above 10^{-6} additional DALYs for Rotavirus.

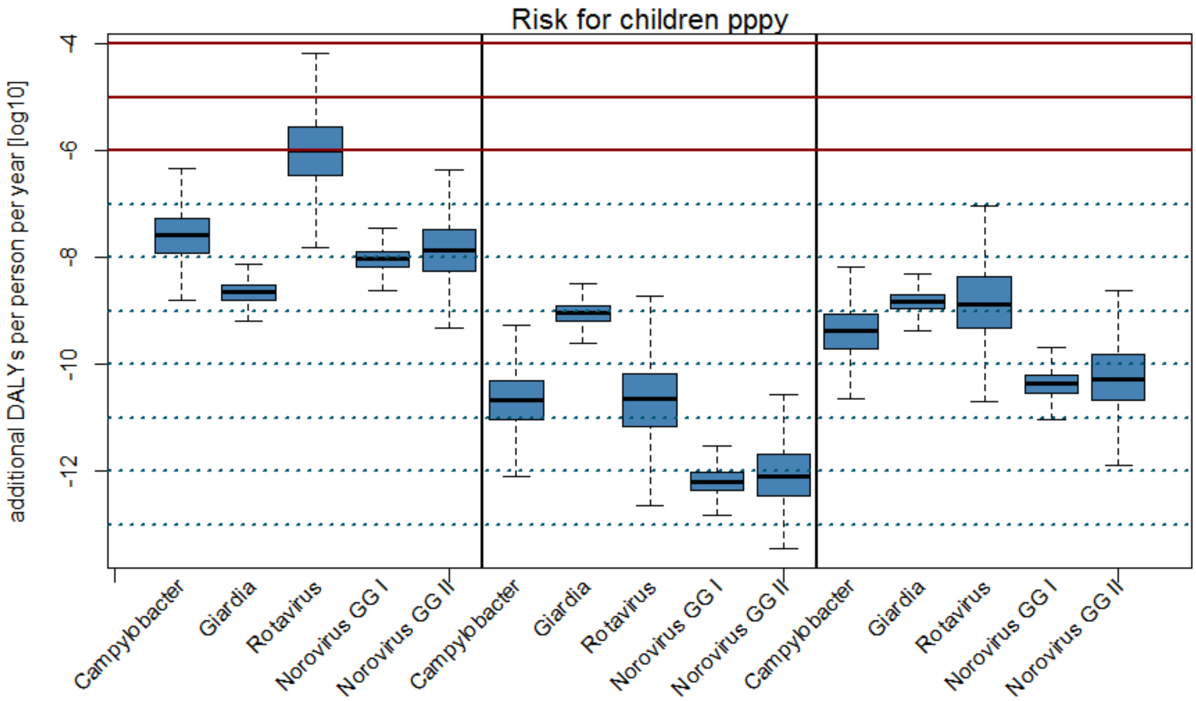


Figure 3-26: Risk expressed in additional DALYs per person per year for children
Solid red lines indicate health based targets of 10^{-6}, 10^{-5}, and 10^{-4} additional DALYs per person per year.

For the resident scenario only Rotavirus and Norovirus GGI showed values slightly above 10^{-6} additional DALYs per person per year. This is due to the extremely low infectious dose of these pathogens. However, the present calculations are based on the minimal distance between sprinkler and private property, where the irrigation machine will not be all of the time, so that lower risk can be expected.

Consequently, risk after both disinfection options are well below acceptable levels for all pathogens.

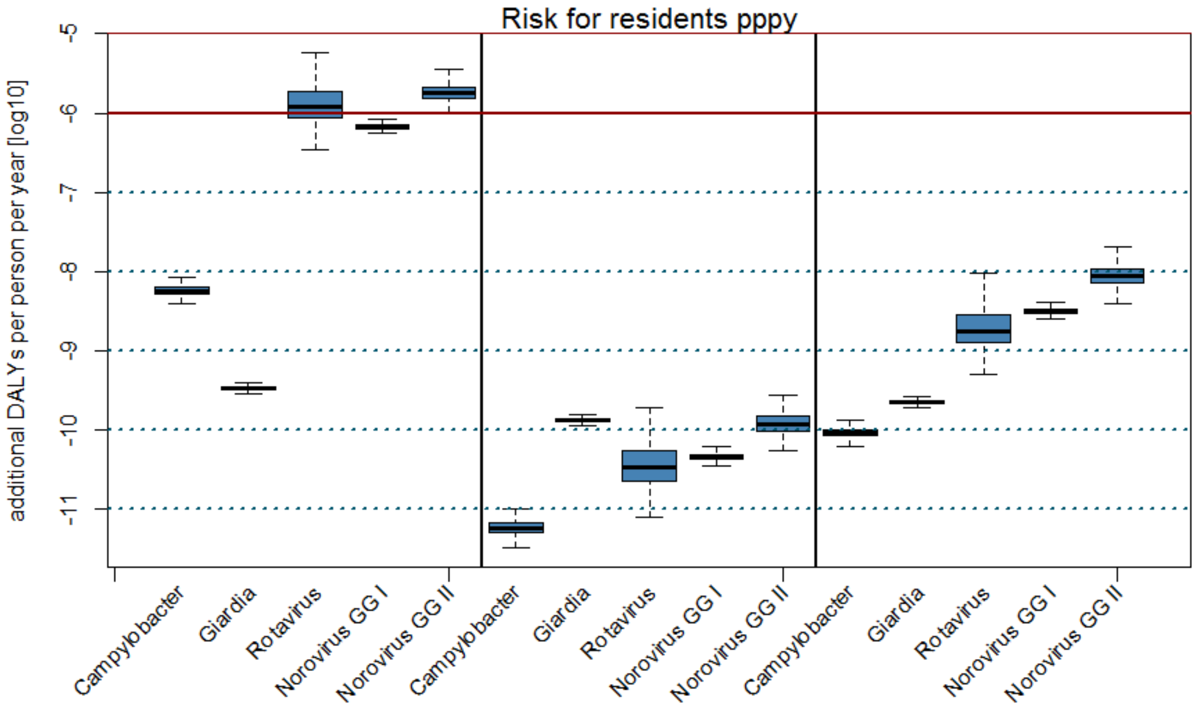


Figure 3-27: Risk expressed in additional DALYs per person per year for residents
Solid red lines indicate health based targets of 10^{-6}, 10^{-5}, and 10^{-4} additional DALYs per person per year

Scenarios for irrigation on demand

For irrigation with secondary effluent a change of the irrigation regime does not suffice to reach a health target of 10^{-6} for any of the assessed pathogens. After PFA disinfection risk was estimated between 10^{-4} and 10^{-6} additional DALYs per person per year. Only UV disinfection was able to reduce pathogen risk below 10^{-6} for 4 out of 5 assessed reference pathogens. Considering the additional risk reduction pre-filtration might provide the results of UV disinfection are the most promising from a health risk point of view.

For the exposure scenarios for children and residents, irrigation on demand reduces risk for most pathogens already below 10^{-6} additional DALYs per person per year. Thus, the exposure prevention methods mentioned above suffice to reduce risk below acceptable levels.

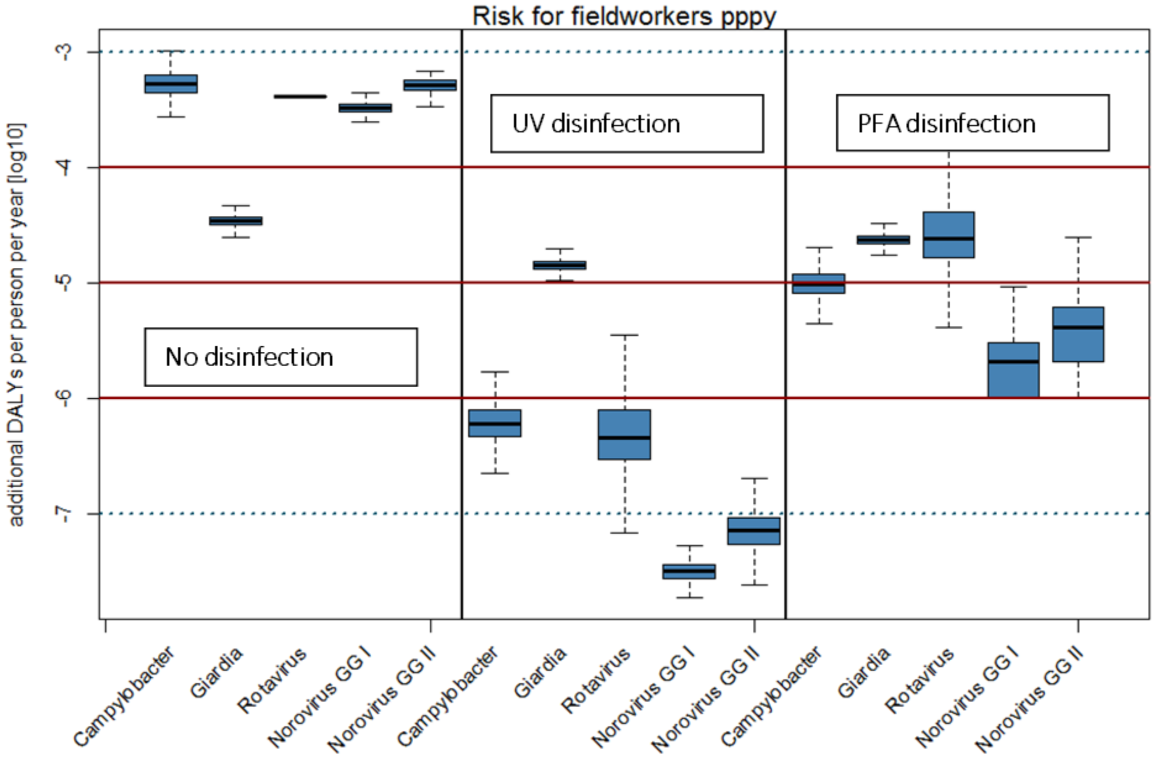


Figure 3-28: Risk expressed in additional DALYs per person per year for fieldworkers with irrigation on demand
Solid red lines indicate health based targets of 10^{-6}, 10^{-5}, and 10^{-4} additional DALYs per person per year.

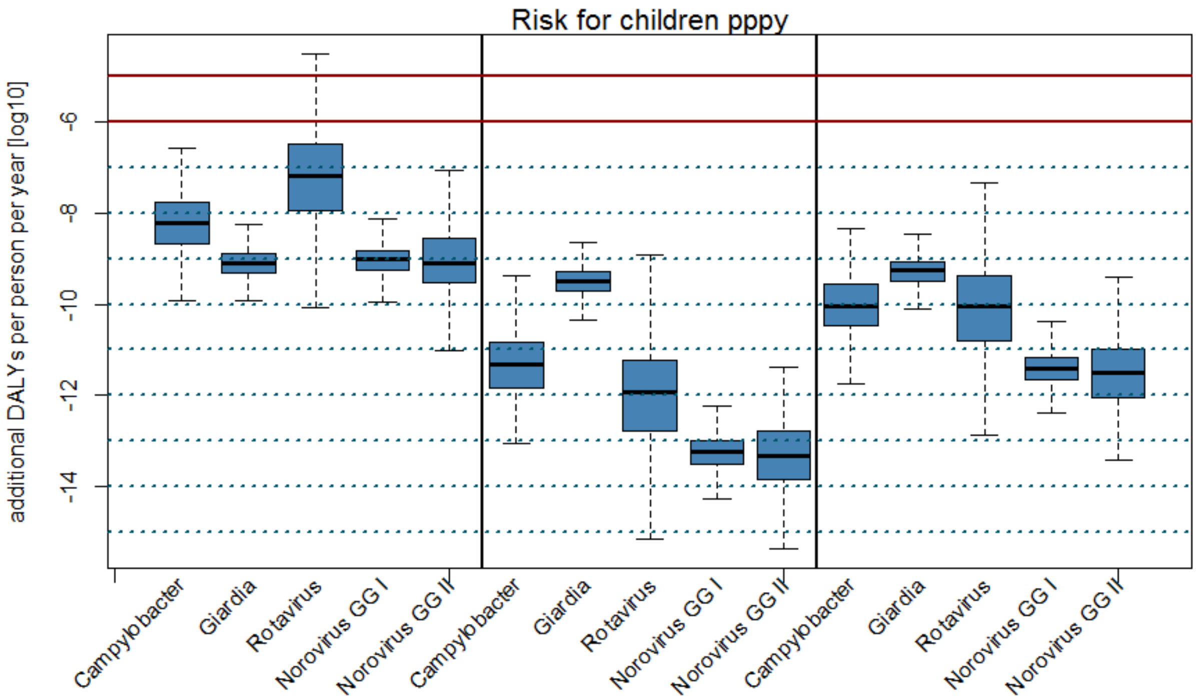


Figure 3-29: Risk expressed in additional DALYs per person per year for children with irrigation on demand
Solid red lines indicate health based targets of 10^{-6}, 10^{-5}, and 10^{-4} additional DALYs per person per year.

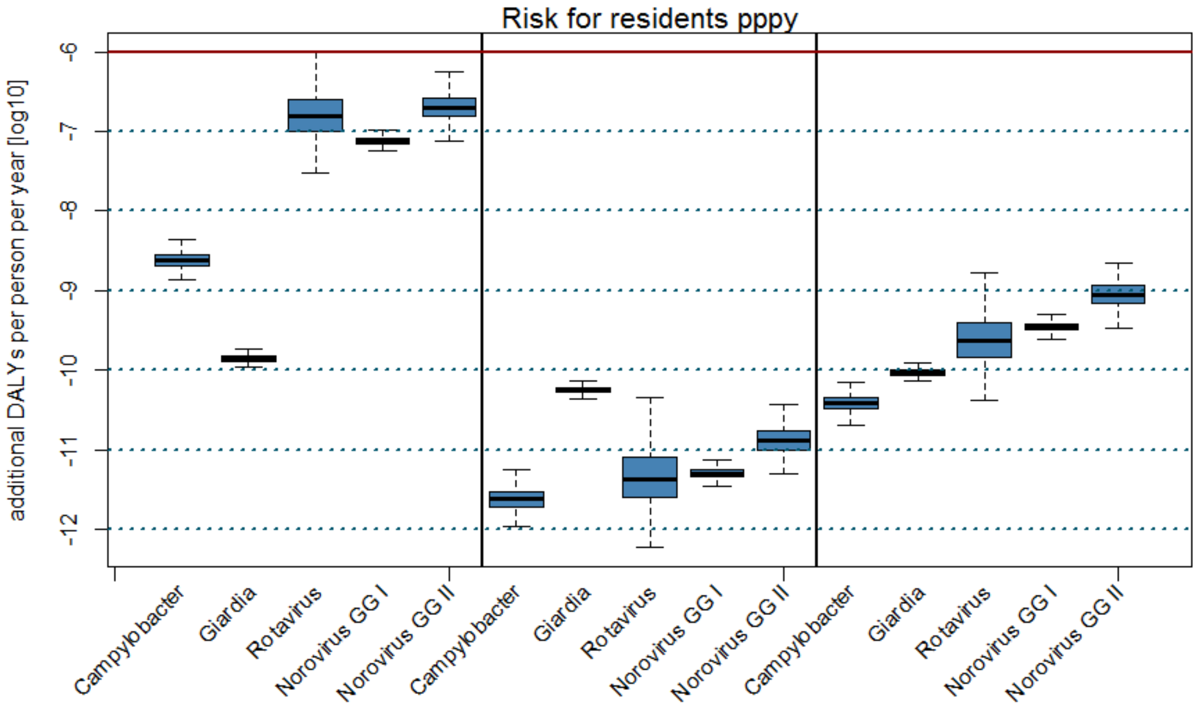


Figure 3-30: Risk expressed in additional DALYs per person per year for residents with irrigation on demand
Solid red lines indicate health based targets of 10^{6}, 10^{-5}, and 10^{-4} additional DALYs per person per year.

Summary

Against the background of the made assumption and the results from the conducted monitoring campaigns and disinfection trials the following conclusion can be derived.

Regarding the considered population groups fieldworkers are exposed to reclaimed water most regularly and to the highest volumes and were the population group at highest relative risk. Risk decreased in the order:

Fieldworkers > children ingesting soil > local residents

Moreover, given the information of the disinfection trials UV disinfection generally showed better disinfection performance and consequently reduced risk more effectively. Regarding both treatment options measures for further improvement became evident. Mixing and reaction time was not optimal during the PFA disinfection trials and consequently the outcomes have to be treated with caution and cannot be seen as a generic assessment of this treatment option. For UV disinfection the secondary effluent was treated without filtration prior to UV disinfection. This may explain the treatment efficiency of only 0.5 log units against Clostridium Perfringens. A comparable setup in El Port de la Selva with prior filtration showed reductions of up 3.5 log reduction for this indicator.

Against this background the elevated risk for parasitic pathogens may be reduced by implementing a filtration step prior to UV disinfection as well as by reassessing PFA performance with controlled reaction time and complete mixing. Treatment performance of PFA can potentially be improved by controlling mixing rates and residence time in the reactor by improved dosing and / or reactor geometry

In general, the results show elevated risk for people working on the agricultural areas in Braunschweig. Especially for parasitic pathogen the risk calculation showed that a health based target of 10^{-6} additional DALYs per person per year might not be achieved even with UV disinfection. For the other pathogens only the combination out of UV disinfection and irrigation on demand was able to reduce risk below a health based target below 10^{-6} additional DALYs per year for people working on the area. For the residents and children scenario changing irrigation patterns to an “irrigation on demand” system in combination with the exposure control measures already in place seems to be sufficient to reduce risk below acceptable levels.

Given that the DALY indicator is quite abstract concept it is worth recapitulating what a risk of 10^{-6} additional DALYs actually means, especially against the background of setting the same acceptable level for drinking water supplies.

The following calculations try to express the risk in cases of diarrhoea per person per year working on agricultural areas in Braunschweig:

Acceptable risk of 10^{-6} additional DALY per person per year (pppy) means:

- ~ Probability of illness (Rotavirus): $1.2*10^{-3}$ per person per year
- ~ Probability of illness (Cryptosporidium/Giardia): $6.7*10^{-4}$ per person per year
- ~ Probability of illness (Campylobacter) : $2.2*10^{-4}$ per person per year

Given 50 employees at the wastewater association in Braunschweig 10^{-6} additional DALYs would be equivalent to:

- 1 case of *Rotavirus* disease among 50 people every 17 years
- 1 case of Cryptosporidium disease among 50 people every 30 years
- 1 case of Campylobacter disease among 50 people every 91 year

Expressed as risk of disease per working career assuming 30 years of working at the wastewater association:

- $3.6*10^{-2}$ for Rotavirus disease
- $2.0*10^{-2}$ for Cryptosporidium disease
- $6.6*10^{-3}$ for Campylobacter disease

Comparing these numbers to an average disease probability for diarrhoea of 0.2 per person per year (WHO 2006) in industrialized countries, those numbers underline that using a health based target of 10^{-6} additional DALYs per person per year leads to very low acceptable risk which will not be detectable by any epidemiological survey because they lie two orders of magnitude below the incidence in the general public. Against these background discussions about where to spend limited resources from a public health perspective should keep in mind that even a health based target of 10^{-5} additional DALYs per person might be sufficiently protective in comparison to the risk of mild diarrhoea in the general public.

3.3.7 Chemical risk assessment

The used methodology of quantitative chemical risk assessment (QCRA) follows the methods of the *European Union Technical Guidance Document on Risk assessment [96].* Like QMRA the conducted QCRA is structured in:

- Hazard identification
- Hazard characterization
- Exposure assessment
- Risk characterization

Available local data

Annual mean heavy metal concentrations measured in the STP Steinhof are available (Appendix A). Annual loads are calculated by further using the respective measured influent, effluent and sludge rates. Soil metal contents for the four pumping districts are available for the years from 1993-2010 Moreover, Cadmium concentrations in wheat, corn and sugar beet are available for the time span from 1995 to 2010. Data on soil properties (pH, clay content, content of organic carbon) are available from previous research studies [97]. Climate data (rain rate, average temperature) as well as atmospheric deposition of heavy metals are available from national surveillance programs [98].

Limit values

In Braunschweig, both treated wastewater and sewage sludge are distributed at the agricultural area for water and nutrient supply. The German ordinance for the application of sewage sludge (Klärschlammverordnung,[99]) defines quality limit values for sludge and soil concerning the respective content of heavy metals.

Moreover, a maximum amount 5t of sewage sludge may be applied per hectare of arable land within a three years period. Thus, a maximum annual load of heavy metals is set (Table 3-10).

Table 3-10: maximum allowed heavy metal concentrations of sewage sludge and arable soil as well as the calculated maximum annual load considering the maximum amount of applied sludge of 5t/3years.

Metal	Sludge concentration [$mg/kg_{sludge}(dw)$]	Soil concentration [$mg/kg_{soil}(dw)$]	Maximum annual load [g/ha*a]
Cadmium (Cd)	10 (5)	1.5 (1)	17
Chromium (Cr)	900	100	1500
Copper (Cu)	800	60	1333
Nickel (Ni)	200	50	333
Mercury (Hg)	8	1	13
Lead (Pb)	900	100	1500
Zinc (Zn)	2500 (2000 sandy soils)	200 (150 sandy soils)	4167

According to this ordinance sludge application is prohibited 14 days before harvest [99].

Quality of monitoring data

At the STP Steinhof parameters for nutrients, suspended particles, heavy metals as well as sum parameters for carbon and halogenated organic carbon compounds are measured regularly. Monitoring data are given in Appendix A.

For the risk assessment the data for heavy metals are of special interest. The monitoring program concerning heavy metals is shown in Table 3-11.

Table 3-11: Monitoring program of heavy metals at the STP Steinhof

Sampling site	Sample	Parameter	Frequency	Day	Comments
Influent primary treatment	24h-mixed sample	Cd, Cr, Pb, Cu, Zn, Ni	Daily	Mo-Sun	Complete influent STP
		Hg	2x/month		
Effluent activated sludge treatment	24h-mixed-sample	Cd, Cr, Pb, Cu, Zn, Ni	3x/week	Mo, Wed, Fri	Entering irrigation fields
		Hg	2x/month		
Effluent irrigation	24h-mixed-saple	Cd, Cr, Pb, Cu, Zn, Ni, Hg	2x/week	Di, Do	Effluent STP + treated sludge
Primary Sludge	Grab sample	Cd, Cr, Pb, Cu, Zn, Ni, Hg	1x/week	On varying days	Sludge of primary sedimentation
Activated sludge	Grab sample	Cd, Cr, Pb, Cu, Zn, Ni, Hg	1x/week	On varying days	Activated sludge
Effluent Aue-Oker-Canal	24h-mixed-sample	Cd, Cr, Pb, Cu, Zn, Ni, Hg	1x/week	Tue or Wed	Official effluent STP

On the agricultural areas of the AVBS soil contents of Cd, Cr, Cu, Hg, Ni, Pb, and Zn are measured once a year. Additionally, cadmium concentrations in wheat, sugar beet and corn are measured once a year. For monitoring purposes the 3000ha are divided in four districts. Due to the presence of pumping stations in these four districts, the four districts are referred to as Pumping district I, Pumping district II, Pumping district III and Pumping district IV. The single pumping districts are in turn subdivided in 5-7 areas, respectively.

Existing reduction measures

Concerning heavy metals reduction measures have to be applied before the respective metal enters the sewage system as heavy metals are not biodegradable and thus will not be reduced or eliminated in the STP. In Braunschweig SE|BS monitors the industrial discharges into the sewage system. At over 300 sites concentrations of certain contaminants are measured. Industrial discharges with potentially high heavy metal contents are controlled 6-8 times per year [100]. By this control of industrial discharges annual heavy metal loads could be significantly reduced during the last decades (Figure 3-31).

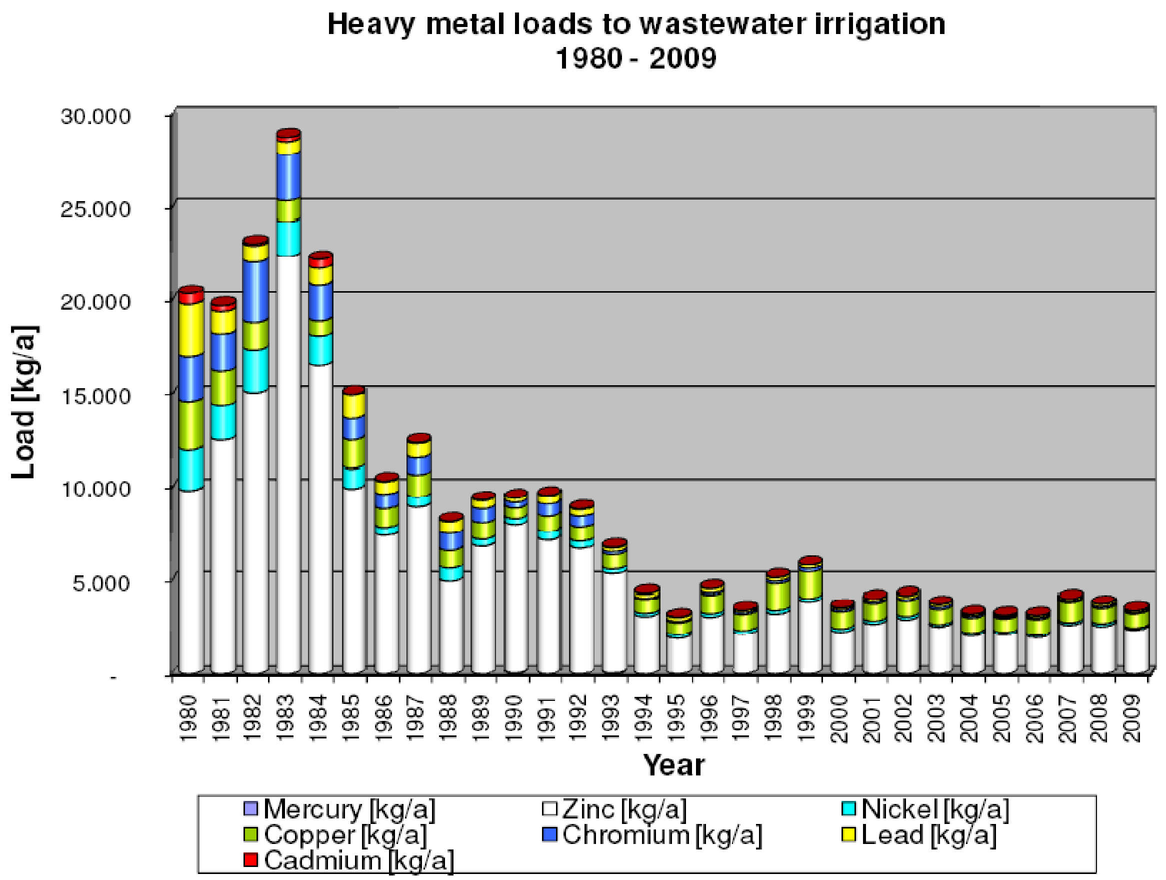


Figure 3-31: annual loads of heavy metals which were applied on the agricultural areas of Braunschweig [63].

Hazard characterization

This section characterizes the different heavy metals concerning their effects on human health and the environment.

Cadmium

Cadmium has toxic effects on human health already at very low concentrations. Cadmium accumulates especially in the liver and the kidney. Due to its long half-life time inside the human body Cadmium concentration increase over life time. A critical kidney concentration of approximately 200µg/g (fresh weight) may cause proteinuria ([101], section 7.3.5.3). Painful bone disorders are another effect of Cadmium exposure, including spontaneous bone fractures. Severe bone disorders due to Cadmium exposure have been observed in Japan (Itai-Itai-disease). Cadmium as classified as carcinogenic [102].

For Cadmium, no biological function is known. Cadmium is toxic for various terrestrial and aquatic organisms. Cadmium is the metal which shows the highest mobility of all heavy metals at moderate pH levels (<6.5). Due to the combination of high mobility and high toxicity Cadmium is of special concern.

Cadmium is classified as a priority substance of the European Water Framework Directive. Today the major sources of Cadmium emissions to water are the result of urban surface runoff, erosion and drainage of agricultural areas as well as municipal sewage treatment plants [103].

Mercury

Like Cadmium Mercury is highly toxic to human health. Long term exposure to even very small Mercury concentrations can cause severe neurological disorders and immune-deficiencies [102]. The biological half-life time inside the human body is about 70 days [101].

As for Cadmium there are no biological function known for Mercury. Mercury is rather immobile and accumulates in the organic layers of the top soil. Because of its immobility plant uptake is of minor importance. Accumulation, especially of organic Mercury-compounds has been found in fish.

Mercury is classified as a priority substance of the European Water Framework Directive. The main sources of Mercury emissions to water are identical to that of Cadmium [104].

Lead

The level of human toxicity of Lead is far below the ones of Cadmium and Mercury. Lead accumulates in bones, teeth, the liver and the kidney [101]. One of the characteristics of long and high lead exposure is anemia. Moreover, lead exposure can lead to hematological and neurological effects as well as to adverse reproductive and development effects [102]. The biological half life time inside the human body lies between 5 and 20 years. Therefore, lead concentration increases over lifetime [101].

There are no biological functions know for lead in the environment. Lead shows adverse effects on plant and terrestrial microorganisms, but to a lesser extent as Cadmium.

Lead is classified as a priority substance of the European Water Framework Directive. The main sources of Lead emission to water are urban and agricultural surface runoff as well as municipal sewage treatment plants [105].

Nickel

Human health effects concerning Nickel exposure are mainly known for the respiratory tract. Long term Nickel inhalation may cause chronic bronchitis and a reduction of lung functions. The most important exposure route is via food intake, but as only a small fraction of the Nickel in food is resorbed by the human body (1-2%) there are currently no known adverse human health effects due to Nickel intake via food [101]. Nickel is classified as carcinogenic and may cause skin irritation [106].

Concerning its environmental relevance Nickel shows high phyto-toxicity and may cause adverse effects on soil organisms.

Nickel is classified as a priority substance of the European Water Framework Directive. The main sources of emission to water are surface runoff and drainage of urban and agricultural areas as well as sewage treatment plants [106].

Chromium

Chromium may be present in the environment as Cr(III) and Cr(VI). Cr(III) is an essential element for human and animals, whereas Cr(VI) is highly toxic. In presence of organic substance Cr(VI) is reduced to Cr(III) in the environment.

Cr(III) is very immobile in the environment. Plants only take up little amounts of the metal via soil solution. Concerning human health issues slightly increased Chromium concentrations in plants would be favorable [101].

Copper

Copper is essential for all living organisms. Chronic effects on human health are rarely known. Nevertheless, in higher concentrations Copper can have phyto-toxic effects on plant and thus may be a hazard for terrestrial ecosystems [101].

Zinc

Zinc is essential for humans, animals and plants. At higher concentrations toxic effects on soil organisms were observed. Adverse effects on human health are currently not known [101].

Tolerable concentrations for human health

Concerning human health risk due to the intake of heavy metals via food, which is grown on the agricultural areas of the AVBS, critical limits in plants and soil are calculated, taking the safety intake parameters (see **Fehler! Verweisquelle konnte nicht gefunden werden.**) and food consumption data as a baseline. From these data tolerable food and soil concentrations are back-calculated.

Tolerable weekly intake values are given in µg/kg_{bw} (bw = bodyweight). An average bodyweight of 70kg is assumed [102]. The TDI value accounts for all exposure routes, including other than food consumption. Following the approach of the WHO [88] and UNECE Expert Meeting [102]the tolerable fraction via food intake is set to 50%. Values for the TWI and UL where taken from [107] and are shown in Table 3-12.

Table 3-12: safety parameters (tolerable weekly intake, upper intake level) for oral human intake for heavy metals [107].

Metals	TWI [µg/kg_{bw}]	UL [mg/day]	Other safety parameters [mg/d]	Publishing Institution	Tolerable daily intake [µg/d] (70kg/person)	Tolerable daily intake via food consumption [µg/d]
Cadmium	2.5			EFSA 2009	25	12.5
Chromium			1	VKM 2007	1000	500
Copper		5		SCF 2003	5000	2500
Lead	25			JECFA 2000	250	125
Mercury	5			JECFA 2003	50	25
Nickel	-	-	-		-	-
Zinc		25		SCF 2003	25000	12500

Data about the quantity of food consumption are taken from the national survey of food consumption in Germany (Nationale Verzehrstudie II) conducted by the Max-Rubner-Institute [108]. Consumption data are shown in Appendix B (section **Fehler! Verweisquelle konnte nicht gefunden werden.**). Wheat is taken as a proxy for all cereals, as it is grown in BS for bread production and because of its affinity of metal accumulation. De Vries at al. state that *"an appropriate indicator for critical load calculation addressing human health effects via food intake is the Cd content in wheat. Keeping a conservative food quality criterion for wheat [...] protects at the same time against effects on human health via other food and fodder crops (including also the quality of animal products), since the pathway of Cd to wheat leads to the lowest critical Cd content in soils"* ([109], p. 15, section 2.1, ll. 16-21).

Based on the mean consumption data an average wheat consumption of 400g/d per person is assumed and an amount of 600g/d for high consuming people (95-percentile). The fraction of heavy metals resorbed by the human body is set to 15% [102]. Thus, the effective amount of metals taken in via food consumption is calculated by:

$$Effective\ consumption = Total\ consumption * 0.15$$

Effective consumption = amount of cereal consumption, whose metal content is completely resorbed [g/d]

Total consumption = total cereal consumption [g/d]

The tolerable heavy metal content in wheat is calculated by dividing the tolerable intake via food consumption (see Table 3-12) through the effective consumption.

$$Tolerable\ wheat\ concentration = \frac{TDI\ (food)}{Effective\ consumption}$$

Tolerable wheat concentration = tolerable heavy metal concentration in wheat [$mg/kg_{freshweight}$]

TDI (food) = tolerable daily intake attributed to food consumption (see Table 3-12)

Effective consumption = amount of cereal consumption, whose metal content is completely resorbed [g/d]

In order to calculate tolerable soil concentrations, plant concentration were back-calculated via:

$$Tolerable\ soil\ concentration = \frac{Tolerable\ wheat\ concentration}{BCF}$$

Tolerable soil concentration = tolerable soil concentration for human health [$mg/kg_{soil}(dw)$]

Tolerable wheat concentration = tolerable heavy metal concentration in wheat [$mg/kg_{freshweight}$]

BCF = Bioconcentration factor

The bioconcentration factor (BCF) is defined as the ratio between plant and soil concentration. Since only cadmium plant concentrations are monitored in Braunschweig, BCFs of the remaining heavy metals were taken from literature [107].

Subsequently, the overall mean wheat concentration was divided by the overall mean soil concentration to determine the BCF. A dry matter content of 86% is applied for wheat (Ripke, personal correspondence).

Both soil and wheat concentrations are measured in Braunschweig. The measurement just takes place once a year. No information is available on the sampling and measurement program. Moreover, paired data of soil and wheat concentrations are just available for the years 1995-1999 and 2009-2010 for the respective pumping districts. Thus, additionally to the soil-wheat relation calculated by the application of a BCF, another soil-wheat relation formulated by De Vries et al. 2003 published in [102] is applied to account for present uncertainties.

$$\log(Csoil_{tol}) = \frac{\log\left(c(Cd)_{plant}\right) - 0.35 + 0.15pH + 0.39\log(OM)}{0.76}$$

$Csoil_{tol}$	= tolerable soil concentration [mg/kg$_{soil}$(dw)]
$c(Cd)_{plant}$	= cadmium concentration in wheat [mg/kg$_{wheat}$(fw)]
OM	= fraction of organic matter [%]

Based on these methods the following tolerable wheat and soil concentrations are calculated for average human consumption and high human consumption (Table 3-13 and 28).

Table 3-13: derived tolerable wheat and soil concentrations for average food consumption

Metal	Effective consumption (average) [g/d]	Critical wheat concentration [mg/kg$_{freshweight}$]	BCF	Critical soil concentration [mg/kg$_{dw}$]	Critical soil concentration [mg/kg$_{dw}$] (De Vries et al. 2003)
Cadmium	60	0.2	0.29	0.69	0.87
Chromium	60	8.3	0.017	490	
Copper	60	41.7	0.26	160	
Lead	60	2.1	0.0009	2314	
Mercury	60	0.4	0.013	32	
Nickel	60	-	0.06		
Zinc	60	208	0.17	1225	

Table 3-14: derived tolerable wheat and soil concentrations for high food consumption

Metal	Effective consumption (high) [g/d]	Critical wheat concentration [mg/kg$_{freshweight}$]	BCF	Critical soil concentration [mg/kg$_{dw}$]	Critical soil concentration [mg/kg$_{dw}$] (De Vries et al.)
Cadmium	90	0.14	0.29	0.48	0.54
Chromium	90	5.5	0.017	327	
Copper	90	27.8	0.26	106	
Lead	90	1.4	0.0009	1543	
Mercury	90	0.28	0.013	21	
Nickel	90	-	0.06		
Zinc	90	138	0.17	817	

Due to its low TDI and high plant uptake Cadmium shows the lowest tolerable soil concentration concerning human health risks. Moreover, the differences in the two different formulation of the soil-wheat relation become visible. The equation proposed by De Vries et al. 2003 leads to higher tolerable soil concentrations. Concerning the remaining metals, the highest tolerable soil concentration is attributed to lead, which is hardly taken up by plants. Moderate concentrations are attributed to Copper, Chromium and Zinc. Mercury shows tolerable soil concentrations significantly higher than the ones calculated for Cadmium. As no intake data for Ni was found no tolerable soil concentration could be calculated concerning human health impacts.

Tolerable concentrations for environmental endpoints

Figure 3-17 gives an overview of relevant receptors which are exposed directly or indirectly via the two ecosystems of concern, namely arable land and the aquatic ecosystem. PNECs and critical concentrations for the respective endpoints are collected from literature. For environmental risk assessment relevant receptors are soil microorganisms, soil invertebrates as well as mammals and birds. For the assessment of risks concerning soil microorganisms and soil invertebrates PNEC$_{soil}$ for the respective metals are used as the tolerable value. Mammals and birds are not directly exposed to metals but are exposed indirectly via the food chain. Schütze and Spranger back-calculated critical soil contents from acceptable daily intakes (ADI) for birds and mammals (Schütze and Spranger 2002). The badger was taken as a reference animal for worm eating mammals, whereas for the calculation of critical soil contents for impacts on worm eating birds the black-tailed godwit was chosen. Within this study the authors state that *"the only metal in which indirect impacts due to accumulation in the food chain may cause lower critical soil metal contents […] is Cd"* ([102], section 4.1.6, ll.11-12). Therefore, environmental risk assessment due to secondary poisoning of mammals and birds is reduced to Cadmium. For the other metals the PNEC$_{soil}$ values are considered to protect also higher trophic levels. Within the European risk assessment report for Cadmium an additional PNEC$_{soil}$ for the assessment of mammals and birds exposure is proposed [110]. Both are used and compared. For environmental impacts on the aquatic ecosystem, algae and crustacea are the receptors of concern. To assess risk on these aquatic organisms PNEC$_{water}$ values are collected.

Figure 3-17 gives an overview on the used literature for the respective PNECs and critical contents.

Table 3-15: sources used for the respective critical soil and water contents and concentrations

Metal	Source of $PNEC_{soil}$	Source of $PNEC_{water}$	Source of critical soil content for birds	Source of critical soil content for mammals
Cadmium	European Chemicals Bureau, 2007	[110]	[102], (ECB 2007)	[102], [110]
Chromium	European Chemicals Bureau, 2005	[107]		
Copper	European Copper Institute, 2008	[111]		
Lead	EURAS, 2008	[112]		
Mercury	Euro-Chlor, Voluntary Risk Assessment, Mercury, 2004	[107]		
Nickel	Danish Environmental Protection Agency, 2006	[113]		
Zinc	VROM, 2008	[114]		

The literature review on critical soil concentrations and predicted no effect concentrations (PNEC) for the respective endpoint led to the following values (Table 3-16).

Table 3-16: PNECs and critical concentrations for the different environmental endpoints of concern

Metal	$PNEC_{soil}$ [mg/kg$_{soil}$(dw)]	$PNEC_{water}$ [µg/L]	Critical soil content for birds (black tailed godwit) [mg/kg$_{soil}$(dw)]	critical soil content for mammals (Badger) [mg/kg$_{soil}$(dw)]	$PNEC_{soil}$ for mammals and birds (EU) [mg/kg$_{soil}$(dw)]
Cadmium	1.15	0.08	0.14*	0.067*	0.9**
Chromium	62	3.4			
Copper	89.6	7.8			
Lead	166	7.2			
Mercury	0.3	0.047+BC			
Nickel	50	5			
Zinc	26+BC	7.8+BC			

*calculated by Schütze et al. 2002

**taken from the European Risk Assessment Report for Cadmium Metal [110]

For cadmium impacts on soil organisms and soil invertebrates lead to the highest tolerable concentrations ($PNEC_{soil}$). The tolerable values the European risk assessment report on Cadmium published for animals is approximately 8 times higher than the ones published by Schütze et al. 2002.

For soil organisms as well as for surface waters, Mercury and Cadmium show the lowest PNECs. The highest are attributed to Zinc and Copper.

Exposure assessment

Tolerable risks for human health concerning heavy metal exposure are a back-calculated to soil and wheat concentrations. For the environmental endpoints soil microorganisms, soil invertebrates, bird and mammals tolerable risks levels via direct and indirect environmental exposure are expressed as critical soil contents as well. Thus, the calculation of soil concentration is the essential step in risk assessment via the terrestrial compartment. Heavy metals are not biodegradable and tend to accumulate in soil. Therefore, the environmental risk is considered to be tolerable if the modeled concentrations and contents do not exceed the PNEC within a hundred years. This approach follows a risk assessment conducted by the Norwegian Scientific Committee for Food Safety [107]. The 100 years are chosen since this time frame seems to be still imaginable and manageable.

Determination of annual loads

The first step of exposure modeling is the calculation of the average annual loads of the respective metal, which are distributed on the agricultural areas. The measured monitoring data for heavy metals are first checked for plausibility and consistency. For this purpose simplified mass balances for the respective metals are calculated and the ratio between effluent to influent loads determined.

$$Cinf * \dot{V}inf = Ceff * \dot{V}eff + Cps * Mps + Css * Mss$$

Cinf = annual mean influent concentration [mg/L]

Vinf = influent rate [m³/year]

Ceff = annual mean effluent concentration [mg/L]

Veff = effluent rate [m³/year]

Cps = annual mean concentration in primary sludge [mg/kg_{sludge}(dw)]

Mps = mass of produced primary sludge [kg_{sludge}(dw)/year]

Css = annual mean concentration in surplus sludge [mg/kg_{sludge}(dw)]

Mss = mass of produced surplus sludge [kg_{sludge}(dw)]/year]

The ratios between influent to effluent loads are shown in Figure 3-19.

Table 3-17: ratios between effluent and influent loads based on the measured monitoring data in STP Steinhof

Metal	Effluent/influent ratio [%]
Cadmium (Cd)	124
Chromium (Cr)	91

Copper (Cu)	87
Lead (Pb)	54
Mercury (Hg)	532
Nickel (Ni)	89
Zinc (Zn)	106

The results are regarded as plausible if the calculated ratio is between 85 and 115%. For the metals Cd, Cr, Cu, Ni and Zn, thus, measured data were used for further calculations.

Because of its high toxicity for Cd effluent measurements are used. The large gap in the Hg mass balance can be explained by the low concentrations in the effluent of the STP Steinhof as well as in the primary sludge. Both fall below the limit of quantification. One has to mention that the latest data which were available for this study were from 2010. The mass balances for Cd, Hg and Pb for 2011 would fulfill the plausibility criterion.

For Pb and Hg annual loads are therefore calculated based on measured influent concentrations. Effluent and sludge concentration were modeled using the formulas from the TGD ([96], part 2, section 2.7.1). Calculations are based in measured influent concentrations.

$$Ceff = Cinf * Fstp_{water}$$

Ceff = concentration in the effluent of the STP [mg/L]

$Fstp_{water}$ = fraction of emission directed to water by STP []

$$Csludge = \frac{Fstp_{sludge} * Erate_{water} * 10^6}{sludgerate}$$

Csludge = metal concentration in sewage sludge [mg/kgdw]

$Fstp_{sludge}$ = fraction of emission directed to sewage sludge by STP []

$Erate_{water}$ = metal emission to water [kg/d]

Sludgerate = rate of sewage sludge production [kg_{sludge}(dw)/d]

$$Erate_{water} = \frac{Cinf * \dot{V}inf}{365}$$

$Erate_{water}$ = metal emission to water [kg/d]

Cinf = annual mean influent concentration [mg/L]

Vinf = influent rate [m³/year]

$$sludgerate = \frac{Mps + Mss}{365}$$

Sludgerate = rate of sewage sludge production [kgdw/d]

Mps = mass of produced primary sludge [kgdw/year]

Mss = mass of produced surplus sludge [kgdw/year]

Values for the fractions of emission directed to wastewater and sewage sludge, respectively, are taken from ([115], p.45)

Table 3-18: fractions of Lead and Mercury directed to sewage sludge after primary sedimentation and activated sludge treatment [115].

Metal	$Fstp_{sludge}$ [%]
Lead	70
Mercury	80

Table 3-19 shows the calculated annual heavy metal loads in the STP Steinhof.

Table 3-19: mean calculated annual load for the STP Steinhof. Influent loads of all heavy metals as well as the annual cumulative loads directed to irrigation for Cadmium, Chromium, Copper, Nickel and Zinc are based on mean measured data. Values for Lead and Mercury were calculated using the models outlined in section 0.

Metal	Influent load [kg/a]	Effluent load to irrigation [kg/a]	Sludge load to irrigation [kg/a]	Annual load [kg/a]
Cadmium	10.8			5.5
Chromium	226.2			90.9
Copper	1837			828.2
Lead	491.5	48.6	229.8	278.4
Mercury	6.7	0.9	2.9	3.8
Nickel	357			94.8
Zinc	5112			2381.8

Calculation of soil concentrations

Based on the annual heavy metal loads soil concentrations are calculated by using the equations of the Technical Guidance Document model (TGD)([96], part 2, section 2.3.8.5). The development of soil concentrations of heavy metals is calculated for a time period of a hundred years.

Once released into the soil environment the behavior of heavy metals strongly depends on the environmental conditions. Metal mobility and availability determine to which amounts metals are taken up by plants, being leached into the groundwater or accumulate in the soil. Mobility and availability in turn are depended on both the physical-chemical properties of the respective metal and the surrounding environmental conditions. Therefore, the surrounding conditions have to be determined. The Technical Guidance Document includes a set of default values for environmental conditions for calculating soil concentrations. As far as local data for the respective parameters are available for Braunschweig, the default data are replaced. Values and sources are presented in Table 3-20.

Table 3-20: values for surrounding conditions used for the calculation of soil concentrations. Moreover, the respective symbols used in the following calculations and data sources are outlined.

Parameter	Symbol	Value	Unit	Source
Rain rate	*RAINrate*	599	mm/a	Climate data Braunschweig
Temperature	*T_C*	9.2	°C	Climate data Braunschweig
Soil pH	*pH*	5.9		Measured data, [97]
Bulk density of soil	rho_{soil}	1700	kg/m³	([96], section 2.3.4)
Mixing depth of soil	$Depth_{soil}$	0.2	m	([96], section 2.3.8.5)
Infiltration rate of rain into soil	$Finf_{soil}$	0.25		([96], section 2.3.8.5)
Fraction of organic carbon is soil	*Foc*	0.9	%	Measured data, [97]

In order to calculate soil metal concentrations annual inputs and outputs are taken into account.

Inputs

The two main inputs of heavy metals in the conducted calculations are the annual loads applied via wastewater and sewage sludge and atmospheric deposition.

The TGD does not consider permanent wastewater irrigation but solely refers to sludge application. Therefore, for the calculation of soil concentration it is assumed that the total heavy metal load is present in sewage sludge which is applied once in the beginning of the each year. The increase of soil concentrations due to sludge application is calculated by:

$$Csludge_{soil} = \frac{Load_{Metal}}{Depth_{soil} * rho_{soil}}$$

$Csludge_{soil}$ = increase of soil metal concentration due to 1 year of sludge application [mg/kg]

$Depth_{soil}$ = mixing depths of soil [m]

rho_{soil} = bulk density of soil [kg/m³]

$Load_{Metal}$ = annual metal load [mg/m²*a]

The increase of soil concentration due to atmospheric deposition is calculated by:

$$Dair = \frac{DEPtotal_{ann}}{Depth_{soil} * rho_{soil}}$$

$Depth_{soil}$ = mixing depths of soil [m]

Rho_{soil} = bulk density of soil [kg/m³]

Dair = aeral deposition flux per kg of soil [mg/kg*d]

$DEPtotal_{ann}$ = annual average total deposition flux [mg/m²*d]

Values for the total annual deposition in Germany are shown in Table 3-21.

Table 3-21: annual atmospheric deposition of heavy metals in Germany

Source	Cd [g/ha*a]	Cr [g/ha*a]	Cu [g/ha*a]	Hg [g/ha*a]	Ni [g/ha*a]	Pb [g/ha*a]	Zn [g/ha*a]
[98]	2	5	30	0.2	15	40	250

Outputs

The TGD model considers biodegradation, volatilization and leaching as the main output fluxes for chemicals. As metals are neither biodegradable and (except from some single organic Hg-compounds) not volatile those outputs are set to zero.

Thus, the overall output constant k is calculated by:

$$k = kleach$$

k = *first order rate constant for removal from top soil* [d^{-1}]

kleach = *pseudo-first order rate constant for leaching from top soil* [d^{-1}]

kleach is calculated by:

$$kleach = \frac{Finf_{soil} * RAINrate}{Kd_{soil-water} * Depth_{soil} * rho_{soil} * 10^{-3}}$$

$Finf_{soil}$ = *fraction of rainwater that infiltrates into soil* []

RAINrate = *rate of wet precipitation* [m/d]

$Kd_{soil\text{-}water}$ = *soil-water partitioning coefficient* [L/kg_{soil}]

$Depth_{soil}$ = *mixing depth of soil* [m]

kleach = *pseudo-first order rate constant for leaching from top soil* [d^{-1}]

Partitioning coefficients are taken from literature (Table 3-22).

Table 3-22: Partitioning coefficients used for calculating leaching processes from top soil

Metal	$Kd_{soi\text{-}water}$[L/kg]	Source
Cd	280	[110]
Cr	3000	[107]
Cu	Log Kd = 1.75 + 0.21pH + 0.51log(Foc)	[111]
Hg	3000	[107]
Ni	Log Kd = 2.86	[113]

Pb	6400	[112]
Zn	Log Kd = 3.07	[114]

Irrigation represents a water flux additional to the annual rain rate. Therefore, the amount of irrigated water is added to the average annual rain rate

The overall soil concentration over the year is calculated by combining input and outputs via:

$$Csoil(t) = \frac{Dair}{k} - \left[\frac{Dair}{k} - Csoil(0)\right] * e^{-kt}$$

with

$$Csoil(0) = Csludge_{soil} + Cinitial$$

Csoil(0) = Soil metal concentration in the beginning of the year after sludge application [mg/kg_{dw}]

$Csludge_{soil}$ = increase of soil metal concentration due to sludge application [mg/kg_{dw}]

Cinitial = soil concentration before the first sludge application [mg/kg_{dw}]

For the determination of the initial soil metal concentration measured monitoring data of the four pumping districts are used. The overall mean of the available data is taken as the initial soil concentration. As mentioned above the pumping districts are subdivided into 5-7 smaller areas, respectively. In the pumping districts I and II the single areas show comparable mean soil concentrations. In pumping district III two out of 7 areas show elevated heavy metal concentrations, which are already above the limit value for sludge application of 1mg/kg_{soil}(dw). Moreover, in pumping district IV one out of 5 areas shows elevated heavy metal concentrations as well. On this area not only the limit for Cd but also the ones for Pb and Zn are exceeded.

The elevated soil concentrations in pumping district III originate from the past, as no wastewater treatment had been in place, yet. In this time wastewater was stored on these areas for the settlement and thus the removal of solid fractions prior to irrigation. On area 5 of pumping district IV the elevated concentrations have a different origin. The metal concentrations in this area are increased because it lies within the flooding area of the river Oker (Ripke, personal correspondence). It happens to be that this river has its source in the Harz Mountains, where extensive mining for metal ores in the past still causes high metal concentrations in the river and its sediment.

Nevertheless, in environmental and human health risk assessment all relevant inputs have to be considered. Statements have to be based on soil concentrations, independently from their origin.

Therefore, the pumping districts III and IV are treated differently from pumping districts I and II. In addition to using mean soil concentrations of the whole district, future soil concentrations are calculated for each single area, using the area specific annual mean concentration, respectively. Table 3-23 shows the mean measured metal concentrations in top soil in the single pumping districts as well as the values of the single areas.

Table 3-23: overall mean heavy metal concentrations in top soil for the respective pumping districts and areas: Highlighted (fat, cursive) values indicated that the respective value exceeds the legislative limit value for sludge application

Pumping district	area	Pb	Cd	Cr	Cu	Ni	Hg	Zn
I	1	16.20	0.36	10.00	7.80	5.50	0.07	37.78
I	2	13.60	0.36	8.60	6.70	4.70	0.06	32.30
I	3	15.20	0.35	8.90	7.20	5.00	0.07	37.20
I	4	17.56	0.71	12.00	10.78	7.00	0.11	54.56
I	5	16.60	0.43	10.60	8.80	7.10	0.07	44.40
Pumping district	area	Pb	Cd	Cr	Cu	Ni	Hg	Zn
II	1a	11.20	0.43	7.40	9.30	4.40	0.07	35.30
II	1b	11.00	0.39	7.60	9.30	4.10	0.07	35.70
II	1c	11.90	0.35	6.50	8.50	3.80	0.07	31.10
II	2	12.30	0.51	9.20	11.00	4.20	0.07	41.70
II	3	16.90	0.41	10.00	8.55	5.82	0.07	45.91
II	4	14.70	0.31	9.00	8.70	6.20	0.05	39.90
II	5	13.00	0.23	8.00	6.10	5.60	0.04	35.40
Pumping district	area	Pb	Cd	Cr	Cu	Ni	Hg	Zn
III	1a	20.40	***1.11***	11.90	18.40	6.80	0.18	63.50
III	1b	20.70	***1.15***	11.40	19.90	6.60	0.20	61.10
III	2a	14.90	0.55	9.10	13.40	5.30	0.11	45.40
III	2b	17.30	0.84	11.80	18.30	7.10	0.17	62.70
III	3	14.44	0.76	9.89	11.56	6.11	0.10	46.44
III	4	11.60	0.48	8.50	8.10	4.70	0.07	35.20
III	5	17.00	0.32	11.40	9.90	5.80	0.07	39.10
Pumping district	area	Pb	Cd	Cr	Cu	Ni	Hg	Zn
IV	1	10.20	0.20	5.10	4.00	2.70	0.05	20.50
IV	2	12.90	0.37	7.80	9.00	3.80	0.06	28.90
IV	3	15.30	0.33	8.50	10.20	4.60	0.06	35.30
IV	4	13.80	0.29	7.80	7.90	4.20	0.06	32.20
IV	5	***157.30***	***2.20***	28.00	41.20	21.60	0.18	***1107.00***

Calculation of PECsoil for terrestrial ecosystems and plant uptake

Sludge application in this model is treated as a single event in the beginning of the year. Soil concentration changes over the year as leaching and atmospheric deposition are continuous fluxes. Thus, an average value has to be determined. This average concentration is defined as the average concentration over a certain time period. The time period depends on the respective endpoint. For calculating the PECsoil, which is the endpoint concentration for terrestrial ecosystems, birds and mammals 30 days are chosen. For plant uptake an average time of 180 days is applied. The endpoint-specific soil concentrations are calculated by:

$$PECsoil(T)_{endpoint} = \frac{Dair}{k} + \frac{1}{kT}\left[Csoil(0) - \frac{Dair}{k}\right] * \left[1 - e^{-kT}\right]$$

Dair	= aeral deposition flux per kg of soil [mg/kg*d]
k	= first order rate constant for removal from top soil [d^{-1}]
T	= endpoint specific averaging time [d]
$PECsoil(T)_{endpoint}$	= predicted environmental soil concentration for the respective endpoint [mg/kg_{dw}]

Calculation of PECwater due to surface runoff

As mentioned above environmental exposure assessment calculates concentrations instead of doses. The maximum concentration of surface waters as a result of any discharge is the concentration of the discharge itself. If the initial concentration of the surface water is already above the concentration of the respective discharge, the discharge would lead to dilution and thus to a reduction of the general concentration. Since this is rarely the case the TGD model assumes a default value for dilution of 10 (see [96], section 2.3.8.3). For Zn, Ni, Cu and Cr measured concentration from STP Steinhof for irrigation are used. For Cd, Hg, and Pb the calculated values are used. The annual load of the respective metal is divided by the annual amount of water and sludge as a first estimate for the concentration in irrigation water. The calculation assumes that sewage sludge has a density of 1kg/L.

Since only the dissolved fraction of the respective metal has toxic effect on water organisms, the partitioning between solids and water has to be considered (Kp_{susp}) as well as the amount of suspended matter in the receiving water body ($SUSP_{water}$). For the latter one the default value of the TGD of 15mg/L is used (see [96], section 2.3.8.3). The partitioning coefficients for the respective metal are taken from literature (Table 3-24).

Table 3-24: used partitioning coefficients for heavy metals in surface water

Metal	Kp_{susp} [L/kg]	Source
Cd	130000	[103]
Cr	150000	Assumed to be comparable to Hg
Cu	30246	[111]
Hg	150000	[104]
Ni	$10^{4.42}$	[113]
Pb	$10^{5.34}$	[112]
Zn	81000	[114]

In PECwater due to surface runoff from agricultural areas is calculated by:

$$PECwater = \frac{Cwater_{irr}}{(1 + Kp_{susp} * SUSP_{water} * 10^{-6}) * DILUTION}$$

PECwater = predicted environmental concentration in surface water [mg/L]

Kp_{susp} = solids-water partitioning coefficient of suspended matter [L/kg]

SUSPwater = concentration of suspended matter in the river [mg/L]

DILUTION = dilution factor

$Cwater_{irr}$ = concentration of the metal in irrigation water [mg/L]

Metal concentrations in surface water

Table 3-25 shows the calculated surface water concentrations due to surface runoff from agricultural areas.

Table 3-25: Calculated surface water concentrations due to surface runoff from agricultural areas

Metal	PEC_{water} [µg/L]
Cadmium	0.015
Chromium	0.22
Copper	4.4
Lead	0.47
Mercury	0.0085
Nickel	0.66
Zinc	8.4

Risk characterization

Risk characterization for humans and environmental endpoints is conducted by calculating the risk quotient for the respective endpoints. Table 3-26 summarizes the used concentrations which are used for risk characterization.

Table 3-26: overview of the PECs, PNECs and critical concentrations (CC) used for the calculation of Risk Quotients (RQ) for the respective human and environmental endpoints

Endpoint/receptor	Risk quotient
Humans average consumption (hac)*	$RQ = \frac{PEC_{soil}180}{CC_{hac}}$
Humans high consumption (hhc)*	$RQ = \frac{PEC_{soil}180}{CC_{hhc}}$
Soil organisms	$RQ = \frac{PEC_{soil}30}{PNEC_{soil}}$
Birds (Schütze et al.)*	$RQ = \frac{PEC_{soil}30}{CC_{brd}}$
Mammals (Schütze et al.)*	$RQ = \frac{PEC_{soil}30}{CC_{mm}}$
Animals (EU)	$RQ = \frac{PEC_{soil}30}{PNEC_{soil}(animals)}$
Algae and crustacea	$RQ = \frac{PEC_{water}}{PNEC_{water}}$

*CC = critical concentration

Risk expressed as risk quotients

This section characterizes the environmental and human health risk with respect to heavy metals using Risk Quotients (see section 0). As mentioned above different averaging times are used for calculating soil concentrations for environmental and human assessment. Since there is just a slight difference between the two calculated soil concentrations just the soil concentration averaged over 30 days is shown in the following figures. The values for both soil concentrations are shown in Annex IV (section **Fehler! Verweisquelle konnte nicht gefunden werden.**).

Risk characterization concerning the terrestrial compartment

Cadmium soil concentrations as well as the points at which the calculated risk quotients for the respective endpoints exceed a value of 1 are illustrated in Figure 3-32, Figure 21, Figure 22 and Figure 23. Figure 21 shows soil concentrations in contrast to tolerable values concerning human health. Figure 22 focuses on environmental endpoints. The figures 23 and 24 show the area specific soil concentration of the pumping districts III and IV against tolerable human health values.

Concerning pumping district I and II Risk Quotients for birds and mammals from Schütze et al. are exceed the value of 1 from the beginning. The PNECs for soil concerning animals and soil organisms calculated by the European Risk assessment Report are not exceeded over 100 years irrigation. Concerning human consumption the current concentration is below both, the critical soil concentration for high and average consumption and show a stable or decreasing development.

Concerning the mean concentration of pumping district III the Risk Quotients for birds (R_{brd}) and mammals (R_{mm}) from Schütze et al. exceed the critical value of 1 at the beginning. The critical value for animals and soil organisms from the European Risk assessment Report are not exceeded and show in decreasing trend. However, if the areas are investigated separately, area 1 and 1b exceed the critical value from the European Risk assessment Report for animals and does not fall below this value, even not in a hundred years.

When it comes to human health effects the areas 1 and 1b exceed all derived critical soil concentrations for average and high consumption. The trend is decreasing but does not fall below any critical value in 100 years. The initial concentrations on area 2b and 3 lies between the two derived critical concentrations for average human consumption. They are both below the critical concentration derived with the DeVries soil-wheat relation and show a decreasing tendency. Nevertheless, they do not fall below the critical concentration for average human consumption derived with the BCF soil –plant relation. Areas 2a and 4 are not relevant for average human consumption. Area 4 falls below all critical values within 100 years. Area 2a falls below the critical value derived with the DeVries- relation but stays above the one derived with the BCF method. The concentrations on area 5 are not relevant for human health.

Concerning pumping district IV the areas 1-5 exceed the critical soil concentrations for birds and mammals from Schütze et al.. Concerning the remaining critical concentrations for environmental endpoints but also for human health areas 1-4 do not exceed any critical value within 100 years period. In contrast, area 5 exceeds all of them significantly and does not fall below any of them in 100 years.

Concerning lead, copper, Chromium, mercury and nickel in non of the pumping districts wastewater irrigation leads to Risk Quotients ≥ 1 over 100 years wastewater irrigation (Figure 3-36 to Figure 3-40).

Concerning zinc the Risk Quotient for soil organisms is exceeded in pumping district IV from the beginning if area 5 is included in the calculations. If not, the exceeding occurs in a time period of 70 years. In the pumping districts I, II, and III Risk Quotients for soil organisms are exceeded within 10 to 35 years. Risk Quotients for human health are not exceeded.

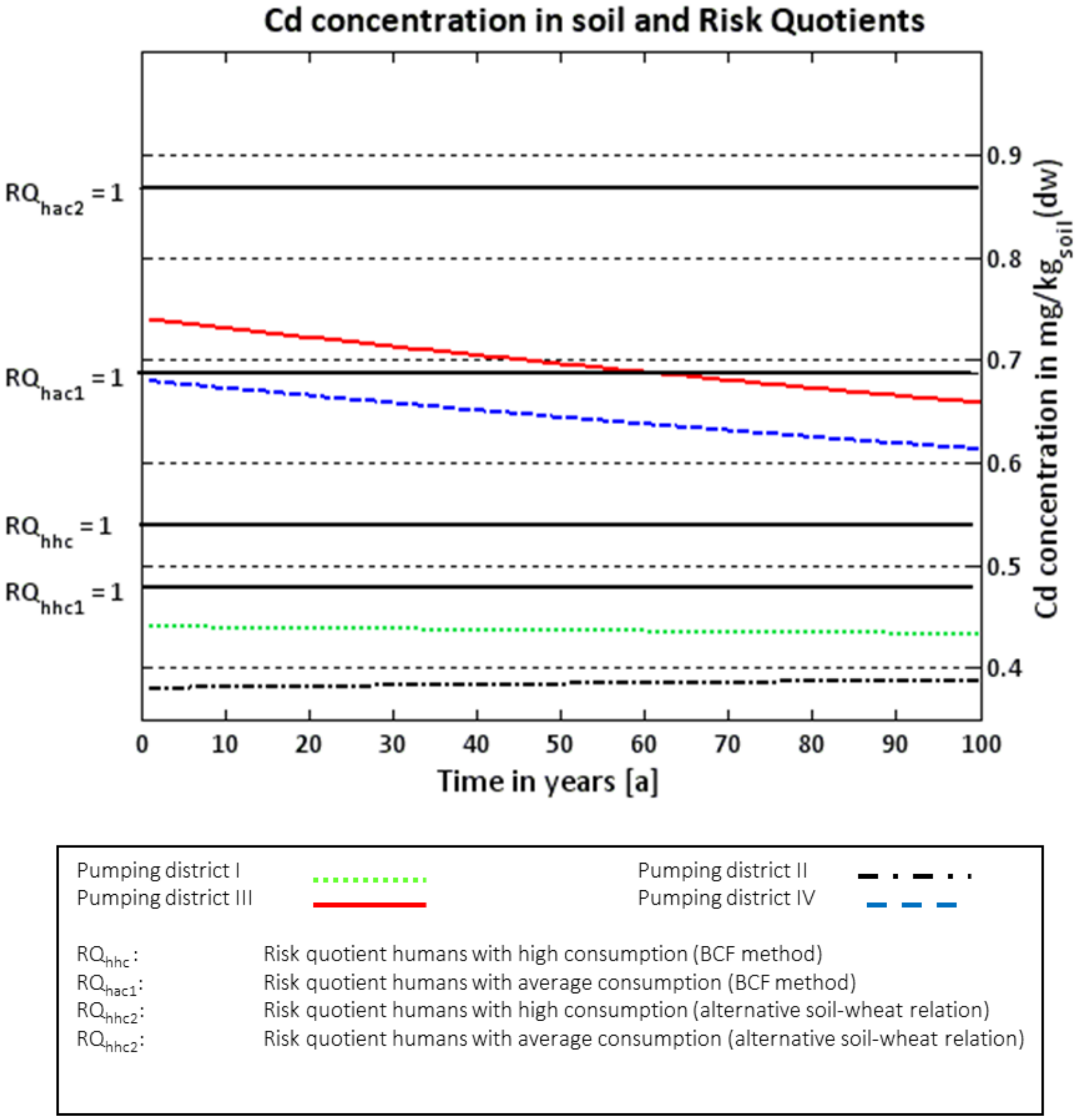


Figure 3-32: Cd concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective human endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

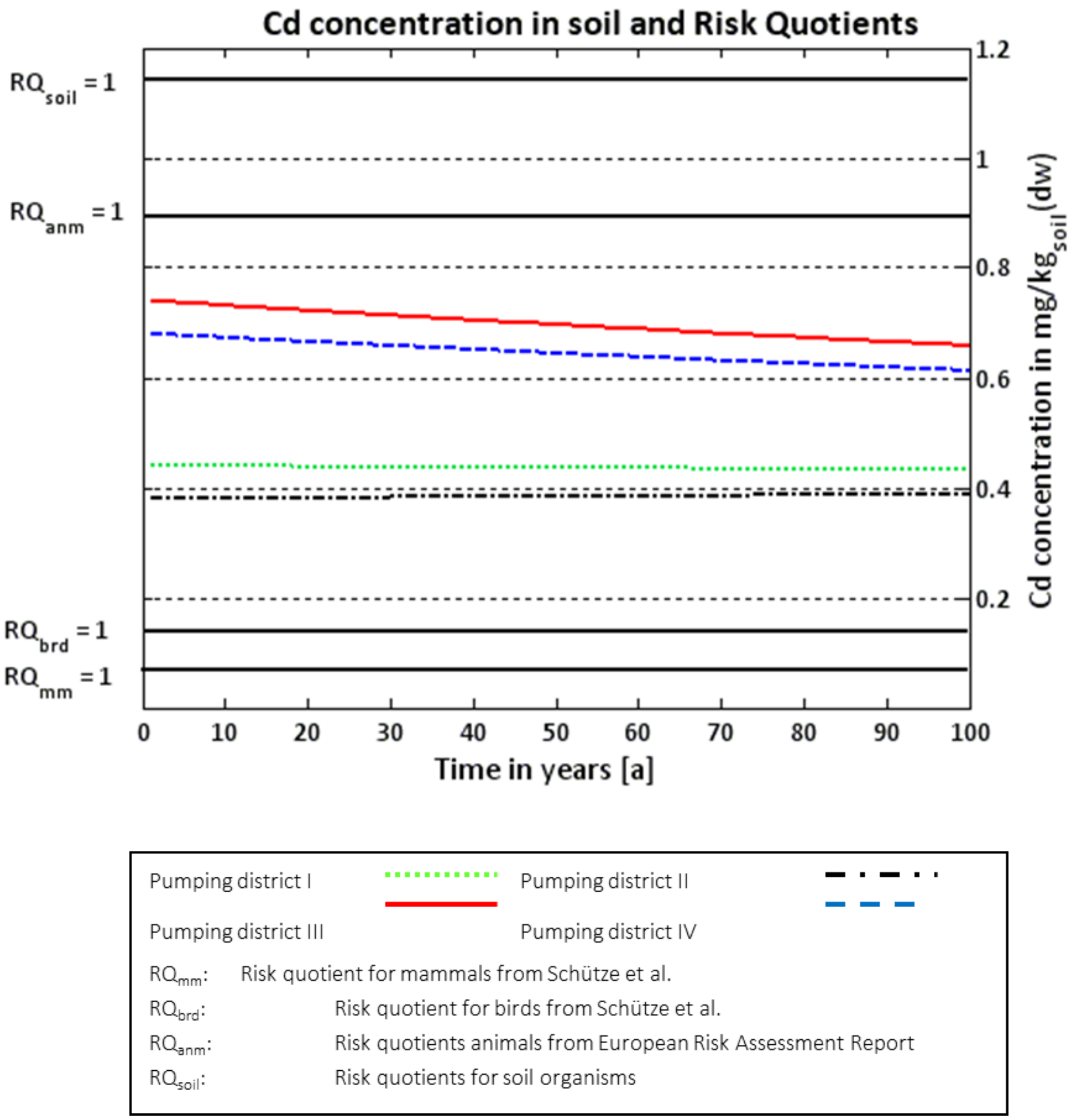


Figure 3-33: Cd concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective environmental endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

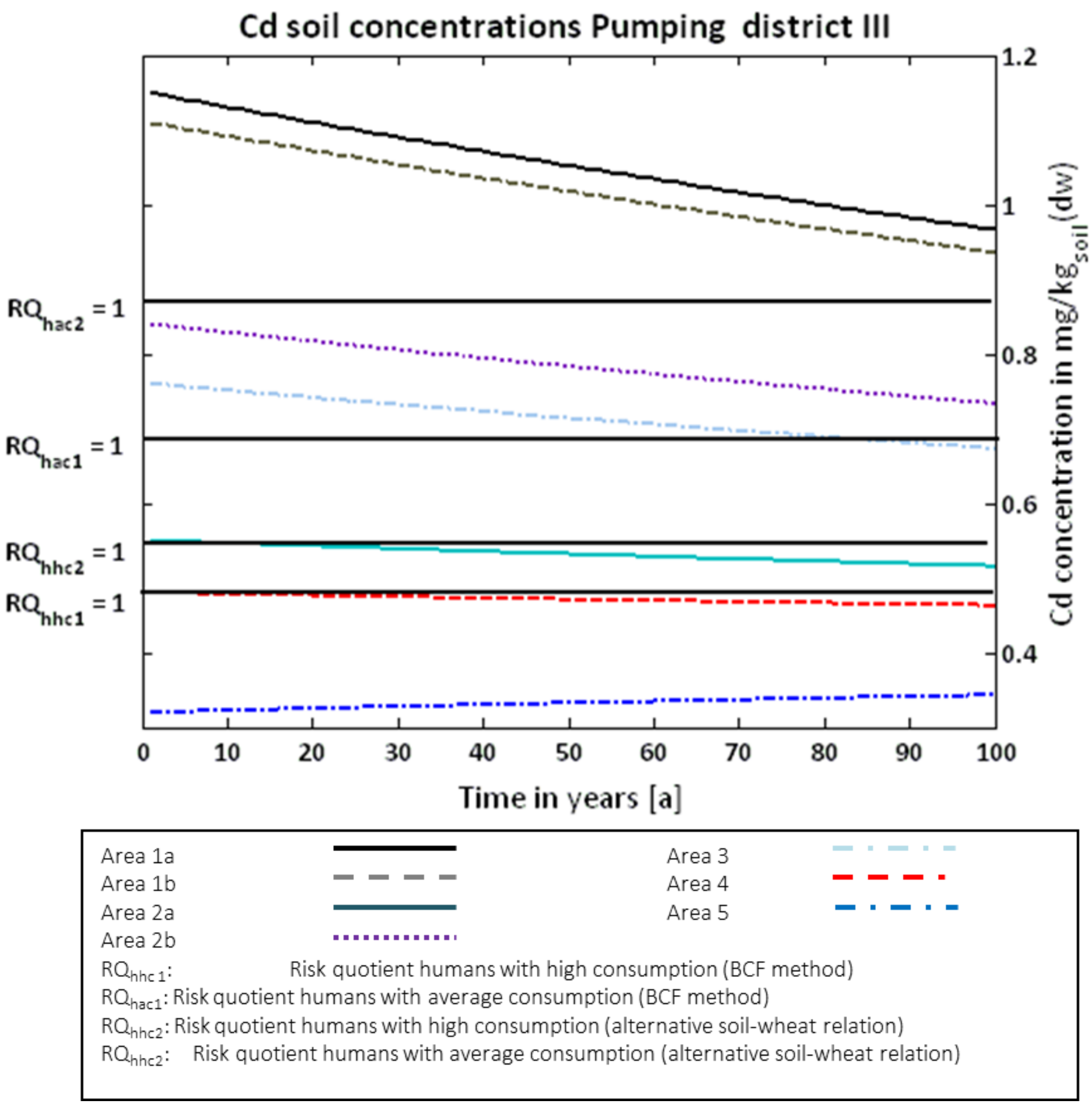


Figure 3-34: Cd concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective human endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

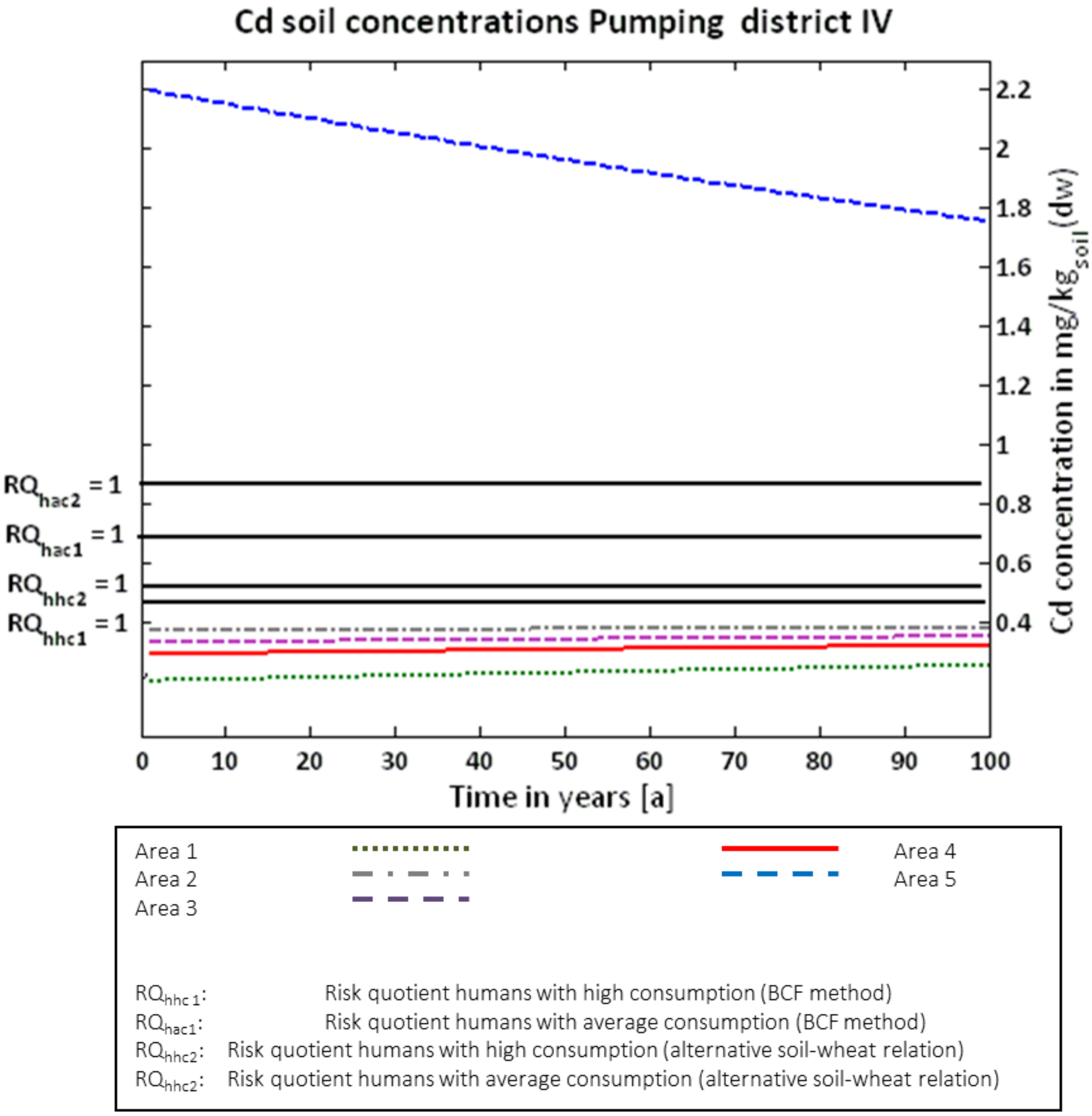
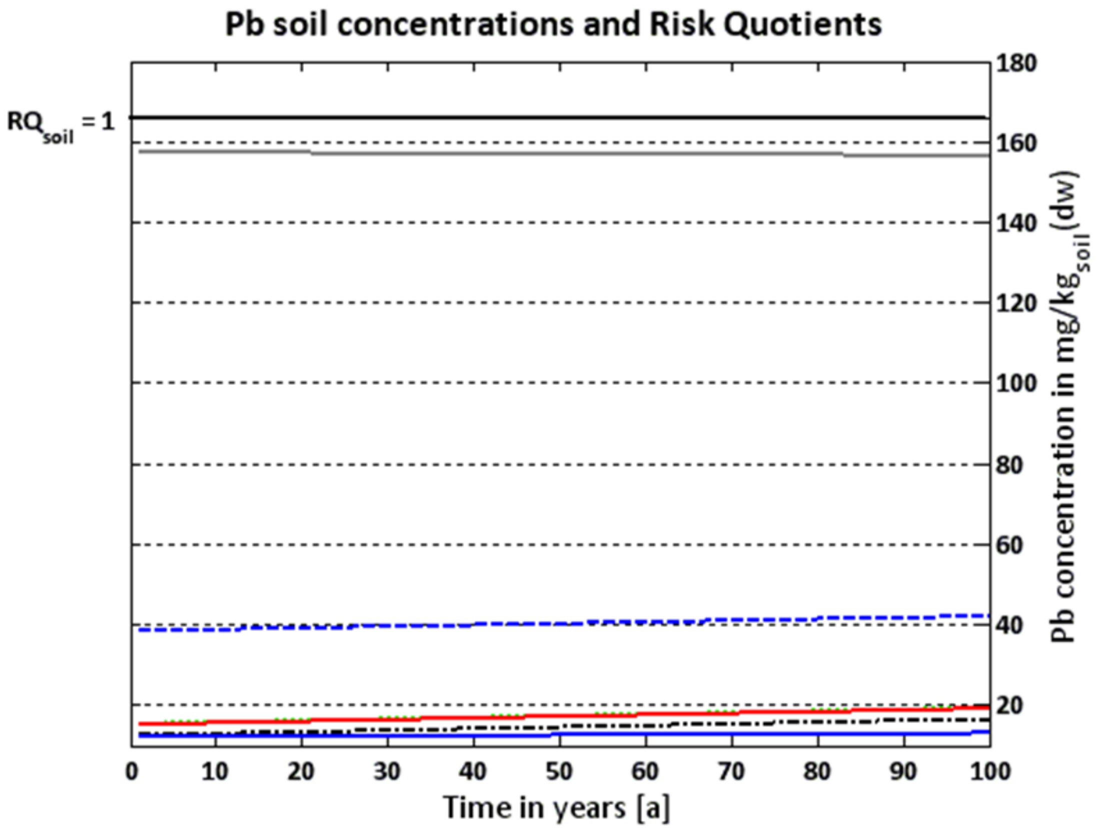


Figure 3-35: Cd concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective human endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.



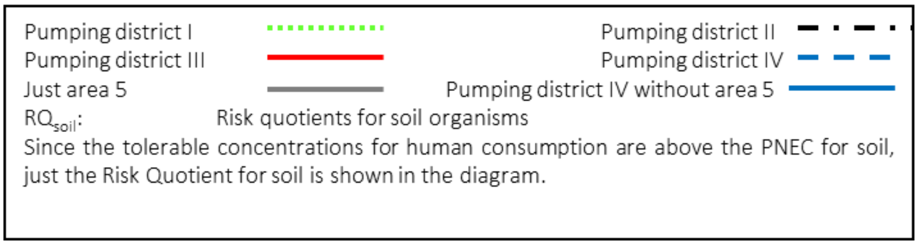


Figure 3-36: Pb concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

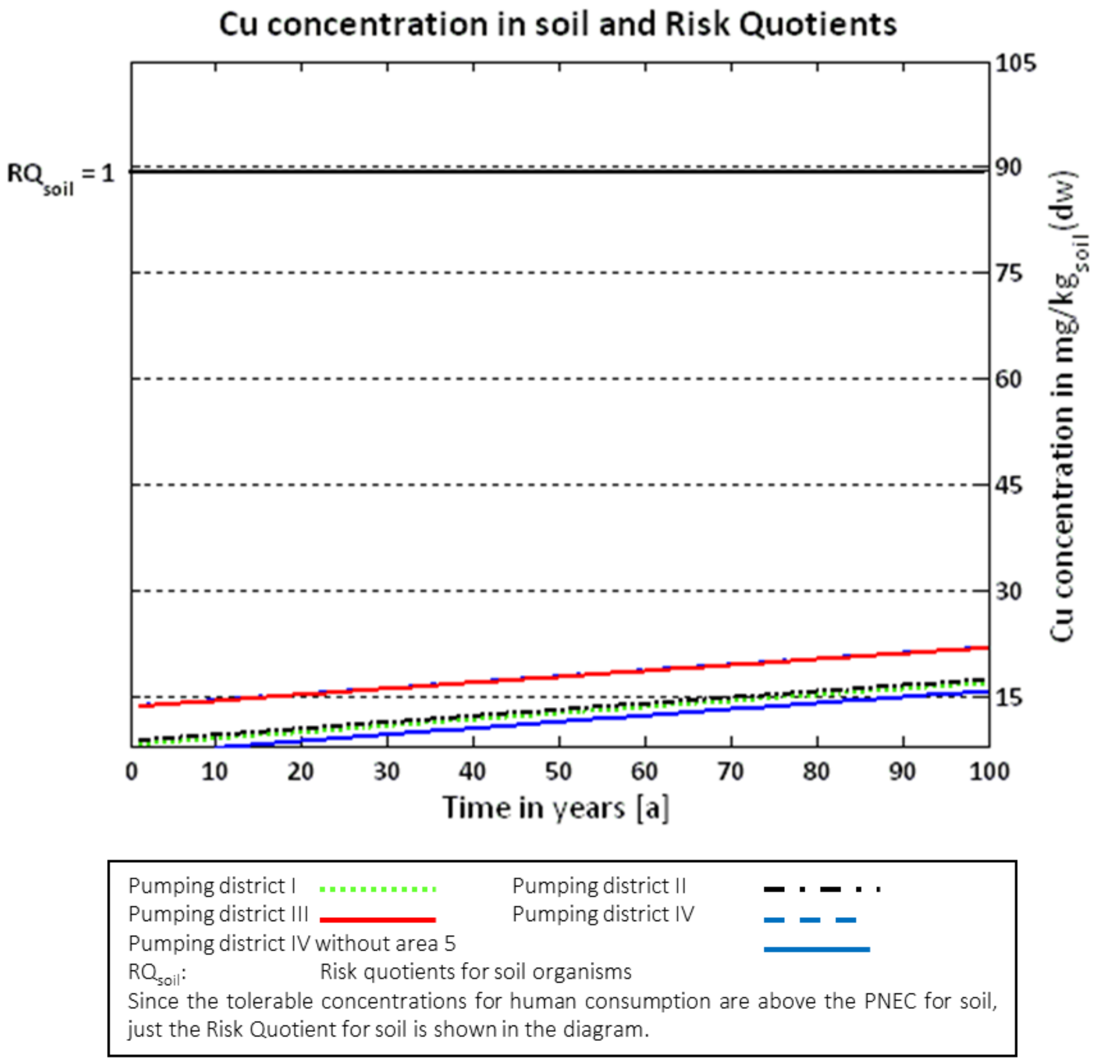


Figure 3-37: Cu concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

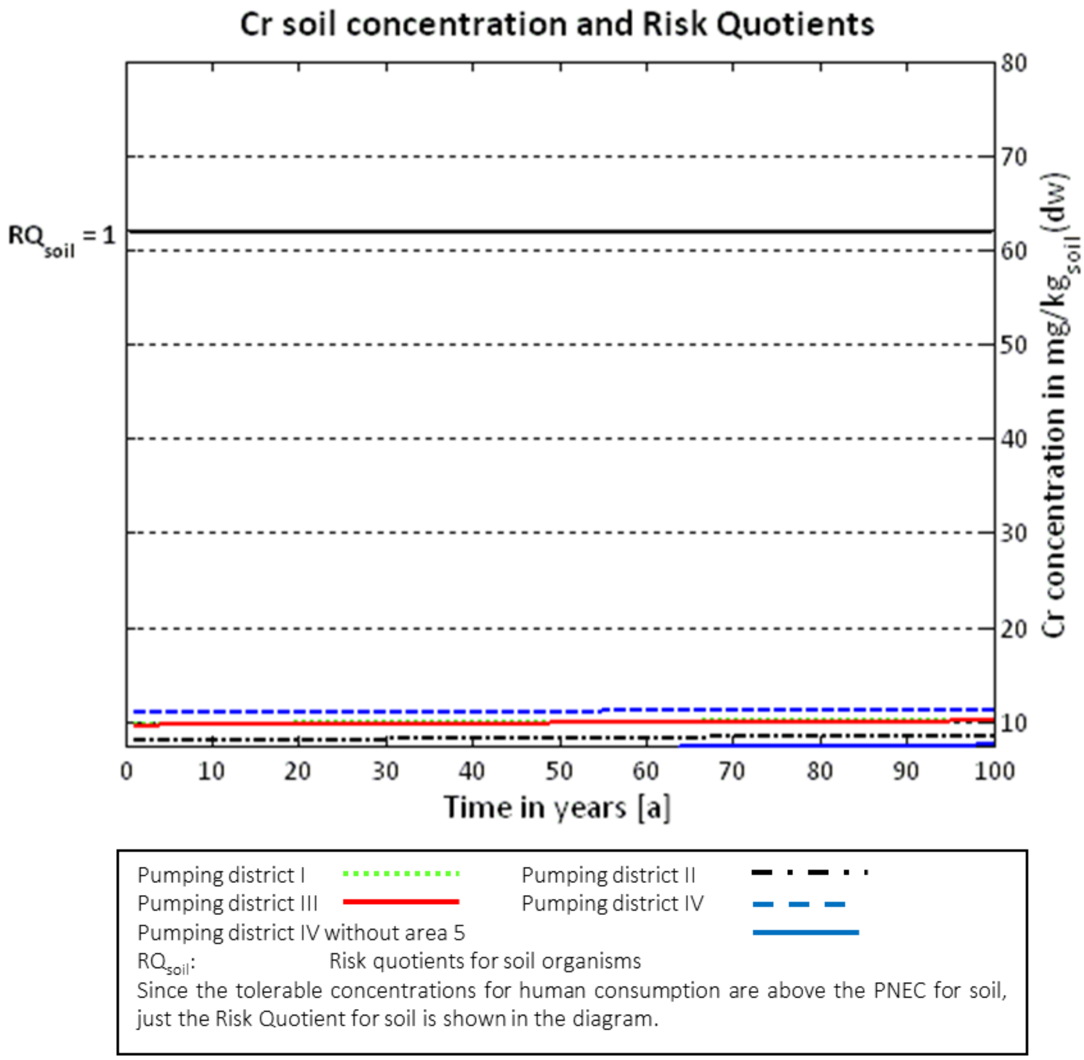


Figure 3-38: Cr concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

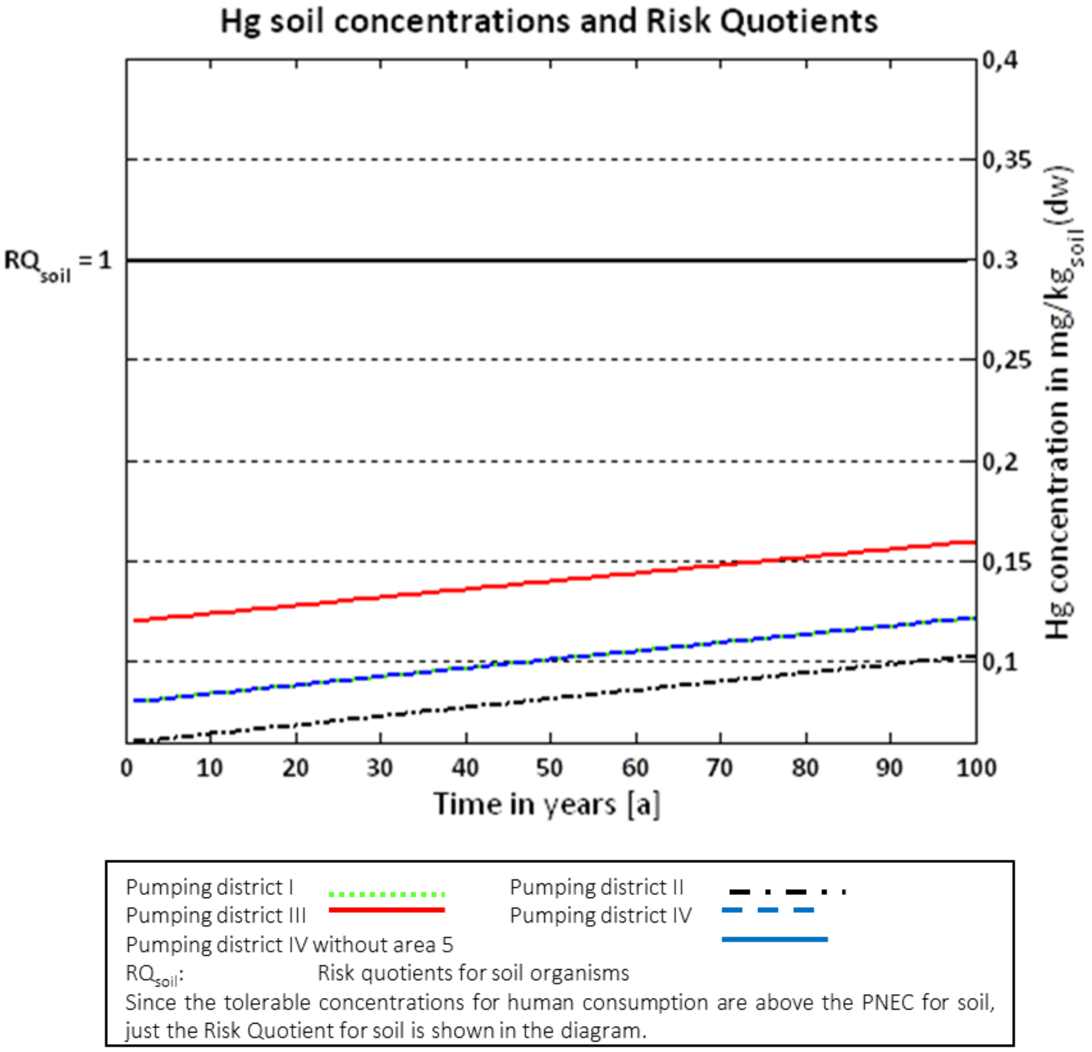
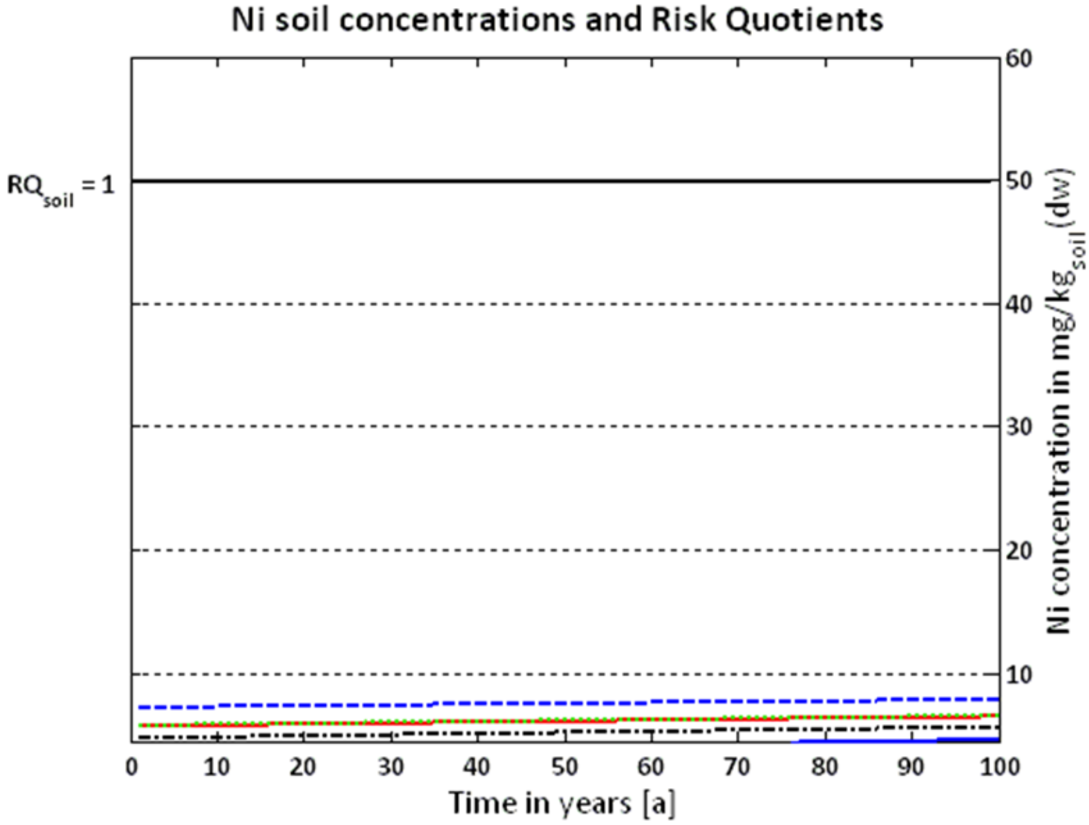


Figure 3-39: Hg concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.



Pumping district I
Pumping district II
Pumping district III
Pumping district IV
Pumping district IV without area 5
RQ_{soil}: Risk quotients for soil organisms
Since the tolerable concentrations for human consumption are above the PNEC for soil, just the Risk Quotient for soil is shown in the diagram.

Figure 3-40: Ni concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

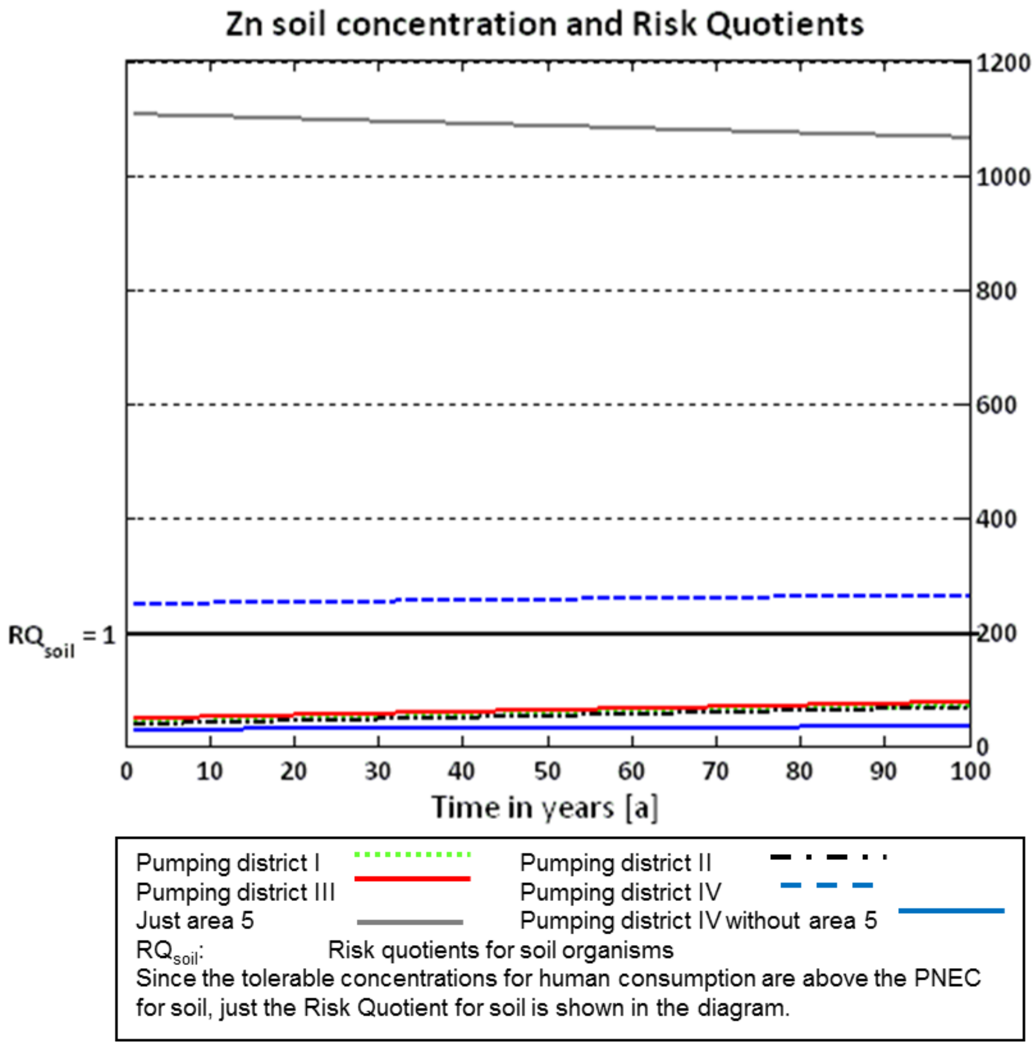


Figure 3-41: Zn concentrations in top soil over a hundred years time period. On the right x-axis the points are shown at which the Risk Quotients of the respective endpoints equal one. Concentrations above the respective line indicate risk for the respective endpoint.

Risk characterization concerning the aquatic compartment

Table 3-27 shows the calculated risk quotients for heavy metals in surface water due to surface runoff from the agricultural areas in Braunschweig.

Table 3-27: Calculated Risk Quotients for heavy metals in surface water due to surface runoff from agricultural areas in Braunschweig

Metal	PEC_{water} [µg/L]	$PNEC_{water}$ [µg/L]	$RQ_{surface\ water}$
Cadmium	0.073	0.08	0.1822
Chromium	0.22	3.4	0.0638
Copper	4.4	7.8	0.5679
Lead	0.47	7.2	0.0654
Mercury	0.0085	0.047+BC	0.1817
Nickel	0.66	5	0.1319
Zinc	8.4	7.8+BC	1.0726

Except from zinc all metal are well below the $PNEC_{water}$, resulting in a Risk quotient smaller than 1. Zinc is the only metal exceeding the $PNEC_{water}$.

Evaluation and discussion

The conducted QCRA of heavy metals show that, except from cadmium and zinc, heavy metals neither exceed the critical soil concentrations for human consumption nor the predicted no-effect concentrations (PNECs) for environmental endpoints.

The present zinc soil concentrations pose no risk for humans. Concerning environmental risks currently the $PNEC_{soil}$ is exceeded in pumping district IV if area 5 is included. The soil concentrations in the other pumping districts as well as when area 5 is excluded will all exceed the $PNEC_{soil}$ in the next 50 to 70 years. The $PNEC_{soil}$ for zinc on area 5 in district IV is exceeded significantly. Soil concentrations thus pose a risk for the terrestrial ecosystem.

Concerning the aquatic environment zinc concentrations exceed the $PNEC_{water}$, leading to a risk quotient of 1.07. The exceeding of the tolerable value (RQ=1) by a value of 0.07 means that modeled zinc concentrations in water pose a risk for algae and crustacea. Nevertheless, against the background of present uncertainties within the model this is not a significant exceeding.

Concerning area 5 in pumping district IV as well as areas 1 and 1b in pumping district III Cadmium concentrations exceed the critical concentrations for all relevant endpoints. Although the concentrations show a decreasing trend, concentrations do not fall below the critical concentrations within 100 years independently from the soil-wheat relation used for deriving tolerable soil concentrations. Against this background Cd poses a risk for human health and the environment on these areas, although present concentrations are not the direct result of present wastewater reuse.

Concerning cadmium concentrations on the other pumping districts the results show that an assessment of present and future risks for the environment and human health depends on the used critical soil concentrations. The critical soil concentration derived by Schütze et al. for mammals and birds lead to present risks for this endpoint, whereas the application of the European predicted no-effect concentration for animals does not.

Moreover, the application of an appropriate soil-wheat relation for deriving tolerable soil concentrations is crucial for the assessment of human health risk concerning area 2b and 3 in pumping district III.

Before making any final statement on the human and environmental risks caused by cadmium, it shall be discussed and derived which of the respective tolerable concentrations for animals and humans is the more appropriate one for this risk assessment.

Tolerable Cd soil concentration for animals

The European Risk assessment Report for cadmium proposes a $PNEC_{soil}$ for animals of 0.9mg/kg_{dw}, whereas Schütze et al. calculate a critical soil concentration of 0.14 for the black-tailed godwit as a reference for worm eating birds and a value of 0.062mg/kg_{dw} for the badger as a reference animal for mammals.

Since both sources are considered to be reliable and trustworthy, the respective outcomes shall not be questioned at this place. Nevertheless, as a personal remark, the critical soil concentrations, which were calculated for badgers and the black-tailed godwit by Schütze et al. are just slightly above or even below the average cadmium concentration in the natural earth crust of 0.1mg/kg ([101]). It may be that even under natural conditions without any anthropogenic influence adverse effects on these two animals may occur. Therefore, the question arises if these two animals are the appropriate reference organisms for assessing environmental risks due to wastewater application. Thus, concerning risk calculation of animals the European PNEC is preferred.

Tolerable Cd soil concentration concerning human health

The result clearly point out that the conclusion, whether present and modeled future soil concentrations pose a risk for humans consuming agricultural products from the areas of the AVBS, depends on the calculated tolerable soil concentration. The calculated Spearman coefficients (see section0) indicate correlation between soil and wheat concentrations. Nevertheless, this does not give any information whether this correlation is linear or not. The BCF method assumes a linear relationship. Instead, the equation formulated by DeVries et al 2003 results in a graph, where wheat concentration does not increase as strong as soil concentrations (Figure 3-42).

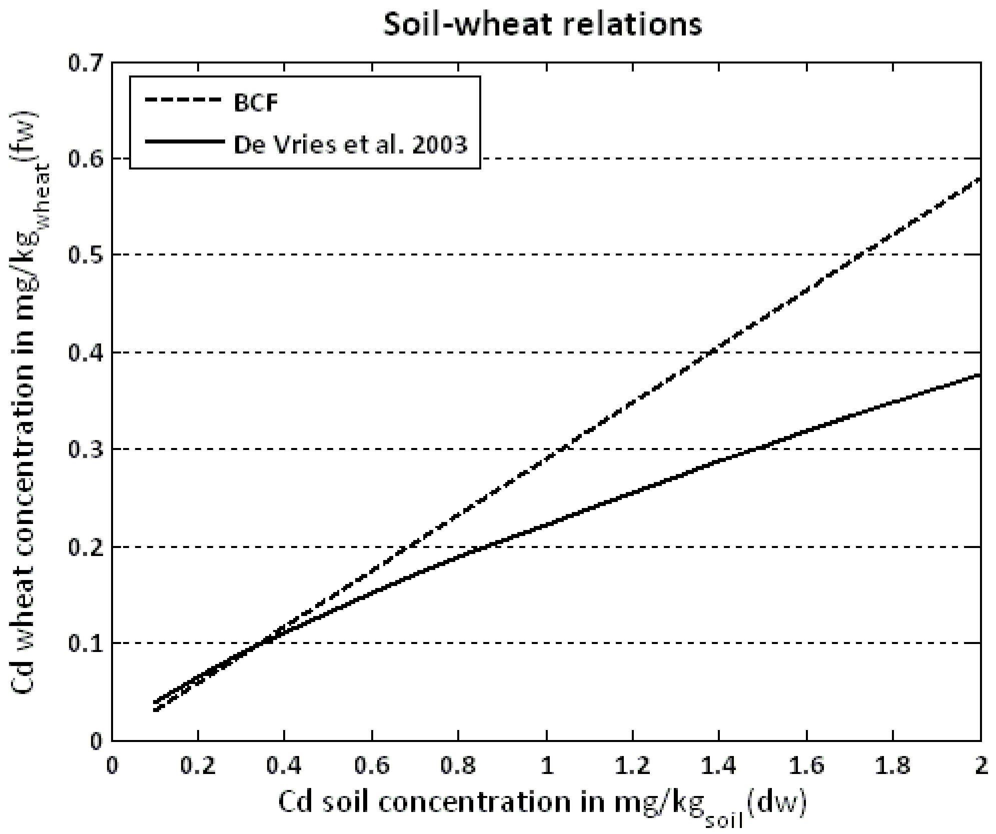


Figure 3-42: Soil-wheat relation using the BCF method and the equation formulated by DeVries at al 2003

Figure 3-42 points out that the two approaches lead to similar results up to a soil concentration of 0.3-0.4mg/kg$_{soil}$(dw). Above this value the BCF method leads significantly higher wheat concentrations as the De Vries method does.

Paired data for soil and wheat concentration are available for the years 1995-1999 and 2009-2010. Measurements take place once a year. Depending on the year 1-4 values for wheat concentration are available per pumping district. This is certainly not enough of a data set to calculate reliable mean values. Thus, the quality of the calculated BCF has to be questioned. Moreover, the use of BCFs for calculating soil-wheat relations in general is subject of discussion. Some publications like ([107]) use it for all metals, whereas others like [102] state that *"only for Cd in wheat some relationship can be discerned. For all other combinations, the BCF concept does not work, since there is simply not such a relationship" (p.51, l.3).*

However, the fact that the BCF method may not be the most appropriate method to calculate tolerable soil concentrations does neither implicate that the equation of De Vries et al. 2003 is a more appropriate approach nor that the BCF methods does not lead to reasonable results for certain cases (e.g. Cd in wheat). Therefore, a comparison is made between the initial modeled wheat concentrations (see Table 3-28) based on the mean soil concentrations and the measured wheat concentrations of the four pumping districts during the time span from 1995-2010. The values are calculated by using the measured dry matter content and applying a dry matter content of 86% (Ripke, personal correspondence) Data for 2006 are missing. Figure 3-43 shows the measured data. Modeled values as well as the mean and the median of the measured data are shown in Table 3-28.

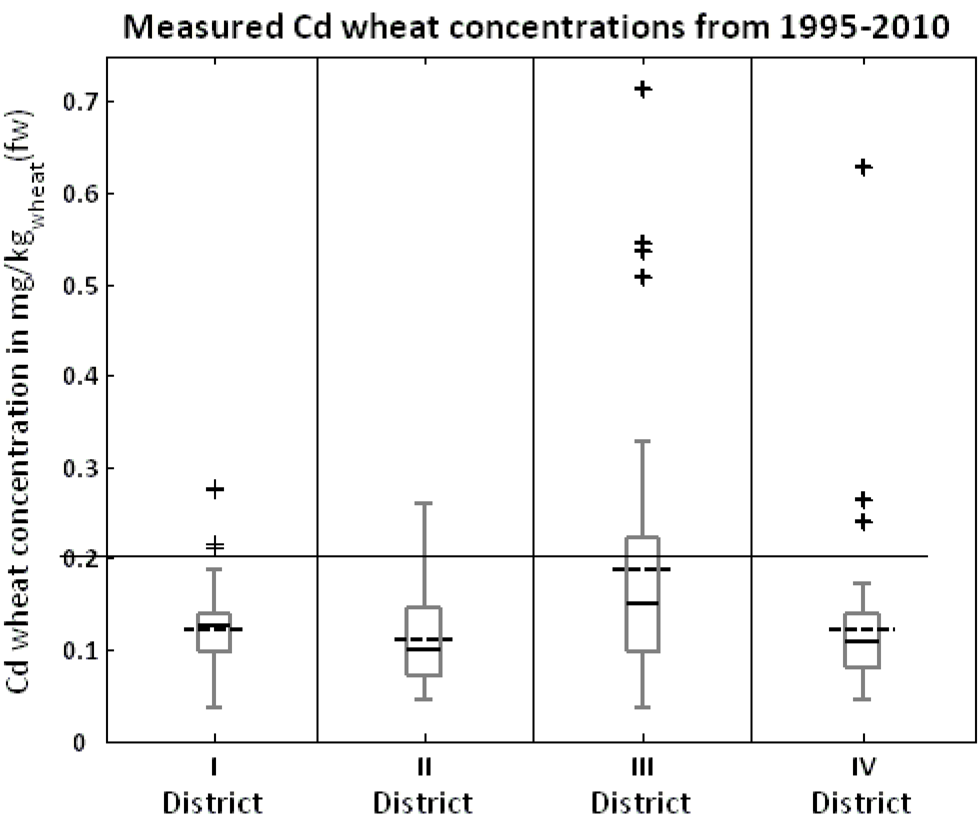


Figure 3-43: Measured wheat concentrations from 1995-2010. Black solid lines show the median, black dotted lines the mean value. The boxes range from the 25 to the 75 percentile. Black crosses indicate outliers. The horizontal line represents the derived critical wheat concentration. (Pumping district I (n=41), Pumping district II (n=32), Pumping district (n=41), Pumping district IV (n=42))

Table 3-28: Modeled and measured wheat concentrations in mg/kg$_{wheat}$(fw)

Pumping district	Modeled (De Vries)	Modeled (BCF)	Measured median	Measured mean
I	0.119	0.128	0.126	0.126
II	0.107	0.11	0.099	0.117
III	0.176	0.206	0.151	0.189
IV	0.166	0.197	0.108	0.126
IV without area 5	0.097	0.084	0.108	0.126

The comparison between modeled and measured data shows that for lower Cd soil concentrations (Pumping districts I, II, IV without area 5) the modeled values correspond to the measured ones for both modeling approaches. Concerning pumping district III which shows higher Cd soil concentrations the both modeled concentrations in wheat exceed the median measured value. Concerning the mean measured value the BCF method overestimates measured concentrations, whereas the equation of De Vries et al.

2003 underestimates the measured mean. The De Vries equation shows a deviation of -0.013 $mg/kg_{wheat}(fw)$, the BCF method one of +0.017 $mg/kg_{wheat}(fw)$.

Nevertheless, it is not only the question if the used model approaches represent reality appropriately, but also if the measured data represent reality in an appropriate way. Within a 15 year time period the number of annual single samples per pumping district ranges from 2 in district II to 3 in the other districts. Assuming that the single pumping districts are equally large, and that cereals are grown on 30% of the agricultural areas of the AVBS (see section **Fehler! Verweisquelle konnte nicht gefunden werden.**), than 1 sample represents an area of 225ha. If additionally, other sources of uncertainties, like the annual variations of environmental conditions (weather), the species of wheat, the sampling methods etc. are taken into account, the question arises if these measured data are sufficiently reliable to validate the respective model. Against this background of present uncertainties, none of the models can be described as completely inappropriate by the comparison to measured data.

Environmental and human health risks due to Cadmium

The critical discussion on the formulated tolerable soil concentrations for animals and human health led to the conclusion that the $PNEC_{soil}$ formulated by the European Union for animals seems to be the more appropriate value for assessing environmental risks for this endpoint. Except from area 5 in pumping district IV and the areas 1 and 1b in district III, the $PNEC_{soil}$ is currently not exceeded and also the model results indicate that this will not be the case within the next 100 years in the other pumping districts. Adverse effects are thus unlikely to occur in those districts.

Concerning risks for human health, there are three types of areas. The Cd soil concentrations on area 5 of district IV and on area 1 and 1b in district III clearly exceed the tolerable value independently from the used soil-wheat relation. According to the used methodology these concentrations hence pose a risk for human health if products for human consumption are grown on them.

The second type of areas is area 2b and 3 in district III. Here statements of current risk depend on the used soil- wheat relation. Concerning the tolerable concentration for human health impacts even the comparison to measured wheat concentrations does not give further information, which of the two approaches is the more appropriate one. Both approaches lead to wheat concentrations comparable to the actually measured ones at low soil concentrations. The two models show higher deviations to measured data for the higher soil concentrations. Nevertheless, against the present uncertainties of the monitoring data, this deviation is too small for being a knock-out criterion for one or both of the models.

Taking this information into consideration, on the one hand, a clear statement whether the present Cd soil concentration on these areas poses a risk for human health is hard to derive, since it does, if the BCF method is applied and it does not, if the De Vries method is applied. On the other hand, it can be stated that also these areas are of concern concerning risk from Cd soil concentrations. Definitely, monitoring should be extended to gather a more reliable data set.

The third type of areas is all the remaining ones. Here, Cd concentrations currently do not pose a risk for average human consumption. The model result show that there is a kind of equilibrium concentration at a soil concentration of about 0.4 $mg/kg_{soil}(dw)$, below which concentrations are slightly increasing and above which concentrations decrease. This stable state depends on the used partitioning coefficient between soil and water used in this model. As this value is taken from literature and there is no local value known yet, this statement is uncertain. Nevertheless, against the background of present soil concentrations and decreasing overall Cd emissions in Germany, the statement can be made, that adverse human health effects resulting from wastewater reuse of these areas are unlikely to occur.

Validation of model results

Not only the derived calculated tolerable soil concentration for Cd but also the calculated soil concentrations have to be checked for plausibility. For this purpose the modeled data will be compared to measured soil concentrations. Subsequently, a sensitivity analysis is conducted to examine the robustness of the calculated results.

Comparison to measured data

Modeled cadmium concentrations show a decreasing tendency at higher concentrations and an increasing one for lower concentrations. This cannot be confirmed by measured mean values (Figure 32).

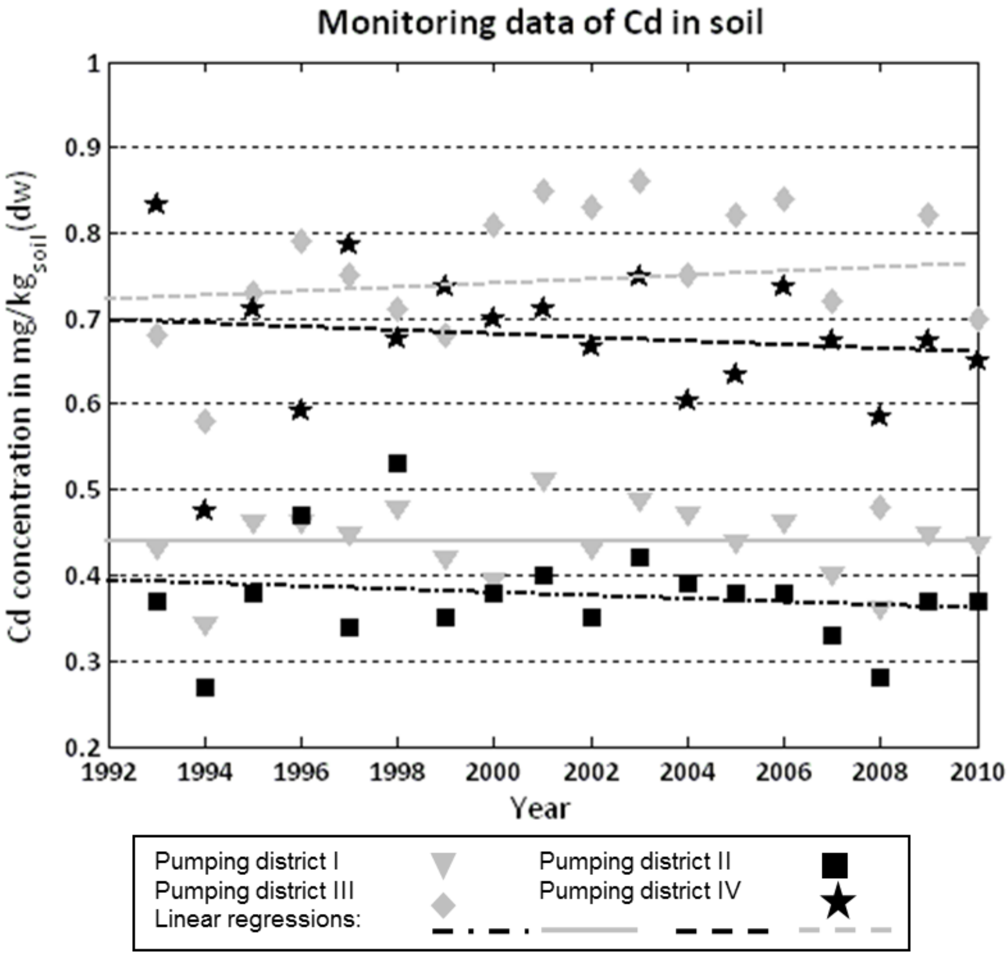


Figure 3-44: Linear regression of the measured soil concentrations in Braunschweig.

Except from the areas with a high initial concentration, especially area 5 in district IV, the model indicates a rather stagnating development. If the mean data are taken as initial concentration Cd soil concentrations show a change of less than 0.1 mg/kg$_{soil}$(dw) within 100 years. In contrast measured data show high annual fluctuations of up to 0.2mg/kg$_{soil}$(dw). Uncertainties like fluctuations in the annual precipitation, different annual cadmium loads and varying sampling locations have thus high impact on the overall results relative to the modeled results. Although the linear regression shows also just slight increases and decreases, respectively, the fit is rather poor and cannot be used for further statements.

Sensitivity analysis

The model for calculating soil metal concentrations is influenced by several factors, including physical-chemical properties of the respective metal, the surrounding environmental conditions and the annual metal loads which are applied on the agricultural areas. Moreover, the model itself may lead to imprecision with regard to the calculated results.

Cadmium is used as reference as it exceeds the most critical concentrations and PNECs. The factors which are analyzed for their respective impact on the overall results are plant uptake, the annual load of Cd applied on agricultural areas and the partitioning coefficient Kd. It will be examined if a change of the respective factor influences the final result of the assessment.

Plant uptake

The model described in the TGD does not consider the uptake of plants as an output factor. The impact on soil concentrations by including an additional removal rate constant for plant uptake is conducted by assuming that wheat is grown on the whole area, as this plant is known for its high Cadmium accumulation. Based on measured Cd concentrations in wheat and the amount of wheat which is harvested per year an additional removal rate constant is calculated.

$$kplant = \frac{Cwheat * Mwheat * DMwheat}{10 * 365 * Depth_{soil} * rho_{soil}}$$

kplant	= first order rate constant for Cd removal from top soil via plant uptake [mg/kg_{soil}*d]
Mwheat	= Mass of wheat harvested per year [kg/ha]
DMwheat	= content of dry matter in wheat [%]
$Depth_{soil}$	= mixing depth of top soil [m]
Rho_{soil}	= bulk density of top soil [kg/m³]

The amount of wheat harvested in Braunschweig is set to 7.8t/ha with a dry matter content of 86%. The overall mean measured Cd concentration is used for Cwheat (AVBS, personal correspondence).

Impacts on the overall result are presented in

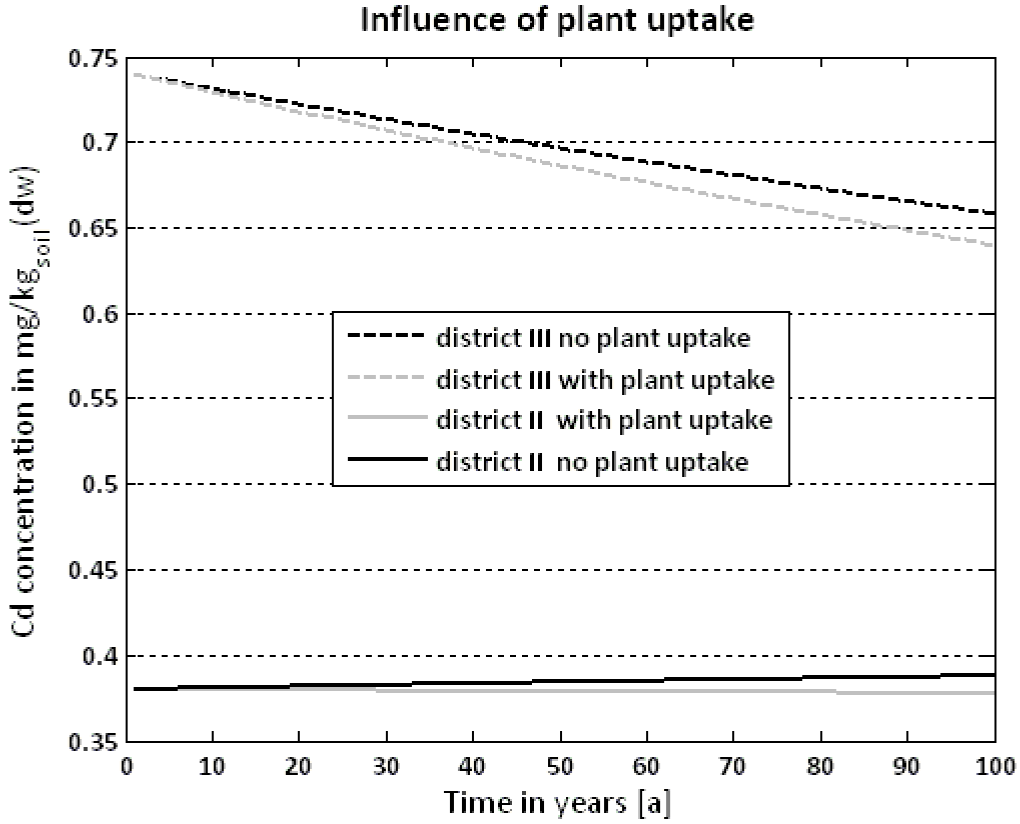


Figure 3-45.The results show that although plant uptake influences soil concentrations the differences do not change the general outcomes of the risk assessment as the overall change is approximately 3% due to plant uptake in respect to the initial concentration.

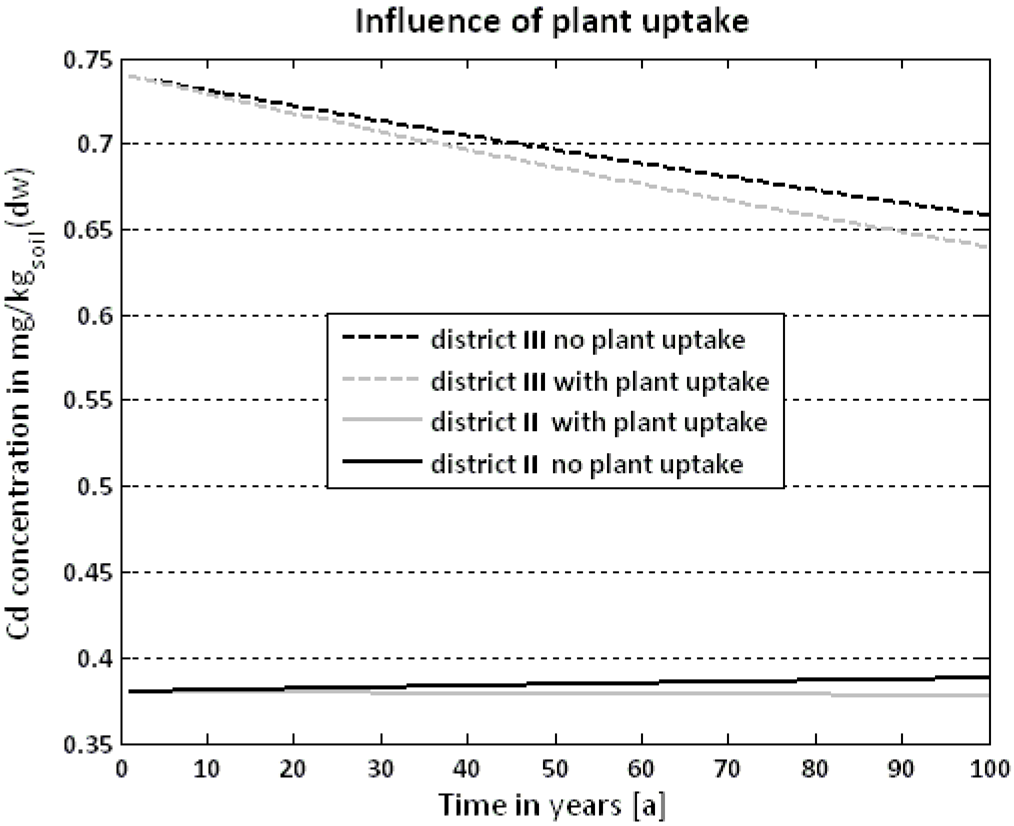


Figure 3-45: Impact of plant uptake on the overall model results. Results are plotted for the pumping districts II (black) and III (red) is the districts with the highest and lowest initial Cd concentration. The respective lower concentrations are calculated if plant uptake is included.

Annual metal loads

Figure 3-46 shows how the model reacts when the annual Cd load is changed by +10%, +20%, -10% and -20% respectively. The influence of 20% change of the annual load changes the final concentration after 100 year of wastewater irrigation of just 1% and has thus no influence on the final result.

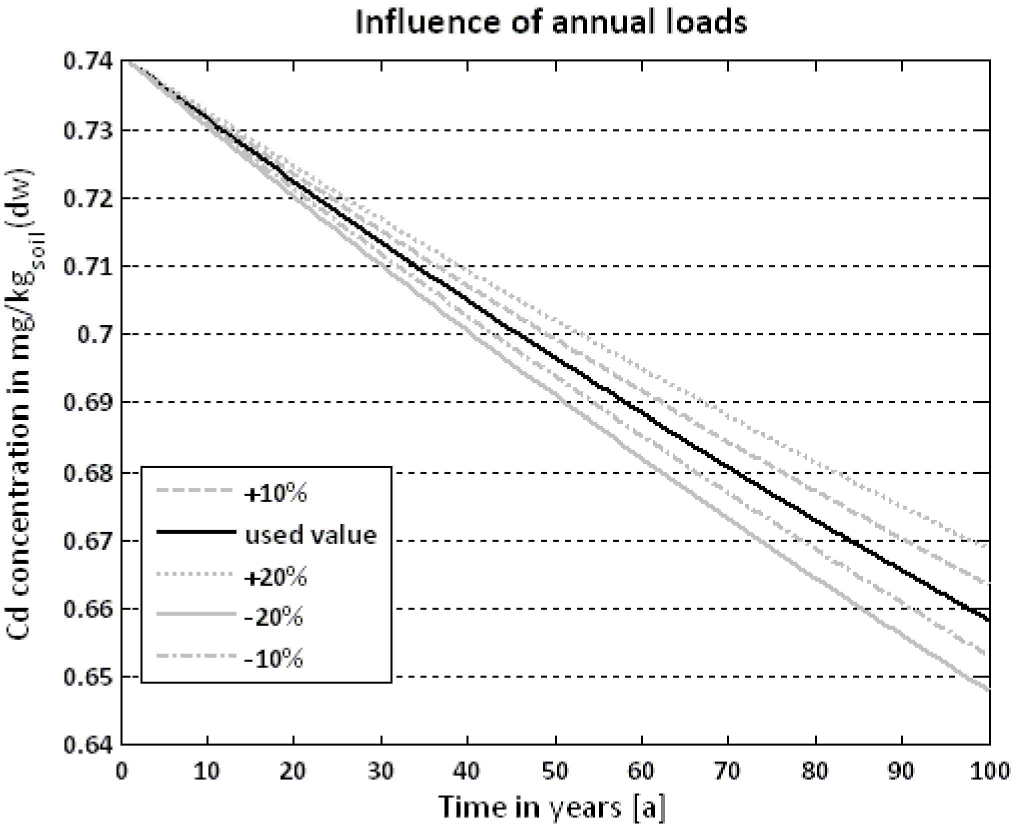


Figure 3-46: Impact of the annual Cd loads on soil concentrations.

Soil-water partitioning coefficient Kd

The Kd value determines the equilibrium between soil and soil solution and thus the amount of heavy metal which is washed out by leaching processes. The Kd value is dependent of several factors, from which the fraction of organic matter and the pH level are the most important ones. Different functional relationships have been formulated describing the relationship between organic content, pH level and the Kd value (see [110] p.190). The referenced equations are used to calculate different Kd values for this sensitivity analysis. The calculated values range from 139-310 L/kg. The higher the Kd value the higher is the amount of Cadmium remaining in top soil (Figure 3-47).

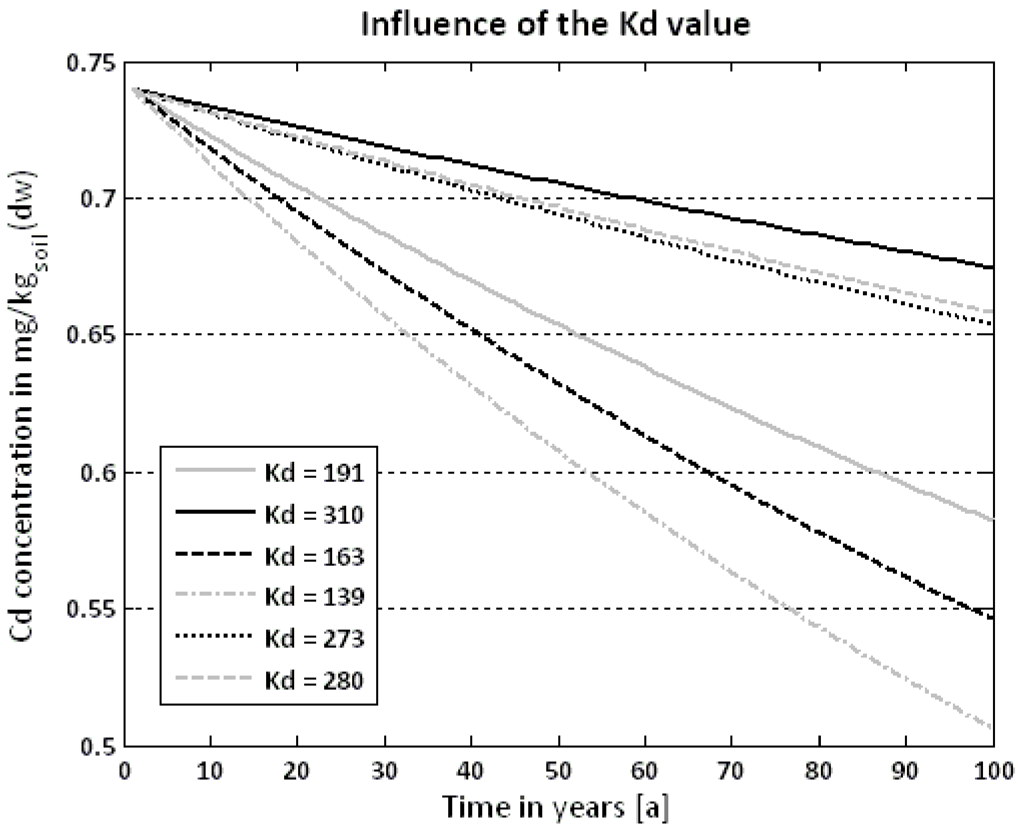


Figure 3-47: Influence of the partitioning coefficient on the modeled Cd soil concentration in pumping district III. Values for Kd are given in L/kg

Figure 3-47 shows that the choice of an appropriate Kd value influences the trend of Cadmium concentrations in top soil. A change of the used Kd value by 32% (191kg/L) results in a change of the final result of 11%. The highest calculated Kd value, which corresponds to an increase of 10% in respect to the used value, changed the final result by 2%.

Conclusions on sensitivity

The sensitivity analysis of the single factors showed that Cd soil concentrations are influenced by the partitioning coefficient Kd, plant uptake and the annual Cd loads which are applied on the agricultural areas in Braunschweig. Varying single factors while keeping the other ones constant (like in the conducted calculations) did not lead to changes in soil concentrations, which would change the overall outcome of the risk assessment in a way that final conclusions would have to be changed.

Risk based targets

Taking all the information of the conducted model and the sensitivity analysis into consideration one has to draw the conclusion that risk reduction measures for human health risks due to Cd exposure have to be considered concerning pumping district III (areas 1 and 1b) and area 5 of district IV.

As humans are indirectly exposed to cadmium via food consumption, as a short term action, risks can be reduced by stopping the production of food crops on the respective areas of the agricultural areas of the AVBS. Another option would be to prevent Cd from being taken up by plants. An increase of the soil pH value through liming could be one option to achieve this.

Nevertheless, it became obvious that in order to achieve a more sustainable solution it makes more sense to express risk-based targets in terms of environmental outcomes. A reduction of soil concentrations below all critical concentrations should be achieved. As the annual heavy metal loads of the STP Steinhof are not the only inputs, which have to be considered (atmospheric deposition, Oker), and against the background of present uncertainties concerning the surrounding environment, this target cannot be expressed as a certain tolerable annual Cd load, yet. Nonetheless, as Cd is highly toxic to humans, animals and the environment, any increase is undesirable and releases into the environment should be reduced to its minimum.

Since zinc shows an increasing tendency as well and will reach the PNEC for soil organisms within the next 20-70 years efforts to reduce zinc loads should be considered as well.

Critical discussion

Concerning the methodological approach the model calculations are based on the widely reviewed *European Technical Guidance Document on Risk assessment*. The used calculations are therefore considered to generate acceptable results concerning general outcomes and overall tendencies. However, environmental modeling can be conducted far more complicated and in more detail. The soil-plant relation was identified to be a source of uncertainty in the mathematical approach. Another weakness is that the conducted calculations are based on total metal contents in soil. No differentiation is made between total and reactive metal contents. Moreover, the speciation of the respective metal is not taken into account. This, in turn depends on the local soil pH and redox conditions. Concerning plant uptake certainly this simplification plays an important role, as just the metal content in soil solution can be taken up by plants.

Another weakness is the use of calculated instead of measured metal loads for lead and mercury. As the mass balance currently does not come out even this influence factor implicates a lot of uncertainties. The sensitivity analysis showed that also the correct value for the partitioning coefficient between soil and water has impacts on the overall tendencies of soil metal concentrations. The validity of the results can thus definitely be improved by replacing this value by an actually measured one, which accounts better for the site specific surroundings. Nevertheless, even if a far lower Kd value would be applied it would not change the final conclusions concerning the current indentified risks.

Concerning the derived critical wheat concentrations concerning human health impacts the calculated critical wheat content is not overly conservative, as it is in line with current European food quality standards (0.2mg/kg_{wheat}(fw) [116]).

The derived critical soil concentrations for Cd are calculated by two different approaches. Since the derived values are within the same order of magnitude as the values set by German legislation (precautionary value 0.4mg/kg_{soil}(dw) [117], limit value 1mg/kg_{soil}(dw) [99]), the derived values are considered to be within a reasonable range. Concerning environmental endpoints the used PNECs are based on widely reviewed European Risk Assessment Reports.

In conclusion, against present uncertainties the conducted model is on the one hand not sufficiently precise to make a statement, whether the whole reuse system of Braunschweig is “safe” or “unsafe”. On the other hand, the results are sufficiently good to identify Cd as a priority for risk reduction measures. Moreover, the whole procedure of risk assessment made weaknesses, like the Cd balance of the STP Steinhof, apparent and transparent. Since the identification of weaknesses is the necessary first step towards any improvement the generated results can be used as a first step towards a more risk based management approach.

3.4 Conclusions

From the risk assessment of the existing reuse scheme in Braunschweig for microbial hazards, the following conclusions can be drawn:

- **Current measures for risk reduction are sufficient to meet the WHO benchmark for water reuse (10^{-6} additional DALYs per person per year) for local residents (including children) for all pathogens. However, fieldworkers have an increased work-related risk of infection which exceeds the WHO benchmark.**
- Based on disinfection pilot trials, **UV disinfection (650 J/m^2) or PFA dosing (2.0 ppm of 13.5% PFA solution) of secondary effluent will be able to reduce risk** by 0.5 - 4 and 0.5 - 3 orders of magnitude, respectively. For disinfection measures to be effective, mixing of digested sludge with irrigation water has to be stopped.
- **Health risks of water reuse can also be reduced by switching from the existing excess irrigation to an "irrigation on demand" system** (adjusting water supply to actual demand of crops from May-October and local climate and weather conditions). This effect is due to a) a reduced virus concentration in raw wastewater during the vegetation period in summer (~ 0.5 log) and b) the reduced Number of exposure events per person per year, as irrigation will be limited to 150 days.
- **For fieldworkers, the WHO target can only be met by combining UV disinfection and irrigation on demand.**
- Compared to the current incidence of mild gastroenteritis in society, **the WHO target of 10^{-6} additional DALYs per person per year for water reuse seems to be very conservative** as a benchmark for all population groups (e.g. fieldworkers). Existing measures for risk management (crop restrictions, air drift control and protective hedges) already reduce health risks from water reuse for the general public to very low levels.

Environmental impacts of the Braunschweig reuse system and several options for improvement have been assessed with LCA. This analysis leads to the following conclusions:

- **The existing system of water reuse in agriculture reduces local nutrient loads** and resulting eutrophication potential of the receiving surface water (Oker) by more than 50% compared to direct discharge of secondary effluent. In addition, **water reuse reduces water footprint of agricultural irrigation** by substituting the use of groundwater resources. On the contrary, **water reuse leads to additional energy demand and associated emissions of the system** (+ 29% in net energy demand) mainly due to water pumping in the irrigation network.
- Hence, the existing reuse scheme in Braunschweig can be principally recommended from an environmental point of view. However, the reuse scheme could be **energetically optimized by switching from excess irrigation to irrigation on demand** (-29% in net energy demand), although nutrient emissions in surface water will then be increased (+ 70%) with more direct discharge of WWTP effluent. In addition, nutrient management should be improved to better match nutrient supply and seasonal demand, especially for nitrogen.
- Reducing hygienic risks of water reuse with **UV or PFA disinfection will slightly increase net energy demand at the WWTP (+6 or +14 %).** Combining disinfection and irrigation on demand will reduce overall energy demand of the reuse system compared to the existing status.
- For disinfection to be effective in risk reduction, mixing of digested sludge and irrigation water has to be stopped. **Continuous dewatering of sludge will then lead to increased net energy demand**

(+ 19%) due to electricity and polymer demand for centrifuge and higher return load. In addition, nutrient supply with the irrigation water will be stopped.

- Existing and potential future drawbacks of energy and nutrient management can be overcome by **optimizing the sludge treatment with DLD for increased energy recovery and nutrient recovery technologies (struvite precipitation and ammonia stripping)**, mitigating or even over-compensating the negative effects of continuous dewatering.
- Based on the outcomes of this LCA study, **an optimized Braunschweig reuse system would include irrigation on demand, UV disinfection and enhanced sludge treatment with DLD and nutrient recovery technologies;** nonetheless this approach will potentially increase eutrophication potential in the receiving surface water by increasing volumes of WWTP discharge to Aue-Oker-Canal. It has to be checked if this additional nutrient load is in line with current legal obligations of the WWTP, and if it could probably lead to potential problems of eutrophication downstream of the WWTP discharge.

4 Case Study of Old Ford Water Recycling Plant

4.1 Introduction and Setting

The Old Ford Water Recycling Plant (OFWRP) is the UK's largest community wastewater recycling scheme. It is located next to the Queen Elisabeth Olympic Park (QEOP) in East London and was first operational for the Olympic Games in 2012. It is owned and operated by Thames Water Utilities Limited (TWUL). The catalyst for the scheme was a strategy to reduce potable water consumption during the Olympic and Paralympic Games and through to the post-games legacy period.

The source of water is raw sewage which is "mined" from a large sewer directly adjacent to the OFWRP. The flow is predominantly made up of domestic and light commercial sewage along with surface drainage captured in the combined sewer. The plant has a capacity to produce 574 m^3/d (0.21 Mio m^3/a) of recycled water using a membrane bioreactor (MBR) with ultrafiltration membranes, granular activated carbon (GAC) filtration and disinfection with dosing of NaOCl (Figure 4-1). The recycled water is supplied to the QEOP via a dedicated 3.6 km network for reclaimed water which was designed to minimise risks of unintended or unapproved uses. The recycled water is used for toilet flushing, topping up rainwater harvesting systems at venues and predominantly for irrigation of the parkland.

The OFWRP is contractually required to achieve a percentile-based water quality standard. This standard was based on the USEPA guidelines for 'unrestricted urban reuse' [96]. Other water quality criteria were specified by customers and consideration was also given to UK drinking water standards.

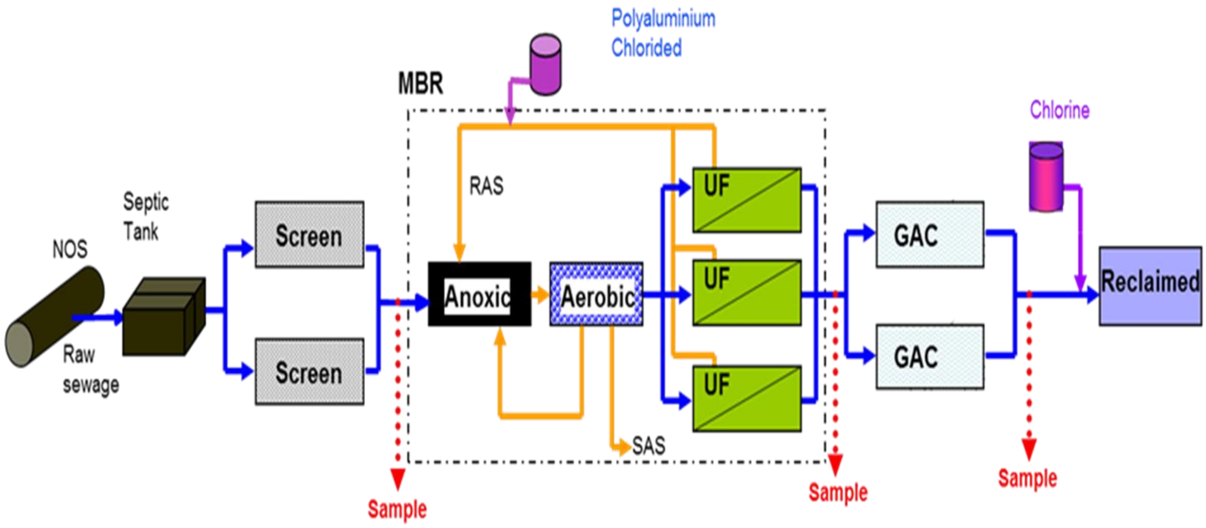


Figure 4-1: Treatment scheme for water recycling at Old Ford Water Recycling Plant in London
MBR: membrane bioreactor, RAS: return activated sludge, SAS: Surplus activated sludge, UF: ultrafiltration membrane, GAC: Granular activated carbon

The Old Ford site itself is a Site of Nature Conservation Importance (SNCI) and the treatment plant building was architecturally designed to include natural materials such as timber cladding, gabion walls and greenroofs (Figure 4-2). As well as contributing to a reduction in potable water use at the QEOP, the scheme also provides a focus point for Thames Water's research into water recycling and reuse. This research is driven by water resource management planning requirements that aim to address future pressures from population growth and changing climatic conditions.

Figure 4-2: Old Ford Water Recycling Plant in the Olympic Park in London (© Thames Water)

4.2 Life Cycle Assessment

4.2.1 Goal and scope definition

The goal of this LCA is to assess the environmental impacts associated with construction and operation of the OldFord Water Recycling Plant (WRP) in the Olympic Park in London and compare them to the conventional water management in London with separate wastewater treatment and drinking water supply. The study should reveal the benefits and drawbacks of water reuse in an urban setting, using the advanced treatment train of OldFord WRP. Hence, this LCA can serve as an example for environmental impacts of a water reuse schceme using MBR technology. The target group of this study consist primarily of the local stakeholders such as the treatment plant operators (Thames Water), but also planers and engineers in the field of wastewater treatment and recycling.

Function/ Functional Unit

The function of the system under study is to provide both wastewater treatment in the north London area and water supply for irrigation of the Olympic Park in London and toilet flushing. The analysed system includes all processes that are related to this function. The functional unit is defined by the annual wastewater pollutant load of OldFord WRP, defined as population equivalents (pe). Referring to the annual COD load of OldFord WRP (64 t COD/a) and assuming a daily load of 120 g COD/ (pe*d) [39], the annual wastewater load entering the plant corresponds to 1 500 pe. The WRP supplies 118’000 m^3/a of reclaimed water for unrestricted urban reuse, and the reclaimed water fulfills the standards of the USEPA guidelines for water reuse [96].

System boundaries

The system boundaries cover the entire treatment train of OldFord WRP, consisting of primary treatment, secondary treatment in MBR, and tertiary treatment. Sludge handling and disposal is also included in the assessment. Reclaimed water distribution is included in terms of pumping energy, but pipe networks are excluded from the LCA. Infrastructure for the treatment plant is part of this LCA. For the conventional scheme, both wastewater treatment and freshwater supply from reservoir are included in the assessment, together with infrastructure for the WWTP. The LCA also includes all background process for production of electricity, chemicals, fuels, materials for infrastructure, and maintenance (Figure 4-3).

Allocation

Credits from co-products such as electricity from biogas valorisation in CHP plants produced during sludge stabilisation in anaerobic digestion or electricity generated in sludge mono-incineration facility is allocated to the function of wastewater treatment.

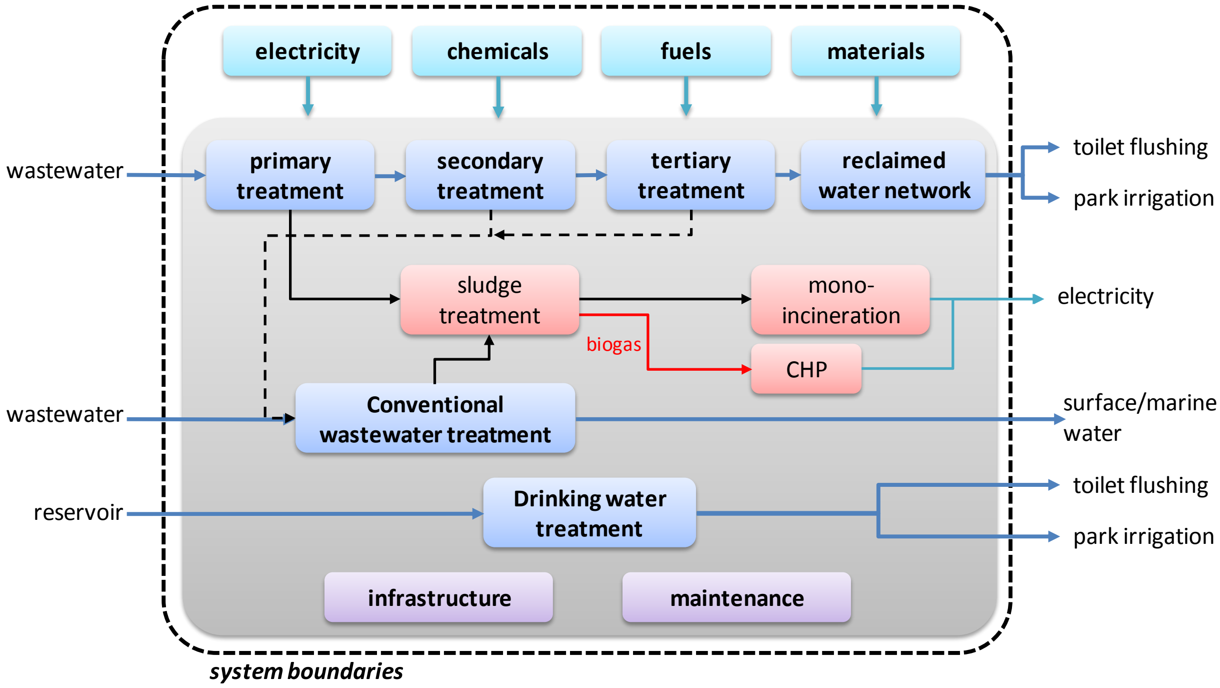


Figure 4-3: System boundaries and scope for LCA study Old Ford

Scenarios

Two scenarios are defined to compare the OldFord WRP with a conventional system of water management in London ('2 No Reuse'). A short overview (Figure 4-4Figure 4-3) is provided below:

1. OldFord represents the existing WRP operating in the Olympic Park since 2012. The plant has a capacity to treat an annual wastewater volume of 125 000 m³ and provides 118 000 m³/year as reclaimed water (94 % recovery rate). Drawing a mixture of municipal wastewater and surface water from the combined sewer system, the plant includes primary treatment with 1mm screens and septic tanks, followed by an activated sludge process including nitrification and denitrification and PACl-dosage for P removal (Figure 4-1). Activated sludge is separated in a dedicated tank using submerged and aerated membrane modules. The biological process and the membrane tank constitute the MBR. Tertiary treatment includes a GAC filter for removal of trace organics and colour and final chlorination with NaOCl to secure disinfection. The reclaimed water is then delivered to a buffer tank, and then pumped into the non-potable water network. Currently, the bulk of reclaimed water (84 %) is currently reused for irrigating the Olympic Park, while 16 % are used for toilet flushing on the Olympic park [97]. Off-gas from the entire treatment train is collected and treated in a biofilter to minimize negative effects in this sensitive area. Primary sludge from septic tanks (4 % dry matter) is transported by truck to the centralized sludge treatment in a large-scale WWTP, while secondary sludge and GAC backwash water are discharged to the local sewer network and also end up in the large-scale WWTP a conventional activated sludge plant. Here, the sludge is thickened, stabilised in anaerobic digestors with valorisation of biogas in CHP plants, dewatered and finally disposed on-site in mono-incineration with partial generation of electricity from off-gas heat in a steam turbine.
2. No reuse represents the conventional water management of this London area as an alternative to the reuse system. In this scenario, the wastewater volume of 125 000 m³/year is treated in a large-scale WWTP, discharging around 122 000 m³/year into the river Thames after treatment.

The large-scale WWTP includes primary treatment and conventional activated sludge tanks, but no enhanced P removal. Primary and secondary sludge is thickened, stabilised in anaerobic digestion with biogas valorisation in CHP plants, dewatered and finally disposed on-site in mono-incineration. Water supply for irrigation of the Olympic Park and for toilet flushing is realized from the drinking water network in this scenario. Raw water (124 000 m³/a) is taken from a reservoir and treated in a drinking water treatment plant (DWTP). Due to lack of information for the existing DWTP in this area, it was assumed that the DWTP process includes ozonation, sand filtration with GAC layer and final disinfection (via dosage of NaOCl) plus addition of corrosion control additives (H_3PO_4).

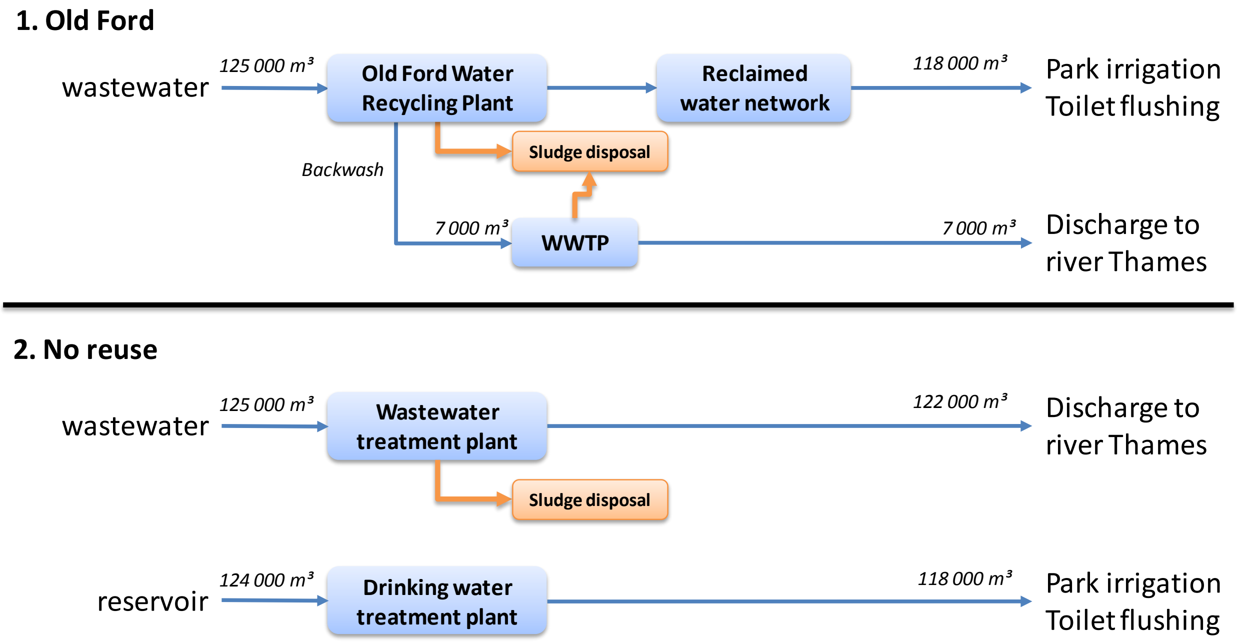


Figure 4-4: Comparative overview of scenarios for LCA Old Ford

Data quality and limitations of this study

Input data for the LCA inventory is discussed below regarding data quality and uncertainties to point out inherent limitations of this study. An overview of data sources and quality is provided in Table 4-1.

- Water quality and quantities: Data on water quality and quantities was provided by the WWTP operator Thames Water [97]. Since this data is collected from long-term measurements and is subject to internal quality control, the data quality is assumed to be very good.
- Sludge production and treatment: Water content and other parameters of different sludge types for the OldFord WRP has been provided by the operator [97] as well as important design parameters (such as sludge age in the MBR) to predict sludge production in reuse and conventional WWTP. Data for the large-scale WWTP sludge treatment system has been estimated based on previous LCA studies of KWB. Overall, this data can be interpreted as representative for a large WWTP, but may not fully represent the actual data from the large-scale WWTP.
- Energy, chemicals and material consumption: A previous carbon footprint study on the OldFord WRP by Thames Water provided a very detailed inventory on electricity consumption of different aggregates and treatment steps as well as for material consumption regarding infrastructure [98]. In addition, input data on electricity and chemical demand for the reuse system was validated again in close cooperation with the operator [97], so data quality is expected to be very good. In

terms of materials for infrastructure at OldFord, not all listed aggregates in the detailed inventory could be transferred into the LCA software, filling existing gaps with estimates. For the large-scale WWTP and also the DWTP, electricity consumption has been roughly estimated by the operator, together with more detailed data of chemical consumption of the DWTP [97]. Infrastructure for the large-scale WWTP has been estimated based on previous studies of KWB [45].

Table 4-1: Overview on data quality of input data

Parameter/Process	Data source	Data quality
OldFord Water Recycling Plant		
Water quality	[97], local operators	very good
Sludge production, treatment, qualities and quantities	[97], estimations	medium
Electricity consumption	[98], local operators	very good
Chemical consumption	[97], local operators	very good
Infrastructure	[98], local operators	good
No reuse (large-scale WWTP and DWTP) – data confidential		
Water quality (WWTP)	[97], local operators	very good
Sludge production, treatment, qualities and quantities	[97], estimations	medium
Electricity consumption (WWTP and DWTP)	[97], estimations	medium
Water quality (DWTP)	[97], local operators	very good
Chemical consumption (DWTP)	[97], local operators	medium
Infrastructure	[45], estimations	medium
Background		
Electricity mix	Mix of UK 2010	good
Chemicals and materials	EU or global datasets	good
Transport	Truck transport (EU)	good

Normalization

Normalisation reveals the contribution of the different systems under study in relation to the total environmental footprint of each citizen in the EU27. Principles for normalization and normalization factors are shown in Annex 9.1.1.

4.2.2 Inventory (Input data)

Primary data

Inventory data for the LCA study was provided by the local operator Thames Water and complemented with estimates based on previous LCA studies of KWB (Table 4-1). For consumptives, Table 4-2 summarizes the electricity demand and Table 4-3 summarizes chemical demand for all scenarios. Rough estimations on materials for infrastructure are shown in detail in Annex 9.4.1.

The OldFord WRP has a higher electricity demand for primary and secondary treatment than the large-scale WWTP, which is mainly due to the operation of the MBR (aeration). However, primary treatment in septic tanks of Old Ford is very efficient, so that more solids and COD are transferred to primary sludge. OldFord WRP also uses PACl dosing in secondary treatment for phosphorus precipitation. MBR system has to be cleaned with chemical backwash (NaOCl, citric acid) to prevent inorganic and biofouling.

Table 4-2: Inventory data for energy (summarized in categories) for LCA Old Ford
different volumes per categories and scenario as in Figure 4-3 primary data mainly provided by Thames Water [97, 98]

	Unit	1. OldFord	2. No reuse
Wastewater treatment, total	**kWh/a**	**129 022**	Data is confidential
Wastewater treatment, total	kWh/m³ wastewater	1.03[6]	
Pre-treatment	kWh/m³ wastewater	0.16	
Activated sludge tank	kWh/m³ wastewater	0.24	
MBR system + clarifier	kWh/m³ wastewater	0.32	
Tertiary treatment (GAC, Cl)	kWh/m³ wastewater	0.07	
Water network	kWh/m³ reclaimed water	0.12	
Pumping sludge in sewer	kWh/m³ sludge	0.05	
Odour control	kWh/a	15 774	
Drinking water treatment, total	**kWh/a**	**4 563**[7]	
Drinking water treatment, total	kWh/m³ drinking water	0.5	
Sludge treatment and disposal, total	kWh/a	- 24 602	
Monoincineration (net)	kWh/kg DS in incineration	- 0.11	
Electricity from CHP via biogas production	kWh/kg COD in digester influent	- 0.82	
Overall electricity demand, total	**kWh/a**	**108 983**	

Sludge production for both MBR and conventional WWTP has been estimated according to German wastewater models (DWA A 131 [39]), assuming a high sludge age for MBR (27d [97]) and a lower for the large-scale WWTP (10d). Consequently, sludge production in secondary treatment is lower in MBR systems, which is also reflected by lower biogas yields when this sludge is stabilised.

In tertiary treatment of OldFord WRP, the GAC filter is estimated to have a lifetime of 5 years for the virgin material (equivalent to an exchange after 15 000 bed volume) before new virgin material is required. Regeneration of GAC is currently not planned at the WRP, even though it would be an option for larger plants. The volume of the GAC filter is 6.5 m³, requiring 2860 kg GAC for filling. The GAC product is produced from renewable raw materials (coconut shells). The backwash volume of the GAC filter is estimated to 4 % of the feed flow, which is discharged into the sewer. For final disinfection 4.5 ppm Cl are added via NaOCl.

For operation of the OldFord WRP, additional drinking water is also used (25 m³/d) for pipe flushing, which is accounted for with comparable efforts than DWTP in scenario ‘No Reuse’.

[6] Including water treatment, odour control, and water distribution in the non-potable network

[7] Small amount of drinking water required at OldFord WRP for pipe flushing

Table 4-3: Inventory data for materials demand for LCA Old Ford

related on different volumes and aggregates, all concentrations per feed volume and chemicals in concentrations with water; MBR = MBR maintenance; GAC = GAC plant at OldFord; Cl = Chlorination at OldFord; DWTP = Drinking Water Treatment Plant [97]

Chemical	Unit	1. OldFord	2. No reuse
Polymer for sludge dewatering	g/kg DS	13.3	Data is confidential
PACl (10 % Al)	mg/L	48.9	
NaOCl (15 %)	mg/L	22.2 (MBR) 60.9 (Cl) 29.7 (DWTP)	
Citric acid (40 %)	mg/L	6.90 (MBR)	
Fresh GAC	kg/a	572 (GAC)	
Regenerated GAC	kg/a	-	
Ozone	mg/L	-	
H_3PO_4 (85 %)	mg/L	-	
Mono-incineration	*Consumption of additives, natural gas, emissions to air and sludge disposal adopted from [71]*		

Water Inventory

Table 4-4 shows the water volumes and qualities for the WWTP influent and the related effluents. Differences between influent and effluent are integrated in sludge.

Table 4-4: Water inventory including WWTP influent and effluent

Measured data by [97], * estimates and model calculations

Parameter	Unit	WWTP influent	1. OldFord effluent	2. No reuse ffluent	Drinking water
Volume	m^3/a	124 597	117 696	Data is confidential	
SS	mg/L	293.3	2.2		
COD	mg/L	515.0	14.7		
DOC	mg/L	98.2	3.9		
TN	mg/L	50.6	15.8		
TP	mg/L	8.3	2.1		
Cd	µg/L	0.2	0.1		
Cr	µg/L	4.7	1.3		
Cu	µg/L	68.2	7.8		
Hg	µg/L	0.3	0.1		
Ni	µg/L	5.1	1.8		
Pb	µg/L	19.1	0.4		
Zn	µg/L	232.3	18.5		

Background data

Information on background processes are shown in the Annex 9.4.1. Datasets for background processes have been extracted from the ecoinvent database v3.1 [76].

Inventory for Water Impact Index

The Water Impact Index (WIIX) is calculated according to the methodology described in *D3.1* [11]. The water scarcity index (WSI) is adopted from the *AWARE method [52]*. Although monthly WSI are available for the London region (see Annex 9.1.1, Table 9-2), an annual mean WSI of 1.8 was used for calculating the WIIX. Monthly differentiation of water withdrawals or releases was not conducted in this study due to the low seasonal variety in water demand and availability in the Olympic Park. The water withdrawals and releases in both scenarios are shown in Table 4-5 according to chapter 4.2.1.

In the OldFord WRP, 118 000 m³ water can be annually reclaimed for reuse. In addition, 25 m³/day (9 125 m³/a) of drinking water is required for pipe cleaning, for which the withdrawal from freshwater resources is fully accounted in the WIIX. From the reclaimed water, 16 % of the volume is used annualy for toilet flushing, which is not accounted as release to the environment as it will end up in the technosphere (i.e. wastewater system). The remaining 84% of reclaimed water (98 865 m³/year) is used for irrigation of the park. During irrigation, only 25% of the water will effectively reach the groundwater after subtracting for evaporation and plant uptake [51]. Hence, only 24 716 m³/year of irrigation water are actively released into the environment in terms of replenishing freshwater resources. This loss in irrigation water is relevant for both scenarios and is an inherent feature of water used for plant irrigation.

As described in Chapter 4.2.1, the large-scale WWTP would release ca. 122 000 m³/year when treating the same WWTP volumes as the OldFord WRP. The WWTP discharges into the river Thames, which is accounted as both a freshwater and marine environment here to reflect the high tidal influence due to the proximity of the North Sea. This aspect is considered in the WIIX by defining the WWTP discharge into 50% freshwater and 50% marine environment, meaning that only 50% (61 000 m³/year) of the WWTP effluent is actually replenishing freshwater resources. For the drinking water production in scenario 'No reuse', 118 000 m³ of drinking water have to be supplied (comparable to the total volume of reclaimed water produced at OldFord WRP), which leads to a withdrawal of 123 891 m³/year from the reservoir due to 95% recovery in the DWTP process.

The water quality index (WQI) is calculated based on water quality parameters of the respective water withdrawn or released in comparison to environmental standards (see Annex 9.4.1 Table 9-14). In WQI calculation, phosphorus determines the WQI of all releases, leading to a WQI of 0.09 for the OldFord WRP product. Similar assumptions are made for the large-scale WWTP and the DWTP.

Table 4-5: Overview on direct water withdrawal and release and corresponding water quality indices (WQI) for the different scenarios

DW = drinking water; IR = irrigation water (Olympic Park); EF = WWTP effluent

Scenario	1. OldFord	2. No reuse
Withdrawals [m³/year]	9 125 (DW)	123 891 (DW)
WQI (Withdrawals)	*0.45*	*0.45*
Releases [m³/year]	24 716 (IR)	24 716 (IR) 61 197 (EF)
WQI (Releases)	0.09 (IR)	*0.45 (IR)* *0.05 (EF)*

4.2.3 Impact Assessment (Results)

Environmental impacts of both scenarios were assessed with a set of 8 impact categories (including water impact index), representing different areas of environmental concern. Results of the impact assessment are discussed in detail below.

Total environmental impacts and benefits of all scenarios

The environmental profile of both scenarios is shown for all selected impact categories in Figure 4-5, setting the gross impact of scenario '2 No reuse' as 100 %.

The fossil and nuclear cumulative energy demand (CED), the global warming potential (GWP) and terrestrial acidification potential (TAP) are strongly influenced by the background processes, such as electricity, chemicals or material production.

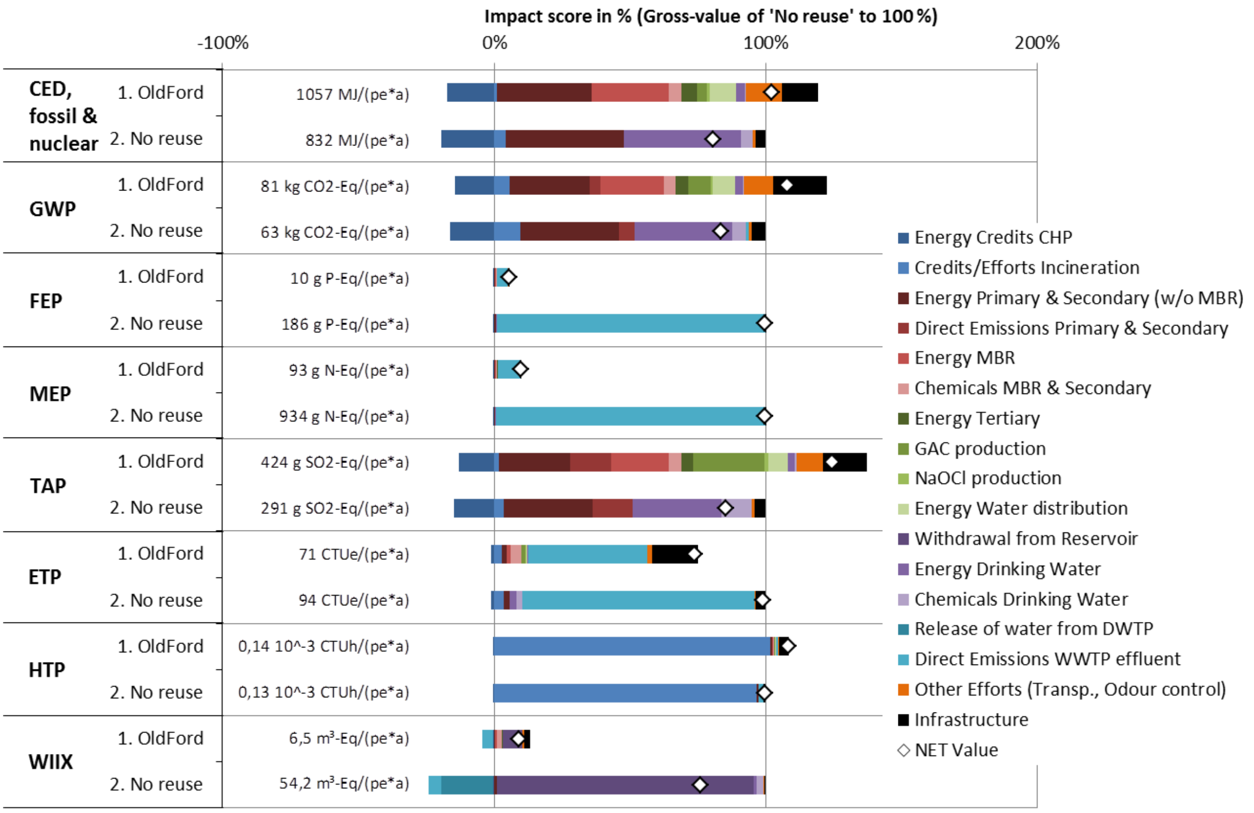


Figure 4-5: Environmental profile for all scenarios related to gross-value of '2 No reuse' (= 100 %) and total net values per scenario and impact category

CED = cumulative energy demand; GWP = global warming potential; FEP = freshwater eutrophication potential; MEP = marine eutrophication potential; TAP = terrestrial acidification potential, ETP = ecotoxicity potential; HTP = human toxicity potential; WIIX = water impact index

The main drivers regarding CED for the reuse scenario 'OldFord' are electricity demand for primary treatment, aeration in the activated sludge process, and the operation of the MBR. In addition, water pumping in the reclaimed water network, the odour control plant and material demand in infrastructure also show relevant contributions to the gross CED. The energy demand of tertiary treatment (GAC filter and chlorination) is comparably low for the reuse system. CED of conventional scenario 'No reuse' is determined by electricity for wastewater and drinking water treatment, while energy demand for chemicals in drinking water treatment is comparably low. For both scenarios, electricity production via CHP from biogas of the anaerobic digester is accounted as credit. This credit is higher for the scenario 'No reuse', because the reuse train produces less excess sludge in the MBR due to the high sludge age. Sludge disposal requires net energy in mono-incineration[8], which is again higher for the 'No reuse' due to higher amount of sludge. Overall, the CED of the OldFord WRP is estimated to 1 057 MJ/ (pe*a) or 13.47 MJ/ m³ water, which is 27 % higher than the conventional system with wastewater and drinking water treatment.

Direct emissions of the WWTP processes (i.e. N_2O and NH_3 from the activated sludge tank) or during sludge incineration are minor in total GWP or TAP compared to the indirect emissions by electricity production. This is due to the high share of fossil fuels in the UK power mix, where 90 % of the electricity is produced from non-renewable energy resources. However, relevant direct emissions for GWP are contributed by N_2O emissions during sludge mono-incineration. For TAP, direct ammonia emissions from the activated sludge tank (stripping of incoming NH_4) are also relevant for impact assessment. Besides direct emissions and electricity, the production of GAC is also associated with relevant emissions in GWP and TAP, e.g. due to significant SO_2 emissions in the production step of activated carbon or related CO_2 emissions in energy-intensive activated carbon production. In total, the OldFord WRP has a net impact in GWP of 81 kg CO_2-eq/ (pe*a) or 1.04 kg CO_2-eq/ m³ water, accounting for a 29 % higher impact in carbon footprint than a conventional system. For the TAP, net impact of OldFord (424 g SO_2-eq/(pe*a)) is even higher with +46% compared to the reference. High contribution of GAC and infrastructure and higher electricity consumption are mainly responsible for the high TAP score of the reuse system.

For water quality, the eutrophication indicators show a significantly lower eutrophication potential for the water reuse scheme, since the WWTP effluent of the OldFord WRP is not directly discharged into surface water. Diverting these nutrient flows from surface or marine water towards irrigation, the release of these nutrients in the Olympic park results in significantly lower eutrophication potentials for the reuse system. In fact, only a small fraction of total phosphorus and nitrogen could eventually be transferred with irrigation water via soil and groundwater into surface or marine environments. In addition, the large-scale WWTP is not equipped with an enhanced P elimination step, which further increases freshwater eutrophication potential (FEP) of the conventional scenario. For the water reuse system, an overall FEP of 10 g P-eq/(pe*a) and a marine eutrophication potential (MEP) of 92 g N-eq/ (pe*a) was calculated, which is only 6 % of FEP and 10 % of MEP of the conventional treatment plant, respectively.

Ecotoxicity potential (ETP) is mainly influenced by the input of heavy metals (Cu, Zn) into surface water with WWTP effluent. As metal removal is higher in the reuse system, ETP of this scenario is considerably lower, because metals are bound in sewage sludge and end up in incineration ash. Human toxicity potential (HTP) is mainly influenced by mercury emissions to air in the mono-incineration process, which relies on mass balance estimates in this LCA.

Water footprinting with the WIIX shows the high reduction of local water stress due to water reuse: whereas WIIX of the conventional system is mainly determined by freshwater withdrawal from the reservoir, water reuse can significantly reduce freshwater withdrawal, only requiring small amounts of drinking water for network flushing. Water release in the environment is of lower quality (e.g. effluent of the large-scale WWTP, reclaimed water) for both systems, and the resulting WIIX credit for water release is higher in the conventional system due to high quality of drinking water compared to reclaimed water. However, credits for irrigation water only account for 25% of the volume, as 75% of the water will be evaporated or incorporated into the plants. Overall, the OldFord WRP has a net WIIX of 6.5 m³-eq/(pe*a) or 0.08 m³-eq/m³ water and thereby only 12 % of the WIIX of a conventional system (0.67 m³-eq/m³ water), reducing the water footprint by 88%.

[8] Mono-incineration requires net energy due to consumption of additives although it produces electricity (see Table 4-2=

Normalization

The score for each impact category per pe (Figure 4-5) related to the normalization data (Table 9-1) per EU-27 citizen is shown in Figure 4-6.

CED, GWP, TAP and ETP contribute approximately 1-2 % to the gross impact per citizen in the EU-27, showing that energy-related impacts of water treatment are only minor contributors to the total environmental footprint of each person. In terms of water quality, FEP and MEP show a higher contribution which is obviously connected to wastewater management being a significant source of nutrient emissions into the environment. Again, it can be observed that water reuse can significantly reduce negative effects of wastewater management on water quality by reducing the risk of freshwater or marine eutrophication due to nutrient emissions in WWTP effluent. The effect is very significant for freshwater eutrophication, as the large-scale conventional WWTP is not equipped with enhanced P elimination and thus emits a considerable load of P into the river Thames. It has to be noted that only 50% of P emissions in the WWTP effluent are accounted here, as the Thames in this area is already affected by tidal flows from the North Sea and could be seen as partially marine environment. Larger normalised scores for HTP in both scenarios are mainly caused by Hg emissions in mono-incineration with off-gas, but this score has to be critically reflected due to high uncertainties in HTP characterization factors which are further discussed in Chapter 8.4.

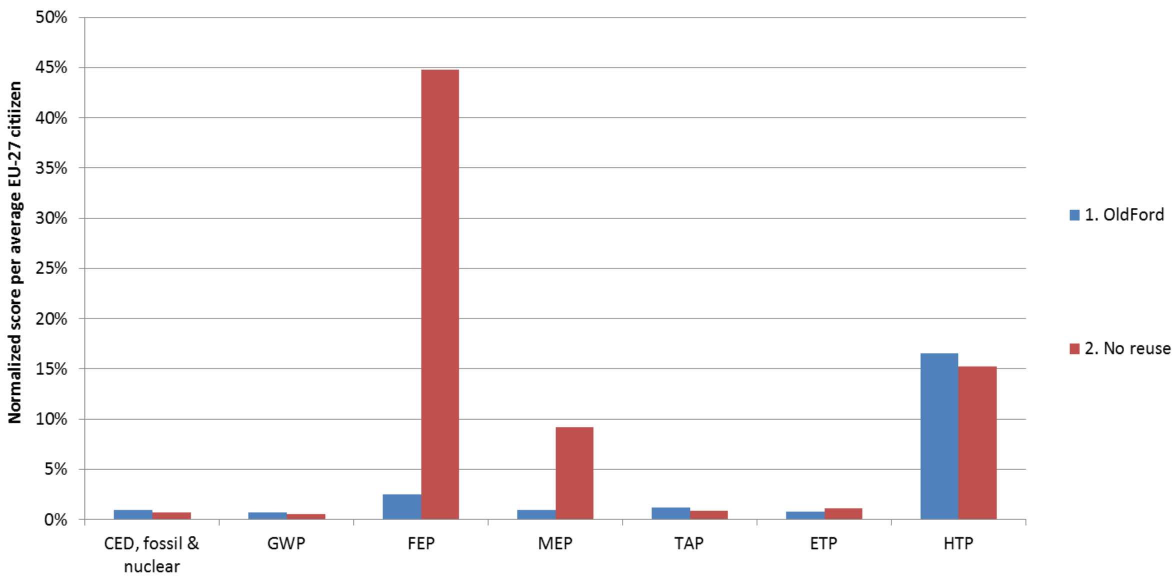


Figure 4-6: Normalized scores for all impact categories per average EU-27 citizen

4.2.4 Interpretation and Discussion

Summary and interpretation of results

Table 4-6 gives a summary on the net environmental efforts and benefits of both scenarios for all impact categories, compared to the conventional system ‘2 No reuse’ as a benchmark.

Table 4-6: Summary of net environmental efforts and benefits of the scenarios for all impact categories, related to '2 No reuse' as reference

Scenario	2. No reuse	1. OldFord
Cumulative energy demand, fossil & nuclear	832 MJ/(pe*a)	+ 27 %
Global warming potential	63kg CO_2-Eq/(pe*a)	+ 29 %
Freshwater eutrophication potential	186 g P-Eq/(pe*a)	- 94 %
Marine eutrophication potential	934 g N-Eq/(pe*a)	- 90 %
Terrestrial acidification potential	291 g SO_2-Eq/(pe*a)	+ 46 %
Ecotoxicity potential	94 CTUe/(pe*a)	- 25 %
Human toxicity potential	0.13 10^{-3} CTUh/(pe*a)	+ 9 %
Water Impact Index	54.2 m^3-Eq/(pe*a)	- 88 %

Overall, the water reuse system at OldFord WRP increases the energetic efforts and associated greenhouse gas emissions of water management compared to a conventional system (+27-29 %), mainly due to the high energy demand for the MBR system. In addition, energy and material demand for MBR and GAC process also causes higher TAP (+46%) of the reuse scheme compared to the reference. However, water reuse also has distinct environmental benefits, namely reducing water footprint by 88% and nutrient-related pollution of freshwater and marine environments by more than 90%. However, it has to be noted that the conventional WWTP can still be optimised in terms of nutrient removal, thus improving the quality of WWTP effluent and decreasing the relative benefit of the reuse scheme in this comparison.

For the reuse scheme, reduction of electricity demand in MBR treatment should be targeted to minimize environmental impacts of reclaimed water production. The combination of an activated sludge process with sludge and effluent recycling for denitrification and a downstream MBR unit acting as a compact clarifier seems to be energetically unfavourable for the reuse system. It may be questioned if an enhanced removal of nitrogen and phosphorus is required if reclaimed water is used for park irrigation, as applied reuse standards for unrestricted urban reuse do not require nutrient removal. Nutrient content may even be seen as beneficial for the irrigation of plants in the park.

For the entire LCA study, it has to be underlined that a small-scale system such as Old Ford WRP (< 1000 m^3/d) is compared here to a large-scale system (i.e. existing WWTP and DWTP in London with > 100'000 m^3/d). Hence, the differences in energy demand may also arise from a scale effect, comparing small and large systems. However, this represents the current reality at Old Ford WRP and illustrates the difficulties to introduce new systems for water management or water reuse which have to compete against optimized large systems in a conventional scheme. In the current state and given the relatively low water scarcity in the London area, the reuse scheme tested in Old Ford WRP is not fully favourable from an environmental point of view due to its high energy demand, which may be a major challenge for future wider implementation of these schemes in the London area.

4.3 Risk Assessment

The microbiological risk assessment (Figure 4-7) starts with the selection and characterisation of relevant pathogens (Chapter 4.3.1). Consequently, for each pathogen multiple input parameters (Chapter 4.3.4) are required for performing the health risk calculation, which results are presented in Chapter 4.3.4.

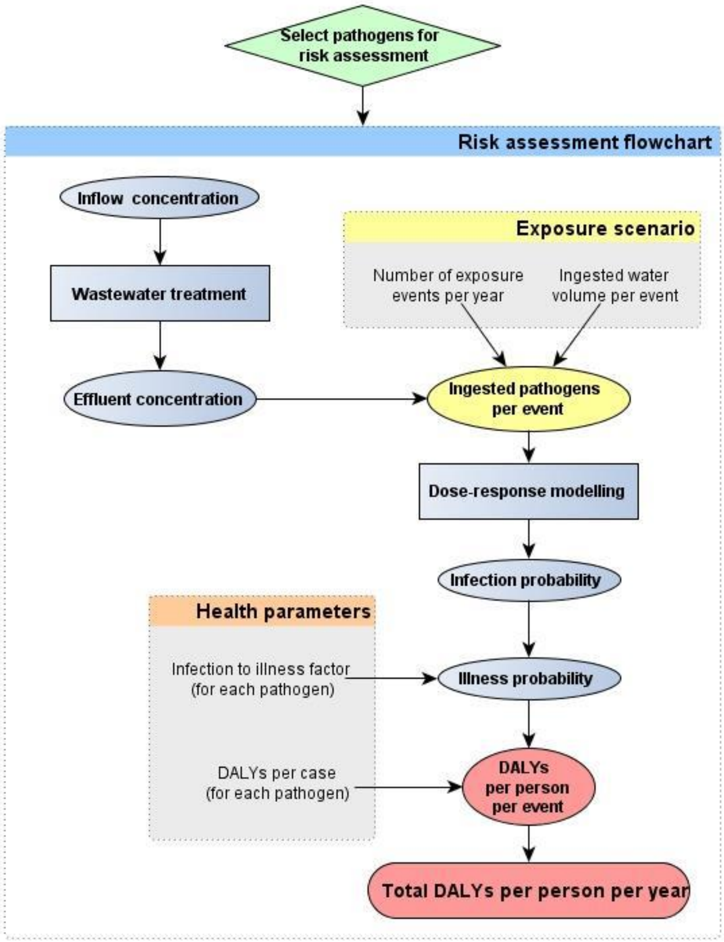


Figure 4-7: Risk assessment flowchart

4.3.1 Hazard selection & characterisation

Campylobacter jejuni, Rotavirus and Cryptosporidium were used as reference pathogens. Epidemiological data from UK and Wales for the period 2002 – 2012 shows (Figure 4-8), that Clampylobacter is the most infectious (median 51932 cases/year), whilst Rotavirus (median 15059 cases/year) and Cryptosporidium (median 3853 cases/year) are much less infectious.

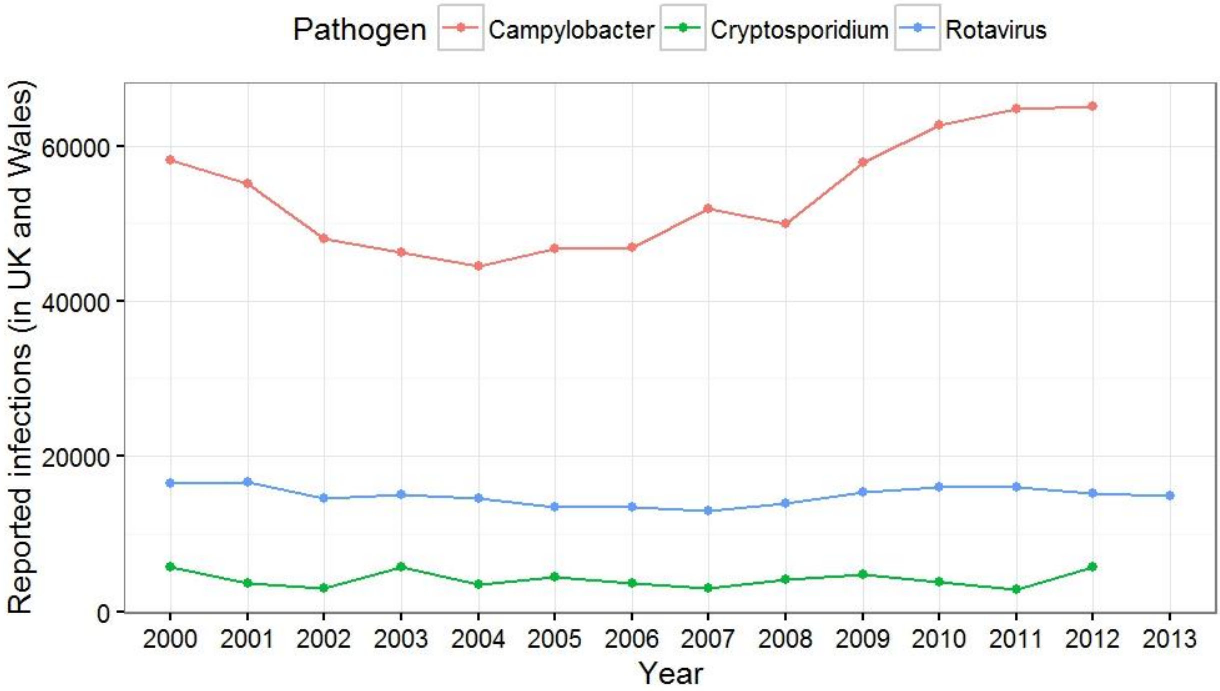


Figure 4-8: Epidemiological data from UK and Wales for the three selected reference pathogens (Source: GOV.UK: Public Health England)

Campylobacter

Pathogens of the genus Campylobacter (C.) are gram-negative rod shaped bacteria. Until now there are over 20 species known. The most important pathogenic species are C.jenuni, C.coli and C.lari.

C.jenuni and C.coli are globally ubiquitous. The bacteria colonize a broad spectrum of animals including. The main reservoir is the digestive tract of warm-blooded animals, especially cattle (dogs, cats and pigs) and birds Campylobacter bacteria are able to survive in the environment for a certain time but are not capable of multiplying outside the host.

The main route of human infection is via food consumption, especially due to the consumption of poultry. Infections also occur due to the consumption of and bathing in contaminated water. Human to human infections are rather rare.

Rotavirus

Rotaviruses belong to the family of the Reoviridae and are the main cause of severe gastroenteritis in children. The main routes of infection are human-to-human transmissions as well as the consumption of contaminated food and water. [81]

Cryptosporidium

Cryptosporidium is a genus of protozoa recognised as a major cause of diarrhoeal illness, contributing significantly to the global burden of gastroenteritis, especially in young children. *Cryptosporidium* occurs worldwide but infection is especially prevalent where drinking water quality and sanitation are poor, and is most significant clinically in young children, malnourished people and immunocompromised patients.

The oocyst stage of the life cycle is shed in faeces of humans and animals and survives many environmental conditions and disinfectants. Oocysts have been detected in surface and ground waters, drinking water, wastewaters, treated and untreated recreational waters. These can be transport vehicles from infected to susceptible hosts, in addition to direct transmission through person-to-person and animal contact. The most common species infecting humans are *C. hominis*, *C. parvum* and *C. meleagridis*, with geographic differences in species and subtype distribution. Prevention and control measures include personal hygiene, effective sanitation and drinking water protection and treatment. There is a lack of effective specific therapy and no vaccine. [99]

4.3.2 Microbiological monitoring data

In total 1820 microbiological samples (data period: 2011-11-17 – 2016-03-30, samples labelled in the raw data as “presumptive” are not considered) are available for the multiple organisms as shown in Figure 4-9. However, out of the three reference pathogens used for risk assessment only for *Cryptosporidium* measurements are available, but without monitoring WWTP inflow concentrations. Thus it is not possible to evaluate the WWTP performance concerning this pathogen.

In general, only 0.55 % (10 out of 1820) of the microbiological samples taken in the WWTP inflow with 2470 MPN / 100ml for all samples of Coliform and E. Coli. However, this value does not represent the real concentration, because it is based on the neat (undiluted) concentration and exceeds the maximum detection limit.

In a nutshell the available microbiological monitoring data were not sufficient for evaluating the log-removal of the WWTP for the three selected pathogens (*Campylobacter*,*Cryptosporidium* and *Rotavirus*) because no samples of these key parameters are available. Thus the QMRA is based not based on montitoring data but on literature values for these pathogens, which is described in more detail in the following paragraph.

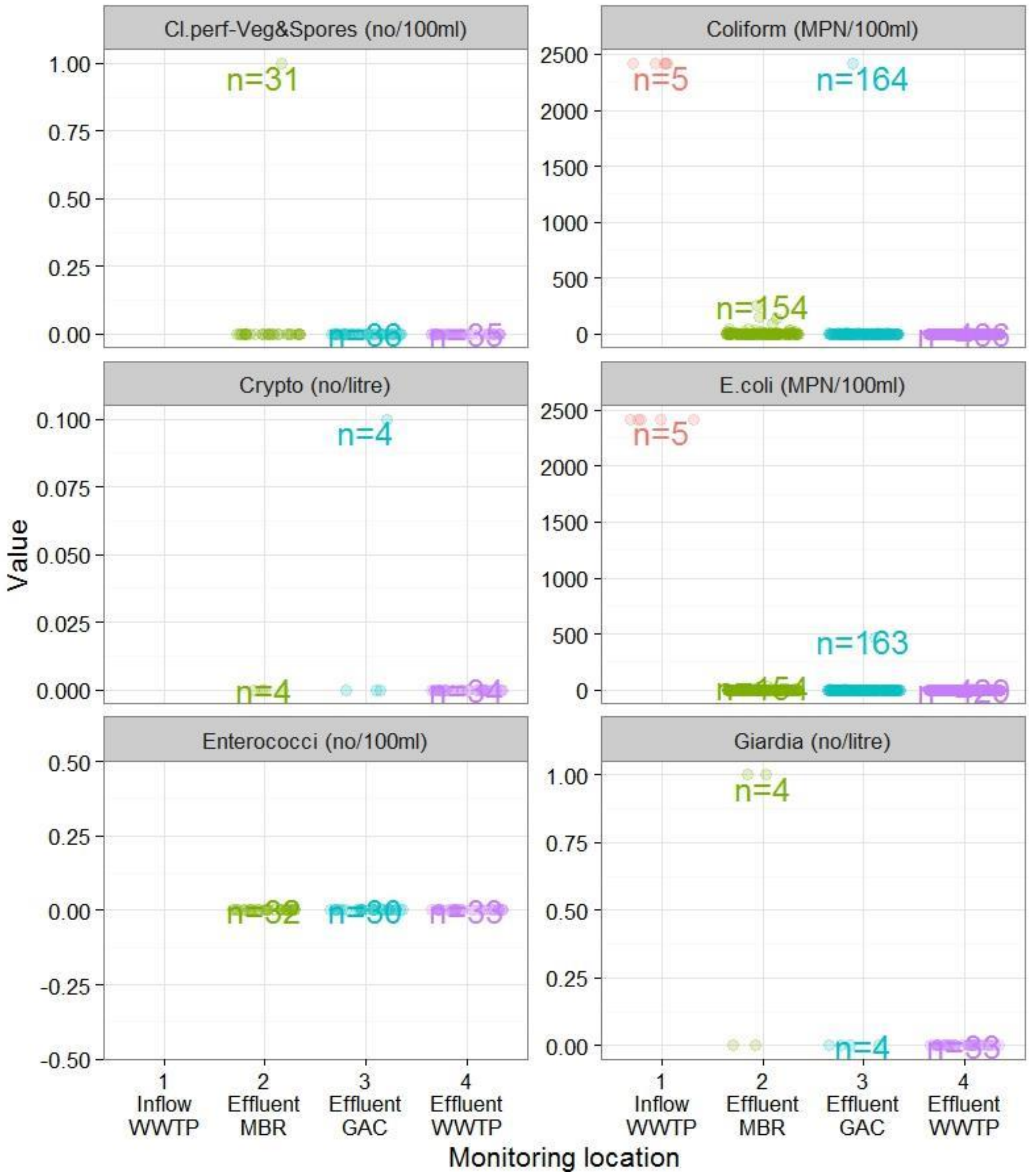


Figure 4-9: Microbiological monitoring data (time period: 2011-11-17 – 2016-03-30) of the Old Ford WWTP. The number of samples for each location and organism is indicated by n

4.3.3 Input parameters

For the risk assessment no monitoring data on the microbial quality of the selected reference pathogens in Old Ford were available for both, wastewater inflow and effluent (see Chapter 4.3.2). Therefore a literature review was conducted to estimate pathogen concentrations as well as removal efficiency of the wastewater treatment plant (WWTP).

Inflow concentrations

The inflow concentrations (Figure 4-10) for the three pathogens are assumed to follow a uniform distribution with minimum and maximum values as defined in Table 4-7. The minimum/maximum values for each pathogen are compiled from literature, because sampling of these pathogens was not performed at the WTTP inflow (see Figure 4-9).

Table 4-7: Typical min/max pathogen concentrations in WWTP inflow based on literature values (WHO [100])

Name	Group	Distribution	Minimum (org./L)	Maximum (org./L)
Campylobacter jejuni	Bacteria	uniform	100	1e+06
Rotavirus	Viruses	uniform	50	5e+03
Cryptosporidium parvum	Protozoa	uniform	1	1e+04

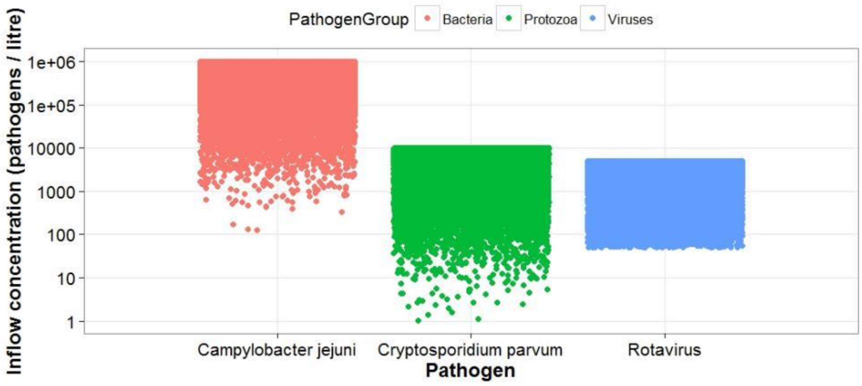


Figure 4-10: WWTP inflow concentrations for the different pathogens
Calculated with R using the built-in function "runif" for generating random uniform distribution for each pathogen with 50000 values and using the min/max parameters listed in Table 4-7

Wastewater treatment plant (WWTP)

The performance ranges of different WWTP processes for reducing the inflow concentrations of the three different pathogen groups (bacteria, viruses and protozoa) are compiled from literature [11] and shown in Table 4-8. For the risk assessment it is further assumed that the reduction performance follows a uniform distribution.

Table 4-8: Treatment process performances [11]

Treatment process	Pathogen group	Log10 reduction	
		Minimum	Maximum
Primary	Bacteria	0	0.5
	Protozoa	0	1
	Viruses	0	0.1
Secondary	Bacteria	1	3
	Protozoa	0.5	1.5
	Viruses	0.5	2
Membrane filtration	Bacteria	3.5	6
	Protozoa	6	6
	Viruses	2.5	6
Chlorination	Bacteria	2	6
	Protozoa	0	1.5
	Viruses	1	3

The treatment scheme of the WWTP Old Ford combines all of the above treatment processes (1st-2nd-MBR-Cl), so that the log10 reductions of all processes need to be summed up. In addition the impact of a partly or total failure of the WWTP Old Ford will be assessed by neglecting some or all treatment processes (Table 4-9).

Table 4-9: Treatment schemes

Treatment schemes	Treatment processes			
	Primary	Secondary	MBR	Chlorination
0				
1st	√			
1st-2nd	√	√		
1st-2nd-MBR	√	√	√	
1st-2nd-MBR-Cl	√	√	√	√

Figure 4-11 shows the total log10 reduction of the different treatment schemes for the three different pathogen groups. The resulting pathogen concentrations in the WWTP effluent can be calculated combining the random inflow concentration c_{inflow} (see Figure 4-10) with the random log-removal log10($WWTP_{reduction}$) for each pathogen in the WWTP (see Figure 4-11) according to the following equation:

Equation 19 $$c_{effluent} = 10^\wedge(log10(c_{inflow}) - log10(WWTP_{reduction}))$$

Note that this approach assumes that the treatment scheme performance is always equal for different pathogens within one pathogen group (e.g. ***Rotavirus*** and *Norovirus)*, which might not be the case in reality.

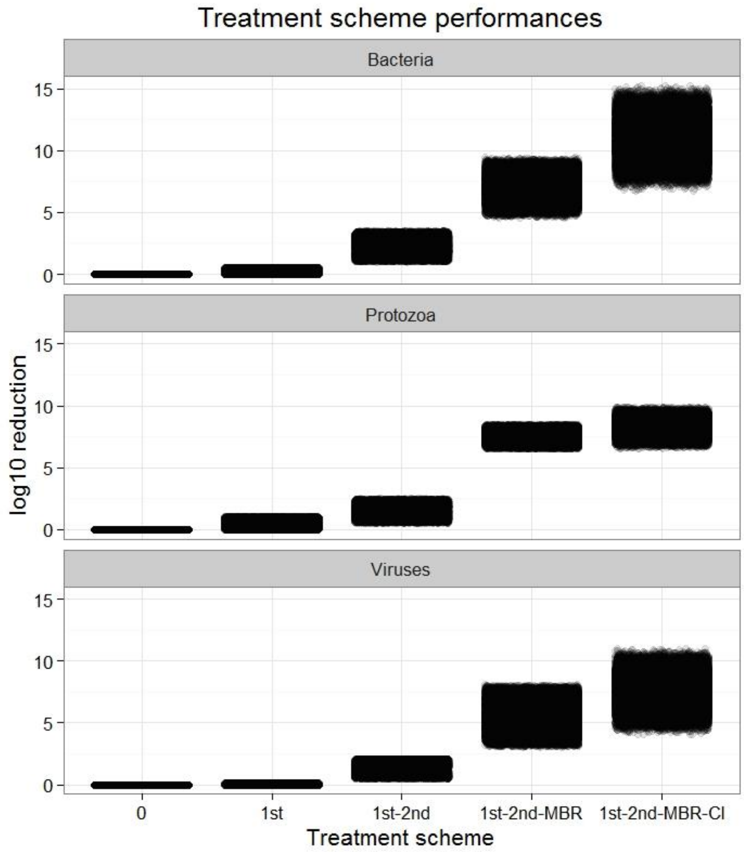


Figure 4-11: Performance of treatment schemes for the different pathogen groups calculated in R by using random uniform distribution for each treatment process
with the min/max log-reduction values of Table 4-8 and finally summing up the results for each treatment scheme (Table 4-9).

Exposure scenarios

The exposure scenarios defined in Table 4-10 were used for performing the risk assessment, which differed only in two parameters:

- **Number of exposure events per year:** determines how often random values for the input parameters inflow concentration and WWTP reduction (following predefined uniform distributions, Table 4-7 and Table 4-8) should be drawn
- **Ingested volume per event:** for all scenarios a constant volume value is assumed for reasons of simplicity

Finally, by multiplying the ingested volume per event with the WWTP effluent concentration (following a random uniform distribution as described in Figure 4-10), for each exposure event the ingested number of pathogens per exposure event is calculated.

Table 4-10: Exposure scenarios based on international guidelines, [101]

Parameters	Scenarios		
	Public irrigation	Toilet flushing	Indirect potable reuse
Exposure events per year	50	1100	365
Ingested volume per event (L / event)	1e-3 (constant value)	1e-5 (constant value)	1 (constant value)

Dose-response modelling

The ingested number of pathogens per exposure event is a required input parameter for performing dose-response modelling, which calculates the infection probability. Both, the selected dose-response model type (exponential or beta-Poisson) and its parameterisation is based on a literature review (Table 4-11). Figure 4-12 shows the resulting infection probabilities different doses for the three assessed pathogens. In case that ingested number of pathogens per exposure event is equal or larger than one, the infection probability can be simply calculated using this approach. However, in case that the ingested pathogen number is below one, the infection probability was calculated by (1) performing dose-response modelling assuming a pathogen dose of one and subsequently (2) multiplying the resulting infection probability with the ingested number of pathogen value [100].

Table 4-11: Dose-response model parameterization [102]

Name	Group	Model	k	alpha	N50	Reference
Campylobacter jejuni	Bacteria	beta-Poisson		0.144	890	Black et al. 1988
Rotavirus	Viruses	beta-Poisson		0.253	6.17	Ward et al. 1986
Cryptosporidium parvum	Protozoa	exponential	0.0572			Messner et al. 2011

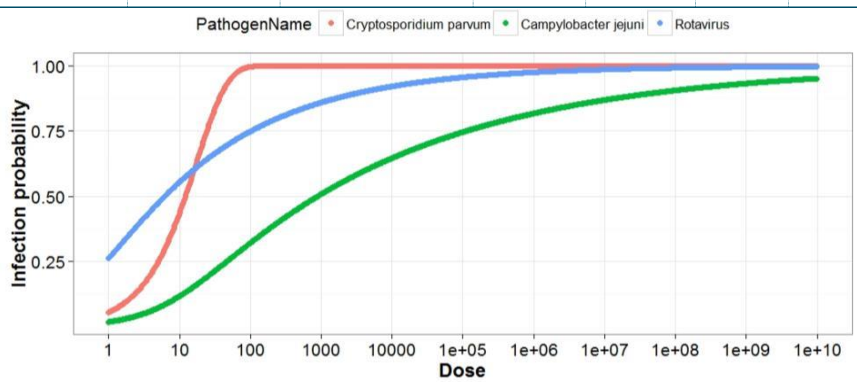


Figure 4-12: Dose-response relationship for the three assessed pathogens

Health parameters

For calculating the health risk of each event, expressed in disability-adjusted life years (DALY), the following two health parameters for each pathogen were required (Table 4-12):

- **Infection to illness factor:** constant factor by which the ***infection probability*** is multiplied for converting it into an ***illness probability***
- **DALYs per case:** expresses the severity of the illness, which needs to be multiplied with the ***illness probability*** for calculating the ***DALYs of each event***

Finally the totals DALY per person per year are calculated by summing up the DALYs of each exposure event for each pathogen, treatment scheme and exposure scenario.

Table 4-12: Health parameterization [100]

Name	Group	Infection to illness factor	DALYs per case
Campylobacter jejuni	Bacteria	0.3	0.0046
Rotavirus	Viruses	0.5	0.0140
Cryptosporidium parvum	Protozoa	0.7	0.0015

4.3.4 Risk characterization

The risk calculation is performed with the software R (www.r-project.org) and the additional R package `kwb.qmra` [103]. The results can be fully reproduced by downloading the source code from [104]. For each exposure scenario, treatment scheme and pathogen, the risk calculation was repeated 1000 times because two input parameters (inflow concentrations, WWTP performance) relied on random sampling from predefined uniform distributions (Monte Carlo method). The resulting risk expressed in disability-adjusted life years per person per year (DALY/pppy) for the different exposure scenarios and treatment schemes is shown in Figure 4-13.

The [WHO [100]] recommends an acceptable threshold of 1 µDALY per person per year, which is satisfied in case of the first two exposure scenarios (public irrigation, toilet flushing) under usual WWTP operation (i.e. all treatment processes working properly; Figure 4-10 treatment scheme: 1st-2nd –MBR-Cl). Only in case of a non-working chlorination (Figure 4-10 treatment scheme: 1st-2nd –MBR), Rotavirus is permanently above the WHO threshold, while the health risk of the two other pathogens is still acceptable. However, in case that only primary and secondary treatment is working properly the health risk for all three assessed pathogens is at least two orders of magnitude above the acceptable WHO threshold.

The third exposure scenario “indirect potable reuse” is a hypothetic one, as this water usage is not implemented yet at the Old Field site. Under these boundary conditions the current WWTP operation will not be able to reduce the health risk below the recommended WHO threshold of. However, at least Cryptrosporidium (median: 1.5 ± 0.15 µDALYs per person per year,Table 4-7) and Campylobacter (median: 3.6 ± 1.3 µDALYs per person per year,Table 4-7) are quite close to this threshold but Rotavirus by more than four orders of magnitude exceeds the health performance target (median: 2402 ± 592 µDALYs per person per year,Table 4-7).

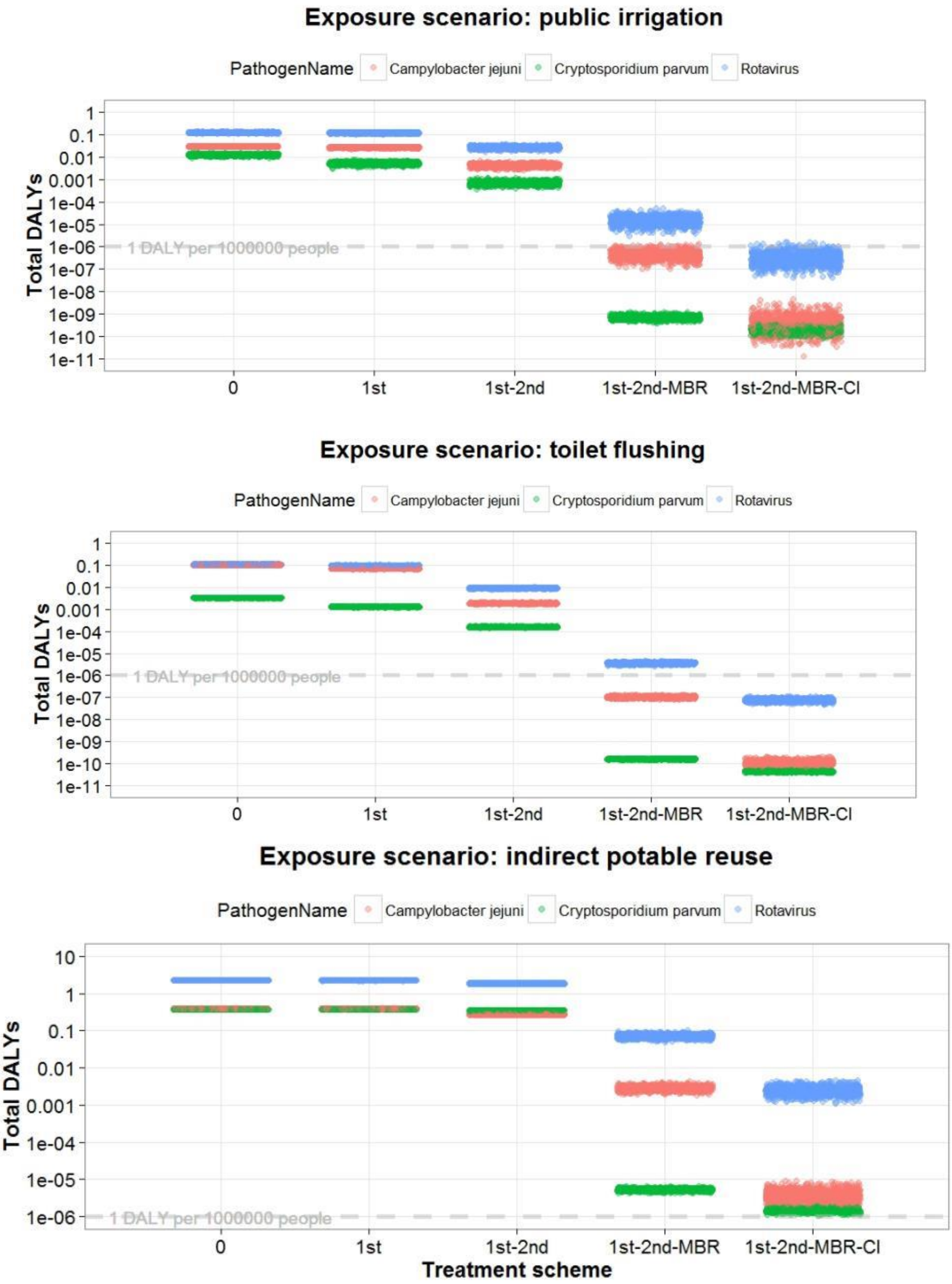


Figure 4-13: Calculated DALYs per person per year for the three different scenarios (Table 4-12) and five treatment schemes (Table 4-11).

The risk calculation was performed 1000 times for each pathogen, treatment scheme and scenario, because it is based on randomised uniform distributions

Table 4-13: Median total DALYs per person per year for the different exposure scenarios & treatment schemes
all values with green and red color are below or above WHO threshold of 1 µDALY per person per year, respectively

Exposure scenario	Pathogen	0	1st	1st-2nd	1st-2nd-MBR	1st-2ndMBR-Cl
Public irrigation	Campylobacter	3.0E-02	2.6E-02	4.3E-03	4.4E-07	3.4E-10
	Cryptospor.	1.3E-02	5.3E-03	6.9E-04	7.1E-10	2.0E-10
	Rotavirus	1.2E-01	1.2E-01	2.7E-02	1.5E-05	2.9E-07
Toilet flushing	Campylobacter	1.0E-01	6.8E-02	1.9E-03	1.0E-07	1.1E-10
	Cryptospor.	3.3E-03	1.3E-03	1.6E-04	1.6E-10	4.4E-11
	Rotavirus	1.1E-01	9.9E-02	9.1E-03	3.6E-06	7.7E-08
Indirect potable reuse	Campylobacter	4.0E-01	3.9E-01	2.7E-01	2.8E-03	3.6E-06
	Cryptospor.	3.8E-01	3.8E-01	3.5E-01	5.3E-06	1.5E-06
	Rotavirus	2.2E+00	2.2E+00	1.9E+00	7.1E-02	2.4E-03

4.4 Conclusions

From the risk assessment of the existing reuse scheme in Old Ford for microbial hazards, the following conclusions can be drawn:

- The available monitoring data show very good microbial quality of the WWTP effluent. However, only Coliforms and E.coli were sampled in the WTTP inflow both with concentrations of 2470 MPN/100ml in the neat , but none of the selected microbial hazards (Clampylobacter, Cryptosporidium and Rotavirus).
- Consequently, for the risk assessment literature values had to be used for both, inflow concentrations and removal efficiencies for all three assessed microbiological hazards. This may lead to an over- or underestimation of the expected risk (depending on the field site characteristics). This uncertainty could only be minimised by the monitoring of all three microbiological hazards in a sufficient spatiotemporal resolution and subsequently using this information as input data for the risk assessment.
- The present treatment scheme of Old Ford is expected to meet the requirements of the WHO guidelines (10^{-6} additional DALYs per person per year) for both scenarios that are currently relevant for this site (public irrigation and toilet flushing). Only in case of a non-working chlorination plant Rotavirus would be permanently above the WHO threshold (median $1.5*10^{-5}$ DALYs per person per year), while the health risk of the two other pathogens is still acceptable (median <= $1*10^{-7}$ DALYs per person per year).
- Only in case of a hypothetic indirect potable reuse scenario, additional treatment barriers would be necessary so that the risk for all three selected microbiogical hazards – especially Rotavirus (median: $2.4*10^{-3}$ DALYs per person per year under present treatment scheme) – are in line with the WHO guideline.

From the environmental assessment of the reuse scheme at OldFord Water Recycling Plant with LCA, the following conclusions can be drawn:

- **Water reuse can significantly reduce local water stress**, reducing the water footprint of the system by 88% due to decreased withdrawal of freshwater from local resources.
- **Water reuse with this configuration (MBR + GAC) increases energy consumption and related greenhouse gas emissions by 27-29% compared to conventional water treatment**, mainly due to additional electricity required for MBR operation. This drawback of water reuse may also be affected by scale effects while comparing the small system of Old Ford WRP to large-scale conventional systems for water management.
- **Water reuse in OldFord WRP reduces nutrient loads in surface or marine waters by diverting treated wastewater towards irrigation or non-potable water use**, thus decreasing the total load of nitrogen (-90%) and phosphorus (-94%) which is emitted into receiving water bodies.

5 Case Study of Sabadell

5.1 Introduction and Setting

Sabadell is a city of 200.000 inhabitants located in the metropolitan area of Barcelona (Catalonia) and crossed by two small rivers (Riu Sec and Riu Ripoll). Drinking water is mainly obtained from the Llobregat River (85%) and from groundwater (15 %). Wastewater treatment is performed in two WWTP, the Riu Sec WWTP and the Riu Ripoll WWTP. CASSA is the company in charge of the whole water management in the city.

Since 2003, Sabadell has promoted the use of non-drinkable water for other uses such as street cleaning or agriculture (with groundwater) or re-establishing the ecological flow of Riu Ripoll (with treated water from the Riu Ripoll WWTP).

In Demoware, the study site is the Riu Sec WWTP, located at the south west of the city (Figure 5-1). This plant has a conventional primary treatment and an advanced secondary treatment that includes tertiary treatment in order to produce reclaimed water (see Figure 5-2). The treatment system has a design capacity of 2500 m^3/h (21.9 hm^3/yr) and features flat-sheet membrane bioreactors (MBR) and disinfection post-step based on UV irradiation and hypochlorite dosing.

Reclaimed wastewater is currently being used for urban purposes, mainly street cleaning and green areas irrigation and for commercial uses such us toilet flushing in a commercial area. However, an ambitious wastewater reuse program is planned. New uses are being promoted in order to supply golf courses in the region for irrigation. For this purpose, a separate distribution network has already been constructed (25 km of non-drinkable water network). Sabadell site in DEMOWARE project is a demonstration site to show that the reuse scheme is not a threat to public health and environment and to increase the confidence, mainly of authorities but also of the general public in water reuse.

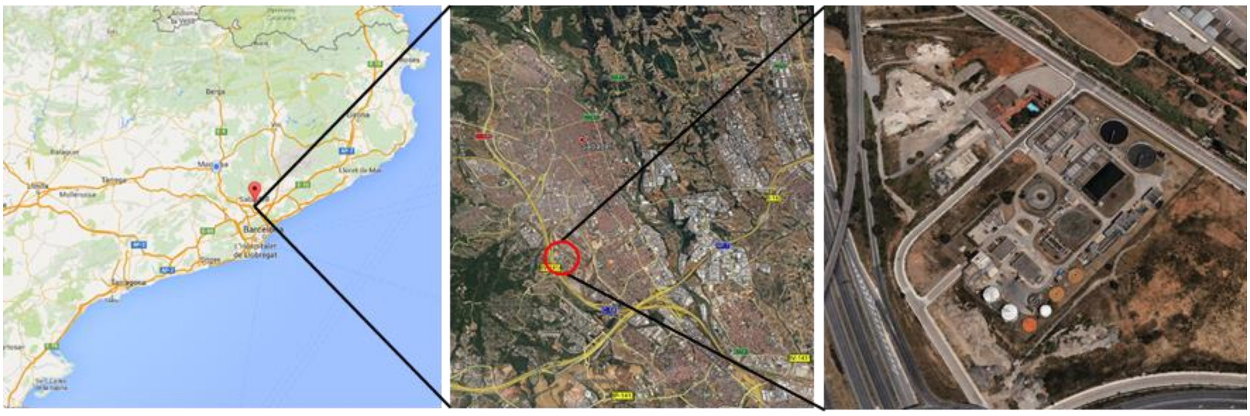


Figure 5-1: Sabadell case study location (Google Earth®)

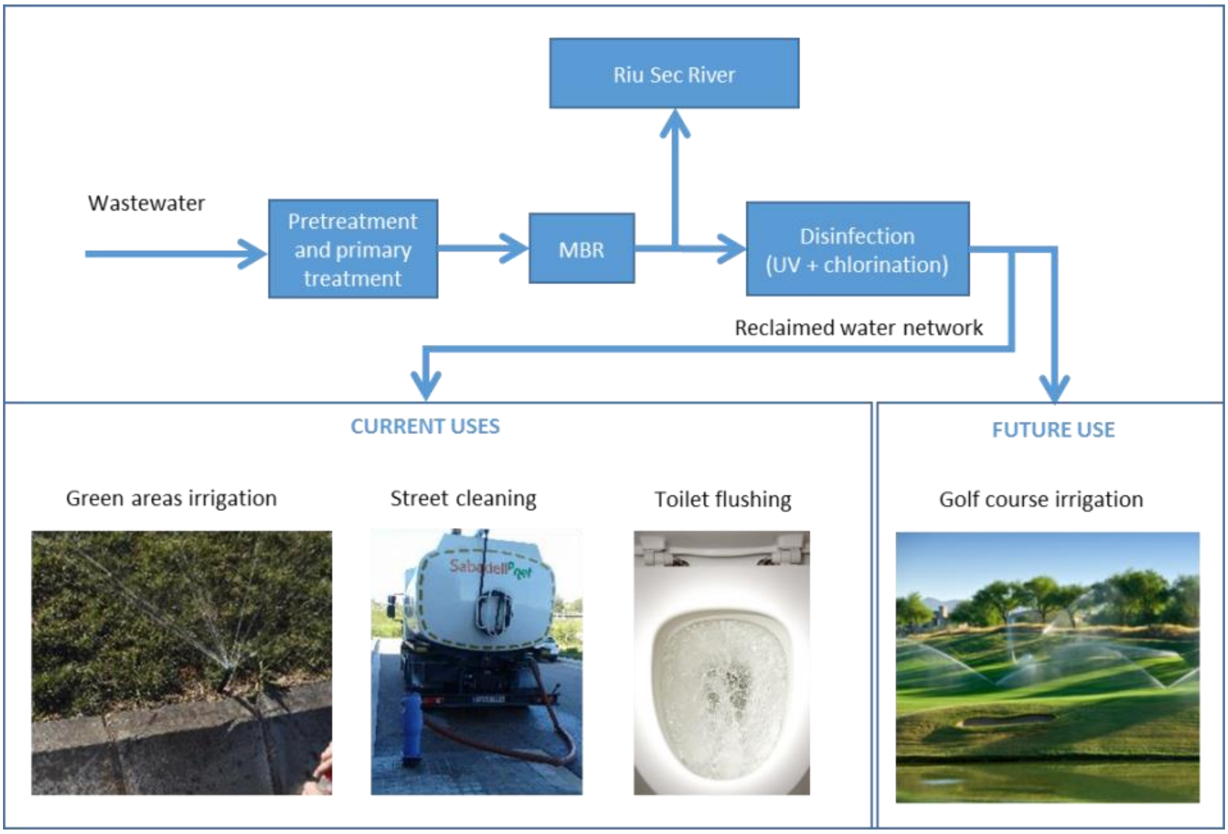


Figure 5-2: Schematic view of Riu Sec WWTP in Sabadell and its reclaimed water uses.

This report shows the human health risk assessment and the environmental assessment of the reuse scheme implemented and Sabadell.

Human health risk assessment is focused both on the chemical risk (human health risk assessment, HHRA) and on the microbiological risk (quantitative microbial risk assessment, QMRA) for the persons that could be in contact with reclaimed water considering its different uses.

Environmental impact assessment, based on Life Cycle Assessment (LCA) methodology is focused on the benefits and drawbacks of the water reuse in general and for the reuse schemes currently implemented.

5.2 Life Cycle Assessment

5.2.1 Goal and scope definition

The goal of this LCA is double: demonstrate the benefits of water reuse in front a non reuse scenario, and evaluate the environmental implications of the reuse of water based in the current reuse schemes of Sabadell WWTP. Additionally, future reuse scenario have been analysed in order to check if an increase in reuse rate can be positive from an environmental point of view. And consequently, the feasibility of increase water to be reclaimed. In this regard, Sabadell currently has a low rate of water reuse, where only 0.21% of total water available is reused. Rest of water is discharged to the Riu Sec River for recreational purposes.

The target group of this study are all stakeholders involved in the water reuse cycle of the municipality: the plant operator, Sabadell local government and obviously the citizens. Also, due to that report will be open to every interested part; the report is also addressed to other agents like planers and engineers in the field of wastewater treatment, policy planners at regional and national level and the scientific community who base their work on LCA, wastewater technologies or both.

Function/ Functional Unit

The function of the system under study is to provide 1 m^3 of water with an optimal quality to be reused at the 3 current uses: irrigation of gardens, street cleaning and toilet flushing in a shopping centre. The quality of the water is defined at the Royal decree 1620/2007 in Annex I.A. and It includes the quality criteria required on the basis of each possible use scenario. Parameters that must always be checked are: intestinal nematodes (1 egg 10 L), escherichia coli (0-10.000 CFU /100 mL), suspended solids (5-35 mg/L) and turbidity (2-15 NTU). Values depend on use. Hence, in order to obtain that quality, Sabadell WWTP has a process scheme train that include among others a MBR unit and a UV-Cl step.

System boundaries

Processes involved are the treatment of wastewater at Sabadell WWTP, all necessary additional processes to send water to the different locations and the processes necessary to use that water in each site. The sub-system WWTP has 4 main stages following the scheme in the Sabadell plant: the pre-treatment stage, MBR, UV-Chlorination (only if water will be reused) and the management of sludge that involves the biogas production and burning in a CHP plant to obtain electricity. That electricity is partially consumed at the plant and the rest is released to the network. Once water is obtained, it can be released to the river for recreational purposes or it is sent via a pipe network to different uses. In these scenarios, pipes, network and transport efforts necessary to supply the reclaimed water are considered. In that boundaries, a series of background process are considered such as energy production, chemicals needed, fuels, materials for the infrastructure, maintenance efforts and direct emission of the plant (Figure 5-3). Below these processes are described in detail.

Impact allocation procedures

All inputs and input and output flows of processes accounted have been allocated to the reference flow stated (total capacity of WWTP) and referred to each subsystem mentioned. Regarding the avoided burden due to the biogas valorisation, all this impact has been allocated to the subsystem "biogas production" due to the lack of data related to how many energy obtained is used in each step of the WWTP. At the same time the environmental implications due to obtain drinking water for the same use is credited as avoided burden for the water reuse scenarios. Hence, a direct balance between environmental impact due to the reuse of water and avoided by tap water is analysed in each scenario.

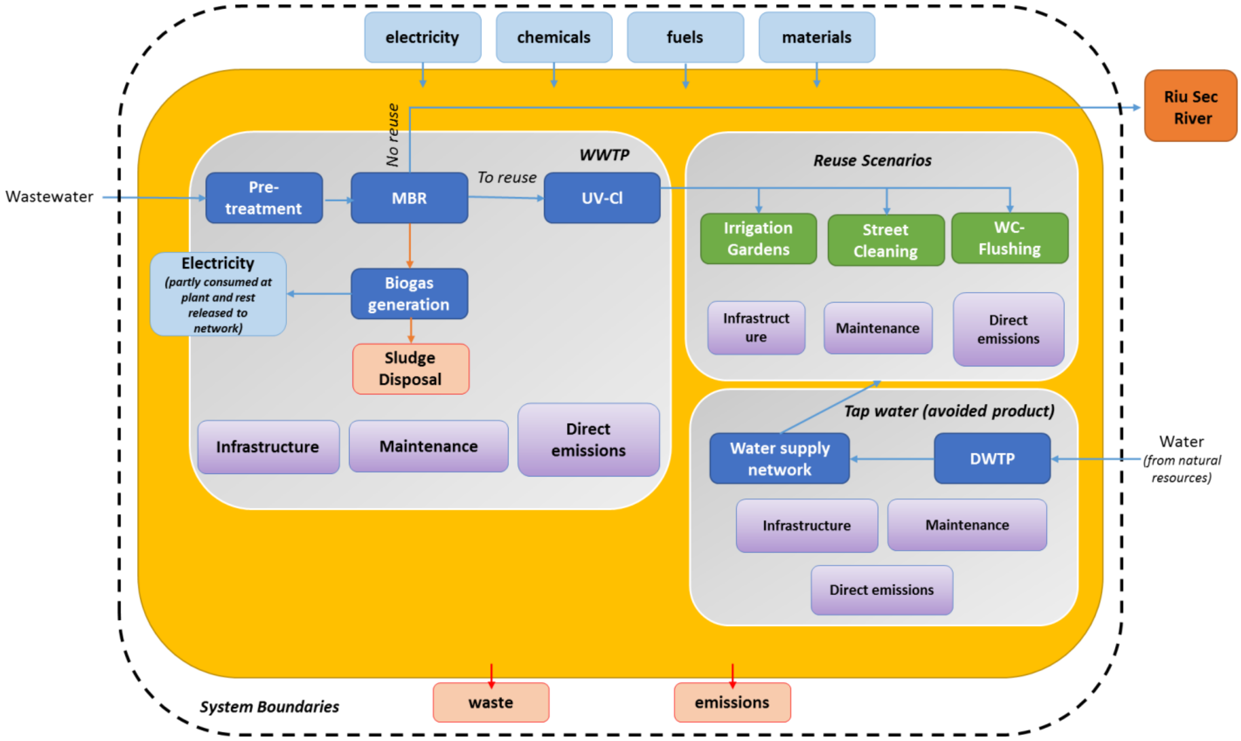


Figure 5-3: System boundaries and scope for LCA Study Sabadell (WWTP: wastewater treatment plant, UV-Cl: ultraviolet-chlorination step)

Scenarios

The scenarios are selected in order to establish a comparison between a scenario scheme where water is released to the Riu Sec river in front of current reuse scenarios implemented in Sabadell. The comparison has been based on a cubic meter of water. Note that is also reflected the avoided impact derived from tap water in reuse scenarios and accounted to for the each entire system: garden irrigation, street cleaning and WC flusing.

1. Baseline represents the impact of obtaining water at the Sabadell's WWTP without the UV-Cl step. That water is released to the Riu Sec River for recreational purposes.
2. Irrigation gardens. After water is treated in Sabadell plant, it is sent to an industrial area near to the plant. Through a piping system and pumped from WWTP, water arrives to the fields, where with the help of a sprinkler system is irrigated in an area of approximately 535 m^2. Previous to this, sodium hypochlorite (NaOCl) is added to the water for assure limit values stated at the national legislation for water reuse (Royal order 1620/2007 of water reuse). Avoided product in that case accounts the production of tap water and the network needed to provide 1 m^3 of water at the point of use as avoided product. Tap water is produced for Sabadell in two drinking water treatment plants (DWTP) located in respectively Abrera (Llobregat river) and Cardedeu (Ter river).
3. Street Cleaning In that scenario water is charged in 4 different vehicles directly at the WWTP and used to clean streets in Sabadell. One tank truck, a hydro street washer and two street sweepers are used to that purpose. The tank truck has to drive 10 km for 3 days a week, to charge water from the reuse site, while the rest of vehicles recharge only once a week. Distances covered when water is released into the city have not been considered. NaOCl consumption and energy for water pumping are also taken into account. Avoided product in that case accounts the production of tap water, the network needed to provide it at the

point of use and the transport needed to charge water. No relevance differences has been stated between distances before and after reclaimed water was used for that purpose. For that reason transport has been considered equal than for the reclaimed water in order to assure study coherence.

4. WC Flushing To a shopping centre near to Sabadell WTTP, water is sent to be flushed in the WC system. In that case, thanks to the pressure achieved at the plant outlet, no energy consumption is needed for pumping. Only piping system to water transport is accounted, among consumption of NaOCl and energy required inside flushing system at the shopping center. Avoided product in that case considers the production of tap water and the network needed to provide it at the point of use.

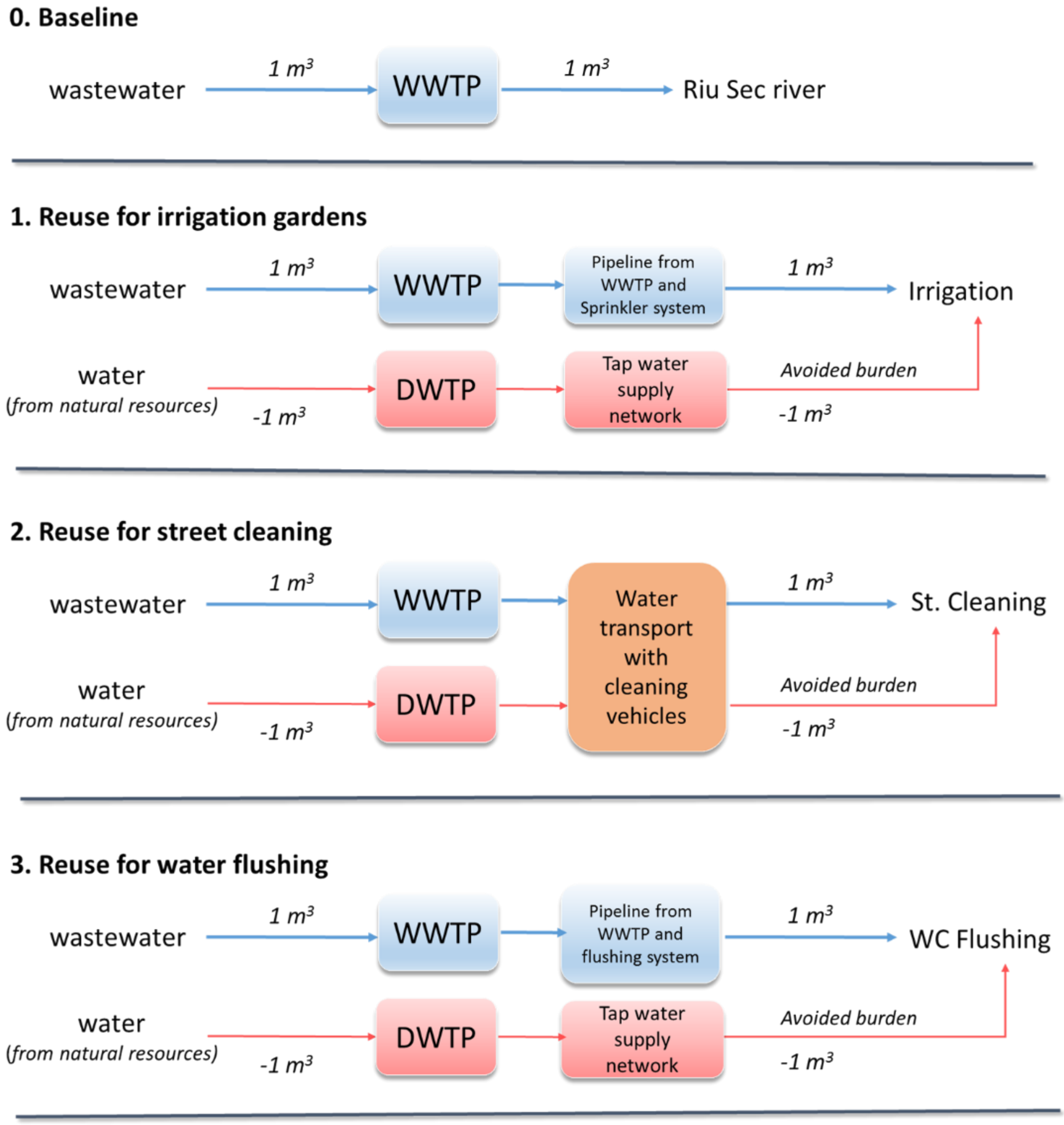


Figure 5-4: Overview of the LCA Scenarios for Sabadell. WWTP: Wastewater Treatment Plant; DWTP Drinking water treatment plant

Data quality and limitations of this study

The following parameters for the LCA inventory are discussed regarding data quality and uncertainties, to clearly point out inherent limitations of this LCA. A summarized overview of data source and data quality is provided in Table 5-1.

- Energy, chemicals and material consumption: The dataset on energy and chemical consumption for the entire WWTP has been recovered from internal registers of Sabadell. Data stated along 2015 were provided by Sabadell owners and questionnaires were completed with average values. In all cases, data referred to regular plant operation and when required, to maintenance period. With that approach, each concept was considered and all data was referred to the total water treated in a year. Thanks to that, uncertainty in data recovered was very limited. In addition, no significant differences were reported by the company due to seasonality. Hence, all these data can be considered of high quality. Existing uncertainties are assumed to be very small.
- Waste management. No evidence about the treatment of sludge generated in plant has been recovered. Hence, general scenario for sludge treatment in Spain has been considered. In this regard, the National Register of Sludge states that in 2014, sludge spreading accounted for an 81% of total wastewater sludge generated in Spain, to landfill 7% and 7% for energy valorisation. Rest (5%) are sent to other uses. For every treatment generic processes of Ecoinvent database have been considered. Due to the national framework has been accounted, it is considered that quality level of data incorporated is good.
- Water quality within WWTP: Data for standard parameters of the plant has been recovered both from plant registers and monitoring campaigns held within DEMOWARE project. Furthermore, these quality parameters have been determined not only before and after the plant, but step by step along different treatments. Hence, a complete study regarding the pollution abatement directly allocated for all steps has been performed. It has been carried out following the net environmental benefit approach stated by Godin et al (2011). Assessment accounts potential impact of releasing wastewater without and with treatment besides assessing the impact of the WWTP's life cycle. Hence, environmental balance between avoided impact due to wastewater treatment and generated impact by the WWTP's life cycle is considered at the study. It is especially relevant for t-N and t-P parameters in FEU and MEU categories as it is detailed below. All these data can be consequently considered as very good.
- Water balance at the WWTP: There is no evidence of water losses according to data provided by the company. Hence, all withdrawal and treated water is released at the end of the plant. It could represent an unequivocal limitation of the study mainly for the water footprint assessment. However, the withdrawal water accounted in the plant according to the methodology employed has to be considered 0.
- Direct emissions. Direct emissions generated in plant like NO_x, CO_2 etc. from different units like the digester gas has been estimated with the help of calculation tool for municipal wastewater treatment plant performed by Doka (2003) included in the Ecoinvent Database v2.1. This represents a source of uncertainty in the study. However, as it is expressed in final results, direct emissions do not represent a great source of impact along the plant and for impact categories reflected.
- Capital goods for the WWTP: Data regarding capital goods at the WWTP has been estimated by general assumptions based in geometry and common material composition according to technical datasheets of different providers and other studies (Cashman et al, 2014) (Remy, 2013). Where available, other considerations based on specific databases like Ecoinvent have been considered. These assumptions result in a higher uncertainty of the plant.

Table 5-1: overview on data quality of input data

Parameter/Process	Data source	Data quality
Reference system: WWTP Riu Sec		
Capital goods	(Cashman et al, 2014) (Remy, 2013) Own estimations, commercial providers	low
Water quality	Plant owners	very good
Water balance	Plant owners	medium
Energy, chemicals and material consumption	Plant owners	very good
Direct emissions	Doka (2003)	low
Waste management (sludge)	National statistics	medium
Reuse scenarios		
Energy for pumping and NaOCl dosage	Plant owners	Very good
Energy & capital goods for irrigation gardens	Estevez-Olea (2015)	medium
Energy, capital goods and transport for street cleaning	Plant owners, own estimations and commercial providers	low-medium
Capital goods for WC flushing	Plant owners, personal communication with engineering expert	low-medium
Avoided product DWTP and water supply network		
Energy, chemicals and material consumption	Environmental declaration (ATLL)	good
Chemicals and materials	Environmental declaration (ATLL)	good
Water supply network	Environmental declaration (ATLL) and open public information	medium
Background		
Electricity mix	Mix of Spain 2010	medium
Chemicals and materials	EU or global datasets	medium
Transport	Truck transport (EU)	good

- Reuse scenarios: Data for infrastructure has been estimated according to distances from WWTP to point uses of water. Piping system material and diameter have been recovered from plant owners. Based in it, weight of pipes per km and consequently, quantity of material needed have been estimated based on public info from different commercial providers.At the scenario of irrigation gardens, energy consumption and capital goods for sprinkler system has been estimated according to Estevez-Olea (2015). Inventory data of the system capital goods has been recovered from the generic process included at the Australian National Life Cycle Inventory Database (AusLCI). In the street cleaning system, vehicles identified are a tank truck, a hydro street washer and two street sweepers. It was checked technical datasets of similar vehicles, considering total weight, motor power and fuel consumed (diesel). After that, it was considered a 7,5-16 ton freight truck for the two street sweepers and the tank truck and a 3.5-7.5 ton freight truck for the hydro street washer as proxys. These generic processes come from the Ecoinvent database and takes into account manufacturing and operation. Finally, at the WC flushing system, pipeline inside shopping centre which has an extension of circa 40.000 m^2, has been estimated accounting a rate of 0.5 m pipe/m^2. This rate is according to personal communication with an expert engineering firm in the installation of that kind of systems.According to the description it is considered that data quality for reuse scenarios is medium.

- **Drinking water treatment plant:** Information about the DWTP held in Sabadell was obtained from the environmental declaration published by ATLL (Aigües del Ter-Llobregat). It is the company that operates the two plants that provide drinking water to Sabadell. Data about the withdrawal and released water, energy consumption at the plant, reagents used (starch, $NaClO_4$, polyelectrolyte, etc.) and waste have been accounted. Capital goods have been estimated from generic processes of Ecoinvent database. It has been also recovered data about the energy consumption for the distribution. Water provided by ETAP to Sabadell city and length of supply network in order to estimate impact from distribution has been obtained from Sabadell council website. (http://ca.sabadell.cat/Aigua/p/abastament_cat.asp). Having into account all described it is considered that data accounted are very good. Only for capital goods needed where once again generic processes from Ecoinvent database have been accounted are important sources of uncertainty.

Normalization

As stated, current reuse rate of water in Sabadell is very low. If it is considered a normalization factor based in population affected, a big uncertainty will be introduced into the study. For that reason, it has been rejected to express results for inhabitant.

5.2.2 Inventory (Input data)

Inventory data for the LCA study were provided by Sabadell Riu Sec WWTP. A detailed energy balance is shown in Table 9-16. For consumptives, Table 5-2 summarizes the materials, reagents and capital goods accounted for each scenario. Energy requirements for all scenarios are detailed in Table 5-3. Finally, at Table 5-4 is shown the inventory data accounted for the avoided product of tap water considered in scenarios 1-3. Furthermore, data regarding the waste management for sludge is also shown.

Table 5-2: Inventory data for materials, reagents and capital goods in the different scenarios

Material	0. Baseline	1Irrigation Gardens	2.Street cleaning	3.WC Flushing	Comments
Concrete for WWTP (kg)	111330	125246	125246	125246	Estimation based in total area and volume occupied by infraestructure, and studies performed by Cashman et al (2014) and Remy (2013). Pumps and piping system is also accounted
Steel for WWTP (kg)	31791	35765	35765	35765	
Sand for WWTP (kg)	137240	137240	137240	137240	
HDPE for WWTP (kg)	1527	1718	1718	1718	
Other plastics (PVC, PU, PET, etc,) (kg)	418	471	471	471	
UV lamps (units)	--	5	5	5	Primary data
NaOCl (35 %) (kg/m3)	0.03	0.16	--	0.09	Primary data
NaOCl (15 %) (kg/m3)	--	--	0.02	--	Primary data
Sprinkler system (m^2 area)	--	535 m^2	--	--	HDPE: 8,9 kg/m^2, LDPE: 5,4 kg/m^2, PVC: 5,8 kg/m^2, Steel: 9,8 kg/m^2 Lifetime: 30 years
Pipe at reuse site for WC flushing	--	--	--	20000 m	Main material PP: 164 kg/km, Lifetime: 30 years, data from local company expert (0.5 m pipe/m^2)
Pumps (pc)	--	3	1	3	To charge vehicles. Primary data. Lifetime: 12 years. Materials asin Halm et al (2003)
Number of vehicles needed	--	--	4	--	A tank truck, a hydro street washer and two street sweepers. Primary data. Generic processes considered, lifetime of vehicles: 540000 km
Distance of truck transport for water (km)	--	--	10	--	From WWTP to the working zone. Primary data.
Piping system from WWTP (HDPE) (m)	--	1500	--	1500	Allocated based in water requirement for each case (65% WC Flushing and 35% irrigation gardens)

Table 5-3: Inventory data for energy requirements

Energy flow	0. Baseline	1Irrigation Gardens	2.Street cleaning	3.WC Flushing	Comments
Electricity for wastewater treatment (kWh/m3)	1.50	1.53	1.53	1.53	No UV-Cl disinfection in baseline scenario
Electricity generated at CHP from biogas valorisation (kWh/m3)	0.41	0.41	0.41	0.41	Anaerobic digestion and biogas usage in CHP plant. Electricity is partially consumed at plant, and rest is fed into the grid
Electricity for pumping (kWh/m3)	--	1.23	--	--	Pumping from WWTP into the sprinkler system, primary data
Electricity for charging vehicles (kWh/m3)	--	--	0.18	--	Pumping from WWTP into vehicles, primary data
Electricity for pumping (kWh/m3)	--	--	--	0.25	Pumping inside WC flushing system, primary data. No pumping required to deliver water from WWTP to building.
Fuel consumption of tanker truck and street sweepers (kg diesel*m^3 water transported*km)	--	--	0.05	--	Generic processes from the Ecoinvent database.
Fuel consumption for hydro street washer (kg diesel*m^3 water transported*km)	--	--	0.11	--	

Table 5-4: Inventory data for waste disposal and avoided burdens

Material	0. Baseline	1Irrigation Gardens	2.Street cleaning	3.WC Flushing	Comments
Starch (kg/m^3)	--	2,42E-05	2,42E-05	2,42E-05	Reagents used at DWTP and accounted as avoided product
$NaClO_4$ (kg/m^3)	--	5,52E-04	5,52E-04	5,52E-04	
$HClO_4$ (kg/m^3)	--	7,79E-04	7,79E-04	7,79E-04	
Polyelectrolyte (kg/m^3)	--	3,93E-03	3,93E-03	3,93E-03	
Cl_2 (kg/m^3)	--	1,95E-04	1,95E-04	1,95E-04	
$AlCl_4$ (kg/m^3)	--	2,67E-03	2,67E-03	2,67E-03	
$KMnO_4$ (kg/m^3)	--	2,48E-05	2,48E-05	2,48E-05	
CO_2 (kg/m^3)	--	2,53E-04	2,53E-04	2,53E-04	
HCl (kg/m^3)	--	1.23E-03	1.23E-03	1.23E-03	
Electricity for DWTP (kWh/m^3)	--	0.52	0.52	0.52	Accounted as avoided product
Tap water supply network (km/m^3)	---	0.03	0.03	0.03	Tap water network. Accounted as avoided product
Fuel consumption of tanker truck and street sweepers (kg diesel*m^3 water transported*km)	--	--	0.05	--	Equal to reclaimed water. Accounted as avoided product
Fuel consumption for hydro street washer (kg diesel*m^3 water transported*km)	--	--	0.11	--	Equal to reclaimed water. Accounted as avoided product
Sludge generated (kg/m^3)	0.26	0.26	0.26	0.26	Sludge disposal: 81% in agriculture, 7% in landfill, 7% in incineration, and 5% to other use (Spanish mean). Avoided product accounted based in generic processes from Ecoinvent

Background data

Information on background processes are shown in the Annex 9.5.1 in Table 9-15.

Water Impact Index Inventory

The WIIX is calculated according to the methodology described in *D3.1* [11]. The water scarcity index (WSI) according to WULCA AWARE [52] is used for calculation. Withdrawals of water and released for direct WIIX calculation is based in Sabadell WWTP. Losses are negligible according to CASSA information and are not considered. Direct WIIX for withdrawal is accounted to 0 as input water does not come from natural resources, but from the technosphere (= city). Released water at the baseline scenario accounts entirely to the environment cause is released to the Riu Sec river and consequently refill water resources.

In the reuse scenarios, effect of avoided released is assumed with drinking water. Furthermore, in the irrigation scenario, it has been considered the same approach that for other case studies. Hence, only 25% of the water reach the groundwater after subtracting for evaporation and plant uptake [51]. From each cubic meter of water reused, 0.25 m^3 is actively released into the environment in terms of replenishing freshwater resources. For street cleaning and WC flushing total balance is 0 due to water released returns to the city.

Table 5-5: Overview on avoided and direct withdrawals, avoided and direct releases and water quality indices (WQI) for the different scenarios.
In irrigation scenario, only 25% of volume are accounted as groundwater recharge

Scenario	0. Baseline	1. Irrigation Gardens	2. Street cleaning	3. WC Flushingk
Withdrawals	0 [m^3/day]	0 m^3	0 m^3	0 m^3
Avoided withdrawal *(water that comes from natural resources*	0 [m^3/day]	-1 m^3	-1 m^3	-1 m^3
WQI (Withdrawals)	*0.02 WWTP* *1 DWTP*	*0.02 WWTP* *1 DWTP*	*0.02 WWTP* *1 DWTP*	*0.02 WWTP* *1 DWTP*
Releases	1 [m^3/day]	0.25 m^3	0 m^3	0 m^3
Avoided release *(drinking water for reuse scenarios)*	0	-0.25 m^3	0 m^3	0 m^3
WQI (Releases)	*0.04 WWTP* *1 DWTP*	*0.04 WWTP* *1 DWTP*	*0.04 WWTP* *1 DWTP*	*0.04 WWTP* *1 DWTP*

The water quality index for withdrawal and released water at the WWTP has been considered based on the intake and effluent water quality parameters directly measured within DEMOWARE project and historical data provided by CASSA. Total Phosphorous and Zn concentrations are considered as limiting parameters for withdrawal and released water respectively according to the existing surface water benchmarks in the EU water framework directive (see detailed inventory in the Annex 9.5.1, Table 9-17).

For tap water as avoided product, an optimal water quality (WQI = 1) is assumed as conservative approximation.

5.2.3 Impact Assessment (Results)

Environmental impacts were assessed with a set of 8 impact categories (including water impact index), representing different areas of environmental concern. After an overview of all indicators, selected impact categories are discussed more in detail to reveal individual contributions of different processes and aggregates to the total environmental impact.

Total environmental impacts and benefits of all scenarios

The environmental profile of all scenarios for all selected impact categories is shown relatively to the gross impact of the existing system '0 Baseline (= ±100 %) in Figure 5-5 and Figure 5-6. Different scenarios are represented for three main parts considered.

Impact of the waste water treatment plant is tagged as "WWTP". At the same time, is noted separately the 5 main stages: Pre-treatment, membrane bioreactor (MBR), UV-chlorination (UV-Cl) and Biogas production from sludge. Among these main stages, it has been reported separately results for the waste management of sewage sludge, construction of the plant, and direct emissions related. The water impact index accounted for the net balance of water at the wastewater treatment plant is tagged as "Direct WIIX". In the reuse scenarios, among the contribution for the treatment of wastewater, is specified the impact of the infrastructure needed to transport water for the different uses, the reagents required to add to the water and the energy consumed for the distribution. Finally, the impact contribution of tap water for both the drinking water treatment plant (DWTP) and the water supply network is considered.

For the water footprint assessment, direct WIIX is tagged separately as the balance between direct withdrawal and release of water. The addition of all the contributions is tagged as "NET value" and it is highlighted in black with no colour.

The fossil and nuclear cumulative energy demand (CED) and the global warming potential (GWP) are categories where energy consumption strongly influence results. In almost all scenarios, direct consumption in plant and energy consumption of vehicles in street cleaning are the main drivers. At the same time, avoided impacts are due to the direct DWTP and distribution network energy requirements. For MEU and FEU, the t-N and t-P removal capacity at the WWTP has a strong contribution for the impact minimization. As mentioned above, the NEB approach stated by Godin et al (2011) has been adopted and its influence is highest in these categories.

Terrestrial acidification potential (TAP) is strongly influenced also by the energy consumption. For that reason, similar contribution of processes are obtained, but with some differences due to increasing importance of background processes, such as chemicals, material production for capital goods, etc.

In ETP once again energy is the main aspect to the impact contribution. However, it is a category strongly influenced by the use of metal resources. Hence, steel in capital goods, metals used and disposed in the manufacture of vehicles or the piping system in WC flushing has a big influence in all scenarios among that energy consumption. Human toxicity indicator (HTP) is mainly influenced by the scenario of sludge spreading and the related application of heavy metals into soil.

Finally, the WIIX is mainly influenced by the volumes of water withdrawals. As a non-natural resource, withdrawal water has no impact. That effect makes avoided impact provided by tap water be the main aspect to impact contribution.

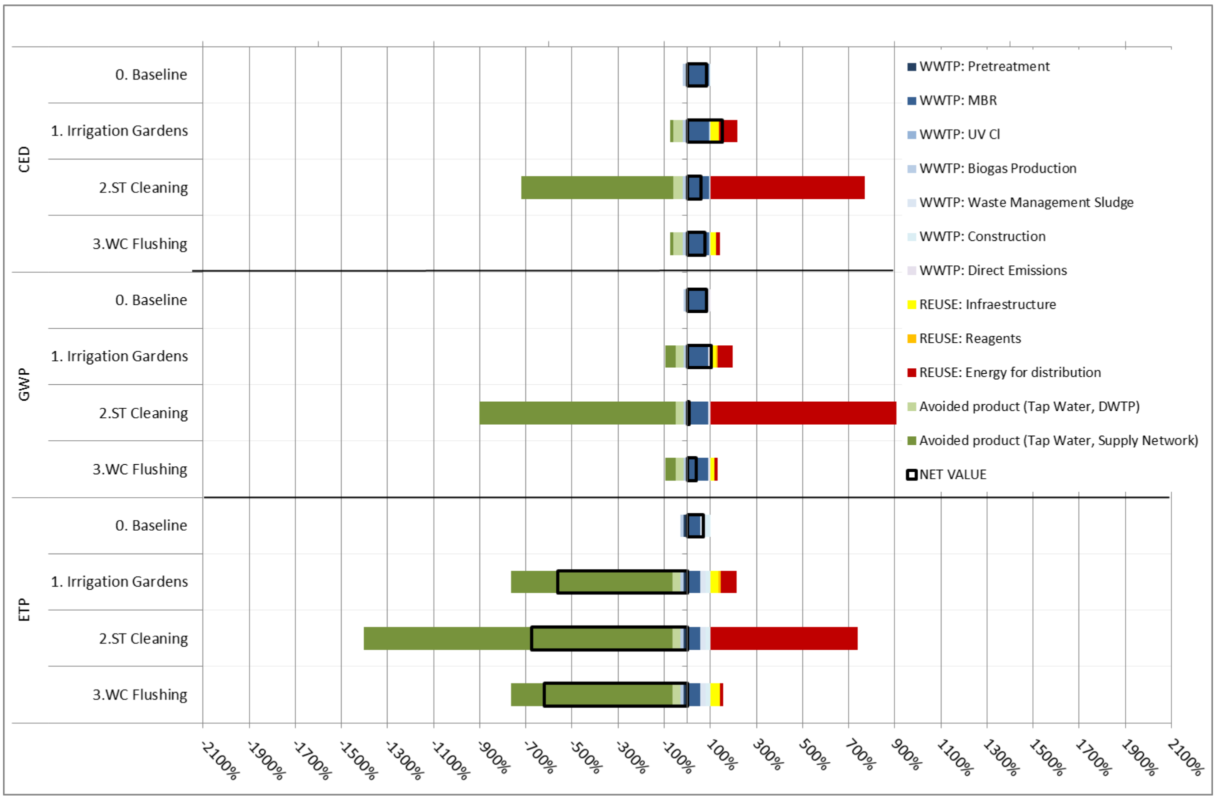
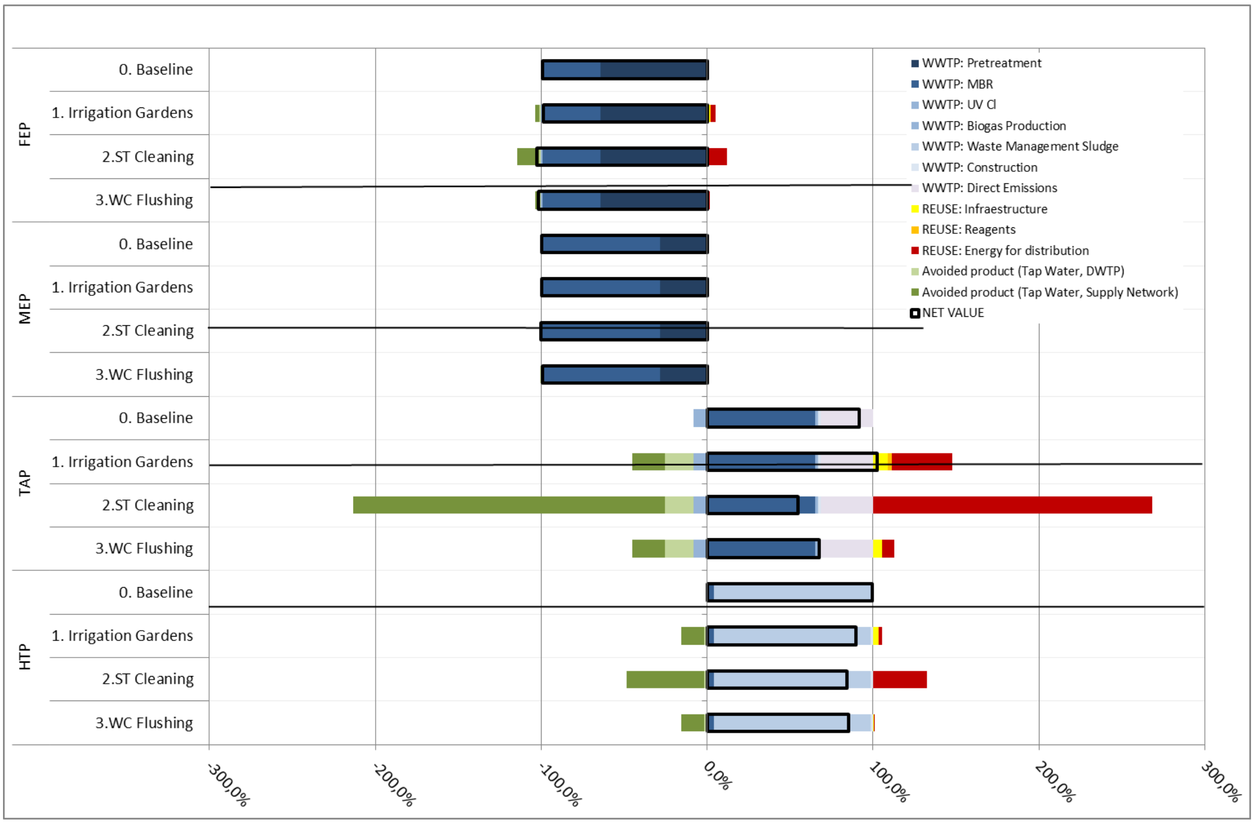


Figure 5-5: Environmental profile for all scenarios related to gross-value of '0 Baseline (= 100 %) and total net values per scenario and impact category

CED = cumulative energy demand; GWP = global warming potential; ETP = eco toxicity potential.



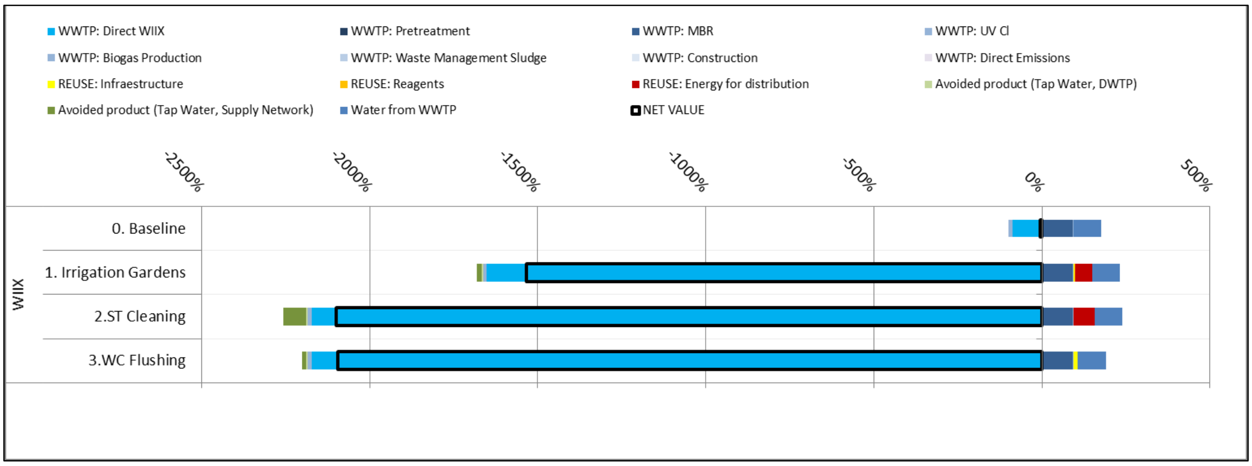


Figure 5-6: Environmental profile for all scenarios related to gross-value of '0 Baseline (= 100 %) and total net values per scenario and impact category

FEP = freshwater eutrophication potential; MEP = marine eutrophication potential; TAP = terrestrial acidification potential, HTP = human toxicity potential; WIIX = water impact index

For a better understanding, impact of the baseline scenario that is referred to the wastewater treatment plant is detailed below. After that, for five main impact categories (CED, GWP, FEP, MEP, WIIX) is discussed separately the impact contribution of different relevant aspects.

Impact of the wastewater treatment plant

Results concerning the production of a 1 m3 of water able to be reused at the point of generation are detailed in Figure 5-7. Relative impact for CC, FEU, MEU and CED are shown separately for the 5 main stages as in the previous figure.

MBR has a strong effect in all categories. It is due to two opposite effects. First, the high energy consumption which accounts of circa 88% of total direct consumption of the plant, as it is stated in Table 5-3. That fact reveals a direct impact for CC and CED as it is reflected in the diagram. Secondly, it is a unit where the pollution abatement in terms of P rand N removal for respectively FEU and MEU is determinant by following the NEB approach. In this regard, only pre-treatment has a higher effect in FEU.

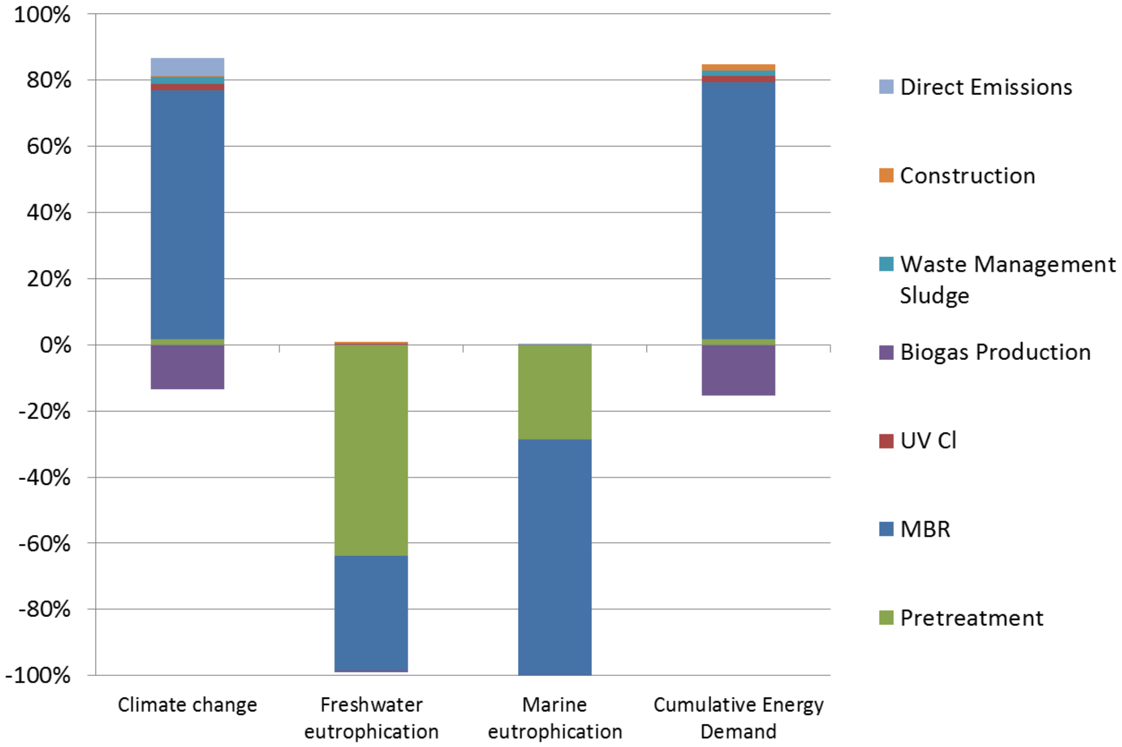


Figure 5-7: Environmental profile for the baseline scenario: the production of water to be reused at the Sabadell WWTP

Other stages (Pretreatment, UV Cl, and Biogas production) have a limited effect in CC and CED. Impact of these units is mainly due to energy consumption, which is much lower than in MBR. Only for FEU and MEU the pre-treatment has a decisive effect. In this regard, P and N removal from wastewater in primary treatment accounts for a net avoided impact for circa 62% and 28% in freshwater and marine eutrophication, respectively.

Biogas production considers the treatment of sludge via anaerobic digestion and the resulting production of energy. For that reason, avoided impact related is reflected in the CED and CC categories. It is worth to mention that all energy produced is partially sent to the network and rest of it is used by the plant. Hence, effect results in a 10% and 12% of avoided impact in CC and CED respectively.

Assessment of the implementation of water reuse

The main objective of the study is to demonstrate the benefits of water reuse in front a non reuse scenario and evaluate the environmental implications of the reuse of water. In other words, the main objective is to evaluate the additional efforts for distribution and treatment and impact related.

Results concerning the production of a 1 m^3 of water at the point of generation are compared in front of different reuse scenarios: irrigation of water in gardens, street cleaning and the use of reclaimed water for WC flushing in a shopping centre. Impact has been detailed in each scenario for the treatment in Sabadell WWTP detailed above. They are referred to the energy for the distribution of water, the reagents needed to be added (NaOCl) prior to the use and the infrastructure needed (pumps, piping, vehicles in the street cleaning, etc.). Furthermore, it is accounted the avoided impact from the production of Tap Water, which is currently used instead reused water. The balance between net impact and the avoided impact is tagged as total and refers the total impact of each scenario. Results for CED and CC, MEU and FET are discussed below.

Comparative Assessment: Cumulative Energy Demand and Climate Change

In Figure 5-8 and Figure 5-9 are shown the relative impact contribution of the stages and aspects detailed in the 4 scenarios defined. As stated, environmental profile for both impact categories are so similar in all scenarios analysed. Energy consumption has a crucial effect in climate change, and for that reason, results can be compared at the same time. In all scenarios, if only is considered the water impact.

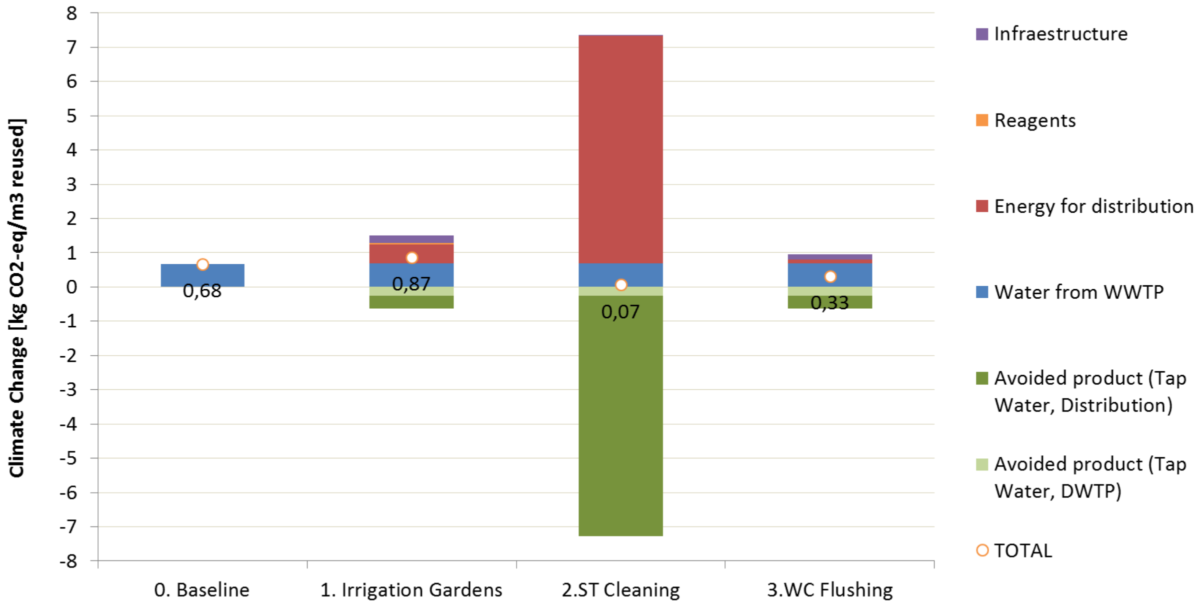


Figure 5-8: Impact for climate change of the different scenarios compared to '0 Baseline' scenario per m³ water reused

When comparing all scenarios for climate change, only in the case where water is reused for the irrigation of gardens, it implies a net increasing of total impact. This additional impact is due to the necessity of water pumping through the piping system and needed infrastructure. In the other two cases, where this requirement is avoided, results denote a decreasing in impact. Despite of more resources are needed, reuse water and its benefit crediting allows having a net minimization of impact.

Results for CED show a decrease in all scenarios regarding the baseline with the exception of garden irrigation. In this scenario, by aspects, WWTP is the main source of impact, followed by the energy necessary to send water to the fields and needed by the sprinkler system. These account for 1.2 kWh/m^3 that in front of the total energy consumption of plant (1.1 kWh/m^3) is quite higher. However effect for non-direct emissions of secondary processes and embodied energy in aspects like the infrastructures, reagents, etc. makes that WWTP has at the end a higher impact. Other aspects like the infrastructure of the piping system coming from the WWTP and the sprinkler system itself has a minor contribution. Finally, the NaClO dosage has an impact contribution almost negligible (<1%) in both CED and GWP.

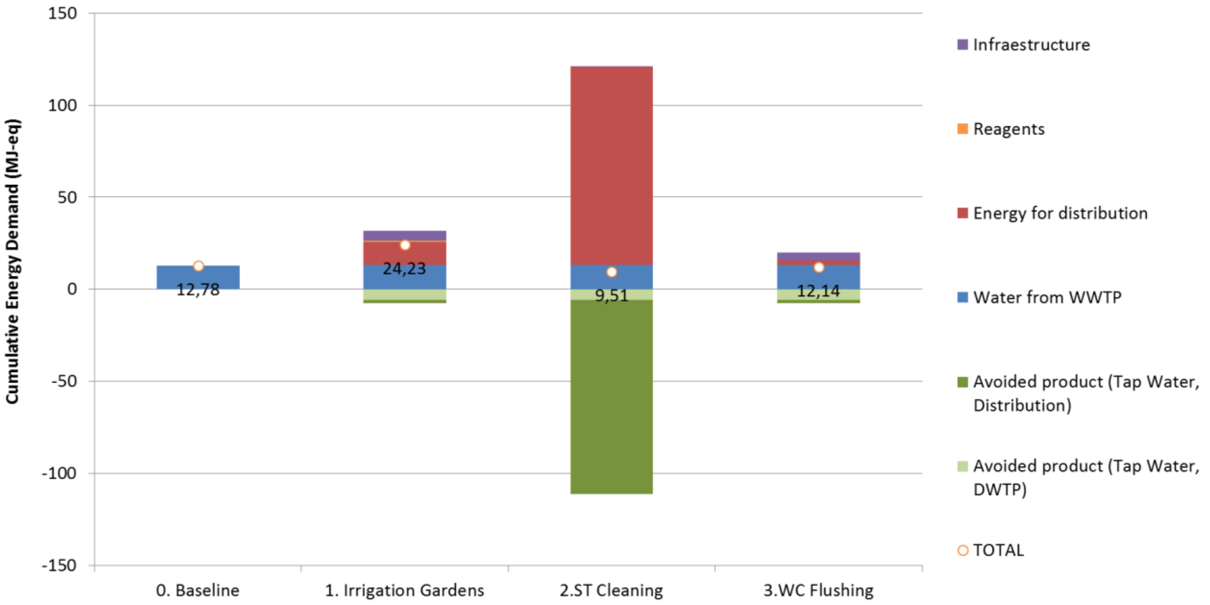


Figure 5-9: Impact for cumulative energy demand of the different scenarios compared to '0 Baseline' scenario per m³ water reused

In the street cleaning scenario, total impact for GWP and CED are 0,07 MJ-eq / m^3 and 9,51 kg CO_2-eq respectively, which means a slight increase in the impact from the baseline in GWP and a net decreasing for CED. In that scenario, the energy, due to fuel used in vehicles represents the main source in impact.

However, this large impact is counterbalanced due to the use of tap water for this scenario has almost equivalent impact. It worth to mention that for tap water, it considers impact due to distribution network, while for reuse water is already included in the impact of the WWTP.

But due to distance to be covered has been stated equal for both sources of water, result is counterbalanced. Hence, differences are stated for the infrastructure requirements. In this regard, no extra capital goods are needed for the reused water due to it is already included at the WWTP meanwhile for tap water it is accounted the needed water network.

Finally, results for WC flushing are 12.14 MJ-eq / m^3 and 0.33 kg CO_2-eq respectively. It implies an increasing impact in CED and a decreasing for GWP. These impacts are mainly determined by the treatment of water. In this regard, it is worth to mention that in this specific case, no extra energy is needed to pump water from WWTP to the shopping centre. Water is generated with enough pressure to avoid the use of a pumping system. Only energy is needed to supply water inside the system at the shopping centre. Hence, net impact is lower than for the baseline case. Like in the irrigation gardens scenario, impacts from other issues have lower influence.

Comparative Assessment: Freshwater and marine eutrophication

Freshwater and marine eutrophication (Figure 5-10 and Figure 5-11) are two impact categories where the abatement potential due to the treatment of wastewater is more determinant to the final results. As it is stated, in all impact categories, influence of other stages like the energy of distribution, reagents used or infrastructure, are almost negligible, especially for the marine eutrophication potential. In this regard, it is also important to note that the water entering at the WWTP has a pollution carrier for total N and total P higher than water entering at the DWTP, which comes from natural resources. That effect makes avoided impact greater in the reuse scenarios. And due to the process associated with this avoided impact is equal in all scenarios; results between them are almost equal. Differences are lower than 1%.

Anyway, it is worth to mention that for freshwater eutrophication, impact caused in street cleaning is some significant. It is due to the manufacturing of vehicles and materials needed. However, that impact is at the same time counterbalanced by the avoided impact effect caused by tap water. As it is reflected above, vehicles used are the same in order to assure coherence between both situations. Hence, total impact is at the end similar in all scenarios.

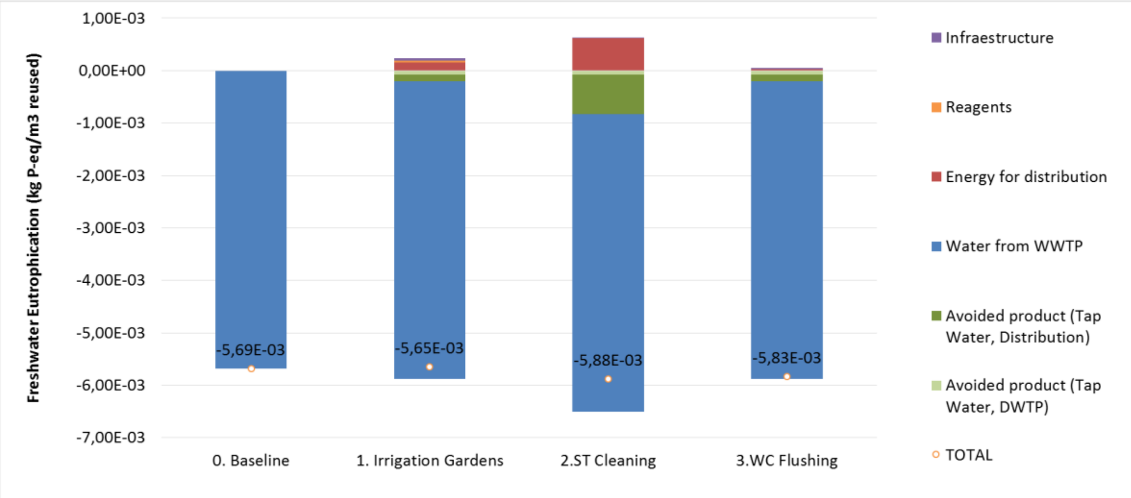


Figure 5-10: Impact for freshwater eutrophication compared to '0 Baseline' scenario per m³ water reused

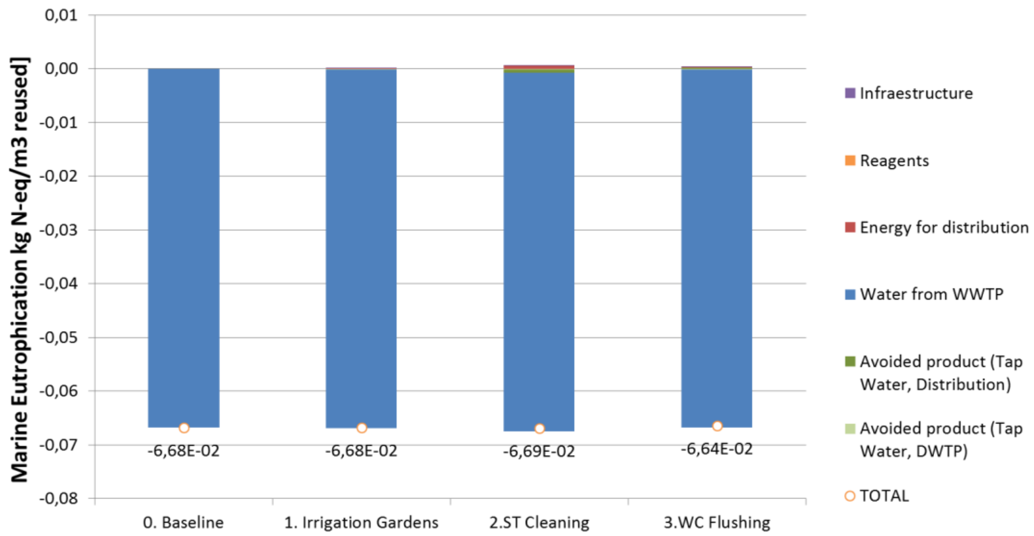


Figure 5-11: Impact for marine eutrophication compared to '0 Baseline' scenario per m³ water reused

Comparative Assessment: Water Footprint

For water footprinting, the results show the relative contribution of aspects accounted in the different scenarios for the local and global water scarcity (= Water Impact Index or WIIX, Figure 5-12). All alternatives for water reuse reduce that index, taking into account local water stress and water quality. In all cases, the index is negative, meaning that water availability increases after the WWTP, independently if the water is discharged directly to Riu Sec river (baseline) or if it is reused. A comparison between scenarios show that efforts to effectively reuse water versus the baseline has greater benefits than if water is released to the Riu Sec river.

The indirect effects, e.g. the water footprint via energy consumption, are relatively small for all scenarios. As expected, this indicates that direct water handling is far more important for water footprinting than indirect water use.

The WIIX in reuse scenarios takes credits from the avoided use of drinking water, which have different quality, and hence different contribution to WIIX than the same amount of water reused (i.e. 1000 m^3 of drinking water equals to 3530 m^3-Eq, which is 25 times higher than the same amount of regenerated water (144 m^3·-eq) with the hypothesis described). As mentioned previously, in the Irrigation garden scenario, part of the reused water returns directly to the environment. This fact means that the volume of avoided drinking water also differs in comparison to the other reuse scenarios. In conclusion, the WIIX is lower in absolute terms in the case of irrigation gardens (-2.49 m^3-eq / m^3) than in comparison with Street cleaning or toilet flushing (-3.41 m^3-Eq / m^3).

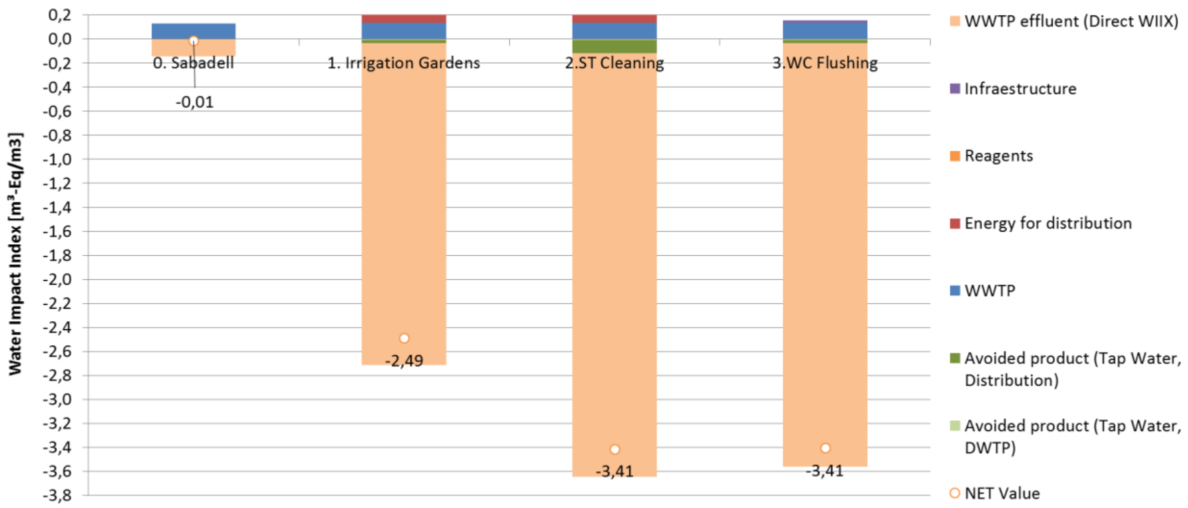


Figure 5-12: Impact for water impact index compared to '0 Baseline' scenario per m³ water reused

5.2.4 Interpretation and Discussion

Summary and Interpretation of results

All results are expressed for the basis that water produced at the WWTP is ready to be reused but released to the riu sec river. It is stated at the baseline scenario. On the contrary, the three additional scenarios reflect the impact due to the reuse of water for the purposes detailed. Table 5-6 gives a summary on the net environmental efforts and benefits of the scenarios for all impact categories, compared to the baseline.

Table 5-6: Summary of net environmental efforts and benefits of the scenarios for all impact categories. Relative changes from"0 Baseline" as reference

Scenario	0. Baseline Scenario	1. Irrigation Gardens	2.Street cleaning	3.WC Flushing
CED	12,78 MJ/ m^3	+90%	-26%	-5%
GWP	0,68 kg CO_2-Eq/ m^3	+28%	-90%	-52%
FEP	-5,69E-03 kg P-Eq/ m^3	± 0 %	-3%	-3%
MEP	-6,68E-02 N-Eq/ m^3	± 0 %	± 0 %	+1%
TAP	8,35E-03 kg SO_2-Eq/	+14%	-39%	-25%
ETP	3,92 CTUe/ m^3	-751%	-920%	-837%
HTP	4,60E-06 CTUh/ m^3	-10%	-15%	-14%
WIIX	3,26E-03 m^3-Eq/ m3	-18666%	-25610%	-25553%

Results show that from the baseline scenario, in almost all categories and reuse scenarios analyzed, the additional resources needed for water reclamation do not increase the environmental impact of the system. Differences for each category are analyzed as follows.

Due to the necessary additional energy and infrastructure, for irrigation gardens the reuse of water increase the CED and GWP of the system. This means that the avoided use of tap water do not counterbalance this additional impact.

Otherwise, in MEP and FEP, the reuse options do not represents a significant change in any scenario. It is need to remember that reduction of nitrogen and phosphorous discharge is mainly affected at the WWTP, equal for all scenarios. At the same time, changes derived from indirect emissions in background processes are limited. It involves a restricted change, and in some cases, under the uncertainty of the study.

In TAP occurs a similar situation than in GWP. For irrigation gardens, it supposes an increasing in the impact but for street cleaning and WC flushing implies a net reduction. Reason is the same: impacts are governed but no only referred to energy consumption. Hence, street cleaning and WC flushing represent a net minimization of impact. Water supply network in tap water and the energy required for pump it across system is the main source of avoided impact.

For ETP and HTP in all scenarios, results show a net reduction of impact. In ETP, impact is mainly governed by the use of chemicals in the production of capital goods. This fact is especially important for the water supply network included as avoided impact.

For HTP that effect is multiplied. Corresponding damage factors are even higher, and impact avoided further high. Anyway, results may be considered with care, due to the hypothesis and limitations stated for the capital goods considered. Furthermore, fate factors for the USEtox toxicity indicators have high uncertainties and should to be considered with caution.

In relation to the water scarcity, extra efforts to reuse water in other scenarios have a high influence on WIIX. In all cases, from an avoided impact stated as the baseline, all scenarios mean an increase in absolute terms. In other words, the negative value of WIIX is increased and at the same time the availability of water resources.

Summarizing the results of this LCA, it can be stated:

- It has been demonstrated that the relative impact of the different reuse scenarios are lower in most cases in comparison with the situation where water is released to the Riu Sec river and in front of the impact of the WWTP in Sabadell, which is able to produce water with enough quality to be reused. Street cleaning and WC flushing are the best options.

- Energy consumption is the main critical factor to implement a water reuse scenario. This energy comes from the overcost to obtain water with higher quality and the requirements for pumping and additional infrastructure. For almost all impact categories, additional energy has a significant contribution to impact.
- The ST cleaning scenario gives the highest environmental gain. It is the scenario where negative effect is more counterbalanced by the avoided tap water impact. On the contrary, irrigation gardens require more energy and infrastructure than for tap water, causing a net positive impact in CED and GWP. In the categories where energy is not an important issue, even for irrigation gardens, water reuse suppose a net impact reduction for the overall system.
- Water reuse is a good measure to minimize adverse effects of climate change, but the direct effect on mitigation of GHG's depends on the specific scenarios.

Scenario analysis: What would be the real environmental impact associated to reuse water in a larger context?

In that scenario is analysed the effect of water reuse leverage produced at the WWTP. As stated above, Sabadell currently has a very low rate of water reuse, where only 0.21% of total water available is reclaimed. Rest of water is flown to the Riu Sec River for recreational purposes. Having into account previous results where currently uses for water reuse are a source of avoided impact, here is analysed the effect to transport all the water available across a water supply network.

Hence, a comparative scenario has been conducted between uses of tap water in front of the hypothetical scenario where all WWTP water produced would be reused. In Table 5-7 are reflected the main data considered in both scenarios.

Table 5-7: Data implemented for reuse all water treated at the WWTP. Comparison in front of currently tap water network operating in Sabadell

Parameter	WWTP	Tap Water
Water released (m^3/day)	27646	21171
Network needed (km)	Scenario 1: 40 Scenario 2: 60	641
Infrastructure lifespan (years)	70	70

The WWTP scenario accounts the treatment of 27646 m^3 of water at the WWTP and the transport through a network with two different lengths: 40 and 60 km. Tap water scenario reflects the current situation stated at Sabadell, where water is treated to be drinkable in two DWTP (see section 1.3.1 for details) and released through a 641 km of network.

Distance differences are due to the fewer number of uses for the reclaimed water and at the same time, the higher consumption of water in each one. For these reasons, it is rational that length needed would be lower than for the tap water scenario. As a conservative approach, construction efforts, capital goods, lose and energy for distribution by km in both networks has been considered equal. Finally, results are shown in Figure 5-13.

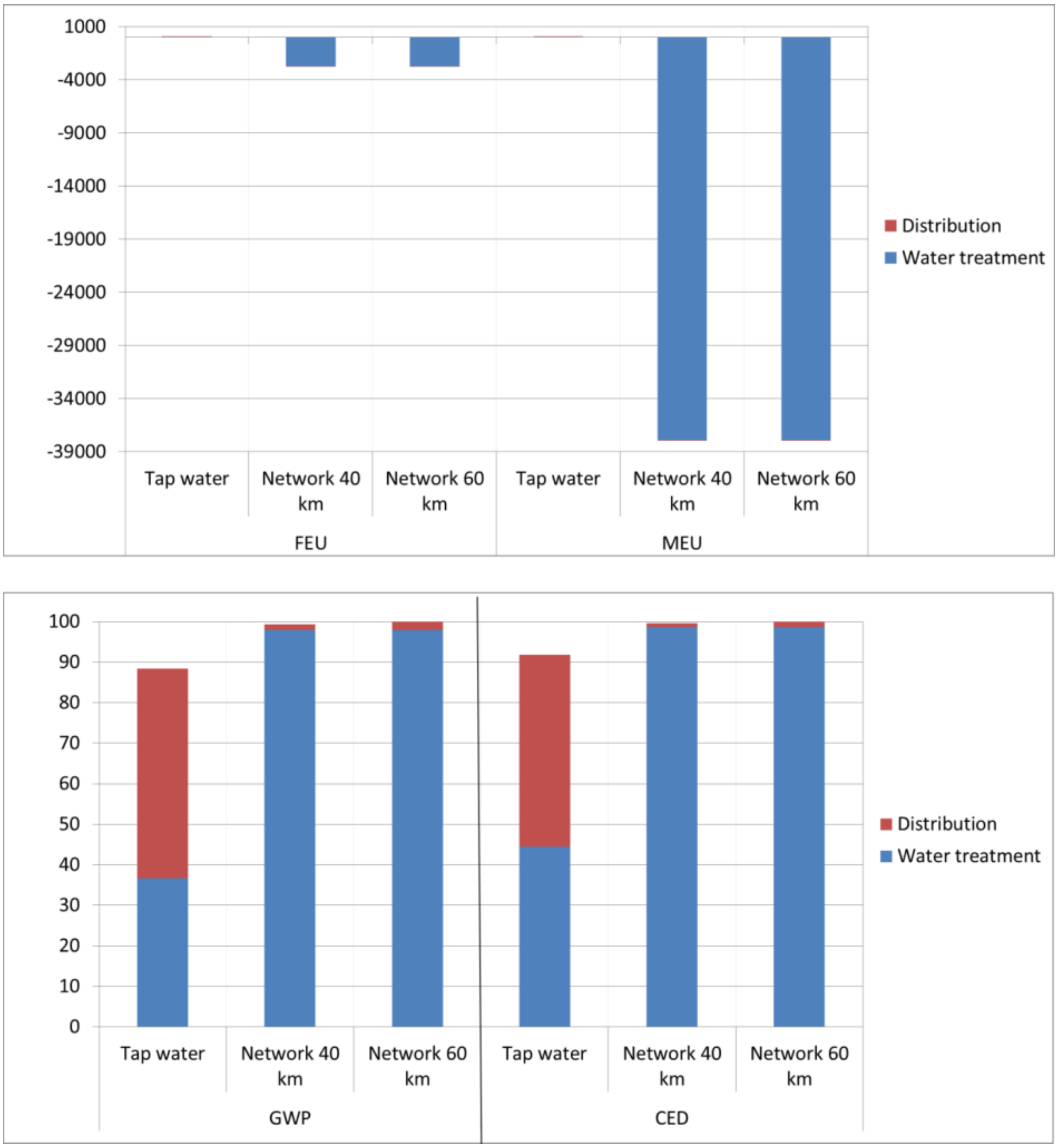


Figure 5-13: Results for the comparative scenario between for the reuse of 27641 m^3 of water from the WWTP through a 40 and 60 km network in front the use of tap water in Sabadell

Figure 5-13 shows the relative contribution of water distribution (water network infrastructure, energy for pumping and loses) and treatment of water in the WWTP or DWTP. As stated, FEU and MEU indicate a net environmental credit regardless of network length for water reuse. This situation denotes the negative contribution to impact held by abatement pollution potential of the WWTP.

Scenarios for reuse at 40 and 60 km involve a net increasing in impact of circa 11% and 12% for GWP and around 8% in both lengths for CED. That increase should not be a barrier to adopt a strategy to increase water reuse in Sabadell. After all, the benefices stated in terms of water pollution potential and for water stress indicate that overcost would be worthwhile.

As a general conclusion, it is worth to mention that water reuse is a good measure to reduce adverse effects of climate change despite of in some scenarios can suppose an increase of GHG’s.

5.3 Risk Assessment

5.3.1 Hazard Identification and selection

Hazards in reclaimed water come mainly from the raw wastewater and also from the chemicals used for the treatment. Municipal wastewater can contain a wide range of agents that can pose a risk to human health, mainly chemicals and pathogenic microorganisms [105]. Main studies focused on the microbial agents since they pose a higher risk when talking about water reclamation, however, the chemicals should also be taken into account for a comprehensive risk assessment.

Chemical Hazards

A wide range of contaminants could be present in wastewater, including inorganic and organic chemicals, pesticides, potential endocrine disruptors, pharmaceuticals and disinfection byproducts. Although WWTPs are designed for the abatement of chemical pollution, some of these contaminants could still be present in the final reclaimed water.

Analyses of treated reclaimed water indicate that chemical quality generally complies with drinking water quality requirements for most parameters, including heavy metals, organic chemicals, pesticides and disinfection by-products [105]. Even though, for each specific case the chemical risk should be assessed considering other exposure routes than oral ingestion, and considering the specific treatments in the WWTP. In this case, it is assumed that the chlorination process in the disinfection could cause high concentration of disinfection by-products (DBPs) such as trihalomethanes (THM) and brominated THMs. These compounds are formed as a result of the reaction of chlorine with organic matter and bromine. The present study focused on the chlorinated and brominated THM since they are the most known and commonly analysed compounds, however other DBP should be considered if a comprehensive study was required. For some THM, it was demonstrated that other routes of exposure than oral route could be much associated to cancer risk, specifically dermal and inhalation routes [106], therefore, these contaminants need to be investigated for water reuse. Other DBPs already identified as having potential impact on public health specifically for drinking water [107, 108].

In this project, the following chemical families were considered: metals, polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCBs), halogenated solvents including disinfection by-products, volatile hydrocarbons (BTEX) and chlorinated pesticides.

Potential endocrine disruptors and pharmaceutical compounds were not included since toxicological data is not much available for chronic exposure. On the other hand, some references show that these compounds are usually removed in MBR systems and disinfection systems (including UV and chlorination) [109].

Potential endocrine disruptors and pharmaceutical compounds and some emerging DBPs were not included since toxicological data is not much available for chronic exposure. Sabadell WWTP is composed about MBR, chlorinated and UV disinfection. Some pharamaceuticals can be removed from aqueous solution thankfull by biologic processes. Some treatment process as MBR, chlorination and UV disinfection showed high removal efficiency of antibiotics andsome pharmaceutical solvents and drugs [109].

Microbiological Hazards

Urban wastewater is a source of multiple microbial agents including virus, bacteria and protozoa, which can survive to the treatments applied in the production of reclaimed water. These agents could cause adverse effects on human health.

The illness most widely linked to polluted water consumption is gastroenteritis, however respiratory and dermal contact can also cause illness.

There are some microbial agents that are used like models or indicators for risk assessment considering gastrointestinal illness as recommended by different guides [105, 110, 111]. Indicator microorganisms should be at human and animals faces; they should not multiply or reproduce in natural waters, and should be in high concentration to be detectable [112].

For bacteria, *Campylobacter* has been used as indicator in scientific studies considering that it is the most common bacteria that causes gastroenteritis. On the other hand, *Clostridium Perfrigens* and *Echerichia Coli* determination are also widely used as indicators of bacteria in full-scale systems. However, some pathogens are more resistant to conventional wastewater treatment plant, including disinfection, than E. Coli, and thus, in some cases it could not be a good indicator of bacteria reduction in reclaimed water. In these cases, *Clostridium Perfrigens* could be the choice since it is very resistant to environmental conditions. Furthermore, *Clostridium* spores, could also be a good indicator for protozoa when membrane based processes are evaluated since protozoa are much larger than *Clostridium* spores [110, 113].

On the other hand, total coliforms can be a good indicator tool for QMRA for the characterization of process efficiency although total coliforms do not pose a health hazard and are not related to a specific fecal pollution source [113, 114]

Cryptosporidium and Giardia pathogens are chosen for protozoa as the best indicators and they are important waterborne human pathogens in developed countries. Cryptosporidium is chosen also because is resistant to chlorination [115].

For virus, rotavirus is the best choice at risk assessment because it is the main cause of diarrhea in many developed countries, they have a relativity high infectivity compared with other waterborne viruses and a dose-response model has been established [115]. In this project, adenovirus, enterovirus, noroviruses and rotavirus are analysed.

Selected microbial agents for this study and the human health effects are shown in Table 5-8.

Table 5-8 Microbial hazards considered and their effects on human health

Family	Microorganism	Effect on human health
Bacteria	*Campylobacter jejunii*	Gastroenteritis
	Clostridium perfringens	Indicator
	Total coliforms	Indicator of removal pathogen
	Escherichia coli	Gastroenteritis
Protozoa	*Cryptosporidium spp.*	Cryptosporidiosis
	Giardia spp.	Giardiasis
Virus	Adenovirus	Gastroenteritis, Respiratory infections, conjunctivitis
	Enterovirus	Different symptoms depends on enterovirus species. (Previous symptoms could be fever, headache or diarrhea)
	Norovirus-I	Gastroenteritis
	Norovirus-II	Gastroenteritis
	Rotavirus	Severe diarrhea among children

5.3.2 Concentration of contaminants in reclaimed water

Reclaimed water quality in terms of chemical and microbiological composition was assessed from September 2015 to May 2016 in four monitoring campaigns (21/09/2015, 1/12/2015, 23/2/2015, and 4/5/2015). Water quality was assessed at different points in the WWTP and in the water reclamation network as shown in Figure 5-14. Other available data is shown in the Annex (section 9.5.3)

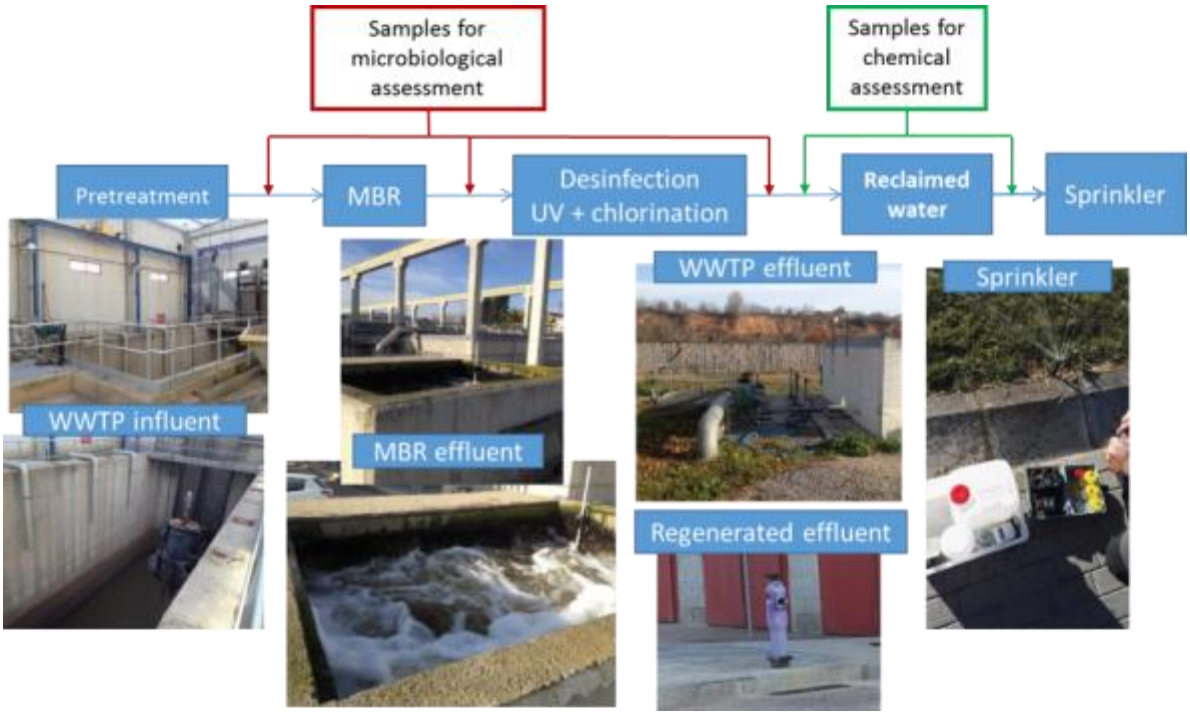


Figure 5-14: Schematic view of monitoring points

1) Chemical contaminants

Samples for chemical analysis of reclaimed water were taken at the effluent of the WWTP (after disinfection). However, for the last two monitoring campaigns, disinfection by-products were also determined at two points of the distribution network: at the reclaimed water supply point for cleaning trucks and at a sprinkler in a green area close to the WWTP.

From all the analysed compounds, only metals and some disinfection by-products were detected in more than one of the sampling campaigns, thus, only these compounds were used in the QCRA. Annex shows the detection limits of all contaminants analysed (section 9.5.3)

Metals concentration in reclaimed water is shown in Table 5-9. The average value was used for QCRA. In this case, for those metals failing under the detection limit, the detection limit was considered. This table also shows the limits for drinking water of this metals for Spain, stablished in [116]. Most of the metals show a concentration below these limits. Only Sb, in one occasion is in higher concentration that the limit.

Table 5-9: Metals concentration in reclaimed water and values used for QCRA

Metals (µg/l)	WWTP effluent 1st Campaign 21-09-15	WWTP effluent 2nd Campaign 01-12-2015	WWTP effluent 3rd Campaign 23-02-16	WWTP effluent 4th Campaign 04-05-16	Concentration for QCRA	Drinking water limits [116] (µg/l)
Cr	1,0	1,3	1,6	1,4	1,3	-
Mn	11,0	32,7	25,3	19,3	22,1	50
Fe	24,0	29,6	50,2	55,0	39,7	200
Ni	8,0	93,5	17,2	14,7	33,3	20
Cu	7,0	3,7	11,1	10,9	8,2	2000
Zn	111,0	191,3	228,3	171,0	175,4	-
As	1,0	1,7	2,1	1,8	1,7	10
Se	1,0	< 1	3,2	<1	2,1	10
Cd	< 0,1	< 0,1	< 0,1	<0,1	0,1	5
Sb	2,0	2,1	6,5	1,9	3,1	5
Hg	< 1	< 5	< 1	<1	1	1
Pb	0,2	0,3	0,3	0,4	0,3	10

Concentration of disinfection by-products detected in reclaimed water is shown in Table 5-10 together with the limits for drinking water in Spain [116]. Only trichloromethane (chloroform), bromodichloromethane, dibromochloromethane and tribromomethane (bromoform) were detected. Chloroform was the compound which showed the highest concentration. Comparing the different campaigns, highest values for all compounds were detected in the fourth campaign. Averaged values were used for QCRA. Comparing with drinking water limits, in most of the sampling campaigns, the sum of these trihalomethanes is above the limit.

Table 5-10: Disinfection by-products concentration in reclaimed water and values used for QCRA

Disinfection by-products (µg/l)	1st Campaign 21-09-15	2nd Campaign 01-12-15	3rd Campaign 26-02-2016			4th Campaign 04-05-2016			Concentration for QCRA	Drinking water limits [116] (µg/l)
	WWTP effluent	WWTP effluent	WWTP effluent	Regenerated effluent	Sprinkler point	WWTP effluent	Regenerated effluent	Sprinkler point		
Trichloro-methane	79,52	81,47	7,29	9,44	16,89	495,4	357,8	46,1	137	Trihalo-methanes sum 100
Bromodichloro-methane	18,01	63,48	2,48	3,11	3,5	165,2	126,9	13,73	49,6	
Dibromochloro-methane	3,04	28,87	<1	1,22	<1	54,93	47,71	2,78	23,1	
Tribromo-methane	<2	4,3	<2	<2	<2	5	9,2	<2	6,2	

Microbial agents concentration results

Samples for microbial characterization were taken before the MBR, after the MBR and after the disinfection (Figure 5-14). All results are available in section 9.5.3. Figure 5-15 shows the results obtained for bacteria and protozoa in the different sampling points. *Cryptosporidium spp.* and *Campylobacter Jejuni* were not included in this figure since the measured concentration was always below the detection limit.

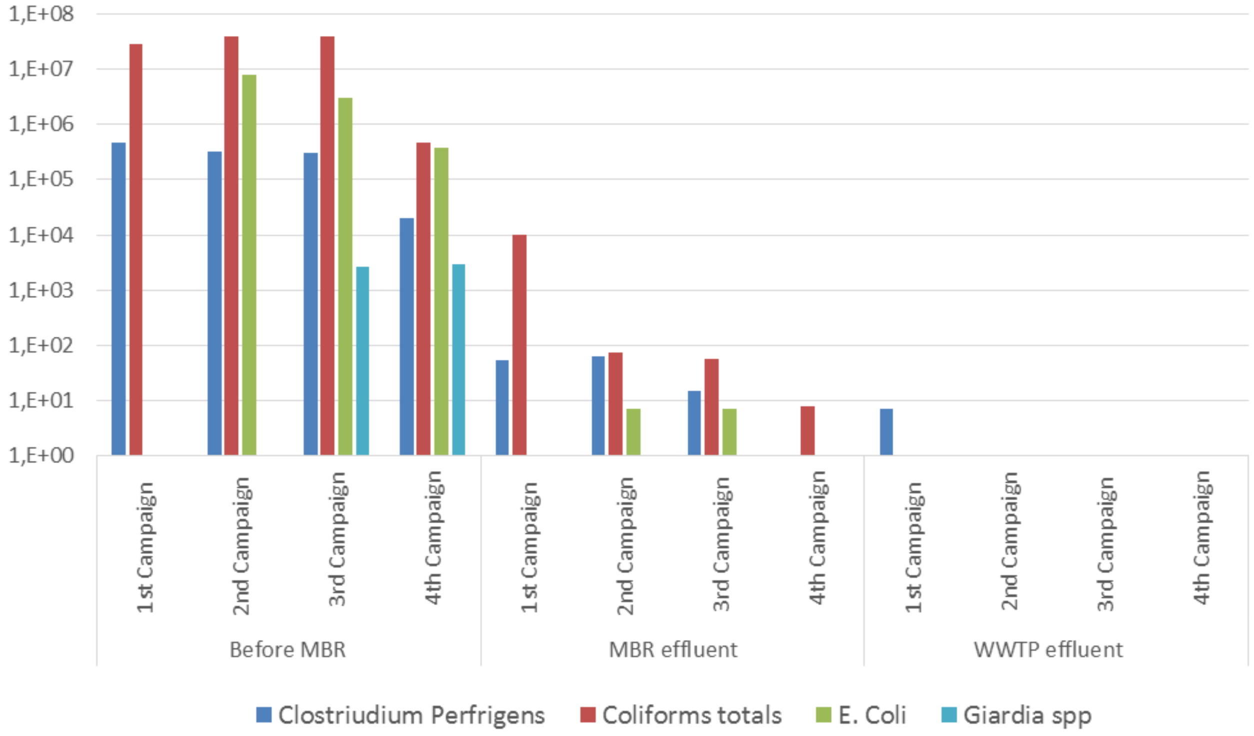


Figure 5-15: Microbial agents' (protozoa and bacteria) concentration at the WWTP sampling points.

Viruses were analysed by the Laboratory of Viruses Contaminants of Water and Food from the Universitat de Barcelona (UB) in collaboration with Demoware Project. Figure 5-15 shows the average values for each sampling point and each sampling campaign. Values indicated by "<" are the detection limits.

Table 5-11: Average values of virus concentration for all sampling campaigns

Virus	Units	Before MBR	MBR effluent	WWTP effluent
Adenovirus	gen copy/L	8,68E+07	<2,14E+02	<2,14E+02
Enterovirus	gen copy/L	< 6,63E+05	<4,14E+03	<4,14E+03
Norovirus-I	gen copy/L	1,61E+06	<4,11E+02	<4,11E+02
Norovirus-II	gen copy/L	1,57E+06	<2,96E+03	<2,96E+03
Rotavirus	gen copy/L	2,37E+06	<3,87E+03	<3,87E+03

In order to validate the efficiency of the two different treatment steps in the WWTP (MBR and disinfection), the reduction of the microorganisms (in log unit reductions, LUR) was calculated and compared with bibliographic values (Table 5-12). *Campylobacter jejuni and Cryptosporidium spp.* were not included in the table since they were not detected in any sampling campaign. Log reduction for the disinfection step could not be calculated for all microorganisms since most results were under the detection limit after disinfection, and in some cases before disinfection. Values given as "above a value (>)" indicate that a detection limit was used for its calculation. This results show that for the MBR, reduction obtained for bacteria and virus are in the range of the literature values. Only *Clostridium perfrigens* shows lower reduction than expected. However, the post-disinfection treatment finally produced an effluent with concentration below the detection limit. As said, due to the lack of quantifiable data, not many conclusions can be derived from the performance of the disinfection step.

Table 5-12: Treatment efficiency in LUR of the MBR and disinfection compared to bibliographic values

	Data from Sabadell (in LUR)		Bibliographic data (in LUR) [105, 110]		
Parameter	**MBR**	**Disinfection (Chlorination + UV)**	**Membrane filtration**	**Chlorination**	**UV radiation**
BACTERIA			3,5 - 6	2-6	2->4
Clostriudium perfrigens	3,7-4,3	≥ 0,9	>6	1-2	
Total coliforms	3,4-5,8	> 0,9			
Escherichia coli	5,6-6,1	>0,8	3,5 -6	2-6	2-4
PROTOZOA				0-1,5	>3
Giardia spp	> 2,5	-	6	0,4-1,5	3
VIRUS			2,5-6	3-6	
Adenovirus	> 5,0	-			>1
Enterovirus	> 2,1	> 0,9			>3
Norovirus-I	> 3,6	-			
Norovirus-II	> 2,7	0,9	2,5-6	1-3	
Rotavirus	>2,8	-			

5.3.3 Effects assessment

Effects assessment is based on toxicity data for each specific chemical contaminant (non-carcinogenic effects and carcinogenic effects) and on dose-response models for biological hazards. It is described in the Annex (section 9.5.4).

The effect-based monitoring tools approach that is being recently developed for measuring early toxic effects even at low dose levels in order to benchmark exposure uncertainties with regards to unknowns and cocktail effects of chemicals without having toxicological data was not considered in this study. In fact, this study was based on the broadly accepted classical risk assessment, which considers contaminant specific-toxicological data.

5.3.4 Exposure assessment

In this study, four different scenarios were been defined for the current and future uses of reclaimed water in Sabadell (see Figure 5-2): (1) irrigation of public gardens, (2) toilet flushing in commercial areas, (3) street cleaning, and (4) irrigation of golf courses. Exposure routes and receptors for each scenario are summarized in Table 5-13. Exposure assessment methodology is defined in section 9.5.2.

Table 5-13: Exposure routes and receptors for each exposure scenario

	Exposure pathway	Water				Air	
	Exposure route	Dermic absorption		Oral Ingestion		Inhalation	
	Receptor	adult	child	adult	child	adult	child
Scenario	Irrigation of public gardens	x	X		x	x	x
	Street cleaning	x	X	x	x	x	x
	Toilet flushing	x	X		x	x	x
	Irrigation of golf courses	x				x	

Irrigation of public gardens

In Sabadell, reclaimed water is currently used for programmed irrigation of public gardens and green areas all over the city. Irrigation is usually performed at night and thus, the exposure to the receptors is probably very low. In order to perform a very conservative study, it has been considered that the irrigation is taking place during the day, and that, adult and child receptors exist in the area every time irrigation takes place.

Therefore, exposure frequency has been considered equal to the irrigation events per year and exposure time has been considered equal to the averaged irrigation time during the year. Dermal absorption and inhalation routes have been considered in both adult and child receptors. Oral ingestion has only been considered in child receptor.

Street Cleaning

All water used for street cleaning in Sabadell is reclaimed water. In this scenario, workers are considered the most sensitive group since they could be exposed to reclaimed water during their whole working time and all working days per year. Exposure pathways considered were inhalation of volatile compounds, accidental ingestion of water and dermal contact.

Children are also considered with lower exposure duration and exposure time but the same frequency than workers taking into a count the most conservative case, for example, a child who is playing every day in the street. Exposure routes for children are the same as for workers.

Toilet flushing

Toilet flushing was implemented in Sabadell in 2015 in a commercial area close to the WWTP. In this scenario, workers at the commercial area are the most exposed receptors, specifically cleaning workers. Therefore, exposure parameters for adult receptor are defined based on the cleaning workers habits. Exposure routes considered are dermal absorption through hands and air inhalation. Exposure time is defined based on the working days per year, and exposure duration is assumed to be one hour per day.

On the other hand, children are also considered as receptors. In this case, a child customer is considered that can be in contact with reclaimed water by dermal contact, incidental ingestion and inhalation. The exposure frequency considered is once every two weeks for an exposure time of ten minutes.

Irrigation of golf courses

Irrigation of golf courses is very similar to the irrigation of public gardens in terms of exposure routes. However, in this case, no children are considered since irrigation takes place while the golf course is closed to the public. Workers are considered the most exposed receptors. Exposure frequency and exposure time are assumed equal to the irrigation frequency and time.

Exposure parameters are shown in Table 5-14 and Table 5-15.

Table 5-14: Exposure parameters for each scenario

	Exposure parameters				
	Exposure Duration (years)	Exposure Frequency (events/year)	Exposure Time (hours/event)	Dermal surface (feet, hands and forearms) (m^2)	Ingestion rate (L/event)
Irrigation of public gardens					
Adult	25	168	0,5	0,31	-
Child	6	168	0,5	0,182	0,001
Reference	*[117]*	*Sabadell data*	*Sabadell data*	*[118]*	*[105]*
Street cleaning					
Adult	25	220	5,5	0,31	0,001
Child	6	220	0,5	0,182	0,001
Reference	*[119]*	*Sabadell data*	*Sabadell data*	*[118]*	*[105]*
Irrigation golf courses					
Adult	25	100	2	0,31	-
Reference	*[119]*	*Golf course data*	*Golf course data*	*[118]*	-
Toilet flushing					
Adult	25	240	1	0,198 (hands)	-
Child	6	27	0,17	0,182	0,0001
Reference	*[119]*	*Assumed*	*Assumed*	*[118]*	*[105]*

Table 5-15: Exposure parameters for all scenarios

	Weight (kg)	Average lifetime (years)
Adult	80	70
Child	15	70
Reference	*[120]*	*[120]*

The exposure of the defined receptors to the detected chemical contaminants was calculated using these exposure parameters, average contaminants data (Table 5-9 and Annex 9.5.3) and volatilization factors (see Annex 9.5.2). Microbiological exposure only takes into account water ingestion, so only ingestion parameters in Table 5-14 and Table 5-15 were used together with concentrations measured (Table 5-11).

5.3.5 Quantitative chemical risk assessment

Total risk calculated for each receptor and each scenario is summarized in Table 5-16. These results are based on methodology described in section 9.5.2. Contaminant-specific results for each exposure route and each scenario are shown in the Annex (section 9.5.4).

Table 5-16: Total risk for each scenario and receptor

	NON-CANCER		CANCER	
Scenario	CHILD	ADULT	CHILD	ADULT
Public irrigation gardens	6,08E-03	3,68E-03	2,39E-07	6,43E-07
Street Cleaning	6,71E-03	2,71E-02	2,54E-07	4,99E-06
Golf irrigation		9,85E-03		1,94E-06
Toilet Flushing	2,84E-04	8,69E-03	1,33E-08	1,81E-06
Threshold value	*1*	*1*	*1,00E-05*	*1,00E-05*

These results show that, although conservative parameters were used for the exposure calculation, the calculated risk is acceptable for all scenarios and all receptors. Therefore, the actual water reuse performed and planned in Sabadell is acceptable in terms of chemical risk for the contaminants considered.

These results also show that the scenario with the highest risk values is the street cleaning, with the highest non cancer risk for children and the highest cancer risk for adults. This means that an increase of the average chemicals concentration in the reclaimed water could cause unacceptable risk, especially for this scenario.

Chemical risk results are presented at Figure 5-16 for non-cancer risk and Figure 5-17 for cancer risk. Calculated risk for each scenario and each specific exposure route is shown. Adults or Childs as a receptors are separately exposed in each figure. Contaminants were clustered in two families, metals and halogenated solvents (chlorination by-products) to show the contribution of each type of compounds.

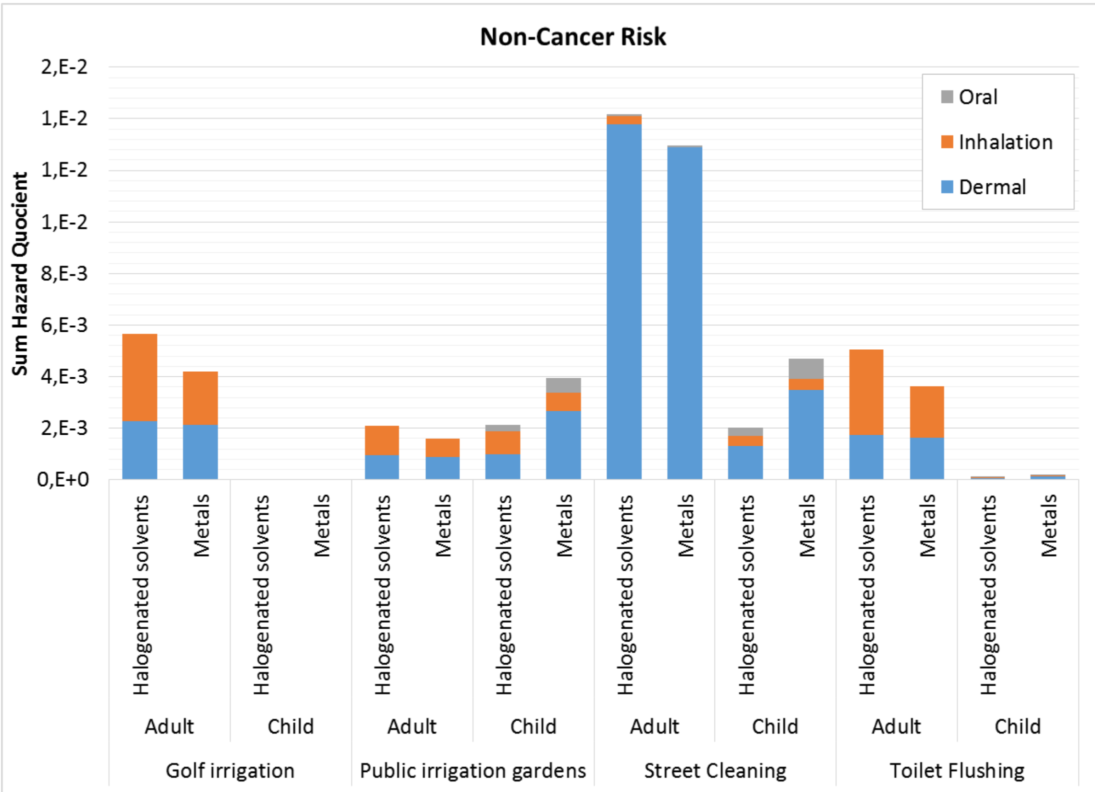


Figure 5-16: Non-cancer risk for all scenarios

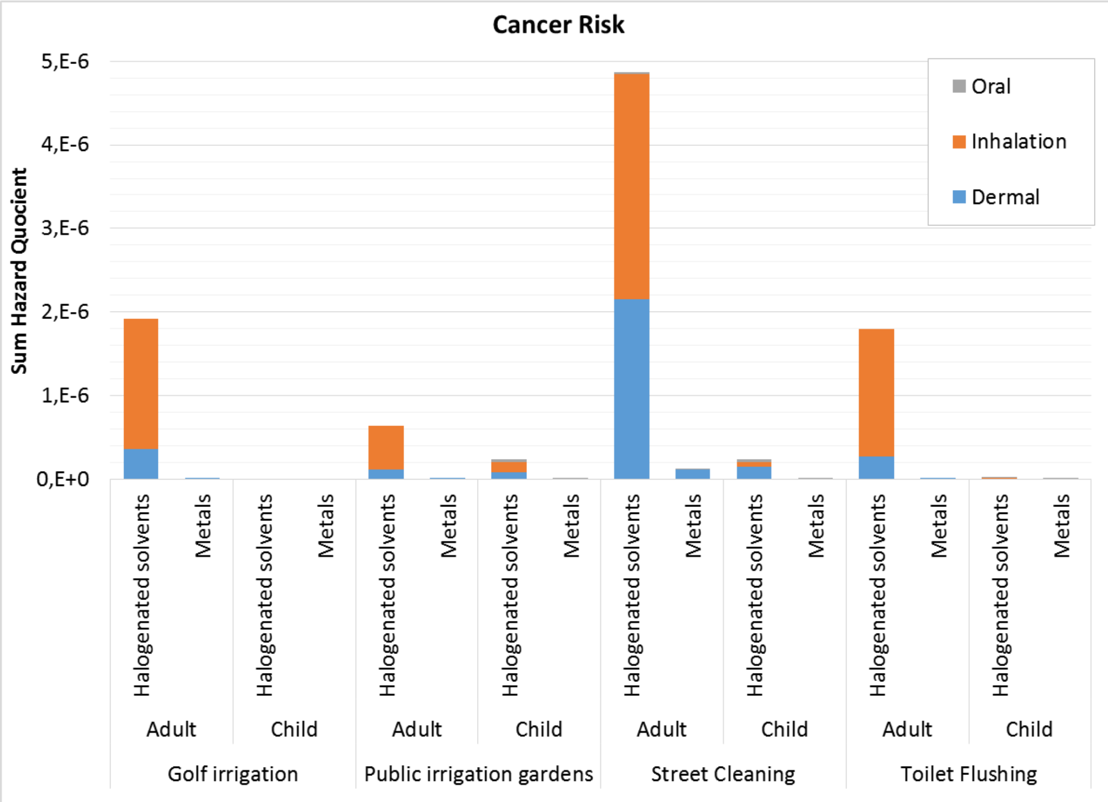


Figure 5-17: Cancer risk for all scenarios

Irrigation of public gardens

As shown, for non-cancer risk, all contaminants and exposure routs have similar contribution to the final risk, being metals through the dermal route the contaminants with highest contribution. For the cancer risk, inhalation is the exposure route that poses higher risk, and in all scenarios, halogenated compounds are the contaminants contributing mostly to the total risk.

It is important to remind that irrigation of public gardens in Sabadell is being performed during the night but that the exposure receptors were defined as if they were always present when the irrigation takes place. This is a very conservative approach. Even though, calculated risk is below the accepted threshold value and thus, it can be assured that the risk is acceptable for this scenario.

Street cleaning

Is it clear from these figures that for the non-cancer risk, the dermal route is the pathway that contributes mostly to the total non-cancer risk, especially for adult receptors, both considering halogenated solvents and metals. For the cancer risk, main contaminants contributing significantly to the total risk are halogenated solvents though the inhalation and the dermal route. Chloroform and bromodichloromethane together sum up to 99.9 % of the inhalation cancer risk and 91.5 % of the total cancer risk for adults.

Since workers were detected as the most exposed receptors in this scenario, a risk mitigation action, in case it was needed, could be to avoid dermal contact with water during the working time (protecting arms and hands) and avoiding inhalation of vapors (using nose protection masks). However, before implementing this actions, a more accurate assessment should be done in which concentration of contaminants in air should be required to avoid using vapor transfer models.

However, using the exposure parameters defined in this work and the average concentration values mesured during the projecte, the risk calculated for the steet cleaning scenario is below the treshold limits, thus it is acceptable.

Golf course irrigation

Golf course irrigation is a future use for Sabadell. As explained, irrigation in golf courses usually takes place during the night, and therefore, no receptors will exist. However, for the risk calculation it was considered that a receptor existed during the whole irrigation period. Results show that for the non-cancer risk, the calculated value is two orders of magnitude below the threshold value. For the cancer risk, the total risk is close to the threshold value, mainly due to the inhalation of chlorinated solvents, and specifically due to chloroform and bromodichloromethane. This results indicate that, in case the irrigation took place while workers or golf parcticers were in the area, the main contribution to the total risk would be the desinfection by-products chloroform and bromodichloromethane.

Toilet flushing

As explained, adult receptor was defined as the cleaning worker of the facilities, since it was considered that this worker was more exposed than the sporadic visitors of the commercial area. On the other hand, in order to consider a child receptor, it was assumed that a child was frequently visiting the commercial area and the toilet.

For non-cancer risk, all exposure routes present hazard quotients below 10E-02, being the inhalation and the dermal exposure, the main exposure routes for both halogenated solvents and metals. For the cancer risk the principal routes of exposure are the inhalation of volatiles and the dermal exposure for adults to these contaminants. Again, the contaminants that contribute greater to the total risk are chloroform and bromodichloromethane.

Results obtained are very sensitive to all the defined parameters, specifically those considered for the calculation of the concentration of contaminants in air. Again, it should be considered that more accurate results could be calculated if measurements of the contaminants in the air were considered in the exposure assessment. However, considering that all the parameters were chosen from a conservative point of view, it should be remarked that the risk associated to this water use is acceptable for all users.

5.3.6 Discussion on QCRA results

Main result of the QCRA performed in this project is that, the risk associated to the chemicals considered and detected in the reclaim water is below the established threshold limits, and thus, the risk is acceptable.

Contaminants that fell under the detection limit of the analytical technique were not considered in this study assuming that their contribution to the total risk would be minimal due to their low concentration. It must be also highlight that not all possible contaminants present in water were analysed, and that the assessment focused to metals and chlorinated compounds.

On the other hand, a number of parameters were assumed based on the case studio (i.e. irrigation time, frequency…) and on bibliographic data (i.e. water ingestion, skin permeability factor…) which makes the results an approximation of the actual risk. However, all parameters were chosen under a conservative point of view to calculate the worst case.

As shown in the previous figures, the contaminants that influenced mostly the final risk were the chlorination by-products, specifically chloroform ($CHCl_3$) and bromodichloromethane ($CHBrCl_2$). This result was due to the combination of their high concentration in reclaimed water, compared to the other chlorinated products and due to their toxicity values for the inhalation route.

For these contaminants and the street cleaning scenario, Figure 5-18 was developed. This figure indicates when the risk would be unacceptable if chloroform or bromo-dichloro-methane were present in reclaimed water (as averaged values) at higher concentrations.

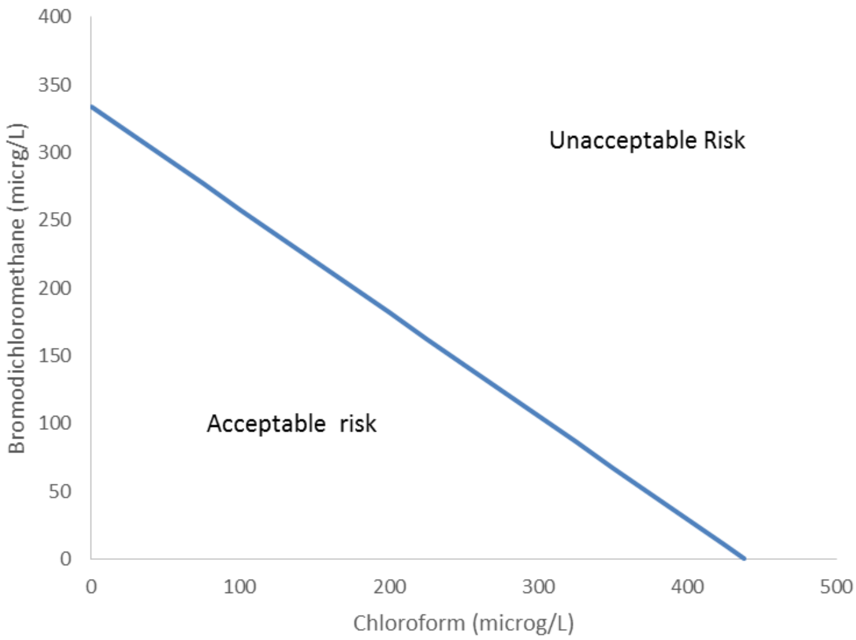


Figure 5-18: Concentration of chloroform and bromodichloromethane causing acceptable risk and unacceptable risk for the street cleaning scenario

However, due to the number of assumptions included in the risk calculation it should be stated that more accurate risk estimation could be performed if air concentration measurements were available and no modelling of the volatilization factors was required.

5.3.7 Quantitative microbiological risk assessment

Quantitative microbiological risk assessment (QMRA) has been performed following the methodology described in deliverable D3.1 and using dose-response models described in Table 9-19 (Annex 9.5.2)

As shown, almost no microbial agent could be detected in reclaimed water (after disinfection) indicating that the treatment barriers of the technology are quite efficient, specifically for bacteria and protozoa. Only some viruses were quantified above the detection limit. However, these results are very dependent on the analytical procedures to detect the target microorganisms and their specific detection limits.

Although the initial idea was to perform a QMRA based on the microbiological analyses performed on reclaimed water, due to the lack of reliable quantifiable results, two approaches were used. These approaches are described in the following sections, focusing on the viruses.

Required virus log-reduction for acceptable microbial risk

Using data from viruses' concentration in the MBR influent, the required LUR was calculated for each scenario and each virus to have a tolerable risk level of 1 additional µDALY pppy. Results are summarized in Table 5-17 which also shows the maximum virus concentration that should be present in reclaimed water.

Table 5-17: Minimum LUR and maximum virus concentration in reclaimed water for a risk level of 1 additional µDaly pppy

Scenario	Toilet flushing		Irrigation of public gardens		Street cleaning	
Pathogen	Max. Conc. in reclaimed water (gc/L)	Minimum LUR	Max. Conc. in reclaimed water (gc/L)	Minimum LUR	Max. Conc. in reclaimed water (gc/L)	Minimum LUR
Norovirus	7,10E-01	8,0	1,14E-02	9,8	8,70E-03	9,9
Adenovirus	1,80E+01	6,4	2,89E-01	8,2	2,21E-01	8,3
Enterovirus	5,35E+02	4,5	8,60E+00	6,3	6,59E+00	6,5
Rotavirus	8,59E-01	7,9	1,38E-02	9,7	1,05E-02	9,8

Since the street cleaning scenario is the strictest scenario (highest exposure frequency), the minimum LUR required is the most restrictive value, and the calculated maximum concentration in reclaimed water is the minimum. Therefore, in order to assure an acceptable microbial risk, a reduction of 9.9, 8.3, 6.5 and 9.8 LUR for norovirus, adenovirus, enterovirus and rotavirus, respectively, should be achieved in the treatment train. These values are in accordance with the theoretical values found in the literature for virus, which range from 5.5 to 12 for the MBR+chlorination (see Table 5-12). Thus, the treatment train (MBR+disinfecton, including UV) should be an effective barrier for the viruses in Sabadell.

Calculated minimum LUR could not be compared to actual LUR in Sabadell (Table 5-12) due to the too high detection limit for the virus analyses. Therefore, for an accurate estimation of risk, an analytical technique with lower detection limits should be required, specifically 8.7E-03, 2.21E-01, 6.59, 1.05E-02 gc/L for norovirus, adenovirus, enterovirus and rotavirus, respectively. Alternatively, a good indicator should be used, which was detectable and had similar behaviour to virus in front of MBR and disinfection systems.

QMRA for virus detected in the reclaimed water

For norovirus and adenovirus, a quantifiable result was obtained when analysing reclaimed water during the third campaign. This data was used in this section to quantify the microbial risk in the scenarios were oral ingestion was considered (irrigation of public gardens, street cleaning and toilet flushing).

Table 5-18 shows the QMRA results for these viruses in all scenarios. Final column shows the calculated risk. This value should be below one to consider that the microbial risk is acceptable, since Risk=1 means that the probability of infection is 1µDaly (accepted WHO value). These results show that in all scenarios and for both viruses, the risk is unacceptable. This is due to the high number of events per year and the high disease/infection ratio of these viruses.

However, even if one exposure event per year was considered, the calculated risk would be also unacceptable (data not shown). These results indicate that a unique ingestion of reclaimed water could cause a probability of infection above the benchmark value of 1 additional µDaly pppy for the receptor.

Table 5-18: QMRA results for norovirus and adenovirus for all scenarios

Public irrigation gardens							
Pathogen	Concentration (gen copy/L)	Exposure (events/year)	Water ingested (L/event)	Dose (gen copy/event)	Pi daily	Pi annual	Risk
NOROVIRUS (Group II)	85,10	168	1,00E-03	0,09	0,04	1,00	718,66
ADENOVIRUS	432,00	168	1,00E-03	0,43	0,16	1,00	50,00
Street Cleaning							
Pathogen	Concentration (gen copy/L)	Exposure (events/year)	Water ingested (L/event)	Dose (gen copy/event)	Pi daily	Pi annual	Risk
NOROVIRUS (Group II)	85,10	220	1,00E-03	0,09	0,04	1,00	719,81
ADENOVIRUS	432,00	220	1,00E-03	0,43	0,16	1,00	50,00
Flushing toilet							
Pathogen	Concentration (gen copy/L)	Exposure (events/year)	Water ingested (L/event)	Dose(gen copy/event)	Pi daily	Pi annual	Risk
NOROVIRUS (Group II)	85,10	27	1,00E-04	0,01	0,01	0,14	103,61
ADENOVIRUS	432,00	27	1,00E-04	0,04	0,02	0,39	19,27

Pi: Probability of infection (daily or annual)

It must be highlight that, in this study, it was assumed that all genome copies measurements represent the infectious concentration of the viruses. Some other authors assumed that the infectious concentration is a fraction of the total gene copies detected for norovirus [121, 122], and others consider that all genome copies are infectious viruses [123]. This parameter, difficult to estimate, adds significant uncertainty in estimates of infectious viral doses. Within Demoware, an infection test for adenovirus detected in the 2nd campling was performed by the Laboratory of Virus Contaminants of Water and Food from the Universitat de Barcelona (UB). Results showed no infection capability for the cellular lines A549 i 293A after 10 days. This indicates that if there were infectious adenovirus in the sample, they belonged to a group that does not infect the cell lines used in this test. Therefore, as a conservative option, infectivity was considered 100% for all genome copies detected.

Therefore, for this specific study, the dose of virus considered, which is calculated using the measured viruses' concentration and the assumed water ingestion volume, is too high and could cause an unacceptable risk of infection.

An inverse risk calculation was performed to determine the values for virus concentration in water and water ingestion volume that could lead to acceptable infection risk. Figure 5-19 shows the results for the scenario of toilet flushing for norovirus and adenovirus considering 27 exposure events per year. The same calculation was also performed for rotavirus and adenovirus (Figure 5-20). In this case, it can be seen that the detection limit falls into the unacceptable risk.

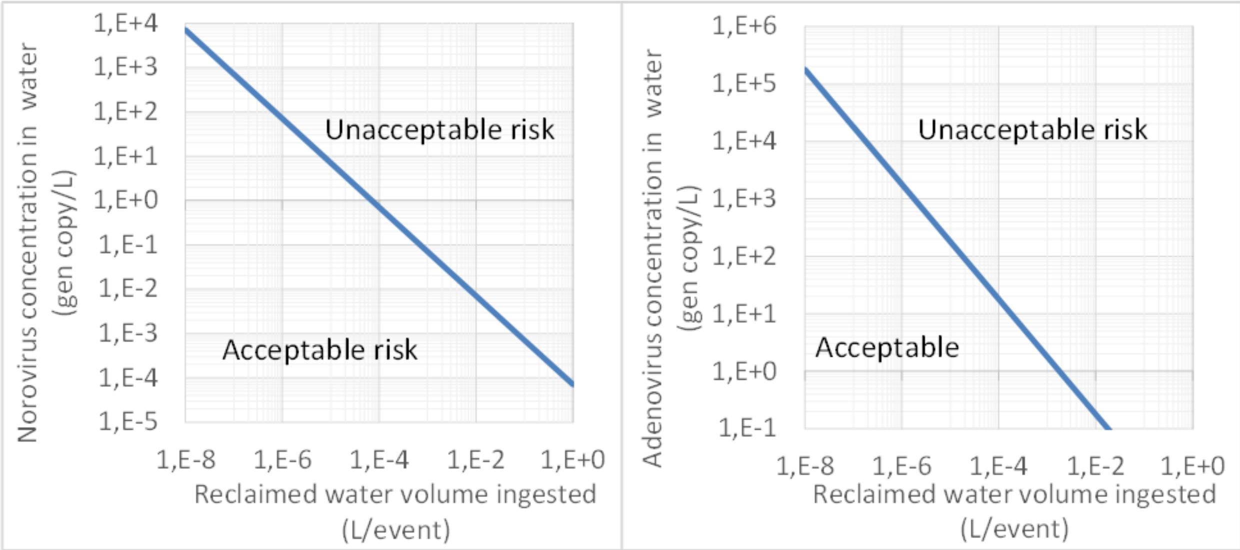


Figure 5-19: Risk of infection for norovirus (left) and adenovirus (right) in toilet flushing scenario based on reclaimed water volume ingested and virus concentration.

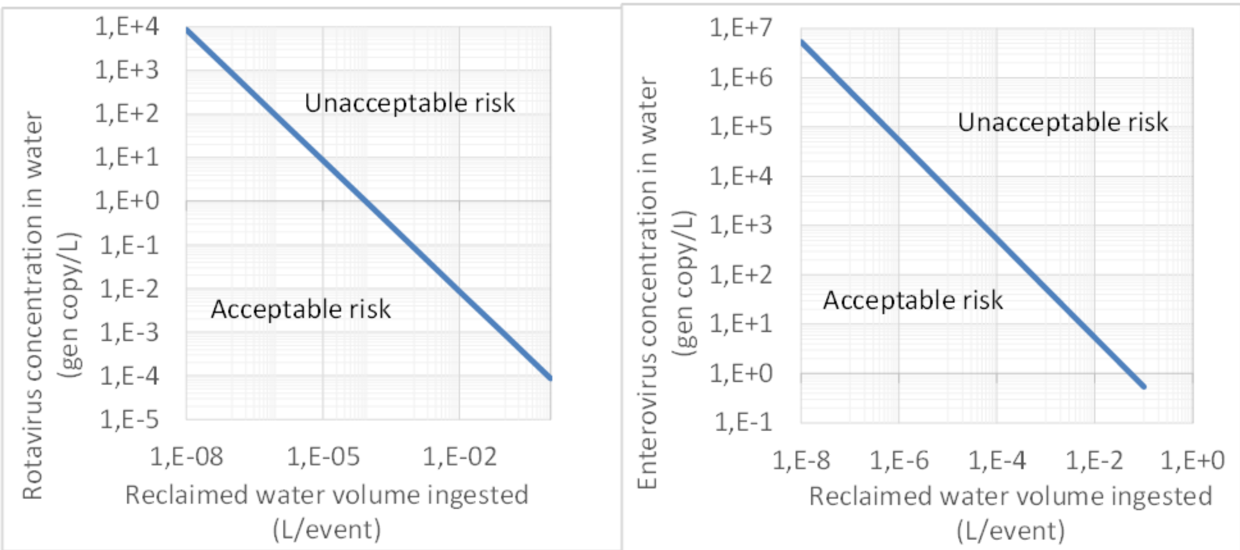


Figure 5-20: Risk of infection for rotavirus (left) and enterovirus (right) in toilet flushing scenario based on reclaimed water volume ingested and virus concentration.

Therefore, for the assessment of the microbial risk in reclaimed water it is crucial to have an accurate estimate of the infectious virus concentration in water and an accurate estimate of the exposure parameters such as the water ingestion volume and the ingestion events per year. Regarding infectious virus concentration in water, the low existing concentration, close to the detection limits of the state-of-the art techniques makes it difficult to be assessed directly. On the other hand, regarding the exposure parameters, they represent the average behaviour of the potential receptors and are based on bibliographic data. They are really difficult to assess with accuracy for each specific scenario.

5.4 Conclusions

LCA and WF results show that water reuse is a key challenge to be addressed in future at Sabadell, being in almost all cases positive in terms of environmental impact and clearly improve their water scarcity.

In terms of LCA, the benefits due to water reuse overtake the drawbacks of reuse, e.g. due to higher energy. In this regard, infrastructure is a determinant source of impact depending on scenario analysed. Environmental gain depends on factors like orography, availability of a network, amortization etc. has to be properly analysed on future actions and will require a detailed study before new water reuse scenarios will be implemented in order to increase the reuse rate and make more profit of an installation that can release water with an optimal quality.

When results for water footprint are analysed it is stated that reuse can be a decisive tool to reduce abatement of water as a natural resource. Even considering that in the baseline scenario, where water is released to the river means a positive impact to water availability, the effect of water reuse severely increase their water availability, mainly due to the boost effect regarding the avoided use of drinking water. By having into account both LCA and water footprint results, it can be concluded that water reuse is an excellent option for the adaptation to climate change, considering at the same time that mitigation can be addressed if solutions are planned properly.

For risk assessment, the following conclusions can be drawn:

- Metals and chlorinated compounds were detected in reclaimed water and were considered in the QCRA. Chlorinated compounds were detected above the limit for drinking water.
- Not all possible disinfection by-products were considered, only chlorinated and brominated compounds where taken into account.
- The human health risk due to chemical contaminants measured and detected in reclaimed water was acceptable for all scenarios considered: irrigation of public gardens, street cleaning, toilet flushing and golf course irrigation.
- Calculated QCRA was based on very conservative exposure parameters.
- Contaminants that influenced mostly in the QCRA were the chlorination by-products, specifically chloroform ($CHCl_3$) and bromodichloromethane ($CHBrCl_2$).
- None of the microbial agents analysed were detected in reclaimed water, only norovirus and adenovirus were detected in one campaign. However, the monitoring did not cover the whole year and thus, the seasonal variation of some viruses could not be monitored.
- The calculated LUR for the Sabadell MBR system was in accordance with literature data.
- No LUR could be calculated for the disinfection system due to the non-detected results.
- For bacteria and protozoa it was assumed that the treatment train was enough to produce a reclaimed water free of these microbial agents.
- QMRA performed using data from the sampling campaigns showed unacceptable risk, considering very conservative assumptions for the exposure. This was due to the poor-quality data in terms of amount of samples.
- There is a need to refine QMRA for viruses, mainly exposure parameters and infecting virus dose.
- A minimum LUR of 9.9, 8.3, 6.5 and 9.8 for norovirus, adenovirus, enterovirus and rotavirus, respectively was calculated for the more strict scenario (street cleaning), considering virus data from the influent of MBR system.
- Comparing required LUR and literature data, the treatment train (MBR+disinfecton, including UV) should be an effective barrier for the viruses in Sabadell.
- Actual virus detection limits for analysis of reclaimed water are too high to perform accurate QMRA.

6 Case Study of Shafdan

6.1 Introduction and Setting

At the Shafdan WWTP near Tel Aviv, a major water reclamation and reuse scheme is operated since the 1970s. Treating the municipal wastewater of more than 2 Mio people in the Tel Aviv-Jaffa area, the WWTP effluent of the activated sludge process is infiltrated into the underground for soil aquifer treatment (SAT) (Figure 6-1). Large infiltration ponds recharge the effluent (ca. 130 Mio m³/year) to the unsaturated zone of the aquifer (Figure 6-2). The use of flooding/drying cycles and frequent mechanical plowing of the ponds should enable a sustainable infiltration rate, prevent clogging of the ponds and introduce oxygen in the upper soil. After travelling for 3-12 month through the soil, the water is recovered in deep groundwater wells which are at 250-1600 m distance to the infiltration area. SAT improves the water quality of the WWTP effluent, removing microbial contamination, particles, organic and inorganic pollutants. The reclaimed water is then transported to the Negev desert where it is used to irrigate agricultural fields (unrestricted irrigation). The reuse system is simple, reliable and has low operational costs, providing an additional underground reservoir to buffer seasonal variations in water availability and demand.

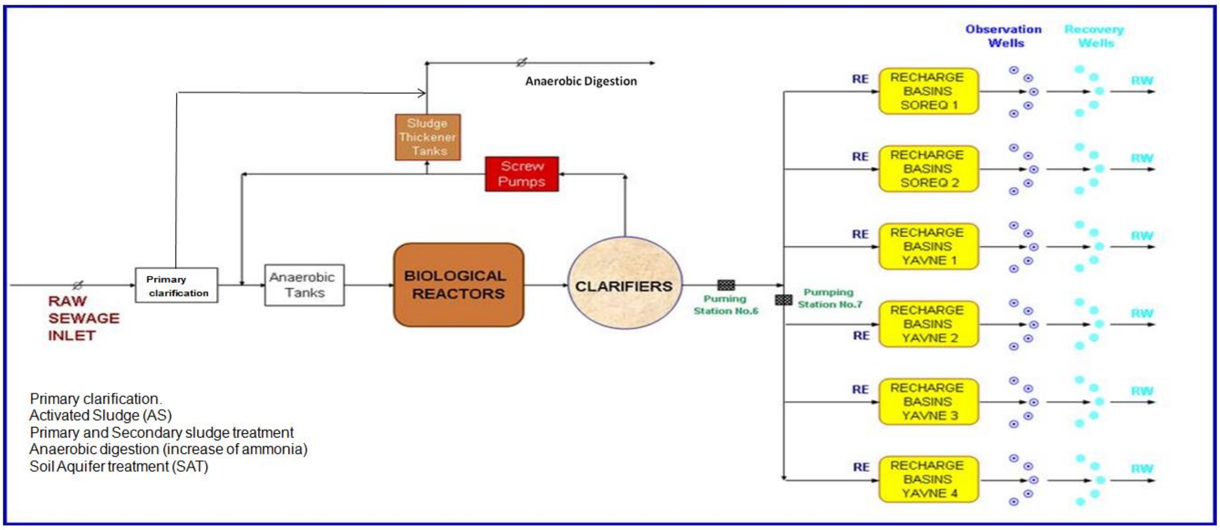


Figure 6-1: Overview of Shafdan scheme for water reclamation

Figure 6-2: SAT infiltration ponds in the Shafdan reuse site (© Mekorot)

Recently, the operation of SAT infiltration ponds has become more difficult due to higher hydraulic loading of the ponds: population growth leads to higher volumes of wastewater, but lack of available land prevents the construction of more infiltration ponds near the WWTP. In addition, the nitrogen load of the WWTP has increased considerably due to the N return load from newly-built anaerobic digestors, which leads to high levels of ammonium in the WWTP effluent. Infiltrating this water with high ammonium levels leads to oxygen consumption in the SAT system, causing reductive conditions and the dissolution of Mn oxides. The dissolved Mn precipitates again when conditions get oxic again (e.g. in pipelines or drip irrigation systems) and causes clogging problems also in the pond operation.

Hence, new schemes for water reuse have been developed and tested in recent years:

- **Pretreatment of WWTP secondary effluent by media filter for nitrification, then ozonation followed by SAT with short retention time (AOP & SSAT):** To prevent clogging of SAT ponds and irrigation systems with Mn precipitates, SAT should be operated in oxic conditions at all times. Ozonation is a good option to introduce oxygen in water and simultaneously provide a barrier against emerging organic micropollutants in the secondary effluent. In addition, a biofilter has been installed upstream of ozonation to provide complete nitrification prior to ozonation and infiltration to prevent excessive oxygen consumption in the SAT due to nitrification. Following the extensive pretreatment, a shorter SAT (1-2 months retention time) is sufficient to reach good water quality and will considerably reduce land requirements for water reuse via SAT.
- **Direct membrane treatment of WWTP secondary effluent:** treating secondary effluent with ultrafiltration and reverse osmosis yields an excellent water quality, but comes at high energy demand and produces brine to be disposed. UF filtrate has already a good quality in microbial parameters, but lacks the removal of dissolved bulk organics (DOC) and organic micropollutants and is not directly suitable for water reuse according to the local standards. Membrane systems have the advantage of a very low footprint, making them suitable as an option to expand the reuse system without the need of land nearby the WWTP.

Both schemes have been piloted in the DEMOWARE project in different configurations. This study analyses all options for water reuse in their potential environmental impacts to reveal benefits and drawbacks of the different systems from an environmental point of view. Using the method of Life Cycle Assessment (LCA), reuse schemes are compared with each other and against a reference scenario without using WWTP effluent for irrigation to illustrate the environmental consequences of water reuse in Israel. A special focus of the study is the aspect of water footprinting to underline the positive effects of water reuse on overcoming the local water scarcity in Israel, taking into account both quality and quantity of freshwater resources.

6.2 Life Cycle Assessment

6.2.1 Goal and scope definition

The goal of this LCA is to analyse and compare different options for the production of potable water (which will be used to supply agricultural fields within the Negev desert in Israel) in their environmental impacts. Six scenarios of wastewater reuse (unrestricted irrigation) and alternative water supply via potable water networks are investigated. This LCA can serve as example for sites with extreme water scarcity, quantifying the environmental profile of different options to overcome this water scarcity. The target group consist primarily of the local stakeholders such as the treatment plant operators (Mekorot), but also planers and engineers in the field of wastewater treatment and water supply.

Function/ Functional Unit

The function of the system under study is a) the enhanced treatment or discharge of WWTP secondary effluent and b) simultaneously providing water supply for agriculture within the Negev desert, including all processes that are related to these two functions. Due to the different scale of the investigated systems, the functional unit of this LCA is defined by the volume of water which is provided for agricultural purposes ("per m³ water supplied") or discharged, which amounts to 130 Mio m³/a originating from reclaimed wastewater for the full-scale SAT system. Quality of the reclaimed water fulfills the local standards for unrestricted irrigation.

While evaluating the water footprint (= water impact index), water losses that come with specific tertiary treatment options (e.g. by backwash/concentrate discharge of membranes) should also be reflected, so the functional unit for WFP is defined by the original feed water volume to the water reuse train or "per m³ tertiary treatment influent". For normalization of LCA results, an alternative functional unit is useful to compare the environmental profile of the Shafdan reuse scheme in relation to the total environmental footprint of each citizen. Hence, the treated volume is related to the functional unit "per pe and year" for normalisation, where the annual water volume of the full-scale SAT system (130 Mio m³/a as capacity of the Shafdan WWTP) refers to 2.5 Mio pe [124] while the size of the demonstration units for alternative schemes refer to 168 000 pe.

System boundaries

This LCA should directly compare the existing scheme of water reuse (= tertiary treatment via a soil-aquifer-treatment (SAT) with long retention times) with recently tested processes for tertiary treatment and other alternatives to water reuse (i.e. external water supply from freshwater resources or via the potable water network of Israel). Hence, it was decided to exclude primary and secondary treatment of the Shafdan WWTP from the analysis, as it will be comparable in all scenarios. However, potential upstream effects of water reuse on the mainstream WWTP (e.g. via recycling of backwash/concentrates of membrane processes) are taken into account using a simplified WWTP model, calculating the efforts for treatment and related direct emissions based on the particular pollutant loads in these recyclates. In case of assessing alternatives to water reuse, the complete secondary effluent of the Shafdan WWTP is hypothetically discharged into surface water (river or lake) after disinfection which is legally mandatory.

The system boundaries therefore include tertiary treatment of wastewater and alternatively potable water supply from different natural resources or seawater desalination. Distribution of water to the Negev (pipe network) is not included in this LCA, but reused water is pressurized to a certain point via pumps of tertiary treatment. Nutrient content in reclaimed water is not credited in this LCA. Finally, all relevant background process for production of electricity, chemicals, fuels, infrastructure materials, and maintenance are considered (Figure 6-3).

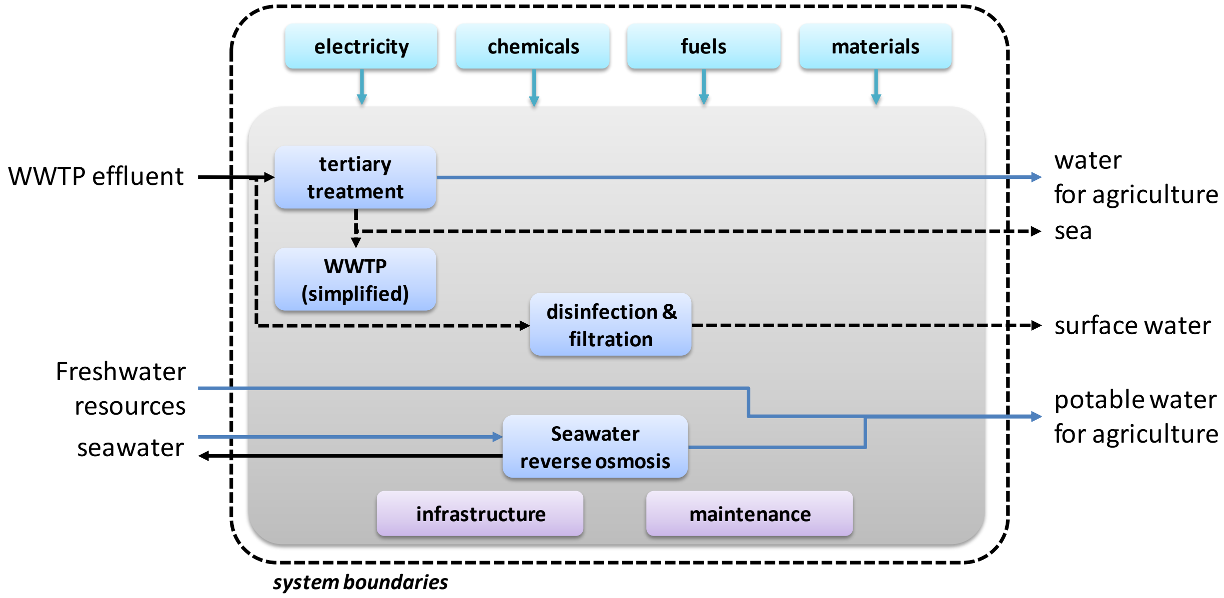


Figure 6-3: System boundaries and scope for LCA study Shafdan

Scenarios

The scenarios are selected to compare different approaches to supply the agriculture in the Negev desert with water. A comparative overview of all scenarios with process descriptions and annual water volumes supplied (Figure 6-4) is provided below:

0a. Natural water resources (NWR): represents a hypothetical reference scenario and alternative to the current water reuse scheme in Shafdan. Assuming that it is possible to supply agriculture in the Negev desert only via water from natural freshwater resources, this water is delivered by 76% from local groundwater from the Tel-Aviv region, by 12% from the Lake Galilee via the national water carrier and by 12% from marginal water (mainly rainwater). It has to be underlined that this water supply option is not a realistic approach for delivering water to the Negev; the scenario is defined to analyse its hypothetic impacts in the water footprint. In this scenario 100% of the WWTP effluent from the Shafdan WWTP is disinfected, filtrated and discharged into surface waters (river or lake) to fulfill local discharge standards for WWTP effluent.

0b. Potable water mix (PWM): represents a more realistic reference scenario and potential alternative to the current water reuse scheme in Shafdan. The supply mix of freshwater from different origins is based on the potable water mix in Israel used for agricultural purposes (if water reuse is not available). The mix consists of 47 % water from seawater desalination, 40 % water from local groundwater from the Tel-Aviv region, 6.5 % water from the Lake Galilee via the national water carrier and 6.5 % water from marginal water (mainly rainwater). As in scenario 'NWR', the WWTP effluent is disinfected, filtrated and discharged into surface water.

1. Long SAT (LSAT): The long soil-aquifer treatment represents the existing full-scale water reuse scheme in Shafdan. The effluent from Shafdan WWTP (130 Mio m^3/a) is infiltrated in the Tel-Aviv region in surface infiltration ponds. After travelling a specific distance (250-1600 m) in the soil with a retention time of 6-12 months for the infiltrated water, the water is recovered via deep wells and delivered to the agricultural fields in the Negev. In addition to the infiltrated volume of secondary effluent, some 19.5 Mio m^3/a of ambient groundwater are additionally abstracted in the well system.

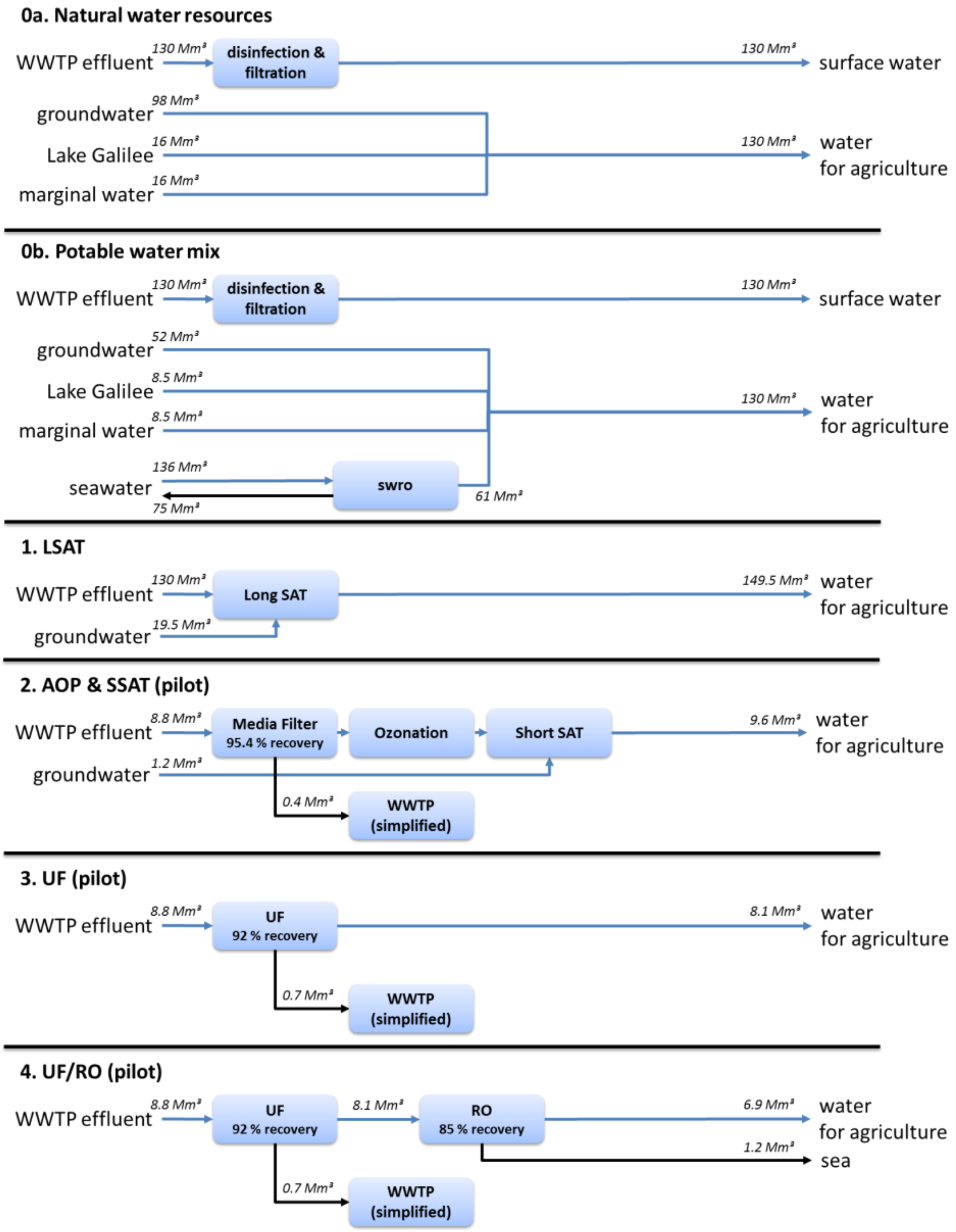


Figure 6-4: Comparative overview of the LCA scenarios for Shafdan and annual water volumes
swro: seawater reverse osmosis; SAT = soil-aquifer treatment; UF: ultrafiltration; RO: reverse osmosis

2. Advanced oxidation process and short SAT (AOP & SSAT): As available surface area is limited in the metropolitan area of Tel-Aviv, future expansion of the reuse system in Shafdan requires a system with much smaller footprint than the existing long SAT. Hence, a shorter SAT scheme has been tested with significantly reduced distance between infiltration and recovery (7-20 m) to

overcome the large footprint needed for the long SAT. For the short SAT, additional pre-treatment steps are required to compensate the lower treatment capacity in the SAT due to shorter retention times (1-3 months), to improve SAT operation by preventing anoxic conditions and to provide additional barriers for emerging pollutants. Within the DEMOWARE project, pre-treatment of secondary effluent via biological media filter and subsequent ozonation has been demonstrated in pilot-scale, followed by short SAT. Oxygen supply of the biofilter is realised by dosing of hydrogen peroxide (27 mg/L H_2O_2) upstream, while coagulant dosing (3 mg/L Al) prior to the filter further reduces COD, DOC and phosphorus in the secondary effluent. Backwash of the media filter (5% of influent volume) is recycled to the WWTP influent, and downstream impacts of the backwash treatment in the WWTP are represented by a simplified WWTP model. After filtration, 7 mg/L of ozone (0.8 mg O_3/mg DOC) are introduced into the filtered water, which is then infiltrated into the soil. This treatment train was demonstrated in DEMOWARE in pilot scale (6 m³/h) and is assumed here in a scale of 1000 m³ feed/h (8.8 Mio m³/a). Backwash losses are compensated by drawing a fraction of ambient groundwater, so that 9.6 Mio m³/a can be recovered in the SAT.

3. **Ultrafiltration (UF):** The treatment of secondary effluent via ultrafiltration membranes was also demonstrated within the DEMOWARE project in pilot-scale (25 m³/h). Outside-in hollow fibre modules (ZeeWeed 1000) have been tested for treatment, using 8% of feed flow for backwash (recovery of 92 %). The UF backwash is recycled back to the WWTP inlet, again using a simplified WWTP model to calculate impacts of UF backwash treatment. UF permeate can be directly used for agricultural irrigation, but has to be blended with other sources to fulfill the stringent quality requirements in terms of DOC (1 mg DOC/L for unrestricted irrigation) which is not removed in ultrafiltration. Membranes are regularly cleaned with chemicals (citric acid, NaOCl) to prevent inorganic and biofouling. For the LCA, a scale of 1000 m³ feed/h (8.8 Mio m³/a) is assumed for this system.
4. **Ultrafiltration + Reverse osmosis (UF/RO):** another option for tertiary treatment is the combination of ultrafiltration and reverse osmosis at the Shafdan site. After UF pretreatment (see scenario UF), the permeate is further treated in spiral-wound RO modules (Filmtec™ LE 4040) in a 3+1+1 configuration, reaching 85% recovery. RO membranes require pH adjustment before filtration (H_2SO_4), addition of anti-scalant and regular cleaning (citric acid, NaOCl) to prevent inorganic and biofouling. The brine of the RO system would be discharged to the Mediterranean Sea (an additional brine treatment may be obligatory but is not considered in this LCA), while the RO permeate can be reused in agriculture in the Negev desert after blending with other sources to reach suitable levels of salinity. The system was demonstrated in DEMOWARE in pilot scale (33 m³/h feed) and is assumed for the LCA in a scale of 1000 m³ feed/h (8.8 Mio m³/a).

It should be noted that different water quantities are provided to agriculture in each scenario due to different recovery rates within the three pilot systems (see Figure 6-4). However, the scaling to a comparable functional unit ("per m³ water supplied") will enable a direct comparison between the systems.

Data quality and limitations of this study

Important parameters for the LCA inventory are discussed below regarding data quality and uncertainties to clearly point out inherent limitations of this LCA. A summarized overview of data source and data quality is provided in Table 6-1.

- **Water qualities and quantities:** The data for water quality and quantities was provided by the operator Mekorot [124] as annual average from the year 2014 for the current reuse scheme.

Since this data is quality controlled, the data quality is assumed to be very good and representative. The water quality data of the different reuse schemes which have been tested in pilot scale is also of high quality, but may not be fully representative of a full scale system. For the alternative water supply schemes without water reuse, information on water quality is very limited and partly based on assumptions, so the quality of this data is assumed to be medium. Water quantities for these scenarios have been estimated to enable a good comparison to the existing reuse scheme.

- **Energy, chemicals and material consumption:** A detailed inventory on electricity and chemical consumption of the different tertiary treatment schemes (both pilot and full scale) was provided by the operator Mekorot [124]. The data quality regarding energy consumption for the current reuse scheme (long SAT and recovery wells) is assumed to be very good. Data quality of energy and chemical consumption for the pilot scale tests is also high, although upscaling may yield different consumptions in electricity compared to pilot systems. Energy consumption for alternatives to water reuse is reported by Mekorot and are based on estimates. For seawater desalination, data is based on literature [47] and feasibility studies for a SWRO system for the Vendee (FR) case study (cf. DEMOWARE deliverable 6.5). Material for infrastructure for all scenarios were estimated by KWB based on previous studies [49] (medium data quality), whereas existing infrastructure (e.g. national water carrier) has not been considered in this LCA.

Table 6-1: Overview on data quality of input data

Parameter/Process	Data source	Data quality
Reuse schemes		
Water quality and quantities (current reuse scheme)	[124], local operators	very good
Water quality (alternative reuse schemes, pilots)	[124], local operators	good
Electricity demand of recovery wells (current scheme)	[124], local operators	very good
Electricity and chemical demand (alternative reuse schemes)	[124], local operators	good
Infrastructure data	[49], estimations	medium
Scenarios for additional water supply		
Water quality (alternatives to reuse, natural water resources)	[124], local operators	medium
Energy demand for delivery with external water	[124], estimations	medium
Energy and chemical demand SWRO	[47, 124],literature, estimations	low-medium
Infrastructure data	[49], estimations	medium
Background		
Electricity mix (separate calculation)	[124], Mix of Israel 2015	medium
Chemicals and materials	EU or global datasets	medium
Transport	Truck transport (EU)	good

Normalization

Normalisation reveals the contribution of the system under study towards the total environmental footprint of each citizen. Principles for normalization and normalization factors are shown in Annex 9.1.1.

6.2.2 Inventory (Input data)

Primary data

Inventory data for this LCA study was provided by local operator Mekorot and complemented with estimates of KWB based on previous studies (Table 6-1). For consumptives, Table 6-2 summarizes the electricity demand and Table 6-3 summarizes chemical demand for all scenarios.

Table 6-2: Inventory data for energy demand (summarized in process modules)
data mainly provided by Mekorot [47, 124]

	Unit	0a. NWR (full-scale)	0b. PWM (full-scale)	1. LSAT (full-scale)	2. AOP & SSAT (pilot)	3. UF (pilot)	4. UF/RO (pilot)
Tertiary treatment, total	**MWh/a**	**5 200**	**5 200**	**41 405**	**4 044**	**3 674**	**8 509**
Pumping WWTP effluent/filter	kWh/ m^3 effluent	0.04	0.04	0.13	0.13	0.13	0.13
Recovery from aquifer (SAT)	kWh/ m^3 withdrawal	-	-	0.16	0.16	-	-
Media filter backwash	kWh/m^3 backwash	-	-	-	0.10	-	-
Ozonation incl. O_2 generation[9]	kWh/m^3 feed	-	-	-	0.15	-	-
Ultrafiltration (UF)	kWh/m^3 filtrate	-	-	-	-	0.28	0.28
Reverse Osmosis (RO)	kWh/m^3 permeate	-	-	-	-	-	0.60
WWTP (simplified)	kWh/m^3 backwash[10]	-	-	-	0.17	0.06	0.06
Freshwater delivery, total	**MWh/a**	**147 224**	**291 879**	**-**	**-**	**-**	**-**
Groundwater pumping	kWh/ m^3 water	1.20	1.20	-	-	-	-
Water from Lake Galilee	kWh/m^3 water	1.80	1.80	-	-	-	-
Marginal water	kWh/m^3 water	0.05	0.05	-	-	-	-
SWRO & delivery	kWh/m^3 water	3.50	3.50	-	-	-	-
Total electricity demand	**kWh/m^3 water supplied**	**1.17**	**2.29**	**0.28**	**0.42**	**0.46**	**1.24**

The scenarios relying on drinking water sources ‘0a NWR’ and ‘0b PWM’ have a higher electricity consumption compared to the current reuse scheme. Recovery of natural groundwater from deeper aquifer layers seems to be associated with higher electricity consumption than the recovery of infiltrated water from SAT. In particular, electricity demand of water supply via the National Water Carrier from Lake Galilee and via seawater desalination is very high. Overall, the supply of agriculture with potable water

[9] 7 mg/L ozone, 15 kWh/kg ozone for ozone generator, 10 kg O_2 per kg ozone required, 0.59 kWh/kg O_2 for oxygen generation

[10] Calculated in relation to COD, N and P load of backwash water

from the current water mix in Israel ('PWM') would increase energy consumption by a factor of 8 compared to the current reuse scheme. The new reuse options are more energy intensive than the existing SAT system. The main contribution to the gross electricity consumption is accredited to the recovery wells (SAT) and the membrane systems.

Chemical consumption is mainly accredited to membrane cleaning and maintenance to prevent membrane fouling (Table 6-3). In addition, pretreatment in the biofilter requires significant amounts of H_2O_2 and coagulant to yield a water quality suitable for ozonation and long-term infiltration without clogging problems.

Table 6-3: Inventory data for materials demand related to different volumes and aggregates

all concentrations per feed volume and referring to listed concentration of chemicals in water; MF = media filter; UF = wastewater UF; RO = wastewater RO; SW = seawater reverse osmosis per SWUF feed volume [47, 124]

Chemical	Unit	0a. NWR (full-scale)	0b. PWM (full-scale)	1. LSAT (full-scale)	2. AOP & SSAT (pilot)	3. UF (pilot)	4. UF/RO (pilot)
PACl (18 % Al)	mg/L	-	-	-	16.7 (MF)	-	-
H_2O_2 (50 %)	mg/L	-	-	-	54.0 (MF)	-	-
HCl (33 %)	mg/L	-	9.21 (SW)	-	-	0.39 (UF)	0.39 (UF)
Citric Acid (100 %)	mg/L	-	2.00 (SW)	-	-	0.63 (UF)	0.63 (UF) 0.06 (RO)
NaOCl (12 %)	mg/L	-	36.0 (SW)	-	-	7.48 (UF)	7.48 (UF) 3.00 (RO)
H_2SO4 (96 %)	mg/L	-	61.0 (SW)	-	-	-	38.0 (RO)
NaOH (40 %)	mg/L	-	1.44 (SW)	-	-	-	0.01 (RO)
$Na_2S_2O_5$ (100 %)	mg/L	-	-	-	-	-	0.42 (RO)
Antiscalant	mg/L	-	[11]	-	-	-	4.00 (RO)
Lime (92 %)	mg/L	-	122.5 (SW)	-	-	-	-
$FeCl_3$ (40 %)	mg/L	-	7.20 (SW)	-	-	-	-

Water Inventory

Table 6-4 shows the water volumes and qualities for the WWTP influent and the related effluents. Differences between influent and effluent are considered via backwash for the scenarios 'AOP & SSAT' and 'UF'.

[11] Antiscalant for SWRO assumed to be citric acid

Table 6-4: Water inventory including WWTP influent and effluent
Measured data by [124], * estimates and model calculations

Parameter	Unit	WWTP effluent	1. LSAT outflow	2. AOP &SSAT outflow	3. UF outflow	4. UF/RO outflow	4. UF/RO brine
Volume	Mm^3/a	130.0	149.5	9.6	8.1	6.9	*1.2 **
SS	mg/L	6.0	*0.5 **	*0.5 **	0.4	0.0	*2.7 **
COD	mg/L	40.0	3.5	4.0	42.0	*0.3 **	*265.3 **
DOC	mg/L	9.8	1.1	1.42	8.5	*0.3 **	*55.3 **
TN	mg/L	7.2	4.8	4.93	5.0	0.5	*33.0 **
TP	mg/L	1.0	0.3	0.3	0.65	0.02	*4.3 **
Cd	µg/L	*0.1 **	*0.1 **	*0.1 **	*1.0 **	-	*0.7 **
Cr	µg/L	*1.5 **	*1.5 **	*1.5 **	*1.5 **	-	*10.0 **
Cu	µg/L	19.0	*1.5 **	*1.5 **	18.0	-	*120.0 **
Hg	µg/L	*0.05 **	*0.05 **	*0.05 **	*0.05 **	-	*0.3 **
Ni	µg/L	*2.0 **	*1.5 **	*1.5 **	*3.0 **	-	*13.3 **
Pb	µg/L	*1.0 **	2.0	*2.0 **	*1.0 **	-	*5.7 **
Zn	µg/L	30.0	30.0	*30.0 **	53.0	-	*200.0 **

Background data

The materials for infrastructure and detailed information on background processes are shown in the Annex 9.6.1. Background datasets are extracted from ecoinvent database v3.1 [76].

Inventory for Water Impact Index

The Water Impact Index (WIIX) is calculated according to the basic methodology described in *D3.1* [11]. For water scarcity assessment, the water scarcity index (WSI) according to WULCA AWARE [52] is used in this study. Monthly WSI for all case studies are shown in the Annex 9.1.1 (Table 9-2). For freshwater resources, the annual WSI of 51.85 for groundwater and marginal water and 51.66 for Lake Galilee water is applied. A monthly differentiation of water withdrawals or releases was not conducted as aspects of seasonal water management were not addressed in this study.

The water quality index (WQI) is calculated based on the intake or effluent water quality parameters (for details see Table 9-37 in the annex). Water quality of the natural freshwater resources for the potable water mix is assumed with optimal water quality (WQI = 1). Similarly, quality of RO permeate is evaluated with WQI = 1 due to removal of most substances to a large extent (see Annex Table 9-37). For water quality of secondary WWTP effluent and UF filtrate (scenario 3 UF), Cu determines the WQI in this calculation and yields a very low quality index (0.07 for WWTP effluent, 0.08 for UF filtrate). However, the very low benchmark for Cu (1.4 µg/L) in WQI calculations is debatable, as the minor quality threshold value for Cu in groundwater by the German *LAWA* is 14 µg/L [77]. Higher Cu content in urban wastewater originates from corrosion of pipe materials. If Cu is neglected in WQI calculations, the WQI for the WWTP effluent is determined by phosphorus (0.20). For UF filtrate and SAT, Zn is now the dominant element for the WQI, but the data seems inconsistent (30 µg/L Zn in WWTP effluent, 53 µg/L Zn in UF filtrate), probably due to seasonal variations. Excluding also Zn from the WQI for all scenarios due to data inconsistency, phosphorus determines the WQI for all scenarios (see Table 9-37 in annex). This is seen as a reasonable basis for comparing all scenarios with the WIIX method, given that phosphorus is also present at significantly higher levels (factor 100) than Cu or Zn.

The water volumes for withdrawal and release are shown in Table 6-5 for all scenarios (cf. chapter 6.2.1). In the reference scenarios without water reuse, the Shafdan WWTP effluent is discharged into surface water and therefore fully accounted as environmental release. The water supplied to agriculture (accounted as well, with significant higher WQI than WWTP effluent) is provided by different sources from natural water (see Table 6-5). Water losses in agriculture are considered for all scenarios, the effective infiltrated water is set to 25 % [51]. Since seawater desalination is neglected in the generic scenario '0a. NWR', shares and volumes of groundwater, water from Lake Galilee and marginal water are higher, respectively.

Table 6-5: Overview on withdrawal and release of water and corresponding water quality indices (WQI) for the different scenarios

EF = WWTP effluent to surface waters; AG = agriculture; GW = groundwater; LW = Lake Galilee; MW = withdrawal of marginal water; SAT = ambient groundwater in SAT

Scenario	0a. NWR (full-scale)	0b. PWM (full-scale)	1. LSAT (full-scale)	2. AOP & SSAT (pilot)	3. UF (pilot)	4. UF/RO (pilot)
Withdrawal [10^3 m³/a]	98 113 (GW) 15 943 (LW) 15 943 (MW)	52 000 (GW) 8 450 (LW) 8 450 (MW)	19 500 (SAT)	1 200 (SAT)	-	-
WQI (withdrawal)	*1*	*1*	0.62	0.61	-	-
Release [10^3 m³/a]	32 500 (AG) 130 000 (EF)	32 500 (AG) 130 000 (EF)	37 375 (AG)	2 403 (AG)	2 015 (AG)	1 713 (AG)
WQI (release)	*1 (AG)* 0.20 (EF)	*1 (AG)* 0.20 (EF)	0.62	0.61	0.31	1.00

For the existing reuse scheme '1 LSAT' a withdrawal of 19.5 Mm³/year as ambient groundwater is accounted. A similar relative share of ambient groundwater is considered for the pilot system with short SAT. Evaporation in the SAT system is neglected in a first estimate. The water losses for the different pilot schemes are due to backwash/brine disposal in the treatment trains (media filter, UF and RO). The backwash of the media filter and UF are recycled to the Shafdan WWTP and not further accounted in WIIX calculations. RO brine is released directly into the ocean, and is also not accounted in WIIX calculations as release.

A detailed sensitivity analysis for the WIIX calculation is conducted in chapter 6.2.4.

6.2.3 Impact Assessment (Results)

Environmental impacts were assessed with a set of 8 impact categories (including WIIX), representing different areas of environmental concern. After an overview of all indicators, selected impact categories are discussed in detail to reveal individual contributions of processes to the total environmental impact.

Environmental impacts and benefits of all scenarios per m³ water supplied for agriculture

The environmental profile of all scenarios for all selected impact categories per m³ water supplied for agriculture is shown relatively to the gross impact of the scenario '0b PWM' (= 100 %) in Figure 6-5.

The fossil and nuclear cumulative energy demand (CED), the global warming potential (GWP) and terrestrial acidification potential (TAP) are strongly influenced by the background processes, such as electricity, chemicals or material production, while electricity production is dominant for all three impact categories due to the fossil-based electricity mix in Israel. The alternatives to water reuse ('0b PWM') show higher CED, GWP, TAP compared to all reuse options. Comparing different alternatives for water reuse in these impact categories, a ranking similar to the gross electricity consumption (Table 6-2) can be observed: treatment via Long SAT has the lowest environmental impact due to its low electricity demand. However, it has to be kept in mind that this option requires the highest area compared to the alternatives. It is followed by AOP & SSAT and UF-treatment, whereby UF filtrate needs blending with other water sources, since it does not fulfil all necessary quality criteria for water reuse. Finally UF/RO is most energy consuming reuse option, but is reveals the smallest footprint regarding area consumption.

The impact category of eutrophication indicates significant benefits for most reuse options. Since the WWTP effluent is not discharged into surface water in reuse scenarios, the direct load of nutrients into surface water is significantly reduced by water reuse. Nonetheless, a certain fraction of N and P may negatively affect the aquatic environment in reuse schemes, e.g. during SAT treatment due to potential transport of nutrients with groundwater. The membrane schemes have similar eutrophication potentials compared to SAT as nutrients are removed from WWTP effluent (P) or directly transferred to agriculture (N). In scenario '4 UF/RO', nitrogen loads of WWTP effluent are concentrated in the brine and directly discharged to the Mediterranean Sea, which results in a high marine eutrophication potential (MEP) comparable to direct discharge of WWTP effluent. Backwash of UF or media filter is recycled back to the WWTP influent and causes significant emissions of N and P loads into sewage sludge which is also disposed in the ocean, causing the related MEP due to its nitrogen content.

In ecotoxicity potential (ETP), heavy metal loads in WWTP effluent are mainly responsible for the potential impacts. Direct emissions of heavy metals in surface water (0a and 0b) are accounted higher than emissions via SAT or in agriculture, so that reuse scenarios have a lower impact in this category.

In human toxicity potential (HTP), 3 out of 4 water reuse scenarios show significantly high potential impacts (factor 9 to the baseline) due to heavy metal emissions into agriculture with reclaimed water. This high score is due to the following reasons:

- Removal of heavy metals is relatively low in UF, media filter or SAT. Only RO membranes show a significant removal of trace metal concentrations present in WWTP effluent of Shafdan.
- The model predicts high human toxicity potentials for heavy metal input in agricultural soils (especially Zn) compared to their input in freshwater. The characterisation factors for human toxicity potential applied in this study are taken from the consensus model USEtox™ and are known to be affected with high uncertainties (see chapter 8.4 for a detailed discussion).
- Heavy metal content in alternative water sources is set to 0 according to information from Mekorot [124]. This simplification will underestimate metal loads into agricultural soils for the reference scenario '0b PWM' and hence distort the comparison between reuse and alternative scenarios.

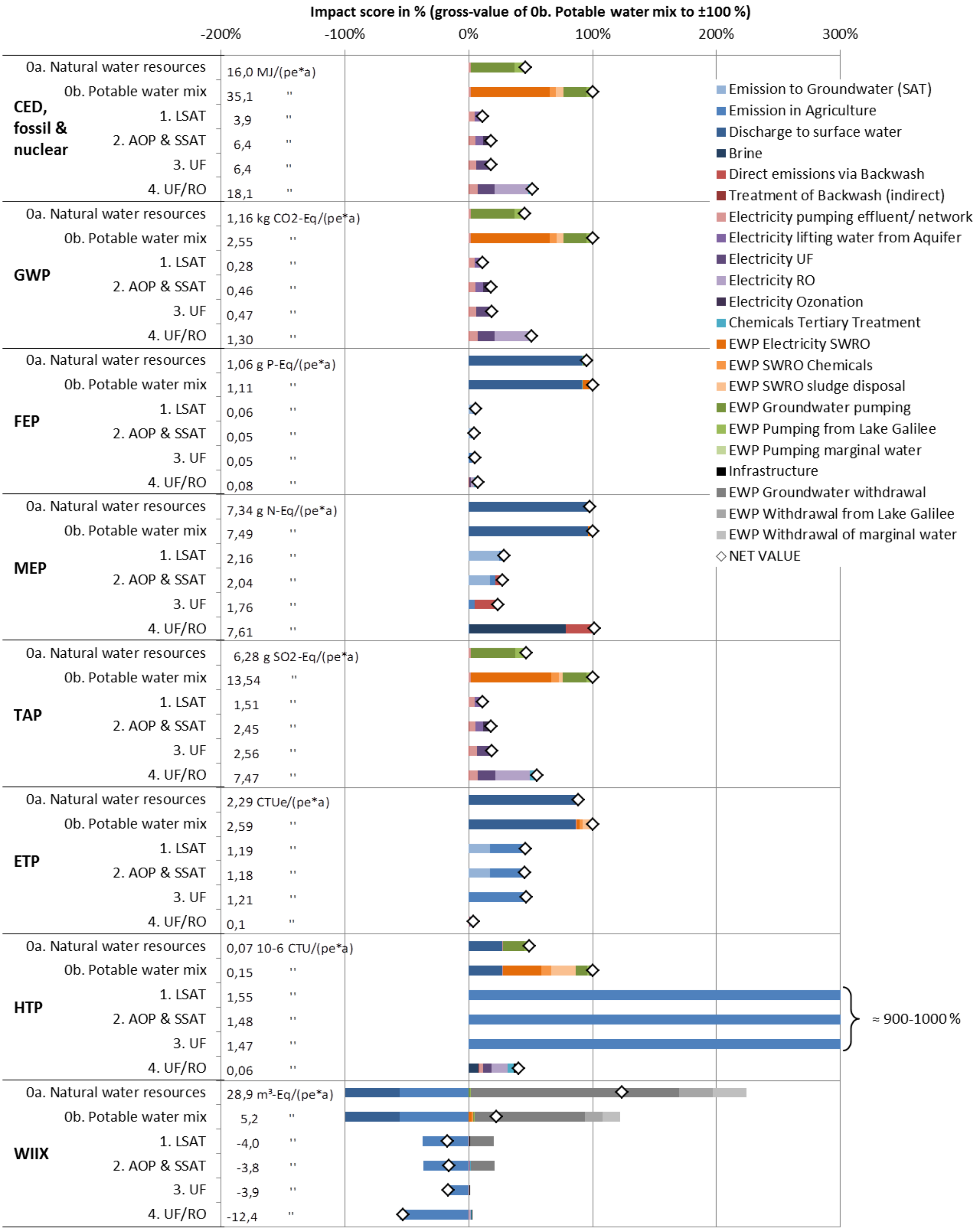


Figure 6-5: Environmental profile for all scenarios related to gross value of '0b. PWM' (= 100 %) and total net values per scenario and impact category

CED = cumulative energy demand; GWP = global warming potential; FEP = freshwater eutrophication potential; MEP = marine eutrophication potential; TAP = terrestrial acidification potential, ETP = eco toxicity potential; HTP = human toxicity potential; WIIX = water impact index; EWP: External water supply

The WIIX is determined by direct water withdrawal and release; indirect contributions by background processes play only a minor role. For the hypothetical scenario '0a NWR', water withdrawal for irrigation and is respectively higher than the subsequent release in agriculture since only 25 % of water is accounted. The net WIIX is also determined by the credits for release of WWTP effluent. Reuse scenarios have no (or marginal) water withdrawal and accredit higher benefits for water released, as secondary effluent is further treated and water quality is improved (higher WQI). RO membrane permeate has the highest quality and consequently highest credits in WIIX of all reuse scenarios. Using the potable water mix for irrigation (0b PWM) with a significant share of SWRO water of highest quality and no direct impact on freshwater resources, WIIX of this scenario is superior to the reuse scenarios. The results for WIIX are further discussed in a sensitivity analysis (see chapter 6.2.4).

Relative comparison of scenarios in energy demand and nitrogen emissions

For a more detailed analysis of environmental benefits and impacts of new water reuse options, the existing reuse system (1 LSAT) is taken as benchmark, and relative changes towards the benchmark are shown for the impact categories CED (Figure 6-6) and MEP (Figure 6-7) for each of the different new reuse schemes tested in DEMOWARE.

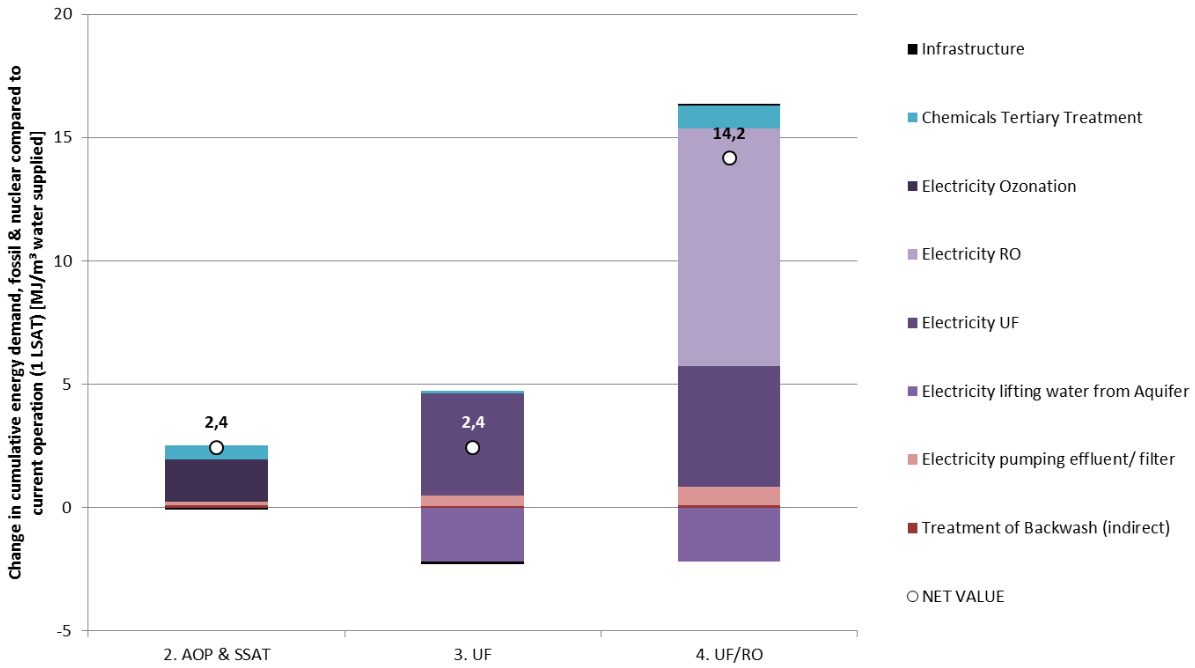


Figure 6-6: Changes in fossil and nuclear cumulative energy demand of reuse schemes compared to '1 LSAT'

With an extensive pretreatment via filtration/ozonation (2 AOP & SSAT), net CED of the reuse scheme increases by 2.4 MJ/m³ or 62% mainly due to electricity for ozonation and use of H_2O_2 as oxygen supply in the biofilter. Well operation is assumed comparable for long and short SAT, so that no energy benefits can be expected for the water recovery. If only UF membranes are operated (3 UF), CED of water reuse also increases by 2.4 MJ/m³ or 62% compared to the existing scheme. Beside higher electricity needs for operation and backwash of the membranes, energy for water recovery after SAT can be avoided, yielding overall in a similar energy footprint as (2 AOP & SSAT). It has to be kept in mind here that UF permeate quality does not allow a direct application in water reuse, but UF product has to be blended with other sources to comply with the legislative standards. A double-membrane system (4 UF/RO) has by far the highest energy demand of all reuse options and would increase CED of the system by 14.2 MJ/m³ or

359%. This is mainly due to the energy and chemcials demand for RO operation, which delivers a very high water quality (cf. WIIX) but comes only at significant efforts in electricity and chemicals.

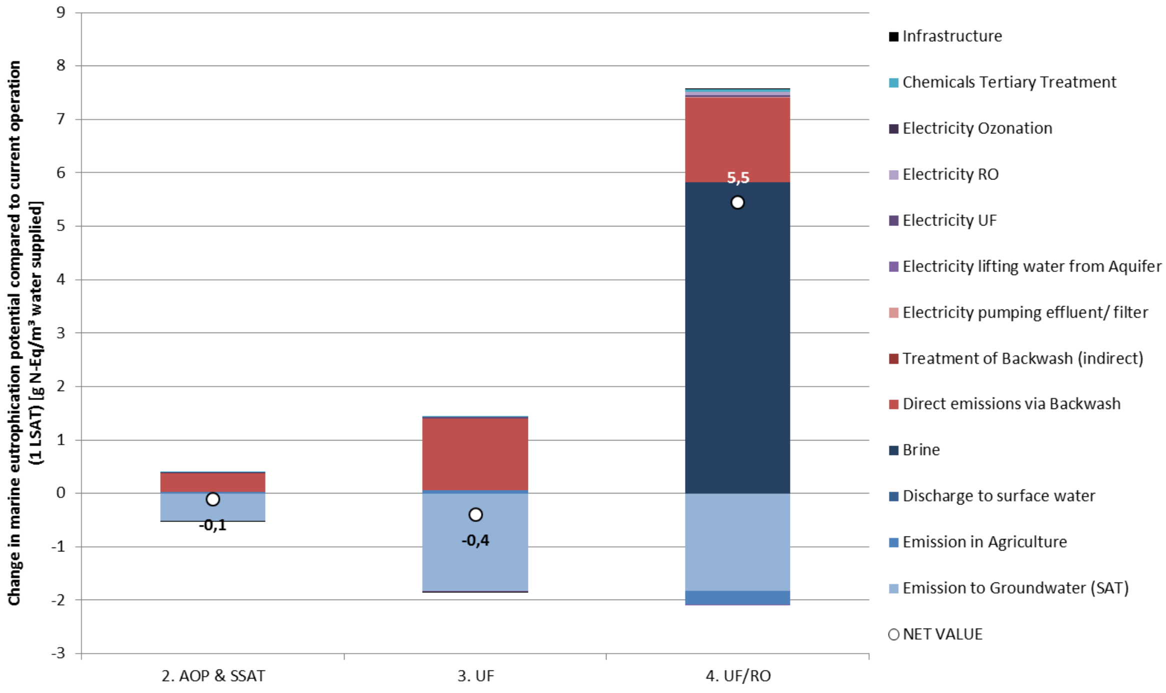


Figure 6-7: Changes in marine eutrophication potential of the reuse schemes compared to '1 LSAT'

In case of MEP (Figure 6-7), the fate of nitrogen in WWTP effluent determines the changes in this impact category between the different reuse schemes. In '2 AOP & SSAT', the media filter will remove some nitrogen which is not introduced into the aquifer, decreasing this impact pathway. However, some nitrogen which is recycled to the WWTP inlet with filter backwash ends up in the sewage sludge (around 30% of recycled N), which is then disposed directly to the sea and causes full eutrophication potential here. UF treatment will directly transfer effluent N to agriculture, omitting N input into groundwater during the SAT stage. However, backwash of UF will be recycled to WWTP inlet and has an impact via sewage sludge disposal in the sea. In total, this scenario will reduce MEP by 18% compared to the existing scheme. In case of '4 UF/RO', nitrogen of WWTP effluent will be fully concentrated in the brine, which brings a high MEP due to its direct discharge into the sea. The latter scenario increases MEP considerably compared to the existing reuse scheme (+ 253%), because nitrogen is directly transferred into the marine environment and will potentially cause direct eutrophication there.

Normalization to the total environmental footprint per person

The normalized score for each impact category (Figure 6-8) shows the contribution of the reuse system to the total environmental impact per person in EU-27 (normalization data can be found in Table 9-1). Specific normalisation data for Israel was not available for this study.

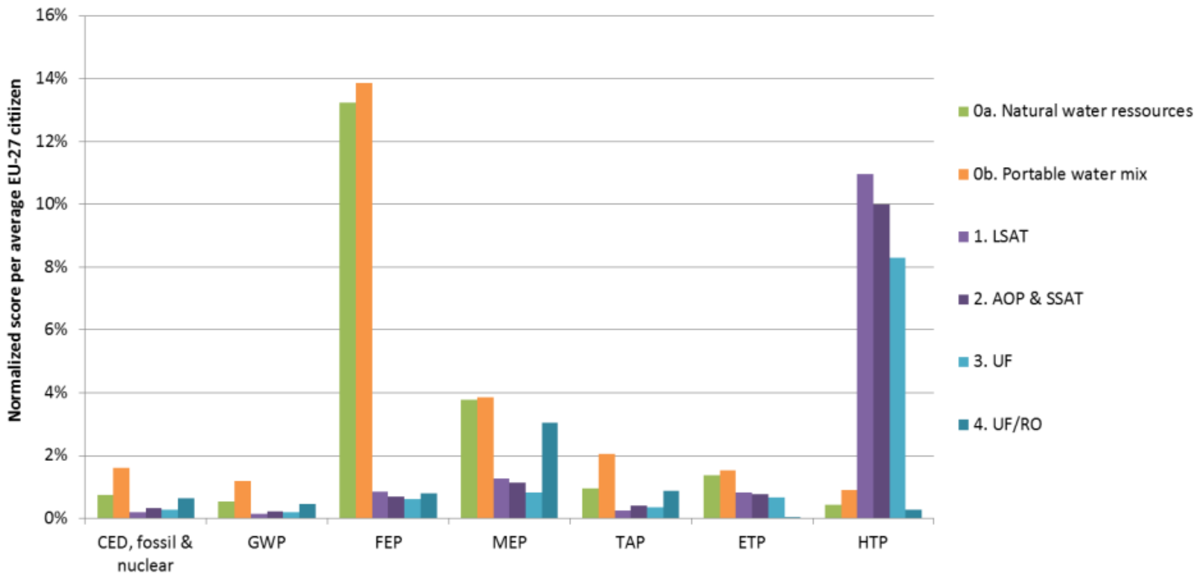


Figure 6-8: Normalized scores for all impact categories per average EU-27 citizen

For all reuse schemes, CED and energy-related emissions GWP and TAP contribute around 0.5 % to the total environmental footprint in these categories per citizen in the EU-27. Using the potable water mix for irrigation increases the score to 1.5-2%, whereas water supply from 100% freshwater resources is in between. For water quality indicators FEP and MEP, water reuse can signicantly reduce the total contribution of 13-14% (FEP) and 4% (MEP) due to fewer emissions with WWTP effluent, eliminating nutrients in tertiary treatment or redirecting them to agriculture. Brine disposal of RO into the sea offsets this benefit for the '4 UF/RO' scenario in case of nitrogen, which should be further treated before dumping brine into the sea. Ecotoxicity potentials can also be reduced with water reuse, but are relatively low (< 2%) for all scenarios analysed. Human toxicity potentials are high for water reuse systems without RO due to associated Zn emissions into agriculture, which has been discussed above. Further discussion of HTP normalisation and impact assessment is provided in chapter 8.4. Overall, normalisation shows that water reuse in Shafdan can significantly improve water quality aspects by avoiding nutrient emissions of WWTP effluent into surface and marine waters, saving energy for water supply and associated emissions of greenhouse and acifidying gases at the same time.

6.2.4 Interpretation and Discussion

Table 6-6 gives a summary on the net environmental efforts and benefits of all scenarios for all impact categories in relation to the existing reuse scheme '1 LSAT' as a benchmark.

Alternative supply options for irrigation water from natural freshwater resources or combined with SWRO water increase environmental impacts in almost all impact categories considerably. They increase energy demand and related emissions, either by factor of 3 in case of 100% freshwater use or by factor 8 in case of the potable water mix with 53% SWRO filtrate. In addition, direct discharge of WWTP effluent will have a negative impact on water quality in receiving water bodies due to increased nutrient loads. Water scarcity will be increased if 100% freshwater resources are used, whereas SWRO water relieves water stress by supplying "new" water and highest quality, but comes with a considerable energy burden.

Table 6-6: Summary of net environmental efforts and benefits of the scenarios for all impact categories, related to the existing reuse system '1 LSAT'

Scenario	1. LSAT	0a. NWR	0b. PWM	2. AOP & SSAT	3. UF	4. RO
CED	3.9 MJ/ m^3	+ 307 %	+ 791 %	+ 62 %	+ 62 %	+ 359 %
GWP	0.28 kg CO_2-Eq/ m^3	+ 309 %	+ 799 %	+ 62 %	+ 65 %	+ 358 %
FEP	0.06 g P-Eq/ m^3	+ 1676 %	+ 1762 %	- 16 %	- 11 %	+ 37 %
MEP	2.16 g N-Eq/ m^3	+ 240 %	+ 248 %	- 5 %	- 18 %	+ 253 %
TAP	1.51 g SO_2-Eq/ m^3	+ 317 %	+ 798 %	+ 62 %	+ 70 %	+ 395 %
ETP	1.19 CTUe/ m^3	+ 92 %	+ 118 %	- 1 %	+ 1 %	- 92 %
HTP	1.55 10^{-6} CTUh/ m^3	- 95 %	- 91 %	- 4 %	- 5 %	- 96 %
WIIX	- 30.3 m^3-Eq/ m^3	+ 67 %	- 11 %	+ 2 %	+ 48 %	- 69 %

The environmental profiles of the new reuse schemes tested in DEMOWARE reveal the following points:

- An advanced SAT system with extensive pretreatment (media filter and ozonation) increases energy consumption and related greenhouse gas emissions by 62%. However, this scheme has a considerably smaller area footprint (see discussion below), provides an additional barrier against phosphorus, residual organics and emerging micropollutants (not reflected in this LCA) and enables a sustainable long-term operation of the SAT system, overcoming existing operational problems due to oxygen deficiency in the SAT and manganese remobilization.
- Applying a UF membrane system as single treatment, energy demand and associated emissions of water reuse increase by 60-70%. However, water quality of UF is not fully comparable to the SAT system and does not comply with the legal standards for water reuse, requiring a blending with other water sources. This lower water quality also leads to a lower credit for the water footprint, indicating that UF water is less valuable than the SAT water. However, as UF does not require SAT treatment, the risk of potential groundwater contamination with nutrients is mitigated. Additionally UF is not remove trace organics, whereby the other treatment options do.
- A full-scale UF-RO scheme for water reuse would more than double the energy consumption and associated emissions of water reuse at Shafdan. In addition, brine disposal into the sea leads to high nitrogen emissions into the marine environment and may cause eutrophication there. However, RO membranes deliver the highest quality of water for reuse, indicated by a significant decrease in freshwater eutrophication, toxicity, and by far the highest credit for reused water in water footprint.

In addition to the environmental profiles analysed in this LCA, other factors are also important for the future upgrade/expansion of the Shafdan reuse system: a) the land footprint due to heavy constraints in available land and b) the overall losses in water during the tertiary treatment, targeting a maximum use of precious water for reuse. Table 6-7 gives an overview of these factors for all tested reuse systems in relation to the existing reuse scheme 'LSAT'. Since the UF-filtrate can only be used for blending water from LSAT by 5 % this mix-scenario is considered in Table 6-7.

Moving to a short SAT system with extensive pretreatment will reduce land footprint requirements by 76%, but comes at the cost of 62% higher energy consumption. Water losses in this system are minor (5%) and can be reused again, as water is recycled to the WWTP inlet. A mixture of UF-treatment and Long SAT-treatment increases the energy footprint by 3 %, while simultaneously the area consumption is reduced by approximately 5 %. Nearly all water from the WWTP effluent is recovered. Pure membrane systems (UF/RO) have the most compact footprint and require less than 1% of the current area of the SAT system, but will additional energy (UF/RO). Another drawback of the latter scheme is the loss of water via brine, which cannot be recovered as it is directly dumped into the sea.

Table 6-7: Comparison of energy demand and required land footprint of potential full-scale systems in relation to the existing reuse scheme (LSAT)

Scenario	2. AOP & SSAT	95 % 'LSAT' + 5 % 'UF'	4. UF/RO
Energy consumption	+ 62 %	+ 3 %	+ 359 %
Area consumption	- 76 %	≈ -5 %	> - 99 %
Water recovery rate	95.4 %	99.6 %	78.2 %

Sensitivity analysis on Water Impact Index

During inventory setup and results discussion of the WIIX for Shafdan, the following aspects have been raised and call for a more detailed discussion in sensitivity analysis:

1. The calculation of the WQI (Cu and Zn were excluded due to data quality and very low benchmarks in the water framework directive)
2. The suitability of the functional unit (m^3 water supplied) given that water losses in tertiary treatment are not reflected in the WIIX then (e.g. brine of RO, but also potential evaporation in SAT ponds)

Both aspects will be analysed separately and in combination in the following discussion.

Figure 6-9 shows the significant impact of the WQI in the WIIX results. If Cu and Zn are included in the WQI calculation, Cu determines the WQI for WWTP effluent (WQI = 0.07) and UF filtrate (WQI = 0.08), while Zn determines the WQI for SAT product (WQI = 0.26). Consequently, water releases of reuse water and WWTP effluent account for less credits in the WIIX, and the net WIIX for all scenarios is higher (= more water stress) except of '4 RO'. In conclusion, WQI has a high impact on WIIX results, so careful analysis of the underlying data quality should be done to validate the list of substances used for WQI calculation.

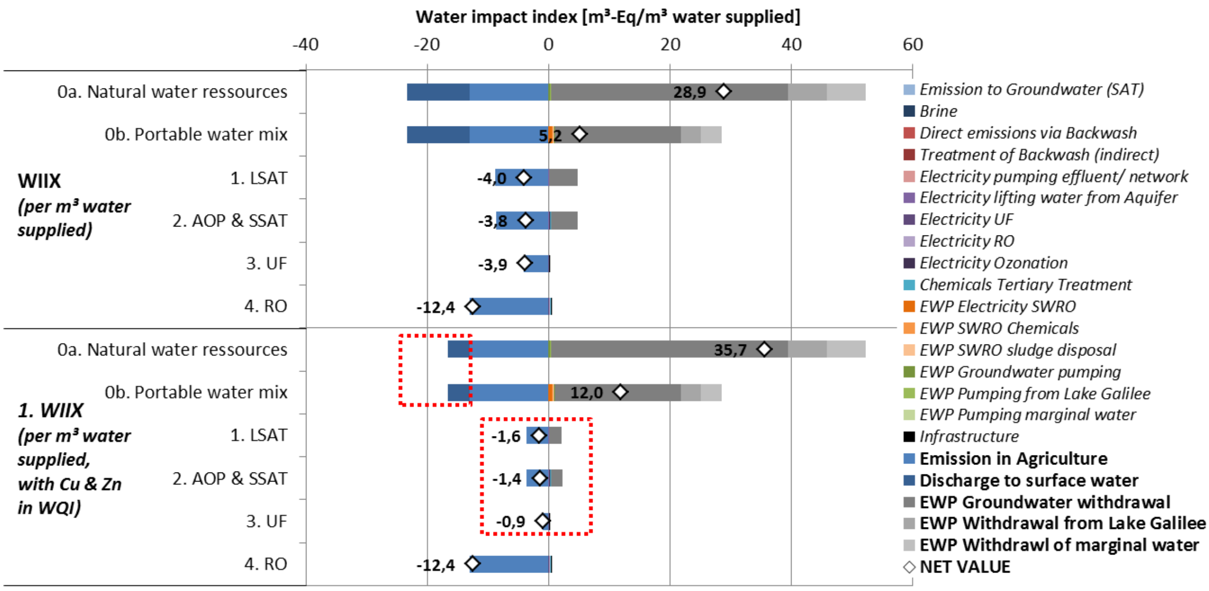


Figure 6-9: Water Impact Index with different WQI (w and w/o exclusion of Cu and Zn in WQI calculation)

Figure 6-10 shows the WIIX using different functional units "per m³ water supplied" and "per m³ influent in tertiary treatment" to better reflect water losses in tertiary treatment. In addition, water losses in SAT ponds have been accounted with estimated 5% water lost by evaporation during infiltration. This assumption refers to an average annual evaporation rate of 1.6 cm/d, which may overestimate real losses since evaporation was assumed with 0.5-1.0 cm/d by Mekorot [124].

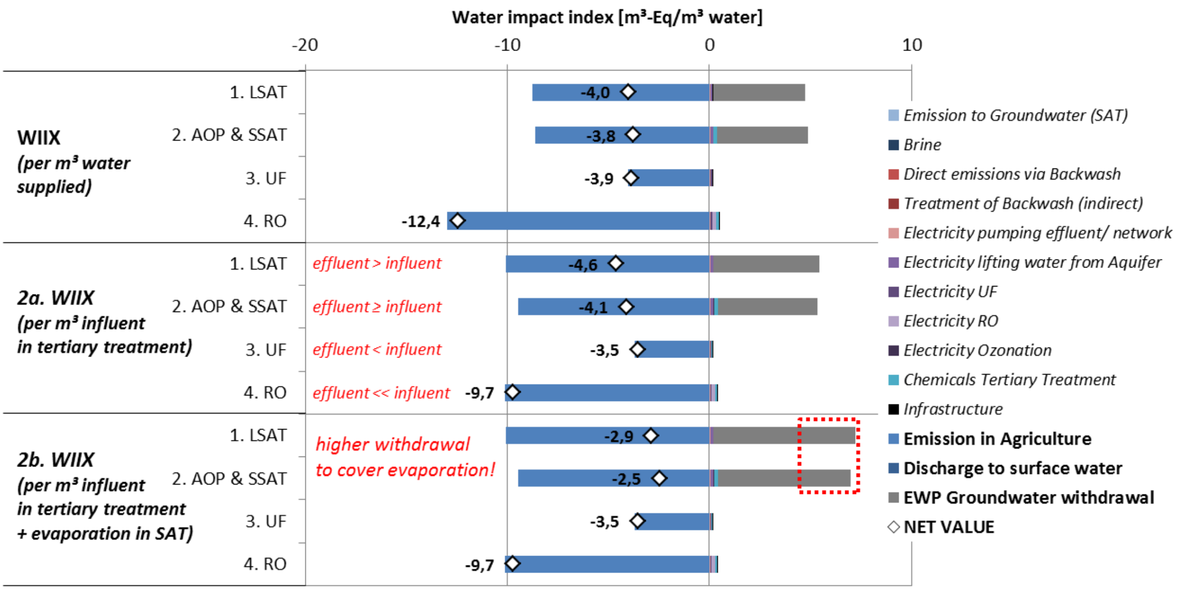


Figure 6-10: Water impact index for reuse schemes considering different functional units (2a) and water losses/consumption by tertiary treatment options (2b)

If WIIX is calculated in relation to the volume available for reuse (m³ influent in tertiary treatment), losses in membrane processes via backwash or brine lead to a lower WIIX credit for water reuse in scenarios '3 UF' and '4 UF/RO'. For SAT systems, WIIX credits get even higher with the new functional unit, because ambient groundwater drawn during SAT increases total water produced in the system, which is now fully allocated to the lower volume entering the SAT system. Relating WIIX to the water input of the reuse systems will show lower benefits for systems with water losses (e.g. membranes), but higher benefits for systems which increase water volumes during treatment (e.g. SAT).

The consideration of evaporation in SAT results in larger WIIX for drawing ambient groundwater, as these will be used to offset losses in evaporation. However, net WIIX scores do not change significantly in the present study, because evaporation is only 5% of the total volume.

Neglecting water quality aspects in water footprinting due to the uncertainties in WQI calculation discussed above, a "water availability footprint" (WAF) can be calculated based only on volumes and scarcity indices (Figure 6-11, inset 3a). Now, all reuse options have comparable benefits in WAF, with a minor disadvantage for SAT schemes due to the need for ambient groundwater. However, water reuse is no longer superior to the scenarios with alternative water supply, because WWTP effluent is accounted with the same credits than agricultural irrigation. In other words, if quality does not matter, WWTP effluent discharge will yield the same benefits in WAF than water reuse. The potable water mix with 50% SWRO has the best WAF, as "new" freshwater is generated from seawater which complements existing freshwater resources in the catchment. Finally, neglecting water quality aspects in water footprinting will not enable to show the benefits of water reuse from WWTP effluent.

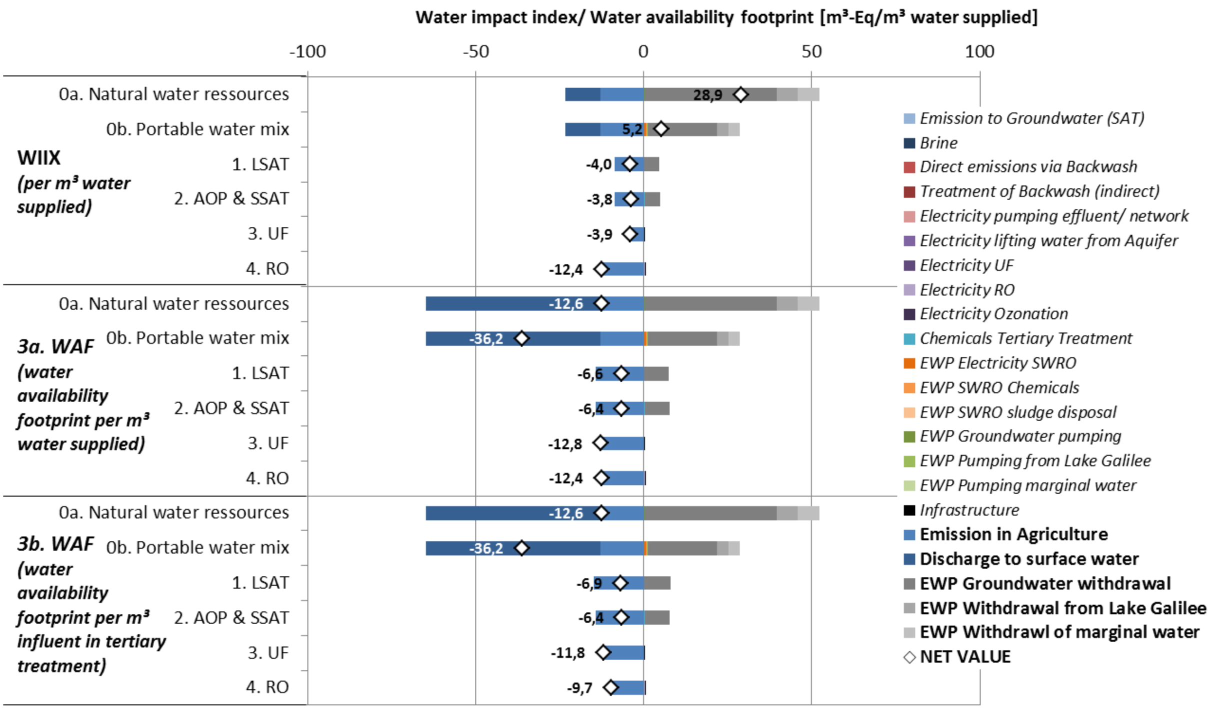


Figure 6-11: Water Impact Index and Water availability footprint (with different functional units in 3a and 3b)

WAF may also be combined with the new functional unit "m³ influent in tertiary treatment", leading to lower WAF credits for those systems with water losses during treatment, i.e. membrane systems (Figure 6-11, inset 3b). As high quality of RO filtrate is no longer reflected in WAF, this scenario has the lowest credits in WAF, and SAT systems are now superior over membrane systems regarding water stress only (and not water quality).

6.3 Conclusions

The following conclusions can be summarized from the LCA study of the Shafdan reuse site:

1. **Water reuse in Shafdan reduces water stress for local freshwater resources and also requires less energy and related greenhouse gas emissions than most other water sources** (e.g. national water carrier, seawater desalination)
2. **Shorter SAT with filtration/ozonation as pretreatment has 62% higher energy demand and GHG emissions than existing SAT**, but requires less area footprint (-76%) and provides an additional barrier against organic micropollutants. Due to complete nitrification and oxygen saturation during pretreatment, potential clogging of infiltration ponds and irrigation systems by Mn precipitates will be mitigated.
3. **Partial UF treatment (5%) of secondary effluent in addition to current the long SAT treatment increases energy demand and associated GHG emissions by 3 %** and reduces area footprint by 5 %. A UF treatment for the entire reuse scheme is not an option since the water quality of UF effluent will require blending with other reclaimed water sources to reach legal standards for irrigation (bulk organics or DOC).
4. **UF/RO treatment requires significantly more energy than the existing system (+360%) and causes high GHG emissions, but also delivers the highest water quality.** Brine disposal of RO can negatively affect marine environment if directly discharged into the sea.
5. **Toxicity effects via heavy metals in reclaimed water are difficult to predict** and should be closely monitored. In addition, quality of freshwater sources and drinking water could be evaluated in detail to verify the comparison between reclaimed water quality and drinking water quality.
6. **Results of comparison of water supply options in WIIX have a high sensitivity to water quality aspects.** Data of water quality and reference concentrations has to be improved to strengthen WIIX results.

7 Case Study of Torreele

7.1 Introduction and Setting

The Intercommunale Waterleidingsmaatschappij van Veurne-Ambacht (IWVA) is one of the pioneers in indirect potable reuse. At the Torreele facility, the municipal wastewater effluent from the adjacent wastewater treatment plant (WWTP) of Wulpen is treated using ultrafiltration (UF) prior to reverse osmosis (RO). After RO treatment, reclaimed water is used to recharge the unconfined dune aquifer of St-André [125], from where drinking water is produced by groundwater abstraction and a multi-stage drinking water treatment plant. This water reuse/infiltration scheme is operational since July 2002.

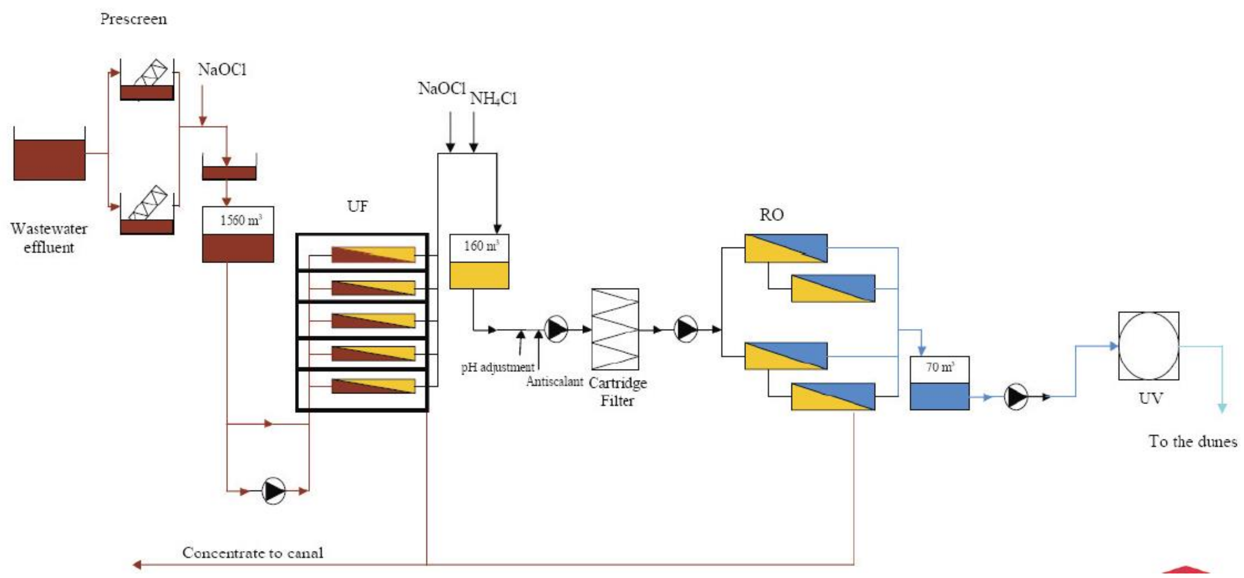


Figure 7-1: Tertiary treatment scheme for water reclamation at Torreele (© IWVA)

A main issue concerning membrane operation is related to the concentrate management. Mixed concentrates of the membrane processes (35% is UF backwash and 65% is RO concentrate) is discharged into the adjacent canal, together with the remaining part of WWTP effluent that has not been treated for water reuse [126]. As the canal is brackish, the salinity of the concentrates (diluted by the WWTP effluent) does not have a major negative effect on the water quality in the canal. Regular sampling showed that the water quality downstream of the Torreele facility is only negatively affected after longer dry periods in the area.

In August 2016 IWVA started reuse of UF backwash water, using a continuous sand filter for backwash water treatment. The remaining concentrate to be discharged is now mainly RO concentrate.

Almost from the start-up of the scheme in Torreele, IWVA performed tests using natural systems to treat the discharged water and mitigate the effect of concentrate discharge. From October 2003 until 2009 the IWVA performed a test using a subsurface flow reed bed (constructed wetland). It proved not to be tolerant for higher salinity. In April 2007, a first test using willows (*Salix*) was performed under the same conditions as reed [125].

In 2010 10 different willow species were tested for their salt tolerance, and in 2011 a test field of 28 m² containing 70 willows of 9 different species was installed and put into operation [127]. The set-up was considered as a Short Rotation Coppice (SRC), a crop of wooden species planted at very high density with the intention to produce wood. 'Short Rotation' reflects to the frequency of harvesting which is in the order of 2 to 3 years and the biomass produced is considered a renewable energy source. It can be used for heating. Within the DEMOWARE project the performance of the willows was investigated and compared to a conventional post denitrification pilot. The test ended at the end of March 2016 and the results are reported in deliverable D1 2 [50].

7.2 Life Cycle Assessment

7.2.1 Goal and scope definition

The goal of this LCA is to analyse and compare different options for wastewater discharge and water supply in the area of the city of Koksijde, including the existing scheme for indirect potable reuse and potential alternatives. This LCA can serve as example for sites with water scarcity situated at the coast, quantifying the environmental profile of different alternatives, such as importing water from an external water supply, seawater desalination or indirect potable reuse (IPR) of tertiary treated wastewater. A second goal of this LCA is to assess additional efforts for and benefits of an innovative brine treatment in the reuse train, which was demonstrated in DEMOWARE [50]. The target group of this study consist primarily of the local stakeholders such as the treatment plant operators (IWVA), but also planers and engineers in the field of wastewater treatment and water supply.

Function/Functional Unit

The function of the system includes enhanced treatment or discharge of WWTP secondary effluent and the provision of drinking water for the city of Koksijde and its surroundings, considering all processes that are related to these functions. Consequently, the functional unit of this LCA is defined by providing this service annually, scaled to the organic load of the wastewater treatment process measured in population equivalents (pe), i.e. “per pe and year” or $(pe*a)^{-1}$. The Wulpen WWTP treats wastewater of 68 000 pe in annual average [48], amounting to a total volume of 2.7 Mio m^3/a of secondary effluent which is used as feed water for the reuse train. The scenarios also supply 3.3 Mio m^3/a of drinking water to the customers.

System boundaries

System boundaries of this LCA include tertiary treatment of secondary WWTP effluent (and brine treatment before discharge) as well as infiltration of this water, its recovery in groundwater pumping, and the final drinking water treatment at Koksijde. For scenarios of alternative water supply, water treatment and transport are included within the study as well. Background process for production of electricity, chemicals, fuels, materials, infrastructure and maintenance are also accounted (Figure 7-2).

Primary and secondary wastewater treatment at WWTP Wulpen is excluded from the LCA, because water reuse has no effect on the upstream mainline of the WWTP, as UF backwash water and RO brine of tertiary treatment are discharged to a nearby cannel which drains to the North Sea. For the other scenarios of water supply, the complete secondary effluent of the Wulpen WWTP is also discharged into this canal.

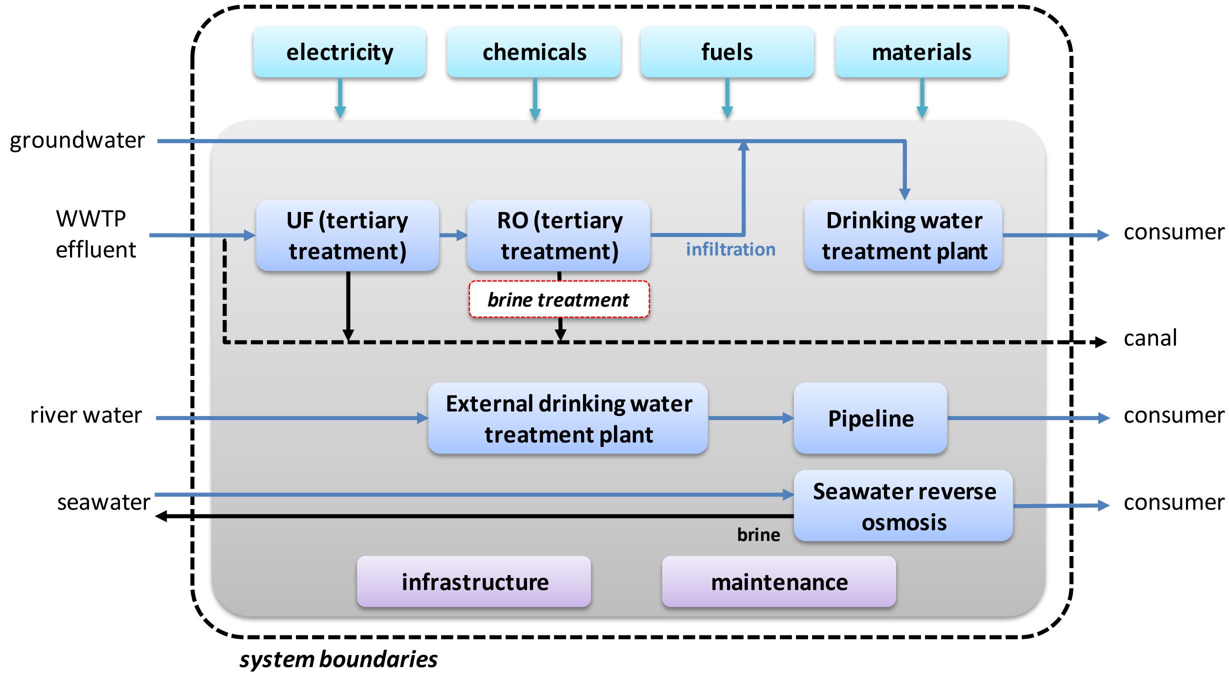


Figure 7-2: System boundaries and scope for LCA study Torreele*Scenarios*

The scenarios represent different approaches to increase the local availability of drinking water in Koksijde, including indirect potable reuse of WWTP effluent, water import, and seawater desalination. For the IPR scheme, a modified scenario with innovative brine treatment is also included. A comparative overview of all scenarios including scenario description and annual water volumes (Figure 7-3) is provided below:

1. IPR: indirect potable reuse represents the existing situation since implementation of the reuse scheme in 2002. Part of the effluent of the Wulpen WWTP (2.73 Mio m³/a) is treated via ultrafiltration (UF) and reverse Osmosis (RO) before infiltrating the RO permeate (1.88 Mio m³/a) in the dunes next to Koksijde. UF backwash water and RO brine are discharged into the canal which drains to the sea. Water is recovered after groundwater recharge, drawing 3.32 Mio m³/a as a mixture of reclaimed water and ambient groundwater. This water is then treated in a drinking water treatment plant with aeration, filtration, and finally UV disinfection before distribution to the consumers.
2. IPR & Willows: This scenario extend scenario '1. IPR' with an innovative polishing treatment for RO brine using a willow field (see D1.2 [2]). The RO brine is discharged into a willow field (10'000 m²), where carbon, nitrogen and phosphorus are biologically degraded or taken up by plants. The effluent of this field with lower COD, TN and TP loads is then discharged into the canal. The willows are harvested regularly, and the biomass is valorised in a biogas plant with CHP unit to produce electricity and/or heat.
3. Network: This scenario represents the import of drinking water from northern Wallonia via a pipeline network (120 km). Imported water from Wallonia originates from surface water resources, so that surface water treatment is included in this scenario, as well as the efforts for water transport in the pipeline. For this study, surface water treatment consists of ozonation, sand filtration, GAC filtration, and final disinfection via chlorination. The volume of imported water is calculated by using the maximum permitted groundwater withdrawal of 1.7 Mio m³/a in the Koksijde plant (without water reuse), complementing this volume with 1.62 Mio m³/a drinking water via import to deliver a comparable total amount of drinking water as in scenario

IPR. Without water reuse, the total volume of secondary effluent of the Wulpen WWTP is discharged into the cannel.

4. SWRO: Seawater desalination can also be used for complementing the water supply of Koksijde, producing drinking water in a UF/RO system from seawater. UF backwash water and RO brine of seawater desalination are directly discharged back to the North Sea. The volume of additional drinking water production is calculated to 1.62 Mio m³/a in analogy to scenario '3. Network' to end up with a comparable amount of drinking water in all scenarios.

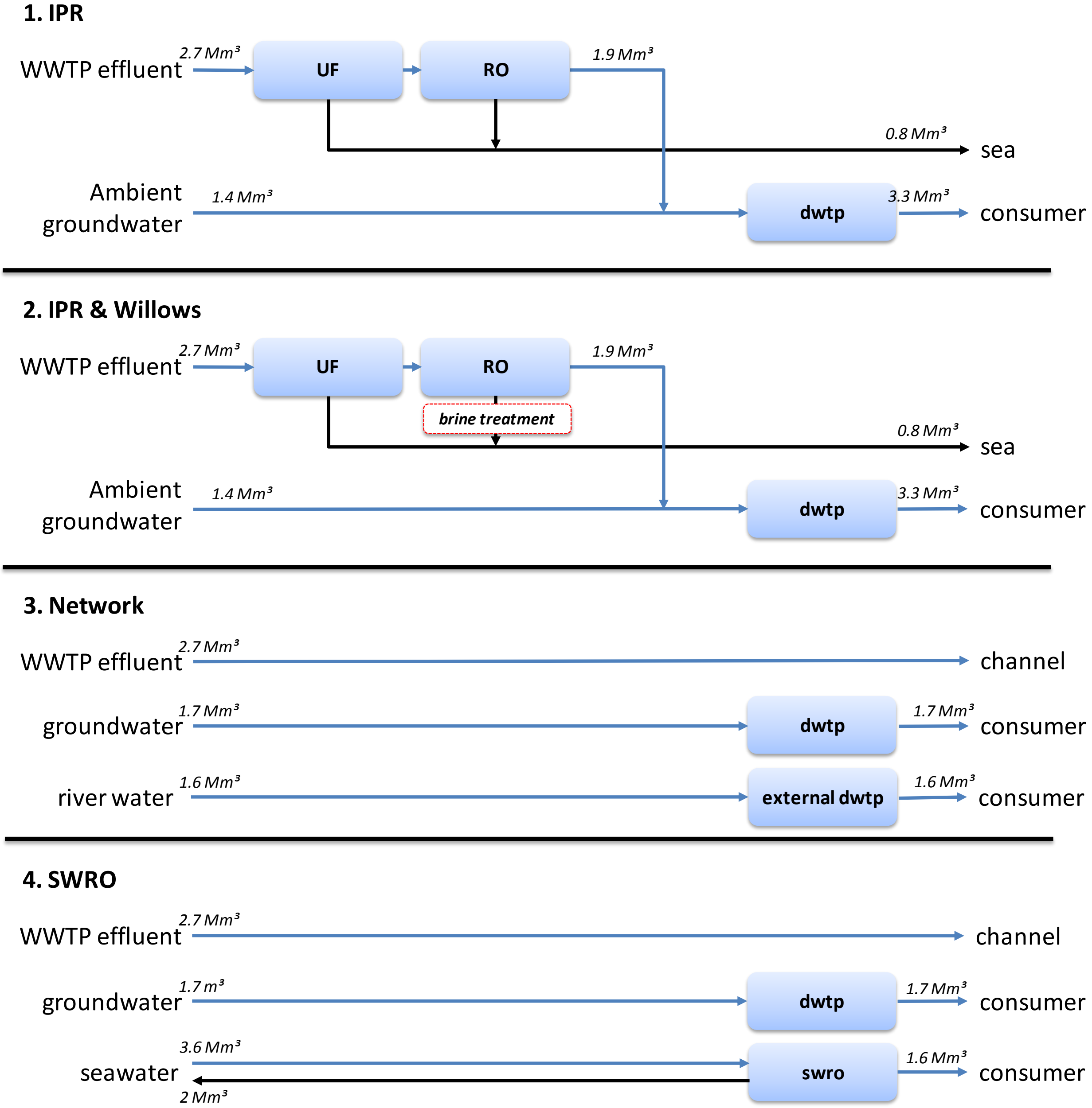


Figure 7-3: Comparative overview of the LCA scenarios for Torreele and annual water volumes
dwtp: drinking water treatment plant; swro: seawater reverse osmosis; UF: ultrafiltration; RO: reverse osmosis

Allocation

Co-products from the system include the biomass from the willow field in scenario 'IPR & Willows'. This biomass is harvested and used in a biogas plant for energy production, valorising the biogas in a CHP unit to produce electricity and heat. These co-products are fully allocated to the reuse system.

Data quality and limitations of this study

In the following, input parameters for the LCA inventory are discussed regarding data quality and existing uncertainties to clearly point out inherent limitations of this LCA. A summary of data sources and data quality is provided in Table 7-1.

- Water quality and quantities: The database on water quality and quantities was provided by the operator IWVA [48], taking mean operational data of 3 years (2013-15). Since this data is quality controlled by the operator, data quality is assumed to be very good. Water quality parameters are limited to organic sum parameters (COD, DOC) and nutrients (TN and N species, TP and PO_4-P), as no data was collected for heavy metals[12] or organic micropollutants[13]. Hence, results of toxicity indicators should be interpreted with care, as metals are known to have a large influence on these impact categories. Water quality data and treatment efficiency for the willow fields were taken from D 1.2 [50] which were tested within DEMOWARE in 2014-15.
- Energy, chemicals and material consumption: A detailed inventory on electricity and chemical consumption on tertiary treatment was provided by the operator IWVA [48] based on mean operational data of 3 years (2013-15), resulting in high quality of primary input data. Recovery rates of the membranes as well as electricity and chemical consumption of the reuse train have been optimized over the last 10 years. Data on electricity consumption for the recovery well and drinking water treatment plant in Koksijde were also provided by IWVA from operational data [48] with high quality. Biomass harvested from the willow field is estimated by IWVA [50], while valorisation of biomass is based on estimates of KWB from previous studies [66], leading to medium-good data quality for this process.

  For the alternative scenarios of water supply, primary data of other studies has been used. Surface water treatment in northern Wallonia ('Network') is estimated based on KWB data for ozonation, sand filtration, GAC filtration, and chlorination, which may not fully represent the actual situation and thus is of medium quality. The electricity consumption for pumping drinking water from northern Wallonia to Koksijde was estimated by KWB based on information provided by IWVA [48]. For seawater desalination, process data is based on literature [47] and feasibility studies for a SWRO system for the Vendee (FR) case study (cf. D 6.5).

  Material for infrastructure for all scenarios was roughly estimated by KWB based on previous studies [49] with medium data quality, whereas material for the pipeline from northern Wallonia to Koksijde was supplied via contacts of IWVA [48].

[12] According to Flemish legislation (VLAREM) IWVA must yearly perform a 5-day measurement campaign on its dioscharge water. The following metals have to be measured: As, Ag, Cr, Zn, Cu, Cd, Pb, Hg, Ni. In the campaign of June 2016, except for As and Zn, alll measurements were below detection limit. The average value for As and Zn was respectively 10 and 69 µg/l, thus very low.

[13] The presence of trace organics was investigated within the RECLAIM WATER project

Table 7-1: overview on data quality of input data for LCA Torreele

Parameter/Process	Data source	Data quality
Reuse schemes		
Water quality (only standard parameters)	[48], local operator	very good
Electricity consumption of membrane scheme	[48], local operator	very good
Chemical consumption of membrane scheme	[48], local operator	very good
Energy recovery from willows	[66], estimations	medium-good
Electricity consumption of DWTP Koksijde	[48], local operator	good
Infrastructural efforts	[49], estimations	medium
Scenarios for additional water supply		
Energy & chemical consumption external DWTP N. Wallonia	KWB estimations	low-medium
Electricity consumption water pumping (pipeline)	[48], KWB estimations	low-medium
Energy & chemical consumption SWRO	[47],literature, estimations	low-medium
Infrastructural efforts	[49], estimations	medium
Infrastructure for pipelines	[48], local operator	very good
Background		
Electricity mix	Mix of Belgium 2010	medium
Chemicals and materials	EU or global datasets	medium
Transport	Truck transport (EU)	good

Normalization

Normalisation reveals the contribution of the system under study towards the total environmental footprint of each citizen. Principles for normalization and normalization factors are shown in Annex 9.1.1.

7.2.2 Inventory (Input data)

Primary data

Inventory data for the LCA study was provided by local operator IWVA and complemented with estimates of KWB based on previous studies (Table 7-1). For consumptives, Table 7-2 summarizes the electricity demand and Table 7-3 summarizes chemical demand for all scenarios. Material demand for infrastructure is listed in detail in Annex 9.7.1. For the infrastructure of the pipeline from northern Wallonia, a fraction of 10 % is accounted for scenario 'Network', since the pipeline is not exclusively built for Koksijde and hence has a higher capacity of water transport than is required here (pipe diameter is DN 700-1000).

In the scenarios 'IPR' and 'IPR & Willows' tertiary treatment in the UF-RO system is the main driver regarding energy and chemical consumption. The electricity consumption is 0.12 kWh/m³ filtrate for UF (89% recovery) and 0.58 kWh/m³ permeate for RO (77% recovery) [48]. UF backwash water and RO brine are directly discharged into the cannel. Chemical consumption for disinfection prior to membranes (NaOCl and NH_4Cl), pH adjustment (caustic and acid) or cleaning in place (citric acid) is listed in Table 7-3. The willow field in scenario 'IPR & Willows' removes 10% of COD, 30% of TN, and 30% of TP from the RO brine prior to discharge into the cannel [50]. Biomass yield of the willow field is estimated with 20 t dry biomass/ha*a, which is valorised in a biogas plant with electricity and heat production in a CHP unit. Assuming a COD content of 46% of the dry matter and 50% degradation in the biogas plant, specific biogas yield can be calculated for the biomass of the willows. Finally, electricity and heat credits can be calculated for biogas valorisation with net efficiencies of electricity and heat production of 38% for the CHP unit (Table 7-2).

Table 7-2: Inventory data for energy demand (summarized in categories)
different volumes per categories and scenario as in Figure 7-3, primary data mainly provided by IVWA [47, 48, 66]

	Unit	1. IPR	2. IPR & Willows	3. Network	4. SWRO
Tertiary treatment, total	**kWh/a**	**1 366 937**	**1 344 034**	**-**	**-**
Ultrafiltration (UF)	kWh/m³ filtrate	0.12	0.12	-	-
Reverse Osmosis (RO)	kWh/m³ permeate	0.58	0.58	-	-
Electricity credits (valorisation of willows in biogas plant)	kWh/kg COD in willows	-	- 0.70	-	-
Heat credits (valorisation of willows in biogas plant)	MJ/kg COD in willows	-	- 2.50	-	-
Drinking water treatment/ pumping, total	**kWh/a**	**365 720**	**365 720**	**1 882 398**	**6 685 936**
Pumping and treatment of groundwater (Koksijde)	kWh/m³ groundwater	0.11	0.11	0.11	0.11
Treatment of surface water (Northern Wallonia)	kWh/m³ water	-	-	0.35	-
Pumping via water network	kWh/m³ water	-	-	0.74	-
Treatment via seawater reverse osmosis	kWh/m³ product water	-	-	-	4.00
Overall electricity demand, total	**kWh/a**	**1 732 657**	**1 709 754**	**1 882 398**	**6 685 936**

Treatment of imported water from northern Wallonia includes sand filtration, ozonation (3 ppm O_3), GAC filtration (runtime of filters: 55 000 bed volume), and disinfection (2 ppm Cl as NaOCl). Electricity consumption of this DWTP is estimated to 0.35 kWh/m³, including filtration (0.1 kWh/m³), ozonation (0.05 kWh/m³; ozone production from liquid oxygen, ozone input and cooling are estimated with 15 kWh/kg O_3) and other additional efforts of the DWTP (0.2 kWh/m³). The lignite-based GAC (23 t in the entire filter) is regenerated in an interval of 2-2.5 years with 10% make-up of virgin coal. The expected lifetime of virgin GAC including regeneration is 30 years.

The scenario 'SWRO' includes a pre-treatment of seawater with pre-chlorination, coagulation with $FeCl_3$, flocculation and ultrafiltration. After RO membrane, sulfuric acid and lime are dosed for remineralisation of the desalinated product water. The detailed chemical consumption is shown inTable 7-3.

Table 7-3: Inventory data for materials demand related on different volumes and aggregates

all concentrations per feed volume and chemicals in concentrations with water; UF = wastewater UF; RO = wastewater RO; SFW = surface water treatment in northern Wallonia; SW = seawater reverse osmosis per UF feed volume [47, 48]

Chemical	Unit	1. IPR	2. IPR & Willows	3. Network	4. SWRO
NaOCl (15 %)	mg/L	40.3 (UF)	40.3 (UF)	29.7 (SFW)	28.2 (SW)
NaOH (29 %)	mg/L	4.5 (RO)	4.5 (RO)	-	1.80 (SW)
Citric acid (40 %)	mg/L	0.53 (UF) 0.90 (RO)	0.53 (UF) 0.90 (RO)	-	0.80 (SW)
Antiscalant	mg/L	2.46 (RO)	2.46 (RO)	-	[14]
H_2SO_4 (32 %)	mg/L	52.0 (RO)	52.0 (RO)	-	61.0 (SW)
$NaHSO_3$ (39 %)	mg/L	1.48 (RO)	1.48 (RO)	-	-
NH_4Cl (50 %)	mg/L	4.00 (RO)	4.00 (RO)	-	-
Fresh GAC	kg/a	-	-	759 (SFW)	-
Regenerated GAC	kg/a	-	-	10 124 (SFW)	-
Ozone	mg/L	-	-	3.00 (SFW)	-
Lime (92 %)	mg/L	-	-	-	123 (SW)
HCl (32 %)	mg/L	-	-	-	9.50 (SW)

Water Inventory

Table 7-4 shows the water volumes and qualities for the WWTP effluent and the related effluents.

Table 7-4: Water inventory for all scenarios including WWTP influent and effluent

Measured data by [48, 50]

Parameter	Unit	WWTP effluent	1./2. Infiltrated water	Potable water	1. UF concentrate + RO brine w/o willow treatment	2. UF concentrate + RO brine with willow treatment
Volume	m^3/a	2 730 984	1 881 370	variable	849 614	849 614
COD	mg/L	-	-	-	109.9	103.3
BOD	mg/L	3.7	-	-	10.6	10.6
TOC	mg/L	-	1.0	2.5	29.1	29.1
TN	mg/L	8.4	4.6	1.6	29.7	15.2
TP	mg/L	1.2	0.1	0.1	3.8	2.9

Background data

Information on datasets for background processes is shown in Annex 9.7.1.

Inventory for Water Impact Index

The Water Impact Index (WIIX) is calculated according to the methodology described in *D3.1* [11]. The water scarcity index (WSI) according to WULCA AWARE [52] is used for calculation, taking an annual WSI

[14] Antiscalant for SWRO assumed to be Citric Acid

of 4.40 for withdrawal and release in the Koksijde area and 1.22 for surface water withdrawn in northern Wallonia. Monthly differentiation of water withdrawals or releases was not within the scope of this study.

Water withdrawals and releases in all scenarios are shown in Table 7-5 (see also Chapter 7.2.1). The water reuse scheme in Torreele recovers 1.88 Mio m³ water per year, which is infiltrated in the dunes. Assuming an evaporation rate of 5 % during infiltration, the annual accounted release volume into groundwater is 1.79 Mio m³. The withdrawal from groundwater at DWTP Koksijde is 3.32 Mio m³/a, drawing another 1.53 Mio m³/a of ambient groundwater on top of the reclaimed water. The UF backwash water and RO brine (sum of 0.85 Mio m³/a) is discharged to the canal and fully accounted as release in the WIIX.

For the alternative scenarios, secondary effluent of WWTP Wulpen is fully discharged to the cannel (2.7 Mio m³/a). Local drinking water treatment extracts 1.7 Mio m³/a from groundwater resources, while the remainder is either withdrawn in northern Wallonia (1.62 Mio m³/a) for water import or supplemented from seawater, which is not accounted in WIIX.

Water quality index (WQI) of RO brine varies between scenarios '1 IPR' and '2 IPR & Willows' due to higher quality of brine after willow treatment (lower P concentration), although the effect is not very strong (Table 7-5). Details for WQI calculation can be found in Table 9-40 in the annex.

Natural groundwater and reclaimed water have both optimum quality (WQI = 1) in the WIIX calculation. Optimum quality is also assumed for surface water withdrawn in the import scenario, although water quality data for this resource was not available. For secondary effluent of WWTP, a WQI of 0.11 is calculated based on the P concentration. However, the total P load to the canal is comparable between water reuse and direct discharge of secondary effluent, as the tertiary treatment (RO) removes the entire P load and transfers it to the brine, which ends also in the canal. It is debatable if a higher concentrated P discharge (RO brine) or a more diluted P discharge (WWTP effluent) is more preferable to prevent eutrophication in the downstream river, but the WQI method acounts higher quality factors to the more diluted stream, even if the total P load is comparable.

Table 7-5: Overview on direct withdrawals and releases and water quality indices (WQI) for the different scenarios

INF = infiltration; EF = effluent to the cannel (WWTP effluent or UF backwash water and RO brine); GW = Withdrawal of Groundwater in Koksijde; RW = Withdrawal of River Water in northern Wallonia

Scenario	1. IPR	2. IPR & Willows	3. Network	4. SWRO
Withdrawals [m³/a]	**3 324 734 (GW)**	**3 324 734 (GW)**	**1 700 000 (GW)** **1 624 734 (RW)**	**1 700 000 (GW)**
WQI (Withdrawals)	1	1	1	1
Releases [m³/a]	**1 787 302 (INF)** **849 614 (EF)**	**1 787 302 (INF)** **849 614 (EF)**	**2 730 984 (EF)**	**2 730 984 (EF)**
WQI (Releases)	1 (INF) 0.05 (EF)	1 (INF) 0.07 (EF)	0.11 (EF)	0.11 (EF)

7.2.3 Impact Assessment (Results)

Environmental impacts were assessed with a set of 8 impact categories (including water impact index), representing different areas of environmental concern. After an overview of all indicators for all scenarios, selected impact categories are discussed more in detail to reveal relative changes due to change in water supply or brine treatment compared to the existing reuse scheme.

Total environmental impacts and benefits of all scenarios

The environmental profile of all scenarios for all selected impact categories is shown relatively to the gross impact of the scenario ‘1 IPR’ (= 100 %) in Figure 7-4. The fossil and nuclear cumulative energy demand (CED), the global warming potential (GWP) and terrestrial acidification potential (TAP) are strongly influenced by the background processes, such as electricity, chemicals or material production. For all scenarios, electricity consumption for water treatment and transport is dominant in CED and GWP. Water reuse and water import are comparable in CED and GWP, whereas seawater RO increases energy-related impacts by around 400% due to the high demand of electricity. Electricity consumption in water reuse and SWRO originates from UF/RO stage, whereas electricity demand in the import scenario is mainly caused by water transport in the pipeline (120 km). Chemical production for water treatment is more relevant for indicators GWP and TAP, since the Belgium power mix 2010 consists of less fossil fuels and more nuclear-based power, giving electricity demand a lower score in GWP and TAP. Infrastructure plays only a minor role in CED, GWP and TAP indicators for all scenarios.

The innovative brine treatment in scenario ‘IPR & Willows’ leads to less impacts in freshwater and marine eutrophication, by reducing P and N discharge into the canal. CED and GWP are only marginally influenced by the additional treatment. The eutrophication indicators are discussed more in detail below.

The absolute scores for toxicity indicators are comparably low, since only background processes have been considered. As described in Chapter 7.2.1 heavy metals where not considered in the inventory, so no potentially toxic water emissions have been assessed for the different water reuse/water supply schemes, neglecting any direct effect of the schemes. With these assumptions, infrastructure has some contribution to toxicity indicators due to cast iron production for the pipeline in ‘3 Network’ and sludge disposal from the SWRO-system. However, the uncertainty of toxicity indicators is relatively high and should lead to a careful interpretation of these results.

The WIIX is mainly influenced by the volumes of water withdrawals, and also by the different water quality of water releases and also by energy intensive background processes in case of seawater desalination. In total, water reuse leads to a lower WIIX compared to both water import and seawater desalination. The high water withdrawal in Koksijde from local groundwater resources in the reuse scenarios is partially offset by infiltrating reclaimed water with comparable quality. The direct discharge of this water as WWTP effluent into the cannel in scenarios ‘3 Network’ and ‘4 SWRO’ only leads to lower credits due to its lower quality reflected in the WQI. Effects of water scarcity are reflected in the different WIIX of water extraction in Koksijde and northern Wallonia: although comparable volumes of water are withdrawn from the environment, the WIIX for water withdrawal is significantly lower in Wallonia due to the lower scarcity. The relative changes in the WIIX are discussed in detail below.

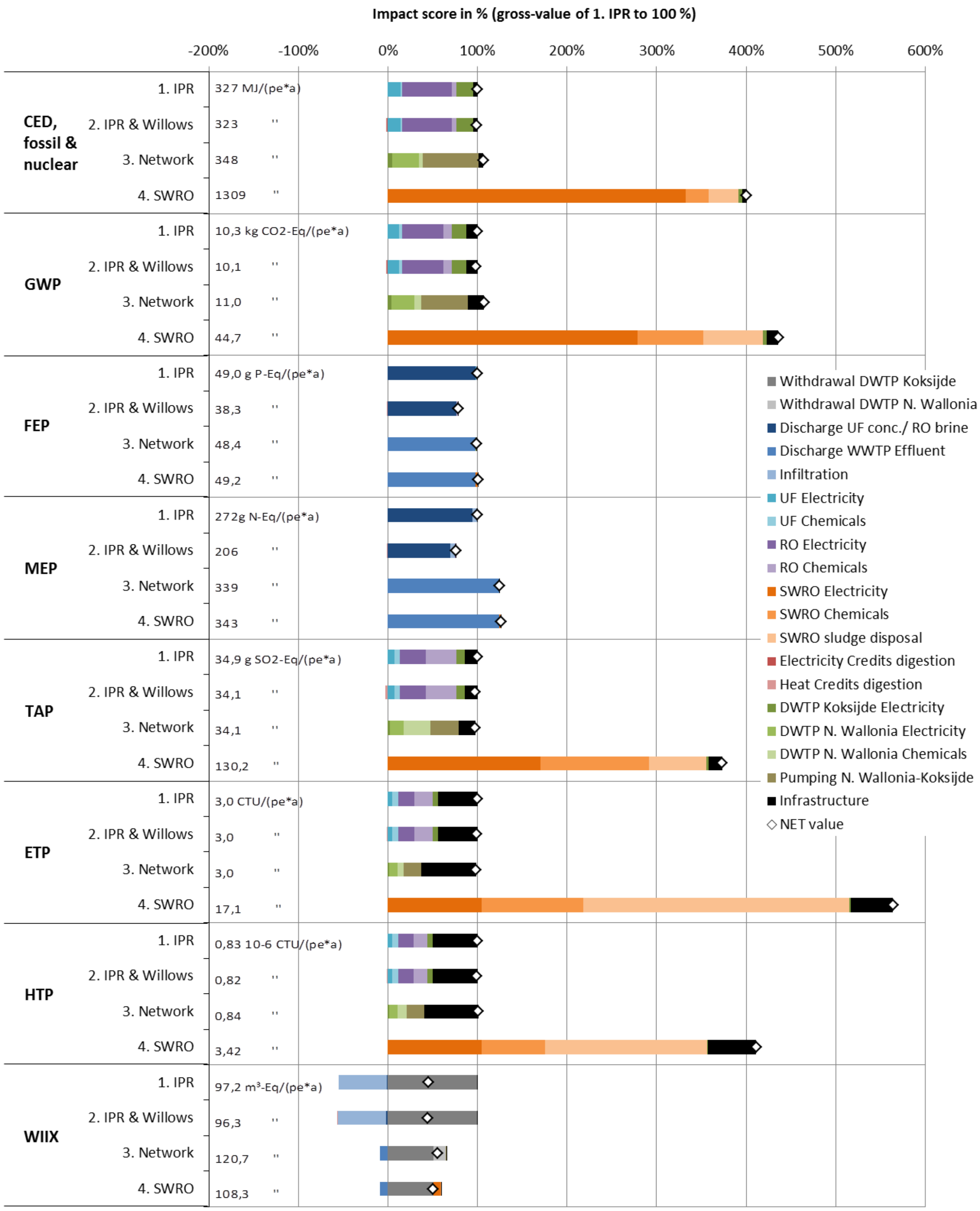


Figure 7-4: Environmental profile for all scenarios related to gross-value of '1 IPR' (= 100 %) and total net values per scenario and impact category

CED = cumulative energy demand; GWP = global warming potential; FEP = freshwater eutrophication potential; MEP = marine eutrophication potential; TAP = terrestrial acidification potential, ETP = eco toxicity potential; HTP = human toxicity potential; WIIX = water impact index

Relative changes for selected impact categories

The relative changes of environmental impacts between different scenarios and the existing situation ('1 IPR') are discussed below. Thus, the specific effects of implementing willow treatment in full-scale or of switching to another water supply option on a switch to external water supply can be shown for the impcat categories of freshwater eutrophication potential (FEP), marine eutrophication potential (MEP) and the WIIX.

The changes in FEP reflect a change in P emissions into the aquatic environment (Figure 7-5). The brine treatment with willows reduces total P emissions of the reuse scheme (49.0 g P-Eq/ (pe*a) or 3.92 g P-Eq/ m³ discharged water) into the cannel by 22%. For the alternative supply options ('3 Network' and '4 SWRO'), no net effect on FEP can be detected, as the total P load in WWTP effluent is now directly discharged to the cannel. This exemplifies the need for a brine treatment to reduce the P emissions into the cannel, because water reuse does not reduce P emissions of the WWTP process if UF/RO membrane concentrates are then directly discharged into the freshwater environment.

The changes in MEP reflect a change in N emissions into the environment (Figure 7-6). In general, similar effects can be detected for MEP and FEP. Brine treatment in willows can reduce total MEP of the reuse scheme (272 g N-Eq/(pe*a) or 22 g N/m³ discharged water) by -24%. Due to incomplete N retention in the UF/RO reuse train, reuse scenarios have less direct N emissions than alternative water supply options with direct discharge of WWTP effluent. Remaining N in the reclaimed water is infiltrated into the local groundwater, where a smaller fraction is accounted for MEP due to potential transfer into marine environments.

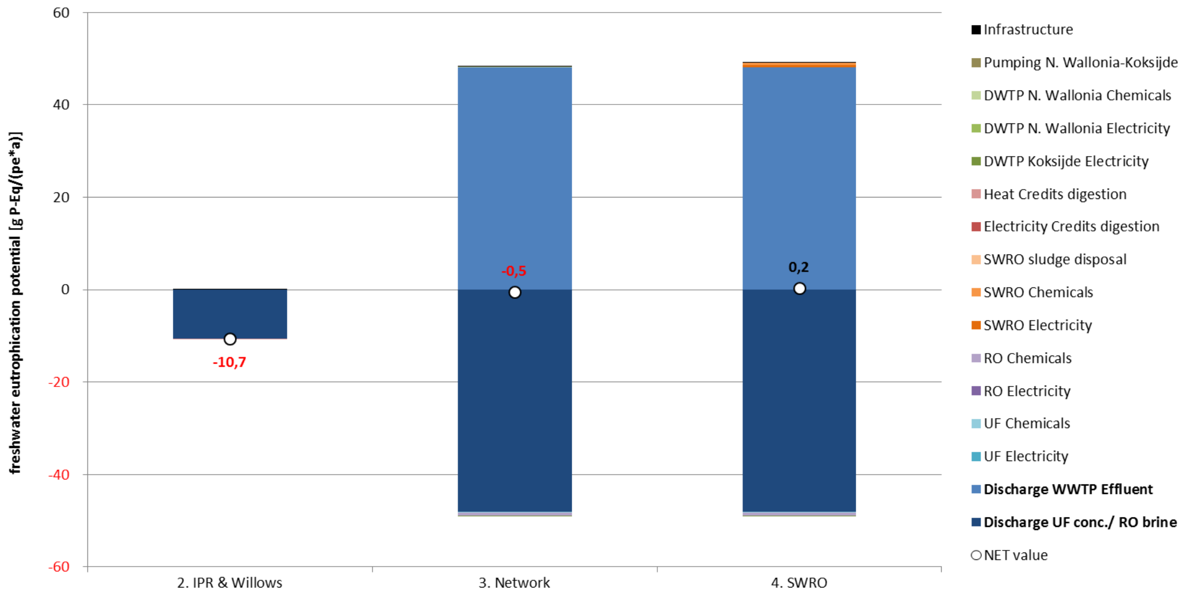


Figure 7-5: Changes in freshwater eutrophication potential of different scenarios compared to '1 IPR'
direct effects bold in the legend

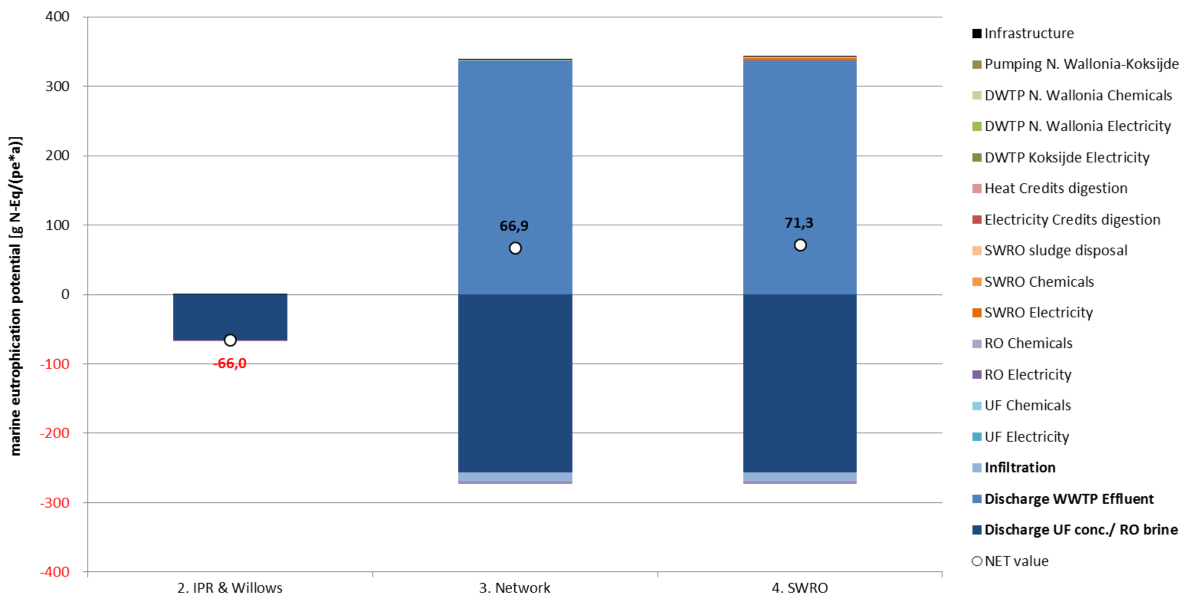


Figure 7-6: Changes in marine eutrophication potential of different scenarios compared to '1 IPR'
direct effects bold in the legend

Overall, LCA assesses FEP and MEP by accounting total loads of P and N which are discharged into the environment. The actual P and N concentration within the discharged water is not accounted, although eutrophication is known to occur when certain tresholds in receiving waters are exceeded. Hence, it should be noted that the local risk for eutrophication may well be higher for more concentrated discharge (e.g. brine) than for more diluted discharge (e.g. WWTP effluent), although the total nutrient loads are comparable or smaller in water reuse.

The changes in WIIX are shown in Figure 7-7 in relation to a total WIIX of 97.2 m^3-Eq/ (pe*a) for the reference scenario '1 IPR'. The willow treatment has only marginal effects on the WIIX by slightly improving the quality of brine release, which is still accounted with a low quality index.

Switching to alternative sources for water supply will increase the WIIX of the current system. Local recharge of groundwater with reclaimed water will be reduced, but this is mostly offset by the reduced extraction of local groundwater for drinking water production. Only evaporated water in infiltration ponds (5%) is "lost" in the reuse scheme. In comparison, net WIIX of the import scenario '3 Network' is higher than in the reuse scheme, as water sources in Wallonia are exploited which have a higher WIIX than the WWTP effluent now discharged into the environment. However, these resources have a relatively low scarcity (WSI) compared to Koksijde groundwater extraction, leading to a relatively low net WIIX per m^3. For seawater desalination, net WIIX of water supply is determined by WWTP effluent discharge and indirect WIIX of electricity production, resulting in an overall lower net WIIX than water import.

From the WIIX analysis, it can be concluded that water reuse by IPR of WWTP effluent has a lower water footprint than both water import and seawater desalination. While the WIIX of water import is determined by the scarcity and quality of the related resource, WIIX of seawater desalination is determined by indirect water uses for electricity production. Both cases lead to the direct discharge of WWTP effluent into the environment, but this gives only low credits due to the low quality of this water.

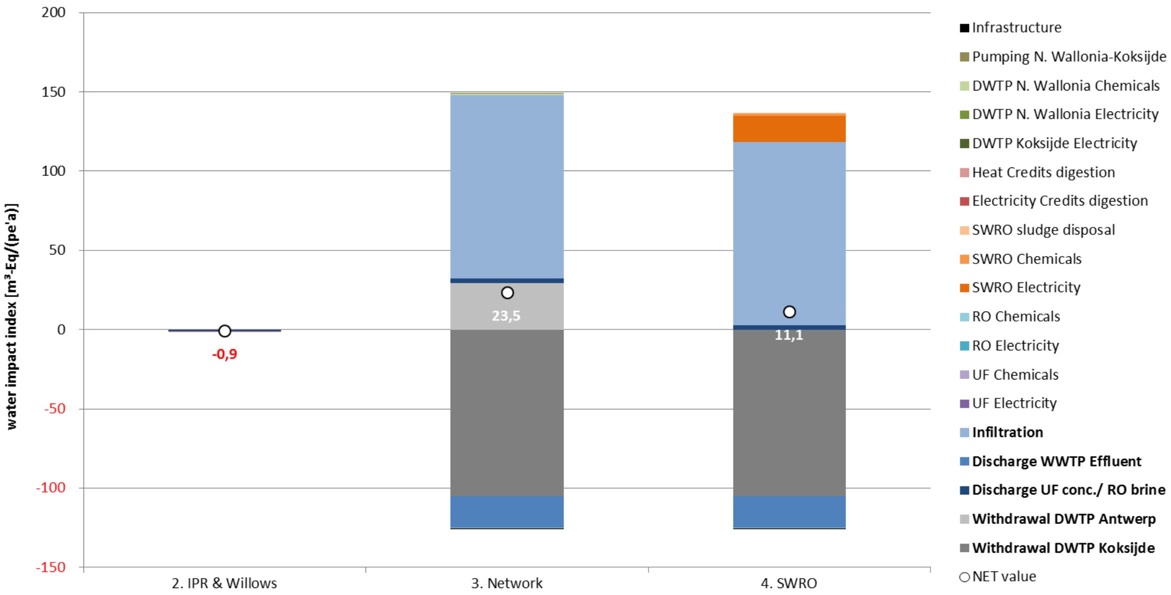


Figure 7-7: Changes in the water impact index of different scenarios compared to '1 IPR'
direct WIIX bold in the legend

Detailed contribution analysis for the UF/RO reuse scheme regarding its environmental impacts

Referring to the optimized water reuse scheme in Torreele, a detailed investigation of associated efforts for operation of the membrane scheme is conducted below for indicators CED, GWP and TAP. Figure 7-8 shows the relative share of electricity and chemicals for the total scores of CED, GWP and TAP. The gross energy demand of the membrane scheme is dominated by electricity consumption (91 %). The production of chemicals incl. transport is estimated with only 9 %. Due to the comparably low share of fossil fuels in the electricity mix of Belgium (in 2010), GWP and TAP are more influenced by production and delivery of chemicals (45 % share of gross GWP and 53 % share of gross TAP). Furthermore, Figure 7-8 shows the contribution of the singular chemicals used in the membrane scheme and their share to the gross contribution of chemicals in the selected impact category.

Based on this detailed analysis, 37-40% of the total gross chemical impact in CED, GWP and TAP is caused only by the production of antiscalant. It should be noted that exact chemical composition of the antiscalant for wastewater RO will not be disclosed by the supplier, so antiscalant was estimated to be based on phosphonates [124], assuming a mixture of dichloromethane and organophosphorus compounds in this LCA. However, the contribution analysis revealed a high impact of antiscalant compared to other chemicals, although the dose is relatively low (2.5 ppm) compared to other chemicals such as NaOCl (40 ppm) and H_2SO_4 (52 ppm)(see Table 7-3). Therefore, the environmental effort for production of antiscalant may be overestimated in this LCA.

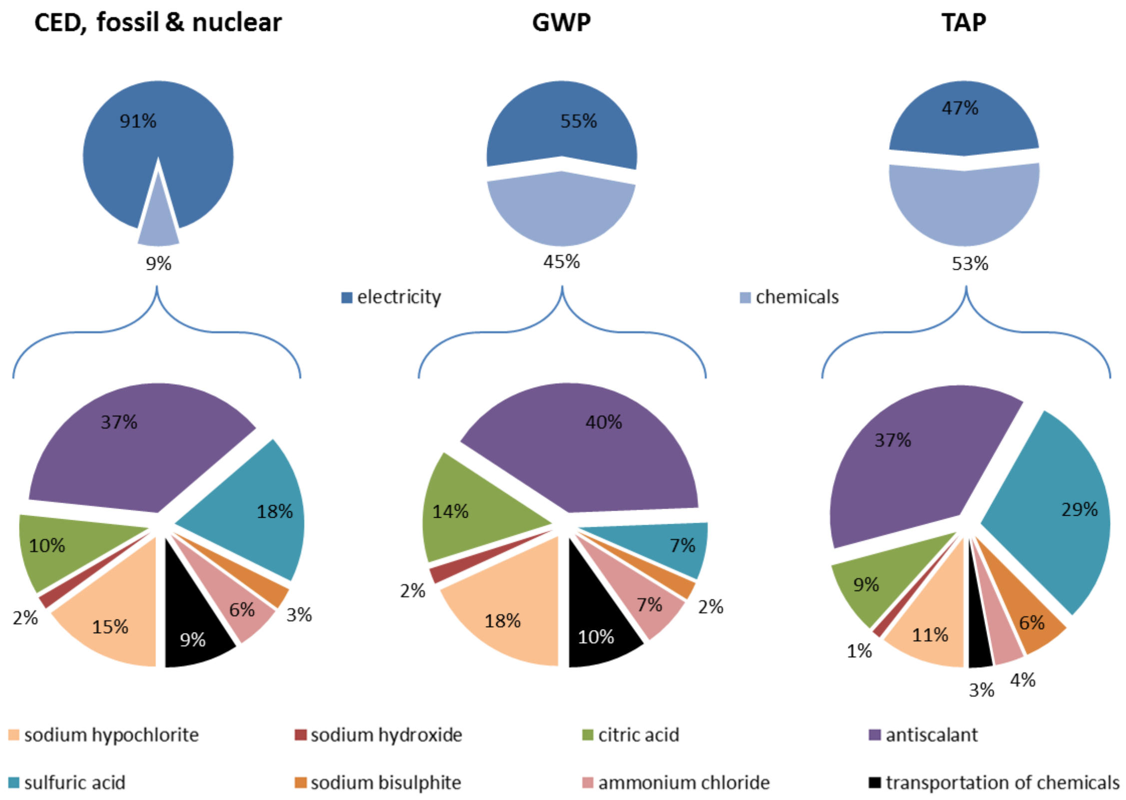


Figure 7-8: Relative composition of environmental efforts in CED, GWP and TAP for the UF/RO-system in Torreele
top: share of electricity and chemicals per impact category; bottom: share of different chemicals used in the membrane scheme per impact category

Normalization

The score for each impact category per pe (Figure 7-4) related to the normalization data (Table 9-1) per EU-27 citizen is shown in Figure 7-9.

CED, GWP, TAP, ETP and HTP contribute approximately 1 % to the gross impact per citizen in the EU-27, which is a realistic magnitude of CED, GWP and TAP for water treatment processes. Referring to the toxicity indicators it should be noted that direct emissions (e.g. heavy metals) have not been considered in this LCA for all scenarios, leading to a potential underestimation of the impacts in these categories.

The high normalized score for FEP (9-12 %) and MEP (2-3 %) are more significant, and are caused by the WWTP effluent discharge or the UF/RO concentrates of the reuse scheme. Again, the need for concentrate treatment becomes obvious, and the willow treatment reduces these impacts.

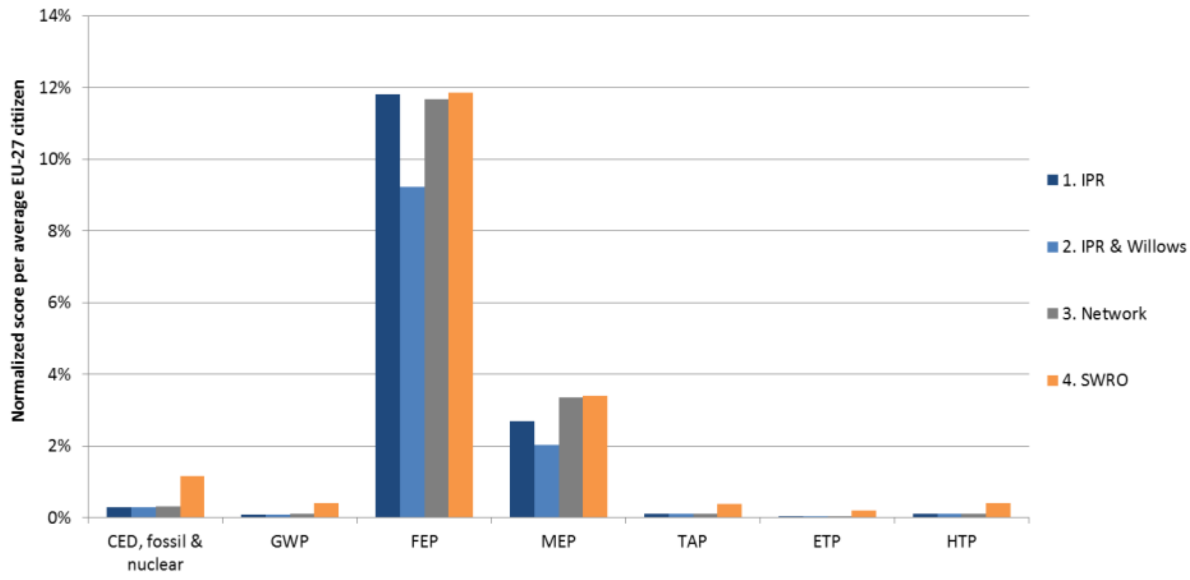


Figure 7-9: Normalized scores for all impact categories per average EU-27 citizen

7.2.4 Interpretation and Discussion

Summary and Interpretation of results

Table 7-6 gives a summary on the net environmental efforts and benefits of the scenarios for all impact categories, refering to the existing reuse scheme '1 IPR'.

Table 7-6: Summary of net environmental efforts and benefits of the scenarios for all impact categories, related to the scenario '1 IPR' as reference

Scenario	1. IPR	2. IPR & Willows	3. Network	4. SWRO
CED	327 MJ/ (pe*a)	- 1 %	+ 6 %	+ 301 %
GWP	10.3 kg CO_2-Eq/ (pe*a)	- 2 %	+ 7 %	+ 336 %
FEP	49.0 g P-Eq/ (pe*a)	- 22 %	- 1 %	± 0 %
MEP	271 g N-Eq/ (pe*a)	- 24 %	+ 25 %	+ 26 %
TAP	34.9 g SO_2-Eq/ (pe*a)	- 2 %	- 2 %	+ 273 %
ETP	3.03 CTUe/ (pe*a)	- 1 %	- 1 %	+ 464 %
HTP	$0.83\ 10^{-6}$ CTUh/ (pe*a)	- 1 %	+ 1 %	+ 311 %
WIIX	97.2 m^3-Eq/ (pe*a)	- 1 %	+ 24 %	+ 11 %

Overall, the water reuse scheme in Torreele is environmentally beneficial to both alternatives of water supply. It is comparable in energy demand and related emissions with water import from Wallonia, while seawater desalination would increase energy demand by a factor of 4. By avoiding direct discharge of WWTP effluent into the cannel, water reuse also reduces nitrogen emissions into the marine environment. In addition, water reuse can reduce the water footprint of water supply in the region, as both water import and seawater desalination have higher water footprints in their life cycle.

Brine treatment in the willow field can be recommended from an environmental point of view, as it can significantly reduce nutrient emissions into the canal by 22-24%. The system adds only a low energy demand to the overall scheme. However, biomass yield and energetic valorisation of the willows via anaerobic digestion does not yield a significant energy credit for this scenario either. This option should be further pursued in the future to reduce impacts of the water treatment and supply system on the local environment. It has to be noted that toxicity-related impacts were not evaluated in this LCA, which is a clear limitation of the study.

7.3 Conclusions

The following conclusions can be summarized from the LCA study of the Torreele water reuse scheme:

1. **Water reclamation in Torreele for indirect potable reuse reduces water stress for local freshwater resources and also requires less energy and related greenhouse gas emissions than other available options for water supply** (e.g. importing water via pipeline network, seawater desalination). Compared to seawater desalination, water reuse can reduce energy demand and associated emissions of water supply by > 75%.
2. The **brine treatment with willows removes nutrients from the brine and thus decreases eutrophication potential by 25%**. In addition, **willow treatment has no additional net energy demand**, if the harvested willows are used for biogas production in a digestor to produce electricity and heat.

The future challenge for the innovative brine treatment is seen in upscaling this demonstrated technology to full-scale and improving the removal rate of phosphorus and nitrogen by optimized operation or increased capacity.

8 Conclusions and summary

The different case studies analysed within this report have shown that environmental impacts and benefits of water reuse should be assessed on a site-specific basis together with a transparent and comprehensive risk assessment of the reuse system. While the latter is required to keep the risks of water reclamation for humans and ecosystems within tolerable limits, the environmental assessment can help to illustrate the benefits of water reuse for the local environment, but also to quantify the additional efforts needed (e.g. electricity, chemicals, infrastructure) and compare them to other alternatives of water supply. Here, available local alternatives for water supply, different reuse purposes, but also other locally specific boundary conditions can all influence the outcomes of the comparison and the relative ranking of water reuse as an environmentally preferable alternative.

However, certain conclusions can also be drawn from the assessment results of the case studies on a more general level. These conclusions relate to different aspects of the main outcomes and also the methods applied in this study:

- General conclusions from the environmental and risk assessment of water reuse schemes
- Inherent trade-off between environmental efforts (e.g. energy demand) and benefits (e.g. reduced water stress) of water reuse
- Method discussion of the Water Impact Index as an indicator for water footprinting of reuse systems
- Critical discussion of toxicity-related impact categories of LCA
- Combination of results from LCA and risk assessment towards the endpoint human health to illustrate trade-offs in risk reduction

8.1 General conclusions for environmental benefits (and risk reduction) in water reuse schemes

Summarizing the main outcomes of the different case studies analysed in this report, some more general conclusions can be drawn for the assessment of water reuse:

- **Reuse of water reduces local water stress**, and local water scarcity is the main driver for water reuse. Using reclaimed water from WWTP effluent can also reduce nutrient loads to receiving surface waters, while metals and other trace contaminants in reclaimed water pose a potential risk compared to the use of natural water sources.
- On the other hand, **additional treatment of reclaimed water requires investing energy and associated environmental impacts** to reduce risk for humans and the environment to a tolerable level.
- **In general, additional environmental impacts from water treatment depend on the use category of reclaimed water.** The higher the target quality of reclaimed water, the higher the additional environmental impact from energy and chemical demand. Hence, the types of reclaimed water use and potential local alternatives of water supply have to be taken into account when assessing the overall sustainability of water reuse schemes in comparison to other options.
- In principle, **available treatment technologies are capable of producing any predefined water quality in reuse schemes.** However, higher risk reduction is often associated with higher environmental impacts of the reuse system, showing an **inherent trade-off between invested resources and accepted residual risk**.
- Therefore, a **common understanding of an accepted level of residual risk is a prerequisite for a wide application of water reuse in Europe**. This acceptable level of risk and required risk reduction has to be defined by the regulators. Currently, a **lack of clear risk targets is one of the major barriers for water reuse**, as the pre-cautionary principle ("no risk") prohibits many potential reuse applications.
- **With a clear target of risk reduction, adequate measures for risk reduction can be set** (e.g. target performance of water treatment processes). Resulting environmental impacts from water reuse schemes can then be compared to other alternatives for water supply on a site-specific basis to identify the preferred solution, also taking into account environmental benefits of water reuse (e.g. reduced water stress).
- Adequate risk management of **water reuse requires risk assessment** based on site-specific data and information.
- **Probabilistic approaches** should be used to **summarize and communicate incomplete scientific knowledge.**
- **Potentially negative impacts of trace pollutants for both reclaimed water and natural water sources are difficult to assess with LCA and WIIX** due to the high uncertainty of pollutant loads in the various water streams (e.g. heavy metals which are often below LOQ) and the method limits of WQI for WIIX and toxicity LCIA methods for LCA.

8.2 Trade-off between energy consumption and water footprint in water reuse

In principal, local scarcity of freshwater resources for human use can be overcome by several measures, e.g. importing water from areas of higher availability, desalination of seawater, or water reclamation and reuse. As water reclamation does often include the reuse of water with lower quality (e.g. secondary WWTP effluent) than the other resources, it may be straight-forward to prefer other options of water supply to minimise additional risks for humans or ecosystems originating from low-quality sources of water reuse. However, these alternatives of water reuse often come with a considerable demand of energy and infrastructure, namely in the case of seawater desalination and pipeline transport of water over long distances. In addition, water import from other areas may also increase problems of water stress there, which will only shift the problem of water scarcity between regions.

From an environmental point of view, water reuse should enable the recycling of locally available secondary water sources to decrease pressure on local freshwater resources while investing a reasonable amount of effort into adequate measures for risk reduction. This inherent trade-off between effort and benefit is characteristic for water reuse, and it should be compared with other available options for water supply to choose the most sustainable option on a case-by-case basis.

This trade-off is further illustrated in selected case studies of this report, taking the indicators of cumulative energy demand and Water Impact Index as a proxy for additional efforts and benefits, respectively. Relative impacts and benefits for water reuse are shown in Figure 8-1 in relation to the current status of water supply as a baseline.

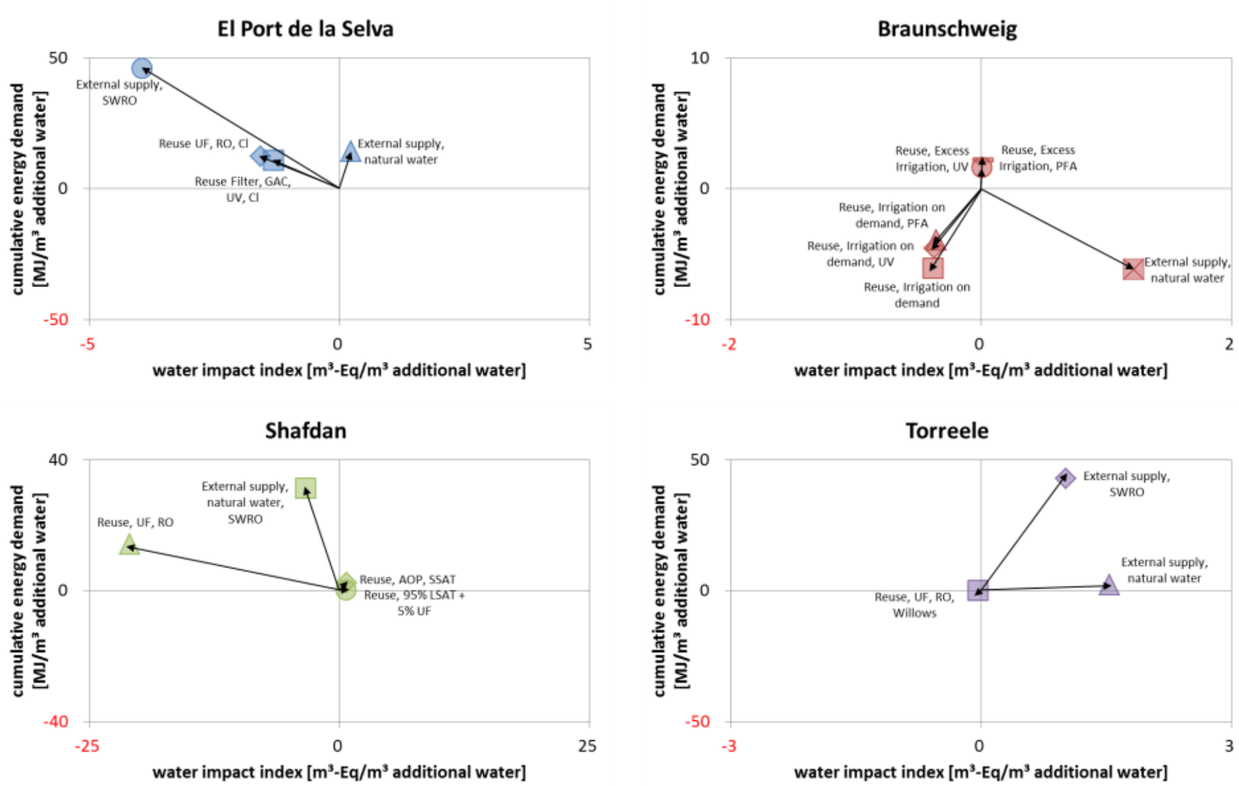


Figure 8-1: Changes in cumulative energy demand and water impact index to the scenario representing the current status (in El Port de la Selva before starting infiltration) for selected reuse sites

These four examples are discussed from a holistic view below, pointing out different targets of system change or optimisation and the related consequences from an environmental point of view:

- **"Invest in reuse scheme"** (El Port de la Selva): The reference system for this study represents the status before operation of the reuse scheme. All investigated options for additional water supply will increase energy consumption of water supply, while they also increase the quantity of available freshwater resources. However, just importing water from another area will not lead to a lower water stress in the catchment region, as water scarcity is assessed with comparable figures at the alternative site, leading to a mere shift of problems. Both water reuse and seawater desalination will significantly reduce local water stress, as they do not further exploit limited freshwater resources. However, seawater desalination is associated with a significantly higher energy demand than water reuse and thus may not be preferable from an environmental point of view. Similarly, water import also needs some energy for water treatment and transport, which is comparable to water reuse in this case. Finally, water reuse is seen as most efficient option here with moderate efforts in treatment and no additional water stress in the region. Risk assessment shows that additional risks of water reuse for human health can be controlled if adequate measures for monitoring and treatment are in place.
- **"Optimise reuse scheme for risk mitigation and efficiency"** (Braunschweig): Here, risk assessment showed that an optimisation of the existing reuse scheme may be required to reduce health risks for local workers to tolerable levels of the WHO. Naturally, the future implementation of a suitable disinfection scheme in Braunschweig is associated with a slight increase in energy consumption, but does not affect the direct water footprint as microbial parameters have not been considered in the assessment of water quality. Choosing a disinfection process with lower energy demand (= UV) will minimise the environmental impact of risk mitigation measures. Moreover, demand-oriented irrigation management can substantially improve the environmental profile of the reuse scheme, as pumping of excess water can be avoided and water discharge to the local river is increased. Further optimisation potential is illustrated by the new sludge treatment line which enables the decoupling of water and nutrient management, further decreasing the use of mineral N fertilizer in agriculture [50].
- **"Increase quality of reused water"** (Shafdan): Results show that an upgrade of the existing SAT site with AOP or 5% UF filtrate does not significantly increase the overall energy demand of the reuse system. Higher water quality of reclaimed water can be realized with a double membrane system, which is reflected with a lower water footprint of this option, but also a significant increase in energy consumption and higher water losses. Trade-offs between higher water quality and lower quantity are not visible here for the membrane system, but should be taken into account in the final discussion. Both SAT-based and RO-based reuse systems are less energy-intensive than using the potable water mix for agricultural irrigation, which is partially based on seawater desalination.
- **"Verify benefits of reuse scheme"** (Torreele): operating since 2002, this scheme of indirect potable reuse has proven that water reuse is superior in its environmental profile to other alternatives such as water import or seawater desalination. Having lower energy consumption and also a lower water footprint, the existing IPR scheme shows that water reuse is highly competitive in this setting and decreases environmental impacts of water supply considerably compared to available alternatives. However, further optimisation of the scheme by low-energy brine treatment in willow fields helps to decrease existing environmental impacts from nutrient emissions via RO brine into the local freshwater systems.

This variety of motivations and results of the different case studies illustrates that a general conclusion from the assessment is difficult to draw. Finally, a **direct transfer of LCA results and conclusions from one case study to another is not meaningful**, as site-specific conditions may affect both the absolute scores, but also the relative comparison of water reuse to other alternatives. However, few general remarks can be stated:

- Water reuse with 'high-tech' tertiary treatment (i.e. based on membranes) consume around 25% of energy compared to seawater desalination, while both approaches have high benefits in water footprint due to the high water quality delivered
- Water reuse with 'low-tech' tertiary treatment (filtration, UV, Cl) need less energy and should be preferred if a tolerable level of risk reduction can be guaranteed (e.g. in combination with natural barriers such as SAT or groundwater recharge). Benefits in water footprint are less distinct, as lower product water quality reduces credits of reclaimed water in this study.
- Seawater desalination has by far the highest energy consumption and associated environmental impacts (e.g. GHG emissions) of all alternatives and has to be seen as the "last resort" of water supply. However, water quality is superior to all other alternatives and results in the highest credits for water footprint in this study, as desalination generates "new" freshwater.
- Water import from other areas might lower water footprint locally if the import region has a lower water stress, and energy consumption is strongly dependent on both treatment and required transport (pumping) over longer distances. The environmental comparison of water import and water reuse is site-specific and should be made on a case-by-case basis.

8.3 Method feedback of applying the Water Impact Index for water reuse

Water footprinting (WF) is a collective term for methodologies assessing the impacts of human water consumption of different processes or systems. For this purpose, a multitude of different approaches and methods have been proposed in recent years. Depending on the specific method, aspects of water scarcity, water quality, withdrawal from and release to different types of water resources as well as evaporation and other water-related emissions are considered and assessed differently. Generally the WF methods can be distinguished into 'stand-alone methods' and 'methods according to the LCA approach'. The WF methods according to LCA can range from simple water inventories to more complex impact assessment methods on midpoint or endpoint level [128] and follow the requirements of LCA regarding goal and scope definitions.

Water inventories and stand-alone methods for WF use volumetric approaches describing water withdrawal, consumption, releases and losses in various categories. Impact-oriented water footprint methods also take into account aspects of water scarcity and water quality in addition to the volumes, thus illustrating the impact of human water use on the availability and the quality of water resources. While the latter are in accordance to the new ISO 14046 standard on WF [129], volumetric approaches do not fulfil the ISO criteria of WF which require a form of impact assessment.

In the present study, the WF method of Veolia as a project partner is used per definition, which is named "Water Impact Index (WIIX)" [15]. The WIIX method is described in detail in DEMOWARE Deliverable 3.1 [11]. The WIIX is an impact-orientated WF on midpoint level, considering aspects of both water scarcity and quality. It can be used as one of many impact indicators of an LCA, although the reflection of water quality aspects in WIIX may also be seen as "double counting" if other LCA indicators (e.g. eutrophication, toxicity) are already influenced by these emissions. This study applies the WIIX indicator in the field of water reuse and should also provide a method feedback and discussion of the WIIX features and their consequences for results and conclusions of WF.

In general, the WIIX can be divided into a direct and an indirect part: The direct WIIX assesses the WF directly associated with the process (i.e. withdrawal and discharge during water treatment), while the indirect WIIX quantifies the WF associated with background processes (such as production of electricity, chemicals or materials). For many industrial processes (e.g. manufacturing of products) the indirect WIIX is of major importance. However, when assessing the environmental impacts of water treatment processes, it becomes clear that direct withdrawal and release of water in the process are responsible for the major part of the WIIX, while the indirect WIIX is often negligible. Moreover, water reuse is often meant to mitigate local water stress by reducing direct use of scarce freshwater resources, also focussing on the direct consequences of water management on the local situation. Hence, the following discussion of the WIIX method is mainly based on the calculation of the direct WIIX. However, some aspects will also relate as well to the indirect WIIX, although this part is not in the focus of the present study.

Several methodological challenges have been encountered while calculating the WIIX and interpreting the respective results of the different case studies which relate to:

- Choice of function, functional unit, and system boundaries of a WIIX study
- Accounting for water withdrawal and release
- Calculation of Water Scarcity Index
- Calculation of Water Quality Index

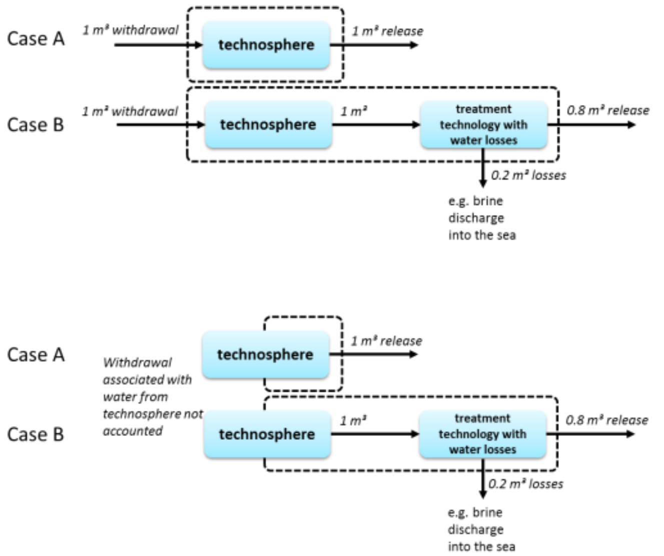
In addition, a direct comparison of WIIX to volumetric approaches or water availability footprints (i.e. not taking into account aspects of water quality) is described to discuss benefits and drawbacks of the WIIX method compared to other approaches. This part is meant to provide method feedback from the application of WIIX in the different case studies and help to improve WF methods in the future.

8.3.1 Functional unit, function and system boundaries

Addressing water losses in treatment

The choice of functional unit and adequate system boundaries is of major importance to properly address water consumption/losses of different water treatment processes in the WIIX. Figure 8-2 shows different choices of functional unit (influent or effluent volume) and system boundaries (with or without water withdrawal) and the calculated WF for two theoretical systems. For reasons of simplicity, aspects of water scarcity and quality are neglected here to focus on the volume-related effects.

If system boundaries include water withdrawal into the technosphere (e.g. drinking water production), water losses during treatment are properly reflected in the WF for both functional units, showing a higher WF for that system with water losses (Figure 8-2). However, if the system boundaries are restricted to the downstream side of the technosphere (e.g. the WWTP and potential water reclamation of secondary effluent), water losses are not properly reflected in the WIIX if product or effluent water is taken as the functional unit. This effect is caused by the "free" delivery of WWTP effluent in this perspective, resulting in a credit for WF by releasing this water into the environment again. Thus, losses during water reclamation (e.g. by evaporation in infiltration systems, or brine disposal into the ocean for membrane processes) are not accounted in the WF if effluent water is taken as a functional unit. This fact is highly relevant for those areas where water scarcity is high, and water reclamation should target the maximum use of reclaimed water without major losses.



	$withdrawal - release$	$\frac{withdrawal - release}{FU_{effluent\ water}}$	$\frac{withdrawal - release}{FU_{influent\ water}}$
Case A	$WF = 1 - 1 = 0$	$\frac{WF}{FU} = \frac{0}{1} = 0$	$\frac{WF}{FU} = \frac{0}{1} = 0$
Case B	$WF = 1 - 0.8 = 0.2$	$\frac{WF}{FU} = \frac{0.2}{0.8} = 0.25$	$\frac{WF}{FU} = \frac{0.2}{1} = 0.2$

	$withdrawal - release$	$\frac{withdrawal - release}{FU_{effluent\ water}}$	$\frac{withdrawal - release}{FU_{influent\ water}}$
Case A	$WF = 0 - 1 = -1$	$\frac{WF}{FU} = \frac{-1}{1} = -1$	$\frac{WF}{FU} = \frac{-1}{1} = -1$
Case B	$WF = 0 - 0.8 = -0.8$	$\frac{WF}{FU} = \frac{-0.8}{0.8} = -1$	$\frac{WF}{FU} = \frac{-0.8}{1} = -0.8$

Figure 8-2: water footprint related to different system boundaries and functional units for two theoretical systems A and B with and without water losses in treatment

Hence, a careful choice of system boundaries and functional unit has to be made to reflect all relevant aspects of water management and water reuse. WF results in relation to produced water may not be able to reflect water losses properly and can lead to an overlooking of these effects. If possible, the entire water management system including water production for the technosphere as well as water treatment and release into the environment should be assessed. However, case studies in the present study are all restricted to the downstream use of water after the technosphere (= city) due to the particular focus on water reuse schemes, and should thus be related to influent water as a functional unit if water losses are to be properly reflected in WF.

8.3.2 Accounting of water withdrawals and releases

Withdrawals and releases of water are only accounted in the WIIX if they occur from or to natural freshwater resources. Consequently, water withdrawal from or release to the ocean are not accounted in the WIIX, as are withdrawals of release from the technosphere (e.g. a city).

This can lead to misinterpretation of WIIX results in case of water reuse systems that are operating near the ocean, or that use groundwater recharge to replenish local aquifers. For coastal WWTPs discharging either in a nearby river or directly into the ocean, WIIX of this release is evaluated quite differently: ocean release is not accounted, whereas river release is fully accounted even though this water will end up in the ocean soon after release, without any major downstream use for humans or ecosystem.

For reuse systems targeting groundwater recharge or agricultural irrigation, potential water losses through evaporation or evapotranspiration will decrease the effective fraction of water that reaches the groundwater table. Depending on the setting of system boundaries, these water losses will lead to lower credits for the water reuse system compared to the direct discharge of water into a river. However, use of reclaimed water for groundwater recharge or agriculture will eventually lead to a decrease in local water scarcity, because other natural water resources are less exploited. However, this fact may not be properly reflected in the WIIX results, which prefer direct discharge of water into a river over groundwater recharge (with evaporation losses) or agricultural irrigation. In case of agriculture, the cut-off of system boundaries at the place of water delivery may overcome this problem and enable the full accounting of reclaimed water delivered to agriculture in the WIIX. Otherwise, only 25% of applied irrigation water may be accounted as water release into the groundwater based on a European estimate of evapotranspiration losses and water uptake by plants [51].

For groundwater recharge, water in aquifer should probably be fully accounted as water release, even though some water losses occur. From a water management point of view, replenished groundwater in aquifer is valuable to overcome seasonal problems of water scarcity by the storage function of the aquifer. This fact may be reflected best in LCA by regarding the relevant part of the aquifer as a “technical system for water storage” and include it into the system boundaries (cf. chapter 2).

8.3.3 Water scarcity factor

Local water scarcity is an important driver for the implementation of a water reuse scheme. As local water availability and consumption can be highly heterogenous in a region, water scarcity and also water reuse are directly related to the local water balance. However, water scarcity calculation in LCA is currently based on catchment or water-shed based information of water balances, so that scale and resolution of this information does not always match locally detected levels of water scarcity. In addition, local measures of improved water management (e.g. import of water from nearby regions with higher water availability) will not be properly reflected in the WF if they are located in the same water shed and the scarcity index for both areas is the same.

This LCA study uses the new AWARE method [52] to determine factors for water scarcity for WIIX calculation. Although the AWARE method improves specific short-comings of the previous water scarcity index by Pfister et al. [130], it may still not properly reflect actual levels of water scarcity on a local basis. This leads to potential bias in WF in case of locally dependent options to improve water management, as could be seen in the case of El Port De La Selva (cf. chapter 2). When using the AWARE method for WIIX calculations of water reuse systems, it should be clearly communicated that this assessment is based on catchment-based water data and may not correctly reflect the local situation of water scarcity in a specific place (e.g. a coastal village). Localized factors for water scarcity could probably be produced within AWARE, but the application of the AWARE method to set up locally specific water balances has not been tested so far.

8.3.4 Water quality index

Within the WIIX method, water quality is assessed by using the Water Quality Index (WQI). The WQI is defined by benchmarking all substances found in the water against a specific reference concentration for each pollutant, taking the minimum of all quotients as the final WQI. This "one-beats-all" approach is very simple to apply, but has an inherent drawback: if only one substance is close to or higher than the reference concentration, WQI for the water will be determined only by that substance, even though other concentration of pollutants may be far below targets for good water quality. Hence, WQI evaluates water quality not based on the mixture of pollutants, the potential uses of the water or the type of treatment required, but only towards a strict reference for each single substance. It thus can happen that water resources are evaluated with a low WQI and hence a low WIIX in a water-scarce area just because one substance does not hold the benchmark. This may clearly underestimate the value of water resources in water scarce areas, where quantity rather than quality is the major problem.

In addition, WQI is very sensitive if pollutant concentrations are near the benchmark: Figure 8-3 illustrates the WQI for a range of concentrations and a hypothetical target value (0.1 mg/L). A two-fold higher concentration compared to the benchmark reduces WQI from 1 to 0.5 (-50%), whereas a 10-fold concentration reduces WQI to 0.1 (-90%).

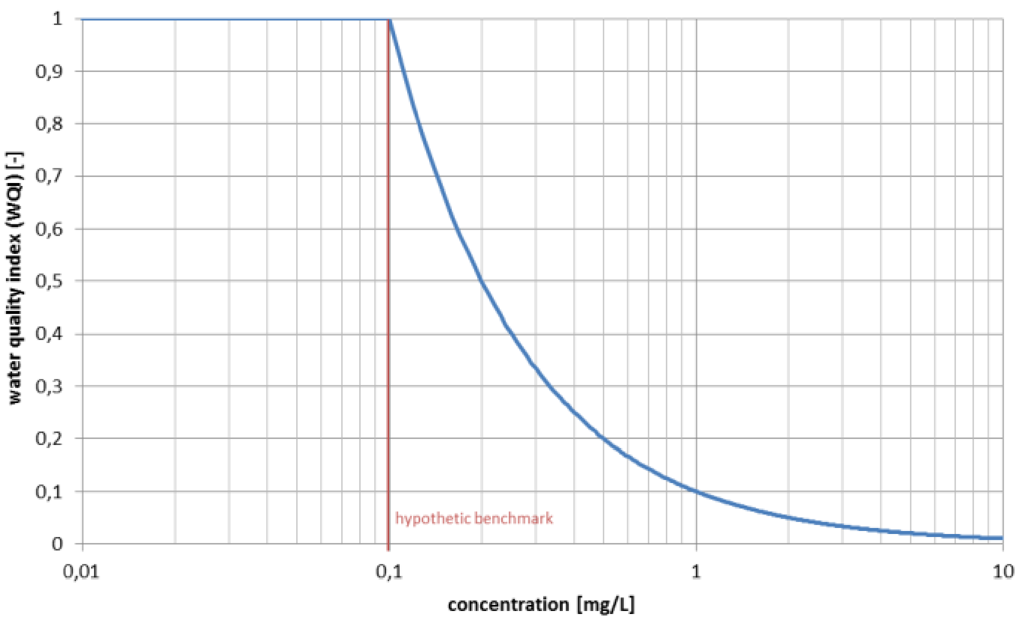


Figure 8-3: Water Quality Index (WQI) for different concentrations of a pollutant and a hypothetic benchmark of 0.1 mg/L

As a result of this approach, the choice of reference concentration or benchmark strongly affects the WQI and the total WIIX score. In this study, the reference concentrations are defined as the limit values defined in the EC directive 2008/105/EC for environmental quality standards (EQS) in the field of water policy [20]. Table 8-1 provides an overview of EQS concentrations for a set of substances related to good surface water quality. Some reference concentrations (e.g. metals) are dependent on the local geochemical background and other water quality parameters such as water hardness.

Table 8-1: Overview of environmental quality standards for selected substances and regulatory values/ recommendations to permit good surface water quality [20]

Parameter	Unit	Reference concentration for good surface water quality
Solids	mg/L	35.0
COD	mg/L	30.0
TN	mg/L	2.0 (NH_4^+: 0.5; NO_3^-: 50)
TP	mg/L	0.2
Cd	µg/L	0.45
Cr	µg/L	3.4
Cu	µg/L	1.4
Hg	µg/L	0.05
Ni and compounds	µg/L	20
Pb	µg/L	7.2
Zn	µg/L	7.8

For specific substances, using the EQS in water reuse assessment may be debatable: in reality, tertiary treated wastewater used for artificial groundwater recharge needs to fulfil other criteria than secondary treated wastewater discharged to surface water. For water reuse in agriculture, benchmarks for nutrients in irrigation water are not meaningful for a low water quality: in contrast, nutrients may even be beneficial for agricultural soil. In addition, specific benchmarks (e.g. for Cu and Zn) are rather low for evaluating the "value" of reclaimed water, even though these substances do not reflect a large hazard potential for human health or ecosystems. Finally, release of reclaimed water into the environment will only be credited a low WIIX score even if the quality of the reclaimed water is assumed to be sufficient for several uses. With WQI having a large impact on the final WIIX score, aspects of water quantity could be masked although they are very relevant for assessment of water reuse strategies (e.g. releasing 10 m³ of water with WQI = 0.1 due to "high" Zn concentration of 78 µg/L will be less beneficial for WIIX than releasing 2 m³ with WQI = 1). Alternative methods for assessing water quality in WF are currently under development, e.g. with a specific method defining water quality via specific use categories of the water [131]. This may help to overcome the drawbacks of a standard reference concentration for all water uses which is independent of the projected use of the water.

Care has to be taken in WQI calculation for groundwater recharge systems when defining the boundary between technosphere and environment: passage of infiltrated water through an unsaturated soil will lead to a further improvement of water quality, which may be seen as a "nature-based" treatment system. Hence, water quality data for the WQI could also be defined at the interface between the unsaturated soil and the aquifer, taking into account the positive effects of soil passage on water quality for the final WIIX of infiltrated water.

Bias in WQI calculation can also originate from incomplete datasets for the different water flows. In the present study, source water for drinking water production (as alternative to water reuse) is often assumed with a WQI of 1 if no quality data is available. This may overestimate the WIIX of available freshwater resources and influence the comparison between different alternatives of water management.

8.3.5 Alternatives to WIIX: water availability footprint

Due to the shortcomings of the WQI assessment discussed above, it may be useful to exclude the aspect of water quality in the WF study. This will also prevent the problem of "double-counting" of water pollutants both in traditional LCA indicators (e.g. eutrophication, toxicity) and in the WIIX.

WF that account only for water volume and water scarcity are known as "water availability footprint (WAF)" (ISO 14046). As has been shown in case studies of El Port de la Selva (cf. chapter 2) and Shafdan (cf. chapter 6), water reuse would be assessed differently while using a WAF. If local problems of water management are more related to water quantity, a WAF may be more useful to show the benefits of water reuse in increasing the available water volume. However, a high quality of reclaimed water is not reflected in the WAF, meaning that reuse systems with higher quality (e.g. polishing of secondary effluent to higher quality) may not be detected as beneficial in the WAF as total volumes released to the environment are comparable. Finally, the user has to decide and communicate openly about the choice of methods for water footprint assessment and provide clear guidance to the target audience about the consequences of this choice.

8.3.6 Conclusions

In summary, the calculation of the WIIX was found to be useful for showing the benefits of water reuse to the local water balance. However, the simplified method of the WIIX may not be fully suitable to reflect the more complex issues of water reuse, especially regarding water quality assessment (WQI with "one-beats-all" approach) and setting appropriate system boundaries to include the water flows originating from the technosphere (e.g. WWTP effluent). Summarizing aspects of water volume, local water scarcity, and water quality into a single indicator score seems to somehow mask valuable information on the different aspects, overlaying impacts of quantity, scarcity and quality. Finally, benefits of water reuse may not be adequately reflected if only the WIIX indicator is used to evaluate these aspects. In fact, minor changes in definitions or inventory data may have a major impact on the WIIX score, and hence on the comparison between water reuse and other alternatives of water supply.

Overall, the WIIX in its current version seems to favour reuse systems delivering high water quality (e.g. RO) over those systems with lower quality due to the high sensitivity of the WIIX score to the WQI, even if the water losses in the high-quality treatment are substantial (e.g. 50%). In contrast, water scarcity is defined per watershed, underlining the general need for water reuse in water-scarce areas by high WIIX scores, but not reflecting the benefits of water management options on a more local scale.

Some aspects have to be clearly communicated when using the WIIX for assessment of water reuse systems:

- Scope of the LCA study (system boundaries and functional unit) has to be analysed properly and may require adaptation for a meaningful calculation of the WIIX
- Summarizing different aspects of water management (quantity, quality, and scarcity) into one indicator can mask certain effects of water reuse. Calculating a water availability footprint can help to clarify the influence of water quantity and water quality on the comparison.
- The choice of reference concentrations for water quality assessment has a high impact on the WIIX score and should probably be adapted to the type of targeted water use.
- It may be difficult to compile consistent datasets for water quality parameters for all relevant water flows (e.g. freshwater, groundwater, reclaimed water in planned reuse scheme), which further increases the lack of confidence in the WQI calculations.

- **Water scarcity is assessed on a watershed level in the WIIX and may not properly reflect the local situation of water availability.** In consequence, certain benefits of local water management and water reuse may not be fully reflected in the WIIX if affected water resources are all located within the same watershed.

8.4 Evaluation of Toxicity Impact Categories of LCA

In several case studies which include disposal of sewage sludge in agriculture, the impact indicator for human toxicity based on the consensus USEtox model revealed very high absolute scores for the systems (up to > 100 % in normalization). In particular, these high scores can be traced back to the associated direct emissions of heavy metals into agricultural soils with sewage sludge. However, the high impact of heavy metal input with sewage sludge on human toxicity has to be critically discussed regarding the underlying uncertainty of the characterization factors of this model. Below, the case study of Braunschweig with agricultural use of sludge is used to further illustrate the short-comings of the toxicity assessment, but similar conclusions may be drawn for the other case studies.

In general, the characterization factors (CFs) for heavy metals derived in the USEtox model are "[…] classified as interim due to the relatively high uncertainty of addressing fate and human exposure[…]" [132]. Furthermore the authors of the USEtox model argue that "[…] interim CFs might be used in LCA studies, but with great caution and under awareness of their large inherent uncertainty. In the case that an LCA result is dominated by impact scores based on interim CFs, we advise to proceed with great caution to their interpretation underlining that these factors are neither recommended nor endorsed." [132]. As already mentioned, human toxicity assessment of most case studies is dominated by the contribution of heavy metals, mostly via application of sewage sludge in agriculture. However, the correlating residual errors and square of the log-normally distributed standard deviation are particularly high for USEtox CFs addressing human health and exposure via agricultural soil [132], focussing on human exposure via plant uptake of heavy metals and food consumption. In contrast, pathways of groundwater transport have no date been disregarded for calculating the human toxicity CFs of heavy metals [76].

Focussing on the example of Braunschweig sewage sludge, Figure 8-4 shows the relative mass contribution of each heavy metal to the total load in the sludge. As previous risk assessment studies on wastewater and sludge already indicated [133], Zn and Cu dominate the total load of metals in sludge with more than 90%, originating from the relatively high content of Zn and Cu in municipal wastewater. In contrast, Pb, Cr, and Ni constitute around 10% of the total metal load, whereas Cd and Hg are below 1%. Based on these total loads, Figure 8-5 shows the impact score for human toxicity and the contribution of the different metals, both for the USEtox mode and also an alternative toxicity model from ReCiPe [12] (USES-LCA [134]).

In the USEtox model Zn is responsible for almost 90 % of the total score, whereby Hg, Pb, and Cd contribute around 10%. In this model, the contribution of Cr, Cu, and Ni is negligible for human toxicity. In contrast, the ReCiPe model calculates the highest contribution for Cd (58%), then Zn (37%), Pb and Hg (sum of 5%) and all other metals below 1%. This exemplifies the high differences in relative contribution of each metal for the total toxicity score between the two models. Furthermore, absolute scores measured at endpoint level (DALY) show that USEtox calculates 820 DALY/a for the Braunschweig sludge, whereas ReCiPe calculates only 5 DALY/a (Figure 8-6). The expected exposure group in endpoint assessment on continental scale also differs between USEtox and ReCiPe [132, 135]. In relation to population equivalents USEtox human toxicity leads to a scores of 2.73 µDALY/(pe*a) and ReCiPe human toxicity to 0.01 µDALY/(pe*a) for the application of Braunschweig sludge in agriculture. This difference (factor 270) also has high consequences for normalised scores (EU27), where USEtox leads to a significantly higher score than ReCiPe (Figure 8-6).

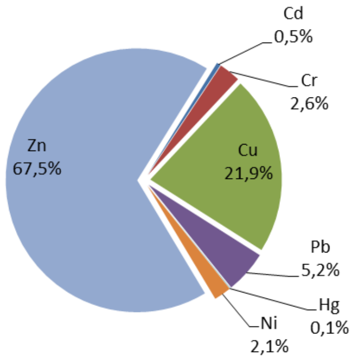


Figure 8-4: Composition of heavy metals in Braunschweig sewage sludge based on mass balances

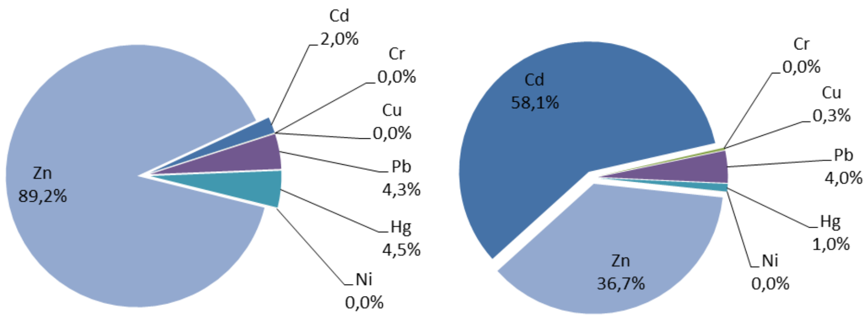


Figure 8-5: Composition of human toxicity indicator score by applying USEtox model (left) and ReCiPe model (right) based on mass balances

The shortcoming in LCA endpoint assessment and the related toxicity indicators (especially for USEtox) is underlined by the following consideration: the Braunschweig WWTP recycles approx. 4 kt DS sludge each year, in Europe more than 4 000 kt DS sludge/year are recycled to agricultural soils [136]. Assuming the same quality for all sludge reused in EU agriculture as comparable to the high quality of the Braunschweig sludge (optimistic approach), the resulting DALY values would be increased by a factor of 1000. Consequently in average the European citizen's lifetime would be reduced or negatively influenced by 5.2 years (USEtox) and approx. 9 hours (ReCiPe) only due to sewage sludge application in agriculture. It should again be noted that 90 % of the USEtox score originates from Zn, which is present in many other products for human consumption and is also used as a nutritional supplement in some countries (reduced or negatively influenced lifetime is evoked by zinc). Furthermore the normalized score per EU citizen is accounted with 102 % for USEtox and only 3 % for ReCiPe (Figure 8-6). Reflecting human life and other zinc immissions humans are exposed too (e.g. dermal path via air or consumption of sanitary products (toothpaste with zinc)) the normalization as well as the endpoint assessment of USEtox is implausible.

An important short-coming of toxicity assessment in LCA may be the linear scaling of environmental doses to effects on humans, and the additivity of several effects (CTU or DALY). A better approach for assessing potential impacts from sewage sludge application is a quantitative risk assessment, as has been done for the Braunschweig sludge in another EU-funded project [133]. Based on a long-term scenario of 100a of sewage sludge application in agriculture, none of the metals exceeded the PNEC (Predicted No Effect Concentration) which represents a benchmark for negative effects on human health. The weighed quotient of PEC and PNEC of each metal in relation to the other is shown in Figure 8-7, showing the relevancy of heavy metals compared to the others regarding human health/ toxicity among the food chain (similar to Figure 8-5).

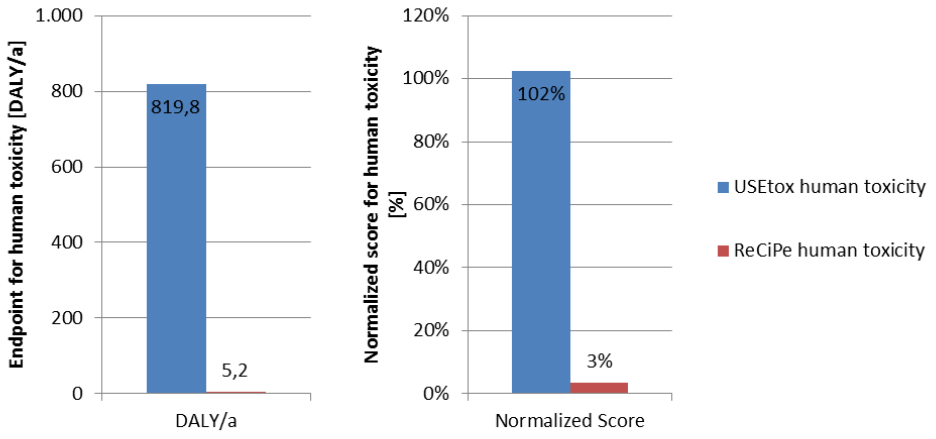


Figure 8-6: Endpoint assessment and Normalization for human toxicity indicators of USEtox and ReCiPe

Within this risk assessment, Cd is identified as priority hazard for human health due to potential accumulation along the food chain. This is due to the high human toxicity of cadmium and the proven accumulation rate of cadmium in plants. Nonetheless, also the calculated PEC (Predicted Environmental Concentration) for Cd is only around 10 % of the PNEC in this study, underlining the low risk for human health of applying this sludge to agriculture on a long-term basis. Both Cu and Zn show relatively high shares because of their comparably high concentrations in sewage sludge (and environment) and average accumulation rates. However, human toxicity effects of both metals are relatively low and do not lead to a high risk of negative effects on human health.

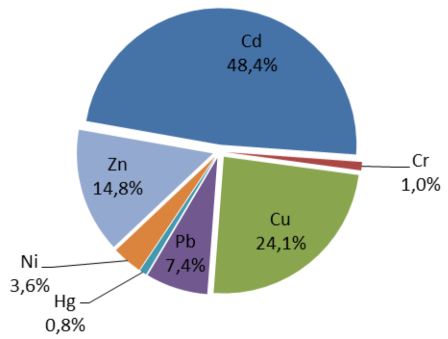


Figure 8-7: Composition of risk characterization ratios based on chemical risk assessment for Braunschweig sewage sludge [133]

This comparison between different LCA models for toxicity assessment and a risk assessment study reveals that both absolute effects and also relative contribution of metals to potentially negative effects on human health are highly different between the models.

This outcome underlines the high uncertainty associated with toxicity assessment in LCA, and consequently the low level of confidence in the validity of its results. In summary, high normalization scores in the human toxicity indicator in several case studies of this report should be interpreted with care and not be used for decision making between several alternatives.

8.5 Combining RA and LCA to reveal potential trade-offs in risk mitigation

Both microbial risk assessment and LCA describe and assess potentially negative effects on human health or ecosystems from the activities of water reuse. Whereas RA aims at quantifying the probability of adverse effects conditioned on local conditions and system specific infromation, LCA has a more global perspective and assesses potential impacts throughout the life cycle of the process with non-specific exposure models.

The implementation of a water reuse scheme has both consequences on the local and on the global level. Risk assessment gives information about potential local risks for human health due to water quality provided by the reuse system. For reducing this risk below acceptable levels, different types of treatment of reclaimed water can be applied. The case studies have shown that those measures for risk reduction come with a certain "environmental footprint" due to the demand for electricity, chemicals and infrastructure for water treatment and transport. However, this footprint of water treatment and transport can also have negative impacts on human health, e.g. by emitting atmospheric pollutants in power plants or chemical production facilities, or by causing climate change via global warming.

This inherent trade-off of technical risk reduction measures in water reuse (= reducing local risks by increasing global impacts) leads to the question of how to weight local against global impacts in a decision-making process. Reaching an adequate level of risk reduction in water reuse without imposing major global impacts due to water treatment and transport could be a target for planning a water reuse system. A scientific approach to compare local and global impacts has to compare the results of LCA and risk assessment on a meaningful basis. Several studies have tried to blend elements of risk assessment and LCA, but the methods have different characteristics which may lead to potential pitfalls during combination [137]. A recent study combines results of quantitative microbial risk assessment and LCA for the impact assessment of sewage sludge disposal in agriculture, focussing on the risk from pathogenic microorganisms [138, 139]. Results of risk assessment and LCA can be integrated if comparable units are used (e.g. DALY for impacts on human health). However, risk assessment often takes a worst-case approach comparing exposure to certain threshold levels and is limited to localized effects at one site, whereas LCA does not take into account time-related and spatial aspects and calculates potential impacts over the entire life-cycle for average operating conditions. In addition, modelling choices in risk assessment will also have an impact on the comparability of the two methods [140].

Nevertheless, the combination of risk assessment and LCA can reveal additional insights into the illustrated trade-off of technical risk reduction measures in water reuse between local and global impacts. To illustrate this combination, results of risk assessment and LCA are combined and integrated in the following chapter, taking the disinfection step for secondary effluent in Braunschweig as an example (cf. chapter 3). In principle, positive impacts of disinfection on human health due to reduced risk of exposure to pathogenic microorganisms for locals are weighted against global impacts of disinfection on human health from the production of electricity, chemicals, and infrastructure.

8.5.1 Human health (endpoint) assessment with LCA

Global impacts on human health originating from efforts for disinfection (electricity, chemicals, infrastructure) are evaluated with LCA, following the respective endpoint approach of the ReCiPe method [12]. For the endpoint assessment, five midpoint indicators are used and converted into the endpoint indicator for damage to human health measured in DALY (see Table 8-2). Additionally to the midpoint indicator of climate change (cf. chapter 3) and of direct human toxicity (cf. method discussion in chapter 8.4), three other midpoint indicators are chosen for the endpoint assessment according to the ReCiPe-standards [135]. All five indicators as well as their potential harmfulness and the main contribution in the Braunschweig scheme are listed below:

- **Global warming potential (GWP):** as the most popular impact category of LCA assessing the contribution of the scheme to anthropogenic climate change; beside the indirect emissions of fossil-based carbon dioxide (CO_2) mainly from electricity consumption, the potential direct formation of dinitrogen monoxide (N_2O) in the activated sludge tank is a relevant contributing source.
- **Human toxicity potential (HTP):** as an impact category describing toxic effects on human health. Although the normalized score is very low (see Figure 8-6), mainly cadmium (Cd) and zinc (Zn) within sewage sludge showing a relevant contribution when sewage sludge is recycled.
- **Photochemical oxidant formation potential (POFP):** as an impact category for the formation of tropospheric ozone which is mainly influenced by the direct and indirect (via electricity consumption) formation of nitrogen oxides (NO_X)
- **Particulate matter formation potential (PMFP):** as an impact category for the formation potential of dust, mainly influenced by direct and indirect formation of aerosol forming emissions to air, such as ammonia (NH_3), sulfur dioxide (SO_2) and nitrogen oxides (NO_X) as well as the direct formation of particulates in electricity production.
- **Ionising radiation potential (IRP):** as an impact category showing the radiation potential mainly evoked by the share of nuclear power in electricity production.

Table 8-2: Conversion of midpoint to endpoint impacts on human health for ReCiPe (Europe, hierarchical perspective) [135]

Midpoint impact category	Endpoint of damage to Human Health in DALY (hierarchical perspective)
Global warming potential (GWP)	$1.40 \cdot 10^{-6}$ DALY/kg CO_2-Eq
Human toxicity potential (HT)	$7.00 \cdot 10^{-7}$ DALY/kg 1.4-DCB-Eq
Photochemical oxidant formation potential (POFP)	$3.90 \cdot 10^{-8}$ DALY/kg NMVOC
Particulate matter formation potenitial (PMFP)	$2.60 \cdot 10^{-4}$ DALY/kg PM_{10}-Eq
Ionising radiation potential (IRP)	$1.64 \cdot 10^{-8}$ DALY/kg U^{235}-Eq

The calculated endpoint results for the different scenarios are shown in Figure 8-8 as total scores and in relation to the exposed population for the global impacts (assumption: exposed population is equivalent to EU population [135]). Figure 8-9 shows the changes between reference and disinfection scenarios, i.e. the global impacts related to the implementation of disinfection processes. From the endpoint assessment, additional damage on human health from global impacts of disinfection amounts to 0.4-1 DALY/a or 0.001-0.002 µDALY/(pe*a) if related to the total exposed population (EU). In contrast, damage on human health can be substantially reduced if water management is optimised (irrigation on demand), avoiding more than 2 DALY/a or 0.004 µDALY/(pe*a) for the exposed population. For both effects, the main contributing pathway of the additional or reduced damage is via global warming, whereas global impacts via human toxicity have only a low contribution since the input of heavy metals on arable land via sewage sludge or wastewater recycling is not affected by disinfection stages.

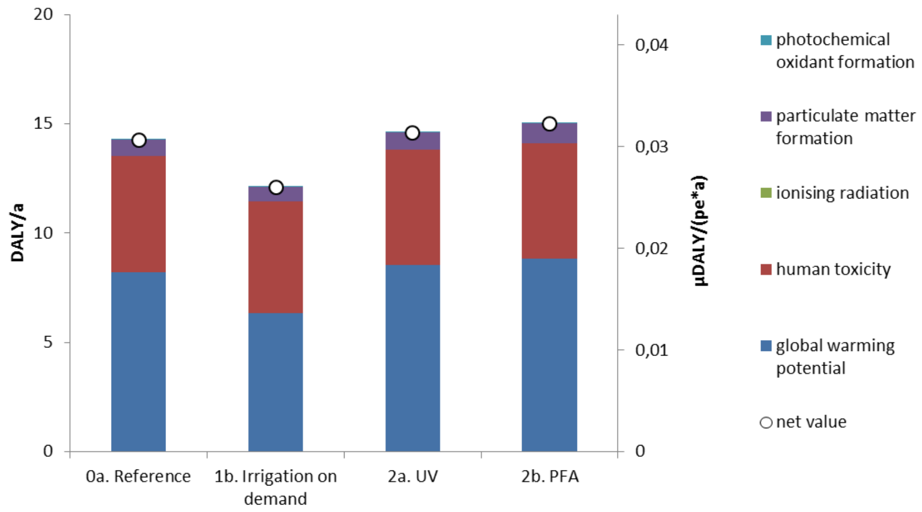


Figure 8-8: Total LCA impact (left axis) and LCA impact related to exposed EU population (right axis) of endpoint assessment towards human health for different scenarios in Braunschweig

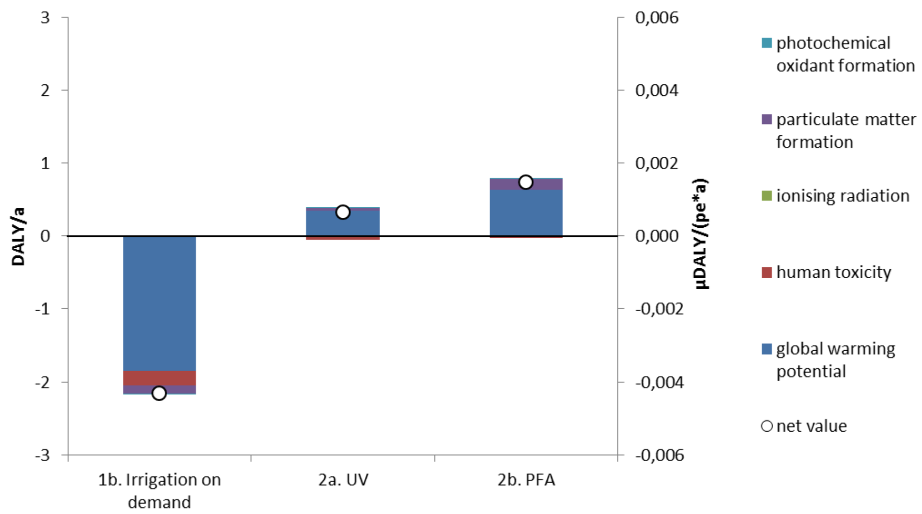


Figure 8-9: Changes in total LCA impact (left axis) and LCA impact related to exposed EU population (right axis) of endpoint damage to human health assessment for selected scenarios compared to the 'reference'

8.5.2 Human health assessment with risk assessment

In contrast to LCA, QMRA already expresses risks and human health effects in DALYs. To simplify the comparison of LCA and QMRA results, only the scenario for fieldworkers is considered. Since the risk for fieldworkers is orders of magnitudes higher than the risk for the other scenarios, the contribution of the other scenarios to the total DALY score is neglegible. Following a worst-case approach, the respective maximum DALY-values were chosen for each scenario, meaning different pathogens are of main interest addressing priority risks for different teriary treatment schemes. These pathogens and respective maximum DALYs are shown in Table 8-3.

Figure 8-10 and Figure 8-11 show the resulting overall DALY/a and µDALY/(pe*a) for risk assessment similar to the previous figures for LCA (in total and the changes compared to the reference scheme). It becomes apparent that the factor between DALY/a and µDALY/(pe*a) is different compared to LCA results, since both impacts of the assessments are allocated to different exposed groups of people: QMRA shows the benefits of disinfection for human health of 50 fieldworkers, and LCA describes potential health impacts for a group of 464’815’432 European citizens [135].

Table 8-3: Scenarios, respective pathogens and maximum DALYs

Scenario	Pathogens contributing the most to existing risk	Risk score [DALY per pe*a]	Related figure
Reference/status-quo (no disinfection)	*Campylobacter*	$6.60 \cdot 10^{-3}$	Figure 3-25
Irrigation on demand (no disinfection)	*Campylobacter*	$1.80 \cdot 10^{-3}$	Figure 3-28
Status-quo – UV disinfection	*Giardia*	$6.08 \cdot 10^{-5}$	Figure 3-25
Status-quo – PFA disinfection	*Rotavirus*	$8.34 \cdot 10^{-4}$	Figure 3-25

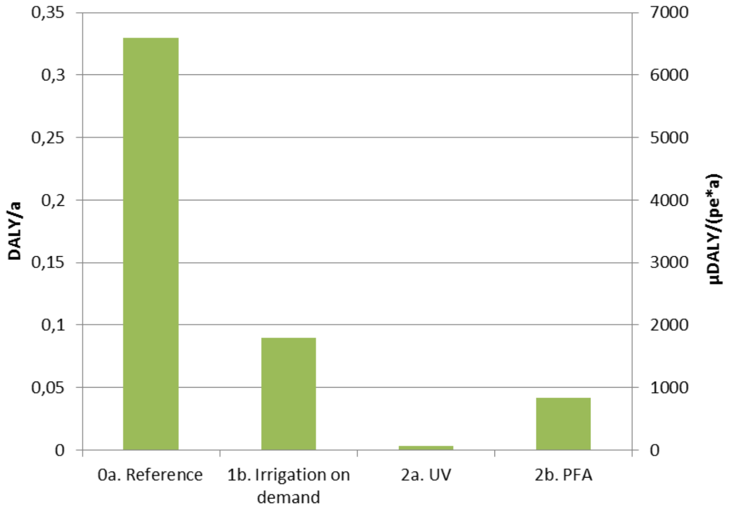


Figure 8-10: Total QMRA impact (left axis) and QMRA impact related to exposed group of fieldworkers (right axis) for different scenarios in Braunschweig

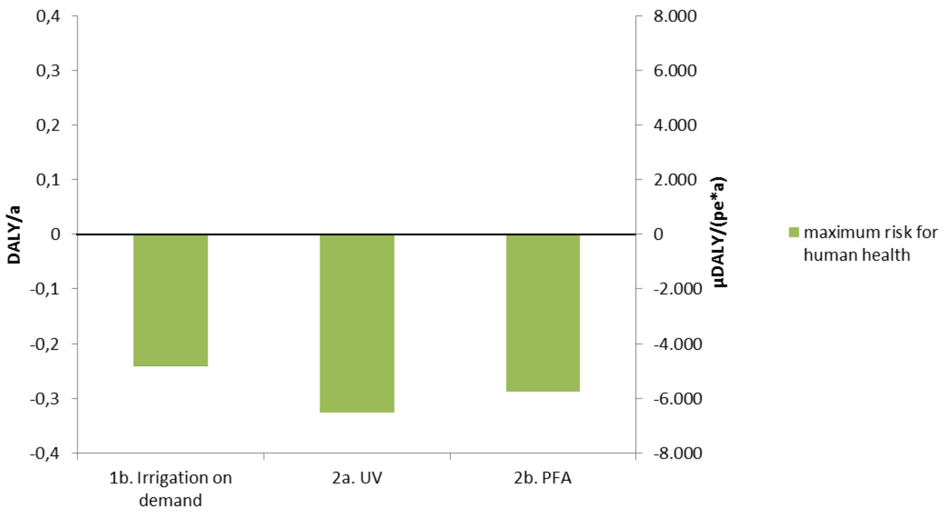


Figure 8-11: Changes in total QMRA impact (left axis) and QMRA impact related to exposed fieldworkers (right axis) for selected scenarios compared to the ‘reference’

8.5.3 Combining QMRA and LCA

Since the exposed groups of people for the DALYs of QMRA and LCA are different, the combination of both results can be established in two perspectives. First, Figure 8-12 shows the "global" or European perspective in total DALY/a: the net efforts or benefits of LCA and risk reduction potential of QMRA are not allocated to a specific exposed group, but summarized in total.

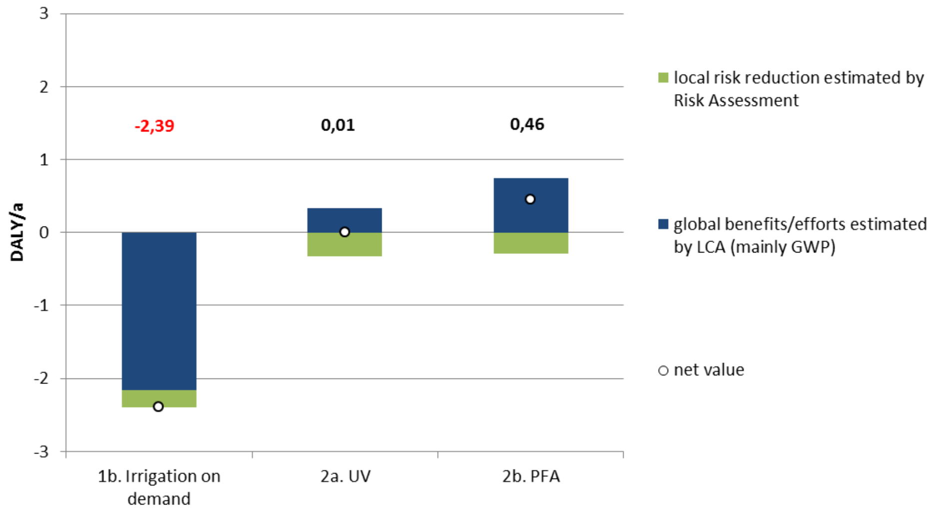


Figure 8-12: Combination of LCA and QMRA in total DALY per year ("global"/European perspective)

As indicated in Figure 8-12, the scenario 'Irrigation on Demand' reduces total DALYs per year related to the net value of the reference scheme, since both DALYs from LCA and also QMRA are reduced. For LCA this is a consequence of the lower energy consumption by avoiding the pumping of reclaimed water from the WWTP to the agricultural fields beyond actual plant demand ("excess irrigation"). In addition, the risk of infection originating from the use of reclaimed water is reduced in QMRA, mostly due to reduced exposure time for fieldworkers (irrigation in summer only when plants actually need water) and due to reduced pathogen concentration in raw wastewater and WWTP effluent in this summer season (cf. Chapter 3.3.5). Hence, this scenario can be described as an option to reduce both local and global potential damages for human health from water reuse ("win-win").

In contrast to the scenario 'Irrigation on demand' discussed above, the disinfection scenarios 'UV' and 'PFA' show an increasing impact on human health in LCA results: the implementation of a disinfection system is associated with resulting energy consumption and related emissions, and consequently with additional DALYs. These additional DALYs per year are more than double as high for 'PFA' compared to 'UV' (cf. related impact categories in Chapter 3.2.3), mostly due to the use of chemicals in PFA. In contrast, the local risk reduction of both disinfection systems is almost comparable, with slightly higher pathogen removal for the 'UV' scenario. Overall, the additional "global" DALYs of LCA and the reduced "local" DALYs of QMRA almost offset each other for the UV system (Figure 8-12), meaning that the implementation of a UV disinfection in Braunschweig will not improve nor decrease the cumulative impact of the water reuse scheme on human health from a "global" or European perspective. For the PFA disinfection, "global" efforts in LCA are higher than "local" benefits of QMRA in the "global" or European perspective. Hence, this type of disinfection induces more potential damage to human health by electricity and chemicals demand than it reduces from eliminating microbial pathogens in the Braunschweig reuse scheme. Of course, this comparison is highly influenced by local conditions and exposure models, and cannot be generalized from this study.

In fact, the actual decision whether or not to implement a disinfection step in Braunschweig will not be made using this “global” perspective. In reality, the decision will be based on the risk reduction targets and requirements by the local health authority for the exposed population in Braunschweig, taking a “local” perspective. The local decision-makers will not take into account potential effects of increased energy and chemical consumption towards globally distributed damages to human health.

Following this “local” perspective, Figure 8-13 shows the combination of QMRA and LCA impacts for the group of local fieldworkers in the Braunschweig reuse scheme. Now, the local benefits of risk reduction with a disinfection step are fully accounted for each fieldworker, whereby the additional DALYs from LCA are equally distributed to all European citizens as exposed population group for the global effects. Consequently, this additional “global” effort is very low for all 50 fieldworkers, and the local benefits for all three scenarios are predominant.

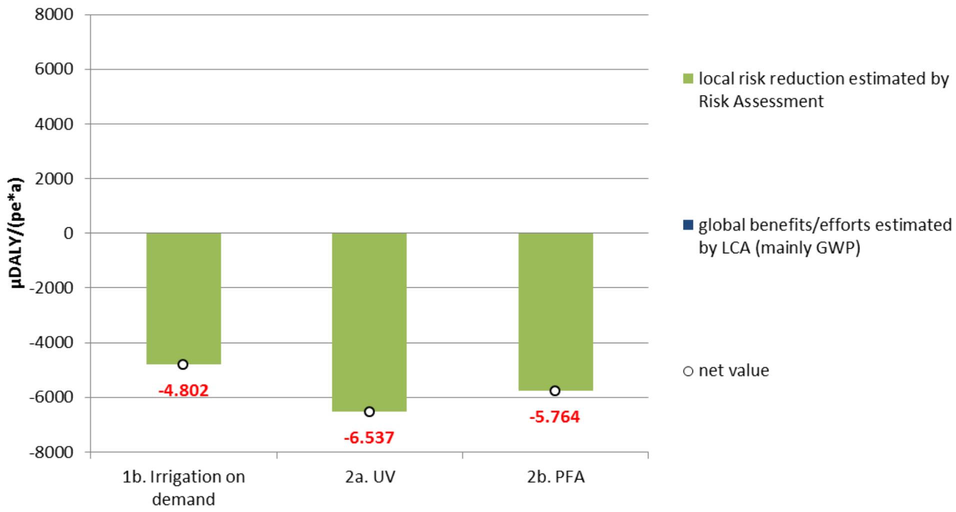


Figure 8-13: Combination of LCA and QMRA in µDALY/pe*a (local/fieldworker perspective)

In summary, the implementation of an additional disinfection system (or in general each technology for tertiary treatment to improve water quality and protect human health) reduces health risks on a local level, while globally additional efforts may cumulate in higher “potential damages” compared to the avoided “damage” on a local scale.

8.5.4 Discussion

In general, the direct combination and offsetting of DALYs derived from LCA and QMRA results can be challenged in its consistency, mostly due to the different methodological approaches in both types of assessment:

- LCA traditionally derives impact assessment via midpoint indicators and further modelling of the cause-effect chain until the endpoint. Hence, the calculation of DALYs for the endpoint human health is linked with high uncertainties in the model and on a very generic continental fate and exposure model.
- Midpoint assessment of LCA itself is affected by intrinsic uncertainties, assuming that cumulative environmental or health-related effects are linear to emission loads. This linearity of emission to effect is represented by constant characterization factors (CF) for each type of emission in LCA, and it is different to compare with the approach in QMRA (e.g. for calculating toxicity potentials).

- In terms of transferring the midpoint assessment to an endpoint assessment in LCA, double counting of certain emissions cannot be excluded. For excample, nitrogen oxides (NO_X) are accounted in midpoint assessment for potential effects in photochemical formation and particulate matter formation, which is consistent to the methodology of midpoint assessment, since NO_X has the potential for both effects. However, in endpoint assessment both potential effects are aggregated and 1 mol NO_X creates a consequential DALY-value via photochemical and particulate matter formation, although it is doubtful that the same molecule of NO_X can have both effects with the full impact at the same time (= potential double counting).
- QMRA assesses and aggregates DALYs from microbial pathogens on a site-specific level and usually takes realistic worst-case assumptions [137]. In contrast, LCA uses mean concentrations of emission loads and estimated potential damage from them.
- For QCRA, a similar aggregation to DALYs by potential effects of chemical substances (e.g. trace organics) is missing and difficult. The method of QCRA calculates a risk ratio (PEC/PNEC) which cannot be directly transfered into DALY in a simple and methodologically consistent way. The latter effect of local emissions of chemicals may then be assessed with the global model of LCA. However, this use of LCA for assessing local effects contradicts to the approach to use a) site-specific risk assessments for assessing the local emissions and b) LCA only for global impacts.
- Another approach could be the integration of pathogen-related risks into an impact category of LCA, and the understanding of QMRA or RA in general as part of an LCA impact assessment. Even though this approach reflects the current trend of localization in LCA impact assessment, it may be doubted whether the tools of RA and LCA can be combined due to the intrinsic differences in methodology of the two approaches [137].

In the authors' opinion, RA should not be considered as a part of LCA or the other way around. Both methodologies are necessary to address relevant information in a different ways. The localisation trends in LCA impact assessment are seen rather critically. For most site-specific assessments and even for generic assessments where a linearity of CFs between emission, exposure and effect is not meaningful, RA with local and specific information is seen as a better choice than LCA with its global and generic models. This clear segregation between different kinds of assessment and their stand-alone interpretation is more useful for practitioners and underlines the complimentarity of the methods.

Finally, both LCA and QMRA are often based on a limited number of measurement data in this study. For a deeper evaluation of this method comparison, a broader dataset and full parameterisation is necessary. Furthermore, it is disputable if DALYs of LCA and QMRA are comparable if both tools are not based on the same amount and quality of input fata.

Despite existing uncertainties of this comparison, the principle message is clear and can be discussed under moral or ethical aspects. Tertiary treatment of reclaimed water to reduce pathogens or harmful chemicals in the product is generally associated with an increase in energy consumption and consequential potential global damages. The basic question arising from this comparison of LCA and QMRA is about the reasonable ratio between an adequate efforts and benefits:

- What are realistic health targets for water reuse schemes, taking into account also associated efforts of technical barriers?
- What are recommended health targets in different industrialized and developing countries and what are the consequential efforts?
- What about fairness between different groups of exposed people and environmental justice?
- How many people had to be protected by a tertiary treatment to make the efforts for a system justifyable?

- How do local health authorities and decision makers in policy observe local risks and risk reduction strategies in contrast to global challenges as climate change?
- Is a link between local benefits and global efforts established in decision making?

8.5.5 Conclusions

The combination of DALYs derived from QMRA and LCA has to interpretated with care due to different purposes the methods are designed and used for. However, bringing the results of LCA and QMRA together allows for illustrating the following aspects:

- Expand the discussion on water treatment requirements to a wider context of environmental and human health.
- Connect local benefits and global impacts of water treatment and thus address issues of acceptable effort-benefit-ratios and related environmental justice
- Support current debates regarding the proportionality of new water treatment technologies targeting emerging hazards with quantitative information on impacts and benefits

For the example of the Braunschweig reuse scheme, measures for local risk reduction can be basically divided into two categories:

- Measures reducing local risks whith increased global efforts (most likely technical measures such as disinfection as tertiary treatment)
- Measures reducing local risks without significantly increasing global efforts (e.g. improved management towards a demand orientated system operation or other non-technical measures of risk reduction)

In the latter example, it could be shown that a combination of irrigation on demand and UV-disinfection would reduce both global and local DALYs compared to the reference situation, being beneficial from a global and local perspective (cf. Chapter 3.2.4 and 3.3.6).

8.6 Outlook

The present report used the methods of water footprinting, LCA and health risk assessment in order to evaluate various aspects related to water reuse systems in Europe. Questions which have been addressed included:

1. Is water reuse feasible from a global environmental perspective?
2. Can the existing hygienic and chemical risk been quantified and controlled?
3. Does the use of water footprinting indicators provide additional insight into the decision making problems related to water recycling and reuse?

By implementing the named assessment approaches to different case studies the added value each of the assessment methods can provide became evident. While health risk assessment allowed summarizing available information to transparently assess the degree of confidence that the system under study is able to deliver “safe” water LCA provided deep insights into how water reuse can be implemented while minimizing the environmental footprint of the system.

Moreover, it could be shown that the combination and comparision of risk assessment and life cycle assessment may be an instrument to foster discussions about environmental justice. The approach provided a critical perspective about how much technology is actually needed. Furthermore, new methodological approaches have been proposed and implemented regarding the use of external information. Such approaches will be further pursued as they may help to communicate results and the underlying assumptions more transparently and reproducibly.

Especially regarding environmental and health risk assessment various notions about the nature of risk and how to communicate them remain areas which can be further improved. On the one hand, risk competence requires an at least basic understanding of probability. On the other hand, an overly academic approach to the subject might hinder the implementation of risk based approaches on an operator’s level, especially for small water reuse systems.

Multi stage implementation routines and checklist approaches, which require an increasing level of detail and information after each implementation phase, like e.g followed within the Australian Guidelines for Water Recycling might be a way to overcome this obstacle and should be considered on a Euopean level.

Finally, future work should address the issue about a common understanding about an acceptable level of residual risk, which in the end cannot be answered by science alone as well as the aspect of how to include both risk based and LCA based approaches into a common implementation framework.

9 Annex

9.1 General Annex

9.1.1 Life Cycle Assessment

Normalization

LCA indicator results are related to the total environmental impact per inhabitant in a reference area (here: EU-27). Normalized results reveal the individual contribution of each impact category to the total environmental footprint of societal activities, indicating if a specific environmental impact of the treatment schemes has higher or lower contribution. Normalization data are collected for all indicators with exception of the water footprint from latest available sources for EU-27 countries (Table 9-1).

Table 9-1: Normalization data for impact categories in EU-27

Indicator	Unit	Total impacts in EU-27	Source
CED	MJ/(pe*a)	122 950	[141]
GWP	kg CO_2-Eq/(pe*a)	11 215	[135]
FEP	kg P-Eq/(pe*a)	0.415	[135]
MEP	kg N-Eq/(pe*a)	10.12	[135]
TAP	kg SO_2-Eq/(pe*a)	34.4	[135]
ETP	CTU_e/(pe*a)	8 720	[142]
HTP	CTU_h/(pe*a)	8.47E-4	[142]

Water Scarcity Factors

Table 9-2: Water scarcity factors according to Pfister et al. [130] and AWARE [52]
AWARE factors represent "non-agri" data

Site	El Port de la Selva		Braunschweig		OldFord Water		Sabadell		Shafdan		Torreele	
WSI	Pfister	Aware	Pfister	Aware	Pfister	Aware	Pfister	Aware	Pfister	Aware	Pfister	Aware
JAN	0.026	0.63	0.025	0.46	0.404	0.73	0.831	0.99	0.012	0.35	-	1.32
FEB	0.021	0.63	0.023	0.51	0.520	0.79	0.730	1.03	0.013	0.42	-	1.52
MAR	0.020	0.73	0.018	0.47	0.351	0.83	0.512	1.21	0.018	0.89	-	1.66
APR	0.019	0.63	0.025	0.61	0.760	0.97	0.888	1.53	0.048	16.60	-	2.16
MAY	0.024	0.62	0.076	0.88	0.995	1.30	0.865	1.89	0.263	100.0	-	3.09
JUN	0.047	1.09	0.217	1.26	1.00	1.91	1.00	3.71	0.880	100.0	-	4.56
JUL	0.191	4.14	0.584	1.78	1.00	2.89	1.00	13.03	1.00	100.0	-	6.15
AUG	0.532	6.30	0.968	2.62	1.00	3.36	1.00	9.07	1.00	100.0	-	9.43
SEP	0.649	3.07	0.963	3.05	1.00	3.61	1.00	5.24	1.00	100.0	-	11.86
OCT	0.238	1.13	0.710	2.16	1.00	2.59	1.00	2.15	1.00	100.0	-	6.89
NOV	0.061	0.85	0.072	1.16	0.972	1.62	1.00	1.35	0.397	3.41	-	2.64
DEC	0.034	0.68	0.027	0.67	0.712	1.01	0.984	1.15	0.014	0.56	-	1.49
MEAN	**0.155**	**1.71**	**0.309**	**1.30**	**0.818**	**1.80**	**0.902**	**3.53**	**0.470**	**51.85**	**0.970**	**4.40**

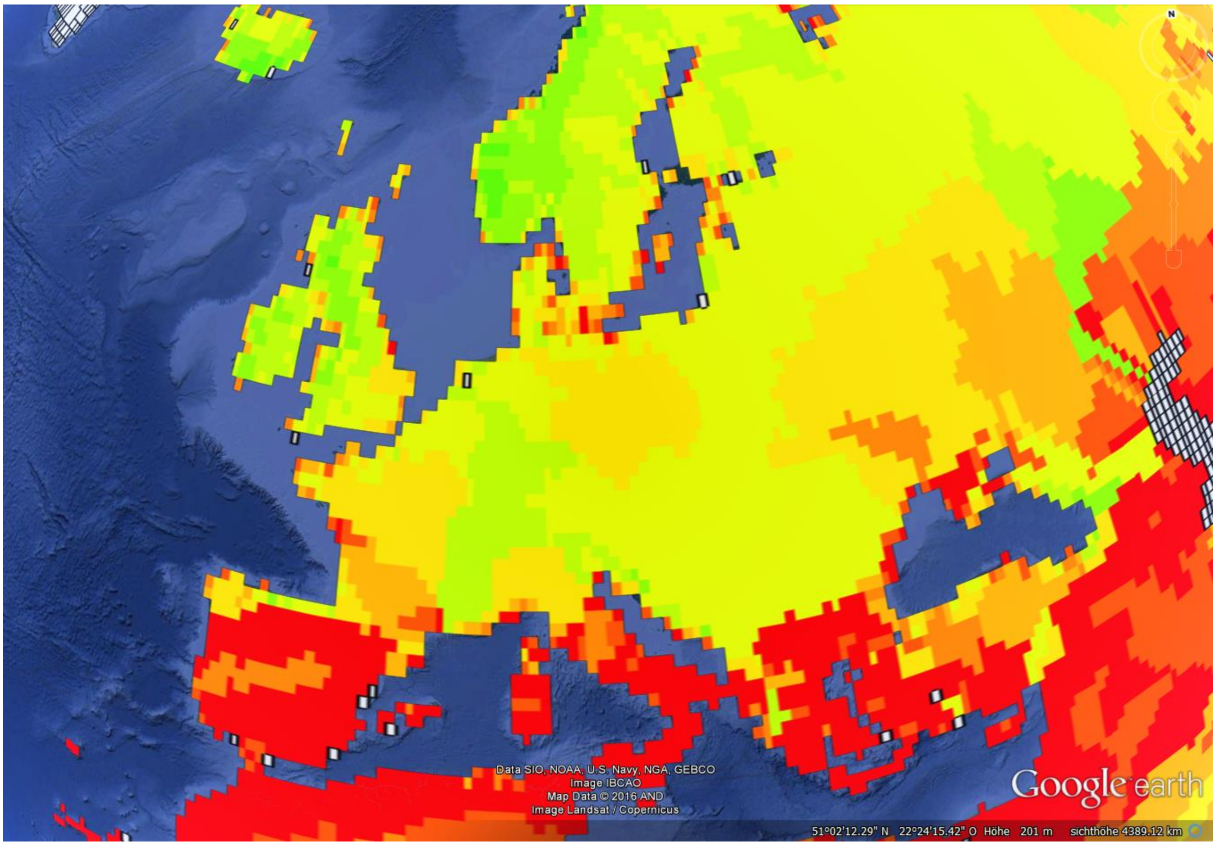


Figure 9-1: Map of annual AWARE index for Europe [16], measuring water scarcity per watershed
water scarcity decreases with colour range from red to orange to yellow to green

Indirect Water Impact Index

The section shortly describes the methodological approach to calculate the indirect water impact index (WIIX). The total WIIX is the sum of direct and indirect WIIX.

Equation 20 $$WIIX_{total} = WIIX_{direct} + WIIX_{indirect}$$

The calculation of direct WIIX and corresponding assumptions for water withdrawals, releases, water quality index and scarcity index are discussed in the inventory of each case study.

The indirect WIIX for the background processes has been calculated based on the related input/output water flows from Ecoinvent database for each case study and scenario.

The indirect WIIX is the product of the water scarcity index (WSI) and a "water volume quality footprint" (WVQF) (see below):

Equation 21 $$WIIX_{indirect} = WSI_X \cdot WVQF_{indirect}$$

The WSI is variable depending on the respective background process. For each case study the distinction between the national and the European WSI were conducted:

- processes related to electricity production were calculated in LCA with national electricity mix, and were consequently calculated with national mean WSI for each country
- processes related to chemicals production, transport and infrastructure were calculated in LCA with EU mix, and were consequently calculated with EU-mean WSI

An exemption is the case study of Shafdan in Israel, where Israeli WSI was accounted for both electricity and other background processes.

Table 9-3 shows the different WSI according to AWARE [52]. This simplified approach accounts indirect WIIX depending on the region where water withdrawal or release takes place for background processes. A more detailed estimation of localized effects of background processes is not necessary, since preliminary analysis revealed that the indirect WIIX is mostly negligible compared to the direct WIIX for all case studies.

Table 9-3: National WSI for selected countries and European Union according to Aware [52]

Country	WSI_X
Spain	31.49
Germany	1.24
United Kingdom	3.13
Israel	52.25
Belgium	1.16
European Union	5.19

The generic WVQF is expressed as the sum of two energy- and one process-based WVQF (see below):

Equation 22 $$WVQF_{indirect} = WVQF_{cooling} + WVQF_{turbine} + WVQF_{process}$$

For a given background process, the Ecoinvent LCI of this process gives the water inputs (W) for the three water usages and an overall water output (R), representing the sum of the three water usages minus the water evaporated in each usage.

Losses for water withdrawals by cooling and turbined water are accounted as follows

Equation 23 $$WVQF_{cooling} = W_{cooling} \cdot WQI_W \cdot EVAP_{cooling,i}$$

Equation 24 $$WVQF_{turbine} = W_{turbine} \cdot WQI_W \cdot EVAP_{turbine,i}$$

The withdrawal volumes ($W_{cooling}$ and $W_{turbine}$) where take from the LCA flow inputs. The WQI_W (WQI for withdrawals related to cooling and turbine water) is set to 1 (= optimal water quality for withdrawn cooling water). The evaporation factors ($EVAP_{cooling,i}$ and $EVAP_{turbine,i}$) represent the fraction which evaporates depending in use type (cooling or turbine water) and manufacturing item (chemicals & materials or electricity). This means that these WVQF for cooling or turbined water express the volume of "lost" water while considering optimal water quality of the discharged flows (= water quality is not degraded during water use for cooling or turbine purpose). Evaporation factors for cooling and turbine usages are specific to the considered process. However, typical ranges of values were estimated for cooling and turbine usages for the chemical/goods manufacturing industries and for electricity production system based on the Ecoinvent v3 water LCIs. Table 9-4 gives an overview on these assumed evaporation factors.

Table 9-4: Evaporation factors EVAPx for water uses (cooling or turbine water) and respective type of background process

Use category x	$EVAP_x$ for Chemicals, materials	$EVAP_x$ for Electricity
Cooling water	0.3875	0.032
Turbined water	0.001	0.0003

Unlike for cooling and turbine water usages, the quality of the water used for process purpose is degraded between the input and the output of the process (WQI ≠1). The process-based WVQF is thus more complex to evaluate as both evaporation factor and output flow (R) have to be estimated. Therefore a "work-around" has been developed by project partner VERI to roughly estimate the WVQF associated to a given process. The process-based WVQF is defined as follows:

Equation 25 $$WVQF_{process} = W \cdot WQI_W - R \cdot WQI_R$$

The process withdrawal W is the sum of withdrawals including LCA input flows (WQIW is set to 1 with optimum water quality):

- Water, lake [natural resource/in water]
- Water, river [natural resource/in water]
- Water, unspecified natural origin [natural resource/in water]
- Water, well, in ground [natural resource/in water]

For water releases in proce, water quality is estimated with WQI_R = 0.05, taking a conservative approach and assuming low water quality of the released water. The corresponding release-function R is defined as follows:

Equation 26 $$R := \begin{cases} R_A - WVQF_{cooling} - WVQF_{turbine} \geq 0 & \rightarrow \quad W - (R_A - WVQF_{cooling} - WVQF_{turbine}) \\ R_A - WVQF_{cooling} - WVQF_{turbine} < 0 & \rightarrow \quad W \end{cases}$$

R_A is the process water release calculated by the sum of water releases from LCA output flows (net of water evaporating to air):

- Water [air/lower stratosphere + upper troposphere]
- Water [air/non-urban air or from high stacks]
- Water [air/unspecified]
- Water [air/urban air close to ground]

Since the quantities of water evaporated by cooling or turbine water were calculated before, these are subtracted from the total evaporated volume. If this sum (representing the water release) is higher than the water withdrawal W, the corresponding release R is defined to be equal with the withdrawal W.

This is done for each background process to calculate a singular indirect WIIX for each dataset to respectively show the indirect WIIX for the different contributing background processes.

9.1.2 Risk Assessment

In the following the procedure of sampling from a normal distribution with non-informative prior distribution is explained.

The non-informative prior density is formulated as:

Equation 27 $$p(\mu, \sigma^2) \propto (\sigma^2)^{-1}$$

The multiplication of this prior with the formula for the normally distributed data leads to the joint posterior distribution of the parameters μ and σ^2 given the data (y):

Equation 28 $$p(\mu, \sigma | y) = \sigma^{-n-2} \exp\left(-\frac{1}{2\sigma^2}[(n-1)s^2 + n(\bar{y} - \mu)^2]\right)$$

where:

Equation 29 $$s^2 = \frac{1}{n-1} \sum_{i=1}^{n} (y_i - \bar{y})^2$$

is the sample variance of the elements y_i of the vector y of n independent observations. Using the product rule on the joint posterior it can be written as:

Equation 30 $$p(\mu, \sigma | y) = p(\mu | \sigma^2, y)\, p(\sigma^2 | y)$$

Thus, in order to create samples from the joint posterior distribution first the variance is sampled from scaled inverse *chi*-squared density,

Equation 31 $\sigma^2|y \sim Inv - X^2(n-1, s^2)$

which is followed by the sampling of μ from a normal distribution conditioned on the data and the sampled values for σ:

Equation 32 $\mu \mid \sigma^2, y \sim N\ (\bar{y}, \sigma^2/n)$

This conditional posterior distribution is used for further risk calculation and expresses the plausibility of the possible values of μ given the data and unknown σ.

9.2 Case Study of El Port de la Selva

9.2.1 Life Cycle Assessment

Primary Data for Infrastructure

Table 9-5: Inventory data for material infrastructure for LCA El Port de la Selva
ST = secondary treatment, TT = tertiary treatment, RWN = reclaimed water network; IP = infiltration ponds; WN = water network to Llanca; SW = seawater reverse osmosis

Material	Unit	Lifetime [a]	0. Status	1. Reuse A	2. Reuse B	3. Network	4. SWRO
Concrete	m^3	30	450 (ST)	450 (ST) 17.0 (TT) 92.0 (RWN)	450 (ST) 35.0 (TT) 92.0 (RWN)	450 (ST)	450 (ST) 134(SW)
Excavation volume	m^3	30	-	31.3 (TT)	101 (TT)	-	381 (SW)
Reinforced steel	t	30	73.5 (ST)	73.5 (ST) 2.91 (TT) 4.37 (RWN)	73.5 (ST) 6.37 (TT) 4.37 (RWN)	73.5 (ST)	73.5 (ST) 24.1 (SW)
Sand	t	30	-	1.15 (TT) 400 (IP)	400 (IP)	-	-
Anthracite	t	30	-	1.21 (TT)	-	-	-
Low alloyed steel	t	30		0.03 (TT)	1.38 (TT)	-	5.20 (SW)
Stainless steel	t	12	0.45 (ST)	0.45 (ST) 0.09 (TT)	0.45 (ST) 0.33 (TT)	0.45 (ST)	0.45 (ST) 1.24 (SW)
Copper	t	12	-	2.2E-3 (TT)	5.0E-3 (TT)	-	0.02 (SW)
PE	t	12 50	-	8.2E-3 (TT) 49.3 (RWN)	0.54 (TT) 49.3 (RWN)	-	2.02 (SW)
Cast iron	t	12 50	-	0.01 (TT)	0.25 (TT)	99.8 (WN)	0.93 (SW)
PVC-U	t	7	-	-	0.17 (TT)	-	0.34 (SW)
Epoxy resin	t	7	-	-	0.09 (TT)	-	0.16 (SW)
UV lamps	-	3	-	8 (TT)	-	-	-

Background data

Background processes for the schemes are modelled with dataset from ecoinvent v3.1 database [76] as described below (Table 9-6). Market datasets are used for all chemicals and materials as available. Additional transportation of chemicals and materials is considered.

Table 9-6: Ecoinvent dataset for background processes in LCA El Port de la Selva

Material	Ecoinvent v3.1 dataset	remark
Electricity	Electricity, voltage transformation, high to medium [ES]	Mix for Spain 2010
$FeCl_3$ (40 %)	iron (III) chloride production, product in 40% solution state [RoW]	
NaOCl (15 %)	sodium hypochlorite production, product in 15% solution state [RER]	
NaOH (50 %)	market for sodium hydroxide, without water, in 50% solution state [GLO]	
Citric Acid (40 %)	market for citric acid [GLO]	
H_2SO_4 (32 %)	market for sulfuric acid [GLO]	
$NaHSO_3$ (39 %)	market for sodium hydrogen sulfite [GLO]	
NH_4Cl (50 %)	market for ammonium chloride [GLO]	
Chlorine gas	market for chlorine, gaseous [RER]	
Lime (92 %)	market for lime, hydrated, packed [GLO]	
HCl (32 %)	market for hydrochloric acid, without water, in 30% solution state [RER]	
GAC fresh/reg.	electricity, high voltage, production mix [CN]; steam production, in chemical industry [RoW]; hard coal briquettes production [RoW]	
Antiscalant	dichloromethane production [RoW]; market for organophosphorus-compound, unspecified [GLO]: 50 % for each compound	for Wastewater RO
PACl (10 %)	aluminium hydroxide, at plant [RER]; electricity, medium voltage, at grid [ES]; hydrochloric acid, 30% in H2O, at plant [RER]	
Concrete	Market for concrete, for de-icing salt contact [GLO]	Waste treatment included
Excavation volume	Excavation hydraulic digger [RER]	
Reinforced steel	Reinforced steel production [RoW]	
Sand	silica sand production [RoW]	
Anthracite	market for hard coal [WEU]	Waste treatment included
Low alloyed steel	Steel production, low-alloyed, hot rolled [RoW]	
Stainless steel	Steel production, electric, chromium steel 18/8 [RoW]	
Copper	Copper production, primary [RoW]	
PE	Polyethylene production, low density, granulate [RER]	Waste treatment included
Cast iron	Cast iron production [RoW]	
PVC-U	Market for polyvinylchloride, bulk polymerized	
Epoxy Resin	Market for epoxy resin, liquid [GLO]	
UV lamps	Flat glass production, uncoated [RER]; steel production low-alloyed, hot rolled [RER], copper production, primary [RoW], market for mercury [GLO]	Mixture of materials

Inventory for Water Impact Index

A detailed inventory on the water quality index including concentrations and reference concentration for the reuse schemes is shown in Table 9-7. Reference concentrations are taken from EC Directive 2008/105/EC [143].

Table 9-7: Reference concentration, effluent concentrations and water quality index for different parameters in LCA El Port de la Selva

	SS	COD	TN	TP
reference concentration (C_{ref}) [mg/L]	35.0	30.0	13.7	0.2
'Reuse A'				
Concentration summer (C) [mg/L]	2.3	56.7	23.7	4.3
C_{ref}/C summer (WQI)	15.23	0.53	0.58	**0.05**
Concentration winter (C) [mg/L]	2.1	55.5	7.7	1.2
C_{ref}/C winter (WQI)	16.67	0.54	1.78	**0.17**
'Reuse B'				
Concentration summer (C) [mg/L]	0.3	53.6	20.4	3.6
C_{ref}/C summer (WQI)	116.67	0.56	0.67	**0.06**
Concentration winter (C) [mg/L]	0.01	1.0	0.6	0.1
C_{ref}/C winter (WQI)	3500.00	30.00	22.80	2.00 **(1.00)**

9.2.2 Risk assessment

Table 9-8: Results of screening level monitoring campaign in El Port de la Selva

Parameter	Quantity [ng/L]	LOQ	Selected	
Naphtalene	<LOQ	8.54		PAHs
Acenaphthene	0.66	0.33		
Fluorene	1.55	0.75		
Acenaphthylene	1.40	0.09		
Phenantrene	<LOQ	1.64		
Anthracene	1.05	0.12		
Fluoranthene	1.05	0.36		
Pyrene	2.04	0.47		
Benzo[a]anthracene	0.12	0.05		
Chrysene	0.24	0.07		
Benzo(b)fluoranthene	0.09	0.04		
Benzo(k)fluoranthene	<LOQ	0.03		
Benzo(a)pyrene	0.32	0.03		
Indeno(1,2,3-cd)pyrene	0.13	0.05		
Benzo(g,h,i)]perylene	0.13	0.05		
Dibenzo[a,h]anthracene	<LOQ	0.03		
BDE-28	<LOQ	0.01		PBDEs
BDE-47	0.07	0.02		
BDE-99	0.07	0.01		

Continuation of Table 9-8

Parameter	Quantity [ng/L]	LOQ	Selected	
BDE-100	0.02	0.01		PBDEs
BDE-153	<LOQ	0.01		
BDE-154	<LOQ	0.01		
BDE-183	0.01	0.01		
BDE-197	<LOQ	0.01		
BDE-209	1.46	0.03		
1,2,3-Trichlorobenzene	0.28	0.01		OCLs
1,2,4-Trichlorobenzene	0.46	0.01		
1,2,5-Trichlorobenzene	0.31	0.01		
Hexachlorobutadiene	<LOQ	0.35		
Pentachlorobenzene	0.06	0.05		
Hexachlorobenzene	0.11	0.05		
α-Hexachlorocyclohexane	0.26	0.05		
β-Hexachlorocyclohexane	0.52	0.02		
γ-Hexachlorocyclohexane	1.66	0.05		
δ-Hexachlorocyclohexane	<LOQ	0.02		
o,p'-DDE	<LOQ	0.03		
p,p'-DDE	0.13	0.04		
o,p'-DDD	<LOQ	0.03		
p,p'-DDD	0.07	0.05		
o,p'-DDT	<LOQ	0.03		
p,p'-DDT	<LOQ	0.05		
Aldrin	<LOQ	0.01		
Isodrin	<LOQ	0.01		
Dieldrin	0.40	0.05		
Endrin	<LOQ	0.04		
α-Endosulfan	<LOQ	0.05		
β-Endosulfan	<LOQ	0.19		
Endosulfan sulphate	<LOQ	0.1		
Nonylphenols	<LOQ	30.00		NP
Di(2-ethylhexyl)phathalate	222.00	120.00		DEHP
Chloroalkanes,	<LOQ	20.00		C10-13
Hexabromocyclodecane	<LOQ	0.60		HBCD
Isoproturon	1.17	0.08		Pesicides
Diuron	492.80	1.54	x	
Trifluralin	<LOQ	0.09		
Desethyl-s-atrazine (DEA)	<LOQ	0.19		
Dimethoate	15.10	2.66		
Simazine	1.95	0.14		
Atrazine	<LOQ	0.14		
Terbuthylazine	11.20	0.04		
Diazinon	3.17	0.17		

Continuation of Table 9-8

Parameter	Quantity [ng/L]	LOQ	Selected	
Alachlor	<LOQ	0.14		Pesicides
Terbutryn	163.50	1.33	x	
Metolachlor	0.16	0.02		
Chlorpyrifos	5.08	0.09		
4,4'-Diclorobenzophenone	<LOQ	0.04		
Heptachlor epoxide B	<LOQ	0.03		
Cholorphenvinfos	<LOQ	0.08		
Cybutrine	3509.00	0.42	x	
Aclonifen	<LOQ	0.11		
Quinoxyfen	<LOQ	0.21		
Dicofol p,p	11.12	1.01	x	
Cypermethrin	<LOQ	2.90		
Benzene	13.00	2.00		VOCs
1,2-dichloroethane	<LOQ	1.00		
Trichloromethane (chloroform)	2.00	1		
Trichloroethylene	<LOQ	1.00		
Carbon tetrachloride	<LOQ	1		
Tetrachoroethylene	5.00	1.00		
Perfluorobutanoic acid	10.60	0.50		PFCA
Perfluoropentanoic acid	<LOQ	5.00		
Perfluorohexanoic acid,	<LOQ	0.50		
Perfluoroheptanoic acid,	<LOQ	0.50		
Perfluorooctanoic acid,	7.50	1.00		
Perfluorononaoic acid,	5.00	0.50		
Perfluorodecanoic acid,	8.00	5.00		
Perfluorobutanesulfonic acid	<LOQ	5.00		PFSA
Perfluorohexanesulfonic acid,	<LOQ	0.50		
Perfluorooctanesulfonic acid,	<LOQ	5.00		

9.3 Case Study of Braunschweig

9.3.1 Life Cycle Assessment

Primary Data for Infrastructure

Table 9-9: Inventory data for material infrastructure in the LCA Braunschweig

	Unit	Lifetime [years]	0a. reference	0b. status	1a. no reuse	1b. irrigation on demand	2a. UV-disinfection	2b. PFA-disinfection	3a. DLD-NR w HR	3b. DLD-NR w/o HR
WWTP baseline										
Concrete	m³	35	23 358							
Reinforced steel	t	35	3 183							
Asbestos cement	t	80	8.6							
PE	t	30	0.84							
PVC	t	30	0.84							
Stainless steel	t	14	253							
Cast iron	t	14	67							
Irrigation system (and water distribution network)										
Concrete	m³	80	7 273	7 273	-	7 273	7 273	7 273	7 273	7 273
Reinforced steel	t	80	1 000	1 000	-	1 000	1 000	1 000	1 000	1 000
Asbestos cement	t	80	45	45	-	45	45	45	45	45
PE	t	40	19.5	19.5	-	19.5	19.5	19.5	19.5	19.5
PVC	t	40	163	163	-	163	163	163	163	163
Stainless steel	t	25	0.35	0.35	-	0.35	0.35	0.35	0.35	0.35
Low alloyed steel	t	25	1.85	1.85	-	1.85	1.85	1.85	1.85	1.85
Disinfection system										
Concrete	m³	30	-	-	-	-	42	32	-	-
Reinforced steel	t	30	-	-	-	-	4.32	-	-	-
PE	t	30	-	-	-	-	-	0.50	-	-
PVC	t	30	-	-	-	-	0.51	-	-	-
Stainless steel	t	12	-	-	-	-	2.42	0.20	-	-
Low alloyed steel	t	12	-	-	-	-	-	0.20	-	-
Cast iron	t	12	-	-	-	-	2.42	-	-	-
Plastics	t	12	-	-	-	-	-	0.20	-	-
UV lamps	1	3	-	-	-	-	135	-	-	-
DLD-NR system										
Concrete	m³	35	-	-	-	-	-	-	450	450
Reinforced steel	t	35	-	-	-	-	-	-	37.0	37.0
PE	t	20	-	-	-	-	-	-	39.0	39.0
Stainless steel	t	14			-	-	-	-	18.6	18.6

Background data

Background processes for the schemes are modelled with dataset from ecoinvent v3.1 database [76] as described below (Table 9-10). Market datasets are used for all chemicals and materials as available. Additional transportation of chemicals and materials is considered.

Table 9-10: Ecoinvent dataset for background processes in the LCA Braunschweig

Material	Ecoinvent v3.1 dataset	remark
Electricity	Market for electricity, medium voltage [DE]	Mix for Germany 2010
$FeCl_2$ (10 %)	iron(II) chloride production [GLO]	
Polymer	market for acrylonitrile [GLO], water	
$MgCl_2$ (30 %)	Potassium hydroxide, heat production, natural gas, at industrial furnace low-NOx >100kW [Europe without Switzerland]	
NaOH (50 %)	market for sodium hydroxide, without water, in 50% solution state [GLO]	
H_2SO_4 (40 %)	sulfuric acid production [RER]	
H_2O_2 (50 %)	hydrogen peroxide production, product in 50% solution state [RER]	
Formic acid	market for formic acid [RER]	
Natural gas	market for heat, district or industrial, natural gas [Europe without Switzerland]	
Concrete	Market for concrete, for de-icing salt contact [GLO]	Waste treatment included
Reinforced steel	Reinforced steel production [RoW]	
Low alloyed steel	Steel production, low-alloyed, hot rolled [RoW]	
Stainless steel	Steel production, electric, chromium steel 18/8 [RoW]	
PE	Polyethylene production, low density, granulate [RER]	Waste treatment included
Cast iron	Cast iron production [RoW]	
PVC	Market for polyvinylchloride, bulk polymerized	Waste treatment included
Epoxy resin	Market for epoxy resin, liquid [GLO]	
UV lamps	Flat glass production, uncoated [RER]; steel production low-alloyed, hot rolled [RER], copper production, primary [RoW], market for mercury [GLO]	Mixture of materials, Waste treatment included
Asbestos cement	market for asbestos, crysotile type [GLO]	
Injection moulded plastics	injection moulding [RER]	
Transportation	market for transport, freight, lorry 16-32 metric ton, EURO5 [GLO], transport, freight train [DE]	Depending on transportation distance

Inventory for Water Impact Index

A detailed inventory on the water quality index including concentrations and reference concentration for the scenarios is shown in Table 9-11. Reference concentrations are taken from EC Directive 2008/105/EC [143].

Table 9-11: Reference concentration, effluent concentrations and water quality index for different parameters for the LCA Braunschweig

	SS	COD	TN	TP	Cd	Cr	Cu	Hg	Ni	Pb	Zn
reference concentration (C_{ref}) standard parameters [mg/L]; heavy metals [µg/L]	35.0	30.0	13.7	0.2	0.45	3.4	1.4	0.05	20.0	7.2	7.8
‘reference’, ‘no reuse’, ‘irrigation on demand’, ‘UV-disinfection’, ‘PFA-disinfection’ *scenarios which do not effect water quality regarding standard parameters*											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	9.3	42.8	11.9	1.0	0.13	2.4	8.3	0.04	3.4	2.3	17.6
C_{ref}/C (WQI)	3.76	0.70	1.15	**0.19**	3.39	1.39	**0.17**	1.34	5.83	3.09	0.44
‘status’											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	9.3	42.8	11.1	1.0	0.13	2.5	8.5	0.04	3.5	2.4	18.1
C_{ref}/C (WQI)	3.76	0.70	1.23	**0.19**	3.36	1.37	**0.17**	1.29	5.73	3.04	0.43
‘DLD-NR w HR’, ‘DLD-NR w/o HR’											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	9.3	42.8	10.7	1.1	0.13	2.5	8.5	0.04	3.5	2.4	18.3
C_{ref}/C (WQI)	3.76	0.69	1.27	**0.19**	3.37	1.36	**0.16**	1.27	5.72	3.03	0.43

9.4 Case Study of OldFord Water Recycling Plant

9.4.1 Life Cycle Assessment

Primary Data for Infrastructure

A rough estimation on infrastructure was conducted by implementing data from a carbon footprint study by Thames Water [98] into this LCA as possible. Additional infrastructure e.g. for membrane production has not been considered. Estimates for the 'No Reuse' scenario are based on previous KWB studies [45] and linearly scaled to the volume to be treated. Infrastructure for drinking water treatment plant has not been considered. Finally, the rough estimation just considers tanks and technical aggregates in both WWTPs. It becomes apparent that the required material per m³ treated wastewater is considerably higher for a small WWTP such as 'OldFord' (1 500 pe) compared to a larger WWTP system in London.

Table 9-12: Inventory data for material infrastructure for the LCA OldFord Water Recycling Plant

Material	Unit	Lifetime [years]	1. OldFord	2. No Reuse
Concrete	m^3	30	559	187
Excavation volume	m^3	30	2111	595
Reinforced steel	t	30	105.7	25.9
Low alloyed steel	t	30	14.4	-
Stainless steel	t	12	11.2	2.5

Background data

Background processes for the schemes are modelled with dataset from ecoinvent v3.1 database [76] as described below (Table 9-13). Market datasets are used for all chemicals and materials as available. Additional transportation of chemicals and materials is considered.

Table 9-13: Ecoinvent dataset for background processes for the LCA OldFord Water Recycling Plant

Material	Ecoinvent v3.1 dataset	remark
Electricity	electricity voltage transformation from high to medium voltage [GB]	Mix for UK 2010
PACl (10 % Al)	hydrochloric acid, 30% in H2O, at plant [RER]; aluminium hydroxide, at plant [RER]; electricity voltage transformation from high to medium voltage [GB]	
NaOCl (15 %)	sodium hypochlorite production, product in 15% solution state [RER]	
Citric Acid (40 %)	market for citric acid [GLO]	
Oxygen (100%)	market for oxygen, liquid [RoW]	
H_3PO_4 (85 %)	market for phosphoric acid, industrial grade, without water, in 85% solution state [GLO]	
GAC fresh/reg.	electricity, high voltage, production mix [CN]; steam production, in chemical industry [RoW]; hard coal briquettes production [RoW]	
Concrete	Market for concrete, for de-icing salt contact [GLO]	Incl. waste treatment
Excavation volume	Excavation hydraulic digger [RER]	
Reinforced steel	Reinforced steel production [RoW]	
Low alloyed steel	Steel production, low-alloyed, hot rolled [RoW]	
Stainless steel	Steel production, electric, chromium steel 18/8 [RoW]	

Inventory for Water Impact Index

A detailed inventory on the water quality index including concentrations and reference concentration for the scenarios is shown in Table 9-14. Reference concentrations are taken from EC Directive 2008/105/EC [143].

Table 9-14: Reference concentration, effluent concentrations and water quality index for different parameters for the LCA OldFord Water Recycling Plant

	SS	COD	TN	TP	Cd	Cr	Cu	Hg	Ni	Pb	Zn
reference concentration (C_{ref}) standard parameters [mg/L]; heavy metals [µg/L]	35.0	30.0	13.7	0.2	0.45	3.4	1.4	0.05	20.0	7.2	7.8
Effluent 'OldFord'											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	2.22	17.4	15.8	2.1	0.18	1.3	7.8	0.12	1.8	0.38	18.5
C_{ref}/C (WQI)	1.73	15.8	0.87	0.09	2.47	2.58	0.18	0.42	10.9	19.1	0.42
Effluent 'No reuse'											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	Data is confidential										
C_{ref}/C (WQI)	0.05										
Withdrawal from the reservoir (DWTP influent)											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	Data is confidential										
C_{ref}/C (WQI)	0.45										

9.5 Case Study of Sabadell

9.5.1 Life Cycle Assessment

Background data

Background processes for the schemes are modelled with dataset from Ecoinvent v3.1 database [76] as described below (Table 9-15). Market datasets are used for all chemicals and materials as available. Additional transportation of chemicals and materials is considered.

Table 9-15: Ecoinvent dataset for background processes for the LCA Sabadell

Material	Ecoinvent v3.1 dataset	remark
Electricity	Electricity, medium voltage [ES]	Mix for Spain 2010
NaOCl (15 %)	sodium hypochlorite production, product in 15% solution state [RER]	
Concrete	Market for concrete, for de-icing salt contact [GLO]	Waste treatment included
Excavation volume	Excavation hydraulic digger [RER]	
Reinforced steel	Reinforced steel production [RoW]	
Sand	silica sand production [RoW]	
Low alloyed steel	Steel production, low-alloyed, hot rolled [RoW]	
Stainless steel	Steel production, electric, chromium steel 18/8 [RoW]	
HDPE	Polyethylene, high density, granulate [RER]	Waste treatment included
PVC	Market for polyvinylchloride, bulk polymerized	
Copper	Copper production, primary [RER]	
UV Lamps	Flat glass production, uncoated [RER]; steel production low-alloyed, hot rolled [RER], copper production, primary [RoW], market for mercury [GLO]	Mixture of materials
Sprinkler system	Polyethylene, high density, granulate [RER], Extrusion, plastic pipes [RER], Polyethylene, low density, granulate [RER], Polypropylene, granulate [RER], Polyvinylidenchloride, granulate [RER], Steel, low-alloyed [RER], Drawing of pipe, steel [RER], Aluminium, primary, ingot [RoW], Section bar extrusion, aluminium [RER]	

Energy Balance at Sabadell WWTP

Energy consumption and generation at Sabadell WWTP is detailed in Table 9-16.

Table 9-16: Energy Balance at Sabadell WWTP

Material consumption	Comments	Amount	Units	Amount	Units by m^3 of water released)
Pretreatment	Direct Energy demand	716,00	kWh/day	0,03	kWh/m^3
MBR	Direct Energy demand	37322,1	kWh/day	1,35	kWh/m^3
UV-Cl	Direct Energy demand	1,50	kWh/day	0,03	kWh/m^3
Biogas Production	Electricity demand	3432,00	kWh/day	0,12	kWh/m^3
Total WWTP	**Direct energy consumption**	**41471,60**	**kWh/day**	**1,53**	**kWh/m3**
Biogas Production	Total electricity produced	12242,00	Kwh/day	0,44	kWh/m^3
Biogas Production	Losses	945,00	kWh/day	0,03	kwh/m3
Biogas Production	Energy consumed in WWTP	4347,00	kWh/day	0,16	kwh/m3
Biogas Production	Energy delivered to electrical grid	6950,00	kWh/day	0,25	kwh/m3
Total	**Net energy production considering losses**	**11297,00**	**kWh/day**	**0,41**	**kWh/m3**
Total	**Energy Balance (Direct energy consumption – net energy production considering losses)**	**30174,60**	**kWh/day**	**1,12**	**kWh/m3**

Water Impact Index Inventory

A detailed inventory on the water quality index including concentrations and reference concentration for the scenarios is shown in Table 9-17.

Table 9-17: reference concentration, effluent concentrations and water quality index for different parameters for Sabadell

	COD	TN	TP	Cr	Cu	Ni	Pb	Zn
reference concentration (C_{ref}) standard parameters [mg/L]; heavy metals [µg/L]	30	50	0.2	3.4	1.4	20	7.2	7.8
Influent 'Sabadell'								
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	670.0	77.0	11.0	2.9	39.0	19.7	1.3	110.1
C_{ref}/C (WQI)	0.04	0.6	**0.02**	1.2	0.04	1.0	5.5	0.07
Effluent 'Sabadell'								
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	50.0	27.4	0.2	1.3	3.7	93.5	0.3	191.3
C_{ref}/C (WQI)	0.6	1.8	2.5	2.7	0.4	0.2	23	**0.04**

9.5.2 Risk Assessment methodology

Effects assessment

Effects assessment is the estimation of the relationship between dose or level of exposure to a substance, and the incidence and severity of an effect [144]. This chapter shows dose-response parameters and models used for both chemical and microbiological hazards identified in this project.

1) Toxicity parameters for chemical hazards

Different dose-response relationships may be found if a substance produces different toxic effects. For example, a short term exposure to high concentration of high toxic element may produce lethal effects, whereas cancer may be induced as a result of long-term exposure to relatively low concentrations.

Toxicity values for non-carcinogenic effects are expressed as reference doses for oral exposures and reference concentrations for air exposure via inhalation. Reference dose (RfD) is an estimate of a daily oral exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It is expressed as mg/kg·d. Reference concentration (RfC) is an estimate of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. It is expressed as µg/m^3.

Chronic exposures are defined as repeated exposures for more than approximately 10 % of the life span in humans whereas subchronic exposures are defined as repeated exposures for more than 30 days, and up to approximately 10 % of the life span in humans. In this study, chronic toxicity data will be considered for adults and subchronic toxicity data for children (exposure time of 6 years).

Toxicity values for carcinogenic effects are expressed as slope factors for oral exposures and inhalation unit risks for air exposure via inhalation. Slope Factor (SF) is the plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime $(mg/kg\text{-}d)^{-1}$. Inhalation Unit Risk (IUR) is defined as the upper-bound excess lifetime cancer risk estimated to result from continuous exposure to an agent at a concentration of 1 µg/m^3 in air. It is expressed as $(\mu g/m^3)^{-1}$.

Toxicity parameters used as dose-response assessment for chemical hazards were obtained from USEPA database RAIS (Risk Assessment Information System) through its web page [145], which compiles the available toxicity data for the risk assessment of contaminated sites. Toxicity parameters used in this study are shown in Table 9-18. Only compounds that were detected in the sampling campaigns, and thus, were considered in the risk assessment are shown in this table.

Dermal toxicity values (RfD_D and SF_D) were calculated from oral toxicity values and the Gastrointestinal Absorption Factor (GAF) using the following equations [111]

Equation 33 $$\mathrm{RfD_D} = \mathrm{RfD_O} \cdot \mathrm{GAF}$$

Equation 34 $$\mathrm{SF_D} = \frac{\mathrm{SF_O}}{\mathrm{GAF}}$$

GAF values are also shown in Table 9-18.

Table 9-18: Toxicity parameters for chemical hazards used in this study and Gastrointestinal Absorption Factor

Contaminant	Non-carginogenic effects				Carcinogenic effects		
	Chronic Oral Reference Dose (mg/kg·day)	Subchronic Oral Reference Dose (mg/kg·day)	Chronic Inhalation Reference Conc. (mg/m^3)	Subchronic Inhalation Reference Conc. (mg/m^3)	Oral Slope Factor (mg/kg day)$^{-1}$	Inhalation Unit Risk (µg/m^3)$^{-1}$	Gastroinestinal Absoption Factor
	RfD_O	$sRfD_O$	RfC_I	$sRfC_I$	SF_O	IUR_I	GAF
Metals							
Antimony	4,00E-04	4,00E-04		4,00E-04			1,50E-01
Arsenic	3,00E-04	5,00E-03	1,50E-05		1,50E+00	4,30E-03	1,00E+00
Cadmium	5,00E-04		1,00E-05			1,80E-03	5,00E+00
Chromium (III)				1,00E-04			1,30EE-02
Copper	4,00E-02	1,00E-02					1,00E+00
Iron	7,00E-01	7,00E-01					1,00E+00
Lead	3,50E-03				8,50E-03	1,20E-05	1,00E+00
Manganese	2,40E-02		5,00E-05				4,00E-02
Mercury	1,60E-04		3,00E-04	3,00E-04			1,00E+00
Nickel	2,00E-02	2,00E-02	9,00E-05	2,00E-04		2,60E-04	4,00E-02
Selenium	5,00E-03	5,00E-03	2,00E-02				1,00E+00
Zinc	3,00E-01	3,00E-01					1,00E+00
Halogenated Solvents							
Bromoform	2,00E-02	3,00E-02			7,90E-03	1,10E-06	1,00E+00
Bromodichloromethane	2,00E-02	8,00E-03		2,00E-02	6,20E-02	3,70E-05	1,00E+00
Chloroform	1,00E-02	1,00E-01	9,77E-02	2,44E-01	3,10E-02	2,30E-05	1,00E+00
Dibromochloromethane	2,00E-02	7,00E-02			8,40E-02		1,00E+00

1) Dose-response models for microbiological hazards

Dose-response models for the microbiological hazards were obtained from literature [55, 110, 146]. They are shown in Table 9-19. Only microbial agents detected in the monitoring campaigns are indicated.

Table 9-19: Dose-response parameters for microbial hazards used in this study [55, 147, 148]

	Beta-poison			Exponential		DALY per case of disease	Infection ratio
Pathogen	Alpha	Beta	N50	k	r		
Bacteria							
Campylobacter jejeuni	0,145		896				
Protozoa							
Cryptosporidium spp.				238	0,0042	1,60E-02	0,3
Giardia spp				50,23	0,0199	1,50E-03	0,3
Virus							
Norovirus	0,04	0,055			0,0069	9,00E-04	0,8
Enterovirus	0,67	47,9			0,009102	1,00E-02	0,05
Adenovirus				6,07E-1	0,4172	1,00E-04	0,5
Rotavirus	0,2531	0,422				1,40E-02	0,05

Exposure assessment

Exposure assessment for cancer risk and non-cancer risk follows different equations since the daily exposure dose is calculated based on the total lifetime for carcinogenic assessment and on the total exposure time for non-cancer risk).

2) Oral ingestion

Ingestion exposure considers ingestion of reclaimed water accidentally or on purpose. It is calculated as oral ingestion dose using the concentration of the chemicals in reclaimed water and the daily ingestion rate, as follows:

Oral ingestion dose (ODose) for exposure to noncarcinogenic contaminants:

Equation 35
$$ODose, i\left(\frac{mg}{kg*day}\right) = \frac{(Cw,i*IR*EF)}{(BW*365\ days/year)}$$

Where;

- Cw,i: Concentration of contaminant (i) in water (mg/L)
- IR: Ingestion rate (L/event)
- EF: Exposure frequency (events/year)
- BW: Body weight (kg)

Oral ingestion dose (ODose') for exposure to carcinogenic contaminants:

Equation 36
$$ODose', i\left(\frac{mg}{kg*day}\right) = \frac{(Cw,i*IR*EF*ED)}{(BW*365\ days/year*AT)}$$

Where;

- Cw,i: Concentration of contaminant (i) in water (mg/L)
- IR: Ingestion rate (L/event)
- EF: Exposure frequency (events/year)

- ED: Exposure Duration (year)
- AT: Average lifetime (years)
- BW: Body weight (kg)

2) Dermal contact

Substances may have the ability to penetrate intact skin and become absorbed into the body. Dermal exposure can be influenced by the amount and concentration of the substance, the area of skin exposure and the duration and frequency of exposure. Hand transfer of contamination to other parts of the body is an important source of skin exposure. Dermal exposure is expressed in terms of the mass of contaminant per unit surface area of the skin exposed and calculated considering the concentration of the contaminant in water and a skin permeability constant that is contaminant-specific, using the following equations:

Dermal contact dose (DDose) for exposure to noncarcinogenic contaminants:

Equation 37
$$DDose, i \left(\frac{mg}{kg*day}\right) = \frac{(C_W, i*1000*SA*Kp*ET*EF)}{(BW*365\ days/year)}$$

Where;

- Cw: Concentration of contaminant (i) in water (mg/L)
- SA: Total skin surface area (cm^2)
- Kp: Permeability constant (cm/h)
- ET: Exposure time (h/event)
- EF: Exposure frequency (events/year)
- BW: Body weight (kg)

Dermal contact dose (DDose’) for exposure to carcinogenic contaminants:

Equation 38
$$DDose', i \left(\frac{mg}{kg*dia}\right) = \frac{(C_W, i*1000*SA*Kp*ET*EF*ED)}{(365\ days/year*BW*AT)}$$

Where;

- Cw: Concentration of contaminant (i) in water (mg/L)
- SA: Total skin surface area (cm^2)
- Kp: Permeability constant (cm/h)
- ET: Exposure time (h/event)
- EF: Exposure frequency (d/event)
- ED: Exposure duration (year)
- AT: Average lifetime (years)
- BW: Body weight (kg)

3) Inhalation of volatile contaminants

Exposure by inhalation is expressed as the concentration of the substance in the breathing zone. Its calculation is based on the assessment of an equivalent concentration (EC) of each contaminant in air for the exposure period [147]. The equivalent concentration is calculated as shown in the following equations:

Equivalent concentration (EC) for inhalation exposure to non-carcinogenic contaminants:

Equation 39
$$EC, i \left(\frac{mg}{m^3}\right) = \frac{(Ca*ET*EF)}{\left(365\frac{days}{year}*24\frac{hours}{day}\right)}$$

Where;

- Ca: Concentration in air (mg/m^3)
- ET: Exposure time (hours/event)
- EF: Exposure frequency (events/year)

Equivalent concentration (EC') for inhalation exposure to carcinogenic contaminants:

Equation 40 $$EC',i\left(\frac{mg}{m^3}\right)=\frac{(Ca*ET*EF*ED)}{\left(AT*365\frac{days}{year}*24\frac{hours}{day}\right)}$$

Where;

- Ca: concentration in air (mg/m^3)
- ET: Exposure time (h/event)
- EF: Exposure frequency (events/year)
- ED: Exposure duration (years)
- AT: Average lifetime (years)

Concentration of the contaminant in air is required for the calculation of the equivalent concentration. Transport models are used to estimate the contaminants concentration in the air from the contaminant concentration in reclaimed water using a volatilization factor (VF) as described in Martí *et al.*, (2014):

Equation 41 Air Concentration = Water Concentration · Volatilization Factor

Volatilization factors are scenario-specific and depend on the air-water surface transfer (flat or spherical) and on the indoor or outdoor situation. In outdoor scenarios, wind is the main factor for the final air pollutants concentration. For indoor scenarios, the key factor is the rate of air room removal. The outdoor scenarios in this study are the irrigation of public gardens, street cleaning and irrigation of golf courses. Toilet flushing is the only indoor scenario considered. Volatilization factors were calculated as describen in [149]. Table 9-20 shows the volatilization factors obtained for each scenario.

Table 9-20: Volatilization factor used for QCRA (L/m3)

Contaminant	Public garden irrigation	Street cleaning	Golf course irrigation	Toilet flushing
Bromodichloromethane	3,03E-02	1,00E-02	**3,78E-02**	3,08E-02
Bromoform	2,74E-02	7,02E-03	**3,43E-02**	2,86E-02
Chloroform	3,10E-02	1,14E-02	**3,87E-02**	3,15E-02
Dibromochloromethane	2,86E-02	8,21E-03	**3,58E-02**	2,96E-02
Mercury	2,17E-02	1,01E-02	**2,71E-02**	2,21E-02

Chemical properties of contaminants

The following table shows the chemical properties used for exposure assessment.

Table 9-21: Chemical properties

Chemical	CAS	Diffusivityin Air (m2/s)	Diffusivityin Water (m2/s)	Unitless Henry`s Law Constant	Skin Permeability Constant (cm/hr)	RAGS Part E Dermal Absorption Factor	RAGS Part E Gastrointestinal Absorption Factor
Antimony	7440 36 0				1,00E-03		1,50E-01
Arsenic	7440 38 2				1,00E-03	3,00E-02	1,00E+00
Bromodichloromethane	75 27 4	5,63E-06	1,07E-09	8,67E-02	4,02E-03		1,00E+00
Bromoform	75 25 2	3,57E-06	1,04E-09	2,19E-02	2,35E-03		1,00E+00
Cadmium	7440 43 9				1,00E-03	1,00E-03	5,00E-02
Chloroform	67 66 3	7,69E-06	1,09E-09	1,50E-01	6,83E-03		1,00E+00
Chromium	7440 47 3				1,00E-03		1,30E-02
Copper	7440 50 8				1,00E-03		1,00E+00
Dibromochloromethane	124 48 1	3,66E-06	1,06E-09	3,20E-02	2,89E-03		1,00E+00
Iron	7439 89 6				1,00E-03		1,00E+00
Lead	7439 92 1				1,00E-04		1,00E+00
Manganese	7439 96 5				1,00E-03		4,00E-02
Mercury	7439 97 6	3,07E-06	6,3E-10	5,20E+01	1,00E-03		1,00E+00
Nickel	7440 02 0				2,00E-04		4,00E-02
Selenium	7782 49 2				1,00E-03		1,00E+00
Zinc	7440 66 6				6,00E-04		1,00E+00

Quantitative chemical risk assessment theoretical background

Risk is the estimation of the incidence and severity of the adverse effects in a human population due to actual or predicted exposure to a substance (Leeuwen and Vermeire 2007). Risk assessment includes information on exposure to contaminants (exposure assessment) and toxicity of contaminants (effects assessment).

The equation for calculating systemic toxicity (i.e. noncarcinogenic hazard) for oral exposure and dermal exposure is:

Equation 42 $$HQ = \frac{Dose,i}{RfD,i}$$

Where *Dose* could be oral dose (*ODose*) or dermal dose (*DDose*), and *i* is the specific contaminant considered.

In case of inhalation exposure, the equation to calculate the systemic hazard is:

Equation 43 $$HQ = \frac{EC,i}{RfC,i}$$

The equation for calculation excess lifetime cancer risk for oral exposure and dermal exposure is:

Equation 44 $$Risk = SF,i * Dose',i$$

Where Dose’ could be oral dose (*ODose’*) or dermal dose (*DDose’*), and i is the specific contaminant considered.

In case of inhalation exposure, the equation to calculate the cancer risk is:

Equation 45 $$Risk = IUR,i * EC',i$$

Both hazard quotient and cancer risk may be calculated for each exposure route and each contaminant. For each exposure route, the sum of the contribution of each contaminant will give the risk of the exposure route. The total risk includes the contribution of all contaminants and all exposure routes for a specific receptor.

Threshold levels for unacceptable risk are described in Table 9-22. Thus, when HQ is above 1, the non-cancer risk will be unacceptable, and when Risk is above 10^{-5}, the cancer risk will be unacceptable.

Table 9-22: Threshold levels for unacceptable risks and hazards.

Effects	Threshold level
Non-cancer hazards	1
Cancer Risks	10^{-5}

9.5.3 Monitoring campaing results for risk assessment

Available data and information

Monthly analyses of the chemical and microbiological composition of reclaimed water produced in Riu Sec WWTP are being performed by CASSA following Spanish Regulation for water reuse [150]. The required parameters are intestinal nematodes, *Escherichia coli* and *Legionella sp*. for microbial pathogen. Average values for these parameters at the WWTP after disinfection are shown in Table 9-23. CASSA is also characterizing the reclaimed water at the irrigation point. Table 9-24 shows the results for this monitoring. Reclaimed water quality in terms of macro contaminants is shown in Table 9-25.

Table 9-23: Reclaimed water composition after disinfection
Values from 2015, 12 samples

Parameter	Units	Results	Spanish limit value (RD 1620/2007)
E. coli	ufc/100 mL	0	200
Turbidity	NFU	0,2-1,4	10
Suspended solids	mg/L	2-4	20
Conductivity	µS/cm	1246-1650	-
Nematode eggs	eggs/10L	0	1

Table 9-24: Reclaimed water composition at the irrigation network
Values from January 2014 to January 2016, 11 samples

Parameter	Units	Results	Spanish limit value (RD 1620/2007)
E. coli	ufc/100 mL	0	200
Turbidity	UNF	0,3-1,4	10
Suspended solids	mg/L	2-6	20
Conductivity	µS/cm	1112-2000	-
Nematode eggs	eggs/10L	0	1
Legionella	ufc/L	<25	100

Table 9-25: Water characterisation after disinfection
Sampling date: 21/09/2015

Parameter	Units	Results
pH	upH	7.97
COD	mg/L	<50
DBO_5	mg/L	5
Chloride	mg/l	174
Nitrate	mg/l	18
Sulphate	mg/l	79
Phosphate	mg/l	3
Calcium	mg/l	59,0
Potassium	mg/l	21,5
Magnesium	mg/l	11,1
Sodium	mg/l	131,0

Detection limits for compounds not detected in reclaimed water

Polycyclic Aromatic Hydrocarbons (PAHs) were analysed in the 1st and 2nd sample campings at WWTP effluent as is described in section 5.3.1. PAHs include the following compunds: naphthalene, acenaphthylene, acenaphthene, fluorine, phenanthrene, anthracene, fluoranthene, pyrene, benzo(a)anthracene, chrysene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, indeno(123cd)pyrene, dibenzo(ah)antracene, benzo(ghi)perylene. The detection limits for all compounds was 5 ng/l.

Polychlorinated biphenyls (PCBs) were analysed in the 1st and 2nd sample campings at WWTP effluent as is described in section 5.3.1. PCBs include the following compunds: PCB-28, PCB-52, PCB-101, PCB-118, PCB-153, PCB-138 and PCB-180. The detection limits for all compounds was 5 ng/l.

Volatile hydrocarbon fractions (BTEX) were analysed in the 1st and 2nd sample campings at WWTP effluent as is described in section 5.3.1. VPH-BTEX include the following compunds: benzene, toluene, etilbenzene, m,p-xylene, o-xylene. The detection limits for all compounds was 0.5 µg/l.

Chlorinated Pesticides were analysed in the 1st and 2nd sample campings at WWTP effluent as is described in section 5.3.1. The following compund were analyzed: pentachlorobenzene, alpha-HCB, hexachlorobenzene, gamma-HCH, delta-HCH, heptachlor, aldrin, isodrin, beta-Heptachlor peroxide, alpha-heptachlor peroxide, op-DDE, endosulphan-I, pp-DDE, dieldrin, op-DDD, endrin, endosulfan-II, pp-DDD, op-DDT, endrin aldehyde, endosulfan sulfate, pp-DDT. The detection limits for all compounds was 10 ng/l.

Results and detection limits for the chlorinated disinfection by-products

The following table shows the results for the chlorinated disinfection by-products and the detection limits of the compounds not detected.

Table 9-26: Halogenated compounds results

Halogenated compounds	1st S. C. 21-09-15 WWTP effluent (µg/l)	2nd S. C. 01-12-2015 WWTP effluent (µg/l)	3rd Sampling Campaign 26-02-2016			4th Sampling Campaign 04-05-2016		
			WWTP effluent (µg/l)	Regenerated effluent (µg/l)	Sprinkler point (µg/l)	WWTP effluent (µg/l)	Regenerated effluent (µg/l)	Sprinkler point (µg/l)
Ninyl Chloride	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Trichlorofluoromethane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,1-Dichloroethene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Dichloromethane	<5	< 5	< 5	< 5	< 5	< 5	< 5	< 5
t-1,2-Dichloroethene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,1-Dichloroethane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
c-1,2-Dichloroethene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Trichloromethane	79,52	81,47	7,29	9,44	16,89	495,4	357,8	46,1
1,1,1-Trichloroethane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Tetrachloromethane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,2-Dichloroethane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Trichloroethane	<0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5
1,2-Dichloropropane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Bromodichloro methane	18,01	63,48	2,48	3,11	3,5	165,2	126,9	13,73
c-1,3-Dichloropropene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
t-1,3-Dichloropropene	<2	< 2	< 2	< 2	< 2	< 2	< 2	< 2
1,1,2-Trichloroethane	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Tetrachloroethene	<0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5
Dibromochloromethane	3,04	28,87	<1	1,22	<1	54,93	47,71	2,78
Chlorobenzene	<0,5	< 0,5	<0,5	<0,5	<0,5	<0,5	<0,5	<0,5

Continuation of Table 9-26

Halogenated compounds	1st S. C. 21-09-15 WWTP effluent (µg/l)	2nd S. C. 01-12-2015 WWTP effluent (µg/l)	3rd Sampling Campaign 26-02-2016			4th Sampling Campaign 04-05-2016		
Tribromomethane	<2	4,3	<2	<2	<2	5	9,2	<2
1,1,2,2-Tetrachloroethane	<20	< 20	<20	<20	<20	<20	<20	<20
1,3-Dichlorobenzene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,4-Dichlorobenzene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,2-Dichlorobenzene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,3,5-Trichlorobenzene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
1,2,4-Trichlorobenzene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1
Hexachlorobutadiene	<0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5	< 0,5
1,2,3-Trichlorobenzene	<1	< 1	< 1	< 1	< 1	< 1	< 1	< 1

Microbiological results from monitoring campaigns

Table 9-27: Microbiological results for bacteria and protozoa before MBR and MBR effluent

			Before MBR				MBR effluent			
Parameter	Method	units	1st Sampling Campaign	2nd Sampling Campaign	3rd Sampling Campaign	4th Sampling Campaign	1st Sampling Campaign	2nd Sampling Campaign	3rd Sampling Campaign	4th Sampling Campaign
Campylo-bacter jejeuni	PE Special isolation	absence / presence	absence	< 1	n.a.	n.a.	absence	< 1	n.a.	n.a.
Clostriudium Perfrigens	A-E-PE_0048 Membrane filtration	cfu/100ml	4,70E+05	3,20E+05	3,00E+05	2,00E+04	5,30E+01	6,20E+01	1,50E+01	1,00E+00
Coliforms totals	A-E-PE_0061 Culture isolation	cfu/100ml	2,80E+07	3,80E+07	4,00E+07	4,70E+05	1,00E+04	7,60E+01	5,80E+01	8,00E+00
E. Coli	A-E-PE_0061Culture isolation	cfu/100ml	0	8,00E+06	3,00E+06	3,70E+05	0	7	7	1
Cryptospo-ridium spp.	immunofluorescence PE-B_MU/036	Oocysts / 1L	0	0	0	0	0	0	n.a.	n.a.
Giardia spp	immunofluorescence PE-B_MU/036	cysts / 1L	> 300	> 300	2700	3020	0	0	n.a.	n.a.

Table 9-28: Microbiological results for bacteria and protozoa in WWTP effluent

			WWTP effluent (regenerated water)			
Parameter	Method	units	1st Sampling Campaign	2nd Sampling Campaign	3rd Sampling Campaign	4th Sampling Campaign
Campylo-bacter jejeuni	PE Special isolation	absence / presence	absence	< 1	n.a.	n.a.
Clostriudium Perfrigens	A-E-PE_0048 Membrane filtration	cfu/100ml	7	0	0	0
Coliforms totals	A-E-PE_0061 Culture isolation	cfu/100ml	0	0	1	0
E. Coli	A-E-PE_0061Culture isolation	cfu/100ml	0	0	0	0
Cryptospo-ridium spp.	immunofluorescence PE-B_MU/036	Oocysts / 1L	0	0	n.a.	n.a.
Giardia spp	immunofluorescence PE-B_MU/036	cysts / 1L	0	0	n.a.	n.a.

Table 9-29: Microbiological results for virus

		MBR influent				MBR effluent				WWTP effluent			
Virus	Units	1st Campaign	2nd Campaign	3rd Campaign	4th Campaign	1st Campaign	2nd Campaign	3rd Campaign	4th Campaign	1st Campaign	2nd Campaign	3rd Campaign	4th Campaign
Adeno-virus	gen copy/L	3,76E+03	>6000	5,80E+07	8,68E+07	<300	<300	<300	<214	<300	<300	4,32E+02	<214
Entero-virus	gen copy/L	<6,63E+5	7,31E+07	5,53E+05	<6,63E+5	<4140	3,66E+04	<4140	<4140	<4140	<4140	<4140	<4140
Noro-virus-I	gen copy/L	1,78E+06	2,76E+07	1,94E+07	1,61E+06	<2000	<2000	<2000	<411	<2000	<2000	<2000	<411
Noro-virus-II	gen copy/L	7,63E+03	2,29E+08	2,93E+07	1,57E+06	<3000	<3000	6,26E+02	<2959	<3000	<3000	8,51E+01	<2959
Rota-virus	gen copy/L	1,63E+04	6,59E+07	1,97E+08	2,37E+06	<3870	<3870	<3870	<3873	<3870	<3870	<3870	<3873

9.5.4 Quantitative chemical risk assessment results

Table 9-30: Public irrigation garden scenario results

Contaminant	Inhalation				Dermal contact				Ingestion		Non cancer		Cancer	
	Non cancer		Cancer		Non cancer		Cancer		Non cancer	Cancer	Total risk for each contaminant			
	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Child	Child	Adult	Child	Adult
Antimony					1,46E-03	4,65E-04			2,40E-04		1,70E-03	4,65E-04		
Arsenic					9,23E-06	4,91E-05	5,93E-09	7,89E-09	1,01E-05		1,94E-05	4,91E-05	5,93E-09	7,89E-09
Bromodichloro-methane	7,19E-4	7,19E-4	4,56E-8	1,90E-7	6,95E-04	8,88E-05	3,62E-11	4,82E-11	1,90E-04	8,08E-09	1,60E-03	8,08E-04	5,37E-08	1,90E-7
Bromoform			1,53E-10	6,37E-10	1,35E-5	6,46E-06	2,74E-10	3,65E-10	6,31E-06	1,28E-10	1,98E-05	6,46E-06	5,55E-10	1,00E-09
Cadmium					1,12E-4	3,57E-05			6,14E-06		1,18E-04	3,57E-05		
Chloroform	1,66E-4	4,16E-4	8,01E-8	3,34E-7	2,61E-4	8,33E-04	6,93E-08	9,22E-08	4,20E-05	1,11E-08	4,69E-04	1,25E-03	1,61E-07	4,26E-07
Chromium														
Copper					2,28E-05	1,82E-06			2,51E-05		4,79E-05	1,82E-06		
Dibromochloro-methane					2,66E-5	2,98E-05	1,34E-08	1,79E-08	1,01E-05	5,10E-09	3,67E-05	2,98E-05	1,85E-08	1,79E-08
Iron					1,58E-06	5,06E-07			1,74E-06		3,33E-06	5,06E-07		
Lead					2,42E-07	7,72E-08	6,16E-13	8,20E-13	2,66E-06	6,77E-12	2,90E-06	7,72E-08	7,39E-12	8,20E-13
Manganese					6,42E-04	2,05E-04			2,82E-05		6,70E-04	2,05E-04		
Mercury					1,75E-04	5,57E-05			1,92E-04		1,06E-03	7,48E-04		
Nickel					2,33E-04	7,43E-05			5,12E-05		2,84E-04	7,43E-05		
Selenium					1,18E-05	3,76E-06			1,29E-05		2,47E-05	3,76E-06		
Zinc					9,79E-06	3,13E-06			1,79E-05		2,77E-05	3,13E-06		
											6,08E-03	**3,68E-03**	**2,39E-07**	**6,43E-07**
Halogenated solvents	8,86E-04	1,13E-03	1,26E-07	5,24E-07	9,96E-04	9,58E-04	8,30E-08	1,10E-07	2,48E-04	2,45E-08				
Metals	6,92E-04	6,92E-04			2,67E-03	8,94E-04	5,93E-09	7,90E-09	5,88E-04	6,77E-12				
Total risk for each route	**1,58E-03**	**1,83E-03**	**1,26E-07**	**5,24E-07**	**3,67E-03**	**1,85E-03**	**8,90E-08**	**1,18E-07**	**8,36E-04**	**2,45E-08**				

Table 9-31: Street cleaning scenario results

	Non cancer		Cancer		Non cancer		Cancer		Non cancer	Non cancer	Cancer	Cancer	Total risk for each contaminant			
	HQ inf	HQ inh	HQ'inh	HQ'inh	HQ dermal	HQ dermal	HQ' dermal	HQ' dermal	HQ oral	HQ oral	HQ'oral	HQ'oral	Non cancer		Cancer	
Contaminant	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Adult
Antimony					1,91E-3	6,69E-3			3,14E-4	8,83E06			2,22E-3	6,70E-3		
Arsenic					1,21E-5	7,08E-4	7,77E-9	1,1E-7	1,33E-5	2,49E-6	8,53E-9	6,67E-9	2,54E-5	7,10E-4	1,63E-8	1,20E-7
Bromodichloromethane	3,12E-4	6,87E-5	1,98E-8	9,08E-7	9,10E-4	1,28E-3	3,87E-8	5,7E-7	2,49E-4	4,67E-5	1,0E-8	8,26E-9	1,47E-3	1,39E-3	6,91E-8	1,48E-6
Bromoform		5,98E-6	5,13E-11	2,35E-9	1,77E-5	9,31E-5	3,59E-10	5,3E-9	8,26E-6	1,55E-6	1,6E-10	1,31E-10	2,59E-5	1,01E-4	5,78E-10	7,73E-9
Cadmium					1,46E-4	5,14E-4			8,04E-6	7,53E-8			1,54E-4	5,14E-4		
Chloroform	8,06E-5	2,16E-4	3,87E-8	1,78E-6	3,42E-4	1,20E-2	9,07E-8	1,3E-6	5,49E-5	1,03E-5	1,46E-8	1,14E-8	4,77E-4	1,22E-2	1,44E-7	3,12E-6
Copper					2,99E-5	2,62E-5			3,28E-5	6,15E-6			6,27E-5	3,24E-5		
Dibromochloromethane		2,62E-5			3,49E-5	4,29E-4	1,76E-8	2,6E-7	1,33E-5	2,49E-6	6,68E-9	5,22E-9	4,81E-5	4,57E-4	2,43E-8	2,62E-7
Iron					2,07E-6	7,29E-6			2,28E-6	4,27E-7			4,35E-6	7,72E-6		
Lead					3,16E-7	1,11E-6	8,07E-13	1,2E-11	3,48E-6	6,52E-7	8,87E-12	6,92E-12	3,79E-6	1,76E-6	9,67E-12	1,87E-11
Manganese					8,41E-4	2,95E-3			3,70E-5	2,77E-7			8,78E-4	2,95E-3		
Mercury					2,29E-4	8,03E-4			2,51E-4	4,71E-5			9,03E-4	8,51E-4		
Nickel					3,05E-4	1,07E-3			6,70E-5	5,02E-7			3,72E-4	1,07E-3		
Selenium					1,54E-5	5,42E-5			1,69E-5	3,18E-6			3,24E-5	5,73E-5		
Zinc					1,28E-5	4,51E-5			2,35E-5	4,40E-6			3,63E-5	4,95E-5		
													6,71E-3	**2,71E-2**	**2,54E-7**	**4,99E-6**
Halogenated solvents	3,93E-4	3,17E-4	5,86E-8	2,69E-6	1,30E-3	1,38E-2	1,47E-7	2,16E-6	3,25E-4	6,10E-5	3,20E-8	2,50E-8				
metals	4,24E-4	1,40E-6			3,50E-3	1,29E-2	7,77E-9	1,14E-7	7,70E-4	7,41E-5	8,55E-9	6,68E-9				
Total risk for each route	**8,17E-4**	**3,19E-4**	**5,86E-8**	**2,69E-6**	**4,80E-3**	**2,67E-2**	**1,55E-7**	**2,27E-6**	**1,09E-3**	**1,35E-4**	**4,06E-8**	**3,17E-8**				

Table 9-32: Golf course irrigation scenario results

	Non cancer	Cancer	Non cancer	Cancer	Total risk for each contaminant	
	HQ inh	HQ' inh	HQ dermal	HQ' dermal	Non cancer	Cancer
Contaminant	Adult	Adult	Adult	Adult	Adult	Adult
Antimony			1,11E-03		1,11E-03	
Arsenic			1,17E-04	1,88E-08	1,17E-04	1,88E-08
Bromodichloromethane	2,14E-03	5,66E-07	2,11E-04	9,37E-08	2,35E-03	6,59E-07
Bromoform		1,89E-09	1,54E-05	8,68E-10	1,54E-05	2,76E-09
Cadmium			8,49E-05		8,49E-05	
Chloroform	1,24E-03	9,93E-07	1,98E-03	2,20E-07	3,22E-03	1,21E-06
Copper			4,34E-06		4,34E-06	
Dibromochloromethane			7,08E-05	4,25E-08	7,08E-05	4,25E-08
Iron			1,20E-06		1,20E-06	
Lead and Compounds			1,84E-07	1,95E-12	1,84E-07	1,95E-12
Manganese			4,88E-04		4,88E-04	
Mercury	2,06E-03		1,33E-04		2,19E-03	
Nickel			1,77E-04		1,77E-04	
Selenium			8,95E-06		8,95E-06	
Zinc			7,45E-06		7,45E-06	
					9,85E-03	**1,94E-06**
Halogenated solvents	3,38E-03	1,56E-06	2,28E-03	3,57E-07		
Metals	2,06E-03	0,00E+00	2,13E-03	1,88E-08		
Total risk for each route	**5,44E-03**	**1,56E-06**	**4,41E-03**	**3,75E-07**		

Table 9-33: Toilet flushing scenario results

	Non cancer		Cancer		Non cancer		Cancer		Non cancer	Cancer	Total risk for each contaminant			
	HQinh	**HQinh**	**HQ'inh**	**HQ'inh**	**HQ dermal**	**HQ dermal**	**HQ' dermal**	**HQ' dermal**	**HQ oral**	**HQ'oral**	Non cancer		Cancer	
Contaminant	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Child	Child	Adult	Child	Adult
Antimony					7,80E-5	8,48E-4			3,85E-7		7,83E-5	8,48E-4		
Arsenic					4,94E-7	8,96E-5	3,18E-10	1,44E-8	1,63E-8	1,05E-11	5,11E-7	8,96E-5	3,28E-10	1,44E-8
Bromodi chloro methane	3,92E-5	2,09E-3	2,49E-9	5,53E-7	3,72E-5	1,62E-4	1,58E-9	7,17E-8	3,05E-7	1,29E-11	7,68E-5	2,25E-3	4,09E-9	6,25E-7
Bromoform			8,54E-12	1,90E-9	7,23E-7	1,18E-5	1,46E-11	6,65E-10	1,01E-8	2,05E-13	7,33E-7	1,18E-5	2,34E-11	2,56E-9
Cadmium					5,98E-6	6,51E-5			9,86E-9		5,99E-6	6,51E-5		
Chloroform	9,06E-6	1,21E-3	4,36E-9	9,69E-7	1,40E-5	1,52E-3	3,71E-9	1,68E-7	6,74E-8	1,79E-11	2,31E-5	2,73E-3	8,09E-9	1,14E-6
Copper					1,22E-6	3,32E-6			4,03E-8		1,26E-6	3,32E-6		
Dibromo chloro methane					1,43E-6	5,43E-5	7,18E-10	3,25E-8	1,63E-8	8,19E-12	1,44E-6	5,43E-5	7,27E-10	3,26E-8
Iron					8,49E-8	9,23E-7			2,80E-9		8,77E-8	9,23E-7		
Lead					1,29E-8	1,41E-7	3,30E-14	1,50E-12	4,27E-9	1,09E-14	1,72E-8	1,41E-7	4,39E-14	1,50E-12
Manganese					3,44E-5	3,74E-4			4,54E-8		3,44E-5	3,74E-4		
Mercury	3,78E-5	2,01E-3			9,35E-6	1,02E-4			3,08E-7		4,74E-5	2,12E-3		
Nickel					1,25E-5	1,36E-4			8,22E-8		1,26E-5	1,36E-4		
Selenium					6,31E-7	6,86E-6			2,08E-8		6,52E-7	6,86E-6		
Zinc					5,25E-7	5,71E-6			2,88E-8		5,54E-7	5,71E-6		
											2,84E-4	**8,69E-3**	**1,33E-8**	**1,82E-6**
Halogenated solvents	4,83E-5	3,30E-3	6,86E-9	1,52E-6	5,34E-5	1,75E-3	6,03E-9	2,73E-7	3,99E-7	3,93E-11				
Metals	3,78E-5	2,01E-3			1,43E-4	1,63E-3	3,18E-10	1,44E-8	9,44E-7	1,05E-11				
Total risk for each route	**8,61E-5**	**5,31E-3**	**6,86E-9**	**1,52E-6**	**1,96E-4**	**3,38E-3**	**6,35E-9**	**2,88E-7**	**1,34E-6**	**4,98E-11**				

Table 9-34: Toilet flushing scenario results

	Non cancer		Cancer		Non cancer		Cancer		Non cancer	Cancer	Total risk for each contaminant			
	HQinh	**HQinh**	**HQ'inh**	**HQ'inh**	**HQ dermal**	**HQ dermal**	**HQ' dermal**	**HQ' dermal**	**HQ oral**	**HQ'oral**	Non cancer		Cancer	
Contaminant	Child	Adult	Child	Adult	Child	Adult	Child	Adult	Child	Child	Child	Adult	Child	Adult
Antimony					7,80E-5	8,48E-4			3,85E-7		7,83E-5	8,48E-4		
Arsenic					4,94E-7	8,96E-5	3,18E-10	1,44E-8	1,63E-8	1,05E-11	5,11E-7	8,96E-5	3,28E-10	1,44E-8
Bromodi chloro methane	3,92E-5	2,09E-3	2,49E-9	5,53E-7	3,72E-5	1,62E-4	1,58E-9	7,17E-8	3,05E-7	1,29E-11	7,68E-5	2,25E-3	4,09E-9	6,25E-7
Bromoform			8,54E-12	1,90E-9	7,23E-7	1,18E-5	1,46E-11	6,65E-10	1,01E-8	2,05E-13	7,33E-7	1,18E-5	2,34E-11	2,56E-9
Cadmium					5,98E-6	6,51E-5			9,86E-9		5,99E-6	6,51E-5		
Chloroform	9,06E-6	1,21E-3	4,36E-9	9,69E-7	1,40E-5	1,52E-3	3,71E-9	1,68E-7	6,74E-8	1,79E-11	2,31E-5	2,73E-3	8,09E-9	1,14E-6
Copper					1,22E-6	3,32E-6			4,03E-8		1,26E-6	3,32E-6		
Dibromo chloro methane					1,43E-6	5,43E-5	7,18E-10	3,25E-8	1,63E-8	8,19E-12	1,44E-6	5,43E-5	7,27E-10	3,26E-8
Iron					8,49E-8	9,23E-7			2,80E-9		8,77E-8	9,23E-7		
Lead					1,29E-8	1,41E-7	3,30E-14	1,50E-12	4,27E-9	1,09E-14	1,72E-8	1,41E-7	4,39E-14	1,50E-12
Manganese					3,44E-5	3,74E-4			4,54E-8		3,44E-5	3,74E-4		
Mercury	3,78E-5	2,01E-3			9,35E-6	1,02E-4			3,08E-7		4,74E-5	2,12E-3		
Nickel					1,25E-5	1,36E-4			8,22E-8		1,26E-5	1,36E-4		
Selenium					6,31E-7	6,86E-6			2,08E-8		6,52E-7	6,86E-6		
Zinc					5,25E-7	5,71E-6			2,88E-8		5,54E-7	5,71E-6		
											2,84E-4	**8,69E-3**	**1,33E-8**	**1,82E-6**
Halogenated solvents	4,83E-5	3,30E-3	6,86E-9	1,52E-6	5,34E-5	1,75E-3	6,03E-9	2,73E-7	3,99E-7	3,93E-11				
Metals	3,78E-5	2,01E-3			1,43E-4	1,63E-3	3,18E-10	1,44E-8	9,44E-7	1,05E-11				
Total risk for each route	**8,61E-5**	**5,31E-3**	**6,86E-9**	**1,52E-6**	**1,96E-4**	**3,38E-3**	**6,35E-9**	**2,88E-7**	**1,34E-6**	**4,98E-11**				

9.6 Case Study of Shafdan

9.6.1 Life Cycle Assessment

Primary Data for Infrastructure

No estimations for infrastructure had been made for scenario '0a NWR' (groundwater wells and national water carrier). For the scenario '0b PWM' only estimations for seawater desalination had been considered.

Table 9-35: Inventory data for material infrastructure for the LCA Shafdan

Material	Unit	Lifetime [years]	0a. NWR (full-scale)	0b. PWM (full-scale)	1. LSAT (full-scale)	2. AOP & SSAT (pilot)	3. UF (pilot)	4. RO (pilot)
Concrete	m^3	30	-	28 806	-	487	815	2 688
Excavation volume	m^3	30 50	-	82 105	 1 100 000	 19 315	2 322	7 623
Reinforced steel	t	30	-	5 187	-	73.1	147	484
Sand	t	50	-	-	1 694	27.7	-	-
Anthracite	t	30	-	-	-	92.4	-	-
Basalt	t	30	-	-	-	108	-	-
Low alloyed steel	t	30	-	1 120	1 175	19.2	31.7	105
PE	t	30	-	436	-	-	12.3	40.7
Stainless steel	t	12	-	267	4.00	0.07	7.55	24.9
Copper	t	12	-	3.68	-	-	0.10	0.34
Cast iron	t	12	-	200	-	-	5.70	18.7
PVC-U	t	7	-	144	-	-	4.08	13.5
Epoxy Resin	t	7	-	70.6	-	-	2.00	6.59

Background data

Background processes for the schemes are modelled with dataset from ecoinvent v3.1 database [76] as described below (Table 9-36). Market datasets are used for all chemicals and materials as available. Additional transportation of chemicals and materials is considered.

Table 9-36: Ecoinvent dataset for background processes for the LCA Shafdan

Material	Ecoinvent v3.1 dataset	remark
Electricity	50 % electricity production, natural gas, at conventional power plant [RoW]; 50 % electricity production, hard coal [RoW]; 2 % losses by converting electricity high voltage into medium voltage	Mix for Israel 2015
NaOCl (12 %)	sodium hypochlorite production, product in 15% solution state [RER]	
NaOH (40 %)	market for sodium hydroxide, without water, in 50% solution state [GLO]	
Citric Acid (40 %)	market for citric acid [GLO]	
H_2SO_4 (96 %)	market for sulfuric acid [GLO]	
$Na_2S_2O_5$ (100 %)	200 % market for sodium hydrogen 311ulphite [GLO]	
Chlorine gas	market for chlorine, gaseous [RER]	
Lime (92 %)	market for lime, hydrated, packed [GLO]	
HCl (33 %)	market for hydrochloric acid, without water, in 30% solution state [RER]	
Antiscalant	dichloromethane production [RoW]; market for organophosphorus-compound, unspecified [GLO]: 50 % for each compound	for Wastewater RO
PACl (18 %)	aluminium hydroxide, at plant [RER]; electricity, medium voltage, at grid [ES]; hydrochloric acid, 30% in H2O, at plant [RER]	
H_2O_2 (50 %)	hydrogen peroxide production, product in 50% solution state [RoW]	
$FeCl_3$ (40 %)	iron (III) chloride production, product in 40% solution state [RoW]	
Concrete	Market for concrete, for de-icing salt contact [GLO]	Waste treatment included
Excavation volume	Excavation hydraulic digger [RER]	
Reinforced steel	Reinforced steel production [RoW]	
Sand	silica sand production [RoW]	
Anthracite	market for hard coal [WEU]	Waste treatment included
Basalt	market for basalt [GLO]	
Low alloyed steel	Steel production, low-alloyed, hot rolled [RoW]	
Stainless steel	Steel production, electric, chromium steel 18/8 [RoW]	
Copper	Copper production, primary [RoW]	
PE	Polyethylene production, low density, granulate [RER]	Waste treatment included
Cast iron	Cast iron production [RoW]	
PVC-U	Market for polyvinylchloride, bulk polymerized	
Epoxy Resin	Market for epoxy resin, liquid [GLO]	

Inventory for Water Impact Index

A detailed inventory on the water quality index including concentrations and reference concentration for the reuse schemes is shown in Table 9-37. Reference concentrations are taken from EC Directive 2008/105/EC [143].

Table 9-37: Reference concentration, effluent concentrations and water quality index for different parameters for the LCA Shafdan

	SS	COD	TN	TP	Cd	Cr	Cu	Hg	Ni	Pb	Zn
reference concentration (C_{ref}) standard parameters [mg/L]; heavy metals [µg/L]	35.0	30.0	13.7	0.2	0.45	3.4	1.4	0.05	20.0	7.2	7.8
Withdrawal from LSAT/ release to agriculture of water from LSAT											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	0.50	3.5	4.8	0.3	0.1	1.5	1.5	0.05	1.5	2.0	30.0
C_{ref}/C (WQI)	70.0	8.57	2.85	**0.62**	4.50	2.27	0.93	1.00	13.3	3.60	**0.26**
Withdrawal from SSAT/ release to agriculture of water from SSAT											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	0.50	4.0	4.9	0.3	0.1	1.5	1.5	0.05	1.5	2.0	30.0
C_{ref}/C (WQI)	70.0	7.50	2.77	**0.61**	4.50	2.27	0.93	1.00	13.3	3.60	**0.26**
Release to agriculture of water from UF											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	0.4	42.0	5.0	0.65	1.0	1.5	18.0	0.05	3.0	1.0	53.0
C_{ref}/C (WQI)	87.5	0.71	2.74	**0.31**	0.45	2.27	**0.08**	1.00	6.67	7.20	**0.15**
Release to agriculture of water from RO											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	-	0.25	0.54	0.02	-	-	-	-	-	-	-
C_{ref}/C (WQI)	-	120	25.3	10.0 (1.0)	-	-	-	-	-	-	-
Release of WWTP effluent to surface water											
Concentration (C) standard parameters [mg/L]; heavy metals [µg/L]	6.0	40.0	7.22	1.0	0.1	1.5	19.0	0.05	2.0	1.0	30.0
C_{ref}/C (WQI)	5.83	0.75	1.89	**0.20**	4.50	2.27	**0.07**	1.00	10.0	7.20	0.26

9.7 Case Study of Torreele

9.7.1 Life Cycle Assessment

Primary Data for Infrastructure

Note the Pipeline from Northern Wallonia to Koksijde is accounted for only 10 % in this study, since it provides a higher transport capacity and is not exclusively built for Koksijde supply.

Table 9-38: Inventory data for material infrastructure for the LCA Torreele

TT = tertiary treatment, GW = groundwater water treatment plant in Koksijde; RW = river water treatment plant in northern Wallonia; WN = water network (pipeline); SW = seawater reverse osmosis

Material	Unit	Lifetime [years]	1. IPR	2. IPR & Willows	3. Network	4. SWRO
Concrete	m^3	30 100	1378 (TT) 745 (GW)	1378 (TT) 745 (GW)	381 (GW) 355 (RW) 6 378 (WN)	381 (GW) 1805 (SW)
Excavation volume	m^3	30 100	3929 (TT) 1423 (GW)	3929 (TT) 1423 (GW)	728 (GW) 696 (RW) 2 702 (WN)	728 (GW) 5144 (SW)
Reinforced steel	t	30	284 (TT) 131 (GW)	284 (TT) 131 (GW)	67.8 (GW) 83.9 (RW)	67.8 (GW) 325 (SW)
Sand	t	30	26.2 (GW)	16856 (GW)	13.4 (GW) 12.8 (RW)	13.4 (GW)
Anthracite	t	30	39.5 (GW)	39.5 (GW)	20.2 (GW) 19.3 (RW)	20.2 (GW)
Low alloyed steel	t	30	53.6 (TT) 1.37 (GW)	53.6 (TT) 1.37 (GW)	0.70 (GW) 0.67 (RW)	0.70 (GW) 70.2 (SW)
PE	t	30	20.9 (TT) 0.38 (GW)	20.9 (TT) 0.38 (GW)	0.19 (GW) 0.07 (RW)	0.19 (GW) 27.3 (SW)
Stainless steel	t	12	12.8 (TT) 4.10 (GW)	12.8 (TT) 4.10 (GW)	2.10 (GW) 1.48 (RW)	2.10 (GW) 16.7 (SW)
Copper	t	12	0.18 (TT) 0.10 (GW)	0.18 (TT) 0.10 (GW)	0.05 (GW) 0.05 (RW)	0.05 (GW) 0.23 (SW)
Cast iron	t	12	9.58 (TT) 0.57 (GW)	9.58 (TT) 0.57 (GW)	0.29 (GW) 0.27 (RW)	0.29 (GW) 12.5 (SW)
PVC-U	t	7	6.90 (TT)	6.90 (TT)	-	9.04 (SW)
Epoxy Resin	t	7	3.38 (TT)	3.38 (TT)	-	4,42 (SW)
UV Lamps	-	3	32 (GW)	32 (GW)	17 (GW)	17 (GW)

Background data

Background processes for the schemes are modelled with dataset from ecoinvent v3.1 database [76] as described below (Table 9-39). Market datasets are used for all chemicals and materials as available.

Table 9-39: Ecoinvent dataset for background processes for the LCA Torreele

Material	Ecoinvent v3.1 dataset	remark
Electricity	Electricity, voltage transformation, high to medium [BE]	Mix for Belgium 2010
Heat	market for heat, district or industrial, other than natural gas [Europe without Switzerland]	
NaOCl (15 %)	sodium hypochlorite production, product in 15% solution state [RER]	
NaOH (50 %)	market for sodium hydroxide, without water, in 50% solution state [GLO]	
Citric Acid (40 %)	market for citric acid [GLO]	
H_2SO_4 (32 %)	market for sulfuric acid [GLO]	
$NaHSO_3$ (39 %)	market for sodium hydrogen sulfite [GLO]	
NH_4Cl (50 %)	market for ammonium chloride [GLO]	
Chlorine gas	market for chlorine, gaseous [RER]	
Lime (92 %)	market for lime, hydrated, packed [GLO]	
HCl (32 %)	market for hydrochloric acid, without water, in 30% solution state [RER]	
GAC fresh/reg.	electricity, high voltage, production mix [CN]; steam production, in chemical industry [RoW]; hard coal briquettes production [RoW]	
Antiscalant	dichloromethane production [RoW]; market for organophosphorus-compound, unspecified [GLO]: 50 % for each compound	for Wastewater RO
PACl (10 %)	aluminium hydroxide, at plant [RER]; electricity, medium voltage, at grid [ES]; hydrochloric acid, 30% in H2O, at plant [RER]	
$FeCl_3$ (40 %)	iron (III) chloride production, product in 40% solution state [RoW]	
Concrete	Market for concrete, for de-icing salt contact [GLO]	Waste treatment included
Excavation volume	Excavation hydraulic digger [RER]	
Reinforced steel	Reinforced steel production [RoW]	
Sand	silica sand production [RoW]	
Anthracite	market for hard coal [WEU]	Waste treatment included
Low alloyed steel	Steel production, low-alloyed, hot rolled [RoW]	
Stainless steel	Steel production, electric, chromium steel 18/8 [RoW]	
Copper	Copper production, primary [RoW]	
PE	Polyethylene production, low density, granulate [RER]	Waste treatment included
Cast iron	Cast iron production [RoW]	
PVC-U	Market for polyvinylchloride, bulk polymerized	
Epoxy Resin	Market for epoxy resin, liquid [GLO]	
UV Lamps	Flat glass production, uncoated [RER]; steel production low-alloyed, hot rolled [RER], copper production, primary [RoW], market for mercury [GLO]	Mixture of materials

Inventory for Water Impact Index

A detailed inventory on the water quality index including concentrations and reference concentration for the reuse schemes is shown in Table 9-40. Reference concentrations are taken from EC Directive 2008/105/EC [143].

Table 9-40: Reference concentration, effluent concentrations and water quality index for different parameters for the LCA Torreele

	COD	BOD	TOC	TN	TP
reference concentration (C_{ref}) [mg/L]	30.0	6.0	7.0	13.7	0.2
'1 IPR' and '2 IPR & Willows'					
Concentration (C) [mg/L] for reclaimed water for infiltration	-	-	1.0	4.6	0.1
C_{ref}/C (WQI)	-	-	7.00	2.98	2.00 **(1.00)**
'1 IPR'					
Concentration (C) [mg/L] effluent from UF concentrate and RO brine	109.9	10.6	29.1	29.7	3.8
C_{ref}/C (WQI)	0.27	0.57	0.24	0.46	**0.05**
'2 IPR & Willows'					
Concentration (C) [mg/L] effluent from UF concentrate and RO brine	103.3	10.6	29.1	15.2	2.9
C_{ref}/C (WQI)	0.29	0.57	0.24	0.90	**0.07**
'3 Network' and '4 SWRO'					
Concentration (C) [mg/L] WWTP effluent	-	3.7	-	8.4	1.8
C_{ref}/C (WQI)	-	1.62	-	1.63	**0.11**
All scenarios					
Concentration (C) [mg/L] Withdrawal (exemplary DWTP Koksijde)	-	-	2.5	1.6	0.1
C_{ref}/C (WQI)	-	-	2.80	8.55	2.00 **(1.00)**

10 References

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