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MODELLING ENVIRONMENTAL IMPACTS OF TREATED MUNICIPAL WASTEWATER REUSE FOR TREE CROPS IRRIGATION IN THE MEDITERRANEAN COASTAL REGION

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Abstract

Wastewater reuse provides valuable solutions to solve the societal challenges of decreasing availability and limiting access to secure water resources. The present study quantifies the environmental performance of nectarine orchards irrigation using TMW and surface water using a unique dataset based on field experimental data. One open field nectarine cultivation orchard located in Apulia Region, South of Italy, has been selected as case study. Climate change, toxicity (for human and freshwater), eutrophication (marine and freshwater) and acidification impacts were analysed using the impact assessment method suggested by the International Reference Life Cycle Data System (ILCD). The water footprint associated to the life cycles of each system has been estimated using the Available WAter REmaining (AWARE) method. Monte Carlo simulation was used to assess data uncertainty. The irrigation of nectarine orchards using TMW performes better than the irrigation using surface water for eutrophication impact categories. Compared with surface water resources, the potential impacts of TMW reuse in agriculture on climate change and toxicity are affected by the wastewater treatment phase (WWT). Only eutrophication and acidification burdens are generated by in-field substitution of surface water with TMW. Considering human and ecosystem water demand, the irrigation with TMW increases water consumption of 19.12 m^3 per kg of nectarine produced. Whereas, it shows a positive contribution to water stress (-0.19 m^3) if only human water demand is considered. This study provides important results that allow for a better understanding of the potential environmental consequences of TMW reuse in agriculture. It suggests that embracing the type of WWTs, the replacement of fertilizers, the effects on water scarcity and ecostytem quality might be useful to redefine water reuse regulations and increase public acceptance for the reuse of TMW in agriculture. Moreover, this study reveales the need for developing consensus and standardized guidance for life cycle analysis of water reuse applications.

Keywords:

- Wastewater reuse;
- Surface water;
- Nectarine orchards production;
- Life cycle assessment;
- Water footprint;
- Water scarcity.

1 Introduction

Continuous population growth and increasing urbanization are challenging secure water availability and wastewater disposal (Angelakis and Snyder, 2015). However, freshwater resources are mostly consumed in water scarce regions (Scherer and Pfister, 2016) and the exploitation of unsustainable water resources is predicted to increase in the future (Wada and Bierkens, 2014). Water reuse provides valuable solutions to solve the societal challenges linked to water scarcity and environmental pollution (Asano, 1994). The contribution of agriculture to total freshwater consumption can reach 80% in Southern European Regions (EEA, 2009). This is partly on account of the unique climate and geographical conditions of the Mediterranean coast and because of freshwater shortage. The reuse of treated municipal wastewater (TMW) as an alternative water source in agriculture has been recommended in many studies (Bedbabis et al., 2015; Grattan et al., 2015; Vivaldi et al., 2013; Weber et al., 2014). However, the high variability of soil-climate conditions, the large number of wastewater treatment technologies and water management options, make the estimation of the environmental impacts of water reuse systems complicated, mostly in regions where agriculture needs a large amount of water.

Recognizing that Apulia region has the smallest amount of available water resources (136 m^3/capita/year) and the lowest rainfall average value (i.e. about 660 mm/y) in Italy (Lopez et al. 2010), water reuse becomes a promising alternative for conserving and extending available water sources. Despite a large amount of TMW that is locally available (more than 100 million m^3/year), only 5 million m^3 are recovered yearly (Arborea et al. 2017), while the remainings are dumped into the sea. The lack of wastewater reuse is due to inadequate infrastructures for storage and distribution, restrictive legislation on reuse of wastewater, and scarce public acceptance fostered by inadequate information on the environmental impact of this alternative water source. Spain is the leading country on life cycle studies of water reuse in agriculture. Rodriguez-Garcia et al. (2011) analysed the life cycle eutrophication and global worming potentials (GWP) of six different wastewater treatment plants and associated water reuse applications (agriculture, industrial and aquifer recharge). They found the performance of TMW reuse in agriculture as worse compared with the other alternatives in terms of eutrophication, while TMW resulted as beneficial compared with acquirer recharge for Global Warming Potential (GWP). While analysing different tertiary treatments and wastewater reuse scenarios, Meneses et al. (2010) compared agriculture with landscape irrigation wastewater reuse, finding nutrient replacement as the main driver for the more beneficial outcomes of the former. Amores et al. (2013) compared the environmental profile of the current water cycle in Tarragona (Spain), with alternative scenarios where tertiary treatment and water reuse in agriculture were added to the current system. Reuse showed to be beneficial only for water use impact which was reduced when TMW is reused. However, these studies relied on previous studies to estimate of the environmental consequences of water reuse for non-potable applications. To the authors' knowledge, only Muñoz et al. (2009)) used experimental data to analyse the environmental impacts of TMW reuse. They found that irrigation with TMW performed better than using desalinated water or groundwater, in terms of eutrophication, aquatic ecotoxicity and energy use. To establish a scientific basis to evaluate environmental performances of TMW reuse for fruit orchards irrigation, the environmental impacts of nectarine grove irrigation with TMW and surface water in southern Italy are compared based on a unique experimental dataset.

2 Materials and method

2.1 General description of the field experiment.

An experimental trial was carried out in Trinitapoli in a commercial nectarine grove (Apulia Region, Southern Italy, 41°22'.92" N; 16°03'16.27" E; Altitude 1m). Two different water sources were compared for the irrigation of the nectarine orchard: treated municipal wastewater (TMW), made by a membrane filtration public plant located near the experimental site, and surface water (SW) that represents the conventional water used in this area. Acknowledging the high variability of water quality observed during a previous water monitoring campaign (data not published), the water fertilization contribution was not examined and the experiment was not set up to maximize fertilisers efficiency. Therefore, the same amount of fertilizers (102, 27, 0 kg ha^{-1} of N_2-P_2O_5-K_2O, respectively) were applied to both treatments. Moreover, as reported in Pedrero et al. (2018), the irrigation volume was calculated by the water balance method with restitution of 100% crop evapotraspiration[1] (ETc) lost in each irrigation interval, and the same amount of water was applied to both treatments. The yearly average (2012–2014) yield resulted higher in the SW (18,444 kg × ha^{-1}) compared with the TMW (17, 932 kg ha-1) treatment. Additional information about the experimental trial, irrigation and fertilization treatments are reported in Vivaldi et al. (2015) and provided in the supplementary materials (S1). Descriptive statistics of the physical and chemical characteristics of soil and water types are provided in the supplementary materials (Table S1 and Table S2).

2.2 Orchard modelling and boundaries definition.

Two cradle-to-farm gate scenarios were modelled for the irrigation of nectarine orchards using TMW (Scenario 1) and surface water (Scenario 2). In Scenario 1, the agricultural production process interacted with other product systems: (i) the secondary wastewater treatment (hereinafter referred to as WWT), generating an open loop recycling, and (ii) the water system by avoiding freshwater withdrawals. Along with the decrease in freshwater withdrawals, the reuse of TMW rise the same issue of multi-functionality as in the case of systemes producing more than one product (co-production) (Schrijvers et al., 2016). However, if a recycled material is used as input for a subsequent life cycle the allocation of the impacts should be tackled with the same procedures as for co-products (Guinée, 2002). As described in Schrijvers et al. (2016), the "cut-off" approach could be use to include treated wastewater in the analysis. However, Pradel et al. (2016) showed that this procedure holds only if treated wastewater is considered a final waste. In the study, the wastewater is considered a product containing nutrients and minerals, which directly affect agricultural production. Therefore, the system boundaries in Sceanrio 1 include the whole treated wastewater life cycle (Pradel et al., 2016). As suggested by the ISO standards 14044:2006 (ISO, 2006), the multi-functionality issue is addressed in this study by using system expansion, which guarantees the comparability between

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[1] The crop evapotranspiration (ETc) was calculated using the equation recommended by the FAO: ETc = Kr × Kc × ET_0, were Kr is the reduction coefficient (Kr= 0.75), Kc (0.50 Kcini, 1.15 Kcmid, 0.85 Kcend) is the crop coefficient, and ET_0 is reference evapotraspiration.

multi-dimensional (Scenario 1) and mono-dimensional (Scenario 2) systems (Schrijvers et al., 2016; Weidema, 2001). In system expansion, the functional unit (hereinafter referred to as FU) must be enlarged to include the function or service provided by the recycled product and the co-product (Hauschild et al., 2013). Therefore, Scenario 1 has been expanded by including the impacts generated by the avoided freshwater withdrawals and the discharging of TMW into the sea (*green boxes in* Figure 1). All agricultural processes (e.g. sowing, pruning, harvesting) were managed using the same equipment and machineries in the experimental field. Therefore, no machineries and equipment, fuels, or energy consumptions were accounted in the studied systems. However, fertilizers and pesticides production was incorporated in the system boundaries due to their impact in agricultural LCA (Brentrup et al., 2004; Margni et al., 2002). The application of fertilizers was carried out using the same equipment in both scenarios. Previous studies reported negligible environmental impact of the construction, dismantling, of a WWT plant compared to the operational phase (Lorenzo-Toja et al., 2016a; Muñoz and Fernández-Alba, 2008; Raluy et al., 2006). Moreover, water storage and distribution infrastructures are large-scale infrastructures managing large amounts of water and serving not only the agricultural sector, and they were thus expected to have negligible impacts on the environmental profile of the studied systems.Therefore, wastewater treatment plant, surface water storage infrastructures and the distribution network (e.g. dam, pumps and water network) were excluded from the analysis.

Figure 1: Material flows diagrams for nectarine field irrigation with TMW and surface water

The limitations arising from using a mass-based functional unit (FU) have been reported by several studies (Milà i Canals et al., 2006; Mouron et al., 2012; van der Werf et al., 2007) In this regard, Cerutti et al. (2014) recommend the use of both mass-based and land-based FU to avoid the overvaluation of the more resource efficient farming system. However, the experimental trial was set-up to maximize yield. Therefore, a mass-based FU (one kg of nectarines produced) reflecting a product-oriented expression of the agricultural systems (Hayashi, 2013) was used in this study.

2.3 Modelling assumptions and parameters specification

Field operations data, including the water inventory, were directly measured during the three years of the experimental trial (2012 - 2014). Background data for fertilizers production was provided by the ecoinvent v3 database (Wernet et al., 2016). Although Lorenzo-Toja et al. (2016b) reported considerable differences in terms of eutrophication net environmental impact and GWP according to the size of the WWT plant, materials flows, energy consumption and operational data for the WWT plant were modelled adopting an average WWT plant in the Mediterranean countries serving 144 000 population equivalent as defined by Amores et al. (2013) and Pasqualino et al. (2009). Data and assumptions on the WWT processes were presented in the supplementary materials (S2).

Table 1. Inputs-outputs inventory for Scenario 1 and 2 (per kg of nectarine produced).

All the emissions generated by fertilizers and water application have been accounted for. Ammonia, nitrate and nitrous oxides emissions were estimated according to Emmenegger et al. (2009). To accommodate the model specification, the average nitrogen uptake by nectarine trees reported by Johnson and Urui (1989) has been assumed. The SALCA – Phosphorus model (Prasuhn, 2006), including the modification proposed by Emmenegger et al. (2009) was used to estimate phosphorus emissions evaluating only run-off and leaching risks. According to the model, phosphorus emissions from fertilizers application depend on the slope of the field. The latter was 0% in the studied orchards, thus, only phosphorus leaching due to soil erosion has been accounted in the model. Field emissions from pesticide were estimated using PestLCI 2.0 model (Dijkman et al., 2012) and the approach suggested by Margni et al. (2002). Both irrigation water types included considerable concentration of macro and micronutrients (Table S1). However, the experimental trials in this study were not arranged to capture detailed data on plant uptakes and nutrients mobilisations at the soil-plant interface. Therefore, calcium, sulfur, magnesium and the other micronutrients in both water types have been accounted as indirect emissions directly reaching soil and water bodies.

2.4 Impact assessment

To address the lack of standardized impact categories in LCA, the International Reference Life-Cycle Data System (ILCD) recommendations (European Commission - Joint Research, 2011) were used in this study to define the best-suited methods for characterizing the impact categories at the midpoint level. Moreover, Bessou et al. (2013) and Cerutti et al. (2014) found global warming, eutrophication and toxicity impacts as the most affected by agriculture production. Accordingly, the following impact categories were quantified: Climate change (CC); Human toxicity (HT); Acidification (AC); Freshwater and marine eutrophication (FE, ME) and Freshwater ecotoxicity (FEc). The impacts for these categories were assessed using the IPCC 2007 GWP at 100 years (Ipcc, 2014), USEtox (Rosenbaum et al., 2008), and Accumulated Exceedance (Seppälä et al., 2006) methods, respectively. The rationale behind the selection of impact categories lay in the fact that the chosen methods are scientifically robust for assessing water degradation categories (Hauschild et al., 2013). In LCA, the direct impacts of water consumption and increasing water competition among different uses were recently addressed (Boulay et al., 2015b). Starting from the general framework proposed by Bayart et al. (2010), several methods addressing these issues were developed. Most of the LCA methods focused on water use impact (midpoint) while some addressed also the potential effects on human health and ecosystem quality (endpoint). At the midpoint level, the existing methods quantify water scarcity based on a use-to-availability ratio, which was referred to as a water stress index (Boulay et al., 2015b). The use-to-availability ratio approach assumes different theoretical and practical forms across the different methods. Withdrawal-to-availability (WTA) and consumption-to-availability (CTA) approaches only account for human water use, failing to examine ecosystem water requirements (Boulay et al., 2015b; Kounina et al., 2013). A water consumption pathway leading to an indicator, which encompassed potential deprivation of both ecosystem and human, was developed by including the demand-to-availability approach (DTA) in the water use characterization pathway (Boulay et al., 2015a). Among the three approaches proposed by the WaterLCA working group of the UNEP-SETAC Life Cycle Initiative (Boulay et al., 2017; Boulay et al., 2015a), the consensus was to use the inverse of water availability minus the water demand approach (1/AMD) and the Available WAter REmaining (AWARE) method. The AWARE method was used in this study to represent the impact of scenarios 1 and 2 on water use. This method builds on the assumption that the potential to deprive another user of water is directly proportional to the water consumed and inversely proportional to the available water remaining per unit of surface and time in a region (Boulay et al., 2018). The sensitivity of the modelled systems to the AWARE method assumptions was analysed by comparing the water use impact with the Water Stress Indicator (hereinafter WSI) proposed by Boulay et al. (2011). The WSI is the result of a CTA-based model for assessing the potential impacts from water use, which accounts only for human water consumption in the definitions of the water stress characterization factors (CFs) (Boulay et al., 2011). Since LCI databases only provide unit process for water production with global or country annual average resolution (Quinteiro et al., 2017), country level CFs were used in this study to guarantee the adequate connection with the inventory flows. For the life cycle impact assessment SimaPro 8.4 software (PRé Consultants, 2017) was used. The uncertainty arising from the variability in the experimental data was assed using Monte Carlo simulation (10,000 trials) with all input-output parmeters varying according the assumed distributions (Table 1).

3 Results

3.1 Life cycle impact assessment (LCIA)

Results are expressed at midpoint level, which translates impacts into the six selected environmental themes to identify key differences between nectarine orchards irrigation with TMW or surface water. Table 2 reports the LCIA outcomes, which are expressed in units per kg of nectarines produced at the farm gate. Scenario 1 performs worse than Scenario 2 for CC, HT, AC, FEc, while it achieves better outcomes for FE and ME. The irrigation with TMW (Scenario 1) generates lower euthrophication impacts (net environmental benefit for ME) compared with surface water (Scenario 2) due to the avoided discharging of TMW into the sea. The differences between the two scenarios are less emphasised for CC, HT and FEc, with Scenario 1 preforming worse than Scenario 2. The differences range from around 9% to 25% for the above mentioned impact categories and they are mainly attributable to WWTs.

Table 2: Characterization of the environmental impacts for the modelled scenarios. (FU = 1 kg of nectarines)[Standard Deviation]

The cultivation phase is the main phase responsible for the impacts on AC and FE (Table 3) in both scenarios. Accounting for the more than the 70% of the whole impact for CC and toxicity (both for humans and freshwater), the WWT phase is the main driver of the negative outcomes of Scenario 1 for these environmental categories. Electricity consumptions and the incineration of sewage sludge are the main contributor to the environmental impacts of the WWT phase.

Table 3: Process contributions to the midpoint environmental impacts of Scenario 1 and Scenario 2.

Figure 2 shows the environmental impacts of Scenario 1 and Scenario 2 normalised according to Benini et al.(2014). The category with the highest normalized impact is FEc (82% and 80% respectively) in both scenarios. CC accounts for almost 20% and 15% in both scenarios, respectively. AC and eutrophication impacts categories show only marginal effects on the environmental performance of Scenario 1 (lower than 1%), while AC and FE contribute for 4% and 2% respectively to the normalized impact of Scenario 2. Electricity consumption and the incineration of sewage sludge in the WWT phase are the main contributors to the environmental performance of Scenario 1 for CC, HT and FEc, while the outcomes of Scenario 2 for these impact categories are mainly driven by fertilizers and pesticides production. In both scenrios, the impacts on AC and eutrophication (both FE and ME) are mainly generated by the field operations.

Figure 2: Normalized processes contribution to environmental impacts of Scenario 1 and Scenario 2. (FU = 1 kg of nectarines)

3.2 Water footprint

Substituting surface water (Scenario 2) with TMW (Scenario 1) generates an almost 10 m^3 increase in water consumption per kg of nectarine produced (Table 4). The WWT is the main contributor to the water use impact of Scenario 1 (around 27 m^3). The contribution of fertilizers and pesticides production to the water use impact of Scenario 1 is almost three order of magniture lower compared with WWT. Whereas, field irrigation with TMW generates net benefit arising from the avoided freshwater withdrawals (-9.07 m^3). The overall water use impact of Scenario 2 resulted in almost 10 m^3, of which only 1.33 m^3 are associated with fertilisers and pesticides production. Field operations contributes for more than 90% to the burden associated to nectarine orchard irrigation with surface water (8.8 m³). According to the WSI method, Scenario 1 shows a positive contribution to water stress (-0.19 m³) which is attributable to the avoided surface water consumption for irrigation (Table 4). The WWT results in a water stress impact almost five order of magnitude lower compared with the water use impact. Scenario 2 shows to be detrimental (0.19 m³), with surface water withdrawals for field irrigation accounting for more than the 90% of the overall water stress impact. For both scenarios, the water stress impact of fertilizers and pesticides production is three order of magnitude lower than field operation.

Table 4: Water footprint and Water stress for both scenarios 1 and 2. (FU = 1 kg of nectarines) [Standard Deviation]

Furthermore, the comparison of CC and water use performances of both modelled systems with different scenarios, where the energy production mix used in the WWT process has been changed, shows that the climate change and water use performances of Scenario 1 could differ substantially according to the energy production mix used in the WWT phase (Figure 4).

4 Discussion

The substitution of surface water with TMW for nectarine orchards irrigation generates environmental trade-offs. The irrigation of nectarine orchards using TMW (Scenario 1) performes better than the irrigation with surface water (Scenario 2) for eutrophication impacts categories. It performs worse for CC, AC, toxicity (both for humans and freshwater bodies), and water use. T-test stastics report significant differences between Scenaio 1 and 2 for all impact categories (p-values < .001). The probability of Scenario 1 performing better than Scenario 2 is null for CC, AC and water use, it equals 100% for ME, and it varies within a range from 14% to 72% for HT, FEc, and FE respectively (Figure 3).

Figure 3: Probabilities density functions of the differences between Scenario 1 and Scenario 2 for HT, FEc, FE and water use impact categories (FU = 1 kg of nectarines).

Although applying different assumptions on system boundaries, scales and water reuse scenarios, several studies found the reuse of TMW more beneficial for the environment than desalinated water, conventional potable water and groundwater (Meneses et al., 2010; Muñoz et al., 2009; Pasqualino et al., 2011), but none of them contemplated surface water. TMW reuse for irrigation provides several direct benefits to water resources and crop productivity (Angelakis et al., 2003; Asano and Levine, 1996; Pedrero et al., 2010). Among these, nutrients recycling through the direct use of TMW for crop irrigation is beneficial for resources depletion and indirectly decreases the energy and water intensity of agricultural production (Mo and Zhang, 2013). However, the contribution of the fertilizers production phase results at least one order of magnitude lower compared with WWTs for all impact categories. Therefore, even a complete replacement of fertilizers with the nutrients provided with TMW is expected to provide only marginal benefits on the potential environmental impact of Scenario 1. A large share of the impacts generated in the cultivation phase originate in the use of fertilizers and pesticides. The excess chemicals, macro and micronutrients accumulate into soils, leach into groundwater and pollute aquatic ecosystems (Tuomisto et al., 2012). Moreover, in Scenario 1, the avoided discharge of TMW into the sea generates net environmental benefits for ME, which are attributed to the cultivation phase. Besides AC and eutrophication impacts, which are driven by nutrients accumulation and leaching into soil and water bodies respectively, the WWT stage in Scenario 1 resulted as the main contributor to the performance of Scenario 1 for water use, CC and toxicity impacts (Table 3). Energy production accounts for more than 65% and 90% of the impacts of the WWT process on CC and water use respectively (Table S4). Although, the high energy consumption of WWTs operations and their indirect contribution to GHG emissions and human toxicity have been documented by several studies (Amores et al., 2013; Lorenzo-Toja et al., 2016a; Rodriguez-Garcia et al., 2011), the implication on water availability is unclear. In this regard, the Italian energy production mix strongly relies on hydropower generation mix (around 20%, (Eurostat, 2017). The use of surface water for electricity generation prevents water resources to be allocated to other uses (drinking water, irrigation, etc.). According to the AWARE model specifications, this results in less water available after the demand of humans and ecosystems has been met, which can explain the estimated impact of WWT on water use. Therefore, changing the energy production mix towards less water dependent generation pathways could potentially modify the overall outcomes of the study.

Figure 4 Water use and climate change impacts of Scenario 1 using different energy production schemes in the WWT process.[The impact scores estimated for the different alternative energy production mixes in the WWT process are normalized using the impacts Scenario 1 as baseline] *(FU = 1 kg of nectarines)*

Although the use of water-use-intensive energy generation sources, such as hydropower and nuclear, generate beneficial effects on climate change impacts, they cause worse potential impacts on water use compared with the baseline Scenario 1 (Figure 4). The inverse trade-off is achieved when “water-free” generation technologies as biomass and fossil fuels are included in the energy production mix, reducing the burden on water resources while charging the system with higher potential impacts on climate change. Beneficial effects for both environmental impact indicators could result only if more than the 20% of both “carbon and water-free” energy generation technologies (e.g. wind and solar) provide the electricity used in the WWT process in Scenario 1.

The model chosen to assess the minimum water requirements to maintain freshwater ecosystems brings a high degree of uncertainty in the AWARE CFs (Boulay et al. 2017; Damiani et al. 2018). The freshwater ecosystem demand is defined according the method proposed by Pastor et al. (2014), which sets the 30-60 % of mean monthly flow as degradation limits for maintaining the freshwater ecosystem in a fair ecological condition. These water flow dynamics require a level of temporal resolution which is beyond the impact assessment capabilities most impact assessment methods.

The sensitivity of the AWARE model to this assumption has been tested by comparing water use impacts with the WSI of both scenarios respectively. The different impacts on water resources resulting from the two models are a result of the different underlying approaches used to define the CFs: CTA and DTA, respectively. The former considers local water scarcity as being affected only by the water that is actually allocated to human activities, while the latter contemplates both ecosystem water demand and human consumption as factors affecting local water availability. According to the WSI method, the water used for hydropower generation in the WWT stage, which is then returned to freshwater bodies, does not affect local water availability. In contrast, the AWARE model specification regards the water allocated to hydropower generation as unavailable for freshwater ecosystems and therefore reducing local water availability.

This study sought to estimate the environmental impacts of field application of TMW as substitute of surface water for trees irrigation. Therefore it used data from Amores et al. (2013) to model the background WWT process, the quality and amount of water influent, the technology usedfrom a WWT plant located in the Mediterranean region. Although Meneses et al. (2010) reported similar environmental profiles for different wastewater treatment technologies and wastewater treatment plants located in the Mediterranean region, this assumption increases the uncertainty of the outcomes, and might led to unrealistic scenarios especially in relation to water (Risch et al., 2014) and energy (Baresel et al., 2015) use in the WWT process. Unlike regionalized LCA water stress CFs, the higher spatial resolution ones (global or country level) might generate results bias when targeting obvious local environmental issues such as water scarcity (Bai et al., 2017). However, Quinteiro et al. (2017) and Yang et al. (2017) underlined the need to establish a consistent link between local CFs and the related spatial inventory to better recognise the relevance of the impact results on local water use. The latter remains a major challenge in LCA (Yang, 2016; Yang et al., 2017). Since water inflows are unevenly distributed through Italy (Scarascia et al., 2006), the use of national LCI databases for background processes and country level CFs might have led to underestimate or overestimate the water use impacts generated by surface water use for irrigation purposes in the Southern regions.

5 Conclusion

This study sought to contribute to the water recycling literature by measuring the environmental profile of fruit orchard irrigation with TMW and by comparing it with the use of surface water. The use of TMW achieved better outcomes for eutrophication related environmental burdens, while it performed worse for climate change, toxicity (humans and freshwater), acidification and water use. Although the input-output flows are directly related to the specific location and characteristics of the experimental field, the study has identified areas of potential beneficial contribution and the main potential sources of environmental harmof TMW reuse in agriculture. Compared with conventional water resources, the potential environmental impacts of TMW reuse in agriculture are affected by the

WWT phase. This is true for the potential impacts on climate and both humans and freshwater toxicity. In-field substitution of surface water with TMWgenerates both environmental burdens, for the potential impacts on acidification and freshwater eutrophication, and benefits for the potential impacts on marine eutrophication.

Moreover, the study shows that the assumptions embedded in the water footprint characterization models have a crucial role in defining the water footprint of TMW reuse in agriculture. This issue, together with the development of a consistent local inventory, deserves particular attention and should be the matter of future research for reducing the uncertainty of water use impacts on local water scarcity. This study provided important results that allow for a better understanding of the potential environmental consequences of TMW reuse in agriculture. The existing regulations on the reuse of TMW for irrigation distinguish the use or non-use options based on the microbiological quality of the effluents and the type of irrigated crops. The outcomes of this study suggest that a broader environmental perspective, embracing the WWTs technologies, the replacement of fertilizers, the effects on water availability and ecostytem quality might be useful to redefine these regulations and increase public acceptance for the reuse of TMW in agriculture.

Moreover, it reveal the need for developing consensus and standardized guidance for life cycle analysis of water reuse applications.

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References

Amores MJ, Meneses M, Pasqualino J, Antón A, Castells F. Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach. Journal of Cleaner Production 2013; 43: 84-92.

Angelakis AN, Bontoux L, Lazarova V. Challenges and prospectives for water recycling and reuse in EU countries. Water Science and Technology: Water Supply 2003; 3: 59-68.

Angelakis AN, Snyder SA. Wastewater Treatment and Reuse: Past, Present, and Future. 2015: 4887-4895.

Asano T. Reusing Urban Wastewater—An Alternative and a Reliable Water Resource. Water International 1994; 19: 36-42.

Asano T, Levine AD. Wastewater reclamation, recycling and reuse: Past, present, and future. Water Science and Technology 1996; 33: 1-14.

Bai S, Wang X, Huppes G, Zhao X, Ren N. Using site-specific life cycle assessment methodology to evaluate Chinese wastewater treatment scenarios : A comparative study of site-generic and site-specific methods. Journal of Cleaner Production 2017; 144: 1-7.

Baresel C, Dahlgren L, Almemark M, Lazic A. Municipal wastewater reclamation for non-potable reuse - Environmental assessments based on pilot-plant studies and system modelling. Water Science and Technology 2015; 72: 1635-1643.

Bayart JB, Bulle C, Deschênes L, Margni M, Pfister S, Vince F, et al. A framework for assessing off-stream freshwater use in LCA. 15, 2010, pp. 439-453.

Bedbabis S, Trigui D, Ben Ahmed C, Clodoveo ML, Camposeo S, Vivaldi GA, et al. Long-terms effects of irrigation with treated municipal wastewater on soil, yield and olive oil quality. Agricultural Water Management 2015; 160: 14-21.

Bessou C, Basset-Mens C, Tran T, Benoist A. LCA applied to perennial cropping systems: a review focused on the farm stage. International Journal of Life Cycle Assessment 2013; 18: 340-361.

Boulay A-m, Bare J, Benini L, Berger M, Lathuillière MJ, Manzardo A, et al. The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE). The International Journal of Life Cycle Assessment 2017.

Boulay A-M, Bayart J-B, Bulle C, Franceschini H, Motoshita M, Muñoz I, et al. Analysis of water use impact assessment methods (part B): applicability for water footprinting and decision making with a laundry case study. The International Journal of Life Cycle Assessment 2015a; 20: 865-879.

Boulay A-m, Bayart J-b, Desch L, Margni M. Regional Characterization of Freshwater Use in LCA : Modeling Direct Impacts on Human Health. Environmental Science & Technology 2011; 45: 8948-8957.

Boulay A-M, Motoshita M, Pfister S, Bulle C, Muñoz I, Franceschini H, et al. Analysis of water use impact assessment methods (part A): evaluation of modeling choices based on a quantitative comparison of scarcity and human health indicators. The International Journal of Life Cycle Assessment 2015b; 20: 139-160.

Boulay AM, Bare J, Benini L, Berger M, Lathuillière MJ, Manzardo A, et al. The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available water remaining (AWARE). International Journal of Life Cycle Assessment 2018; 23: 368-378.

Brentrup F, Küsters J, Lammel J, Barraclough P, Kuhlmann H. Environmental impact assessment of agricultural production systems using the life cycle assessment (LCA) methodology II. The application to N fertilizer use in winter wheat production systems. European Journal of Agronomy 2004; 20: 265-279.

Cerutti AK, Beccaro GL, Bruun S, Bosco S, Donno D, Notarnicola B, et al. Life cycle assessment application in the fruit sector: State of the art and recommendations for environmental declarations of fruit products. Journal of Cleaner Production 2014; 73: 125-135.

Dijkman TJ, Birkved M, Hauschild MZ. PestLCI 2.0: a second generation model for estimating emissions of pesticides from arable land in LCA. The International Journal of Life Cycle Assessment 2012; 17: 973-986.

EEA. Water resources across Europe — confronting water scarcity and drought. EEA Report 2/2009, 2009.

Emmenegger M, Reinhard J, Zah R, Ziep T. Sustainability Quick Check for Biofuels - intermediate background report. Rsb.Epfl.Ch 2009: 1-29.

European Commission - Joint Research C. International Reference Life Cycle Data System (ILCD) Handbook - Reccomandation for Life Cycle Impact Assessemnt in the European context. Luxemburg, 2011.

Eurostat. Electricity production and supply statistics - Statistics Explained, 2017.

Grattan SR, Díaz FJ, Pedrero F, Vivaldi GA. Assessing the suitability of saline wastewaters for irrigation of Citrus spp.: Emphasis on boron and specific-ion interactions. Agricultural Water Management 2015; 157: 48-58.

Guinée J. Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards. Vol 7: Springer Netherlands, 2002.

Hauschild MZ, Goedkoop M, Guinée J, Heijungs R, Huijbregts M, Jolliet O, et al. Identifying best existing practice for characterization modeling in life cycle impact assessment. International Journal of Life Cycle Assessment 2013; 18: 683-697.

Hayashi K. Practical recommendations for supporting agricultural decisions through life cycle assessment based on two alternative views of crop production: the example of organic conversion. The International Journal of Life Cycle Assessment 2013; 18: 331-339.

Ipcc. Climate Change 2014: Impacts, Adaptation, and Vulnerability Part A: Global and Sectoral Aspects. (Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change), 2014.

ISO. Iso 14040. Euro code SS-EN-1191-2 2006; 1997.

Johnson RS, Urui K. Mineral Nutrition. In: LaRue JH, Johnson RS, editors. Cooperative Extension, University of California, Division of Agriculture and Natural Resources, 1989, pp. 246-246.

Kounina A, Margni M, Bayart J-B, Boulay A-M, Berger M, Bulle C, et al. Review of methods addressing freshwater use in life cycle inventory and impact assessment. Int J Life Cycle Assess 2013; 18: 707-721.

Lorenzo-Toja Y, Alfonsín C, Amores MJ, Aldea X, Marin D, Moreira MT, et al. Beyond the conventional life cycle inventory in wastewater treatment plants. 2016a.

Lorenzo-Toja Y, Vázquez-Rowe I, Amores MJ, Termes-Rifé M, Marin-Navarro D, Moreira MT, et al. Benchmarking wastewater treatment plants under an eco-effi ciency perspective. Science of the Total Environment 2016b; 566-567: 468-479.

Margni M, Rossierb D, Crettaz P, Jolliet O. Life cycle impact assessment of pesticides on human health and ecosystems. Agriculture, Ecosystems & Environment 2002; 93: 379 - 392.

Meneses M, Pasqualino JC, Castells F. Environmental assessment of urban wastewater reuse: Treatment alternatives and applications. Chemosphere 2010; 81: 266-272.

Milà i Canals L, Burnip GM, Cowell SJ. Evaluation of the environmental impacts of apple production using Life Cycle Assessment (LCA): Case study in New Zealand. Agriculture, Ecosystems & Environment 2006; 114: 226-238.

Mo W, Zhang Q. Energy-nutrients-water nexus: Integrated resource recovery in municipal wastewater treatment plants. Journal of Environmental Management 2013; 127: 255-267.

Mouron P, Heijne B, Naef A, Strassemeyer Jr, Hayer F, Avilla J, et al. Sustainability assessment of crop protection systems: SustainOS methodology and its application for apple orchards. Agricultural Systems 2012; 113: 1-15.

Muñoz I, Del Mar Gómez M, Fernández-Alba AR. Life Cycle Assessment of biomass production in a Mediterranean greenhouse using different water sources: Groundwater, treated wastewater and desalinated seawater. Agricultural Systems 2009; 103: 1-9.

Muñoz I, Fernández-Alba AR. Reducing the environmental impacts of reverse osmosis desalination by using brackish groundwater resources. Water Research 2008; 42: 801-811.

Pasqualino JC, Meneses M, Abella M, Castells F. LCA as a Decision Support Tool for the Environmental Improvement of the Operation of a Municipal Wastewater Treatment Plant. Environmental Science & Technology 2009; 43: 3300-3307.

Pasqualino JC, Meneses M, Castells F. Life Cycle Assessment of Urban Wastewater Reclamation and Reuse Alternatives. Journal of Industrial Ecology 2011; 15: 49-63.

Pastor AV, Ludwig F, Biemans H, Hoff H, Kabat P. Accounting for environmental flow requirements in global water assessments. Hydrology and Earth System Sciences 2014; 18: 5041-5059.

Pedrero F, Camposeo S, Pace B, Cefola M, Vivaldi GA. Use of reclaimed wastewater on fruit quality of nectarine in Southern Italy. Agricultural Water Management 2018; 203: 186-192.

Pedrero F, Kalavrouziotis I, Alarcón JJ, Koukoulakis P, Asano T. Use of treated municipal wastewater in irrigated agriculture—Review of some practices in Spain and Greece. Agricultural Water Management 2010; 97: 1233-1241.

Pradel M, Aissani L, Villot J, Baudez J-C, Laforest V. From waste to added value product: towards a paradigm shift in life cycle assessment applied to wastewater sludge – a review. Journal of Cleaner Production 2016; 131: 60-75.

Prasuhn V. Erfassung der PO 4 -Austräge für die Ökobilanzierung. SALCA-Phosphor. Agroscope FAL Reckenholz 2006.

Quinteiro P, Ridoutt BG, Arroja L, Dias AC. Identification of methodological challenges remaining in the assessment of a water scarcity footprint: a review. International Journal of Life Cycle Assessment 2017: 1-17.

Raluy G, Serra L, Uche J. Life cycle assessment of MSF, MED and RO desalination technologies. Energy 2006; 31: 2025-2036.

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

Rodriguez-Garcia G, Molinos-Senante M, Hospido A, Herná Ndez-Sancho F, Moreira MT, Feijoo G. Environmental and economic profile of six typologies of wastewater treatment plants. Water Research 2011; 45: 5997-6010.

Rosenbaum RK, Bachmann TM, Jolliet O, Juraske R, Koehler A, Hauschild MZ. USEtox — the UNEP-SETAC toxicity model : recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. International Journal of Life Cycle Assesment 2008: 532-546.

Scarascia MEV, Battista FD, Salvati L. Water resources in Italy: availability and agricultural uses. Irrigation and Drainage 2006; 55: 115-127.

Scherer L, Pfister S. Dealing with uncertainty in water scarcity footprints. Environmental Research Letters 2016; 11: 054008-054008.

Schrijvers DL, Loubet P, Sonnemann G. Developing a systematic framework for consistent allocation in LCA. The International Journal of Life Cycle Assessment 2016; 21.

Seppälä J, Posch M, Johansson M, Hettelingh J-p. LCA Methodology Country-Dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator. International Journal of Life Cycle Assesment 2006; 11: 403-416.

Tuomisto HL, Hodge ID, Riordan P, Macdonald DW. Comparing energy balances, greenhouse gas balances and biodiversity impacts of contrasting farming systems with alternative land uses. Agricultural Systems 2012; 108: 42-49.

van der Werf HMG, Tzilivakis J, Lewis K, Basset-Mens C. Environmental impacts of farm scenarios according to five assessment methods. Agriculture, Ecosystems and Environment 2007; 118: 327-338.

Vivaldi GA, Camposeo S, Rubino P, Lonigro A. Microbial impact of different types of municipal wastewaters used to irrigate nectarines in Southern Italy. Agriculture, Ecosystems & Environment 2013; 181: 50-57.

Wada Y, Bierkens MFP. Sustainability of global water use: past reconstruction and future projections. Environmental Research Letters 2014; 9: 104003-104003.

Weber E, Grattan SR, Hanson BR, Vivaldi GA, Meyer RD, Prichard TL, et al. Recycled water causes no salinity or toxicity issues in Napa vineyards. 68, 2014, pp. 59-67.

Weidema B. Avoiding Co-Product Allocation in Life-Cycle Assessment. Journal of Industrial Ecology 2001; 4: 11-33.

Wernet G, Bauer C, Steubing B, Reinhard J, Moreno-Ruiz E, Weidema B, et al. The ecoinvent database version 3 (part I): overview and methodology. The International Journal of Life Cycle Assessment 2016; 21: 1218-1230.

Yang Y. Toward a more accurate regionalized life cycle inventory. Journal of Cleaner Production 2016; 112: 308-315.

Yang Y, Tao M, Suh S. Geographic variability of agriculture requires sector-specific uncertainty characterization. International Journal of Life Cycle Assessment 2017: 1-9.

Tables

Table 1: Inputs-outputs inventory for Scenario 1 (waste water field irrigation) and 2 (surface water field irrigation). *(FU = 1 kg of nectarines)*

Input/output	Average Nominal Range [min; max]				Units	Distribution
	Scenario 1		Scenario 2			
Field operation		-				
Water	-	-	0.190	[0.68; 0.21]	m³	Triangular
Ammonium sulphate	0.005	-	0.005	-	kg	-
Phosphate fertilizer	0.001	-	0.001	-	kg	-
Treated wastewater	0.202	[0.17; 0.22]	0.196	[0.16;0.21]	m³	Triangular
Pyridate [b]	0.043	[0.039;0.046]	0.042	[0.038; 0.044]	g	Triangular
Tebuconazole [b]	0.009	[0.006; 0.11]	0.009	[0.006; 0.11]	g	Triangular
Thiram [b]	0.595	[0.44; 0.69]	0.578	[0.43; 0.67]	g	Triangular
Benzoate [b]	0.455	[0.39; 0.59]	0.443	[0.38; 0.57]	g	Triangular
White mineral oil [b]	1.56	[1.17; 2.34]	1.52	[1.14; 2.28]	g	Triangular
Emissions to air						
Ammonia (fertilizers)	0.530	-	0.530	-	g	-
Ammonia	1.400	[SD^2: 1.19]	1.010	[SD^2: 1.04]	g	Lognormal
Nitrogen oxides	1.740	[SD^2: 1.74]	1.080	[SD^2: 1.00]	g	Lognormal
Pyridate [b]	0.34	-	0.33	-	mg	-
Tebuconazole [b]	0.23	-	0.23	-	mg	-
Thiram [b]	2.16	-	2.11	-	mg	-
Benzoate [b]	0.18	-	0.18	-	mg	-
White mineral oil [b]	0.16	-	0.15	-	g	-
Emissions to water						

Nitrate (freshwater)	1.180	[SD^2: 1.00]	2.140	[SD^2: 1.00]	g	Lognormal
Phosphorous	2.310	[SD^2: 2.31]	2.520	[SD^2: 2.52]	g	Lognormal
Pyridate [b]	0.003	-	0.003	-	mg	-
Tebuconazole [b]	0.29	-	0.28	-	mg	-
Thiram [b]	9.85E-5	-	9.57E-5	-	mg	-
Benzoate [b]	16.37	-	15.92	-	mg	-
White mineral oil [b]	0.13	-	0.13		g	-
Nitrate [a]	-1.280	[SD^2: 1.54]	-		g	Lognormal
Phosphate [a]	-2.780	[SD^2: 5.15]	-		g	Lognormal
Ammonium [a]	-8.180	[SD^2: 1.89]	-		g	Lognormal
Fluoride [a]	-1.290	[SD^2: 1.52]	-		g	Lognormal
Chloride [a]	-29.350	[SD^2: 4.20]	-		g	Lognormal
Sulphate [a]	-19.490	[SD^2: 1.75]	-		g	Lognormal
Sodium [a]	-8.180	[SD^2: 1.89]	-		g	Lognormal
Potassium [a]	-5.360	[SD^2: 1.71]	-		g	Lognormal
Calcium [a]	-17.720	[SD^2: 3.36]	-		g	Lognormal
Magnesium [a]	-4.080	[SD^2: 2.09]	-		g	Lognormal
Emissions to soil						-
White mineral oil [b]	1.19	-	1.16	-	g	-

[a] Avoided emission into the sea

[b] Active principle

Table 2: Characterization of the environmental impacts for the modelled scenarios. *(FU = 1 kg of nectarines) [Standard Deviation]*

	Unit	Treated municipal wastewater (S1)		Surface water (S2)	
Climate change	kg CO_2 eq	2.29E-01	[SD: 0.03]	2.18E-02	[SD: 2.56E-3]
Human toxicity	CTUh	3.02E-08	[SD: 2.08E-8]	7.84E-09	[SD: 4.11E-9]
Acidification	molc H+ eq	7.17E-03	[SD: 3.94E-4]	5.02E-03	[SD:7.86E-5]
Freshwater eutrophication	kg P eq	1.47E-03	[SD: 1.27E-3]	2.51E-03	[SD: 1.19E-3]
Marine eutrophication	kg N eq	-6.41E-03	[SD: 2.20E-3]	6.55E-04	[SD: 3.17E-6]
Freshwater ecotoxicity	CTUe	1.04E+00	[SD: 1.32]	1.22E-01	[SD: 2.45E-2]

Table 3: Process contributions to the midpoint environmental impacts of Scenario 1 and Scenario 2.

	Unit	Processes contribution						
		Treated municipal wastewater (S1)				Surface water (S2)		
		WWT[a]	Fertilisers	Pesticides	Field operation	Fertilisers	Pesticides	Field operation
Climate change	kg CO_2 eq	2.09E-01	1.11E-02	1.06E-02	-	1.08E-02	1.03E-02	-
Human toxicity	CTUh	2.23E-08	2.26E-09	5.75E-09	8.01E-12	2.15E-09	5.60E-09	7.78E-12
Acidification	molc H+ eq	8.61E-04	8.92E-05	2.20E-04	6.00E-03	2.97E-10	2.14E-04	4.72E-03
Freshwater eutrophication	kg P eq	6.13E-05	3.94E-06	3.94E-06	1.39E-03	3.75E-06	3.83E-06	2.52E-03
Marine eutrophication	kg N eq	1.67E-04	8.27E-06	2.15E-05	-6.58E-03	8.59E-05	2.09E-05	6.25E-04
Freshwater ecotoxicity	CTUe	9.19E-01	3.56E-02	5.45E-02	3.27E-02	3.38E-02	5.30E-02	3.17E-02

[a] Adapted form (Amores et al., 2013) according to the specific case study (Supplementary Materials S2).

Table 4: Water footprint and Water stress for both Scenarios 1 and 2. (FU = 1 kg of nectarines) [Standard Deviation]

	Units	**Reclaimed wastewater (S1)**				**Surface water (S2)**			
		Average	**SD**	**Process contribution**		**Average**	**SD**	**Process contribution**	
Water use	m³	19.12	[3.31]	*Field operation*	-9.07E+00	10.06	[0.50]	*Field operation*	8.80E+00
				Fertilizers	7.22E-03			*Fertilizers*	6.76E-03
				Pesticides	1.36E+00			*Pesticides*	1.32E+00
				WWT	2.71E+01				
Water stress	m³	-0.19	[0.02]	*Field operation*	-1.99E-01	0.19	[0.009]	*Field operation*	1.94E-01
				Fertilizers	1.14E-04			*Fertilizers*	1.06E-04
				Pesticides	4.18E-05			*Pesticides*	4.07E-05
				WWT	2.93E-04				

Figure captions

Figure 1: Material flows diagrams for nectarine field irrigation with treated wastewater and surface water. *(FU = 1 kg of nectarines)*

Figure 2: Normalized processes contribution to environmental impacts of Scenario 1 (waste water field irrigation) and Scenario 2 (surface water field irrigation). *(FU = 1 kg of nectarines)*

Figure 3: Probabilities density functions of the differences between Scenario 1 and Scenario 2 for HT, FEc., FE and water use impact categories. *(FU = 1 kg of nectarines)*

Figure 4: Water use and climate change impacts of Scenario 1 (waste water field irrigation) using different energy production schemes in the WWT process. [The impact scores estimated for the different alternatives energy production mix in the WWT process are normalized using the impacts Scenario 1 as baseline] *(FU = 1 kg of nectarines)*