Life cycle assessment and costing of urine source separation: Focus on nonsteroidal anti-inflammatory drug removal

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1 **ABSTRACT**

2 Urine source separation has the potential to reduce pharmaceutical loading to the environment,
3 while enhancing nutrient recovery. The focus of this life cycle assessment (LCA) was to evaluate
4 the environmental impacts and economic costs to manage nonsteroidal anti-inflammatory drugs
5 (NSAIDs) (i.e., diclofenac, ibuprofen, ketoprofen and naproxen) and nutrients in human urine.
6 Urine source separation was compared with centralized wastewater treatment (WWT) (biological
7 or upgraded with ozonation). The current treatment method (i.e., centralized biological WWT)
8 was compared with hypothetical treatment scenarios (i.e., centralized biological WWT upgraded
9 with ozonation, and urine source separation). Alternative urine source separation scenarios
10 included varying collection and handling methods (i.e., collection by vacuum truck, vacuum
11 sewer, or decentralized treatment), pharmaceuticals removal by ion-exchange, and struvite
12 precipitation. Urine source separation scenarios had 90% lower environmental impact (based on
13 the TRACI impact assessment method) compared with the centralized wastewater scenarios due
14 to reduced potable water production for flush water, reduced electricity use at the wastewater
15 treatment plant, and nutrient offsets from struvite precipitation. Despite the greatest reduction of
16 pharmaceutical toxicity, centralized treatment upgraded with ozone had the greatest ecotoxicity
17 impacts due to ozonation operation and infrastructure. Among urine source separation scenarios,
18 decentralized treatment of urine and centralized treatment of urine collected by vacuum truck
19 had negligible cost differences compared with centralized wastewater treatment. Centralized
20 treatment of urine collected by vacuum sewer and centralized treatment with ozone cost 30%
21 more compared with conventional wastewater treatment.

22 **Keywords:** Life cycle assessment; life cycle costing; urine source separation; nonsteroidal anti-
23 inflammatory drugs; ion-exchange; wastewater treatment

1. Introduction

Approximately 64% of ingested pharmaceuticals intended for human use are excreted in urine as the parent compound or metabolites (Lienert et al., 2007a). Human urine is the primary contributor of pharmaceuticals in municipal wastewater but only constitutes 1% of the total volumetric flow (Joss et al., 2005, Larsen and Gujer, 1996). Urine source separation has been proposed as a more efficient method to remove and/or destroy pharmaceuticals as opposed to centralized biological wastewater treatment because pharmaceuticals are present at much higher concentrations in undiluted urine (Lamichhane and Babcock, 2012). In addition, human urine contributes 80% of the nitrogen (N) and 50% of the phosphorus (P), indicating separate treatment of urine may have significant impacts on centralized wastewater treatment (Larsen and Gujer, 1996). Furthermore, human urine may be utilized as an alternative fertilizer source because N and P are essential nutrients used in agriculture (Kirchmann and Pettersson, 1995). Therefore, effective separation of pharmaceuticals from nutrients is necessary to obtain a “contaminant free” nutrient product. From previous research, sorption processes are an effective method to selectively remove nonsteroidal anti-inflammatory drugs (NSAIDs) from urine with minimal co-sorption of nutrients, and may be effectively regenerated using a 5% sodium chloride, 50% methanol solution (Landry and Boyer, 2013, Landry et al., 2015). Furthermore, sorption is attractive for pharmaceutical removal because it is low energy and has different treatment configurations such as fixed-bed or mixed reactors, continuous flow or batch, and sorbent regeneration or disposal (Crittenden et al., 2012). The basis of this research is that removing pharmaceuticals from undiluted urine would be more effective and efficient than in centralized wastewater, particularly for pharmaceuticals primarily excreted in urine (Lienert et

al., 2007a, Winker et al., 2008a), however, the environmental impacts of using sorption processes to remove pharmaceuticals in urine is unknown.

Life cycle assessment (LCA) applied to urine source separation is an emerging research area with only one study considering removal of pharmaceuticals. The primary focus of several papers included the source separation system (i.e., urine piping, collection, and storage), fertilizer offsets, wastewater treatment offsets, and potable water offsets (Ishii and Boyer, 2015, Lam et al., 2015, Maurer et al., 2003, Remy, 2010). Remy (2010) conducted an LCA that included an ozonation process for pharmaceutical destruction in source separated urine, however they did not evaluate the specific toxicity of pharmaceuticals in the model. Previous LCA studies have evaluated the environmental impacts of pharmaceuticals in wastewater effluent. A study by Muñoz et al. (2008) concluded that pharmaceuticals were a significant contributor to the toxicity of the studied wastewater. Conversely, it was found that pharmaceuticals in decentralized hospital wastewater exhibited negligible environmental impact compared with the impacts generated by wastewater treatment (Igos et al., 2013, Igos et al., 2012). Advanced treatment of decentralized hospital wastewater would not decrease pharmaceutical toxicity in total wastewater effluent because the contribution of pharmaceuticals from hospital wastewater was much smaller than other sources (e.g., pharmaceutical usage at homes, office buildings, etc.) (Igos et al., 2012). Ort et al. (2010) estimated that hospital wastewater contributed 15% of pharmaceuticals to centralized wastewater. This suggests that an alternative approach to treating municipal wastewater at the community-level, such as urine source separation, could be more effective at reducing pharmaceutical loading to the environment. To date, there is not a published LCA study on urine source separation considering removal of pharmaceuticals by sorption process and corresponding reduction in toxicity. The goal of this research was to compare the overall

environmental and economic impacts of pharmaceutical removal from urine generated in a university community by centralized wastewater treatment, advanced treatment of centralized wastewater, and centralized and decentralized treatment of source separated urine. The pharmaceuticals investigated in this study were from the NSAID pharmaceutical class and included diclofenac (DCF), ibuprofen (IBP), ketoprofen (KTP), and naproxen (NPX). They were selected because of high ecotoxicity potential, prevalence, and variable removal rates in biological wastewater treatment (Hernando et al., 2006, Lienert et al., 2007b, Verlicchi et al., 2012). For the reasons given above, NSAIDs have been the focus of ion-exchange removal studies in urine. This study utilizes lab-scale experimental data to build a robust framework and conduct a baseline assessment that may be augmented with new pharmaceutical adsorption data as it becomes available.

2. Life cycle model

2.1. Scope of the study

The functional unit for this study was the conveyance, storage, pharmaceutical management (i.e., ion-exchange treatment), and nutrient management (i.e., struvite precipitation) of 11,184 m³ of urine, which is equivalent to the estimated annual production of urine at the University of Florida (UF) campus in Gainesville, Florida, USA. This study builds upon the LCA model developed by Ishii and Boyer (2015) by expanding the system boundary from residence halls to include the entire UF campus, and pharmaceutical management. Detailed calculations used to determine the functional unit are provided in the Supplementary Material 1 (SM1, section S1).

Figure 1 shows the wastewater management scenarios considered in this LCA. Scenario **A_{WWT}** serves as the baseline scenario and included combined collection of urine, feces, and greywater, and biological treatment at the UF Water Reclamation Facility. The upstream system

boundary includes operational inputs for potable flush water production at the nearby drinking water treatment facility. The construction and decommission phase of the wastewater treatment plant was not included in this assessment because they were assumed to be equal across all scenarios, thus negating the contributions of these phases to the environmental assessment. Scenario **B_{wwt,o3}** is a hypothetical scenario which included combined collection of urine, feces, and greywater, and biological treatment at the UF Water Reclamation Facility upgraded with an ozonation process for pharmaceutical destruction (Ternes et al., 2003). The construction phase for the ozone contactor and operational phase of the ozone process were included in this assessment. Decommission of the ozone system was not taken into consideration. It was assumed that no nutrients were recovered for reuse as fertilizer in **A_{wwt}** and **B_{wwt,o3}**. Land application of biosolids was excluded from the system boundary because the local utility ceased land application and currently disposes of biosolids in a landfill. Furthermore, the effect that urine source separation has on the composition of biosolids at the centralized wastewater treatment plant is unknown. Modeling the composition of biosolids at the wastewater treatment plant after urine source separation was considered outside the scope of this model.

Scenarios C–H were the hypothetical urine source separation scenarios. The system boundary includes the processes related to potable water production, urine source separation, and treatment (i.e., storage disinfection, pharmaceutical removal by ion-exchange, and struvite precipitation), centralized wastewater treatment, and wastewater discharge to surface water and landscape irrigation. For scenarios **C_{truck,landfill}** and **D_{truck,regen}**, urine was collected by a vacuum truck and transported to a central location for processing. In scenarios **E_{sewer,landfill}** and **F_{sewer,regen}**, urine was conveyed by vacuum sewer to a central location for processing. In scenarios **G_{decen,landfill}** and **H_{decen,regen}**, urine was collected and processed at the building level for

decentralized treatment. For scenarios $C_{\text{truck,landfill}}$, $E_{\text{sewer,landfill}}$, and $G_{\text{decen,landfill}}$, it was assumed spent anion exchange resin (AER) was transported and disposed of in a landfill. For scenarios $D_{\text{truck,regen}}$, $F_{\text{sewer,regen}}$, and $H_{\text{decen,regen}}$, it was assumed spent AER was regenerated using 5% NaCl, 50% methanol, and the brine was transported and incinerated at a cement kiln plant for energy recovery. The system boundaries do not include redistribution of struvite to agriculture. It was assumed that struvite fertilizer would replace commercial fertilizers used in A_{WWT} and $B_{\text{WWT,O3}}$ and that struvite fertilizer granules were comparable to commercial fertilizers, allowing the use of commercial fertilizer spreading equipment (Forrest et al., 2008). Furthermore, the ammonia, nitrous oxide, and phosphate emissions for struvite and commercial fertilizer (i.e., monoammonium phosphate (MAP)) were assumed to be equivalent due to identical emission factors and nutrient content (Nemecek and Kägi, 2007). However, the cadmium content of struvite and commercial fertilizers were considered an emission to land (i.e., 0.39 mg Cd/kg P_2O_5 in struvite and 97.5 mg Cd/kg P_2O_5 in MAP) (Lugon-Moulin et al., 2006, Ronteltap et al., 2007). Infrastructure for the urine source separation system (e.g., vacuum sewer, urine piping, and ion-exchange system) and operation (e.g., road transport, energy and chemical requirements) were included within the system boundary. Decommission of the urine diversion and treatment system was not taken into consideration.

2.2. Life cycle inventory

The data sources and design parameters used to develop foreground processes for each treatment scenario are provided in detail in the SM1 (section S2). The life cycle inventory included potable flush water production (section S2.1), centralized wastewater treatment (section S2.2), ozonation of wastewater (section S2.3), urine source separation infrastructure (section S2.4), urine collection by vacuum truck (section S2.5) or vacuum sewer (section S2.6), ion-exchange

infrastructure and treatment (section S2.7), struvite precipitation for nutrient recovery (section S2.8), and estimated pharmaceutical concentrations in urine (section S2.9). Background inventory data for each scenario were designed using existing components in two databases, the Ecoinvent unit processes (version 2.2) and the U.S. Life Cycle Inventory Database (USLCI) (Ecoinvent Centre, 2015, NREL, 2012). Data from the Ecoinvent database is based on either European, Swiss, or North American technologies published between 2007–2009. Data from the USLCI database is based on North American technologies or processes published between 2003–2008. European based data was adopted without any modification for this study.

2.3. Life cycle costing

The economic analysis included infrastructure and operational costs for pharmaceutical removal in urine by the alternative treatment scenarios. Net present value (NPV) was estimated using an interest rate of 3% (National Center for Environmental Economics, 2010). The sources and assumptions underlying all cost estimates are given in the relevant life cycle inventory sections in SM1 (section S2) and are listed in Table S9 (SM1, section S4). All infrastructure costs were updated to 2016 based on inflation. Labor costs were excluded in the cost analysis.

2.4. Life cycle impact assessment

The LCA model for all scenarios was constructed using SimaPro 8.0.3.14 software (PRé Consultants, 2014). The TRACI impact assessment method was used to evaluate the contributions of processes, generated, and avoided impacts to ten midpoint impact categories (U.S. EPA, 2014). This method was selected because the study pertains to wastewater treatment in a U.S. community and TRACI was developed by the U.S. Environmental Protection Agency. The ten midpoint impact categories (e.g., ozone depletion, global warming, smog, etc.) were evaluated with respect to a reference unit (e.g., kg CFC-11 eq, kg CO₂ eq, kg O₃ eq, etc.) and

normalized to obtain a single impact score, expressed in Person-Equivalent (PE). Normalization is a conversion step that compares the magnitude of impacts relative to a common reference. For TRACI 2.1, results were normalized to the average annual impact of a U.S. citizen using 2008 as the reference year (Ryberg et al., 2014).

The UNEP-SETAC toxicity model USEtox is the basis for the TRACI impact categories for human health non-carcinogenic, and ecotoxicity and are expressed in comparative toxic units (U.S. EPA, 2014). Non-carcinogenic human toxicity (CTUh) is characterized by the estimated morbidity increase in the total population per unit mass of emitted chemical (disease cases·kg⁻¹) and ecotoxicity (CTUe) is an estimate of the potentially affected fraction of species over time and volume per unit mass of chemical emitted (PAF·m³·day·kg⁻¹). Characterization factors for DCF, IBP, and NPX were obtained from literature (Alfonsín et al., 2014). The USEtox model was used to determine an ecotoxicity characterization factor for KTP using the model's substance database and ecotoxicity data from literature (Andersson et al., 2007, Hauschild et al., 2015, Morais, 2014). A characterization factor for human toxicity was not determined for KTP due to a lack of data. The characterization factors are listed in Table S10 (SM1, section S5).

2.5. Sensitivity and uncertainty analysis

The uncertainty of input parameters on the impact assessment results for each scenario was evaluated using the integrated Monte Carlo module in SimaPro. In each Monte Carlo analysis, 3000 iterations were conducted. Table S11 (SM1, section S6) lists all of the input parameters, range of variation, justification and assumed distribution considered in the uncertainty analyses. Variability of unit costs were also included to evaluate the uncertainty of assumed input operational costs (SM1, section S6, Table S12). Cost variability of magnesium oxide and liquid oxygen were excluded due to a lack of data. Infrastructure costs were assumed to remain

constant. Additional sensitivity analyses were conducted to evaluate the effect of assumed model inputs (Table S11) and unit costs (Table S12) on the environmental impacts for each scenario. The sensitivity analysis was conducted by varying each parameter individually between the minimum and maximum values. A parameter was considered sensitive if results varied from the baseline $\pm 10\%$.

3. Results and discussion

3.1. Overall comparison of scenarios

Figure 2 provides a comparison of the environmental impacts of the treatment scenarios, subdivided into the contributing processes (e.g., potable water, WWTP electricity, urine transport, etc.), generated impacts (e.g., nutrient and pharmaceutical emissions), and avoided impacts (e.g., fertilizer offsets). Alternatively, Figure S5 (SM1, section S7) shows the same total impact score of each scenario subdivided into the contributing mid-point impact categories. Together, these two figures provide a holistic view of the major contributing processes and impact categories to the total environmental impact. Non-normalized results for individual TRACI impact categories, excluding ecotoxicity and non-carcinogenic impacts are provided in SM1 (section S7, Figures S6–S13).

The order of decreasing total environmental impact was $B_{\text{WWT},\text{O3}} > A_{\text{WWT}} > E_{\text{sewer,landfill}} > F_{\text{sewer,regen}} > C_{\text{truck,landfill}} > G_{\text{decen,landfill}} > D_{\text{truck,regen}} > H_{\text{decen,regen}}$. The trend suggests that centralized wastewater treatment had greater environmental impacts than the source separation scenarios, primarily due to the potable water requirements for flushing, electricity for wastewater treatment, and nutrient emissions. Furthermore, struvite precipitation of source separated urine reduces nutrient emissions, offsets commercial fertilizer production, and reduces cadmium emissions due to commercial fertilizers. These results are similar to other LCA studies that found that potable

water savings, electricity savings, reduction in nutrient loading, and reduced cadmium emissions from commercial fertilizers in the environment are major benefits gained from urine source separation (Berndtsson, 2006, Ishii and Boyer, 2015, Lam et al., 2015, Lamichhane and Babcock, 2012, Ronteltap et al., 2007). Results of the Monte Carlo simulation indicate that uncertainty does not affect the overall trends for the total environmental impact, where A_{WWT} and $B_{WWT,O3}$ have greater observed environmental impact compared with the urine source separation and the vacuum sewer scenarios had the largest impact among the source separation scenarios. However, there was uncertainty between scenarios $C_{truck,landfill}$, $D_{truck,regen}$, $G_{decen,landfill}$, and $H_{decen,regen}$ at the 97.5% confidence interval suggesting that the difference in environmental impact of urine collection by vacuum truck or decentralized treatment is not significant. However, the total environmental impact for the resin disposal scenarios was less than the scenarios where resin was disposed of in a landfill (e.g., $C_{truck,landfill} > D_{truck,regen}$). Within impact categories, A_{WWT} and $B_{WWT,O3}$ maintained the greatest impact for all categories except ozone depletion. However, some uncertainty was observed within impact categories for the source separation scenarios, with the exception of the eutrophication impact category.

Replacing conventional fixtures with urine diverting flush toilets and waterless urinals would conserve 2.6×10^5 m³ of potable flush water and \$231,000 annual potable water savings. The implications of potable water savings would be of particular importance in areas that face water scarcity and quality issues (Ishii and Boyer, 2015). Accounting for the reduction in potable flush water, influent flow at the wastewater treatment plant would decrease by 17%. This reduction in influent flow could reduce the electricity requirements for wastewater treatment. A limitation of this study is that quantifying electricity use at the plant simplifies the impact urine source separation can have on centralized wastewater treatment. Jimenez et al. (2015) found that

urine source separation can reduce influent N and P loading and potentially eliminate the need for nitrification, reduce sludge age, and reduce chemical requirements for chemical P removal.

Table 1 provides a summary of the estimated economic impacts associated with infrastructure, energy, potable flush water, chemicals, and urine-based fertilizer revenue for each treatment scenario. The NPV and EAC of scenarios $C_{truck,landfill}$, $D_{truck,regen}$, $G_{decen,landfill}$, and $H_{decen,regen}$ vary from A_{WWT} by only ± 2 –13%. These scenarios could be considered comparable to A_{WWT} due to the preliminary nature and uncertainty of the economic evaluation. However, the economic costs of $B_{WWT,O3}$, $E_{sewer,landfill}$, and $F_{sewer,regen}$ was 21–45% greater than A_{WWT} . Due to the uncertainty of the input parameters and unit costs, Monte Carlo analysis show that the cost of each scenario increases and decreases at the 2.5 % confidence interval (CI) and 97.5% CI, respectively. Compared across scenarios, cost savings for urine source separation may be even greater (i.e., 48%–69% less than A_{WWT}) at the 2.5% confidence interval (CI). However, at the 97.5% CI, observed trends were the same as the baseline values. This suggests that scenarios $C_{truck,landfill}$, $D_{truck,regen}$, $G_{decen,landfill}$, and $H_{decen,regen}$ have lower environmental impact and similar or lower economic cost compared to A_{WWT} . This result is similar to Ishii and Boyer (2015) which concluded that urine source separation and struvite precipitation for maximum P recovery had significantly lower environmental impact but negligible cost differences, compared with centralized wastewater treatment. This suggests that the cost of additional pharmaceutical treatment of source separated urine would not limit implementation.

3.2. Urine source separation

The urine source separation scenarios had a lower impact for all impact categories compared with A_{WWT} and $B_{WWT,O3}$, with the exception of the ozone depletion impact category (Figure 2). Anion exchange resin is the major contributing process to the ozone depletion impact category

because of the trichloromethane solvent used to add quaternary amine functional groups to the polymer backbone for a type I AER (SM1, section S7, Figure S6) (Althaus et al., 2007). This result differs from Choe et al. (2013) who found that ion-exchange resins dominate all impact categories except for ozone depletion, however ion-exchange resin was modeled as a general polystyrene and did not include the additional functionalization step in resin manufacturing. Eutrophication is the greatest contributor to the total environmental impact in the source separation scenarios, which is primarily due to the N remaining in urine after struvite precipitation for P recovery (SM1, section S7, Figure S5 and Figure S10). However, Ishii and Boyer (2015) found that struvite precipitation for maximum P and N recovery had a greater environmental impact than struvite precipitation for maximum P recovery due to the chemical inputs. This suggests that alternative N removal or recovery technologies should be explored. Alternatively, direct application of stored liquid urine could serve as a complete nutrient source (Kirchmann and Pettersson, 1995), with the added benefit of reducing both N and P loading at the wastewater treatment plant and subsequent receiving waters. Coupled with the fact that MgO and struvite storage requirements had the second greatest environmental impact in $C_{truck,landfill}$, $D_{truck,regen}$, $G_{dececn,landfill}$, and $H_{decen,regen}$, application of liquid urine may reduce the total environmental impact. However, the social implications of applying liquid urine compared to a urine-derived solid fertilizer should be considered. In general, user acceptance surveys found that >79% of respondents approved from urine source separation technology in residence halls, public buildings, or workplaces (Ishii and Boyer, 2016, Lienert and Larsen, 2010), and 85% approved of urine-derived fertilizers (Lienert and Larsen, 2010). However, only 50% of farmers regarded urine fertilizer as a good idea primarily due to fear of liability (Lienert and Larsen, 2010). Farmers in Switzerland preferred a grainy and odorless ammonium nitrate fertilizer, but

they were willing to use odorous urine fertilizer in fields (Lienert et al., 2003). This suggests that a mineral fertilizer (e.g., struvite) may be more appropriate for application in urban areas compared with liquid urine, which could be applied in rural and agricultural settings.

Scenarios $E_{\text{sewer,landfill}}$ and $F_{\text{sewer,regen}}$ have the largest environmental impact among the source separation scenarios due to the vacuum sewer infrastructure and operation. This indicates that the method of urine collection and handling (e.g., vacuum sewer vs. vacuum truck vs. decentralized treatment) is a critical consideration during the design phase. The material and process inputs for the vacuum sewer system is largely dependent on the geographical size of the collection area. Conversely, the process inputs for collection by vacuum truck is dependent on both distance traveled (size of collection area) and volume of urine produced. As shown in Figure S15, the total normalized impact of the vacuum sewer exceeds vacuum truck collection when plotted as a function of total pipe length of the sewer system or distance traveled (km). This suggests that a vacuum sewer system would have a greater environmental impact than vacuum truck collection, regardless of the size of the collection area. Overall, centralized urine treatment collected by vacuum truck and decentralized urine treatment had the lowest environmental impact. However, the feasibility of implementing multiple decentralized systems on a large scale must be considered. Facilities and maintenance staff maintain all building services and operations (e.g., janitorial and maintenance) on campus. It is expected that maintenance staff would maintain the ion-exchange system and struvite precipitation operations. This would be a significant new task for maintenance staff to undertake and would likely require hiring personnel to handle these tasks or subcontracting to a private firm. The labor requirements and additional costs were not included in this model but could be considered in future work (Ramos et al., 2014). With respect to other types of communities, decentralized treatment may be

more appropriate in rural areas. For example, Wood et al. (2015) found that urine diversion coupled with conventional septic systems for greywater management exhibited the lowest economic cost and highest cost effectiveness for N mitigation for rural households.

The scenarios where spent resin was disposed of in a landfill had a slightly greater environmental impact than the scenarios where resin was regenerated and the waste regeneration solution (i.e., 5% NaCl, 50% methanol) was incinerated at a cement kiln plant. Although the production of methanol, salt, and potable water used for regeneration generates environmental impact, greater environmental offsets were achieved from incinerating the methanol-containing brine for energy recovery as opposed to fossil fuels used at the cement kiln plant (SM1, section S8, Figure S14). A limitation of the Ecosolvent model used to generate the life cycle inventory for brine incineration is that it represents Swiss technology, however plants may vary according to the kiln and flue gas treatment technology (Seyler et al., 2005). Furthermore, the fuel mix assumed in the Ecosolvent model is different compared to the U.S. cement kiln industry (Hanle, 2004). Choe et al. (2013) and Maul et al. (2014) found that salt requirements for the ion-exchange was the major contributor to the environmental impact of an ion-exchange process. This indicates that improving the sustainability of ion-exchange regeneration (e.g., brine incineration for energy recovery or brine recycling) can make regeneration more favorable than disposing of resin in a landfill. The potential benefits from incinerating the regeneration brine are two-fold: environmental offsets due to reduced fossil fuel consumption and ultimate destruction of the pharmaceuticals to prevent release to the environment. An additional driver for resin regeneration and brine incineration, as opposed to landfill disposal, is the potential for pharmaceuticals to end up in landfill leachate (Lu et al., in press). However, occurrence of pharmaceuticals in landfill leachate was not included within the LCA framework. Alternatively,

a semi-closed loop system may be developed by destroying pharmaceuticals in the regeneration brine by advanced oxidation processes to allow brine recycling (Zhang et al., 2015).

3.3. Pharmaceutical toxicity

Figure 3a shows the TRACI impact results for ecotoxicity (CTU_e) subdivided into the contributing processes, generated impacts, and avoided impact, and Figure 3b shows the ecotoxicity impact only due to pharmaceutical emissions in wastewater effluent discharged to surface water and reclaimed water. A similar figure for non-carcinogenic human toxicity (CTU_h) is given in SM1 (section S10, Figure S16). For brevity, this discussion focuses on ecotoxicity because the same general trends were observed for human toxicity. Overall, the order of decreasing total ecotoxicity was $B_{WWT,O3} > A_{WWT} > E_{sewer,landfill} > F_{sewer,regen} > C_{truck,landfill} > G_{decen,landfill} > D_{truck,regen} > H_{decen,regen}$ (Figure 3a). Ecotoxicity due to pharmaceutical emissions followed a decreasing trend of $A_{WWT} > C_{truck,landfill} = D_{truck,regen} = E_{sewer,landfill} = F_{sewer,regen} = G_{decen,landfill} = H_{decen,regen} > B_{WWT,O3}$ (Figure 3b).

As expected, A_{WWT} had the greatest ecotoxicity due to pharmaceutical emissions because biological treatment only achieves 28–87% pharmaceutical removal and can vary for individual pharmaceuticals (Fernandez-Fontaina et al., 2012, Hollender et al., 2009, Joss et al., 2005, Lindqvist et al., 2005, Rivera-Utrilla et al., 2013, Rosal et al., 2010, Salgado et al., 2012, Ternes, 1998). The fate of pharmaceuticals in raw wastewater is removal by adsorption to sludge and/or biotransformation, or discharged in the effluent (Cook et al., 2012). Furthermore, biological wastewater treatment does not maintain consistent treatment efficiencies as observed by negative mass balances in wastewater effluent (Blair et al., 2015). Although biosolids disposal was not included within the framework of the LCA, land application of biosolids is a common emission source for pharmaceuticals in wastewater treatment. It was estimated that 210–250 tonnes year⁻¹

of 72 pharmaceuticals and personal care products are land applied to U.S. soils from biosolids recycling, nationwide (McClellan and Halden, 2010). The pharmaceuticals investigated in this study, on average, were estimated to be removed 5–7% by sludge adsorption based on their sludge adsorption coefficient (K_d) (Alvarino et al., 2014, Blair et al., 2015, Carballa et al., 2008, Jelic et al., 2011, Jones et al., 2002, Joss et al., 2005, Radjenović et al., 2009, Sipma et al., 2010). High variability of overall removal suggests that biological wastewater treatment cannot consistently achieve effective removal of pharmaceuticals.

Scenario $B_{WWT,O3}$ had the lowest pharmaceutical ecotoxicity due to high pharmaceutical destruction (53–98%) (Hollender et al., 2009, Margot et al., 2013, Rosal et al., 2010, Ternes et al., 2003), however it exhibits the greatest total ecotoxicity due to the additional ozone process (e.g., ozone contactor infrastructure, electricity, liquid oxygen, water, and transport requirements). One of the limitations of this model is that only the ecotoxicity of the parent compound is considered and reduction in ecotoxicity is directly related to its removal. However, ecotoxicity studies have shown that more toxic byproducts may be formed after ozonated wastewater compared to the pre-ozonated water and would require an extended contact time, or an additional treatment step such as sand filtration or activated carbon, to remove the oxidation byproduct (Magdeburg et al., 2012, Sánchez-Polo et al., 2008, Stalter et al., 2010).

For the urine source separation scenarios, the total ecotoxicity impact is 90% less than A_{WWT} and $B_{WWT,O3}$, primarily due to the reduction in potable water use and electricity at the wastewater treatment plant (Figure 3a). Indirect toxicity of producing auxiliary materials and energy for potable water production and electricity use at the wastewater treatment plant is originated by several substances emitted to water during electricity production and lime sludge disposal (for potable water production only). As shown in Figure 3b, the toxicity due to

pharmaceutical emissions is the same for all urine source separation scenarios because equivalent pharmaceutical removal was achieved by ion-exchange. In general, pharmaceutical ecotoxicity followed a decreasing order of IBP > DCF > KTP > NPX. The fact that DCF removal was highest in these treatment scenarios (98% removal) but remains the second most toxic pharmaceutical highlights the importance of evaluating the reduction in toxicity of each pharmaceutical as opposed to the average mass removal. Considering the majority of pharmaceuticals in centralized wastewater come from human urine, separation and treatment of this unique waste stream may be the most effective pharmaceutical management strategy. An expressed limitation of LCA regarding toxicity include not being able to accurately evaluate mixture toxicity (Muñoz et al., 2009). Toxicity studies have observed antagonistic or synergistic toxicity effects in pharmaceutical mixtures (Pomati et al., 2008). However, a recent study by Watanabe et al. (2015) found that concentration addition and independent action are accurate at predicting chronic mixture toxicity of pharmaceuticals at environmentally relevant concentrations. In addition, LCA does not evaluate endocrine disruption potential due to limited information and lack of an epidemiological framework (Finkbeiner et al., 2014). Another limitation of this model is that the pharmaceuticals evaluated in this study (i.e., non-steroidal anti-inflammatory drugs) are not comprehensive of all pharmaceutical compounds with respect to toxicity and removal efficiency.

3.4. Model sensitivity

Supplementary Material 2 (SM2) show the results of the sensitivity analysis as a percent change in each urine treatment scenario's impact within an impact category, total impact, and total cost, relative to the baseline assumption; impact changes within varying ranges (i.e., 10–19%, 20–49%, and $\geq 50\%$) are highlighted in color. Overall, the most sensitive assumptions to the model

were pharmaceutical concentrations in urine, TN and TP in urine, WWTP energy, storage time, and resin capacity. Similar to Ishii and Boyer (2015), the various impact categories within the treatment scenarios were sensitive to the assumed concentration of P in urine, the assumption for electricity use at the wastewater treatment plant, and storage time. In general, A_{WWT} and $B_{\text{WWT},\text{O}_3}$ maintained the greatest total impact throughout the entire sensitivity analysis. Although the model was sensitive to the assumed WWTP energy requirements, and TN and TP concentrations in urine, the observed trend for total environmental impact for the scenarios remained the same.

The source separation scenarios were sensitive to the assumed pharmaceutical concentrations in urine for three out of ten impact categories (i.e., ozone depletion, acidification, and respiratory effects). A decrease or increase in impact was observed when pharmaceutical concentrations were minimized or maximized, respectively. This was attributed to the decrease or increase in resin and chemical requirements (i.e., NaCl and methanol for regeneration), and size of the ion-exchange vessel. An inverse relationship (e.g., a decrease in impact when pharmaceutical concentrations were maximized) was observed for $D_{\text{truck,regen}}$, $F_{\text{sewer,regen}}$, and $H_{\text{decen,regen}}$. This was due to the greater methanol requirements for regeneration and subsequent fossil fuel offsets due to brine incineration at the cement kiln plant. Uncertainty regarding the estimated pharmaceutical concentrations in urine may be improved with increased sampling campaigns, improved understanding of pharmaceutical consumption, and modeling procedures. A model developed by Winker et al. (2008b) to predict pharmaceutical concentrations in urine had a strong correlation but only accounted for prescribed pharmaceuticals, however a significant amount of pharmaceuticals may be purchased over-the-counter. Furthermore, there is a lack of data regarding the amount of over-the-counter pharmaceuticals actually consumed. A similar trend was observed when the column was sized to achieve maximum IBP removal

compared to the baseline, which was sized to achieve maximum DCF removal. Due to the low capacity of the AER for IBP compared to DCF resin, chemical requirements and corresponding fossil fuel offsets from brine incineration increased, resulting in a decrease in impact for the regeneration scenarios, increase in impact for the landfill scenarios, and an increase in total cost.

The results of the economic sensitivity analysis are shown in SM2. Table S2.2 (SM2, section S2) shows the percent change from the baseline NPV values for each scenario, respectively. Table S2.3 (SM2, section S2) shows the percent difference in NPV compared with Scenario A. The cost of urine diverting flush toilets and waterless urinals was the most sensitive cost for the urine source separation scenario and also the largest single economic cost. Urine-diverting flush toilets are not widely used compared with conventional toilets. Considering the material inputs of these fixtures do not differ from conventional toilets, it is reasonable to expect that increasing demand would decrease market price. If market value of these fixtures cost the same as conventional fixtures, urine source separation would cost 18–54% less than A_{WWT}. Total cost was also sensitive to utility rates for potable water and electricity, particularly for A_{WWT} and B_{WWT,O3}. For example, when the cost of potable water was minimized, the cost of urine source separation was 24–75% greater than A_{WWT}. However, when the cost was maximized, urine source separation cost 63–74% less than A_{WWT}. A similar trend was observed for electricity costs. This suggests that potable water or electricity savings may be a driver for or against implementation of urine source separation, depending on the community. Finally, the total cost was sensitive to the assumed interest rate. In general, as interest rate increased, the cost of B_{WWT,O3} increased compared to A_{WWT}, and the cost of urine source separation decreased compared to A_{WWT}.

4. Conclusions

There are numerous environmental benefits associated with urine source separation (e.g., potable flush water savings, electricity savings at the wastewater treatment plant, and nutrient offsets), Compared with centralized wastewater treatment, ozonation of wastewater had a higher environmental impact and economic cost, urine collected by vacuum sewer had lower environmental impact and higher economic cost, and urine collected by vacuum truck collection or treated at decentralized locations had lower environmental impact and similar economic cost. Urine source separation can achieve a high reduction of pharmaceutical toxicity and comparatively low total toxicity from the treatment process compared with B_{WWT,O3}. Additional sorption studies are needed to evaluate the removal of various pharmaceutical compounds from diverse therapeutic classes and chemical structure, the results of which could be incorporated into a future version of this LCA framework. The benefit of this LCA framework is that the environmental impact and economic cost of alternative sorbents can easily be evaluated. Although the AER used for this study may not be the most appropriate to remove all pharmaceuticals, multiple sorbents may be utilized which have a higher selectivity and capacity for the pharmaceuticals of concern. Using a more selective and higher capacity resin would decrease the resin requirements and subsequent costs. In conclusion, the framework created and tested herein estimates the environmental and economic impacts of alternative treatment technologies that remove pharmaceuticals and recover nutrients in source separated urine in a community setting.

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Table 1. Capital and operation and management (O&M) costs, net present value (NPV) for each urine treatment scenario; negative values indicate cost, positive values indicate revenue

Capital Costs	A WWT	B WWT,O3	C truck,landfill	D truck,regen	E sewer,landfill	F sewer,regen	G decen,landfill	H decen,regen
Fixtures ^a	-\$1,590,000	-\$1,590,000	-\$4,880,000	-\$4,880,000	-\$4,880,000	-\$4,880,000	-\$4,880,000	-\$4,880,000
Vacuum sewer	\$0	\$0	\$0	\$0	-\$651,500	-\$651,500	\$0	\$0
Urine piping	\$0	\$0	-\$171,000	-\$171,000	-\$171,000	-\$171,000	-\$171,000	-\$171,000
Urine storage ^b	\$0	\$0	-\$1,300,000	-\$1,300,000	-\$1,300,000	-\$1,300,100	-\$69,000	-\$69,000
Ozone system	\$0	-\$2,810,000	\$0	\$0	\$0	\$0	\$0	\$0
Fiberglass IX vessel	\$0	\$0	-\$8,600	-\$8,600	-\$8,600	-\$8,600	-\$8,600	-\$8,600
Struvite storage	\$0	\$0	-\$400	-\$400	-\$400	-\$400	-\$400	-\$400
O&M Costs								
Diesel fuel ^c	\$0	-\$160	-\$2,900	-\$3,000	-\$4	-\$100	-\$12	-\$100
Potable flush water	-\$234,000	-\$234,000	-\$1,300	-\$1,300	-\$1,300	-\$1,300	-\$1,300	-\$1,300
Electricity at WWTP ^d	-\$36,300	-\$36,300	-\$1,700	-\$1,700	-\$1,700	-\$1,700	-\$1,700	-\$1,700
Ozone operation ^e	\$0	-\$19,900	\$0	\$0	\$0	\$0	\$0	\$0
Vacuum sewer	\$0	\$0	\$0	\$0	-\$36,900	-\$36,900	\$0	\$0
IX resin	\$0	\$0	-\$52,800	-\$52,800	-\$52,800	-\$52,800	-\$52,800	-\$52,800
IX operation ^f	\$0	\$0	-\$1,500	-\$9,000	-\$1,500	-\$9,000	-\$1,500	-\$9,000
Struvite revenue ^g	\$0	\$0	\$12,100	\$12,100	\$12,100	\$12,100	\$12,100	\$12,100
NPV (\$M) ^h	-\$10.1	-\$14.6	-\$10.5	-\$10.7	-\$12.2	-\$12.4	-\$8.83	-\$9.05
2.5% CI	-\$36.2	-\$41.5	-\$16.0	-\$16.7	-\$18.3	-\$18.9	-\$11.2	-\$11.8
97.5% CI	-\$7.47	-\$11.9	-\$7.60	-\$7.68	-\$9.28	-\$9.40	-\$7.04	-\$7.12

^a Cost of conventional toilets and urinals (Scenarios A and B) or urine diverting flush toilets and waterless urinals (Scenarios C–F)

^b Includes centralized (Scenarios C and D) and decentralized (Scenarios C–F) urine storage

^c Cost of diesel for all unit processes (e.g., ozonation process, urine, resin disposal to landfill or brine disposal to cement kiln plant, and/or struvite collection)

^d Only pertains to electricity use based on influent flow at wastewater treatment

^e Includes liquid oxygen, potable water, diesel, and energy requirements

^f Includes potable water, chemical (e.g., NaCl, methanol), and energy requirements

^g Net balance of MgO costs for struvite precipitation and value of struvite

^h 60 year planning horizon and 3% interest rate

ⁱ 3% interest rate

Figure 1. Treatment schematic for scenarios A–H (light gray horizontal arrows) and contributing processes. Single black lines represent urine flow, grey lines represent ion-exchange resin flow, and double black lines represent struvite flow. The solid lines indicate transport through pipes and dashed lines represent road transport by truck.

Figure 2. Normalized TRACI impact score for all scenarios (a) centralized wastewater treatment scenarios (A_{WWT}, B_{WWT,O3}), and (b) urine source separation scenarios (C–H). Each colored bar represents input processes (e.g., potable water, electricity use at the WWTP, urine source separation (USS) infrastructure), avoided impacts (e.g., fertilizer offsets, brine incineration), and generated impacts (e.g., nutrient discharge, pharmaceutical discharge). The brackets around each error bar represent the 95% confidence interval resulting from Ecoinvent database distributions from the Monte Carlo uncertainty analysis.

Figure 3. Comparison of ecotoxicity impact ($\text{CTUe} = \text{PAF} \cdot \text{m}^3 \cdot \text{day}$) due to (a) contributing processes (e.g., flush water, urine transport) and generated emissions (e.g., nutrient discharge, pharmaceutical discharge) and avoided impacts (e.g., fertilizer offsets, brine incineration) in each scenario and (b) pharmaceutical emissions only. The brackets around each error bar represent the 95% confidence interval resulting from Ecoinvent database distributions from the Monte Carlo uncertainty analysis.

Graphical Abstract