Life-Cycle Assessment of Urine Diversion and Conversion to Fertilizer Products at the City Scale

by

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PREFACE

Separate collection of urine to recover nitrogen and phosphorus has been advocated to enhance the sustainability of water management and food production. Urine could provide a renewable source of nitrogen and phosphorus, which are currently extracted from nonrenewable resources. Urine diversion also has the potential to prevent nutrients from entering water bodies and to reduce the amount of energy and chemicals needed to treat wastewater. However, urine diversion would require systems to collect urine, produce urine-derived fertilizers, and to ship them, all of which have their own environmental impacts. This thesis explores the greenhouse gas emissions, cumulative energy demand, freshwater use, eutrophication potential, and acidification potential of systems that recover urine compared to those that do not. It evaluates the importance of location-specific factors by focusing on three locations, and then by conducting further sensitivity analysis. This work has been submitted to the journal *Environmental Science & Technology* (currently in review).

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LIST OF ABBREVIATIONS

LCA	Life Cycle Assessment
p.e.	Person Equivalent
USLCI	United States Life Cycle Inventory
GREET	The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation Model
TRACI	Tool for Reduction and Assessment of Chemicals and Other Environmental Impacts
GWP	Global Warming Potential (100 year)
CED	Cumulative Energy Demand
EP	Eutrophication Potential
AP	Acidification Potential
N	Nitrogen
P	Phosphorus
TN	Total Nitrogen
A2O	Anaerobic/Anoxic/Oxic (Wastewater Treatment Process)
RO	Reverse Osmosis
COD	Chemical Oxygen Demand
BOD5	5 Day Biochemical Oxygen Demand
TSS	Total Suspended Solids
VSS	Volatile Suspended Solids
TKN	Total Kjeldahl Nitrogen
SRT	Solids Retention Time
OHO	Ordinary Heterotrophic Organism
PAO	Phosphorus-Accumulating Organism
MLE	Modified Ludzack-Ettinger
MLSS	Mixed Liquor Suspended Solids
RAS	Return Activated Sludge
MLR	Mixed Liquor Recirculation
WAS	Waste Activated Sludge
TP	Total Phosphorus
WWTP	Wastewater Treatment Plant
DA	Direct + aeration emissions
DAS	Direct + aeration + substrate emissoins
5SB	5-Stage Bardenpho
SAB	Single Aeration Basin

ABSTRACT

Urine diversion has been proposed as an approach for producing renewable fertilizers and reducing nutrient loads to wastewater treatment plants. Life cycle assessment was used to compare environmental impacts of the operations phase of urine diversion and fertilizer processing systems (via 1) a urine concentration alternative and 2) a struvite precipitation and ion exchange alternative) at a city scale to conventional systems. Scenarios in Vermont, Michigan, and Virginia were modeled, along with additional sensitivity analysis to understand the importance of key parameters, such as the electricity grid and wastewater treatment method. Both urine diversion technologies had better environmental performance than the conventional system, and led to reductions of 29-47% in greenhouse gas emissions, 26-41% in energy consumption, approximately half the freshwater consumption, and 25-64% in eutrophication, while acidification ranged between a 24% decrease to a 90% increase. In some situations wastewater treatment chemical requirements were eliminated. The environmental performance improvement was usually dependent on offsetting the production of synthetic fertilizers. This study suggests that urine diversion could be applied broadly as a strategy for both improving wastewater management and decarbonization.

INTRODUCTION

About half of the world food supply depends on synthetic fertilizers produced from nonrenewable resources [1].Phosphate rock is used to produce phosphorus fertilizers. While the extent of the resource base is contested, supply is finite, demand has increased partly due to increased meat consumption and biofuel production, and supplies are dominated by a few countries. [2–5] Production of nitrogen fertilizer depends on natural gas, and is responsible for about 1.2% of world energy use and associated greenhouse gas emissions. [6,7] Prices for phosphate rock and other fertilizer commodities have fluctuated as much as 800% in recent years, which has led to food riots in many countries.[3,4,8] Given the impacts and resource constraints of conventional fertilizers, renewable and reliable alternatives are needed.

Food consumption by humans is the principal source of these vital nutrients in domestic wastewater, and significant resources are invested to remove them to protect the aquatic environment. Water and wastewater systems consume about 3-4% of the total electricity in the United States, with nutrient removal often being one of the most energy intensive processes.[9,10] Some propose separately collecting urine and using it to produce fertilizer.[11,12] Although it comprises less than 1% of wastewater volume, urine contains approximately 50% of the phosphorus and 80% of the nitrogen contained in domestic wastewater. [13–15] As utilities increasingly focus on sustainability, large-scale urine diversion has the potential to improve regional wastewater management, recover essential resources and reduce energy consumed in processes such as aeration. [11,16–19]

Compared to synthetic fertilizers, urine-derived fertilizers recover important nutrients, can be as effective at stimulating plant growth, and contain lower levels of heavy metals. [19–26] However, processing fertilizers from urine will have environmental impacts. [15] Collecting and transporting urine will require new infrastructure systems, such as pressurized pipe networks or truck collection.

Use of acetic acid or other chemicals may be needed to prevent the spontaneous release of ammonia gas and formation of precipitates that clog piping infrastructure. [15,27–29] Urine concentration, through processes such as reverse osmosis, freeze thaw, or distillation, may be required to make nutrient concentrations in urine, which are much lower than synthetic fertilizer, high enough for efficient agricultural application. [15,30–34] Alternatively, nutrients may be concentrated through removal processes such as struvite precipitation, ammonia capture via ion exchange, or urea adsorption. [15,20,35–41] Additional treatment to deactivate pathogens and remove pharmaceuticals found in urine may also be needed.

Life Cycle Assessment (LCA) is well suited to compare the environmental performance of urine diverting systems to conventional systems, determine environmental hotspots, and highlight trade-offs and opportunities for system improvement.[42,43] LCA has been used to compare a range of wastewater treatment alternatives, [44–47] and in most cases has indicated that urine diversion has lower environmental impacts than conventional systems . [13,14,48–51] However, these studies have focused on small scale systems, have evaluated only a few locations and urine-derived fertilizers, and simplified how diverting urine will affect wastewater treatment plants. These studies measure changes to wastewater through volume reduction or a static offset for denitrification, which may not capture significant changes to wastewater treatment as nutrient ratios change, or how urine diversion could change treatment configurations.[48,49,52–54]

This study expands upon previous research by evaluating the environmental impacts of urine diversion and conversion to fertilizer relative to conventional alternatives in large and diverse settings, and by a more detailed assessment of how this will affect wastewater treatment. This conventional alternative manages urine through the wastewater system and produces and transports equivalent amounts of nutrients in the form of synthetic fertilizer. The relative differences between these two different approaches are quantified. Wastewater treatment is modeled in detail to better account for the ramifications of urine diversion. Three distinct locations, namely the States of Vermont, Michigan, and Virginia (referred to subsequently as scenarios) are considered to explore how important parameters such as population, extent of nutrient removal at wastewater treatment plants, electricity grid fuel mix and the amount of urine-derived fertilizer produced influence the environmental performance. Sensitivity analysis is conducted using Monte Carlo in order to further evaluate these parameters and the uncertainty of many others.

2. METHODS AND MATERIALS

2.1 Urine Processing Alternatives

Two distinct urine-derived fertilizer alternatives were evaluated to represent the range of products that can be produced. They consist of (1) concentrated urine, where organics such as pharmaceuticals are removed from diverted urine through activated carbon and urine is subsequently concentrated by reverse osmosis (RO) and then heat pasteurized, and (2) struvite and ammonium sulfate, where urine is processed to produce struvite through precipitation and ammonium sulfate through ion exchange. Use of urine-derived fertilizer products are compared to commercial fertilizers. For the urine-derived fertilizer alternatives it was assumed that 70 percent of urine in each of the three scenarios considered was diverted for fertilizer production. This was done to simulate large-scale collection within these locations but to allow for some inefficiency in collection. As shown in Figure 1, production and distribution of flushwater, collection of wastewater (including separated urine), production and transportation of fertilizers, and wastewater treatment were included in the scope of the study to capture system-wide differences.

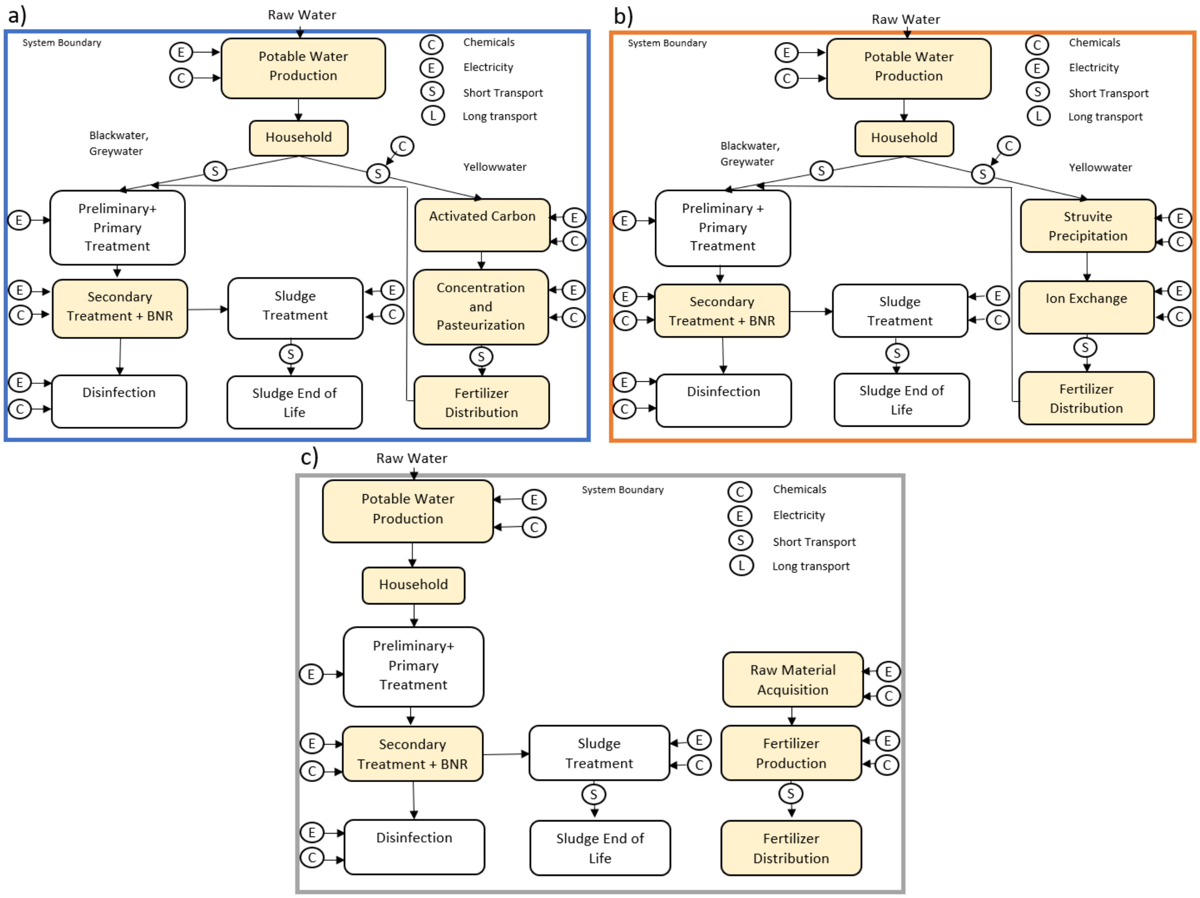

Figure 1 a-c. System Diagram for each alternative.
a) The urine concentration alternative, b) the struvite and ammonium sulfate alternative, c) the conventional system. Yellow boxes indicate that a process is either unique to that alternative, or that urine diversion significantly affects its environmental impact.

The inputs to treat and distribute flush water were determined using the ratio of surface and groundwater treated in each location,[55] and literature data for both types of treatment. [48,56,65–67,57–64] When urine was diverted, urine diversion toilet flush volumes were used. In the conventional alternative, for people not using urine-diverting toilets, and during defecation, low-flow toilet flush volumes were used, as shown in Tables 11 & 12. When urine is diverted, acetic acid is added to stabilize it, followed by transportation to a fertilizer production center via a pressurized pipe system.

Magnesium oxide is added to precipitate phosphorus as struvite and the remaining ammonium from the effluent is captured through ion exchange using a resin such as Dowex Mac

3.[39] The exhausted resin is regenerated with 3 M sulfuric acid, producing a liquid ammonium sulfate fertilizer. Additional acetic acid is needed for the concentrated urine fertilizer to consistently maintain nitrogen in the urea form. Following pharmaceutical removal using activated carbon, urine is concentrated to a fifth of its original volume using reverse osmosis with an energy recovery device (ERD) and then heat pasteurized. Chemical and energy inputs for regeneration of activated carbon[68–72] and reverse osmosis membrane cleanings[73] are included. Effluents from the urine-derived fertilizer production facilities are sent to the wastewater treatment plant, and the urine-derived fertilizers are trucked to a regional fertilizer distributor.

The methodology described in Hilton et al.[74] is used to model wastewater treatment for all alternatives to determine electricity consumption, chemical consumption, secondary sludge production, water and air emissions. All alternatives assumed equal amounts of feces and greywater, steady state conditions, and compliance with all regulatory requirements. Processes that were equivalent in magnitude between alternatives, such as primary sludge treatment and hauling screenings to landfills, were excluded. Further details can be found in the supplemental materials, Figure S1, and Hilton et al.[74]

The production of urea and mono-ammonium phosphate fertilizers and transportation to the regional fertilizer distribution center was used to ensure all alternatives provided the same mass of nitrogen and phosphorus as fertilizer. These synthetic fertilizers were added in the conventional and both diversion alternatives to provide equal amounts of nitrogen and phosphorus despite differing nutrient recovery ratios. Transportation from the regional fertilizer distribution center and application at the farm were not analyzed, as previous research did not find plant uptake and runoff from urine-derived fertilizers to differ from synthetic fertilizers.[25,75–77]

2.2 Life Cycle Assessment

The treatment of one person equivalent's (p.e.) wastewater for one year is the functional unit of analysis used. Treatment of all wastewater produced (including urine as appropriate) is considered because urine diversion can lead to significant reductions in the nitrogen and phosphorus of wastewater arriving at the treatment plant, and can significantly affect treatment. All alternatives provided equal masses of nitrogen and phosphorus in fertilizer. Environmental burdens of capital equipment and the end of life of wastewater and fertilizer infrastructure were excluded because the operational phase impacts are expected to dominate. [78–81]

Parameters used for the life cycle inventory and mass balance were obtained from literature sources and pilot scale systems, and can be found in Tables 1, 11 and 13. The United States Life Cycle Inventory (USLCI) was used for most unit processes, though Ecoinvent was used when unit processes were not available.[82,83] A Life Cycle Impact Assessment was conducted using global warming potential (GWP), cumulative energy demand (CED), freshwater use,[84] eutrophication potential (EP), and acidification potential (AP). These categories represent key impacts for changes in energy use, chemical manufacturing, water quality, and water use that are caused by urine diversion.

Table 1. Important parameters to model urine collection and fertilizer production.

Process	Parameter	Value	Unit	Notes and Sources
Home/Collection	Flushes per person per day	3.8,5.14	/pe · day	Urine only, then total.[85–90]
	Conventional: water per flush	4.84	L/flush	Also used for feces flushes in UD toilets

	Urine diversion: water per flush	0.165	L/flush	Used for urine-only flushes. Personal conversation Raye-Leonard [18]
	5% acetic acid added	0.033-0.04	L/L urine and flushwater	Struvite and ammonium sulfate (Calculated) then Urine Concentration (Experimentally determined [25])
Struvite and Ammonium Sulfate Production	Mg:P ratio for struvite	1.5:1		[48,54,91,92]
	Sulfuric acid per kg N	16.7	liters/kg N	18%. Tarpeh, personal conversation.
	N and P recovery	96, 96	%	[39,48,54,93,94]
Concentrated Urine Production	RO electricity consumption	0.009	kWh/l removed	Noe-Hays, Personal Communication
	N & P Recovery	95, 99	%	[95,96]

2.3 Description of Scenarios Evaluated

Three scenarios were modeled to provide an initial assessment of how location-specific factors affect the environmental merits and drawbacks of urine diversion. The Vermont scenario represents a smaller urban community without strict nitrogen effluent limits located in a largely rural state. The Michigan scenario was developed as a statewide average and was constructed by categorizing the range of communities in the State, the types of wastewater treatment plants found, and wastewater treatment volumes. The Virginia scenario represents a more densely-populated urban location with strict effluent limits. Further description of these scenarios can be found in the supplemental materials, Tables 2 and 14-20, and Hilton et al. [74] All alternatives were evaluated for each scenario.

Table 2. Comparison of Three Scenarios.

Item	**Vermont**	**Michigan**	**Virginia**
Description	Largely rural state with small to mid-size communities	Large state with diverse range of community sizes	Stringent effluent discharge standards
Population Modeled	25,000	150,000	350,000
Effluent Discharge Standards	Secondary, P limits	Secondary, P, some ammonia and TN limits	Advanced Secondary, stringent TN and P limits
Wastewater Treatment Process(es)	Single Aeration Basin	Single Aeration Basin, Nitrification, A2O	5-Stage Bardenpho
Typical Distance to Fertilizer Distributors	50	50	41
GWP of Electricity (kg CO2e/kWh)	0.107	0.544	0.450

2.4 Sensitivity Analysis

Sensitivity analysis was conducted to evaluate the robustness of the results, test urine diversion in a broader range of contexts, and to elucidate how model parameters and key assumptions influenced the environmental performance of urine diversion. Twelve separate simulation scenarios were created. As shown in Figure S2, six of these simulation scenarios modeled the 5-Stage Bardenpho treatment plant because it had the highest level of nutrient removal, while six modeled the single aeration basin with phosphorus removal because it had the lowest level of nutrient removal. Three electric grids, coal, natural gas, and renewable comprised of 50% wind and 50% hydropower were considered for each wastewater treatment type. Both the urine concentration, and struvite and ammonium sulfate urine derived fertilizer alternatives were compared, given six simulations for each wastewater treatment type. Table 21 lists the distributions of each parameter used. The Excel plugin Simvoi was used to conduct a Monte Carlo analysis with 10,000 repetitions for each sensitivity scenario[97].

3. RESULTS

3.1 Life Cycle Impacts Across Scenarios

Urine diversion consistently provides improved environmental performance relative to the conventional system for each scenario for all impact categories, except AP, as shown in Table 3. Both diversion alternatives reduced the GWP, CED, freshwater use, and EP categories from anywhere between 26% to 64%. The urine concentration alternative typically led to larger improvements than the struvite and ammonium sulfate alternative. Urine concentration alternatives decreased the AP modestly compared to the conventional alternative for all scenarios (12-24%), while struvite and ammonium sulfate alternatives increased the AP by 34% to 91% relative to the conventional alternative. Figures 2, 7, and 8 provide the relative differences in environmental performance for each alternative.

Table 3. Life Cycle Impacts per Scenario

Scenario	**Alternative**	**GWP**	**CED**	**Freshwater Use**	**Eutrophication Potential**	**Acidification Potential**
		kg CO_2e	MJ	m^3	kg N eq	kg SO_x eq
Vermont	Urine Concentration	14.6	297	7.28	1.19	0.0510
	Struvite and Ammonium Sulfate	19.7	313	7.37	1.27	0.111
	Conventional	27.6	450	13.7	3.27	0.0581
Michigan	Urine Concentration	30.1	441	7.66	1.44	0.123
	Struvite and Ammonium Sulfate	33.5	456	7.78	1.51	0.180
	Conventional	47.9	616	15.1	3.58	0.135

Virginia	Urine Concentration	22.9	376	6.67	0.295	0.0728
	Struvite and Ammonium Sulfate	26.1	382	6.73	0.302	0.130
	Conventional	36.8	637	12.8	0.405	0.0941

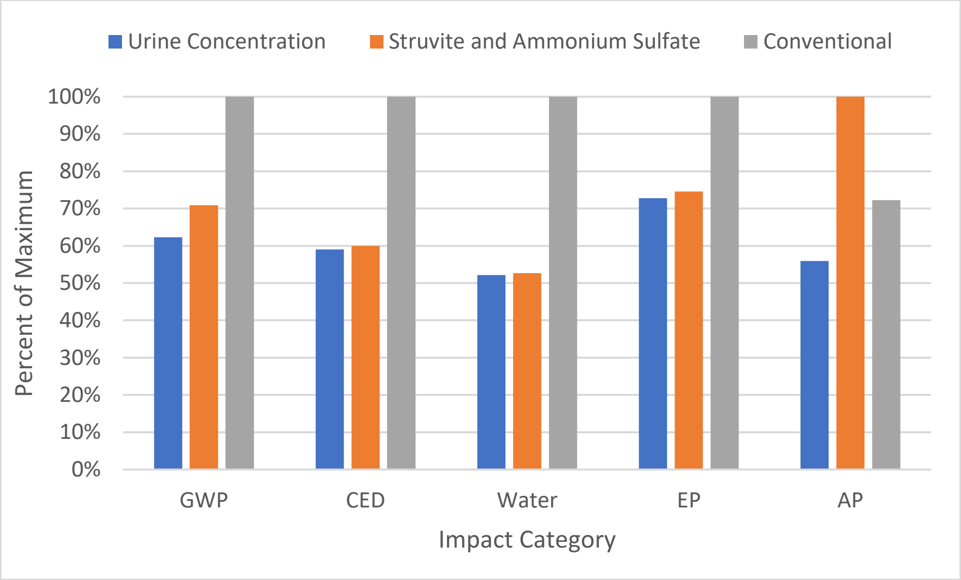

Figure 2. Normalized impacts in Virginia Scenario.
Total impacts in each alternative normalized to the maximum value in each category.

The magnitude of environmental impacts differed substantially between the three scenarios. Michigan had the highest GWP, CED, and AP impacts, while Vermont had the lowest. Much of this is because Michigan’s electricity grid is comprised primarily of fossil fuels and uses natural gas to thermally dry sludge, while Vermont’s electricity grid is mostly comprised of renewable energy sources. The Vermont scenario had an EP approximately four times larger than in Virginia as a result of the large differences in effluent standards. The urine diversion alternatives in states with less stringent effluent standards (Vermont and Michigan) saw the largest decreases in EP. The differences between the urine concentration, and struvite and ammonium sulfate alternatives were smaller for scenarios where the environmental impacts of producing electricity were larger, such as in Michigan.

3.2 Life Cycle Impacts by Process

Figure 3 shows the contribution of system components to greenhouse gas emissions for the Virginia scenario (see Figures S5-S18 for all impact categories and scenarios). Wastewater treatment dominated the eutrophication potential (81-99%), was usually responsible for the largest proportion of impacts in GWP (46-56%) and CED (35-49%) categories, and was a major contributor to AP (16-64%). Fertilizer production had the next largest impacts in the GWP (15-38%), CED (17-30%), and EP (0-17%) categories, and was a major contributor to AP (9-63%). In Michigan and Vermont, the EP from fertilizer production was negligible relative to its contribution from wastewater effluent. Potable water production and urine collection respectively had the next largest impacts in the GWP and CED categories. The largest contributor to AP was sulfuric acid (36%-58% when producing ammonium sulfate) followed by acetic acid (11-17% when producing ammonium sulfate, 20%-48% when concentrating urine).

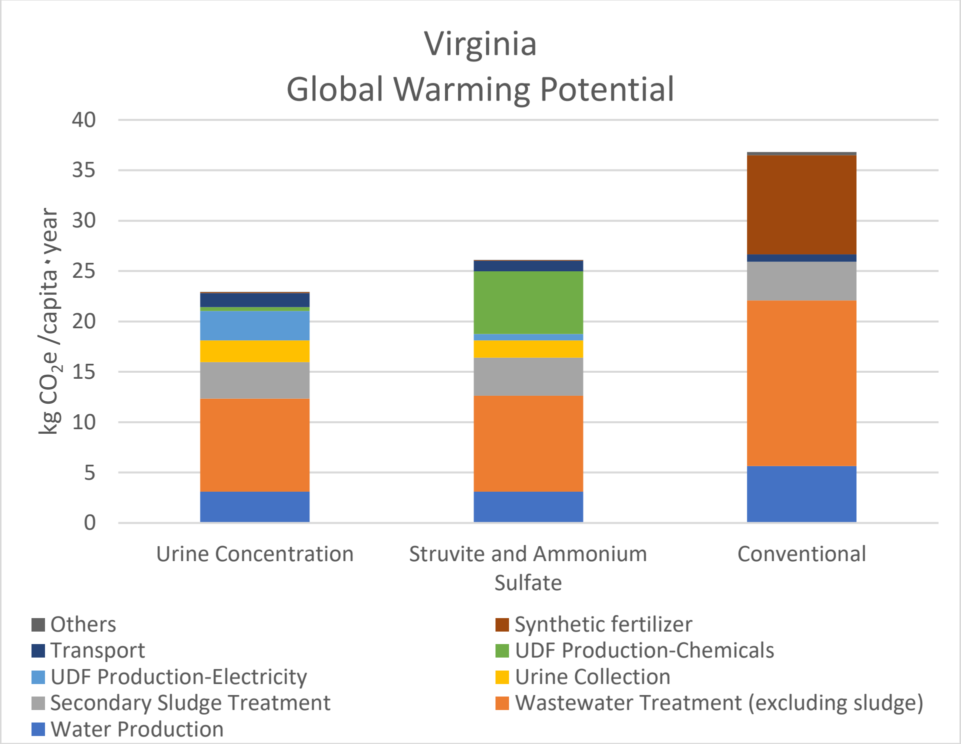

Figure 3. GWP by Process.
Global warming potential of the Virginia alternatives broken down by process.

In the conventional alternative, 10.4 cubic meters of water were needed per person per year for flushing excluding leaks between the drinking water plant and the consumer. This decreases to 5.3 and 3.1 cubic meters for 70% and 100% urine diversion, respectively. Reduced flush volumes from urine-diverting toilets were responsible for the majority of decreased freshwater used although 9 to 11% came from upstream sources such as production of synthetic fertilizer, ferric chloride, and other chemicals.

For urine collection, producing acetic acid led to higher environmental impacts than the electricity consumed to collect urine. More acetic acid was used to ensure that urine remained stable in the urine concentration alternative. While urine diversion reduced the volume of wastewater that needed to be collected, the impacts of collecting and stabilizing urine were substantially larger than any benefits of collecting less wastewater in sewers.

Urine-derived fertilizer production resulted in about 21-75% as much GWP as synthetic fertilizers and decreased most other environmental impacts. The exception was AP, which ranged anywhere from a 77% decrease to a 231% increase from synthetic fertilizers. Offsetting synthetic fertilizers was almost always required to reduce GWP and CED.

The impacts of concentrating urine were dominated by electricity consumed for reverse osmosis. Unless urine diversion led to major reductions in electricity consumed at wastewater treatment plants, such as in Virginia, concentration increased total electricity within a municipality. The environmental impacts of producing concentrated urine were low in Vermont due to the high proportion of renewable energy. The impacts of producing struvite and ammonium sulfate were relatively consistent, with sulfuric acid being responsible for much of the GWP and leading to this alternative always having the largest AP. Processes such as regenerating activated carbon, cleaning RO membranes, producing magnesium oxide and ion exchange resin, and electricity for pumping in the fertilizer production facility had small overall impacts.

The GWP and CED of shipping urine-derived fertilizers to the fertilizer depot comprised a relatively small portion of the net impact, but were up to 3.5 times higher than shipping synthetic fertilizers. Synthetic fertilizers were shipped much longer distances, but only required about 4-8% as much mass, and were more likely to use larger and more efficient transports.

Urine diversion significantly decreased the impacts (GWP, CED, AP) of nutrient removal from treatment plants with stringent effluent limits, whereas more lenient plants reduced the EP of releasing effluent to aquatic ecosystems. As shown in Figure 4, all treatment plants benefitted by reducing the amount of ferric chloride required to remove phosphorus. Treatment plants with stricter effluent limits had larger reductions of electricity, methanol, and nitrous oxide emissions in biological treatment. These benefits were so large in Virginia that even if no synthetic fertilizer were offset, urine diversion would still reduce net greenhouse gas emissions. In certain cases, urine diversion could eliminate the need for ferric chloride and methanol during average conditions. Reducing total wastewater volume, capturing BOD in concentrated urine, and minor changes to secondary sludge production led to small changes in environmental impacts.

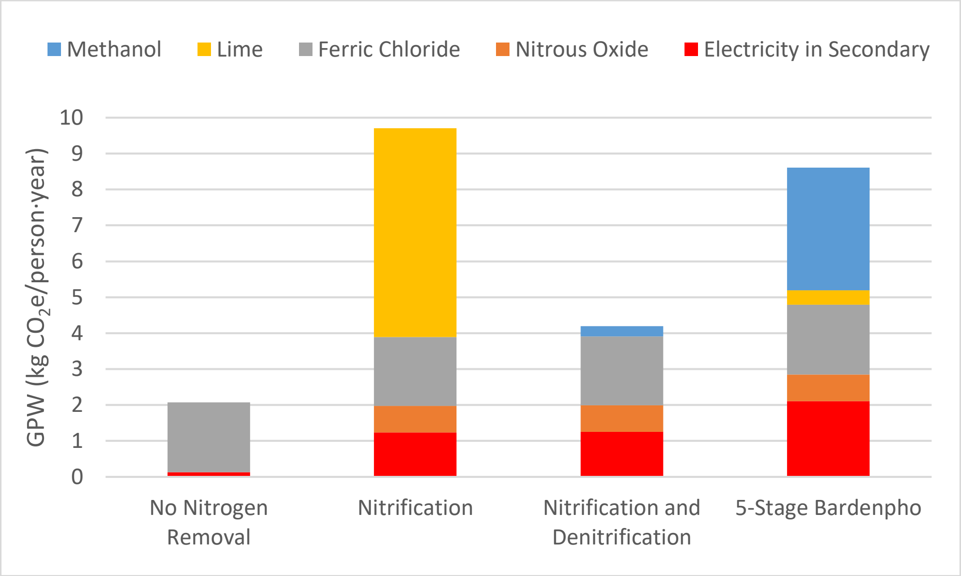

Figure 4. Reductions in GHGs in WWTPs.
Reductions in greenhouse gas emissions from different types of wastewater treatment plants due to 70% urine diversion. All remove phosphorus and use the Virginia electricity grid to allow comparison. The first type has an aeration basin to remove BOD (Vermont). The second category uses nitrification to oxidize ammonia to nitrate. The third category further treats wastewater with denitrification, which converts some nitrate to nitrogen gas. The final category is the 5-Stage Bardenpho treatment method which removes the most nutrients (Virginia).

Figure 23 shows that the methodology used in this study and the simpler methodologies used in other studies to estimate how much urine diversion reduces greenhouse gas emissions from wastewater treatment are within a reasonable range.[48,49] However, the benefits from increasing urine are not linear due to elimination of chemical requirements or changes in wastewater treatment plant configuration, so the use of an linear offset results in some level of inaccuracy.

3.3 Sensitivity Analysis

Figures 24-27 demonstrate that the results of this study were largely robust. Urine diversion always decreased freshwater use and EP. The number of repetitions where urine concentration increased GWP and CED were negligible, but occurred occasionally for struvite and ammonium sulfate when renewable electricity was used. Urine concentration alternatives did increase AP in a few repetitions with the Five-Stage Bardenpho when renewable electricity was used, and approximately 30% of repetitions in the single aeration basin. The AP for struvite and ammonium sulfate was always higher than the conventional alternative even as the efficiency of ammonium sulfate use approached 100%. Figures 28 and 29 show that urine concentration typically had a better environmental performance than struvite and ammonium sulfate. These differences were more pronounced when producing electricity had lower environmental impacts because the added burden of electricity consumption to concentrate urine was lessened. Environmental improvements in GWP, CED, and AP categories are highest in locations with electricity produced from fossil fuels and large levels of nutrient removal, as shown in Figures 24-27. Environmental improvements are also greater in locations with less wastewater volume per person and lower performing aeration systems.

Tables 22 and 23 show that excluding fertilizer offsets and nitrous oxide emissions from effluents from the scope can change the conclusion of the analysis. As the environmental impact of producing electricity decreased, reducing greenhouse gases without considering fertilizer is less likely. The exception is for urine concentration alternatives with limited nutrient removal because net electricity consumption in a municipality increases. When nitrous oxide emissions from effluent were not considered, urine diversion almost never decreased greenhouse gas emissions from single aeration basin systems that use renewable electricity.

4. DISCUSSION

Similar to other life cycle assessments,[48,49,51] this study found urine diversion reduced most environmental impacts. It expanded upon previous research by conducting a more comprehensive characterization of wastewater treatment and by evaluating a range of large-scale systems. Simpler methods to estimate the changes in environmental impacts of treating wastewater are valid as an approximation, but the more complete methods used in this study may be more appropriate when increased accuracy is needed or when different extents of urine diversion are being evaluated. Scenario and sensitivity analyses showed that freshwater use and EP impacts were always reduced, GWP and CED were consistently reduced, and urine concentration usually reduced the AP.

Urine collection is the uncertain aspect of this analysis due to a lack of large-scale examples. This study modeled a centralized system conveying urine from an urban area to a central processing facility in order to create a reasonable estimate of the environmental burdens from urine collection. It suggested the importance of the acetic acid dosage used for stabilization. Other options include a more distributed system consisting of multiple processing facilities strategically located throughout an urban area to reduce both the distance collected urine would need to be transported, as well as the transport time which could reduce urine stabilization requirements. The optimal scale of decentralization of urine collection still needs to be assessed. Significant further development of urine collection is certainly possible which could reduce not only cost but environmental impacts.

The advantages urine diversion provides wastewater treatment are clearly demonstrated in this study and corroborated by previous research.[18,52] Where nutrient removal is practiced, these primarily include elimination of chemical inputs (metal salts for phosphorus removal, supplemental carbon such as methanol for nitrogen removal) and reduced energy use. In many cases urine diversion can eliminate the need to expand existing wastewater treatment plants for nutrient removal capabilities. While not considered in this study, eliminating the need for nutrient removal could allow further changes to treatment process such as increased capture and utilization of organic matter contained in the influent wastewater. In locations where nutrient removal is not a goal for wastewater treatment, eutrophication can be reduced as less nutrients are discharged to local waterways. Urine diversion leads to decreases in environmental impacts through a wide range of conditions, but can be a particularly effective decarbonization strategy in areas with high levels of nutrient removal, electricity produced primarily from fossil fuels, and relatively little wastewater per capita.

Producing fertilizer from urine instead of mineral sources leads to significant environmental benefits. These urine-derived fertilizer production methods were characterized using laboratory and demonstration scale-studies, [25,26,37–39,54,98] but demonstration of other available approaches[15,33,40,41,92] and larger scale systems will provide an improved basis for assessing environmental impacts.[15,33,40,41,92] They were selected to represent a range of fertilizer products and production methods. Urine concentration is more heavily dependent on energy, produces a fertilizer with nitrogen in the form of urea, retains much of the potassium in urine, and has a relatively consistent nitrogen-to-phosphorus ratio (depending on the composition of urine and whether additional nutrients are added). Struvite precipitation and ammonium sulfate largely use chemical inputs and could easily be applied with different nitrogen-to-phosphorus ratios. Throughout all electricity grids, the environmental burdens of producing concentrated urine were usually lower even as the efficiency of sulfuric acid use approached 100%. A comprehensive environmental evaluation of all the different forms of urine derived fertilizers is needed for a complete life-cycle perspective. The environmental burdens of producing them were lower than synthetic fertilizers, and will be significantly improved as use of sulfuric acid for ion exchange and energy for reverse osmosis are optimized, or renewable energy is used for urine concentration.

The urine-derived fertilizers evaluated could be applied similarly to fertilizers commonly used in the US.[99] Beyond the impacts of fertilizer production, other important factors such as the higher popularity of single-nutrient fertilizers will affect which fertilizers are produced.[99] Implementation efforts need to consider the fertilizer demands of adjacent communities and the transportation costs and environmental impacts associated with shipping urine-derived fertilizers from population centers.[12,100]

Urine can replace a significant fraction of synthetic fertilizers. Researchers estimate 16-30 kilograms of nitrogen and 4 kg of phosphorus in fertilizer are currently used per person per year in affluent countries.[101–104] If all nutrients were recovered from domestic wastewater it would likely produce less than 5 kg of nitrogen and 1 kg of phosphorus per person. Regardless, urine diversion can provide significant environmental benefits and can be used with other strategies such as dietary changes, manure application, and reduction of nutrient runoff during mineral extraction and fertilizer application to significantly improve nutrient use efficiency.[101,102]

The development of large-scale urine collection and processing systems is still at a conceptual stage. Research is ongoing to understand and address the many challenges of urine diversion, including economic, market and regulatory acceptance,[12,26,105–108] potential user error,[26,109] risk aversion and lack of confidence in performance,[8,106,107] and lock-in to conventional systems.[107,110] Irrespective of the urine processing method considered, net benefits were observed for each scenario evaluated. In some cases the environmental benefits associated with water and wastewater management alone were sufficient to offset the environmental burden associated with urine collection, processing, and transport. The analyses presented here clearly indicate that the more well-defined benefits (reduced wastewater management requirements and avoided synthetic fertilizer production) exceed the environmental impacts of urine collection, processing, and transport, suggesting that further efforts to develop such systems are warranted.

APPENDIX A: SYSTEM SCOPE AND BOUNDARY

This section will provide a more in-depth description of what is and is not included in the scope of this study. This study only considers the use phase, so burdens from infrastructure and decommissioning are excluded. There are important exclusions from the use phase, so the impacts should not be interpreted as the total impact of the urban water system.

In order to conduct this Life Cycle Assessment (LCA), mass balances were tracked through much of the urban water cycle. The environmental impacts of managing some of these flows were quantified. While this study evaluates certain environmental burdens of processes that remove pharmaceuticals from urine, a mass balance on pharmaceuticals was not conducted. The impacts from releasing pharmaceuticals into aquatic or terrestrial ecosystems were not considered.

Relevant material flows fall into four general categories:

1. Mass, volume, and environmental impacts are considered.
2. Only the mass and volume are considered.
    a. For example, considering other wastewater flows (e.g. stormwater) to estimate wastewater dilution.
3. Mass, volume, and environmental impacts are excluded from consideration due to similarity between alternatives.
    a. For example, primary sludge was not tracked because it is assumed that it will be unaffected by urine diversion.
4. The mass, volume, and environmental impacts were excluded due to a lack of data.

These flows are displayed in Figure 5 and explained below.

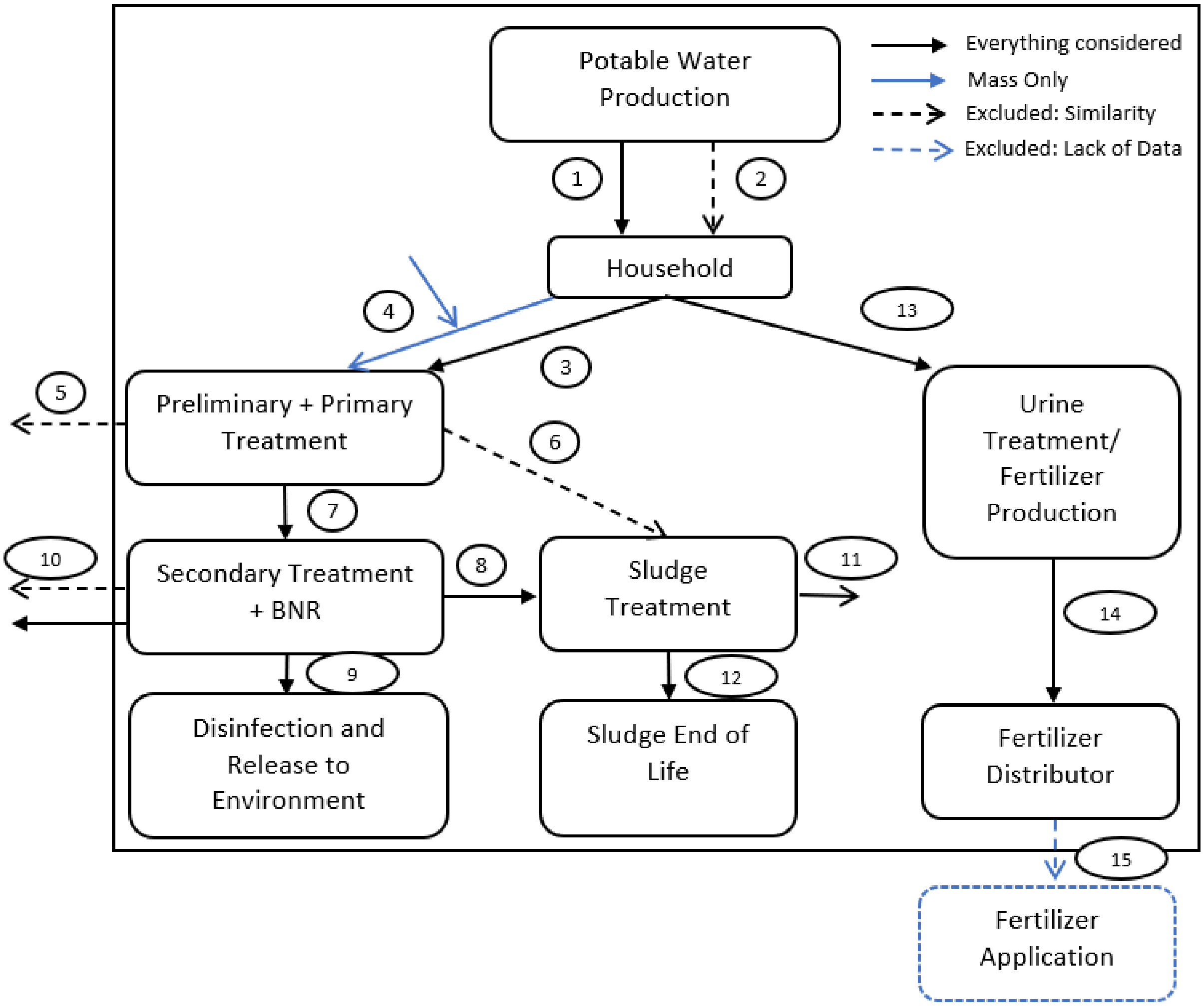

Figure 5. Detailed description of Study Scope.
Depiction of all flows and the extent to which they were accounted for in this study.

1. **Potable water produced for flushing.** The mass, impact of treatment, and impact of delivery to the household were considered.
   a. Impacts of delivering chemicals to the water treatment plant are included. This is also the case for the wastewater treatment plant, the urine-derived fertilizer plant, and the acetic acid used for urine collection.

2. **Potable water produced for other purposes.** This was excluded under the assumption that urine diversion will not affect other uses of water. Activities unrelated to urine diversion such as heating water were also excluded.
3. **Flush water and excrement that are not diverted.** The mass, nutrient composition, and environmental impact of transporting this sewage from households to the sewer were accounted for.
    a. The impact of conveyance only included the energy consumption for sewage lifts. Direct emissions from sewers were not accounted for.
4. **Greywater and rainwater collection:** The impact of collecting this water is not considered in the final analysis. This is because it is assumed that urine diversion will not change the volume of greywater and rainwater collected. However, the masses of pollutants and total volume was considered in order to later determine the wastewater strength.
    a. Flows 3, 4, and reject water from urine derived fertilizer production are treated as if they mix at the wastewater treatment plant. Because these sources of water contribute some pollutants and much of the volume, tracking these flows are essential for accurate wastewater treatment modeling.
5. **Solids from preliminary treatment hauled to a landfill:** The mass and environmental impacts were excluded because it was assumed that urine diversion would not affect the solids removed from grit screens, grit chambers, or other forms of preliminary treatment.
6. **Treatment of sludge produced in primary treatment:** The mass, volume, and environmental impacts were excluded because it was assumed that urine diversion would not affect the mass of primary sludge produced or its composition.

7. **Secondary treatment and nutrient removal:** The total wastewater volume, mass of all major pollutants, and inputs necessary for secondary treatment were included.
8. **Treatment of sludge produced in biological wastewater treatment:** The dry mass of sludge produced, the volume of sludge, and all necessary treatment were included. This was done in order to capture changes in sludge production due to urine diversion.
9. **Disinfection and release of wastewater to the environment:** The amount of water released, the disinfection chemicals and energy, and the eutrophication impact for nutrients were accounted for.
10. **Direct emissions of greenhouse gases during biological treatment:** As wastewater undergoes biological treatment, much of the carbon in wastewater is released as carbon dioxide. Similar to other LCAs, these emissions were excluded under the assumption that they had biogenic origins.[111,112] While some argue that a considerable portion of direct carbon dioxide emissions are not biogenic (e.g. detergents with fossil inputs),[113] urine diversion was assumed to not affect the amount non-biogenic carbon in influent wastewater.
    a. Other gases, notably nitrous oxide, were accounted for.
    b. Urine diversion *could* affect the amount of non-biogenic carbon if it reduces the carbon inputs added during wastewater treatment for denitrification (e.g. methanol). The production of any carbon added is considered, but not the amount converted to carbon dioxide during wastewater treatment.
11. **Production of biogas from anaerobic digestion:** Electricity from biogas production was estimated and subtracted from electricity used in wastewater treatment.

12. **Sludge end of life:** The impacts of transporting sludge to its end of life and environmental burdens thereafter were considered. It was assumed that the end of life for sludge was either land application or disposal in a landfill.
    - **a.** For sludge transportation, the mass depended on whether it was anaerobically digested, composted, lime stabilized, thermally dried, or incinerated.
    - **b.** Included impacts for land application included direct emissions of gases and pollutants to water that can lead to the greenhouse effect, eutrophication, or acidification. Estimates for offset nitrogen and phosphorus fertilizer are also included.
    - **c.** Considered impacts for landfills were fugitive greenhouse gas emissions from uncaptured Landfill Gas.
13. **Collection and treatment of separated urine, flushwater, and acetic acid:** The total mass of nutrients, volume, and impacts from collecting urine and converting it to fertilizer are included.
    - **a.** Infrastructure is excluded, but some equipment is included. This includes the reverse osmosis membranes and the activated carbon columns, including the fiberglass casing.
    - **b.** The impact of packaging urine derived fertilizer was not quantified. This is consistent with the data used for synthetic fertilizers, which also do not include the impact of packaging.
    - **c.** The volume of reject water sent to the wastewater treatment plant, as well as treatment burdens, are considered.

14. **Transportation of fertilizers to a fertilizer distributor:** The mass of fertilizer delivered and the impact of transportation to a fertilizer distributor are accounted for.
15. **Transportation of fertilizers to the farm and application:** The transportation of fertilizer from the distributor to the farm, the energy needed for application, and emissions of the fertilizer after application are not considered.
    - **a.** It is worth noting that the vehicles that are transporting fertilizer are likely less efficient than the vehicles that bring fertilizer to the fertilizer distributor. This could be important as liquid urine-derived fertilizers require more mass and volume than synthetic fertilizers to deliver the same quantity of nutrients.
    - **b.** It is not entirely clear how, if at all, emissions from applying urine-derived fertilizers would differ from applying urine-derived fertilizers. Due to some research on the topic it was assumed they do not differ significantly.[25]
        - **i.** Some researchers suggest that due to the slow releasing nature of fertilizers such as struvite that runoff emissions may be lower.[35,38] This has not yet been quantified well enough to include in this study.
        - **ii.** The extent of nutrient runoff and nitrous oxide emissions likely differ depending on many soil and climate factors, so if these impacts had been quantified the data quality of these emissions would be low at best.

APPENDIX B: WASTEWATER TREATMENT MODELING

B1 Influent, Preliminary and Primary Treatment

The water and excrement from non-diverted flushes, effluent from urine derived fertilizer production, grey water, and rain water were assumed to be mixed before wastewater treatment began. The volume and major constituents of wastewater per capita for the conventional and urine diversion alternatives are shown in Table 4. All scenarios assumed 256 liters of grey and rainwater per person per day based off of current estimates for total wastewater volume[114] and the current average volume of water for flushing.[85] The nutrient content of greywater, urine, and feces per capita were obtained from the literature.[13,114–118] Wastewater per capita was not varied between scenarios because of a lack of data on indoor water use and sewer leaking and infiltration by state.

Table 4. ***Primary Influent per Capita.***

	Struvite and Ammonium Sulfate	Urine Concentration	Conventional
Volume (L/day)	266	266	283
Total Nitrogen (g/day)	5.60	5.72	13
Total Phosphorus (g/day)	1.51	1.46	2.10
COD (g/day)	180	171	180

Electricity consumption for primary and preliminary treatment is listed in Table 9. Table 5 lists the percent of each contaminant removed during primary clarification. The impacts of sending screenings to a landfill, any direct emissions, and primary sludge pumping and production are excluded.

Table 5. Primary Removal Efficiencies.
The following percent of each constituent is assumed to be removed at the primary clarifier. TSS is short for Total Suspended Solids. VSS is short for Volatile Suspended Solids. TKN is short for Total Kjeldahl Nitrogen.

Constituent	**Percent Removal**
Ammonia	0
Total Phosphorus	15%
Organic Phosphate	15%
COD	35%
BOD5	45%
TSS	60%
VSS	60%

B2 Secondary Treatment and Nutrient Removal

Secondary treatment and nutrient removal were modeled to elucidate the environmental ramifications of urine diversion on wastewater treatment. This facilitates determination of whether or not urine diversion can change important operating conditions, such as the solids retention time (SRT), treatment plant configuration, and oxygen demand.[52,119,120] The environmental impacts quantified for this stage of treatment include the eutrophication potential of the effluent, direct emissions of greenhouse gases, electricity consumption, sludge production, and chemical consumption.

B2.1 Conversion Factors Used in Biological Wastewater Treatment Modeling

The constituents tracked in the mass balance needed to be converted using the characterization factors in Table 6 to allow modeling of biological treatment.[121,122]

Table 6. Wastewater Conversion Factors.

Fraction	Value
Readily biodegradable: Total COD	0.15
Non-colloidal slowly biodegradable: Total COD	0.85
Soluble Biodegradable Organic Nitrogen: TKN	0.149
Particulate Biodegradable Organic Nitrogen: Total N	0.195
Particulate Inert Organic Matter: VSS	0.56

This study modeled growth, decay, hydrolysis, ammonification and yield for ordinary heterotrophic organisms (OHOs). Decay and yield were modeled for autotrophic organisms. The parameters listed in Table 7 are used to model microbial metabolic activity for these organisms. The Arrhenius rule is used to adjust these parameters for different temperatures.

Table 7. Parameters used to model metabolic activity.

Parameters	Description	Unit	Value under 20 °C
$i_{O/XB,T}$	Conversion factor from TSS unit to COD units	gCOD/gTSS	1.2
$b_{H,AER}$	Aerobic Decay of OHOs	1/d	0.62
$b_{H, ANX}$	Anoxic/anaerobic Decay of OHOs	1/d	0.3
k_a	Ammonification Rate of OHOs	L/(mgN d)	0.04
f_D	Fraction of active biomass contributing to debris for OHOs	-	0.2
$Y_{H,T}$	Yield of OHOs	mg TSS/mg COD	0.5

b_A	Autotrophic decay	1/d	0.17
$Y_{A,T}$	Yield of autotrophic organisms	mg TSS/mg COD	0.15

This study modeled growth, decay, hydrolysis, ammonification and yield for ordinary heterotrophic organisms (OHOs). Decay and yield were modeled for autotrophic organisms. The parameters listed in Table 8 are used to model microbial metabolic activity for these organisms. The Arrhenius rule is used to adjust these parameters for different temperatures.

Table 8. Parameters used to model metabolic activity.

Parameters	**Description**	**Unit**	**Value under 20 ℃**
$i_{O/XB,T}$	Conversion factor from TSS unit to COD units	gCOD/gTSS	1.2
$b_{H,AER}$	Aerobic Decay of OHOs	1/d	0.62
$b_{H,\ ANX}$	Anoxic/anaerobic Decay of OHOs	1/d	0.3
k_a	Ammonification Rate of OHOs	L/(mgN d)	0.04
f_D	Fraction of active biomass contributing to debris for OHOs	-	0.2
$Y_{H,T}$	Yield of OHOs	mg TSS/mg COD	0.5
b_A	Autotrophic decay	1/d	0.17
$Y_{A,T}$	Yield of autotrophic organisms	mg TSS/mg COD	0.15

B2.2 Effluent Quality

B2.2.1 COD, Total Phosphorus, Ammonia-Nitrogen

The rule-based model used for this study continued treatment until the concentration of all regulated contaminants were at or below regulatory levels. This determined important operating characteristics such as the solids retention time or the treatment plant configuration. The effluent COD concentration is assumed to be equal to the effluent standard.

In cases where the influent concentration of phosphorus is less than the effluent standard, the effluent concentration is determined by the phosphorus in the influent and the phosphorus removed by biomass production. When the secondary influent is in excess of the effluent standard, the effluent concentration is assumed to be equal to the effluent standard. Phosphorus is removed by either biological methods, chemical methods, or both.

Systems with anaerobic zones (e.g. the Five-Stage Bardenpho treatment plant in the Virginia scenario) use Phosphorus-Accumulating Organisms (PAOs) to remove phosphorus. It is assumed that 10.7 grams of Volatile Fatty Acids as COD are required to remove one gram of phosphorus.[122] If there is not enough volatile fatty acids to remove the required phosphorus biologically or if the treatment plant does not use biological phosphorus removal, the remaining phosphorus requiring removal is precipitated with ferric chloride.

When the concentration of ammonia-nitrogen in the influent is lower than the effluent standard, the effluent concentration is assumed to be equal to the influent concentration. When the concentration of ammonia-nitrogen is higher than the effluent standard, nitrification is induced and the concentration is assumed to be equal to the effluent standard.

B2.2.2 Nitrate and Total Nitrogen

Nitrate formed during nitrification

The treatment plant modeled for the Virginia scenario and some of the treatment plants modeled for the Michigan scenario use nitrification and denitrification to reduce the concentration of total nitrogen in the effluent. First, the concentration of nitrate formed during nitrification is calculated using equation 1.[122]

$$S_{NO} = 0.98(S_{N,a} - S_{NH} - S_{NS}) \qquad \text{(equation 1)}$$

Where: S_{NO}=Nitrate Formed by nitrification
$S_{N,a}$=Nitrogen available to nitrifiers
S_{NH}=Effluent Ammonia Nitrogen concentration, set as the effluent standard
S_{NS}=Effluent soluble organic N concentration

The concentration of nitrogen available to nitrifiers is calculated using equation 2.[122]

$$S_{N,a} = S_{NH0} + S_{NS0} + X_{NS0} - NR(S_{S0} + X_{S0} - S_S) \qquad \text{(equation 2)}$$

Where: S_{NH0}= influent soluble ammonia N concentration
S_{NS0}= soluble biodegradable N concentration
X_{NS0}= particulate biodegradable organic N concentration
NR= Nitrogen required for heterotroph growth
S_{S0}= influent readily biodegradable COD concentration
X_{S0}= influent slowly biodegradable COD concentration
S_S= effluent COD concentration, which is determined by standard

Equation 3 is used to determine the nitrogen required for heterotroph growth.[122]

$$NR = 0.087\frac{(1+f_D b_H SRT_{IAZ+PAZ})Y_{H,T} i_{O/XB,T}}{(1+b_H SRT_{IAZ+PAZ})} \qquad \text{(equation 3)}$$

Where: $SRT_{IAZ+PAZ}$=Solids Retention time of the initial anoxic zone and the primary aerobic zone, days

Denitrification

The Modified Ludzack-Ettinger (MLE) treatment plants that were a part of the Michigan scenario had denitrification in one zone. The Five-Stage Bardenpho plant modeled for the Virginia scenario had denitrification in two zones. Some of the processes to model the initial anoxic zone for the Five-Stage Bardenpho plant and the only anoxic zone for the MLE are described below. Then, the processes to model the second denitrification zone for the Five-Stage Bardenpho plant will be described.

Denitrification in the Initial Anoxic Zone

Both readily and slowly biodegradable substrate are assumed to be available for denitrification due to a long enough Solids Retention Time (>3 days). External carbon sources such as methanol are added when there is not enough biodegradable substrate for nitrate removal.

Denitrification associated with both slowly biodegradable substrate and denitrification associated with readily biodegradable substrate are included. Equation 4 is used to estimate the mass rate of denitrification associated with utilization of slowly biodegradable substrate. The concentration of the Mixed Liquor Suspended Solids (MLSS) is assumed to be 3,500 grams of TSS per cubic meter.

$$\Delta N_{XS} = q_{NO/XS} MLSS_{IAZ} \qquad \text{(equation 4)}$$

Where: NXS= mass rate of denitrification, grams nitrate-N per day

$q_{NO/XS}$=Specific Nitrate-N Utilization Rate, grams nitrate-N per gram MLSS per day

$MLSS_{IAZ}$=Concentration of MLSS in the Initial Anoxic Zone

The specific nitrate-N utilization rate associated with slowly biodegradable COD is calculated using equation 5.

$$q_{NO/XS} = 0.018U_{ANX} + 0.029$$

(equation 5)

Where: U_{ANX}= the loading factor for slowly biodegradable substrate to the anoxic zone

The loading factor for slowly biodegradable substrate to the anoxic zone (*UANX*) is calculated using Equation 6.

$$U_{ANX} = F \cdot S_{S0} \cdot MLSS_{IAZ} \qquad \text{(equation 6)}$$

Where: $MLSS_{IAZ}$=Mixed Liquor Suspended Solids in the initial anoxic zone

The denitrification rate associated with readily biodegradable substrate is calculated using equation 7. Then, the effluent nitrate concentration from the primary aerobic zone (CN1) is calculated using equation 8.

$$\Delta N_{SS} = F \cdot S_{S0} \frac{1-Y_{H,TiO} i_{O/XB,T}}{2.86} \qquad \text{(equation 7)}$$

$$CN1 = S_{N0} - \Delta N_{XS} - \Delta N_{SS} \qquad \text{(equation 8)}$$

Denitrification in the Secondary Anoxic Zone

For the secondary anoxic zone, the Burdick empirical relationship (eq. 9) is used to estimate the specific denitrification rate.

$$q_{NO/XB} = 0.12SRT_{TOT}^{-0.706} \qquad \text{(equation 9)}$$

This is then used in equation 10 to estimate the denitrification mass rate.

$$\Delta N_{XB} = q_{NO/XB} MLSS_{SAZ} \quad \text{(equation 10)}$$

Where: $MLSS_{SAZ}$ = Concentration of Mixed Liquor Suspended Solids in the secondary anoxic zone

The effluent loads of these species and TRACI impact factors are used to determine the eutrophication potential of wastewater. [123]

B2.3 Direct Greenhouse Gas Emissions

Wastewater treatment can lead to considerable emissions of carbon dioxide and nitrous oxide. This study did not estimate direct emissions of carbon dioxide under the assumption that they are predominately biogenic, and that urine diversion will not affect the amount of carbon dioxide from fossil origin. In reality, this is a conservative approach because urine diversion may decrease direct carbon dioxide emissions because it can decrease the input of fossil carbon inputs, such as methanol.

Nitrous oxide emissions have been observed in treatment plants that utilize nitrification and denitrification. Nitrous oxide emitted during wastewater treatment and after wastewater is released in the environment can be found in Table 10. It is worth noting that nitrous oxide emissions are very uncertain, and different methodologies have been used in other studes.[121,124–126]

B2.4 Electricity Consumption

B2.4.1 Oxygen Demand

Aeration is one of the largest environmental impacts of wastewater treatment, and be significantly affected by the concentration of nutrients in wastewater. As shown in equation 11, the total oxygen demand depends on the oxygen demand of heterotrophic and autotrophic organisms.

$$RO_{tot} = RO_H + RO_A \tag{equation 11}$$

Equation 12 is used to calculate the oxygen demand from heterotrophic organisms. This is determined by the amount of COD removed and metabolic characteristics of these organisms.

$$RO_H = F(S_{S0} + X_{S0} - S_S)\left[1 - \frac{(1+f_D b_H SRT_{TOT})Y_{H,T} i_{O/XB,T}}{1+b_H SRT_{TOT}}\right] \tag{equation 12}$$

Where: F=m^3 wastewater treated per day

S_{S0}=Influent readily biodegradable COD

X_{S0}=Influent slowly biodegradable COD

S_S= Effluent COD

Treatment plants with ammonia effluent standards also require considerable quantities of oxygen to support autotrophic organisms. In those instances, equation 13 is used to quantify the oxygen demand of these organisms.

$$RO_A = F(S_{N,a} - S_{NH})\left[4.57 - \frac{(1+f_D b_{ATOT} SRT_{TOT})Y_{A,T} i_{O/XB,T}}{1+b_A SRT_{TOT}}\right] \tag{equation 13}$$

B2.4.2 Other Electricity Demand

Electricity is also consumed for mixing and pumping wastewater. For both anaerobic and anoxic zones, it is assumed that 5 watts per cubic meter of reactor volume is needed. It is assumed that influent wastewater, Return Activated Sludge (RAS), Mixed Liquor Recirculation (MLR), and Waste Activated Sludge (WAS) are pumped at 70% efficiency with head losses described in Table 9.

Table 9. Pumping head in secondary treatment.

Pumping Process	**Head (meter)**
Influent wastewater	1
RAS	7
MLR	1
WAS	7

B2.5 Sludge Production

Urine diversion has the potential to prevent the need for more complex wastewater systems, which can change important operating parameters such as the solids retention time. This can ultimately change the total quantity of sludge produced. Equation 14 is used to estimate the dry mass of biosolids produced.

$$WAS_{TSS} = \frac{W(1+\alpha)}{(W+\alpha)MLSS} \qquad \text{(equation 14)}$$

Where: WAS_{TSS}= grams TSS per day of biosolids

W=Wastage Ratio

α=Solids Recycle Ratio, assumed to be 0.5

$MLSS$=Concentration of Mixed Liquor Suspended Solids, grams TSS per cubic meter

The wastage ratio is calculated using equation 15.

$$W = \frac{\alpha \cdot V}{(1+\alpha)SRT_{Tot} \cdot F - V} \qquad \text{(equation 15)}$$

The processes used to treat sludge can be found in Table 11 and section 3.5.

B2.6 Chemical Consumption

This study included chemical consumption for phosphorus removal, disinfection, alkalinity adjustment, and external carbon addition. Ferric chloride was used to precipitate phosphorus. The amount of phosphorus requiring chemical precipitation is described in section 2.2.2.1. Sodium hypochlorite was assumed to be the disinfectant used in all wastewater treatment plants. The dosage used for both can be found in Table 10. Lime was used for alkalinity adjustment, and methanol was used as the external source of carbon. The dosage for each chemical depends on the biological processes occurring during treatment and are described below.

B2.6.1 Alkalinity Adjustment with Lime

The assumed initial alkalinity in all systems was 200 milligrams as calcium carbonate per liter. If enough alkalinity is consumed by biological processes such as nitrification that the alkalinity would drop below 50 milligrams as calcium carbonate per liter, lime is added. Each gram of lime provides 1.35 grams of alkalinity as Calcium Carbonate.[127]

Nitrification of ammonia to nitrate consumes alkalinity.[122] Equation 16 is used to calculated alkalinity destroyed in treatment plants with nitrification but no denitrification.

$$Alk_{des} = 7.23 \cdot S_{N0} \qquad \text{(equation 16)}$$

Where: Alk_{des}= Alkalinity destroyed, in grams per cubic meter

S_{N0}=Concentration of Nitrate-N formed, in grams per cubic meter

Converting nitrate to nitrogen gas through denitrification produces alkalinity. Equation 17 is used for treatment plants with nitrification and denitrification.

$$Alk_{des} = 7.23 \cdot S_{N0} - 3.5 \cdot (S_{N0} - S_N) \qquad \text{(equation 17)}$$

Where: S_N=Concentration of effluent Nitrate-N in grams per cubic meter, set by effluent standard

B2.6.2 External Carbon Provided from Methanol

Many treatment plants add an external carbon source to ensure a high enough carbon to nitrogen ratio during denitrification. As shown in equation 18, this is done by finding the difference between the ideal readily biodegradable substrate and the actual readily biodegradable substrate. Additional substrate is provided in the form of methanol. The assumed COD of methanol is assumed to be 1.2 kg/L.

$$Required\ Additional\ Substrate\ (mg/L) = \Delta S_{Ti} - F \cdot S_{S0} \qquad \text{(equation 18)}$$

Where: ΔS_{Ti}=Ideal mass rate of readily biodegradable substrate for denitrification, grams per day

The ideal concentration of readily biodegradable substrate is found using the required denitrification mass rate and yield, as shown in equation 19.

$$\Delta S_{Ti} = \frac{(\Delta N_{ST} + \Delta N_{XB})}{(1 - Y_{H,T} i_{O/XB,T})/2.86} \qquad \text{(equation 19)}$$

Where: ΔN_{ST}=Required Mass Denitrification Rate from readily biodegradable COD in initial anoxic zone

ΔN_{XB}=Denitrification mass rate in the secondary anoxic zone

The required mass rate of denitrification associated with utilization of readily biodegradable COD in the initial anoxic zone is determined using equation 20.

$$\Delta N_{ST} = F \cdot S_{NO} - \Delta N_{XS} - \Delta N_{XB} - WAS_{TN} - (F - F_W) S_{TN,eff} \qquad \text{(equation 20)}$$

Where: NXS=Denitrification mass rate from slowly biodegradable substrate in the initial anoxic zone

WAS_{TN}=Wastage nitrogen mass rate, grams of nitrogen per day

F_W=Flow rate of waste sludge, cubic meters per day

$S_{TN,eff}$=Concentration of nitrogen in effluent, grams of nitrogen per day

APPENDIX C: LIFE CYCLE ASSESSMENT DATA

A more complete list of parameters used for the Life Cycle Assessment are provided in Table 10, followed by discussion on certain calculations.

Table 10. List of inputs used to model the life cycle assessment.

Used For	Parameter	Value	Unit	Source	Notes
Household Use	Flushes per person per day	5.14	/pe-day	[85–88]	
	Percent flushes only urination	74.1	%	[87,89,90]	
	Conventional Low flow Toilet Water per Flush	4.84	L/flush	Kohler	
	Urine Diversion Toilet Water per flush	0.165	L/flush	[18];Raye-Leonard, personal conversation.	
	Mean Volume of Urine per day	1.62	L/pe-day	[86,89,128]	

	Mean defecation volume per day	0.233	L/pe-day	[13,129]	
	5% Acetic Acid added (Urine Concentration)	0.0417	L/L urine and flushwater	[25]	Likely more than needed, from water with a high pH.[130]
	5% acetic acid added (Struvite and Ammonium Sulfate)	0.0330	L/L urine and flushwater	Calculated	
	Water conveyance losses	13	%	[9,66,131]	
	Electricity to convey water	0.202	kWh/m^3	[16,44,137,64,66,80,132–136]	
Surface Water Treatment	Electricity	0.443	kWh/m^3	[9,57,59–62,65,66,132,138]	
	Alum	0.0491	kg/m^3		
	Ferric Chloride	0.00425	kg/m^3		
	Polymer	$1.19 \cdot 10^{-4}$	kg/m^3		
	Lime	0.00493	kg/m^3		

	Limestone	0.0203	kg/m^3		
	NaOH	0.0166	kg/m^3		
	HSF	0.0011	kg/m^3		
	Ammonia	$2.10 \cdot 10^{-4}$	kg/m^3		
	Phosphoric acid	0.028	kg/m^3		
	CO_2	0.0182	kg/m^3		
	Chlorine gas	0.00148	kg/m^3		
	Sodium hypochlorite	0.0129	kg/m^3		
	Calcium hydroxide	0.0138	kg/m^3		
	$KMnO_4$	$1.96 \cdot 10^{-4}$	kg/m^3		
Groundwater Treatment	Electricity	0.786	kWh/m^3	[48,58–60,63–65]	
	Alum	$1.15 \cdot 10^{-5}$	kg/m^3		
	Lime	0.0484	kg/m^3		

	Limestone	$2.87 \cdot 10^{-6}$	kg/m^3		
	NaOH	0.00997	kg/m^3		
	HSF	$7.00 \cdot 10^{-4}$	kg/m^3		
	CO_2	0.0067	kg/m^3		
	Chlorine gas	0.0063	kg/m^3		
	Sodium hypochlorite	$9.72 \cdot 10^{-4}$	kg/m^3		
	Na_2CO_3	$1.44 \cdot 10^{-5}$	kg/m^3		
	Polyphosphates	$1.40 \cdot 10^{-4}$	kg/m^3		
	Electricity to collect sewage	0.0941	kWh/m^3	[47,134,139–142]	

Wastewater Collection energy	Urine Collection Electricity	0.202	kWh/m^3		Assumed to be same as pressurized water system
Struvite and Ammonium Sulfate Production	Mg:P ratio	1.5:1		[48,54,91,92]	
	Head loss	4.6	meters	Assumed	
	Electricity, Struvite dewatering	0.783	kWh/kg struvite	Bott, personal conversation	
	Dowex Mac adsorption density	4.9	mole/kg resin	[39,49]	
	Sulfuric Acid (18%) per kg N	16.7	L/kg N	Tarpeh, personal conversation	
	Nitrogen concentration in liquid Ammonium Sulfate	59	g/L	[49]	
	Resin replacement frequency	5	years	[49]	

	N recovered from ammonium sulfate	96	%	Tarpeh, personal conversation	
	P recovered in struvite	96	%	[48,54,93,94]	
Urine Concentration	Activated Carbon, pharmaceutical removed	11	g pharmaceuticals/100 g resin	Norit	Based off of methylene blue adsorption
	Pharmaceuticals removed from urine	10	mg pharmaceuticals/L urine		Very uncertain. Most (but not all[143]) studies only focus on a subset of pharmaceuticals.
	Urine concentration electricity consumption	0.009	kWh/L permeate removed	Noe-Hays, personal conversation	
	Uconc N retention	95	%	[95,96]	
	Uconc P Retention	99	%	[95,96]	
	Uconc COD Retetention	99	%	[95,96]	
Wastewater Treatment	Primary + Preliminary Treatment Electricity	0.0301	kWh/m^3	[140,141,144,145]	

	Aeration Electricity	5	kg air/kWh		Diffused air systems
	Ferric Chloride dosage	10.5	g $FeCl_3$/g P removed		Applied when needed to achieve effluent standards
	N_2O emissions during treatment	0.10	% of influent TN	[146]	Applied when treatment plants have intentional nitrification or denitrification. Fraction estimated using values from this study and total nitrogen per person.
	Secondary Clarifier Electricity	0.0035	kWh/m^3	[114]	
	Electricity used for disinfection	$5.50 \cdot 10^{-4}$	kWh/m^3		

	Sodium Hypochlorite used for disinfection	0.006	kg/m^3		
	N_2O emissions after wastewater discharge	$5.0 \cdot 10^{-3}$	kg N_2O/kg N	147	
Sludge Treatment: Anaerobic Digestion	Electricity	163	kWh/dry tonne	148–152	
	Polymer	7.43	kg/dry tonne		
	Finished Solids content	25	%		
	Fugitive Methane	1.84	kg CO_2e/dry tonne	152	
	VS Destruction	40	%	122	Selected from range
	Biogas produced from VS	0.7	m^3 biogas/kg VS destroyed	122	
	Fraction electrons to end products	0.86		122	All other electrons go to biomass. Value selected from range.

	Fraction remaining electrons to methane	0.85		[122]	Other electrons to VFA, alcohols, etc. Selected from range.
	Energy per cubic meter	6.39	kWh/m^3 biogas	[149]	Energy of biogas, not all is captured
	Energy Captured as Electricity	40	%	[114]	Based on internal combustion systems
Sludge Treatment: Lime Stabilization	Electricity	87.9	kWh/dry tonne	[151,152]	
	Polymer	9.5	kg/dry tonne	[151,152]	
	Lime	200	kg/dry tonne		
	Solids Content	25	%		
Sludge Treatment: Incineration	Electricity	188	kWh/dry tonne	[151,153]	
	Natural gas	3490	MJ/dry tonne		
	Polymer	5.26	kg/dry tonne		
	Ash mass/Initial sludge dry mass	0.23			

Sludge Treatment: Thermal Drying	Electricity	242	kWh/dry tonne	[150,152]	
	Natural Gas	11400	MJ/dry tonne		
	Polymer	7.83	kg/dry tonne		
	Solids content	90	%		
Sludge Treatment: Composting	Electricity	97.8	kWh/dry tonne	[150–152]	
	Diesel Consumption in facility	2.83	L/dry tonne		
	Polymer	7.33	kg/dry tonne		
Sludge End of Life: Transportation	Distance sludge shipped to land application	20	km		Class 8 truck, empty backhaul
	Distance sludge shipped to landfill	52	km		Class 8 truck, empty backhaul
Sludge End of Life: Applying Compost	Diesel Consumption	2.68	L/dry tonne	[152]	
	N_2O emissions from application	0.316	kg N2O/dry tonne	[152]	
	Carbon Sequestered	250	kg CO_2e/dry tonne	[152]	

	N Fertilizer Offset	160	kg CO_2e/dry tonne	[152]	
	P Fertilizer Offset	30	kg CO_2e/dry tonne	[152]	
	Diesel Consumption	2.68	L/dry tonne	[152]	
	P emitted to water	0.2	kg/tDM	[154]	
	Methane Emission (storage)	18	kg CO_2e/dry tonne	[152]	
Sludge End of Life: Land Application	N_2O emissions from application	12.8	kg CO_2e/dry tonne	[152]	
	Ammonia emitted to air	1.14	kg/tDM	[154]	
	P emitted to water	0.2	kg/tDM	[154]	
	Carbon Sequestered	250	kg CO_2e/dry tonne	[152]	
	N Fertilizer Offset	200	kg CO_2e/dry tonne	[152]	
	P Fertilizer Offset	38	kg CO_2e/dry tonne	[152]	

	Methane Emissions	1060	kg CO_2e/dry tonne	[78,150–152,154,155]	Assuming 25% Fugitive Emissions. Median value from these sources.
Sludge End of Life: Landfill	N_2O emissions	292	kg CO_2e/dry tonne	[152]	
	NH_3 emitted to air	3.9	kg/dry tonne	[154]	
	Carbon Sequestered	286	kg CO_2e/dry tonne	[152]	
Synthetic Fertilizer Transportation	Distance on Rail	750	miles	[156]	
	Distance on Barge	400	Miles	[156]	
	Distance to Mixer	50	miles	[156]	

C1 Transporting Materials

Environmental burdens of hauling materials from the location of production to the municipality were included. Fuel consumption for transportation of all chemicals, activated carbon, biosolids, and urine-derived fertilizer was estimated using equation 21. All were assumed to be transported with an empty backhaul. This equation was used to account for the distance that synthetic fertilizers were transported via truck, while USLCI data were used for rail and barge transportation.[82]

$$FC = \left(FRV \cdot M_{gv} + K\right) \cdot D + (FRV \cdot M_{vh} + K) \cdot D \qquad \text{(equation 21)}$$

Where: FC = Fuel Consumption, gallons

FRV = Fuel Reduction Value, 0.238 gallons diesel/100 t-mi

M_{gv} = Gross Vehicle Mass, 40 short tons

M_{vh} = Mass of vehicle and trailer, 17.5 short tons

K = 9.7 gallons/100 mi

D = Distance (in 100 mi)

The distance that urine-derived fertilizers and activated carbon were shipped varied between scenarios, while the distance that chemicals were shipped did not. For each scenario, distances urine-derived fertilizers were shipped were estimated by mapping distances from some wastewater treatment plants in the state to large fertilizer suppliers. Assumed distances that chemicals were shipped were informed by the locations of wastewater treatment plants and locations of chemical production plants. These locations were often found using sources such as the *NSF Certified Drinking Water Treatment Chemicals* database.[157]

C2 Household Activities

C2.1 Environmental Burdens of Potable Water

This study differentiated water production in each scenario by considering the ratio of groundwater and surface water. While this is just one of many factors that may affect the environmental impacts of treating potable water, it is useful as a high-level analysis between states.[59,158] Different LCAs were used to determine the inputs per cubic meter of groundwater and surface water found in Table 10.[48,56,65–67,138,57–64] This required weighting inputs found in different studies, which will be described below. Then, the inputs of surface and groundwater were applied in the relevant ratios to characterize a composite cubic meter of water.[55]

Some studies on potable water treatment only quantified energy, while others quantified energy and chemicals consumed. Because of this, different weightings were needed for energy and chemical consumption.

For the electricity consumption of surface water, half of the weight was assigned to values reported in the EPRI report because it was a broad survey focused on the U.S., as opposed to focusing on individual water treatment plants.[59] The electricity consumption in each of the other surface water treatment LCAs was given a weight of 6.25%.[57,61,62,67,138] Chemical consumption was based off of the four studies that quantified chemical use.[58,60,61,65] Each were evenly weighted.

Fewer LCAs on groundwater treatment were found. For electricity consumption, the EPRI report was once again given half of the total weight,[59] while the remaining sources were each given a weight of 12.5%.[48,58,60,63,64] Only three sources contained usable data on chemical consumption. The moderate softening scenario in Godskesen et al. was given a weighting of 20%, the Ishii & Boyer inventory was given a weighting of 35% because it assesses water treatment in the US, and the inventory in the Renzoni & Germain study was given a weighting of 45% because this contained an average value.

C2.2 Water and Acetic Acid for Flushing

The quantities of water and acetic acid used for flushing are listed in Table 10. Table 11 provides further description of when each type of flush is used. Feces only flushes are estimated to only comprise 2% of total flushes and are lumped together with urine and feces flushes.[87] When a user both urinated and defecated, it was assumed that the urine was separated from the feces using a toilet similar to a NoMix toilet.[159] Because this study assumed urine-diversion is very common, it was assumed that users were used to the concept and used the toilets properly without issues found in early studies on urine diversion.[109,160] The nutrient composition of urine and feces were obtained from the literature.[114,115,117,118]

Table 11. Description of flush volumes.

Urine Diversion Alternatives			Conventional Alternative
70% of population diverting urine		30% of population not diverting urine	Entire Population
Urine only	Urine and Feces	All Flushes	All Flushes
UD Flush	Low-flow flush + UD Flush	Low-flow flush	Low-flow flush

Acetic acid is needed in both urine diversion alternatives to delay the process of hydrolysis, which can cause precipitation from urine and can clog pipes.[15,28] The struvite and ammonium sulfate scenario uses less acetic acid because a higher pH is needed to precipitate struvite. It was assumed each flush would provide enough acetic acid to neutralize the alkalinity of urine, which is described in equation 22. The pH of fresh urine is assumed to be 6.2,[27–29] and the concentration of ammonia and phosphate were determined from the nutrient composition of urine. It is assumed that nutrients in potable water were negligible compared to urine.

$$\text{Alkalinity of urine stream} = -[H^+] + [OH^-] + [NH_3] + 2[PO_4^{3-}] \qquad \text{(Equation 22)}$$

The amount of acetic acid needed for the urine concentration was determined through a pilot scale system at the University of Michigan. It should be noted that this location's potable water has a high pH of 9.3[130], which may result in an overestimation of how much acetic acid is required.

C3 Conveyance of Urine

Urine collections is modeled as if it were collected in a pressurized pipe system. The electricity consumption for potable water was deemed as a reasonable estimate. The pipes would be smaller, which would increase head loss. Unlike water, urine collection pipes will not need to be designed to fight fires, reducing required pressure. In addition, they might follow the pathway of sewage collection systems, which already take advantage of gravity.

C4 Urine Derived Fertilizer Production

C4.1 Struvite and Ammonium Sulfate

Struvite is the first fertilizer produced. After the diverted urine arrives at the production center, magnesium oxide is mixed in to provide magnesium and to raise the pH, which causes most of the phosphorus to precipitate as struvite. The precipitated struvite is then dried through draining, an electric dryer, a bucket elevator, and then a vibratory screen (Bott, personal conversation). The effluent from this process continues to ion exchange.

After struvite is precipitated, ion exchange is used to remove the nitrogen and produce a fertilizer. The effluent urine from struvite production is pumped through vessels containing an ion exchange resin such as Dowex Mac 3 resin.[39,49] The ion exchange cartridge collects the ammonia from the urine.[39] To regenerate a cartridge, 18% sulfuric acid is pumped through the filter, which extracts the ammonia and produces a liquid ammonium sulfate fertilizer.[49] Head loss in the cartridge was estimated using the hydraulic conductivity and common column heights.

It was assumed that each urine-derived fertilizer production facility always had enough ion exchange resin to where nitrogen could be removed for 7 days without exceeding the adsorption capacity. The lifetime of the resin is assumed to be 5 years.[49] The environmental burdens of ion exchange resin were not available, so polystyrene was used because it the backbone of the resin.[161]

C4.2 Urine Concentration

The urine concentration scenario removes pharmaceuticals, concentrates urine and deactivates pathogens. While the impacts of pharmaceuticals are not assessed in this study, the inputs for removing them using activated carbon are. The activated carbon is assumed to be shipped back to a regeneration center as a slurry with a 30% solids concentration. A weighted average for the inputs needed to thermally regenerate activated carbon was determined through the literature, and it was assumed that 10% of activated carbon was lost per regeneration.[69–72] It was assumed that each urine-derived fertilizer plant had enough activated carbon to allow continuous removal of pharmaceuticals for 1.1 years.

Urine is concentrated five times its original concentration using reverse osmosis. It was assumed that the membrane received an acid and alkaline wash every six months. The inputs were obtained from a Hydranautics technical report[73]. The only equipment and maintenance considered were the production of and cleaning of the membrane. The necessary area of membrane was based off of the flow rate and the area per influent volume found in other studies.[162–164] A three year membrane lifetime was assumed.

All urine derived fertilizers were assumed to be transported from the fertilizer production facility to regional fertilizer distributers using class 8 trucks with an empty backhaul. Each trip was assumed to haul 22.5 short tons of fertilizer.

C4.3 Synthetic Fertilizer

Synthetic fertilizers were used for the conventional alternatives and to ensure all alternatives (including urine diversion) provided the same mass of nitrogen and phosphorus as fertilizer. Equal mass of nutrients was deemed functionally equivalent because previous research did not find a significant difference in nutrient uptake between these fertilizer alternatives.[25,75–77] The synthetic fertilizers produced were mono-ammonium phosphate (phosphorous and some nitrogen) and urea (nitrogen). Both are commonly used in the U.S.[99]

The urine concentration alternative retains the most phosphorus, and was used to determine how much phosphorus is needed in each alternative. The struvite and ammonium sulfate alternative retained the most nitrogen, and was used to determine how much urea was needed in the other alternatives.

C5 Sludge Treatment

Primary sludge is excluded from the scope of this study because it is assumed that urine diversion will not affect the amount produced or how it is treated. This study does model treatment of the sludge produced from biological wastewater treatment and its disposal.

All secondary sludge is assumed to be thickened, dewatered, and stabilized. In all scenarios, sludge was treated in a combination of anaerobic digestion, composting, lime stabilization, incineration, and thermal drying representative of the state. It is assumed that anaerobic digestion, composting, and lime stabilization produce sludge with 25% solids. It is assumed that thermally drying results in sludge with 90% solids. All sources described treating mixed sludge from primary and secondary treatment. Electricity from biogas production in anaerobic digestion is estimated using equation 23.

$$E_{Electricity} = VS \cdot R_{D,VS} \cdot (1 - Y_{e,b}) \cdot (1 - Y_{e,v}) \cdot P_b \cdot E_{biogas} \cdot e_{electricity} \qquad \text{(equation 23)}$$

Where: $E_{Electricity}$ = Electricity created from biogas, kWh

VS = kg Volatile Solid produced from wastewater treatment and sent to anaerobic digestion

$R_{D,VS}$ = Ratio of Volatile Solids destroyed

$Y_{e,b}$ = Ratio of elections used for biomass production

$Y_{e,v}$ = Ratio of remaining elections used to produce VFAs and other alcohols

P_b = Production of biogas per kg VS destroyed

E_b = Energy per m^3 of biogas

$e_{electricity}$ = Efficiency of converting biogas to electricity

The treated sludge or ash is then shipped to the end of life. The energy use, credits such as fertilizer displacement and carbon sequestration, and emissions to air and water are listed in Table 10. It should be noted that estimates for fugitive methane emissions from landfills varied multiple orders of magnitude.[150–152,154,155]

C6 Data Sources for Unit Processes

Table 12 lists which Life Cycle Databases were used for specific unit processes. GREET was preferably used for electricity and transportation, USLCI was preferably used for other unit processes, and ecoinvent was used in most other cases. Data were not found for some chemicals such as citric acid, so an input-output approach was used in those cases. Data on freshwater use was obtained from Ecoinvent.

Table 12. List of data sources used to quantify environmental impacts.

Input	**Source**	**Item Name**	**Unit**	**Notes**
Electricity	EPA eGRID and GREET		kWh	Grid from eGRID, impact factors from GREET
Natural Gas	USLCI	Natural gas, combusted in industrial boiler/US	m^3	
Urea	ecoinvent	Urea, as N, at Regional Storehouse	kg N	
Monoammonium Phosphate	ecoinvent	Monoammonium phosphate, as P2O5, at regional storehouse, RER S	kg P_2O_5	
Acetic acid (98%)	USLCI	Acetic Acid, at plant/kg/RNA	kg	
Ferric Chloride	ecoinvent	Iron (III) chloride, 40% in H2O, at plant/CH U	kg	
Alum	ecoinvent	Aluminum sulphate, powder, at plant/RER U	kg	
Methanol	USLCI	Methanol, at plant/RNA	kg	
Lime	ecoinvent	Lime, hydraulic, at plant/CH U	kg	
Sodium Hypochlorite	ecoinvent	Sodium hypochlorite, 15% in H2O, at plant/RER U	kg	
Sodium hexametaphosphate	ecoinvent	Sodium phosphate, at plant/RER U	kg	
Sodium Silico Fluoride	ecoinvent	Fluosilicic acid, 22% in H2O, at plant/US U	kg	
CO_2	ecoinvent	Carbon dioxide liquid, at plant/RER U	kg	
Ammonia	USLCI	Ammonia, steam reforming, liquid, at plant/RNA	kg	
Oxygen	USLCI	Oxygen, liquid, at plant	kg	
Sodium Hydroxide	USLCI	Sodium hydroxide, production mix, at plant/RNA	kg	
Magnesium Oxide	ecoinvent	Magnesium oxide, at plant/RER U	kg	

Sulfuric Acid	USLCI	1 kg Sulfuric acid, at plant/kg/RNA	kg	
Low-Sulfur Diesel	GREET		gallon	Used equation 2-3 to calculate diesel consumption in a class 8 truck.
Barge Transport	USLCI	Transport, barge, diesel powered/US	tkm	
Rail Transport	USLCI	Transport, train, diesel powered/US	tkm	
Chlorine Gas	USLCI	Chlorine, gaseous, prodcution mix, at plant/RNA	kg	
Phosphoric Acid	ecoinvent	Phosphoric acid, industrial grade, 85% in H2O, at plant	kg	
Sodium Carbonate	ecoinvent	sodium carbonate from ammonium chloride production, at plant/kg/GLO	kg	
Potassium Permanganate	ecoinvent	Potassium permanganate, at plant/RER U	kg	
Resin	ecoinvent	Polystyrene, general purpose, at pant, CTR/kg/RNA	kg	
Fiberglass	ecoinvent	Glass fibre, at plant/RER U	kg	
Activated Carbon	Agrifootprint	1 kg Activated carbon, at plant/RER Economic	kg	
Citric Acid	USA Input Output Database	Other basic organic chemical manufacturing	US $	
STPP (kg) (sodium tripolyphosphate	ecoinvent	1 kg Sodium tripolyphosphate, at plant/RER U	kg	
Na-DDBS (kg)	USA Input Output Database	Soap and other detergents	US $	
Membrane	ecoinvent	Polyvinylidenchloride, granulate, at plant RER U	kg	PVDF not available[165]

C7 Life Cycle Impact Assessment

The Life Cycle Impact Assessment was conducted using global warming potential (GWP), cumulative energy demand (CED), freshwater use (Water), eutrophication potential (EP), and acidification potential (AP) as indicators. Freshwater use is technically an inventory approach that is calculated by summing the water utilization and consumption for each unit process.[84] Whenever possible, this was measured on Ecoinvent, even if other impacts for a unit process were determined using another database.[83] This was selected over impact assessment methods such as water deprivation for two main reasons. The first reason is that it would be difficult to determine the origin of all supplies when evaluating multiple states. This would make selection of a water stress index value highly uncertain. The second reason is to allow this framework to be easily applied in other locations. Including an impact method that requires geographic specificity inhibits this research goal.

APPENDIX D: SCENARIOS MODELED

D1 Scenario Description

The environmental merits and drawbacks of urine diversion will heavily depend on many location specific factors. To evaluate the environmental performance of urine diversion, three diverse locations were modeled as distinct scenarios. In each of these scenarios, all three alternatives were evaluated.

The Vermont scenario highlights the performance of urine diversion in a region with electricity produced from predominately renewable sources, and wastewater treatment plants that remove phosphorus but do not remove nitrogen. This scenario based its wastewater treatment off of a community serving approximately 25,000 residents.

The Michigan scenario considers urine diversion through a broader range of municipalities and has an electricity grid comprised predominately of fossil fuels. This scenario produces an “average” treatment plant by placing the state’s treatment plants into four categories and considering the volume of wastewater treated in each. All have strict phosphorus removal standards, while the level of ammonia and nitrogen removal varies between categories.

The Virginia scenario was selected to consider urine diversion in a location with strict nitrogen and phosphorus limits. As many regions look to enact more stringent effluent limits, these plants could benefit the most operationally. This scenario was based off of a treatment plant in the Chesapeake Bay region treating wastewater for 350,000 people.

The scenarios vary factors such as the electricity grid, how water is produced, wastewater effluent limits, sludge treatment and disposal methods, transportation distances, and temperature.

D2 Electricity Production

Data on the ratio of fuels used in the electricity grid and the electricity lost in transmission were obtained from EPA eGRID.[166] The Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) Model was then used to determine the net impact of obtaining these fuels, processing these fuels, and combusting them.[167] The impact per kWh generated is listed in Table 13.

Table 13. Environmental impacts of electricity production.

	Global Warming Potential	**Cumulative Energy Demand**	**Water Consumption**	**Eutrophication Potential**	**Acidification Potential**
Scenario	kg CO^{2-} eq/kWh	MJ/kWh	m^3/kWh	kg N-eq/kWh	kg SO_2-eq/kWh
Vermont	0.107	6.45	$7.68 \cdot 10^{-5}$	$1.04 \cdot 10^{-5}$	$2.96 \cdot 10^{-4}$
Michigan	0.544	8.08	$1.54 \cdot 10^{-4}$	$1.70 \cdot 10^{-5}$	$1.26 \cdot 10^{-3}$
Virginia	0.450	8.14	$1.42 \cdot 10^{-3}$	$1.74 \cdot 10^{-5}$	$8.02 \cdot 10^{-4}$

D3 Water Sources

Table 14 shows the ratio of surface and groundwater used in the public supply for each state.[55] This ratio was multiplied by the inputs for each type of water source for a composite cubic meter of water.

Table 14. Sources of Potable Water by Location.
Percentages used with the potable water inventories to create water inventories in each scenario.

Water Source	**Michigan**	**Vermont**	**Virginia**
Surface Water	81%	63%	88%
Groundwater	19%	37%	12%

D4 Wastewater Treatment

Effluent standards for each scenario were determined through assessing technical reports from each state considered.[168] These effluent standards are shown in Table 15. The Virginia and Vermont scenarios focused on one type of treatment plant, while the Michigan scenario evaluated the treatment plants in the state to make an "average" treatment plant. Relevant real wastewater treatment plants were used to select the numbers of basins, solid retention time, MLSS concentration, and secondary clarifier area.

Table 15. Effluent Wastewater Standard for input in Each Location.
All constituents are listed in milligrams per liter.
** Treatment plants with denitrification had a nitrate limit of 3 milligrams of nitrate-N per liter.*
*** Treatment plants with nitrification had an ammonia limit 8 milligrams of ammonia-N per liter.*

Components	Michigan	Vermont	Virginia
BOD5	25	25	5
TSS	30	20	6
Nitrate-N	-*	-	3
Ammonia-N	-**	-	1
TP	0.7	0.2	0.18

In the Vermont scenario, the Rutland wastewater treatment plant was modeled, which has an aerobic zone for COD removal and adds ferric chloride to remove phosphorus.

The Michigan scenario produced an "average" treatment plant from the approximately 400 in Michigan.[168] The treatment plants were put in four categories. Table 16 lists what percent each category treats, which was based on total volume. Both Large-sized wastewater treatment plants and Small-sized treatment plants do not use nitrification or denitrification to remove nitrogen. Some medium-sized treatment plants use nitrification to reduce ammonia, while others use nitrification and denitrification to reduce total nitrogen. All treatment plants are assumed to remove phosphorus, usually with ferric chloride precipitation. Parameters from the Detroit Wastewater treatment plant were used to model the large wastewater treatment plant. The

Downriver wastewater treatment plant was selected to represent the medium sized treatment centers without denitrification. The Ann Arbor wastewater treatment plant was used to represent the medium-sized treatment plant with denitrification. The small wastewater treatment plants were represented by the Pontiac treatment plant.

Table 16. Description of WWTPs in Michigan.
The four types of wastewater treatment plants that are modeled to make the average Michigan treatment plant. The percentage is determined by the volume of wastewater treated.

Categories	**Percent of Total**	**Volume treated per day (MGD)**	**Secondary treatment configuration**
Large-sized WWTP	46	>100	Single aerobic zone with short SRT and chemical addition for phosphorus removal
Medium-sized WWTPs without denitrification	28.8	10-100	Single aerobic zone with long SRT for nitrification and chemical addition for phosphorus removal
Medium-sized WWTPs with denitrification	4.2	10-100	A2O(Anaerobic+Anoxic+Aerobic) process and chemical addition for phosphorus removal
Small sized WWTPs	21	<10	Single aerobic zone with short SRT and chemical addition for phosphorus removal

The treatment plant modeled for the Virginia scenario represented a plant near the Chesapeake Bay estuary, which is sensitive to eutrophication.[169] The Alexandria wastewater treatment plant is located in the eutrophication sensitive Chesapeake Bay estuary and was used to model this scenario. This treatment plant has a 5-Stage Bardenpho configuration.

D5 Sludge Treatment and Disposal

Table 17 displays the ratio of sludge stabilization methods used in each scenario[170–173]. The methods used to dispose of these treated biosolids are listed in Table 18.

Table 17. Sludge Treatment Methods used in each Scenario.
**Due to a very low percent of aerobic digestion, this was included with anaerobic digestion.*

Treatment Process	**Michigan**	**Vermont**	**Virginia**
Anaerobic Digestion*	11%	23.7%	47.8%
Lime Stabilization	44.5%	42.9%	11.4%
Incineration	11.5%	1.6%	40.1%
Thermal Drying	33%	0%	0.667%
Compost	0%	31.8%	0%

Table 18. End of life for biosolids in each scenario.

Treatment	**Michigan**	**Vermont**	**Virginia**
Land Applied (25% solids)	23.5%	48.2%	28.7%
Land applied (90% solids)	33.0%	0%	0.67%
Landfilled (25% solids)	32.0%	50.2%	30.5%
Incinerated	11.5%	1.60%	40.1%

D6 Transportation Distances

Table 19 lists the distances required to transport urine derived fertilizers and activated carbon for each scenario. Google Maps was used to determine the distance between the wastewater treatment plant and the end location. The wastewater treatment plant was selected because it was assumed that the urine derived fertilizer production plant was located near the wastewater treatment plant. For urine derived fertilizers, the distance to the two closest large-scale fertilizer distributors was quantified and averaged to account for the possibility that a fertilizer distributor would not accept all of the fertilizer produced.

Table 19. Shipping distances in different scenarios. All distances are in kilometers.

Material	Vermont	Michigan	Virginia
Activated Carbon	500	350	425
Fertilizer	28.4	17.7	63.2

APPENDIX E: SENSITIVITY ANALYSIS

Sensitivity analyses were used to assess the robustness of the results, to expand the results beyond three specific cases, and to test the uncertainty due to the parameters and choices selected for each scenario. Figure 6 shows the 12 simulation scenarios evaluated using a Monte Carlo with 10,000 repetitions. The important parameters were determined using a one-at-a-time sensitivity analysis. Some parameters that were correlated, such as TSS removal and BOD removal during primary clarification, were then linked. Data on each parameter were collected in order to determine a probability distribution. In cases where there were few data points or these systems are only in early stages of development (e.g. urine collection systems), a conservative approach was used and either triangular or uniform distributions were used. All distributions are listed in Table 20, and some are discussed afterwards.

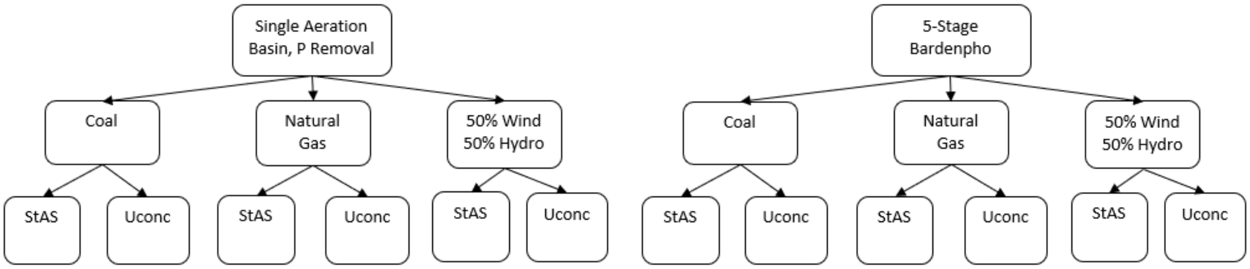

Figure 6. Depiction of simulations ran in sensitivity analysis.
Each of the 12 boxes at the bottom indicate an individual Monte Carlo with 10,000 repetitions.
StAS stands for Struvite and Ammonium Sulfate and Uconc stands for Urine Concentration.

Table 20. List of distributions for parameters in the Monte Carlo analysis.

Section	Parameter	Distribution	Distribution Parameters	Unit	Note
Household & Collection	Ratio urine diverted	Triangular	min=0.5,most likely=0.7,max=0.8		
	Ratio groundwater	Triangular	min=0,most likely=0.37,max=1		
	Average water per flush	Triangular	min=3.5,most likely=4.8,max=9.8	L/L	For low flow and feces flushes
	Acetic acid, urine concentration	Uniform	min=0.015, max=0.025	L/flush	98% acetic acid
	Acetic acid, struvite & ammonium sulfate	Uniform	min=0.025, max=0.041	L/L urine + flushwater	98% acetic acid
	Electricity, urine collection	Uniform	min=0.1, max=1		Based off of water distribution systems
Wastewater Treatment	TSS removed, primary clarification	Normal	med=60,sd=10	%	
	BOD removed with VSS	Normal	med=0.75,sd=0.1	mg/mg VSS	
	Organic N removed with VSS	Normal	med=0.004,sd=0.006	mg N/mg VSS	
	Wastewater per person	Triangular	min=130,most likely=380,max=600	L/p/d	
	Ferric chloride	Normal	med=2, sd=0.5	mole $FeCl_2$/mole P	
	Electricity for aeration	Triangular	min=0.75,most likely=1.2,max=1.5	kg O_2/kWh	
Sludge Treatment and End of Live	Ratio anaerobic digestion	Triangular			Each was set with its prevalence in the US as the most often, and 0 and 1 as triangular.
	Ratio lime stabilization				
	Ratio incineration				

	Ratio drying				These were scaled to equal 1.[174]
	Ratio composting				
	Land applied, 25% solid	Triangular			Ratio of these two types of sludge treatment. Max=ratio of land applied-thermally dried to ratio landfilled. Other end of life determined by treatment method.[174]
	Landfilled				
Struvite & Ammonium Sulfate	Struvite P recovery ratio	Triangular	min=0.9,most likely=0.96,max=0.99		
	Ion exchange N recovery	Uniform	min=0.93,max=0.99		
	Ion exchange stoichiometric ratio	Triangular	min=0.71,most likely=0.91,max=0.98		1 indicates every mole of sulfate binds with ammonium. Performance often increases as systems continue to be developed. Weak Acid Cation Exchange systems often said to operate near 100% efficiency.
Urine Concentration	Uconc N recovery	Uniform	min=0.91,max=0.99		
	Uconc P recovery	Uniform	min=0.98,max=1		
	Uconc COD recovery	Uniform	min=0.95,max=0.99		

	Uconc electricity consumption	Triangular Beta	min=0.002,most likely=0.0035,max=0.036	kWh/L permeate removed	Based off of small scale desalination reverse osmosis systems with Energy Recovery Devices.
Fertilizer Transportation	Ratio synthetic fertilizer sent by truck	Triangular	min=0.015,most likely=0.78,max=0.22,shape=6		Remaining trip 2/3 on rail, 1/3 on barge. From different fertilizer types.[175]
	Distance UD is shipped	Truncated lognormal	median=85,sd=75,min=5,max=500	km	Determined from Google Maps searches,[176] Trimmer and Guest,[100] and US city size statistics. All via truck.

Ion exchange stoichiometric ratio

The amount of sulfuric acid used to regenerate ion exchange resin was one of the most important parameters for the struvite and ammonium sulfate alternative. The stoichiometric ratio is how many ions of sulfuric acid are needed relative to the amount of ammonium ions removed. A weak acid cation exchange resin is used, which are said to often operate near 100% efficiency. The data used in the study are from small-scale experiments, and have a much lower efficiency. The triangular distribution selected assumes that efficiency will improve as the practice becomes more common.

Urine concentration electricity

The electricity demand from reverse osmosis was one of the most important parameters for the urine concentration alternative. The study used a small-scale system for electricity consumption. The Monte Carlo evaluated small-scale desalination systems with energy recovery devices to evaluate how energy intensive reverse osmosis may be at this scale. Desalination was selected as a proxy because much data are available and ocean water has a higher total dissolved solids than urine.

Distance urine is shipped

The probability distribution for distances urine is shipped were estimated by the amount of people in certain sized cities and relevant distances for each. Distances in cities with less than 500,000 were estimated using Google Maps searches. Distances found in Trimmer and Guest[100] were used to estimate distances from larger cities. All transportation was assumed to be via truck, though it is certainly possible that larger cities could use trains.

APPENDIX F: SUPPLEMENTAL RESULTS

More information on the results are shown below. The first section contains graphs comparing the three alternatives in Vermont (Figure 7) and Michigan (Figure 8). The next section has graphs showing each impact measured per component in all scenarios and alternatives (Figures 9-22). Figure 23 then compares different methodologies for estimating the changes in wastewater treatment due to urine diversion.

F1 Comparison of Alternatives

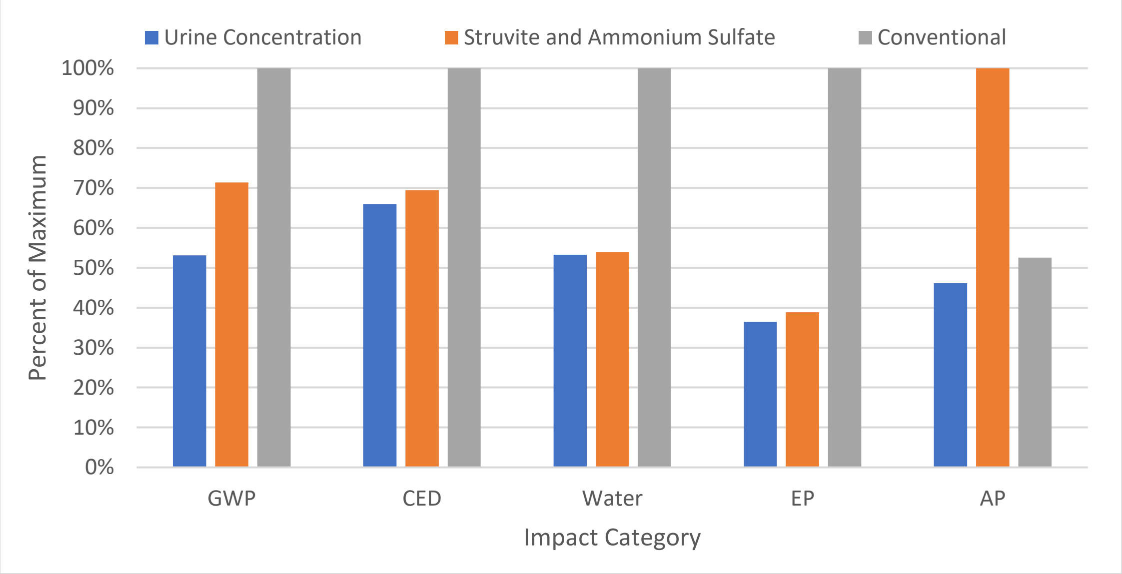

Figure 7. Normalized Impacts in Vermont Scenario.
Total impacts in each alternative normalized to the maximum value.

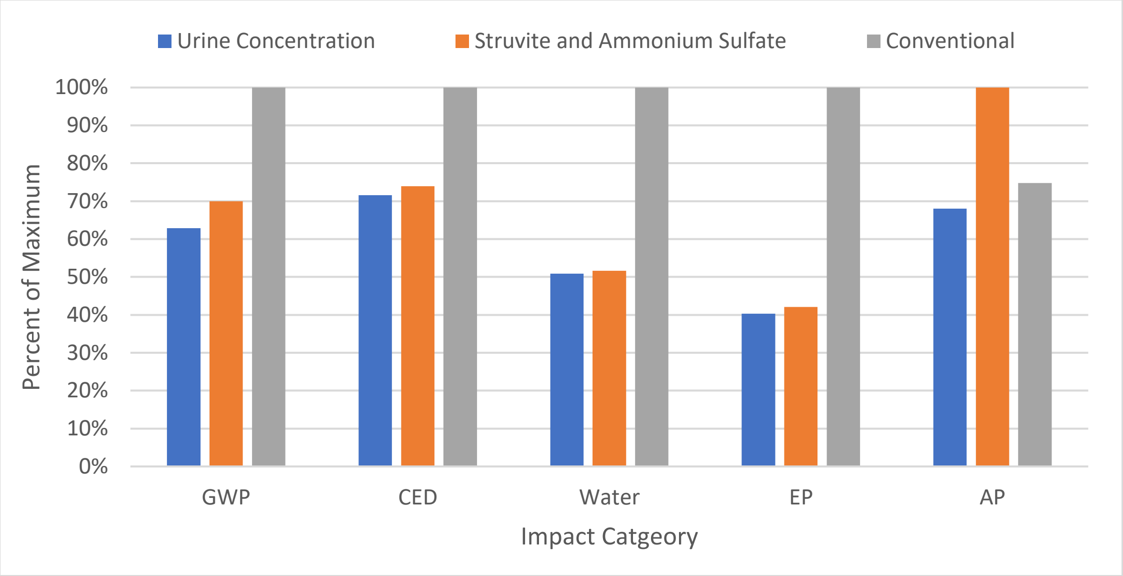

Figure 8. Normalized Impacts in Michigan Scenario.
Total impacts in each alternative normalized to the maximum value.

F2 Impacts per Component

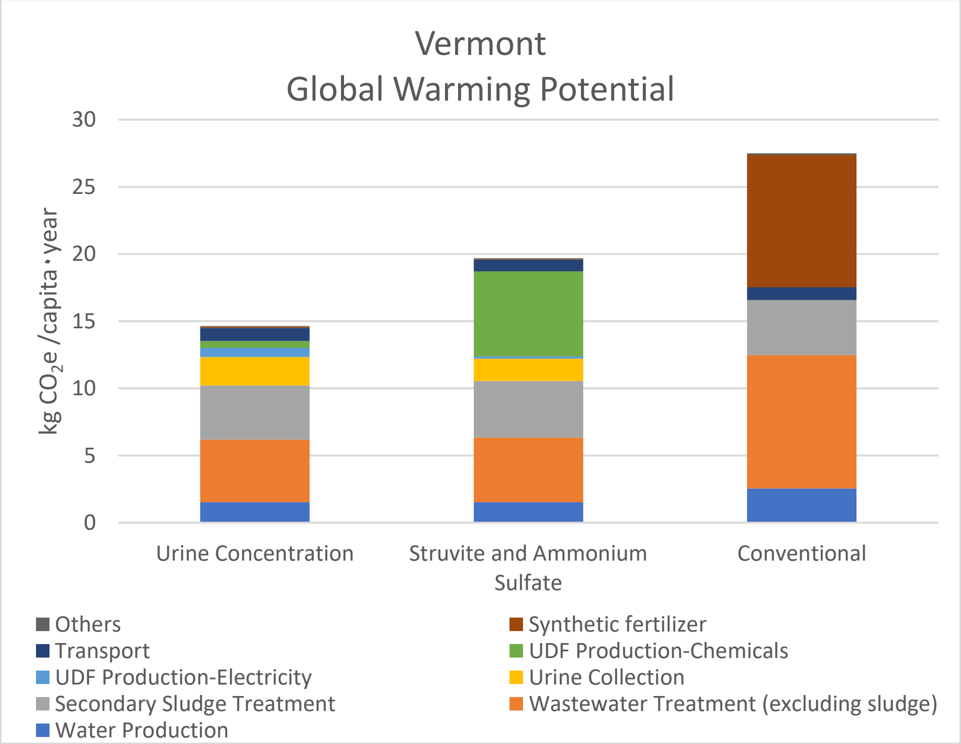

Figure 9. GWP of the Vermont alternatives by Process.

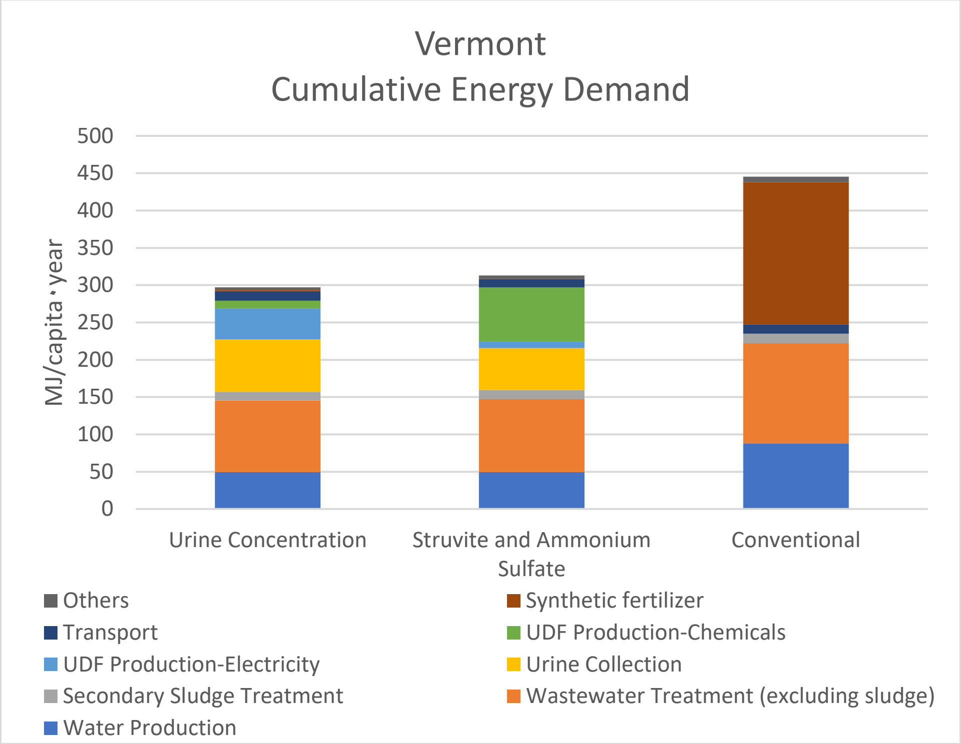

Figure 10. CED of the Vermont alternatives by Process.

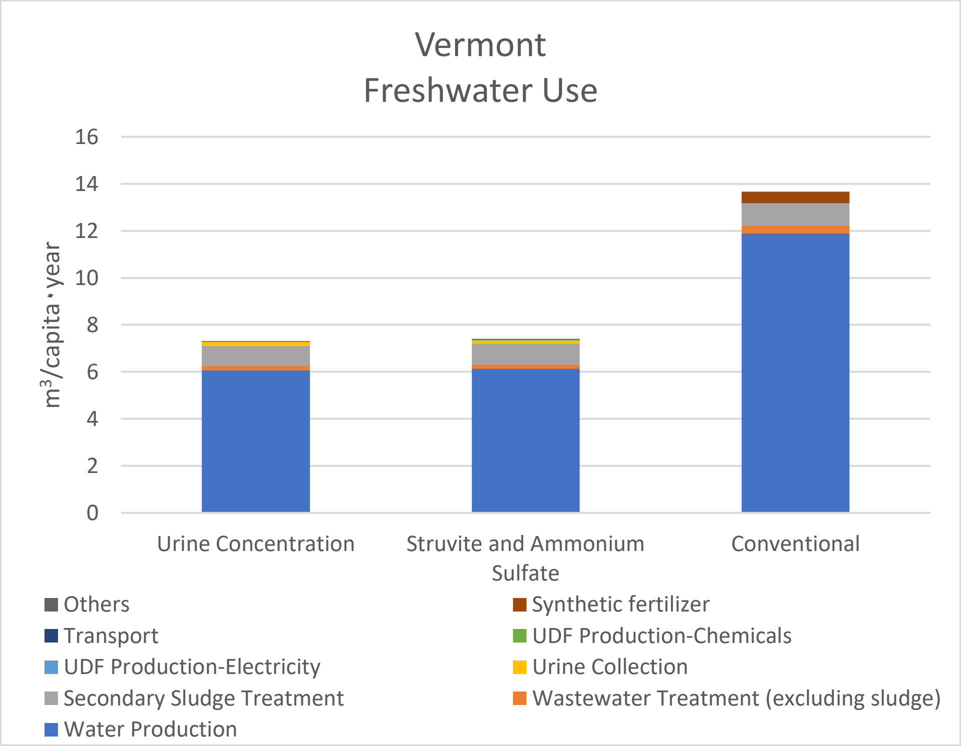

Figure 11. Freshwater use of the Vermont alternatives by Process.

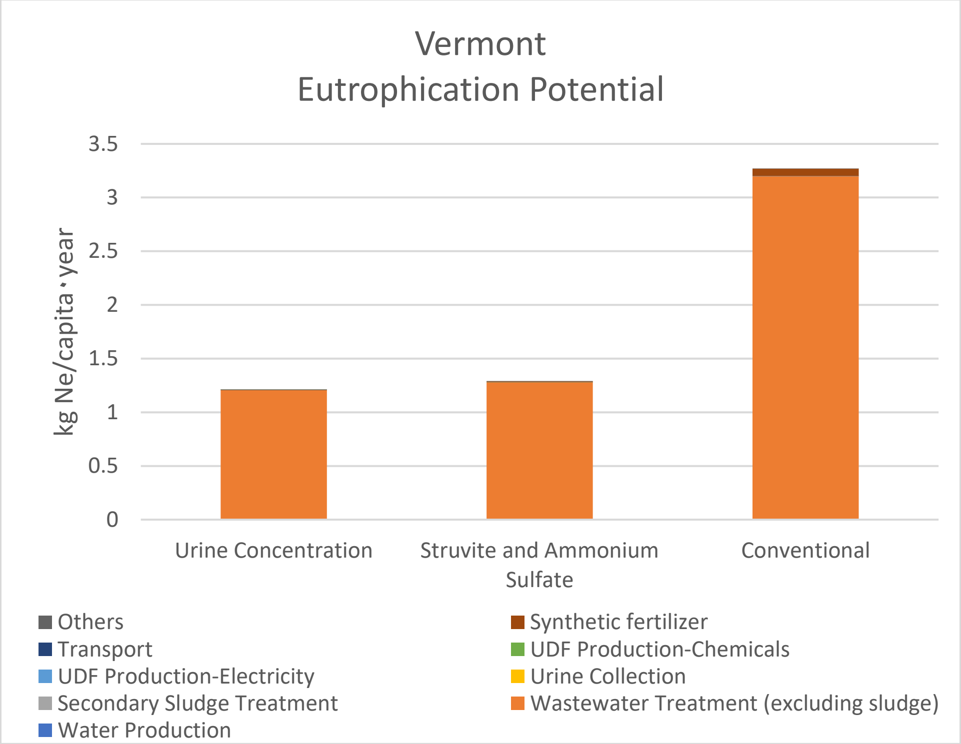

Figure 12. Eutrophication potential of the Vermont alternatives by Process.

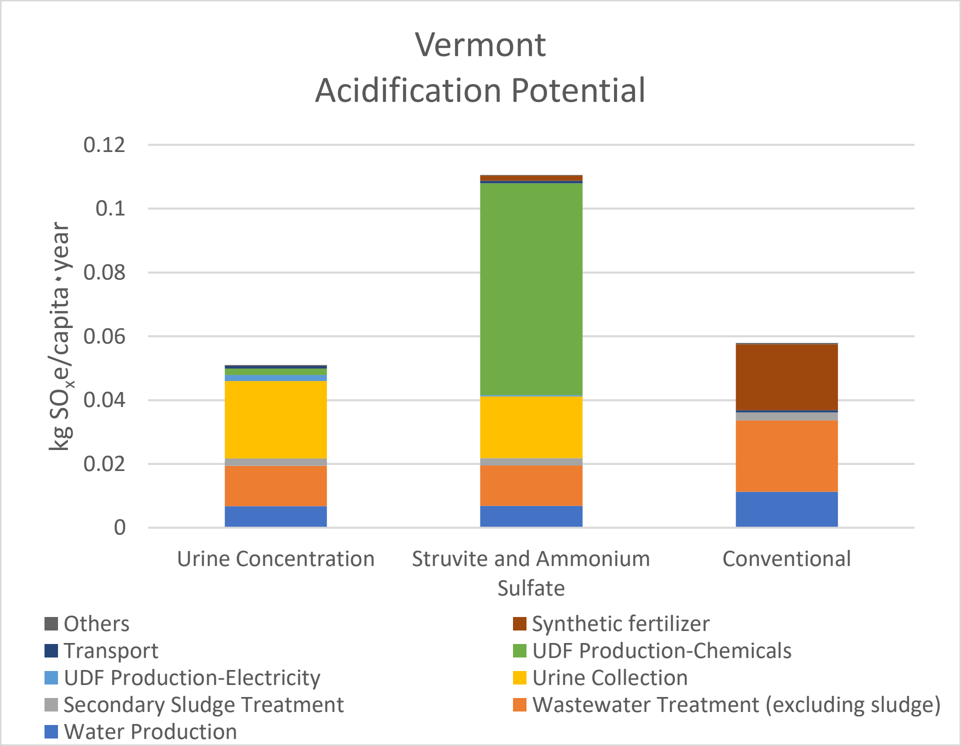

Figure 13. Acidification potential of the Vermont alternatives by Process.

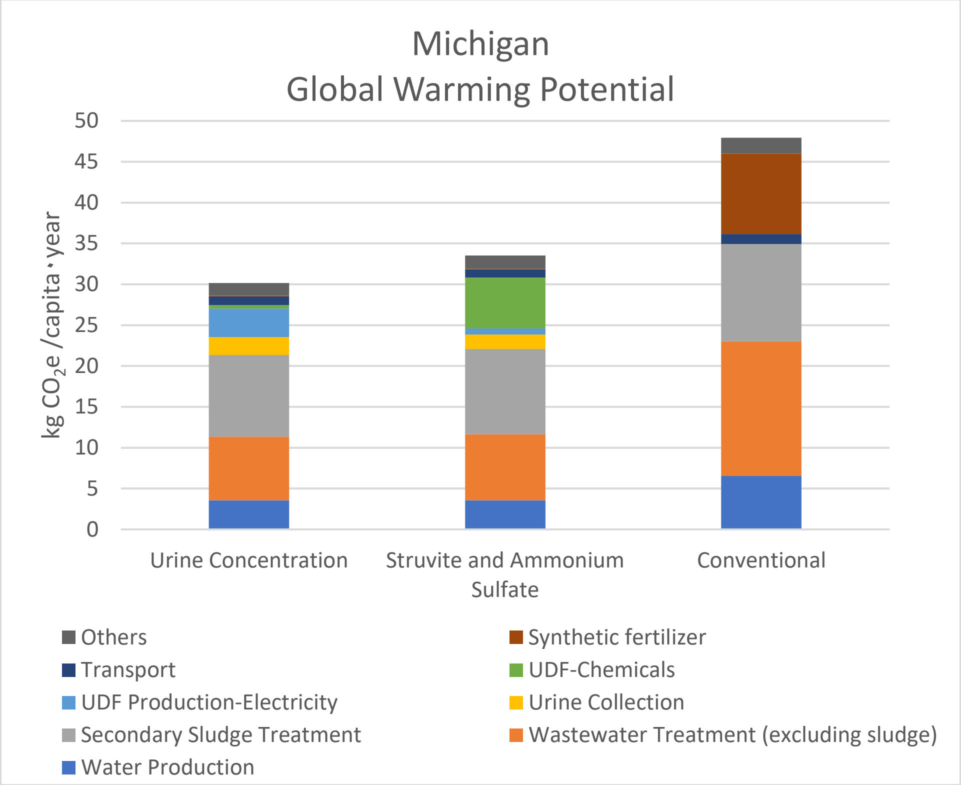

Figure 14. GWP of the Michigan alternatives by Process.

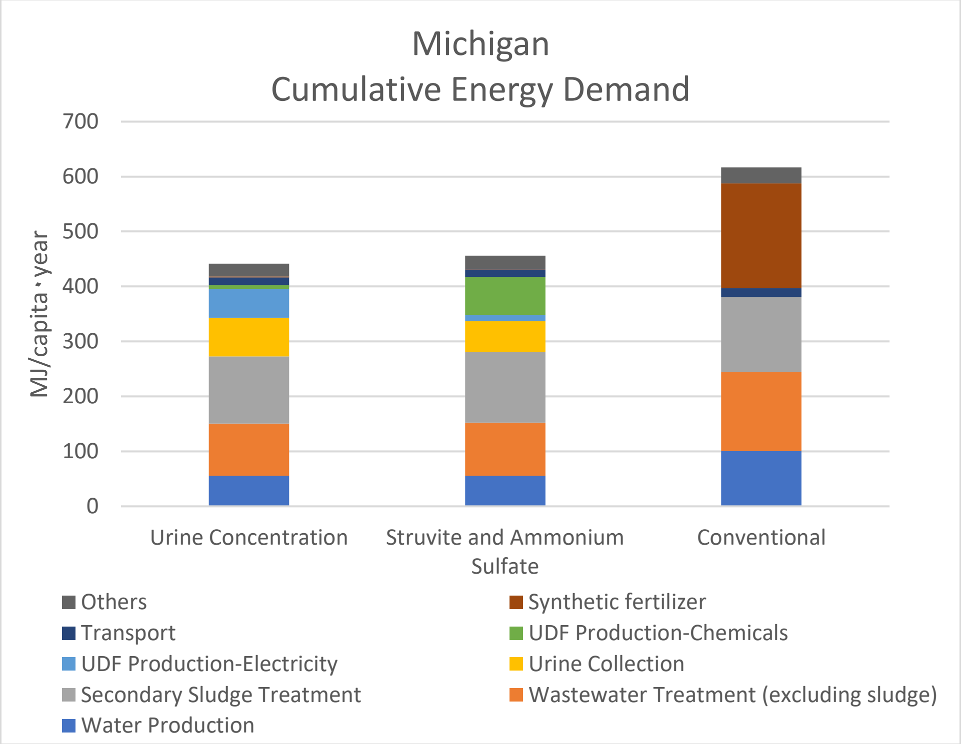

Figure 15. CED of the Michigan alternatives by Process.

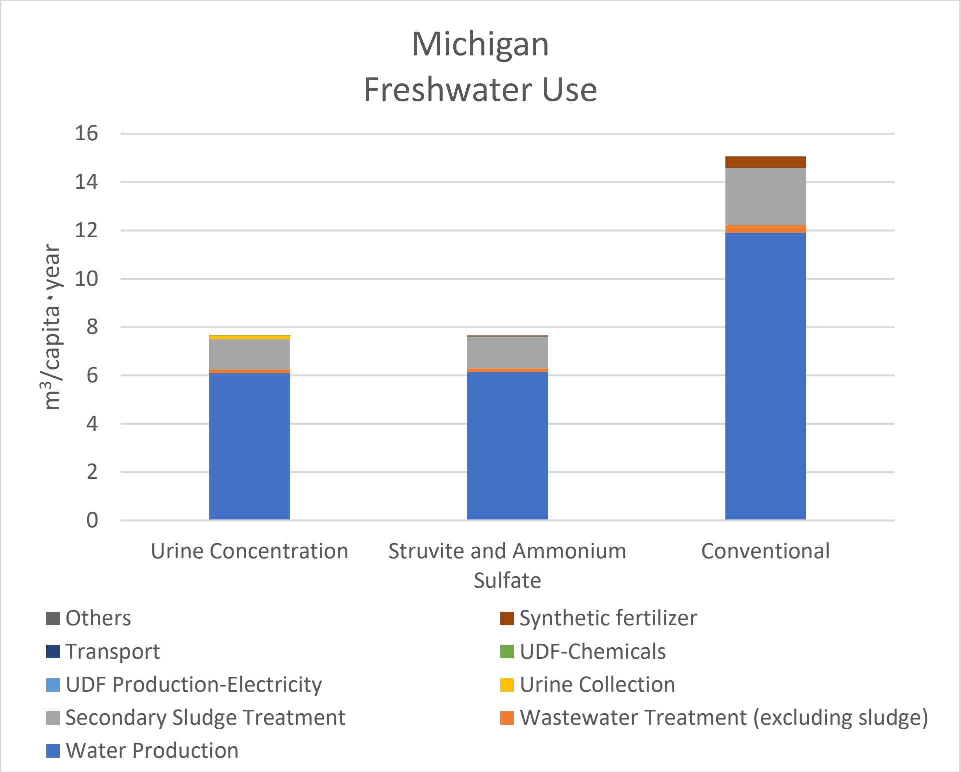

Figure 16. Freshwater use of the Michigan alternatives by Process.

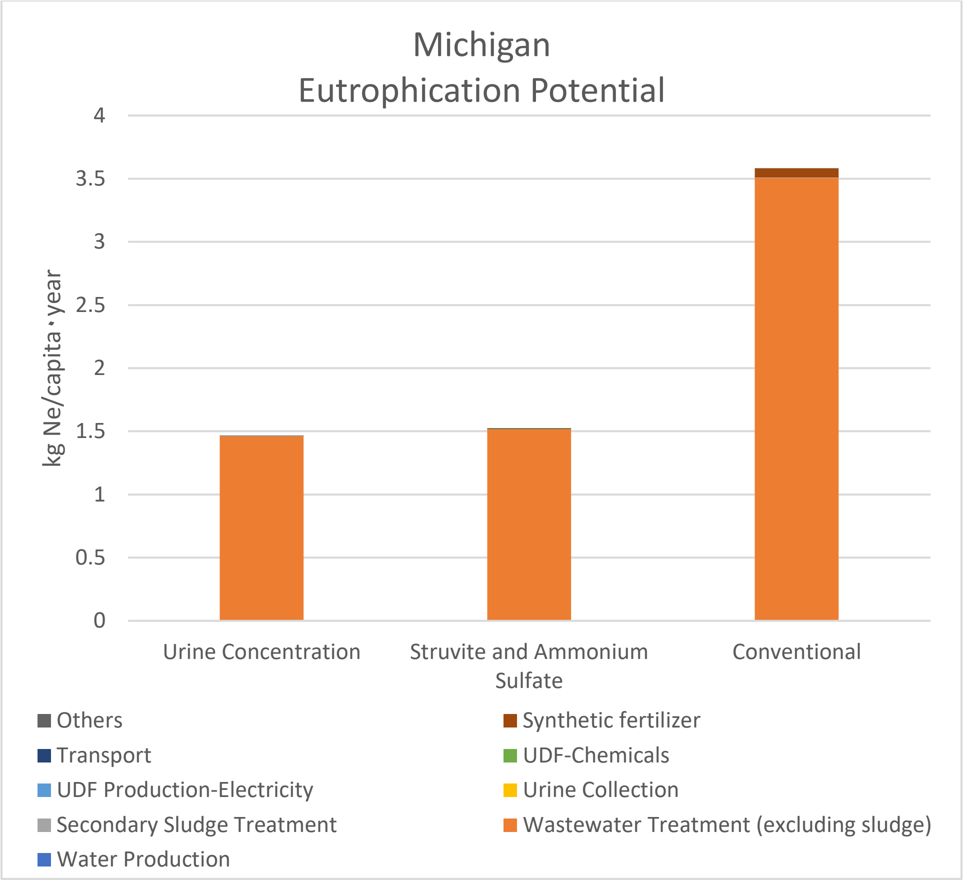

Figure 17. Eutrophication potential of the Michigan alternatives by Process.

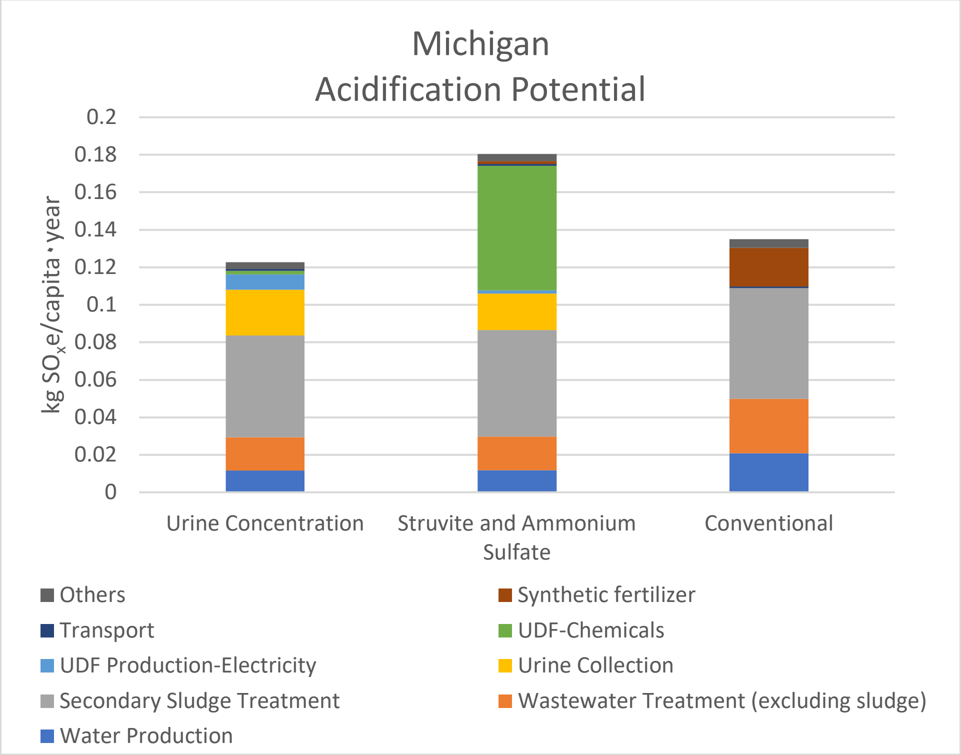

Figure 18. Acidification potential of the Virginia alternatives by Process.

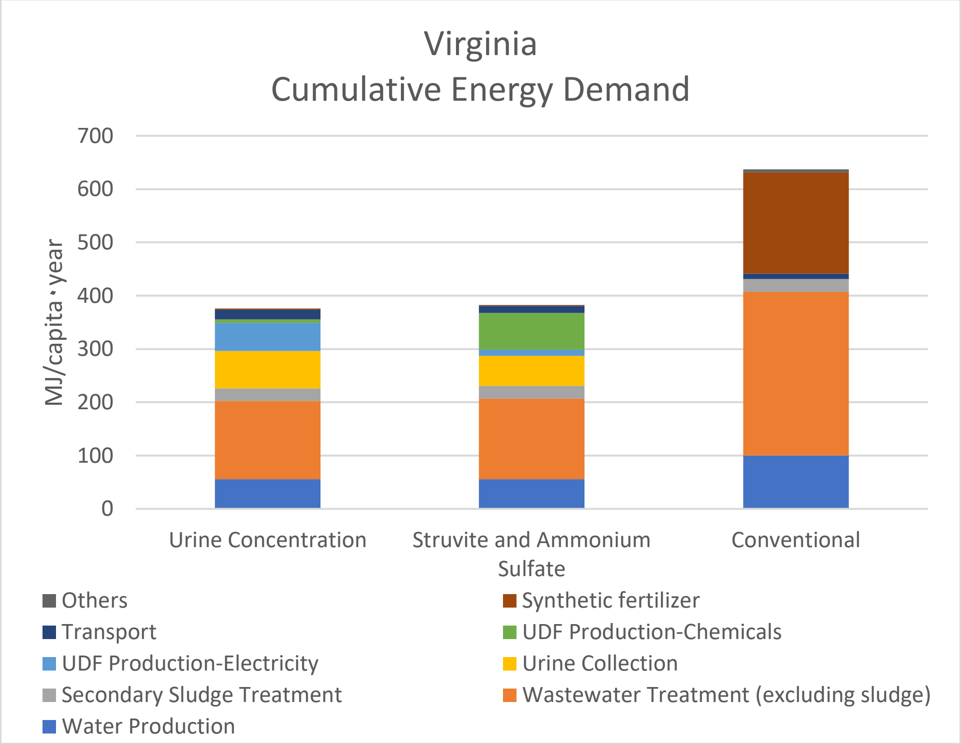

Figure 19. CED of the Virginia alternatives by Process.

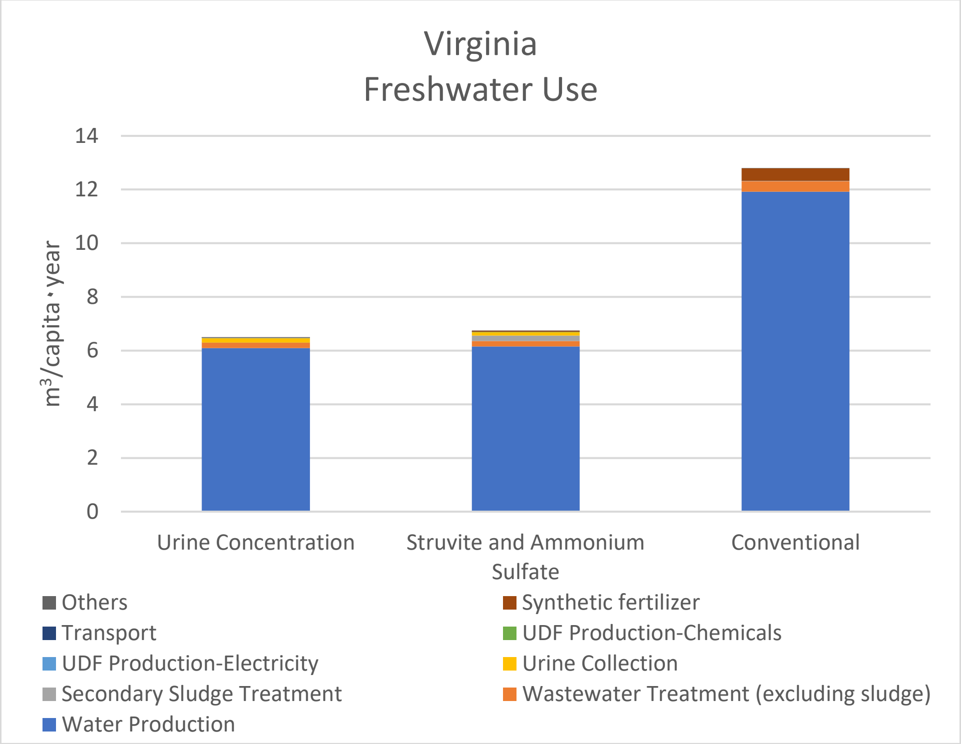

Figure 20. Freshwater use of the Virginia alternatives by Process.

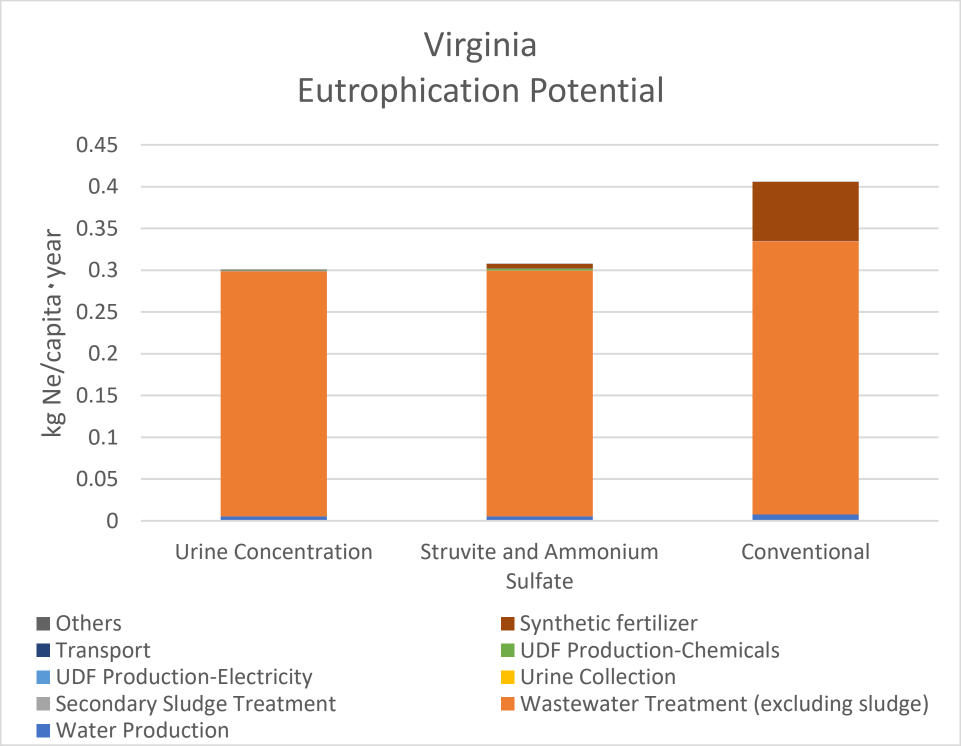

Figure 21. Eutrophication potential of the Virginia alternatives by Process.

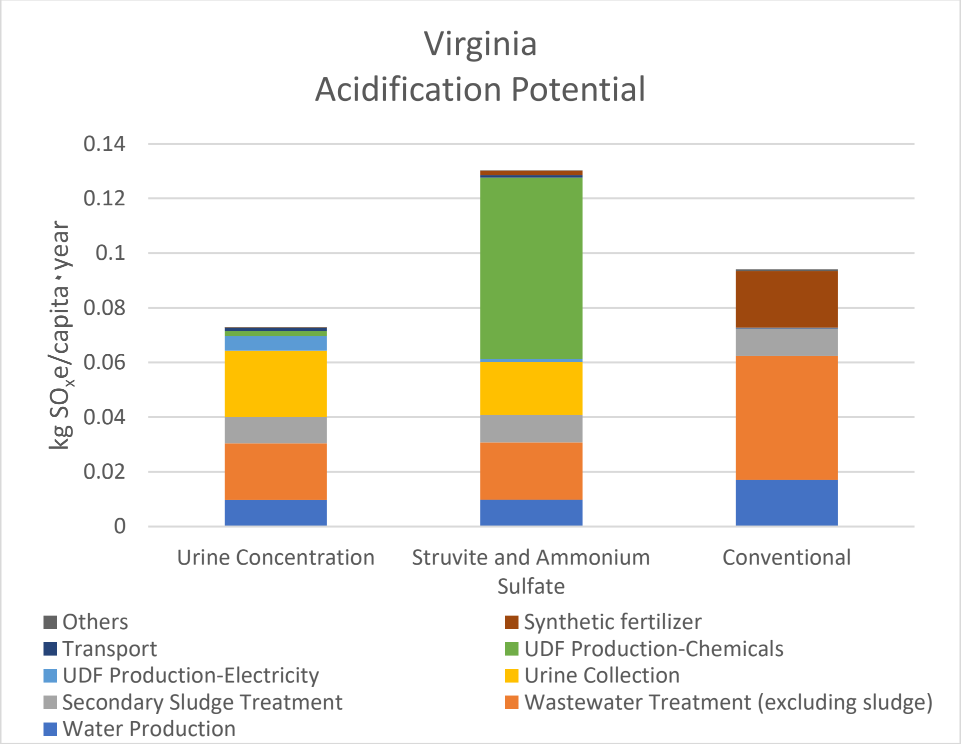

Figure 22. Acidification potential of the Virginia alternatives by Process.

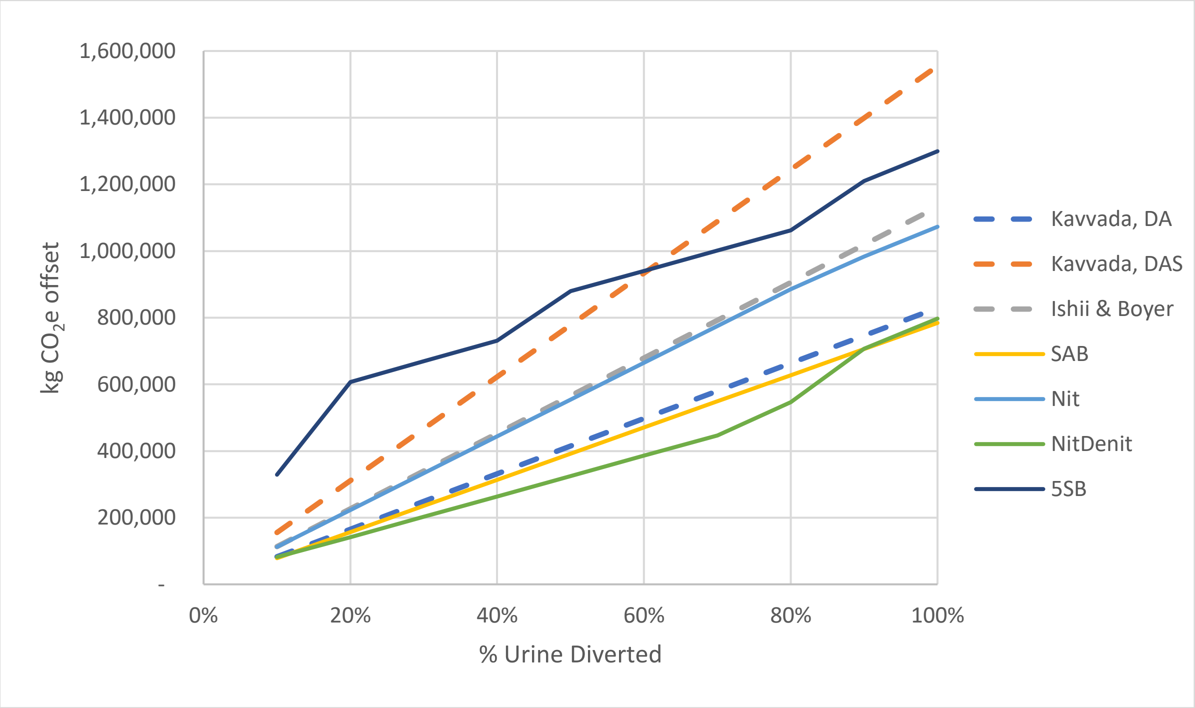

Figure 23. Comparison of WWTP Modeling Methods.

Comparison of GWP savings from wastewater treatment by percent diversion of urine from this study and two other LCAs. All demonstrate urine diversion for a population of 100,000 people and use the average US electricity grid. All measurements are taken at 10% increments. DA is short for "Direct + aeration emissions," DAS is short for "Direct + aeration + substrate *emissions," SAB is short for "Single Aeration Basin," Nit is short for "Nitrification," NitDenit is short for "Nitrification and Denitrification," and 5SB is short for "5-Stage Bardenpho."*

The simpler methods used in other studies to estimate how much urine diversion reduces the greenhouse gas emissions associated with wastewater treatment produce relatively similar results.[48,49] Using a denitrification emission factor without substrate emissions can be a close approximation for less stringent treatment plants, even though some do not actually use denitrification.[49] Using a denitrification emission factor with substrate emissions leads to results relatively close to the stringent 5-Stage Bardenpho treatment plant modeled in this study. Offsetting by volume of wastewater treatment can produce relatively similar results, though it is worth noting that the wastewater treatment plant in Ishii and Boyer's study consumes more electricity per cubic meter of wastewater than most.[48,59] This volumetric approach may underestimate actual savings when the electricity grid uses a large proportion of renewable energy sources, a urine diverting toilet's flush volume is not much lower than the alternative, or when the wastewater treatment plant does not consume an above average amount of electricity per volume wastewater.

Most treatment plants did not have constant improvements as the level of urine diversion increased. Whenever the need for a chemical input is eliminated, increases to urine diversion past that point lead to smaller environmental gains. At certain levels of urine diversion, the treatment configuration of the treatment plant could be simplified, leading to relatively large environmental benefits. Using simplified methods to estimate changes to wastewater treatment will not capture these complexities. They are useful as a reasonable approximation of environmental benefits. The methods in this study would be useful for a more accurate approximation, for a clearer idea of how operations will change (e.g. typically will not need external carbon sources), or comparing the merits and drawbacks of different levels of urine diversion.

APPENDIX G: RESULTS OF THE SENSITIVITY ANALYSIS

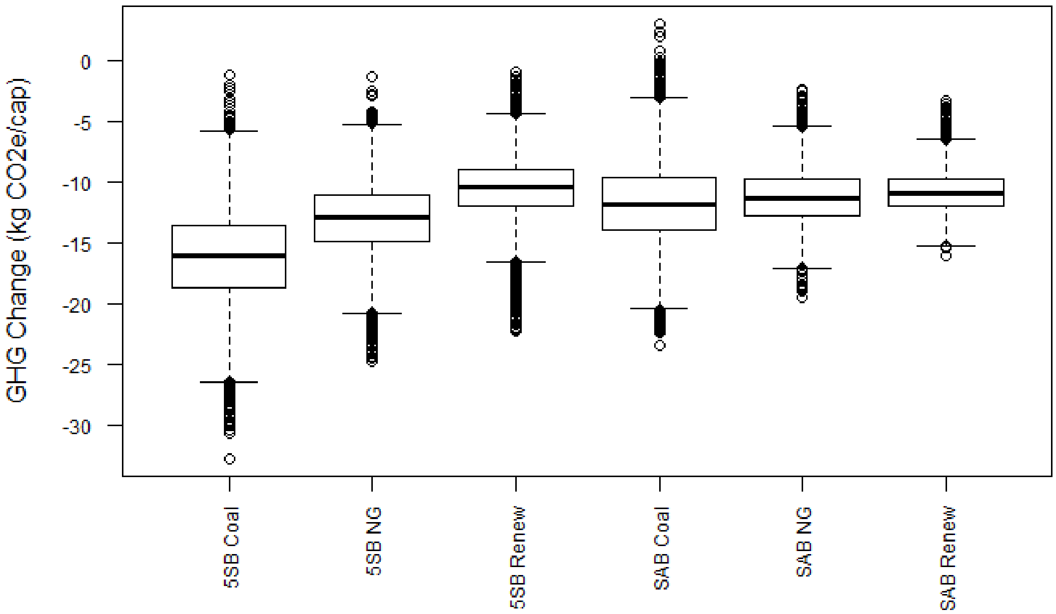

Figure 24. Differences in GHGs from Urine Concentration in sensitivity analysis. Each of the 10,000 simulations calculates the difference between the urine concentration and conventional alternative. The GHG Change is the difference in greenhouse gases. 5SB is short for 5-Stage Bardenpho, and SAB is short for Single Aeration Basin.

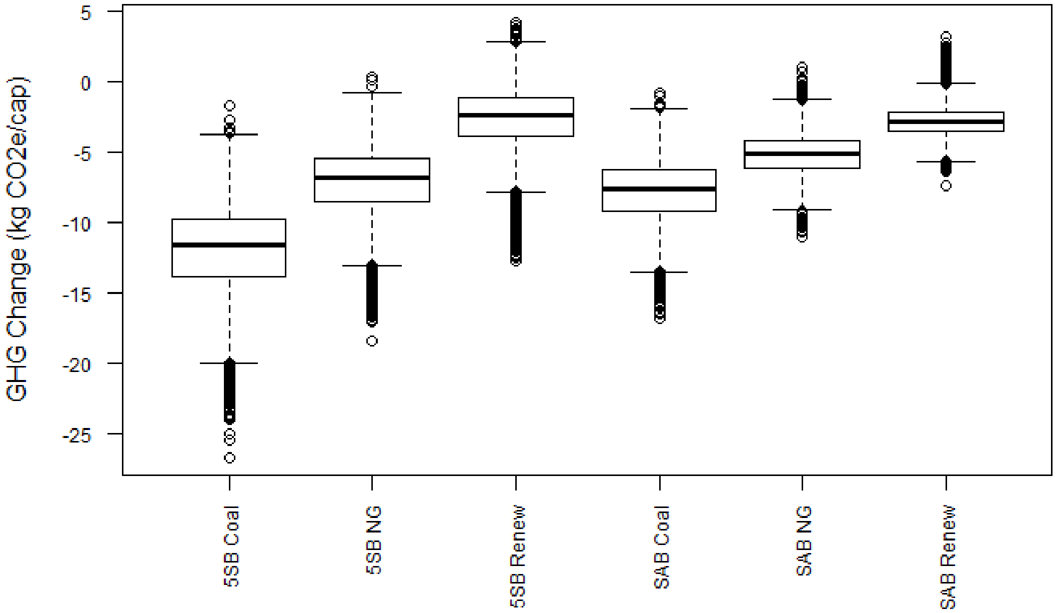

Figure 25. Differences in GHGs from Struvite and Ammonium Sulfate in sensitivity analysis. Box plot of change of greenhouse gas emissions of struvite and ammonium sulfate alternatives compared to conventional scenarios.

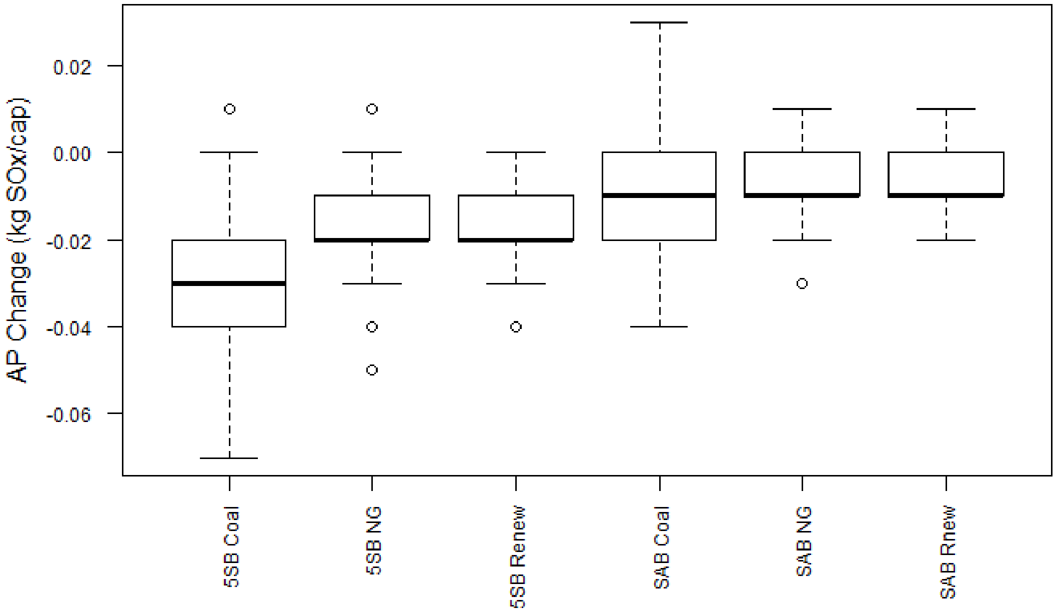

Figure 26. Differences in APs from Urine Concentration in sensitivity analysis.

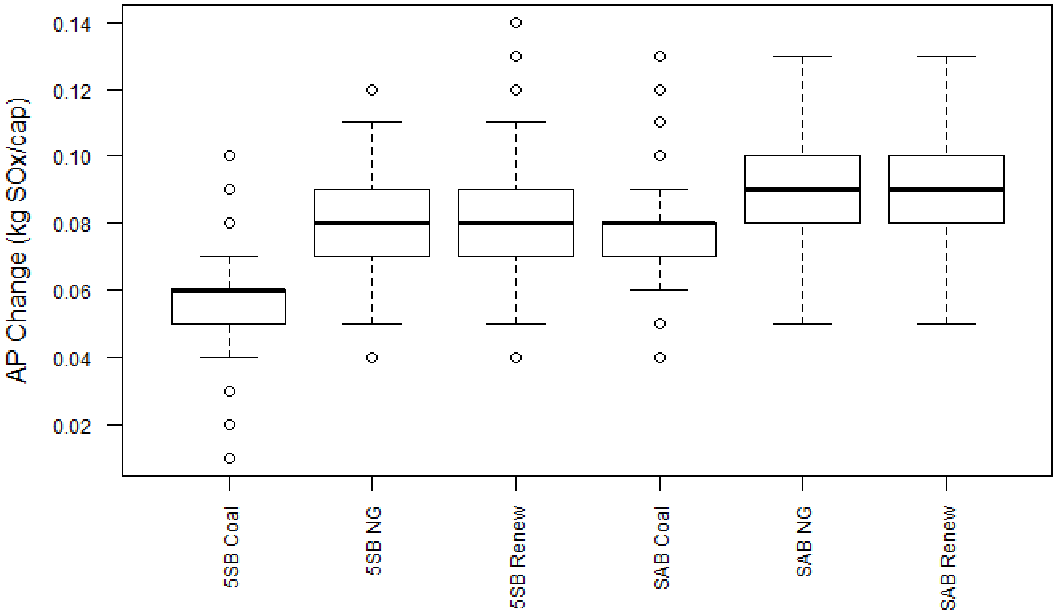

Figure 27. Differences in APs from Struvite and Ammonium Sulfate in sensitivity analysis. Positive values indicate increases in acidification potential.

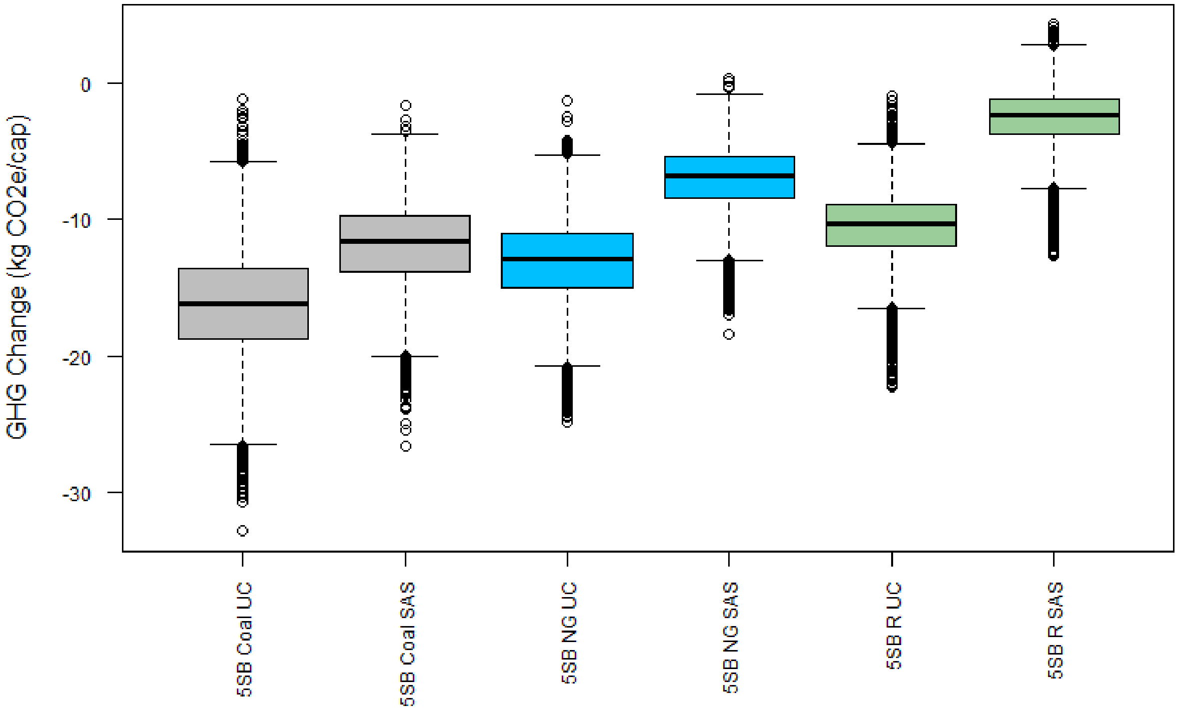

Figure 28. GWP Comparison of Urine Concentration and Struvite and Ammonium Sulfate in 5SB.

All data shown are from the 5-Stage Bardenpho plant modeled. UC is short for Urine Concentration, and SAS is short for Struvite and Ammonium Sulfate. Gray plots indicate coal is used, blue indicate natural gas, and green indicate renewable electricity.

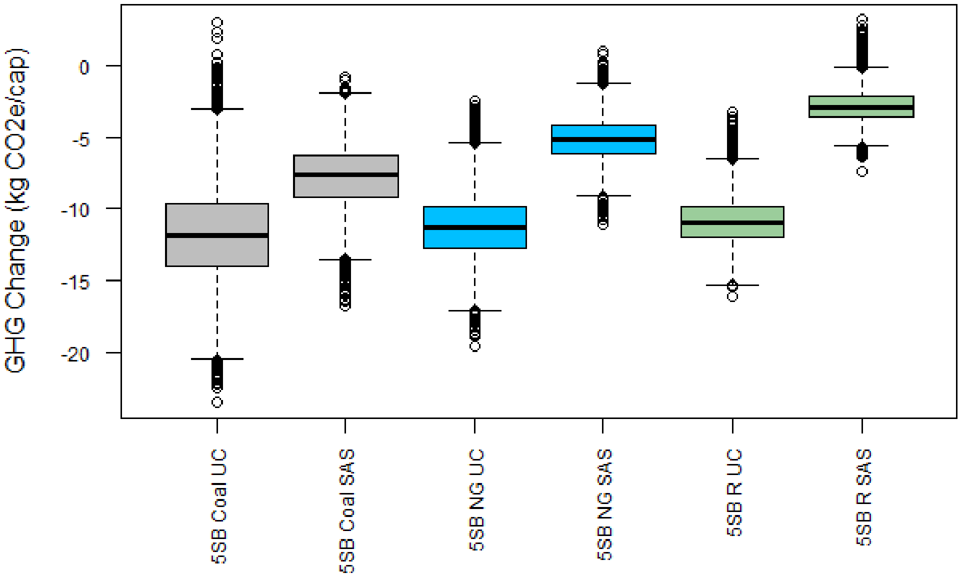

Figure 29. GWP Comparison of Urine Concentration and Struvite and Ammonium Sulfate in SAB.

All data shown are from the single aeration basin plant modeled. UC is short for Urine Concentration, and SAS is short for Struvite and Ammonium Sulfate. Gray plots indicate coal is used, blue indicate natural gas, and green indicate renewable electricity.

Table 21. Urine Concentration Scenarios with lower GWP.
Percent of simulations where urine concentration had lower greenhouse gas emissions than the conventional alternative with different scopes. The first row is what was measured in the study, the second row is if fertilizer offsets were not considered, and the third row is if N_2O emissions from effluent were not considered.

	Urine Concentration					
	5SB Coal	**5SB NG**	**5SB Renew**	**SAB Coal**	**SAB NG**	**SAB Renew**
Standard	100.0%	99.9%	100.0%	100.0%	100.0%	100.0%
W/o fert offset	97.5%	94.5%	77.2%	82.6%	88.7%	92.7%
W/o external N_2O	100.0%	100.0%	100.0%	99.3%	100.0%	1.0%

Table 22. Struvite and Ammonium Sulfate Scenarios with lower GWP.
Percent of simulations where struvite and ammonium sulfate had lower greenhouse gas emissions than the conventional alternative with different scopes. The first row is what was measured in the study, the second row is if fertilizer offsets were not considered, and the third row is if N_2O emissions from effluent were not considered.

	Struvite and Ammonium Sulfate					
	5SB Coal	**5SB NG**	**5SB Renew**	**SAB Coal**	**SAB NG**	**SAB Renew**
Standard	100.0%	100.0%	91.7%	100.0%	99.9%	98.7%
W/o fert offset	85.9%	19.8%	2.6%	27.7%	0.2%	0.0%
W/o external N_2O	100.0%	100.0%	91.4%	99.4%	93.8%	0.4%

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