**Sustainable Wastewater Management:**
**Modeling and Decision Strategies for**
**Unused Medications and Wastewater Solids**

**by**

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# List of Abbreviations

| | |
|---|---|
| 2,4-D | 2,4-Dichlorophenoxyacetic acid |
| ADM1 | Anaerobic Digestion Model No.1 |
| API | Active Pharmaceutical Ingredient |
| BMP | Biochemical Methane Potential |
| C/N | Carbon to Nitrogen |
| CAS# | Chemical Abstract Service Number |
| CFC-11 | Trichlorofluoromethane |
| CSTR | Continuously-Stirred Tank Reactor |
| $CO_2$ | Carbon Dioxide |
| COD | Chemical Oxygen Demand |
| EPA | United States Environmental Protection Agency |
| eq | Equivalents |
| $H^+$ | Hydrogen Ion |
| ISO | International Organization for Standardization |
| LFG | Landfill Gas |
| LCA | Life Cycle Assessment |

| | |
|---|---|
| LCI | Life Cycle Inventory |
| LCFA | Long Chain Fatty Acid |
| LDPE | Low-Density Polyethylene |
| LFER | Liner Free Energy Relationship |
| MSW | Municipal Solid Waste |
| n/a | Not Available |
| N | Nitrogen |
| $NO_X$ | Nitrogen Oxides |
| OFMSW | Organic Fraction of Municipal Solid Waste |
| OLR | Organic Loading Rate |
| ONDCP | White House Office of National Drug Control Policy |
| op-LFER | One-Parameter Liner Free Energy Relationship |
| $PM_{2.5}$ | Particulate Matter (that is 2.5 micrometers in diameter and smaller) |
| pp-LFER | Poly-Parameter Liner Free Energy Relationship |
| TRACI | The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts |
| U.S. | United States of America |
| VFA | Volatile Fatty Acid |
| WWTP | Wastewater Treatment Plant |

# Chapter 1.
# Introduction

With growing environmental, economic, and social challenges, it is becoming increasingly difficult for wastewater treatment plants (WWTPs) to achieve their goal of protecting human and environmental health. These challenges include insufficient funding to maintain and update deteriorating infrastructure[1], a lack of societal acceptance for newer wastewater management approaches (e.g., [2]), and a trend of increasingly stringent effluent permits (e.g., [3,4]). To improve the sustainability of wastewater treatment and address some of the challenges, new management approaches and technologies are being developed, including technologies aimed at recovering resources (e.g., energy, water, nutrients) from wastewater[5–9]. As research offers an ever increasing set of strategies and technologies, comprehensive and comparative assessment methodologies are needed to help decision-makers evaluate new practices for sustainable wastewater management.

While the use of life cycle assessment (LCA) to evaluate the environmental impact of current and new technologies is growing[10–13], there is a need to expand the LCA methodology to improve its usefulness for sustainability decision-making[14]. The development of new technologies and evaluation techniques are creating exciting opportunities to reduce the environmental, economic, and social impacts of current wastewater management approaches. However, new technologies have achieved only limited penetration into practice. This is largely because decision-strategies are not readily available to support the evaluation of new technologies from the economic, environmental, social, and functional (or performance) perspectives, which can be considered a "quadruple bottom line". This dissertation aimed to advance the sustainability of wastewater treatment by developing a quadruple bottom line assessment framework through the evaluation of two case studies focused on key WWTP sustainability issues: unused medication disposal and energy recovery using anaerobic codigestion.

## 1.1  Effluent Quality and Unused Medication Disposal

A key sustainability concern in wastewater treatment involves the quality of a WWTP's effluent[10–13]. The direct environmental emissions of macropollutants (i.e., nitrogen and phosphorus) and micropollutants (e.g., pharmaceuticals, heavy metals) from wastewater effluent and biosolids result in eutrophication and toxicity impacts[10–13]. Progress in reducing eutrophication impacts is being made with advanced nutrient removal technologies and decreasing regulatory emission limits for nitrogen and phosphorus. The emission of micropollutants and their resulting environmental and human health impact, though, are a growing area of concern. Researchers have concluded that micropollutants are an important contributor to wastewater environmental toxicity impacts[15]. While the impacts of many micropollutants still need to be well characterized[12,16], the emission of these compounds from WWTPs is becoming a focus of sustainability and public health efforts. For example, the European Union now requires the removal of several micropollutants from wastewater effluents with its Water Framework Directive[3,4].

The most effective way to reduce micropollutant emissions from wastewater is to prevent pharmaceuticals from entering WWTPs[17]. While there are many sources of pharmaceuticals (e.g., excretion, industrial processes, etc.), the direct disposal of unused medication has been identified as an important source to control[18]. Currently, 40% of U.S. consumers flush their unwanted and unused medications down the toilet and 60% put them in the trash[19]. These medications are a source of pharmaceutical emissions to WWTPs when discarded using a toilet and when put in the trash (due to landfill leachate). As a result, national governments and environmental professionals are increasingly recommending that consumers return their unused medications to a central location for incineration, thereby avoiding introducing pharmaceuticals into the water. This "take-back" recommendation ignores the possibility of exacerbating other known problems, such as air pollution, and neglects the fact that pharmaceutical pollution is expected to be predominately a result of human and animal excretion. Therefore, Chapter 3 of this dissertation determined whether toilet, trash, or take-back disposal for unused pharmaceuticals is best when considering environmental as well as cost and social factors.

Specifically, Chapter 3 describes the integration of LCA methodology with waste treatment models, disposal system models, and uncertainty assessments to quantify and compare the environmental emissions associated with the following disposal options: (i) take-back disposal,

where pharmaceuticals are driven to a pharmacy to be incinerated as hazardous waste; (ii) toilet disposal, where pharmaceuticals are flushed down a toilet to be treated as domestic wastewater; and (iii) trash disposal, where pharmaceuticals are mixed with an unpalatable item (e.g., used coffee grounds or cat litter) to deter diversion and put in the household trash to be managed as municipal solid waste. To quantify the environmental emissions of pharmaceuticals, this study focused on the active ingredient in medication, which is called the active pharmaceutical ingredient (API). For each disposal option, an API was traced through the waste treatment and conveyance process. Models were developed to determine what eventually happens to an API once it enters a WWTP, landfill, and incinerator. These waste treatment models predicted the final fate of APIs, such as emission to the environment or retention in a landfill. The non-API environmental emissions (e.g., greenhouse gas emissions) were estimated by developing disposal system models. These models described the main processes and resources required for each disposal option (e.g., the amount of energy used to drive unused medication to a pharmacy for take-back disposal), and they used U.S. national averages for energy and transportation. Outputs from these disposal system models were translated into life cycle environmental emissions data using life cycle inventory databases. The robustness of the modeling results was tested with an extensive uncertainty and sensitivity analysis. Finally, these results were combined with participation, safety, and cost data from the literature to evaluate the overall sustainability of each disposal option.

## 1.2 Net Energy Use and Energy Recovery from Waste using Anaerobic Codigestion

Another key sustainability concern in wastewater treatment is the net energy use of a WWTP[10–13]. About three percent of the total electric energy used in the U.S. is for wastewater treatment[20]. Current research efforts have been focused on the recovery of energy from wastewater[5–9] as a means of offsetting the environmental impacts and costs associated with a WWTPs use of non-renewable energy sources. For example, the anaerobic digestion of sewage sludge can produce about 15,000 kWh of energy each year for every 1,000 cubic meter of digester volume, which is equivalent to the yearly energy demand of more than two average households[21]. Further, the codigestion of WWTP solids with other organic wastes, such as food waste, can significantly increase this energy production. Food waste was found to produce three times as much methane as WWTP solids[21]. Since anaerobic digestion and codigestion can generate methane from

wastewater solids and reduce a WWTPs net energy balance and environmental emissions, this practice has been found to improve the sustainability of WWTPs[12,22–26]. However, given the tendency of digesters to become unstable under certain conditions[27,28], which can be exacerbated by codigestion[29], the potential of codigestion to reduce the environmental impacts of WWTPs can only be realized if the digester does not fail (i.e., it consistently treats waste and generates energy).

To better understand the link between influent composition and digester stability during codigestion, researchers have conducted batch experiments to determine the methane potential of various waste combinations[29–33] and continuous-flow experiments to evaluate overall digester performance[34–37]. However, these types of evaluations can be resource intensive (e.g., require significant amounts of time, money, laboratory equipment and space). Also, these experimental evaluations need to be repeated for each possible waste and waste combination being considered for codigestion, which also limits the use of a purely experimental approach to determine optimal operating conditions and influent compositions. These limitations exemplify why it is not feasible to experimentally test all possible waste combinations and thus motivate a model-based approach that systematically evaluates the impact of different waste combinations on process stability. So far, codigestion modeling efforts have focused on determining the operational parameters that maximize energy production[38–40]. Of these studies, the one that attempted to experimentally validate their results found their optimization protocol could not select methane-maximizing influent blends unless it considered nutrient limitations[39]. This example highlights that modeling approaches without stability considerations provide limited insight for codigestion operation. Therefore, in Chapter 4 we developed a comprehensive stability assessment and coupled it with an anaerobic digestion biological process model to study the potential impacts of influent composition on process stability.

The boundaries of stable digester operation were defined with multiple stability indicator values that show the potential for process and treatment failure, based on full-scale and lab-scale operation. To test the applicability of the stability assessment and evaluate a large variety of potential waste combinations, codigester performance was simulated for a comprehensive influent set of 10,000 influent compositions. Monte Carlo-based uncertainty assessments were used to evaluate the impact of 35 biochemical parameters on the stability results. These data were evaluated to identify the influent compositions and characteristics that result in stable digester

operation. This information and stability assessment tool can be used to inform the design of codigestion systems that can maximize energy recovery without compromising process stability.

## 1.3  Sustainability Decision Framework

Through the evaluation of these two case studies focused on key WWTP sustainability issues, the need for a sustainability decision framework to incorporate uncertainty assessments and the four pillars of sustainability was highlighted. To this end, this dissertation developed a quadruple bottom line assessment framework. The resulting sustainability Decision Framework (Figure 1) combines the environmental, economic, and social aspects of a waste management approach by building upon the ISO 14040 LCA methodology[41] and incorporating economic and treatment performance models. In order to integrate the performance and reliability of the treatment technologies in the analysis and decision-making process, biological process models were included in the inventory phase of an LCA. Additional analyses considered the evaluation of multiple design and implementation scenarios, along with uncertainty. This framework increases the ability of the wastewater field to engage in sustainability decision-making, and it can be extended to other contemporary issues in environmental engineering.
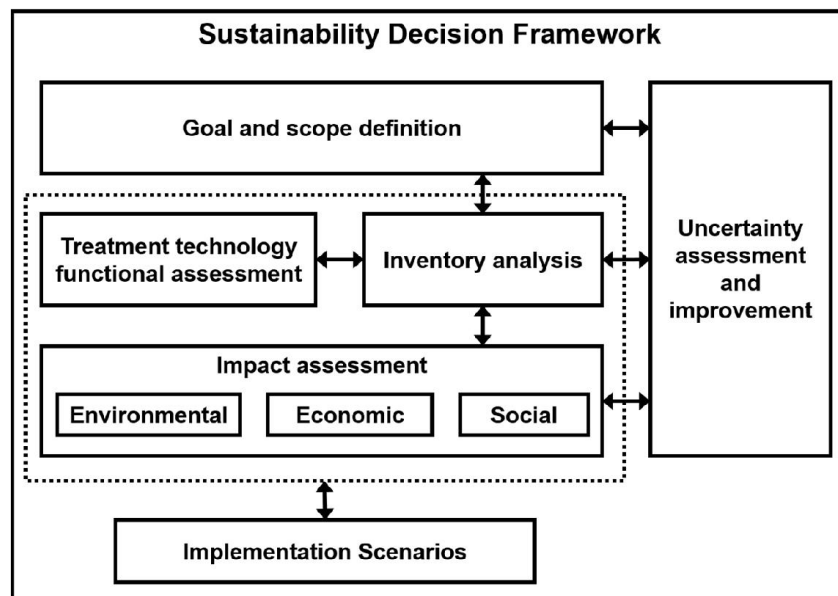


**Figure 1. This dissertation's sustainability decision framework that uses a quadruple bottom line approach for evaluating WWTP sustainability and is based on the LCA framework[41]. A solid line symbolizes a phase. The dotted line symbolizes that the implementation scenario phases will involve changes to all or some of the enclosed phases.**

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# Chapter 2.
# Background

## 2.1 Management of Unused Medications

### 2.1.1 Medications in the Environment

Pharmaceuticals are generally recognized as being widespread in the aquatic environment[42] and this is spurring interest and research on chronic human health impacts as well as ecological health. The main compounds in medications that have pharmaceutical effects are called the active pharmaceutical ingredients (APIs). Advances in analytical instrumentation and methods have enabled the measurement of nanogram per liter concentrations of APIs in the environment[43]. APIs have been found throughout natural aquatic environments[18,44] as well as in wastewater biosolids[45] and in treated drinking water[46]. The risk to human and ecological health posed by this environmental contamination is a current area of research. Researchers have already been able to identify specific compounds that have negative impacts on human and environmental health[18,47–50]. Another area of growing research is the formation, impact, and measurement of API transformation products, also called metabolites. Studies are starting to identify and detect the presence of these transformation products in the natural environment as well[42,43,51].

The environmental impact due to the direct disposal of unused medications to wastewater treatment plants and landfills is not known. To determine the relative impact of direct disposal versus other sources of APIs in the environment, there are three important factors to consider: the total mass of APIs, the type of APIs, and the timeframe of discharge. Sources of APIs in the environment include human and animal excretion (point sources and non-point sources), industrial wastewaters, and direct disposal[44,52]. The relative mass contribution of each source to environmental API emissions is not currently known[52–56], but it is expected that excretion is the largest source[57]. The type of API being released to the environment from each source also needs to be considered and better understood. For example, medications that are ingested can be

extensively transformed into a compound that is very different from the original, parent API. In particular, parent APIs and transformation products can have different environmental fates and toxicological effects[50,58]. The timeframe of an emission of APIs can also influence the environmental impact. For example, the direct disposal of medications to the wastewater collection system of a large quantity over a very short timeframe could result in a pulse load with concentrations that are orders of magnitude higher than continuous API loads due to excretion[53]. Given these uncertainties and unknowns, direct disposal is considered an important source of APIs in the environment.

### 2.1.2 Unused Medication Disposal

Unused medications are expired or unwanted medications that are usually stored in the household. The main risks posed by unused medications are the risk of accidental poisoning, diversion, and abuse[18,55,56,59] and the risk of environmental contamination from pharmaceuticals[18,55,56]. The goal of "proper disposal" is to minimize both of these risks. Historically in the U.S., consumers were instructed to flush their unused medications down the toilet[18]. Since this disposal method can increase emissions of pharmaceuticals to the environmental if the APIs are not removed during wastewater treatment, this recommendation is now discouraged and reserved for only a few medications that are acutely toxic or have a high risk of abuse[18,60]. More recently, the White House Office of National Drug Control Policy has instructed consumers to mix their unused medications with coffee grounds or cat litter before putting it in the household trash[18,60]. Starting around 2007, The U.S. government organizations started advising consumers to use "take-back" programs for disposal. These programs usually consist of consumers bringing their unused medication to a central location in order to be incinerated[18,60].

U.S. legislation is creating the opportunity for the national implementation of take-back programs[61] and to ban other disposal options. For example, the Illinois Safe Pharmaceutical Disposal Act bans flushing disposal by health care institutions[62,63]. In 2014, Michigan passed a bill to promote local collection events[64] and California was trying to pass legislation for the implementation of a state-wide program[65]. Several pilot programs have been operated in the U.S.[66–70] to gain logistical information about a nation-wide take-back program. Table 1 summarizes these pilot programs along with some well-established programs. For many of the pilot and on-going programs the direct cost of participation was free, and the costs of starting and

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running the programs were usually not available. In addition to these disposal sites and programs, several non-profit organizations have focused on increasing the awareness of take-back programs. They usually provide information about the location of take-back programs as well as information regarding the perceived risks associated with storing unused medications in the household and with toilet and trash disposal.

The "Take Back Your Meds" organization is based in Washington State and supports the creation of a statewide take-back program that is financed by pharmaceutical manufacturers. This organization provides information on Washington State take-back disposal options[71]. The organization www.dontflushdrugs.com! provides information about unused medication and disposal risks along with lists of pharmacies participating in take-back programs for Washtenaw County, MI[72].  The Great Lakes Clean Water Organization started a take-back program called "Yellow Jugs Old Drugs" that provides containers and disposal services for local pharmacies[73]. This program was started in Michigan in 2009 and was expanded to Illinois and Wisconsin in 2012.  The program is expected to be available in Ohio and Indiana starting in 2014. Also, the California State government has created a Facility Information Toolbox that provides a directory of locations for the free disposal of sharps and medications within the state[74]. The organization Dispose My Meds provides a free online resource for locating any pharmacy participating in take-back disposal across the entire U.S and is presented by the National Community Pharmacists Association Foundation[75].

**Table 1. A list of unused medication take-back programs in the U.S. as of May 2014.**

| Program | Location | Types of Medications | Duration or Start Date | Direct Cost to Participant | Reported Outcome* | Citation |
|---|---|---|---|---|---|---|
| Clark County Take-Back Program | Clark County, WA; law enforcement locations for controlled substances; pharmacy locations for non-controlled substances | Controlled and non-controlled substances | Started October 2003 | Free | 300 pounds of controlled and 700 pounds non-controlled substances collected in 2008. | [66,68] |
| PH:ARM Pilot Program | Washington State; 37 pharmacies | Non-controlled substances | October 2006 – October 2008 | Free (cost of pilot was $134,000, not including costs estimates for donated time) | 15,798 pounds collected. | [66] |
| San Mateo County | San Mateo County, CA; police stations | Controlled and non-controlled substances | Started 2006 | (not available) | 25,200 pounds collected during the years 2006-2009. | [66] |
| Green Pharmacy Pilot Program | San Francisco Bay Area, CA; 12 pharmacy and office locations; 4 one-day collection events | Non-controlled substances | June 2007-June 2008 | Free | 2,000 pounds collected; identified commonly returned medications and reason for disposal. | [69] |
| Maine Mail-Back Pilot Program | 150 pharmacies across Maine; envelopes were provided for mail-back disposal | Controlled and non-controlled substances | 2007 | Free ($150,000 EPA grant for pilot program) | 250 pounds controlled and 2,123 pounds non-controlled substances collected; program had a 42% envelope return rate. | [68,70] |

*Reported masses represent the "total drug waste" collected (i.e., medication and containers).

**Table 1. Continued.**

| Program | Location | Types of Medications | Duration or Start Date | Direct Cost to Participant | Reported Outcome* | Citation |
|---|---|---|---|---|---|---|
| Chicago | Chicago, IL; police stations | Controlled and non-controlled substances | Started August 2008 | (not available) | 1,000 pounds collected in the first 7 months. | [66] |
| Yellow Jug Old Drugs Program | Pharmacies in the Great Lakes Area (Michigan, Illinois, Wisconsin, Ohio, and Indiana) | Non controlled substances | Started in MI April 2009; Started in IL and WI February 2012; Starting in OH and IN in 2014. | Free | 91,500 tons collected as of January 2014. | [73] |
| DEA's Prescription Drug Take Back Day | More than 5,500 sites across the U.S.; locations at police departments, transfer stations and other community centers | Targeted controlled substances | Started September 2010; Single day events that happen twice a year (eighth event was on April 26, 2014). | Free | 3.4 million pounds collected as of November 2013. | [76,77] |
| TakeAway Environmental Return System™ | Pharmacies across the U.S. (e.g., CVS, Kroger, and RiteAid); envelopes were provided for mail-back disposal | Non-controlled substances | Started services in 2011 | $3.99 to $2.99 per envelope | (not available) | [78–80] |
| Walgreens | Walgreens Stores across the U.S. | Non-controlled substances | Started September 2012 | $2.99 per envelope | 25,000 pounds collected (timeframe n/a). | [81,82] |
| Big Red Barrel Program | Police department lobbies in Livingston and Washtenaw Counties, MI | Controlled and non-controlled substances | Started 2013 or earlier | Free | (not available) | [72,83] |

*Mass collected is of "total drug waste" (i.e., medication and containers).

The effectiveness of take-back programs has been measured by the public's willingness to participate, actual participation rates, and risk assessments. Previous studies have investigated consumers' current disposal practices[18] and the risk management goals of a disposal system for unused medications[84]. More specifically, studies have evaluated consumers' willingness to participate in take-back programs[66,85,86] and implementation strategies to maximize the convenience of take-back programs[67]. They found that 74%-90% of survey respondents said that they would likely participate in a take-back program[66,86]. However, these studies also revealed that disposal inconvenience would increase significantly[67,86]. Interestingly, participation in take-back pilot programs was very low (e.g., during the three years of a collection program in Clark County, Washington, only 30 pounds of controlled substances were collected from less than 350 participants[68] out of a population of 380,000[18]). Studies completed in Sweden, which has had a national take-back program for over 40 years, have found that participation has never been more than 50% and that more than 50% of the citizens seem to have responded to this national program by stockpiling unused medications at home[85]. In addition to expected low participation[59] and increased home storage[55], studies have also estimated that societal costs would increase with an estimated amount of 2 billion dollars per year for a nationwide program[87]; a cost that the public was found to be generally unwilling to pay[86]. While the safety, financial, and participatory aspects of take-back programs continue to be evaluated, studies about the impact of this disposal option on the environment are lacking[55,59]. Research is needed to elucidate the environmental implications of unwanted medication disposal options, specifically by understanding the impact of take-back disposal on environmental pharmaceutical emissions as well as other environmental emissions (e.g., greenhouse gas emissions).

### 2.1.3 Pharmaceutical Removal during Wastewater Treatment

Wastewater treatment plants (WWTPs) provide an opportunity to reduce API emissions to the natural aquatic environment[42], so it is important to be able to predict API fate and removal during wastewater treamtent[43,51]. Data about API removal and fate is very limited and strongly depends on the compound's molecular properties, the wastewater composition, and the treatment processes[88,89]. Since volatilization is expected to be negligible[90], the two main mechanisms of removal are sorption and biodegradation[43,89]. Due to the complexity of pharmaceutical degradation and sorption, the most common approach for reporting pharmaceutical fate is by determining the

overall percent removal of an API parent compound during wastewater treatment[91–98]; the overall percent removals are determined by comparing influent API parent compound concentrations with the WWTP's liquid effluent concentrations. Removal mechanisms can be estimated using this overall percent removal, but this approach does not account for the formation of transformation products. API parent compounds and transformation products can have different environmental fates and toxicological effects[50,58]. While research on transformation products is growing[42,43,51], this section and Chapter 3 will only focus on the "removal" of parent API compounds during wastewater treatment. A further discussion of the limitations of the field's current use of an overall percent removal approach is discussed in the literature (e.g., [91]). Table 2 shows the physicochemical properties (octanol/water partition coefficient, acid dissociation constant, and Henry's law constant) and overall percent removal during wastewater treatment for selected APIs. These APIs were used to represent the most commonly returned medications[69]. This information can be used to estimate API fate during wastewater treatment based on sorption and biodegradation calculations and assumptions.

**Table 2. Pharmaceutical identification, physicochemical properties, and overall percent removal during wastewater treatment, as found in the literature. (n/a is not available)**

| Compound | CAS# | log $K_{ow}$ (citation) | $pK_a$ (citation) | $K_h$ (atm-L/mol)[99] | WWTP Overall Removal |
|---|---|---|---|---|---|
| Acetaminophen | 103-90-2 | 0.46 [99] | 9.4 [99] | $6.42 \cdot 10^{-10}$ | 95% [93] |
| Aspirin | 50-78-2 | 1.2 [99] | 3.5 [99] | $1.30 \cdot 10^{-6}$ | 86% [92,93] |
| Vitamin E | 59-02-9 | 12 [99] | 11 [100] | n/a | 100% [94] |
| Prednisone | 53-03-2 | 1.5 [99] | 12 [100] | $2.83 \cdot 10^{-7}$ | 96% [93,96] |
| Ibuprofen | 15687-27-1 | 4.0 [99] | 4.9 [99] | $1.50 \cdot 10^{-4}$ | 90% [93,97] |
| Warfarin | 81-81-2 | 2.6 [99] | 5.1 [99] | $2.77 \cdot 10^{-6}$ | 80% [95] |
| Topiramate | 97240-79-4 | 1.3 [101] | 9.2 [100] | n/a | 15% [98] |
| Etodolac | 41340-25-4 | 3.9 [99] | 4.7 [98] | n/a | 45% [98] |
| Gabapentin | 60142-96-3 | -1.1 [99] | 4.7 [100] | $1.81 \cdot 10^{-7}$ | 99% [93] |

### 2.1.3.1 Sorption

Collecting experimental data on API sorption to biomass during wastewater treatment is difficult as well as expensive and time-consuming[43]. Therefore, a theoretical modeling approach based on chemical structure is a common method for estimating API sorption. These estimations are based on the concentration of the sorbent (i.e., biomass concentration during mainstream wastewater treatment) and the biomass-water partition coefficient ($K_{biomass}$). The biomass-water partition

coefficient can be calculated using one of two main theoretical approaches for modeling pharmaceutical sorption: one-parameter linear free energy relationships (op-LFERs) and poly-parameter linear free energy relationships (pp-LFERs)[102,103].

For decades, op-LFERs have been developed and applied to estimate sorption of many chemical classes[102]. Using this method, an unknown partition constant is determined by relating it to a known partition constant using a linear double logarithmic relationship. This approach works best when both sorbents have similar properties (e.g., soil and biomass). Commonly known partition coefficients include[102]: octanol-water partition coefficient ($K_{ow}$), which is based on the total change in Gibbs free energy for the transfer of one mole from aqueous solution to octanol; soil-water distribution coefficient ($K_d$), which is the ratio of the concentrations of the chemical in water and in the soil or sediment at equilibrium; and octanol-carbon partition coefficient ($K_{oc}$). There are two main resources for calculating biomass-water partition coefficients using this approach. The Sewage Treatment Plant in Windows is a modeling program available from the U.S. Environmental Protection Agency (EPA) that calculates expected chemical removal during wastewater treatment[90]. Also, Cunningham[58] estimated biomass-water partition coefficients ($K_{biomass}$) from octanol-water distribution coefficients ($D_{ow}$) for different chemical functionalities: acidic, doubly acidic, basic, and zwitterionic.

However, op-LFER-based API sorption estimations may be inaccurate[43,102–104] since pharmaceuticals tend to be large, multifunctional, and ionizable organic compounds[58]. Using a more mechanistic approach that overcomes this limitation, pp-LFERs account for individual intermolecular interactions, such as Van der Waals interactions, polar interactions, hydrogen-bond donor interactions, hydrogen-bond acceptor interactions, and cavity formation[102,103,105]. Overall, pp-LFERs result in more accurate sorption estimations, but the extensive data needed to use this approach are difficult to determine[103] and are not yet available for most compounds[102]. Due to this lack of data, op-LFERs are currently the best available method for determining biomass-water partition coefficients for APIs.

### 2.1.3.2  Biodegradation

Biodegradation is the other main mechanism for pharmaceutical removal during wastewater treatment. There are three main fates of API compounds when biologically degraded; they can be:

(1) mineralized to form carbon dioxide[52]; (2) assimilated into biomass[106]; and (3) transformed into a different compound (i.e., transformation product or metabolite)[107]. Data about pharmaceutical degradation can be even more limited than sorption data, based on the API[52], especially since many APIs can be degraded into a large number of mostly unknown transformation products[107]. While little is known about the reaction pathways for many APIs, researchers have found that dehalogenation, dehydrogenation, sulfur addition, and oxidation are common transformation reactions[108–110]. More data are becoming available on transformation products as analytical methods are improving the detection of these compounds[50,111].

Similar to sorption, the extent of biodegradation is specific to an API. For example, researchers have found that many important antibiotics, such as ciprofloxacin and sulfamethoxazole, were not readily biodegradable[52,112,113]. On the other hand, common pain killers like aspirin and ibuprofen have been found to biodegrade[88,90]. In addition, API biodegradation is also dependent on the treatment environment, in particular the types of electron donors and acceptors[96,107,114]. For example, anaerobic degradation is expected to be unlikely[107]., but newer research on anaerobic biodegradation shows that it is possible in landfills and anaerobic digesters. Musson et al.[114] found that acetylsalicylic acid (aspirin) was significantly anaerobically degraded under landfill conditions. Further, Carballa et al.[115] found that biomass acclimation to certain pharmaceuticals during anaerobic digestion can increase the anaerobic biodegradation of multiple compounds.

Biodegradation extent and rates are commonly evaluated using batch studies[116–118]. For these studies, an environmentally relevant concentration of an API is placed in a reactor with freshly collected wastewater biomass. Both abiotic (i.e., biological activity is inhibited) and biotic conditions are tested. The reactors are incubated using relevant temperatures, durations, and mixing conditions. APIs with appropriate radiolabeling can be used to elucidate biodegradation pathways explicitly. During the course of the experiment, samples of the solution and gas (as applicable) are periodically collected to determine biodegradation rate (typically pseudo first-order kinetics[117]) and/or transformation pathway. Since redox state[96,107,114] and microbial community[116] impact degradation, researchers have conducted these experiments under a variety of conditions to better understand pharmaceutical fate during wastewater treatment. In addition to using a purely experimental approach to estimate biodegradation, computer models are also available to predict the biodegradation of APIs[93]. One example is the U.S. EPA's BIOWIN[TM] model, which is part of

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the Estimation Program Interface Suite[TM], and it estimates the aerobic and anaerobic biodegradability of organic chemicals [119].

Onesis et al.[93] reviewed the biodegradation literature for many different APIs and reported their expected values. For several compounds, the biodegradation values varied among studies. This is expected because the results of biodegradation studies are highly dependent on the experimental conditions (e.g., acclimation of biomass to substrate)[52], especially experimental substrate concentrations[93,117,120–122]. One expected, main reason for the discrepancies between biodegradation studies is the occurrence of cometabolism[93]. Cometabolism is the degradation of a substrate due to microbial enzymes (i.e., the biomass do not use the substrate as a carbon source)[43]. Some researchers expect that API biodegradation might be due mostly to cometabolism[120], and cometabolism is closely linked with substrate concentrations. Tan et al.[122] found that increasing concentrations of organic carbon increased the degradation rates of the compound estrone. High organic concentrations can support higher biomass concentrations and promote cometabolic degradation[120]. Another possible reason for biodegradation discrepancies is the acclimation of the biomass to the substrate, specifically to the APIs[52]. However, one recent study found that pre-exposing the microbial community to a trace organic compound did not influence the structure or function of the microbial community[123].

Overall, due to the strong dependence of API biodegradation on multiple factors and the large range in biodegradation data for a given API, an API's primary removal mechanism is usually assumed based on overall percent loss and sorption data. For example, since ibuprofen has been observed to have high percent removal values and a low biomass-water partition coefficient (i.e., low sorption expectation), biodegradation is expected to be the main removal mechanism[107] (i.e., more ibuprofen mass is assumed to be removed during wastewater treatment due to biodegradation than sorption). Using this approach, Chapter 3 developed a method to quantify the percent loss of each representative API expected due to sorption and due to biodegradation.

### 2.1.4  Environmental Emissions of Unused Medication Disposal

Life cycle assessment (LCA) is a method to quantify the environmental emissions associated with a product or process. The principles and framework of an LCA are outlined by the ISO 14040 standards[41]. An LCA has four phases: goal definition and scoping; inventory analysis; impact

analysis; and improvement analysis. In the first phase of a comparative LCA, a functional unit is chosen to provide a common, fixed unit that allows for the comparison of all the study's scenarios and systems. Once the study's scope and boundary are set, the major flows of energy and materials (for all of the unit processes within that boundary) are quantified during the inventory analysis. Some common databases to help estimate these mass and energy flows and their resulting chemical emissions include the Ecoinvent,[124] Franklin USA 1998,[125] and U.S. Life Cycle Inventory[126] databases. The environmental impact from the environmental emissions (to air, soil, and water) and raw material use associated with all of the unit processes are then assessed in order to determine the overall impacts of the studied system. A variety of methods are available to complete this impact assessment step. The general method is to aggregate the hundreds of emissions into a predefined set of categories, based on emission equivalence calculations (e.g., global warming potential in equivalent carbon dioxide mass). Each impact assessment method uses a unique set of characterization factors and categorical discrepancies. One such method developed by the U.S. EPA is The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI)[127]. Overall, an LCA can inform the environmental dimension of sustainable design and decision-making, such as by comparing different processes or products or highlighting the most environmentally impactful part of the life cycle.

Studies have quantified and analyzed the "cradle-to-gate" environmental impacts of pharmaceuticals by focusing on synthesis and manufacturing[128,129]. One study found that solvent use and its potential re-use had the greatest influence on overall environmental impact[129]. Another study found that energy efficiency was the most important factor[128]. None of these studies, though, look at the use or end-of-life of pharmaceuticals. One study focused on end-of-life issues by determining the best management practice for the disposal of plastic medication containers[130]. It recommended recycling the containers unless take-back disposal is used, at which point the unused pharmaceuticals should remain inside their original containers for disposal and incineration. Cradle-to-cradle advice for decreasing the environmental impact of pharmaceuticals is discussed in the literature. For example, one study discussed several options for pollution prevention, such as improving drug design and delivery (e.g., doses)[131]. Although many of the concepts for considering the full life cycle of pharmaceuticals are present in the literature, end-of-life impacts from pharmaceuticals are missing. In order to compare disposal methods for unused medications, these end-of-life impacts need to be understood and quantified.

## 2.2 Modeling the Stability of Anaerobic Digestion

### 2.2.1 Anaerobic Codigestion

Anaerobic digestion is the biological degradation of organic material into methane-rich biogas and nutrient-rich effluents. The entire degradation process is highly inter-linked and dependent on interactions among a wide range of microbial populations. The digestion process can be described by the so-called anaerobic food web and consists of four main steps: hydrolysis, acidogenesis, acetogenesis, and methanogenesis. Hydrolysis is the degradation of complex polymers (e.g., carbohydrates, lipids, and proteins) into soluble monomers (e.g., sugars, amino acids, and long chain fatty acids), and it is assumed to be catalyzed by extracellular enzymes[132]. Acidogenesis or fermentation is a microbial process that results in the transformation of soluble substrates into intermediate compounds such as volatile fatty acids (VFAs), alcohols, and hydrogen gas[132]. Fermenting bacteria, such as Clostridia and Bacteroidetes, are mostly responsible for hydrolysis and acidogenesis.[133] Acetogenesis results in the conversion of organic acids into acetate and other simple products such as hydrogen and carbon dioxide, and it is characteristic of syntrophic relationships. For example, the degradation of saturated fatty acids[134] and propionate[135] occurs due to the syntrophic relationship between proton-reducing acetogens and methanogens. Since the oxidation of these VFAs is energetically unfavorable, the acetogens rely on the consumption of hydrogen, formate, and acetate by methanogens[132,135]. Thus, the last step in the digestion process is methanogenesis, which is the formation of methane gas from acetate as well as from carbon dioxide and hydrogen. The acetate-utilizing methanogens are aceticlastic (e.g., *Methanosarcinales*), and the hydrogen-utilizing methanogens are hydrogenotrophic (e.g., *Methanobacteriales* and *Methanomicrobiales*).

Anaerobic codigestion is the simultaneous digestion of multiple types of organic wastes and is usually employed to increase methane production. The benefit of increased energy production is usually due to an improved nutrient balance in the digester[136–138]. Typically, wastes with complementary characteristics are combined to achieve this nutrient balance; for example a high strength wastes with low nutrient concentrations (e.g., potato processing waste) would be combined with a high nutrient content waste (e.g., manure or WWTP sludge) to improve the nutrient balance. Several waste combinations have been found to result in greater methane production during codigestion than when digested alone. For example, animal manure has been

combined with the following wastes to increase energy production: mixed food waste[35,37,38], whey[30,138–140], fruits[32], vegetables[32,35], biodiesel waste[39,138,141], paper sludge[31], and olive mill waste[142]. A couple of examples of wastes that have been successfully codigested with WWTP sludge are: municipal solids waste (MSW)[25,36,143–145], fats, oil, and grease (FOG)[34,146], biodiesel waste[147], food processing wastes[35], and mixed food waste[148–150]. Other benefits of codigestion include reducing waste volumes[151,152], diverting wastes from landfills[36,152], and reducing costs[25,151,153,154]. In addition, the benefits of codigestion can be realized with little to no extra infrastructure by using existing digester capacity, i.e., existing digester can be operated at higher organic loading rates (OLRs).

Despite the many potential benefits of codigestion, the codigestion of several types of organic wastes can increase a digester's tendency to become unstable. Codigestion introduces the risk of inhibition[155,156], poor effluent quality[138], digester foaming[157,158], and potentially process failure in cases when a waste has not been well characterized. Instability is usually due to a disruption of the anaerobic food web related to poor growth or inhibition. It ultimately manifests itself as an accumulation of VFAs and lowering of pH, a reduction in methane production, and the digester's inability to treat wastes (i.e., reduce waste volumes and reduce effluent COD). In the worst case, the digester's OLR would need to be substantially reduced to allow recovery or the digester may even need to be emptied and restarted. Digester instability can increase the cost of operation due to chemical addition to counteract a pH decrease (e.g., alkalinity addition), decreased methane production (and associated energy loss), and decreased capacity to treat waste (especially during reactor downtime)[159].

Lab-scale and pilot-scale reactors have been constructed and operated to test the impact of co-substrates on digester performance and to better understand these instability issues. Several waste combinations have been evaluated for increased methane production using biochemical methane potential (BMP) assays[29–33]. However, BMP data provide only limited insight into overall digester performance and stability[30] since it is derived from short-term batch experiments. Compared to semi-continuous or continuous reactor experiments, these batch studies cannot account for acclimation or shock loads. Batch experiments also have the potential to result in nutrient limitations or product accumulation, which can lead to inhibition that would not be experienced in with a different reactor configuration. Continuously and semi-continuously fed reactors have been

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used to evaluate digester performance during the codigestion of different waste combinations (e.g., [34–37]). While these and similar studies have been able to evaluate the codigestion potential of many types of wastes, it is not feasible to experimentally test the impact of all possible wastes and waste combinations on digester stability. This experimental limitation can limit the amount of waste that is anaerobically digested and therefore limit the amount of resources that can be recovered from waste. An entirely experimental approach hinders the ability to design digesters that can maximize energy production without compromising digester stability. A comprehensive modeling approach, as presented in Chapter 4, could systematically evaluate more waste combinations than would be experimentally possible. It would also allow for a better understanding of the link between influent composition and digester stability.

### 2.2.2  Assessing Digester Process Stability

Since anaerobic digestion is a complex biological process that involves several interdependent microbial groups, the ability to thoroughly understand the various process steps improves the assessment of overall process stability[159]. Process stability is affected by several environmental conditions, such as the presence of inhibitory compounds and the OLR[27,28]. An ideal stability indicator is a parameter that reflects the metabolic status of a digester and is easy to measure[160]. The most commonly used and recognized stability indicators are biogas composition and production, pH, concentrations of alkalinity and VFAs, and solids or chemical oxygen demand (COD) reduction[151,159,161].

Stability indicators and their target values commonly found in the literature are compiled in Table 3. Among these indicators, biogas production is the most common[161]. Methane production provides insight about overall digester performance since it is the final product of digestion and a form of energy. While methane generation can be measured in real-time[161], biogas data are usually best at indicating failure as opposed to predicting how close a digester is to failure[159,162]. Similar to methane generation, solids reduction or COD removal provides information about the overall digester performance and is another common approach for monitoring digesters[160,163]. Both of these variables are heavily influenced by the influent and waste characteristics, especially biodegradability. Consequently, they serve as important metrics for evaluating the quality of waste treatment and degree of waste reduction. The pH in the digester represents an important environmental variable since it strongly influences microbial function. For example, the optimal

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pH range for syntrophs and methanogens has been found in the range of 6.5-8.2[164–166]. A pH value outside of this range can result in microbial inhibition and a drop in pH can indicate the accumulation of VFAs, which represents an imbalance in the digestion process. Like methane, pH is limited in its ability to predict process failure[159,160]. Alkalinity represents the digester's buffering capacity and ability to handle temporary changes in organic loading and acid production. Coupling pH with the alkalinity concentration can improve digester failure predictions. For example, pH is a better indicator of potential failure in digesters with low alkalinity[161]. Another common stability indicator is the ratio of total VFA concentration to the alkalinity concentration. It has been successfully used to evaluate digester stability[36,37,159,160], but its application to model outputs can be limited since the use of a ratio (and division) can make it harder to identify unrealistic concentrations (i.e., the VFA-to-Alkalinity ratio value may be within expected stable ranges but the absolute VFA concentration could be at inhibitory levels).

**Table 3. Stability Indicators commonly found in the literature.**

| Indicator | Units | Target Values | Citations |
|---|---|---|---|
| **pH** | $\log [H^+]$ | 6.1 – 8.3 | 163,167,168 |
| **Alkalinity** | mg $CaCO_3$/L | 2,000-20,000 | 35,39,138 |
| **Free ammonia** | mg $NH_3$-N/L | Less than 150 | 31,137,163 |
| **Biogas Composition** | % Methane (by volume) | Greater than 55 | 163,167 |
| **Volatile Solids or COD reduction** | (%) | Influent specific | 160,163,169 |
| **Acetate** | mg COD as acetate/L | Less than 850 | 167,170 |
| **VFA to Alkalinity Ratio** | mg acetate equivalent/ (mg $CaCO_3$ equivalent) | Less than 0.4 | 36,159,160 |
| **Long Chain Fatty Acids** | mg COD as LCFA/L | Less than 1,400 | 171,172 |
| **VFA$_{C2-C5}$** | mg COD as C2-C5 VFAs/L | Less than 3,700 | 167,173 |
| **Ammonium** | mg $NH_4^+$-N/L | Less than 5,000 | 31,163 |
| **Methane Yield** | L $CH_4$/kg VS | Comparative or based on baseline | 151,159,161 |
| **Hydrogen** | L $H_2$/kg VS | Dynamic and based on baseline | 159,160,162 |
| **Propionate to Acetate Ratio** | kg propionate as acetate equivalent/kg acetate | Less than 1.4 | 167,170 |

VFAs are important intermediates in the digestion process, and their build-up can reveal slowly developing failures[160]. It is common for digester operators to use only the digester's acetate concentration or only the total VFA concentration to monitor digester stability[161]. The build-up of VFAs can signal a breakdown in the anaerobic food web and cause the pH to drop to inhibitory levels. Acetate is a precursor to methane, and a buildup of acetate can signal a problem with the aceticlastic methanogens[170]. Also, high acetate concentrations can inhibit propionate and ethanol degradation[174]. Compared to the other VFA concentrations, acetate concentrations have been found to respond the fastest to destabilizing conditions[161], which means that acetate might be the most indicative of process stability. In addition, researchers have used operational data to determine maximum acetate concentrations expected in a stable digestion system[167,170].

Since an increase in the propionate concentration relative to the acetate concentration can signal a problem with the hydrogenotrophic methanogens[170], the ratio of propionate to acetate has also been used to indicate stability[167,170]. Furthermore, propionate concentration alone has been found to indicate digester stability[36,161]. However, propionate degradation is the least thermodynamically favorable among the VFA degradation pathways, and propionate degraders are the slowest growing VFA degraders[161]. As a result, high propionate concentrations may be best at indicating instability as opposed to how close a digester is to failure. Due to these slow kinetics, propionate concentrations have been found to remain elevated after an upset event[175]; so propionate has been identified as a good indicator of a digester recovering from an upset event and returning to stable operation. Unlike acetate, maximum propionate concentrations expected in a stable digester are not well cited. One digestion study revealed that stable digestion could be achieved at propionate concentrations near 700 mg/L[176], while another study claimed that the digester propionate concentrations were high near 200 mg/L[177]. A third study observed full-scale digesters and found that digestion instability was likely at acetate concentrations above 800 mg/L and propionate-to-acetate ratios greater than 1.4; the propionate concentrations at failure were not reported[170]. From this study's stability criteria, the absolute maximum propionate concentration for a stable system would have to be less than 1100 mg/L.

In addition to these commonly used stability-indicating parameters, the concentrations of inhibitory compounds also provide important process stability information. Several compounds have been found to cause the inhibition of different microbial groups. Compounds that result in

decreased biological activity, compared to the activity levels experienced in the absence of that compound, include[178]: salts (e.g., sodium, potassium, calcium, magnesium), ammonia nitrogen, sulfides, heavy metals (e.g., copper, zinc, nickel), long chain fatty acids (LCFAs), and organic solvents (e.g., methanol). Specifically, the inhibitory compounds that have been used most often to indicate digester stability are free ammonia[31,137,163], ammonium[31,163], LCFAs[179–182], and hydrogen gas[159,160,162]. Ammonia nitrogen plays an influential role in digester stability since it serves as a source of nitrogen for biological growth and has inhibitory effects at high concentrations, such as around 150 mg $NH_3$-N/L[31,183], specifically to aceticlastic methanogens[132]. The maximum free ammonia concentration to avoid inhibition is dependent on digester operation since a digester's microbial community can become acclimated to higher concentrations, such as concentrations around 500 mg $NH_3$-N/L[163]. The occurrence and mechanisms of LCFA inhibition are less frequent and more complex, respectively, than ammonia inhibition[132]. LCFA inhibition involves multiple LCFA compounds and mechanisms. For example, the toxic effects of oleate and stearate have been examined. One study found that the effect could not be reversed by dilution[179] and other studies found that the addition of calcium could reduce the inhibitory effect of these LCFAs[182]. Also, some researchers have found that the acclimation of a digester to high LCFA concentrations is possible[180,184]. The value of hydrogen gas as a stability indicator has been debated[159,160,162]. Hydrogen can be inhibitory to fatty acid degraders and strongly impact the electron transport chain (and thermodynamics of methanogenesis)[174,185]. Despite the intricate role hydrogen plays in the digestion process, the reason for changes in hydrogen gas concentrations are not usually apparent and the change in concentration is specific to the digester[159,160]. Both of these characteristics limit the use of hydrogen as a universally applied stability indicator.

Several different control schemes have been developed for anaerobic digestion that rely on these key stability parameters. Some of the first control strategies focused only on methane gas[186] or hydrogen gas[185] production. More current strategies have expanded on these approaches by including more stability parameters, such as pH, alkalinity, and VFA concentration. For example, one control strategy used biogas flowrate and pH to determine if the influent OLR could be increased without destabilizing the system[187]. Another study developed a simple model that combined real-time measurements of VFAs and biogas production for dynamic calculations of digester stability[28]. While there is debate in the literature about the best approach for monitoring digester stability[159], the improvement of control strategies has shown that increasing the number

of key stability parameters monitored can improve stability predictions, and therefore digester control. For modeling efforts, the limitation of an easily measured indicator is removed, and this provides an opportunity for an even more inclusive approach for evaluating stability. Given the diverse performance and metabolic data provided by each stability indicator as well as their cited limitations, a comprehensive stability assessment that combines several stability indicators is expected to provide design insight for stable digester operation.

### 2.2.3 Modeling the Anaerobic Digestion Process

There are many models available to describe the anaerobic digestion process. The earliest models described these metabolic processes using a limited number of microbial groups and reactions (e.g., [188–190]). More complexity has been added to these basic models to improve the prediction of digester performance (e.g., [191–198]). In 2002, the International Water Association's Anaerobic Digestion Model No. 1 (ADM1) was published as a result of a collaborative effort[132]. This generalized digestion model was based on the concepts and frameworks of previous models (e.g., [195–198]), and it was developed to provide a tool for digester design, operation, and optimization[132,199]. The ADM1 is a structured mathematical model that simulates the digestion process using five main conversion steps: (1) disintegration of composite material (dead biomass) into carbohydrates, proteins, lipids, and inerts; (2) hydrolysis of carbohydrates to sugars, of proteins to amino acids, and of lipids to sugars and LCFAs; (3) acidogenesis of sugars and amino acids to VFAs and hydrogen; (4) acetogenesis of LCFAs and VFAs to acetate; and (5) methanogenesis from acetate as well as from hydrogen and carbon dioxide. The involved gaseous and aqueous digester components are represented with 26 dynamic state concentration variables and eight implicit algebraic variables (for the differential and algebraic question implementation, which can be used to reduce model stiffness and computational time).

The main limitations of implementing the original ADM1 model are: (1) acidogenesis only results in organic acid formation and other intermediate productions, such as ethanol and lactate, are not included. In systems with high OLRs, predictions of pH and organic acid concentrations are less accurate due to this omission. (2) Sulfate reduction and sulfide inhibition are not included in the ADM1 calculations. For systems receiving sulfate in the influent, the level of biological activity and methane production may be overestimated. (3) Nitrate reduction and nitrogen oxide inhibition are not included in the model, which primarily results in an overestimation of methane production

rates for influents that contain nitrogen oxides. (4) Solids precipitation may be significant in systems with high levels of calcium or magnesium, but this process is not included. (5) LCFA inhibition is not included even though toxic overloads are possible. Aceticlastic methanogens are likely the most sensitive to high LCFA concentrations and methane production may be overestimated for high-lipid influents. (6) Acetate and hydrogen oxidation can be significant anaerobic pathways but are not included. Acetate-oxidation is expected to become a predominant pathway at thermophilic temperatures, and hydrogen oxidation (homoacetogenesis) is expected to be significant at low temperatures (less than 20°C). Due to these limitations and the increasing application of the ADM1, many model extensions and modifications have been developed (reviewed here [200]). There are three main types of sub-models that improve the modeling of codigestion using ADM1: (1) interface models that integrate ADM1 with WWTP mainstream treatment models (by defining the mapping of compounds between models)[201,202]; (2) models that extend ADM1 to simulate the codigestion of multiple waste streams with different waste characteristics[38,203,204]; and (3) models that improve the characterization of waste components[205–207].

ADM1, with and without model extensions, has been validated with full-scale[143] and lab-scale[38,203] codigestion systems, as discussed below. Derbal et al.[143] found that ADM1 simulation results had an acceptable average fit to the full-scale performance data of a system codigesting waste activated sludge and the organic fraction of municipal solid waste. Zaher et al.[38] adjusted ADM1's model structure to separate the hydrolysis kinetics of food waste and manure and found that ADM1 simulations matched experimental gas production data. Boubaker et al.[203] found that the ADM1 simulation results were similar to the biogas flow and composition, pH, and VFA concentration data from a lab-scale system that was codigesting olive mill wastewater and solid waste. In addition, Astals et al.[141] used a modified version of ADM1, which included hydrogen sulfide[208] as a state variable, and found that simulated gas production matched experimental results from the codigestion of pig manure and glycerin. These studies show that ADM1 is a useful tool for modeling the anaerobic codigestion process. Chapter 4 explores the use of ADM1 to inform codigestion design.

Digestion models have been used to determine operational parameters that can maximize methane production, but these optimization efforts have lacked an assessment or inclusion of process

stability. For example, Zaher et al.[38] used ADM1 to estimate the influent blend of manure and kitchen waste that allowed for a maximum methane flowrate. Balmant et al.[40] developed a simplified digestion model and optimization protocol to determine optimal retention times that result in maximum methane generation. While these two studies provided mechanisms to estimate maximum energy production, they did not explicitly consider the impacts of these optimal conditions on process stability and the optimal conditions were not experimentally validated. Alvarez et al.[39], though, developed an optimization protocol to maximize methane production and applied it to identify the optimal blend of manure, fish, and biodiesel waste. This study experimentally tested the optimal influent blend and found that the actual methane production of those blends was less than 10% of the expected (theoretical) methane production. Possible reasons for this discrepancy between the optimization modeling and experimental results include LCFA inhibition and insufficient nitrogen to support biological growth. Further, they found that their modeling approach to determine optimal influent blends had better agreement with experimental results if stability constraints were included (i.e., COD to nitrogen ratio). This example highlights the importance of evaluating modeling results in terms of stability. Overall, there has been limited use of a model-based approach for understanding codigestion stability to inform design. This points to an opportunity to use ADM1 to identify the stable operating space for a digester. With this insight, a digester can be operated to achieve maximum methane production within this stable operating space to promote reliable energy recovery.

# Chapter 3.
# Life Cycle Comparison of Environmental Emissions from Three Disposal Options for Unused Pharmaceuticals

## 3.1 Introduction

The disposal of unused, unwanted, or expired pharmaceuticals contributes to the occurrence of active pharmaceutical ingredients (APIs) in aquatic environments,[18,44] wastewater biosolids,[45] and treated drinking water.[46] Since these compounds can have negative impacts on both human and environmental health,[18,47–50] incineration is being utilized as a way to eliminate unused pharmaceuticals as a source of APIs in the environment. "Take-back" disposal, which entails consumers transporting unused pharmaceuticals to a collection site for incineration, is increasing in popularity as an environmentally-conscious means to dispose of unused pharmaceuticals. Several take-back disposal programs have been recently piloted in the United States (U.S.).[66–69]

Previous studies have investigated current practices[18] and risk management goals[84] for the disposal of unused pharmaceuticals. Other studies have evaluated the cost[87] and convenience[67] of take-back programs, as well as the public's willingness to pay[86] and participate[85] in them. Amidst this research, the number of voluntary take-back disposal programs is increasing, and new U.S. legislation is making the national implementation of take-back programs a possibility.[61] On the other hand, the majority of environmental API emissions may arise from human and animal excretion; further, the impact of the disposal practices on non-API emissions (e.g., greenhouse gas emissions or smog forming potential) is not yet established.[55]

This study quantifies the environmental emissions of APIs as well as the emissions of other non-API substances that result from the disposal of unused pharmaceuticals. Using life cycle assessment methodology, the study compares the three disposal options illustrated in Figure 2: (i) Take-back disposal where pharmaceuticals are driven to a pharmacy to be incinerated as hazardous waste; (ii) Toilet disposal where pharmaceuticals are flushed down a toilet to be treated as domestic wastewater; and (iii) Trash disposal where pharmaceuticals are mixed with an unpalatable item and put in the household trash to be managed as municipal solid waste (MSW), as recommended by the White House Office of National Drug Control Policy (ONDCP).[18]
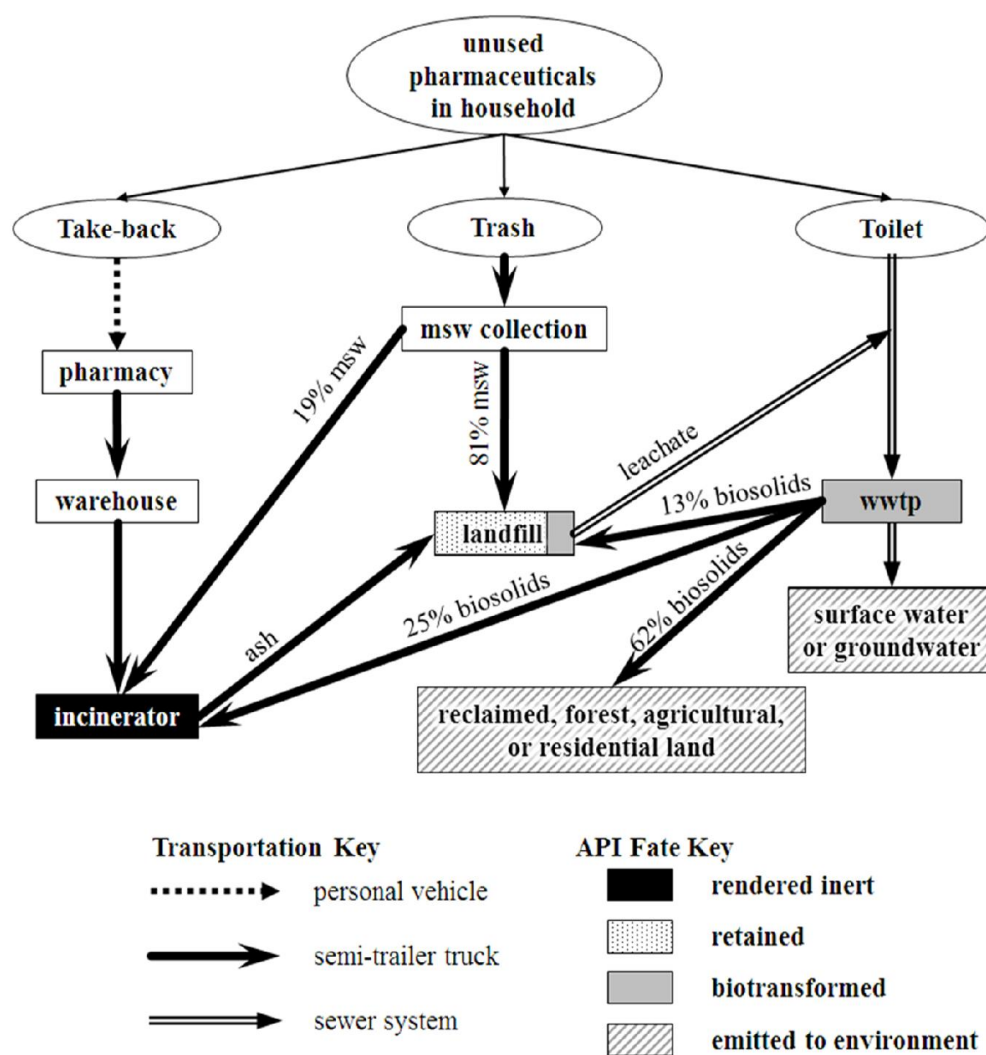


**Figure 2. The waste management systems used by each disposal option, including the transportation and fate of APIs. Abbreviations: msw=municipal solid waste; wwtp=domestic wastewater treatment plant.**

## 3.2  Methods

The three disposal options for unused pharmaceuticals are evaluated using a comparative life cycle assessment following the ISO 14040 framework.[41] Figure 2 lists the major steps for the three disposal options considered in this study. The functional unit is the disposal of an annually accrued mass of unused pharmaceuticals and associated packaging from U.S. households, where it is assumed that the unused pharmaceutical mass is evenly distributed across U.S. households.

The annual mass of unused pharmaceuticals is estimated as 90 million kilograms (200 million pounds).[209] This mass is represented by the ten most commonly returned pharmaceuticals during a take-back pilot program.[69] Each of these pharmaceuticals is assumed to have the same disposal mass. Since two of the ten most commonly returned pharmaceuticals contained acetaminophen as their API, the mass of acetaminophen is estimated at 18 million kilograms while the other eight representative APIs are estimated at 9 million kilograms each. Associated packaging waste is assumed to be 14 million kilograms, based on masses collected during a take-back pilot program.[67] Composition of the packaging material is based on general U.S. packaging waste[210] and pharmaceutical specific packaging.[130] Brief descriptions of data sources, calculations, and assumptions are provided below. Complete information is available in Appendix A and at http://deepblue.lib.umich.edu/ ("Unused Pharmaceutical Disposal").

### 3.2.1  Emissions Categories

In this analysis, ten environmental emissions categories are considered, with nine categories for non-API emissions and one category for emissions of APIs. Non-API emissions arise from the production, use, and/or disposal of materials, fuels, electricity, and infrastructure used in each disposal option. The production of pharmaceuticals is common across all disposal routes and is not included within the LCA system boundary. Non-API emissions are estimated using the Ecoinvent,[124] Franklin USA 1998,[125] and U.S. Life Cycle Inventory (LCI)[126] databases, with adjustments made to reflect typical U.S. transportation and energy system characteristics by substituting U.S. data for European data. The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI)[127] is used to aggregate the hundreds of non-API emissions into the nine non-API emissions categories, based on emission equivalence calculations (e.g., global warming potential in equivalent $CO_2$ emissions).

The fate of APIs in this study include: incineration; retention in a landfill by sorption; biotransformation by aerobic or anaerobic processes within a wastewater treatment plant (WWTP) or landfill; or emission to the environment with WWTP effluent or land applied biosolids. Other possible API fates are assumed to be negligible. For example, the direct emission of APIs from a landfill due to leakage of leachate is assumed to be insignificant based on observed leachate collection efficiencies.[211,212] The incineration process is also assumed to be effective, so the resulting ash does not contain a significant amount of APIs. Therefore, the total API mass emitted to the environment results from the wastewater treatment of flushed pharmaceuticals (toilet disposal) and the wastewater treatment of landfill leachate containing APIs (trash disposal).

Due to uncertainty regarding biotransformation and sorption, an uncertainty analysis is performed with wide parameter ranges, shown in Table 4, to encompass the lack of precise data regarding biotransformation, desorption, sorption equilibrium, and heterogeneity of biomass and MSW. Since knowledge and data about transformation products is not currently available,[50] all biotransformed APIs are assumed to be either: (i) oxidized or assimilated into biomass under aerobic conditions, or (ii) fully converted into landfill gas (e.g., methane, carbon dioxide) in anaerobic environments. Other than these, if the products of biotransformed APIs are shown to be a significant health or environmental hazard, future research should be incorporated within the framework of this study.

**Table 4. Fates of each representative API in a WWTP and a landfill are summarized with the maximum (max) and minimum (min) values used to estimate mass percent biotransformation and sorption. Min and max value calculations are in Appendix A ( page 80).**

| Compound | WWTP Overall % Removal | % Sorption | | % Biotransformation | | Landfill % Biotransformation | | % Sorption | |
|---|---|---|---|---|---|---|---|---|---|
| | (reference) | min | max | min | max | min | max | min | max |
| Acetaminophen | 95% ([93]) | 0% | 82% | 0% | 100% | 0% | 55% | 20% | 100% |
| Aspirin | 86% ([92,93]) | 61% | 100% | 0% | 39% | 0% | 0% | 75% | 100% |
| Vitamin E | 100% ([94]) | 75% | 100% | 0% | 25% | 0% | 0% | 75% | 100% |
| Prednisone | 96% ([93,96]) | 51% | 100% | 0% | 49% | 0% | 9% | 66% | 100% |
| Ibuprofen | 90% ([93,97]) | 65% | 100% | 0% | 35% | 0% | 0% | 75% | 100% |
| Warfarin | 80% ([95]) | 55% | 100% | 0% | 45% | 0% | 0% | 75% | 100% |
| Topiramate | 15% ([98]) | 0% | 40% | 0% | 40% | 0% | 0% | 75% | 100% |
| Etodolac | 45% ([98]) | 20% | 70% | 0% | 50% | 0% | 0% | 75% | 100% |
| Gabapentin | 99% ([93]) | 0% | 25% | 49% | 100% | 0% | 99% | 0% | 100% |

### 3.2.2  Trash

In the trash disposal option, participants mix their pharmaceuticals with a waste item (e.g., coffee grounds) inside a low-density polyethylene (LDPE) bag and remove the labeling from the associated packaging as suggested in the 2007 ONDCP statement.[18] Since our model assumes all trash participants follow the ONDCP recommendation, the LDPE bag and packaging are placed in household trash and no pharmaceuticals are placed in a recycling or composting bin. The production of the LDPE bag (but not the waste item, such as coffee grounds) is included in the LCI. Garbage trucks collect this trash with MSW and, according to current U.S. MSW management, haul 19% to an incinerator and 81% to a landfill.[210]

Incineration of the plastic bag and contents produces air emissions and ash. Air emissions and the composition of ash resulting from the plastic bag and packaging are derived from LCI data. Incineration is assumed to completely oxidize APIs, and the resulting air emissions are based on each representative API's chemical formula as well as treatment of the resulting flue gas with currently available technologies.[124,213,214] Energy recovery from incineration is included in the model by offsetting electricity production. The amount of electricity offset is based on an assumed energy density of 0.535 MWh/ton combusted.[215]

The API fates in a landfill include biotransformation, sorption, or removal with the leachate. All bags are expected to lose their structure due to landfill activities (e.g., mechanical compaction) and release the enclosed APIs. Emissions arising from landfilling the plastic bag and pharmaceutical packaging are estimated with LCI data. Emissions from APIs are calculated using API properties to determine leachate and landfill gas (LFG) compositions and production quantities. The anaerobic biotransformation potential in a landfill for APIs is estimated as a fraction of the aerobic biotransformation potential (in the WWTP). This ratio is defined with an uncertainty parameter ($N_{LF/WWTP}$). The resulting LFG is assumed to be 50% methane, consistent with typical compositions.[216] According to the U.S. average, LFG is either flared to reduce harmful gaseous emissions (28%), combusted to generate electricity (31%), or directly emitted to the atmosphere (41%).[217] The model system boundary includes displaced electricity production due to LFG-based energy recovery.

Within the landfill, the mass of APIs that do not biotransform either sorb to MSW or enter the leachate. Sorption of APIs to MSW in a landfill is estimated using MSW generation[210] and decay[212,216] rates, leachate generation rates,[218] and one-parameter linear free energy relationships (op-LFERs).[219] An op-LFER linearly correlates an API's octanol/water partition coefficient with its MSW/leachate partition coefficient.[219,220] Leachate is sent to a WWTP for treatment according to current U.S. landfill practices.[212] APIs in this leachate have the same fate as APIs disposed of by the toilet disposal method.

### 3.2.3 Toilet

In the toilet disposal option, participants flush pharmaceuticals down the toilet and place associated packaging in the trash. It is assumed that each person disposes of unused APIs in a sole purpose flush, at most once a month and at least once a year. Electricity required to treat this wastewater is estimated by assuming typical energy values for domestic wastewater.[163] The values used in the model (530-1100 kWh/million gallons) include only non-aeration energy demands since the oxygen demand of the wastewater from flushing APIs is significantly lower than domestic wastewater.

Flushed APIs undergo aerobic, conventional activated sludge treatment. These APIs are removed from the wastewater's liquid-phase by sorption, removed by biotransformation, or discharged with the effluent. The total mass percentage removed from the WWTP effluent, by sorption and biotransformation, is based on each representative API's observed removal during aerobic wastewater treatment (see Table 4).[92–98] The quantity of APIs sorbed is determined using typical mixed liquor concentrations[163] and op-LFERs, which linearly correlate an API's octanol/water partition coefficient with its biomass/wastewater partition coefficient.[58,107] The quantity of APIs biotransformed is the difference of the total expected removal and calculated sorption. Heterotrophic yields, from zero to typical observed values,[163] are used to estimate the amount of biotransformed APIs assimilated by biomass.

The biomass formed by API assimilation and the sorbed APIs undergo solids management. It is assumed APIs remain sorbed during sludge stabilization,[45] so sorbed APIs are transported with biosolids from the WWTP to three disposal locations:[221] a landfill (13%), an incinerator (25%), or a land application site (62%). All APIs sorbed to biosolids are also expected to sorb in the landfill,

which has a much higher solids concentration. The incineration of biosolids renders the sorbed APIs inert. Land applied APIs are considered an emission to the environment. Land application sites for biosolids include agricultural lands, forests, reclaimed areas, and residential lands.[221]

### 3.2.4  Take-back

For take-back, participants drive a personal vehicle from their residence to the closest pharmacy to return unused pharmaceuticals and packaging. The nine U.S. rurality categories defined by the U.S. Office of Management and Budget[222] are used to estimate personal driving distances. Take-back participants are proportionally assigned to one of the nine rurality categories based on the geographic distribution of the U.S. population.[223] Three cities were selected to represent metropolitan counties with populations of: more than 1 million (Chicago, IL); 1 million to 250,000 (Bridgeport, CT); and less than 250,000 (Bay City, MI). Six cities were selected to represent non-metropolitan counties, both adjacent to and not adjacent to a metropolitan area, with populations of: more than 20,000 (Georgetown, DE and Coquille, OR); 20,000-2,500 (Abbeville, SC and Baileyboro, TX); and less than 2,500 (Woodruff, UT and Plankinton, SD). For each of these representative cities, Google Maps™ was used to estimate the distances from 50 random addresses to their closest pharmacies. Additional factors considered when calculating personal driving emissions include the type of personal vehicle, the number of return trips a participant makes in one year, how many of these trips are combined with other errands, and what percentage of a combined trip's miles are allocated to taking back unused pharmaceuticals.

Once at the pharmacy, the unused pharmaceuticals and packaging are placed in a collection bin, which is a steel barrel with a removable, cardboard box liner.[66–68,224] Full cardboard boxes are transported to a warehouse for secure storage. Once enough boxes accumulate to completely fill a truck, they are hauled to the closest of the 22 commercial hazardous waste incinerators[225] for destruction. The resulting ash is hauled to the closest of the 21 commercial hazardous waste landfills in the U.S.[226]

### 3.2.5  Uncertainty and Sensitivity Analysis

A Monte Carlo analysis was undertaken to estimate the aggregate impact of parameter uncertainty on the ten emissions categories for each disposal option. Table 14 in Appendix A lists the 62 uncertainty parameters, which represent energy consumption values, energy sources, API fate, API

chemistry, driving behaviors, and vehicle fuel economy. Each uncertainty parameter has a maximum and minimum value assigned from the literature or determined based on what is physically possible. Parameter ranges are characterized with a uniform distribution given the absence of data to justify assigning any other type of probability distribution (including the absence of mean values). In addition, the ranges are selected to be as broad and conservative as possible so that observed differences in emissions between disposal options are likely to be real. The uncertainty ranges for emissions associated with each disposal option are calculated from 100,000 Monte Carlo simulations. A sensitivity analysis was also conducted with the Monte Carlo simulations to determine the sensitivity of each emissions category to each uncertainty parameter. An emissions category was defined as "sensitive" to an uncertainty parameter if the resulting correlation coefficient is greater than +0.8 or less than -0.8.

### 3.2.6 Data Presentation

Emissions data for each disposal scenario are presented in decimal fraction (as emission factors) relative to the emissions of a baseline scenario reflecting current disposal practices for unused pharmaceuticals. Specifically, the baseline scenario for the U.S.[19] is taken as 40% toilet disposal and 60% trash disposal. The emission factor is greater/less than 1.0 for each emission value above/below the baseline. All results presented are the mean values from the Monte Carlo simulations and include an uncertainty range defined by the 25th and 75th percentile values.

## 3.3 Results and Discussion

Figure 3 compares emissions for 100% trash, toilet, and take-back disposal compared with the baseline scenario. It is observed that implementation of take-back programs can eliminate API emissions but results in a significant increase in all non-API emissions. It is also observed that trash disposal can significantly reduce API emissions relative to the baseline without significantly increasing non-API emissions. The Monte Carlo analysis shows that these observations hold when considering the uncertainty in the model parameters.
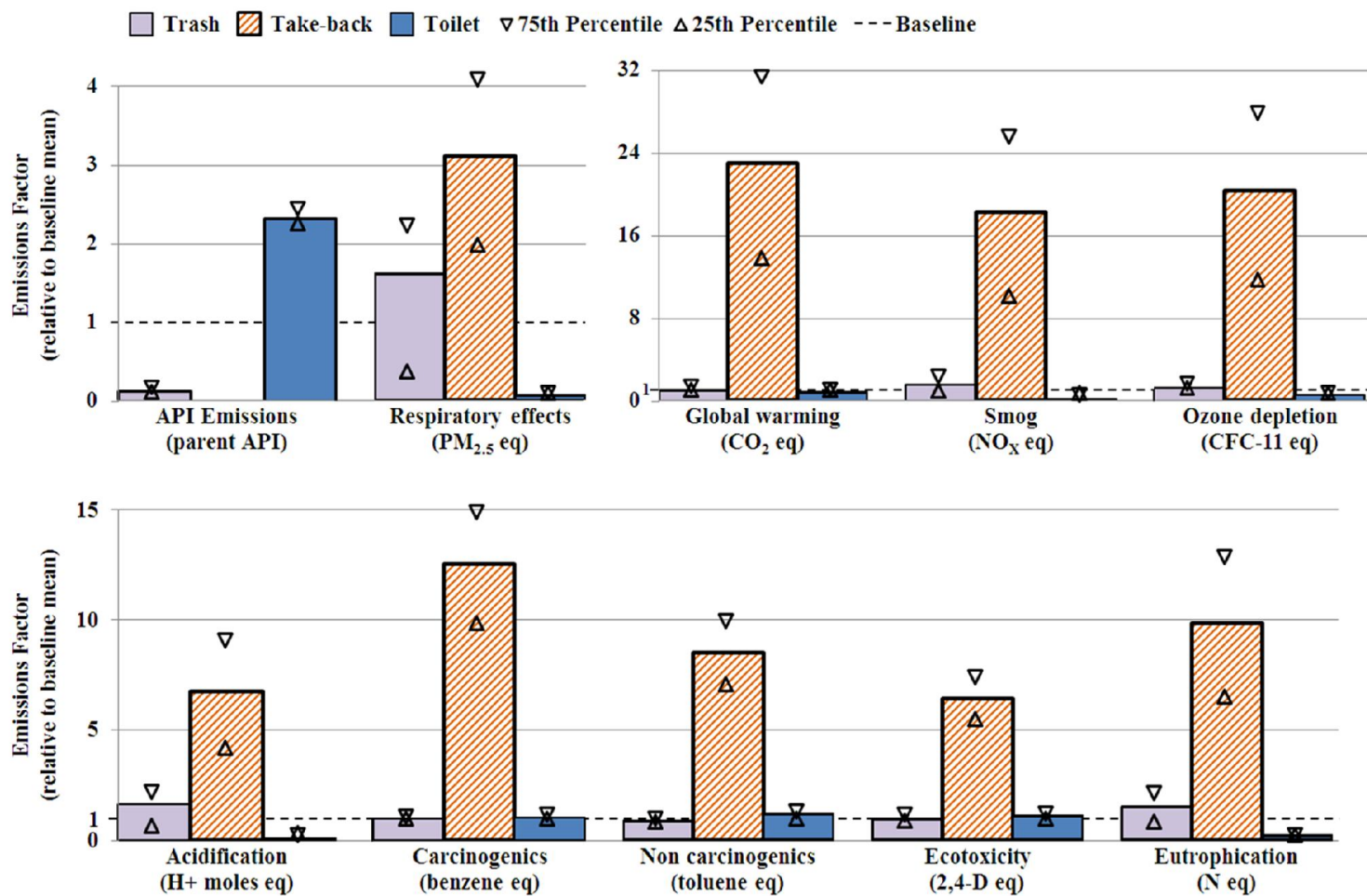
**Figure 3. Results for all 10 emissions categories are presented for 100% participation in each disposal option. Bars are mean values and triangles represent the 75th and 25th percentiles from the Monte Carlo simulations; all are relative to the baseline scenario (40% toilet, 60% trash). Table 13 (in Appendix A) lists the values for the mean and percentile (10, 25, 50, 75, and 90) values.**

### 3.3.1 Toilet

Figure 3 shows that flushing all unused pharmaceuticals down the toilet will more than double API emissions relative to the baseline. It also shows that emissions of carcinogens, non-carcinogens, and ecotoxins increase, though slightly. This is mostly due to the increase of biosolids incineration compared to the baseline. Because the 100% toilet scenario does not require garbage truck transport and only requires the hauling of wastewater treatment byproducts (e.g., biosolids), emissions of eutrophication, smog, and respiratory impacting substances are significantly less than the baseline. Toilet disposal requires about 15 gigawatt-hours per kilogram of unused pharmaceutical (GWh/kg API), which is 8% less energy than the baseline scenario (16 GWh/kg API). Overall, toilet disposal has the highest API emissions, the lowest non-API emissions, and the lowest energy intensity.

### 3.3.2 Trash

The fate estimates for landfilled APIs suggest that most APIs sorb to MSW and are therefore retained in a landfill. As a result, 100% trash disposal reduces API emissions relative to the baseline by 85% - 92%, with a mean reduction of 88%. Given that the landfill model generally used high leachate and MSW decay rates, the 88% API reduction result is likely to be a conservative estimate for 100% trash disposal. Much higher retention of APIs in landfills is possible, especially in arid regions that have low leachate generation rates.[216]

Trash disposal slightly decreases emissions of carcinogens, non-carcinogens, and ecotoxins relative to the baseline. This is due to the decrease in WWTP biosolids production and incineration as well as the electricity offset by recovering energy from 31% of the LFG produced at the landfill. Trash disposal increases the emission of greenhouse gases and ozone depleting substances relative to the baseline by 10% and 30%, respectively. Also, emissions in the categories of acidification, eutrophication, respiratory effects, and smog all increase by at least 50% relative to the baseline. Most of these emissions are from the collection and incineration of MSW as well as from LFG, specifically the 69% that is flared or directly emitted to the atmosphere. The total energy required by trash disposal is 17 GWh/kg unused pharmaceutical, resulting in a 5% increase from the baseline scenario's energy intensity.

### 3.3.3  Take-back

Hazardous waste incineration and then landfilling of unused pharmaceuticals is expected to eliminate all associated API emissions to the environment. Figure 3 also indicates that all non-API emissions increase by more than 200% relative to the baseline. Emissions of carcinogens, non-carcinogens, and substances with eutrophication potential increase by more than 700%. Emissions of global warming, ozone depleting, and smog forming compounds increase by more than 1700%. The increase in global warming compounds represents 1.1-2.6 million metric tons of $CO_2$ equivalent emissions, which is roughly equivalent to the annual carbon dioxide emissions from a 100-300 MW coal-fired power plant.[227] Take-back disposal requires 106 GWh/kg of unused pharmaceutical, which is 560% of the energy demanded by the baseline scenario.

Further work was undertaken to understand how modifications to take-back disposal procedures affect the overall comparison among the three disposal options. One observation from the data is that most of take-back's non-API emissions arise from personal driving (it contributes more than 65% of the emissions in six non-API emissions categories). Also, 72% of the personal driving miles are in rural areas. Therefore, a "best-case" take-back disposal scenario was created that minimizes personal driving by making take-back available only in metropolitan areas (83% national take-back disposal) and assuming half of the metropolitan residents walk to a pharmacy instead of driving (41.5% national take-back disposal via walking). The unused pharmaceuticals located in rural areas are disposed of according to current disposal trends, which translate into 7% national toilet disposal and 10% national trash disposal. In this scenario, API emissions are reduced by 83% relative to the baseline. This "best-case" take-back disposal scenario has more API emissions than the 100% trash scenario (88% API emissions reduction) while still leading to significant increases in non-API emissions relative to the baseline and the 100% trash scenario. In other words, a strategic implementation of take-back programs in metropolitan areas would likely have both higher API and higher non-API emissions than 100% participation in a nationwide trash disposal program.

### 3.3.4  Participation Rates

The analyses with 100% participation show, relative to the baseline, that take-back achieves 100% reduction of API emissions at 106 GWh/kg API, and trash achieves an 88% reduction at 17 GWh/kg API. Comparing these results it can be asked: 1) whether the additional API reduction of

12% is worth the increase in non-API emissions and financial costs of implementing take-back programs, and 2) whether this difference in API emissions is maintained when considering foreseeable participation rates for each disposal option.

With respect to the second question, two U.S.-based surveys suggested that 74%-90% of respondents would likely participate in a take-back program.[66,86] However, a pilot take-back program in Clark County, Washington yielded a participation rate of less than 0.1%.[68] Similarly, after 40 years of a national take-back effort in Sweden, the participation rate has been estimated at only 43%.[85] In addition, it has been reported that more than 50% of people in Sweden store unused pharmaceuticals at home for prolonged periods,[85] which runs counter to the ONDCP goal of safe and fast disposal of unused pharmaceuticals.

To understand the effect of foreseeable participation rates on emissions, a disposal scenario was created where 43% of U.S. households participate in take-back programs and the remaining 57% use trash disposal. Under this scenario, analogous to Swedish participation rates, API emissions are reduced by 93% while all non-API emissions increase by 130-950% relative to the baseline. Under the unlikely case where 43% of unused pharmaceuticals are taken back without any personal driving, the 93% reduction in baseline API emissions still increases all non-API emissions by 30-270% relative to the baseline. Figure 4 summarizes these results and shows, using global warming emissions as an illustrative example, that a take-back program achieving greater reductions of API emissions than a 100% trash disposal scenario will have higher non-API emissions.
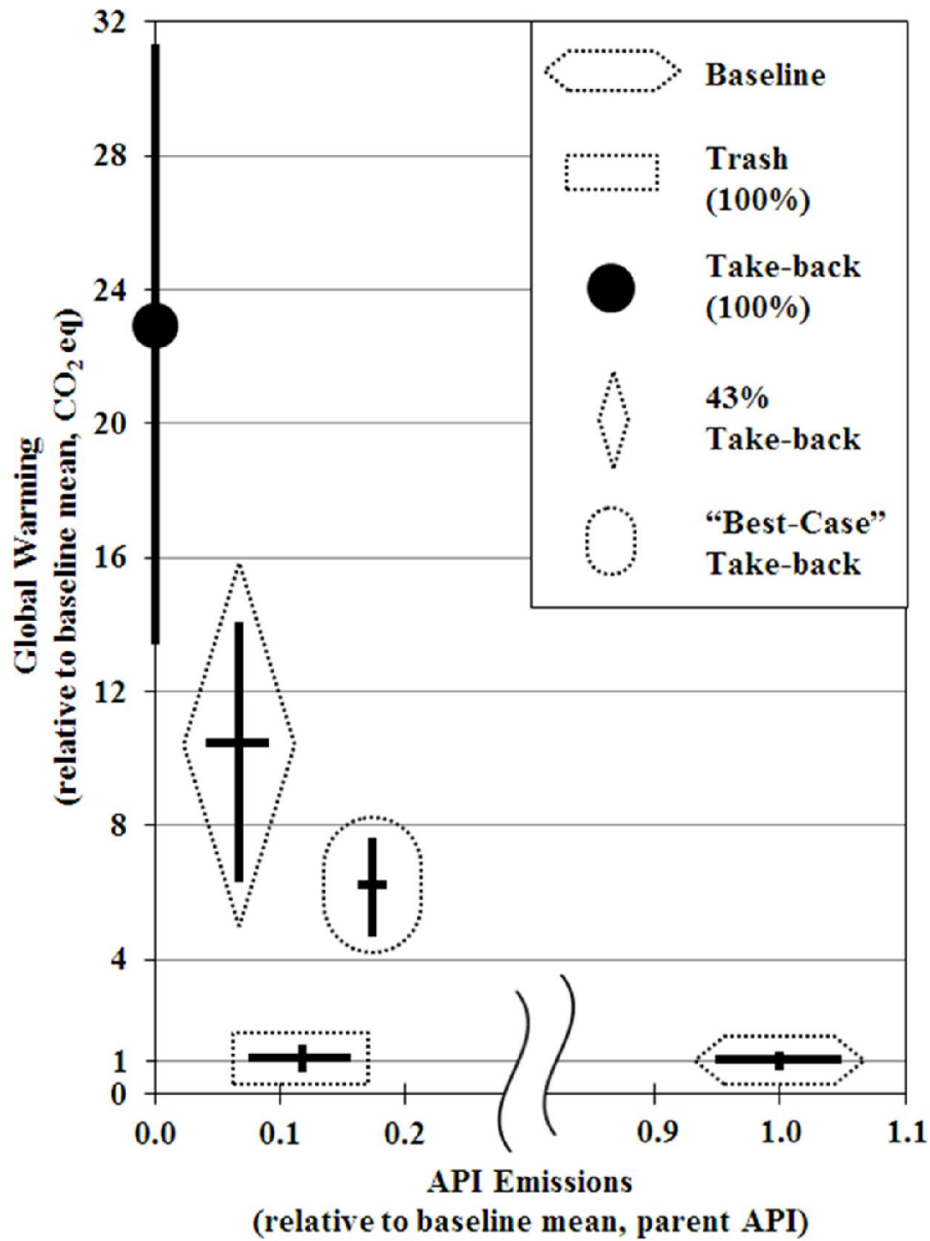
**Figure 4. Results are presented for each scenario's global warming emissions as a function of their API emissions. The uncertainty lines (dark solid lines) show the range of values between the 25th and 75th percentiles. Shapes designate the following disposal scenarios: Baseline (40% toilet, 60% trash); 100% Trash; 100% Take-back; 43% Take-back (43% take-back, 57% trash); and "Best-Case" Take-back (take-back only in metropolitan area with half not driving, 10% trash, 7% toilet).**

### 3.3.5 Disposal Recommendations

If the contribution of environmental API emissions from the disposal of unused pharmaceuticals is determined to be negligible relative to the API contributions from human and animal excretion, then toilet disposal would be the best approach. Toilet disposal has the fewest non-API emissions and is the easiest and fastest way for individuals to remove pharmaceuticals from their homes. Toilet disposal would also be viable if technology could be easily and inexpensively added to all WWTPs to render APIs and their transformation products harmless.

If it is decided that all sources of environmental API emissions should be reduced immediately, then take-back or trash disposal is necessary. Under a take-back participation rate of 43%, take-back reduces 93% of API emissions relative to the baseline (assuming the other 57% is trashed). Trashing all unused pharmaceuticals can reduce baseline API emissions by 88%. Although the implementation of take-back programs might achieve a 5% improvement in API reduction compared to trash disposal under Swedish participation rates, it would come with significant downsides: 1) non-API emissions would increase significantly, 2) societal costs would increase significantly (estimated at 2 billion dollars per year for a nationwide program[87]), 3) disposal inconvenience would increase significantly,[67,86] and 4) home storage of unused pharmaceuticals may increase to an unacceptable level (as in Sweden) due to disposal inconvenience, which increases the risk of poisoning, abuse, and addiction.[55]

Taken together, the results indicate that trash disposal would be nearly as effective in reducing environmental API emissions as take-back programs but without significant increases in non-API emissions or societal costs. Furthermore, since 60% of individuals in the U.S. already trash their unused pharmaceuticals, trash disposal is likely to accomplish faster removal of unused pharmaceuticals from households due to higher participation rates and greater convenience.

A re-evaluation of disposal options for unused pharmaceuticals may become necessary as future waste management strategies, technologies, and research improve our understanding of environmental API sources and impacts. In the meantime, the management of unused pharmaceuticals with MSW can provide a disposal option that is likely to have a high level of compliance, lower costs, and ultimately a similar degree of API removal as compared with take-back programs.

## 3.4 Acknowledgement

Supporting Information: More information about data, assumptions, calculations, and uncertainty parameters are available in the SI. This information is available free of charge via the Internet at http://pubs.acs.org and in Appendix A.

# Chapter 4.
# Evaluations of the Stability and Resource Recovery Potential of Anaerobic Codigestion

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## 4.1  Introduction

Anaerobic digestion is a well-established technology that provides an opportunity for energy and nutrient recovery from organic wastes. Additional benefits of anaerobic digestion can be realized by employing codigestion, which is the simultaneous digestion of two or more types of organic wastes. Benefits of codigestion include diverting wastes from landfills[36], improving a digester's nutrient balance[136], increasing methane production[38], and improving system economics[25,153]. Despite these benefits, codigestion introduces the risk of inhibition[155,156], poor effluent quality[138], digester foaming[157,158], and potentially process failure in cases when a waste has not been well characterized.

To maximize the benefits of codigestion without destabilizing a digester, the operating bounds of codigestion that correspond to stable operation need to be established and then translated into influent composition recommendations. Several researchers have operated codigestion systems in batch, semi-continuous, and continuous mode to evaluate and identify suitable mixtures of organic substrates. Batch assays are commonly used to quickly determine the biochemical methane potential (BMP) of various wastes and waste combinations[29–33]. However, BMP data provide only limited insight into overall digester performance and stability[30] since they are derived from short-term batch experiments. Compared to semi-continuous or continuous reactor experiments, these batch studies cannot account for acclimation or shock loads. Also, they have the potential to result

in nutrient limitations or product accumulation and inhibition due to the reactor configuration. The most effective way used so far to determine the impact of codigestion on process stability has been to evaluate digester performance with continuously or semi-continuously fed reactors. Examples include the codigestion of the organic fraction of municipal solid waste with food industry fats[34], codigestion of domestic wastewater treatment plant (WWTP) sludge with potato processing waste[35], codigestion of WWTP sludge with the organic fraction of municipal solid waste [36], codigestion of agricultural wastes with food industry wastes[35], and codigestion of cattle slurry with fruit and vegetable wastes[37]. All of these studies, as well as many others in the literature, have identified wastes and waste ratios that can result in enhanced methane production and stable digester operation. However, these types of evaluations can be resource intensive (e.g., they can require large amounts of time, money, laboratory equipment and space), and therefore they also tend to provide limited insight into performance optimization. These limitations exemplify why it is not feasible to experimentally test all possible waste combinations that might be considered for codigestion and thus motivate a model-based approach that systematically evaluates the impact of different waste combinations on process stability and performance before an experimental validation phase is considered.

Modeling can be used to reduce the number of experimental conditions that need to be tested to identify the most energy beneficial and stable codigestion waste options. Fortunately, recent advances have been made in anaerobic digestion modeling and enhance the opportunity to conduct such an analysis. However, to date, modeling studies that aimed to select waste combinations for codigestion only focused on maximizing methane production and did not assess process stability. For example, Zaher et al.[38] used the Anaerobic Digestion Model No. 1 (ADM1)[132] to determine the influent blend of manure and kitchen waste needed for maximum methane production. Alvarez et al.[39] developed an optimization protocol that used linear programming and substrate biodegradation potentials to select waste blends of manure, fish, and biodiesel waste that resulted in maximum methane production. When experimentally verifying the optimum blends, they found that their optimization protocol did not select methane-maximizing influent blends unless nutrient limitation was considered. This example highlights that modeling approaches without stability considerations provide limited insight for codigestion operation.

To build on previous modeling efforts and address the need for stability predictions, this chapter aims to elucidate the relationship between influent composition and process stability during codigestion. To achieve this, digester performance was simulated for a range of possible waste combinations using ADM1. Then, the resulting process stability was estimated from this simulated digester performance and steady state model outputs. To estimate process stability, this study developed a comprehensive stability assessment by quantitatively defining the stable boundaries of steady state digester operation using multiple indicators of process stability and failure. This modeling approach and stability assessment were applied to two codigestion scenarios: (i) the codigestion of any possible waste combination, where general and wide ranging influent composition sets were employed, and (ii) the codigestion of WWTP sludge with various organic wastes, where influent compositions were a representative mixture of sludge with waste streams of wide organic waste compositions. Finally, the simulated codigestion performances were evaluated using the stability assessment to identify influent characteristics that could result in stable codigestion.

## 4.2  Model and Stability Assessment Framework

Figure 5 provides an overview of the modeling approach and stability assessment developed to identify operating conditions that could result in stable codigestion systems. First, digester performance is simulated for a wide range of influent compositions. The modeled influent compositions were chosen to have different total organic loadings and relative contributions from four primary waste components: carbohydrates, proteins, lipids, and inerts. All waste streams can be represented in terms of these components, and the different influent compositions can ultimately be translated into different types of wastes and waste combinations. For each simulated influent composition, the digester steady state performance and model outputs were then evaluated to assess the digester's degree of stability. All simulations used a common digester configuration. The simulated digester was a mesophilic (35°C), continuously-stirred tank reactor (CSTR) operated with a 20-day retention time and fed a constant influent, which did not change in composition or loading over time. The steady state model outputs resulting from each influent composition were evaluated to quantitatively assess process stability by considering (i) methane production and (ii) the inhibition of the anaerobic metabolic pathways.
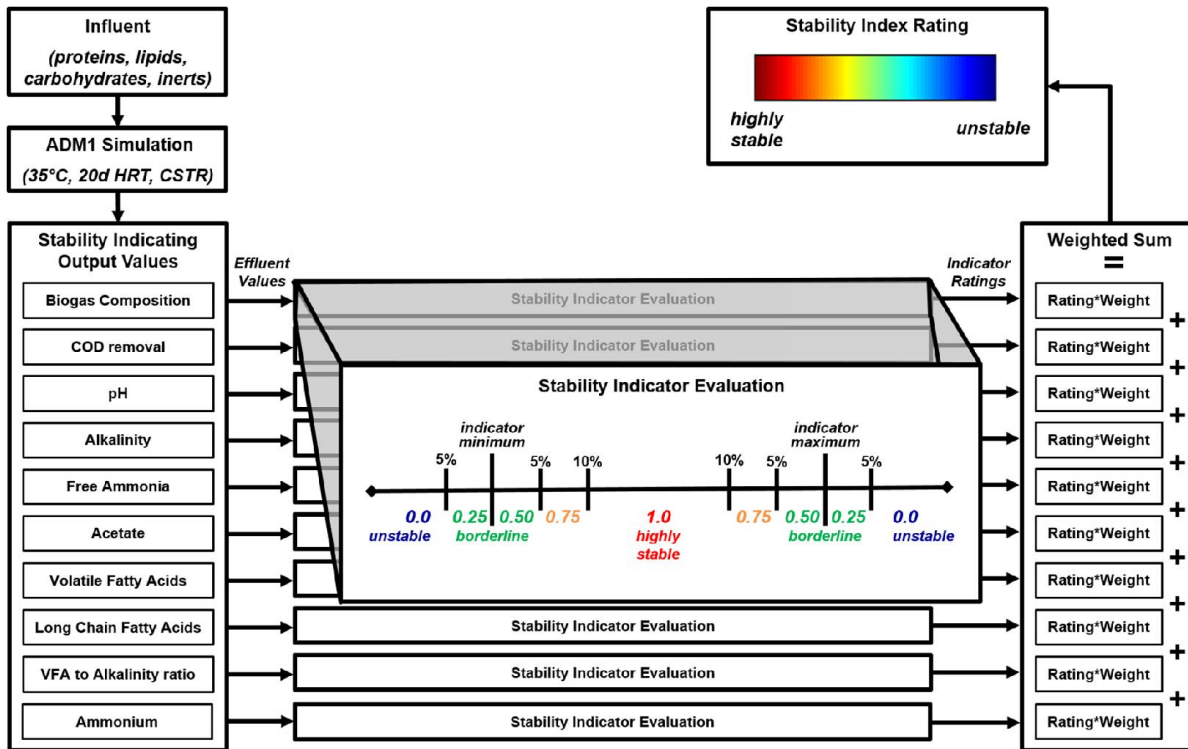
**Figure 5. Overview of stability index and modeling approach used to elucidate the relationship between codigestion influent composition and process stability.**

The quantitative assessment developed herein combines information about the stability of the various microbial and biochemical steps in anaerobic digestion to assess overall process stability. Digester performance metrics (e.g., methane production) and concentrations of chemical intermediates (e.g., acetate) that indicate a digester's metabolic status were used as stability indicators. Table 5 lists the ten stability indicators and their range of values that are expected during stable digestion, as reported in the literature, used in this study. For each simulation, the model outputs were compared to the range of stable values for each of the stability indicators in order to obtain 10 stability indicator ratings. Overall process stability was then represented by consolidating these 10 stability indicator ratings into a single metric using a weighted average. This average value is the stability index rating, which ranges from a value of zero (unstable) to one (highly stable).

## 4.3 Methods

### 4.3.1 Anaerobic Digestion Simulation

#### 4.3.1.1 Influent Compositions

A comprehensive influent space that represents a wide range of potential waste combinations was created by generating 10,000 unique influent compositions that consisted of four primary waste components: carbohydrates, proteins, lipids, and inerts. To help verify stability predictions and explore the potential of uncommon influent compositions, influents that would likely result in both stable and unstable digestion were modeled. Specifically, the 10,000 influents had total organic loading rates (OLRs), reported in units of kilogram chemical oxygen demand (COD) per cubic meter of digester volume per day (kg COD/$m^3$/d), that ranged between 1 and 11 kg COD/$m^3$/d. The upper bound was selected to be slightly greater than an OLR of 9 kg COD/$m^3$/d, above which digester failure is expected[151,173]. This total OLR consisted of separate influent OLRs for carbohydrates, lipids, proteins, and inert compounds. Loadings of carbohydrates, lipids, and proteins were each within a range of 0 to 10 kg COD/$m^3$/d. The OLRs of inert compounds corresponded to the average inert concentrations for various wastes[206] and were within a range of 0-20 percent of the total OLR.

#### 4.3.1.2 Anaerobic Digestion Process Model

The digestion process was simulated using the International Water Association's ADM1[132]. The simulation software package MATLAB[TM]/Simulink® was used to implement the differential and algebraic equations of ADM1, as modified by Rosén and Jeppsson[228]. Typical biochemical and physicochemical parameters for mesophilic digestion[132] were used, and the temperature was set at 35ºC. The simulated digester was a CSTR with a retention time of 20 days. For each influent composition, the simulation was run until the aqueous output concentrations reached a steady state. Steady state was defined as a maximum of five percent change over one retention time after a minimum simulation time of five retention times (i.e., 100 days).

#### 4.3.1.3 Uncertainty & Sensitivity Analysis

To account for the uncertainty inherent to modeling diverse influent compositions, uncertainty assessment tools were applied to 35 ADM1 biochemical parameters. Table 22 in Appendix B lists

all of the parameters and expected range of values as found in the literature. These parameters included disintegration rates, hydrolysis rates, biomass yields, decay rates, half saturation constants, maximum specific uptake rates, and inhibition constants. A maximum and minimum value for each parameter was derived from the literature analysis and was used to characterize each parameter's probability distribution. A uniform distribution was selected based on the absence of data and confidence to justify assigning any other type of distribution. To conduct an analysis of the aggregate impact of parameter variations, Monte Carlo analysis was applied using these uniform probability distributions to randomly generate 1,000 parameter sets. All 10,000 influent compositions were simulated with each parameter set (i.e., each influent composition had 1,000 potentially different outputs). For each influent composition, the estimated stability for all 1,000 outputs (one for each parameter set) were compared to evaluate the impact of biochemical parameter uncertainty and to determine which parameters have a strong influence on the stability assessment.

### 4.3.2  Stability Assessment

The goal of the stability assessment was to quantitatively assess a digester's degree of stability. The quantification and consolidation of stability information from several metrics into a single overall metric was inherently subjective and required the use of stability demarcations and weighting factors. Therefore, the impact of different weighting factors were evaluated by comparing the modeled stability trends with those previously observed in full-scale and lab-scale digester studies. The results of this evaluation and assessments of the utility of the stability index is included in Section 4.4.

### 4.3.2.1  Stability Indicators

The most commonly used stability indicators are biogas composition, methane production, pH, and concentrations of alkalinity and volatile fatty acids (VFAs)[159,161]. In addition to these, this study used a few additional stability indicators and defined their range of stable values, or stability bounds, based on data from operational digesters (see Table 5). The stability indicators were used to compare model outputs of steady state effluent concentrations and gas production against values expected for a stable digestion system. Methane is the desired product of anaerobic digestion and therefore methane production represents overall process performance. This information is captured

with stability indicators for biogas methane composition and COD removal. Thus, these two indicators assessed both process stability and performance. For this study, the digestion goal was to convert organic matter into methane. So the stability threshold value for biogas methane content and percent COD removal were selected to reflect this digestion goal as well as to represent stable values as found in the literature. It should be noted that COD was chosen to assess the conversion of organic matter to methane (e.g., instead of volatile solids reduction) since COD is the base unit of ADM1. The digester pH was selected as a stability indicator since it represents an important variable that is strongly linked to the optimal growth and activity of microbial populations.

**Table 5. Stability indicators that were used to calculate stability index ratings.**

| Indicator | Units | Minimum | Maximum | Weight | Citations |
|---|---|---|---|---|---|
| **pH** | log [H$^+$] | 6.1 | 8.3 | *1* | 163,167,168 |
| **Alkalinity** | mg CaCO$_3$/L | 2,000 | 20,000 | *1* | 35,39,138,163 |
| **Free ammonia** | mg NH$_3$-N/L | *n/a* | 150 | *1* | 31,137,163 |
| **Biogas Composition** | Methane Volume/ Biogas Volume (%) | 55 | *n/a* | *1* | 163,167 |
| **COD removal** | 1-(CODout/CODin) (%) | 45 | *n/a* | *1* | 163,169* |
| **Acetate** | mg COD as acetate/L | *n/a* | 850 | *1* | 167,170 |
| **VFA to Alkalinity Ratio** | mg acetate equivalent/ (mg CaCO$_3$ equivalent) | *n/a* | 0.4 | *0.5* | 36,159,229 |
| **Long Chain Fatty Acids** | mg COD as LCFA/L | *n/a* | 1,400 | *0.5* | 171,172 |
| **VFA$_{C2-C5}$** | mg COD as C2-C5 VFAs/L | *n/a* | 3,700 | *0.5* | 167,173 |
| **Ammonium** | mg NH$_4^+$-N/L | *n/a* | 5,000 | *0.5* | 31,163 |

*The minimum COD removal was derived from reported volatile solids reduction values.

VFAs, specifically acetate, butyrate, propionate, and valerate, are important intermediates in the digestion process. Their build-up can signal a breakdown in the anaerobic food web and cause the pH to drop to inhibitory levels. While, acetate and total VFA are traditional stability indicators[161], propionate has also been found to be good indicator[161], especially for monitoring process

recovery[175]. Acetate was selected as a stability indicator since it can be inhibitory[174], it shows a faster response to destabilizing conditions than other VFAs[161], and previous studies have determined stability threshold values[167,170]. Because stability threshold values for the other individual VFAs are lacking in the literature, the sum of the concentrations of butyrate, propionate, and valerate was used to indicate stability. Alkalinity represents the digester's buffering capacity and its ability to respond to the temporary production of acids due to shock loads. In addition, the VFA to alkalinity ratio was selected as a stability indicator since it has been used to successfully monitor digester performance[229]. Finally, the concentrations of inhibitory compounds, such as free ammonia ($NH_3$), ionized ammonia ($NH_4^+$)[31], and long chain fatty acids (LCFAs)[171] were included as stability indicators.

### 4.3.2.2 Stability Indicator Ratings

For each influent, steady state simulation outputs were compared to all the stability indicator threshold values (see Table 5) and assigned a stability indicator rating. Therefore, 10 stability indicator ratings are assigned and associated with each influent composition. Specifically, how close the correspond model output value was to the stability indicator threshold values was accounted for quantitatively using the stability index rating system described in Table 6. The rating system assumes that a digester with effluent concentrations that are in the stable range and are not close to that stability threshold is more stable than a digester that has concentrations close to the maximum of minimum value expected in a stable system. For each stability indicator, if the corresponding model output value was within a stable range, then a stability indicator rating of at least 0.5 was assigned. A higher rating, up to one, was assigned for model output values that were in the stable range but not close to the stability threshold value. Similarly, a lower rating, down to zero, was assigned for model output values that were outside of the stable range. Two example calculations are included in Appendix B (page 94).

**Table 6. For all 10 stability indicating model outputs, the output value was compared to the corresponding stability indicator threshold value (maximum and minimum values, as applicable) and then assigned a stability indicator rating.**

| Model Output Value (X) Compared to Corresponding Stability Indicator Threshold Values | Stability Indicator Rating |
|---|---|
| X < 95% minimum | 0.0 |
| 95% minimum ≤ X < minimum | 0.25 |
| minimum ≤ X < 105% minimum | 0.50 |
| 105% minimum ≤ X < 110% minimum | 0.75 |
| 110% minimum ≤ X ≥ 90% maximum | 1.0** |
| 90% maximum < X ≥ 95% maximum | 0.75 |
| 95% maximum < X ≥ maximum | 0.50 |
| maximum < X ≥ 105% maximum | 0.25 |
| **Special Conditions when a Stability Indicator only has a Maximum or Minimum* | |
| X ≤ 90% (maximum only) | 1.0 |
| X ≥ 110% (minimum only) | 1.0 |

### 4.3.2.3 Stability Index and Stability Index Rating

For each simulated influent composition, the overall process stability was represented by a stability index rating. This stability index rating was calculated by combining all 10 stability indicator ratings into one weighted average (Equation 1). By definition, the stability index rating ranges from a value of one, representing a highly stable digester, to a value of zero, representing a completely unstable (or failed) digester.

$$Stability\ Index\ Rating = \frac{\sum_{i=1}^{10}(Indicator\ Rating)_i \cdot (Indicator\ Weight)_i}{\sum_{i=1}^{10}(Indicator\ Weight)_i} \qquad \text{(Equation 1)}$$

where $i$=Stability Indicator

The ability of each stability indicator to inform overall process stability was quantified with an indicator weight (see Table 5). Due to limited information about the relative importance of each stability indicator to overall stability estimations, weights were assigned based on current knowledge and agreement about stability threshold values and stability indicator usefulness as

reported in the literature. The most widely recognized stability indicators—biogas composition, pH, alkalinity concentration, acetate concentration, and COD removal—are commonly used because they have been proven to be successful at monitoring operation digesters station received[151,159,161]. Therefore, they were assigned the highest weight of one. The free ammonia stability indicator was also given a weight of one since it is a common inhibitory compound with a well-established stability threshold value. Due to limited application or lack of consistent stability threshold values in the literature, the remaining stability indicators received a weight of 0.5. To further represent the importance of each well-established indicator, a digester was assigned an overall stability index rating of zero and assumed to be completely unstable (failed) if more than half of the outputs were outside of the stability bounds for stability indicators with a weight of one.

## 4.4 Results and Discussion

### 4.4.1 The stability index introduces a quantitative approach for comprehensively evaluating overall process stability.

To test the usefulness of the stability assessment developed in this study, data from operational digesters was used to compare observed digester performance with calculated stability index ratings. Four full-scale WWTP sludge digesters[169], a lab-scale reactor codigesting WWTP sludge and potato processing waste[35], and a lab-scale reactor codigesting manure, slaughterhouse waste, and agricultural waste[35] were evaluated. Table 7 shows the measured digester performance data for each operational data. Using this data, stability indicator ratings and a stability index rating were calculated based on each digester's measured data. Comparing the calculated stability index rating to the observed digester performance shows that the stability index can reflect observed digester process stability.

**Table 7. Stability estimations for six stable digesters that were used to evaluate the applicability of the stability index ratings.**

| Digester | Description | Biogas (% methane) | pH | Alkalinity (mg CaCO$_3$/L) | Free ammonia (mg HN$_3$-N/L) | COD removal (%) |
|---|---|---|---|---|---|---|
| 1 | WWTP sludge | 59 | 7.0 | 2515 | 17 | 52 |
| | Indicator rating: | 0.75 | 1.0 | 1.0 | 1.0 | 1.0 |
| | Rating based on: | Direct measurement. | Direct measurement. | Direct measurement. | Nitrogen mass balance.* | Volatile solids reduction (VSR). |
| 2 | WWTP sludge | 60 | 7.1 | 4308 | 53 | 57 |
| | Indicator rating: | 0.75 | 1.0 | 1.0 | 1.0 | 1.0 |
| | Rating based on: | Direct measurement. | Direct measurement. | Direct measurement. | Nitrogen mass balance.* | VSR measurement. |
| 3 | WWTP sludge | 61 | 7.1 | 4439 | 40 | 52 |
| | Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 |
| | Rating based on: | Direct measurement. | Direct measurement. | Direct measurement. | Nitrogen mass balance.* | VSR measurement. |
| 4 | WWTP sludge | 58 | 7.0 | 3526 | 25 | 45 |
| | Indicator rating: | 0.75 | 1.0 | 1.0 | 1.0 | 0.5 |
| | Rating based on: | Direct measurement. | Direct measurement. | Direct measurement. | Nitrogen mass balance.* | VSR measurement. |
| 5 | Manure, slaughterhouse, and agricultural waste | 71 | 7.9 | 19800 | 480-540 | -- |
| | Indicator rating: | 1.0 | 0.75 | 0.50 | 0.0 | 0.75 |
| | Rating based on: | Direct measurement. | Direct measurement. | Direct measurement. | Direct measurement. | Methane gas yield (comparison to theoretical values). |
| 6 | WWTP sludge and potato processing waste | 67 | 7.2 | 2340 | -- | -- |
| | Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 0.75 |
| | Rating based on: | Direct measurement. | Direct measurement. | Direct measurement. | Influent characterization. | Methane gas yield (comparison to literature values). |

**Table 7. Continued.**

| Description | VFA/ Alkalinity (mg acetate/ mg CaCO$_3$) | Long Chain Fatty Acids (mg COD/L) | VFA (2-5 Carbon) (mg COD/L) | Acetate (mg acetate/L) | Ammonium (mg NH$_4^+$/L) | Stability Index Rating |
|---|---|---|---|---|---|---|
| Digester#1 | 0.02 | -- | 48 | 45 | 1615 | |
| Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 0.97 |
| Rating based on: | Direct measurement. | Influent characterization. | Total VFA measurement.** | Total VFA.** | Nitrogen mass balance.* | |
| Digester#2 | 0.01 | -- | 65 | 61 | 5564 | |
| Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 0.0 | 0.91 |
| Rating based on: | Direct measurement. | Influent characterization. | Total VFA measurement.** | Total VFA.** | Nitrogen mass balance.* | |
| Digester#3 | 0.01 | -- | 63 | 59 | 2971 | |
| Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 1.00 |
| Rating based on: | Direct measurement. | Influent characterization. | Total VFA measurement.** | Total VFA.** | Nitrogen mass balance.* | |
| Digester#4 | 0.02 | -- | 69 | 65 | 2450 | |
| Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 0.91 |
| Rating based on: | Direct measurement. | Influent characterization. | Total VFA measurement.** | Total VFA.** | Nitrogen mass balance.* | |
| Digester#5 | 0.1 | -- | 2400 | 2400 | 10746 | |
| Indicator rating: | 1.0 | 0.0 | 1.0 | 0.0 | 0.0 | 0.50 |
| Rating based on: | Direct measurement. | Influent characterization. | Total VFA measurement.** | Total VFA was mostly acetate.** | Free ammonia measurement. | |
| Digester#6 | 0.1 | -- | 213 | 200 | -- | |
| Indicator rating: | 1.0 | 1.0 | 1.0 | 1.0 | 1.0 | 0.97 |
| Rating based on: | Direct measurement. | Influent characterization. | Total VFA measurement.** | Total VFA. | Influent characterization. | |

* All nitrogen removed from liquid wastewater stream was assumed to go to the digester (this was a conservative assumption that estimated the maximum amount of nitrogen; ionization was calculated from the measured pH.

**This parameter was overestimated to be conservative.

A highly stable system receives a stability index rating around one, while an unstable system obtains a smaller rating closer zero. Evaluation of long-term performance data available for the WWTP sludge digesters (Digesters 1-4) and the lab-scale reactor codigesting manure, slaughterhouse waste, and agricultural waste (Digester 6) suggested each was a highly stable digester. The calculated stability index rating for each of these digesters was close to one (Table 7), which shows agreement between the digester performance data and the calculated stability index rating. In addition, the stability index rating was designed to capture the degree of stability, which can be highlighted with the example of Digesters 2, 4, and 5 from Table 7. Digesters 2 and 4 exhibited high stability index ratings (0.91), but the ratings were lower than those for similar digesters (e.g., Digesters 1 and 3 had ratings of 0.97 and 1.0, respectively). Digester 2 had a lower stability index rating because its biogas methane composition was near the minimum expected for a stable system and its nitrogen concentration was close to inhibitory levels. Digester 4 exhibited low stability since its biogas methane composition and percent COD removal were close to the respective minimum stability threshold values. Digester 5 had the lowest stability index rating of the examined digesters with a value of 0.50, suggesting this digester may have been operating close to failure (borderline stability). This lower rating reflects the high pH, acetate, and ammonia-nitrogen concentrations, which are reactor conditions indicative of aceticlastic methanogen inhibition[35]. Similar to the calculated stability index rating, the operators of this digester concluded that there was a process imbalance. Due to the level of information required to quantitatively evaluate digester stability using the stability index rating and the bias in the literature to report on stable operation, we were able to evaluate only five highly stable digesters and one digester operated close to failure. Future research that collects detailed information from unstable and failed digesters will improve the evaluation and understanding of digester stability.

### 4.4.2 A stable influent space can be estimated using ADM1 and the stability index rating.

Figure 6 shows the estimated relationship between influent composition and digester stability based on stability index ratings. The influent compositions that are expected to result in stable digester operation make up a digester's stable influent space. Each modeled influent includes organic loadings from carbohydrates, proteins, lipids, and inerts. Since only a maximum of three influent components can be visualized at once, just three components of the influents are shown

in Figure 6a-b. In order to further evaluate the correlation between the different influent components and overall digester stability, the four influent components were projected onto two axes (Figure 6c-f). Due to this necessary data visualization approach, all relevant data may not be present on each individual plot. For example, Figure 6e shows stable and unstable data points in close proximity since the specific protein loading values for each point are needed to see the trend between process stability and influent composition; the trend is clearer in figures with a protein loading axis (e.g., Figure 6c-d).

The clearest trends between process stability and influent compositions are shown in Figure 6c-d. These plots show a strong relationship between the stability index rating and influent loadings of proteins. For example, they show that the greatest process stability is achievable with protein loadings around 2 kg COD/$m^3$/d (e.g., Figure 6c, section I). Overall, Figure 6 shows that out of the four influent components, proteins have the strongest relationship with stability. This strong relationship was expected since influent nitrogen can strongly influence process stability[35,141,163], and since this trend matches current knowledge about stable digester performance under steady state conditions. Proteins are the only biodegradable compound in the modeled influent that (i) contain the macronutrient nitrogen, which is required for biological growth, and (ii) release free ammonia upon hydrolysis, which is inhibitory to most microbes when present at high concentrations. These two important roles of nitrogen in digester stability are discussed in detail below.
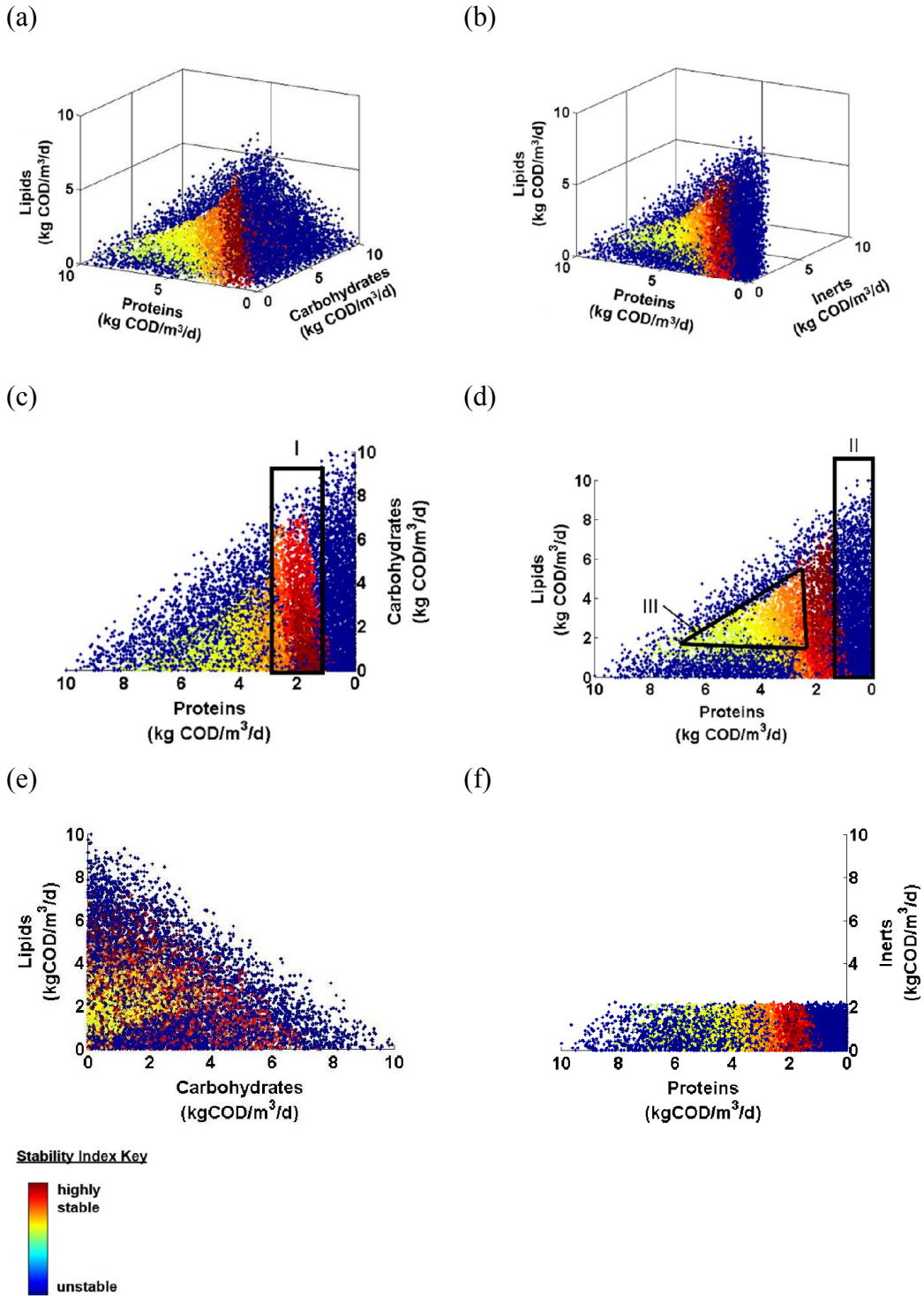
**Figure 6. Stability index ratings calculated for influents consisting of carbohydrate, protein, and lipid loadings each ranging from 0-10 kg COD/m³/d, inert loadings of 0-20% of total influent COD, and total OLRs of 1-11 kg COD/m³/d; (a-b) are 3-dimensional projections and (e-f) are 2-dimensional projections of the 4 dimensional influent compositions.**

#### 4.4.2.1 Minimum Influent Nitrogen Concentration

Figure 6 shows an unstable region for influent compositions with a protein loading less than 1 kg COD /m$^3$/d, which is highlighted with Section II of Figure 6d. This protein loading corresponds to an influent nitrogen concentration of 2.0 g N/L for a retention time of 20 days and assuming 7.0 mole N/kg COD as proteins[132]. The defining characteristic of this low protein loading instability was low total biomass concentrations in the simulated digesters (the mean total biomass concentration for these simulations was approximately 100 mg COD/L, Table 18 in Appendix B). Since the amount of nitrogen needed for growth will depend on the total substrate, the relationship between biodegradable OLRs (i.e., total loading of carbohydrates, lipids, and proteins) and protein loading (representing minimum nitrogen) was analyzed and a linear correlation was found (Figure 16 in Appendix B), as expected. For the influent compositions that resulted in unstable digestion likely due to nitrogen limited growth, a biodegradable OLR around 1.5 kg COD/m$^3$/d had a minimum protein loading requirement greater than 0.75 kg COD/m$^3$/d. A biodegradable OLR around 7 kg COD/m$^3$/d requires more than 1.3 kg COD/m$^3$/d of influent protein. For OLRs in the range of 1 to 8 kg COD/m$^3$/d (corresponding to 20-160 kg COD/m$^3$ for a 20 day retention time), the minimum influent nitrogen concentration required for stable digestion was observed to be from 0.7 to 1.4 kg COD/m$^3$/d protein (corresponding to 2-4 g N/L).

Astals et al.[141] also found similar results with batch experiments that looked at the codigestion of glycerin and pig manure. These experiments found nitrogen-limited biomass growth with an influent nitrogen concentration of 1.0 g N/L, but no biomass growth limitation at influent concentrations of 2.5 g N/L[141]. While the total influent COD concentration was not reported in this study, it was estimated to be within the model range of 20 to 160 kg COD/m$^3$. These results are consistent with this study's modeling results and reveal that an important result of low protein loading is a lack of biomass growth due to nitrogen limitations. Furthermore, they suggest that the minimum influent nitrogen concentration required for digester stability is around 2 g N/L, and the exact value could be greater depending on the total loading rate.

While it is known that a minimum concentration of nitrogen is required for biological growth, the minimum concentration estimated with the ADM1 might be an overestimation for some anaerobic systems. For example, one study observed the stable codigestion of pig slurry and glycerin wastes

with an influent TKN of 1.4 g N/L[138] for a OLR around 3.3 kg COD/m$^3$/d, which is below the minimum influent nitrogen concentration identified with the modeling results. One explanation for this discrepancy is that the ADM1 default parameters overestimated the minimum influent nitrogen concentration. For example, Derbal et al.[143] found that the ADM1 effluent concentration of inorganic nitrogen was underestimated compared to the experimental concentration for the codigestion of waste active sludge and the organic fraction of municipal solid waste. The effluent (and digester) inorganic nitrogen concentration represents the amount of nitrogen in the digester that is available for biomass growth. Therefore, an underestimation of the inorganic nitrogen concentration in the effluent will overestimate the influent nitrogen concentration required for growth. The three main parameters that directly impact predicted digester stability and the inorganic nitrogen calculation are the nitrogen contents of biomass, inert compounds, and proteins. While there are several parameters that influence the inorganic nitrogen concentration (e.g., biomass yield coefficients), these parameters impact many modeling calculations (e.g., biomass concentrations, substrate uptake rates) and the impact on stability predictions is less direct.

An overestimation of the nitrogen incorporated into biomass or of the nitrogen content of inert compounds formed during biomass decay will underestimate inorganic nitrogen. The default parameter values for the ADM1 assume that biomass has 6.1 mole N/kg COD (based on a composition of $C_5H_7O_2N$) and inert compounds have on average of 4.3 mole N/kg COD. While both of these assumptions have been successfully applied to represent wastewater systems (e.g.,[163,168,230] and [132,168,230], respectively), verifying these parameter values for each system modeled may improve the inorganic nitrogen predictions. An underestimation of the average nitrogen content of proteins will underestimate effluent inorganic nitrogen concentration. The ADM1 default protein nitrogen content is 7.0 mole N/kg COD, while other studies have estimated an average protein to have 7.6 mole N/kg COD ($C_{16}H_{24}O_5N_4$)[30] and 7.5 mole N/kg COD ($C_4H_{6.1}O_{1.2}N$)[231]. Comparing these nitrogen contents shows that the default value might be too low and resulting in an underestimation of effluent inorganic nitrogen concentrations.

In addition to the inorganic nitrogen concentration ($S_{IN}$), the inorganic nitrogen half saturation constant ($K_{S,IN}$) is an important modeling parameter for determining nitrogen growth requirements. The ADM1 uses secondary substrate Monod kinetics to describe nitrogen-limited growth with the

following term: $\left(\frac{1}{1+\frac{K_{S,IN}}{S_{IN}}}\right)$. This term results in reduced substrate uptake rates when the concentration of inorganic nitrogen ($S_{IN}$) is less than the half saturation constant ($K_{S,IN}$). This term is included in all kinetic rate equations, except expressions for hydrolysis and decay. Decreasing the ADM1 default value for the inorganic nitrogen half saturation constant will reduce the predicted amount of nitrogen required for growth. Studies that determined or analyzed anaerobic kinetics were reviewed (e.g., [132,195,197,203,228,232–246]), but alternative values of this half saturation constant were not identified. If further evaluation and experimental validation of the stable influent space also show that the minimum nitrogen needed for growth is overestimated, then future research and parameter calibration may be needed.

### 4.4.2.2 Maximum Influent Nitrogen Concentration

The estimated overall stability is impacted by ammonia inhibition when protein loadings are above 3 kg COD/m$^3$/d (about 0.29 kg N/m$^3$/d or 5.9 g N/L for a retention time of 20 days and 7.0 mole N/kg COD as proteins[132]). Section III of Figure 6d shows that a digester is expected to function at these high protein loadings if the protein loading is balanced with the loadings of other non-nitrogenous compounds. However, influents with these high protein loadings (>3 kg COD/m$^3$/d) are expected to be less stable than influents with lower protein loadings (1-3 kg COD/m$^3$/d). The simulation results show that digesters with high protein loadings (>3 kg COD/m$^3$/d) resulted in higher concentrations of free ammonia, an inhibitory compound, and acetate, an intermediate digestion product and inhibitory compound at high concentrations (Table 19 [mean acetate concentration of approximately 30 g/L] and Table 20 [mean NH$_3$-N concentration of approximately 80 mg/L] in Appendix B). Murto et al.[35] found similar results during the operation of three continuously-fed reactors that were codigesting manure, slaughterhouse waste, and agricultural wastes. These reactors had average influent nitrogen loadings around 0.20 kg N/m$^3$/d that were suspected to cause inhibition, as shown by high concentrations of free ammonia and acetate. These experimental observations reasonably match the modeling estimates and suggest that ammonia inhibition plays a strong role in determining digester stability at protein loadings greater than 3 kg COD/m$^3$/d.

The stability modeling approach described in this chapter provides a mechanism for translating inhibitory digester concentrations into useful and specific influent guidelines. Since a digester's influent can be directly controlled, it can be more useful to know the maximum nitrogen loading than the maximum digester concentration of free ammonia. In addition, the microbial community of anaerobic digesters have been found to acclimate to high concentrations of free ammonia[163,247], making the maximum allowable concentration of free ammonia a moving target. To explore the impact of differences in microbial communities, the impact of variable biochemical parameters on predicted digester stability was tested. The results of this uncertainty assessment show that the stability index rating was not sensitive to any one of the varied biochemical parameters (i.e., the stability index rating was not strongly correlated, $|\rho|>=0.8$, with any parameter) for all of the modeled influents. However, the results also show that the stability index rating was sensitive to the entire set of biochemical parameters (i.e., for a given influent, not all parameter sets resulted in the same stability index rating). These results suggest that including more information on microbial community characteristics decreases the uncertainty of stability predictions. In addition to stability insight during steady state operation, information about a digester's microbial community function and structure is needed to fully assess a digester's risk of failure during dynamic operations and response to perturbations. For example, researchers have observed that digestion systems with a history of instability could tolerate organic overload conditions better than digesters without previous instability[248]. Coupling the stability index with microbial community data can improve the prediction of process stability under steady state and dynamic conditions, and it can enable optimal codigestion design for stable systems that maximize resource recovery.

### 4.4.3 Traditional target design values of organic loading rate and carbon to nitrogen ratio guidelines may be too limited for application to codigestion.

The applicability of conventional anaerobic digestion influent design guidelines to codigestion were investigated by comparing the modeled stability to recommended values for OLR and C/N ratio. The traditional OLR design guideline for CSTRs is 1-5 kg COD/m$^3$/d[163] to avoid overload conditions. The C/N ratio design guideline focuses on avoiding ammonia inhibition. Optimal C/N ratios have been determined for different substrate combinations by determining the C/N ratio that achieves maximum methane production and vary from 10:1 to 90:1[137,138]. Figure 7 shows the

modeled influent compositions described by their influent OLRs and C/N ratios. Describing an influent with these two axes highlights a correlation between these combined design guidelines and expected digester stability. This correlation was expected since combining these two metrics provided information about how much substrate (OLR) and what kinds of substrates (C/N ratio) were put in the digester. While using both influent classification approaches together provide useful predictive power, the traditional target values do not align with the predicted digester stability.
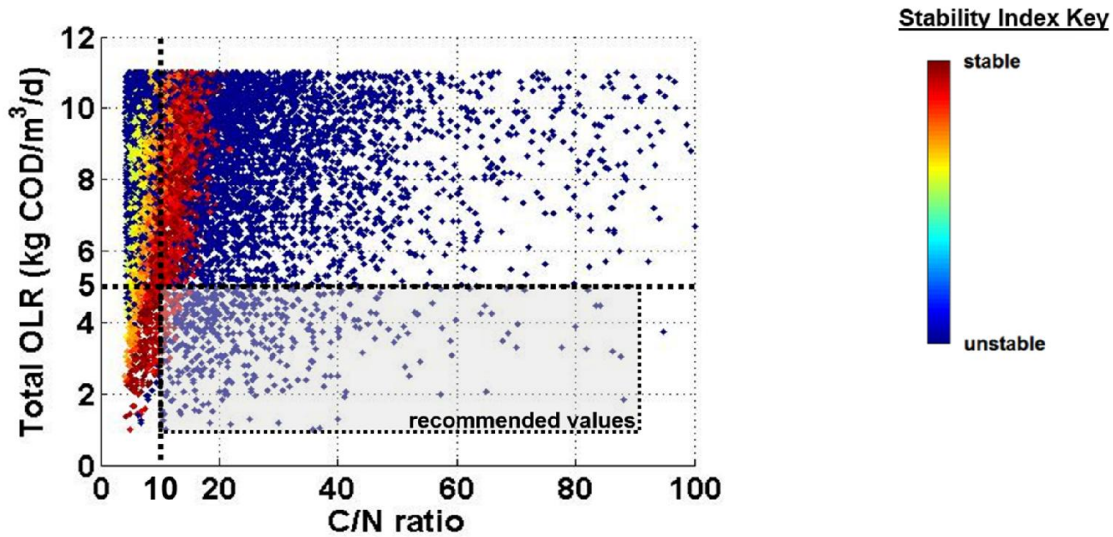


**Figure 7. The influent space described by total organic loading rate (OLR) and carbon to nitrogen (C/N) ratio; the boxed area represents the traditionally recommended design ranges for OLR (1-5 kg COD/m³/d [163]) and C/N ratio (10:1-90:1 [137,138]).**

Figure 7 suggests that the traditional OLR recommendation of 1-5 kg COD/m³/d is too conservative since many influents with loadings greater than this range were predicted as stable. Stable digester operation at OLRs greater than 5 kg COD/m³/d have also been seen experimentally. For example, Gomez et al.[150] observed stable codigestion of primary sludge with fruit and vegetable waste with an OLR around 5 kg COD/m³/d (volatile solids were converted to COD using average food waste data[21,249]), but did not increase the OLR until failure. Björnsson et al.[151] found that an OLR greater than 5 kg COD/m³/d of wastewater sludge and carbohydrate-rich food-processing resulted in stable operation, with failure occurring around 10 kg COD/m³/d. Also, Kusowski et al.[173] reported that an OLR up to 9 kg COD/m³/d resulted in the stable codigestion of

WWTP sludge with industrial wastes. While the modeling results suggest that stable digestion is also possible at OLRs greater than 9 kg $COD/m^3/d$, the current experimental literature has not reported this, and ADM1 effluent predictions have been found to be less accurate at high OLRs[245]. This modeling limitation highlights the need to combine experimental and modeling results to effectively inform codigestion design. Overall, for OLRs up to 9 kg $COD/m^3/d$, the modeling results and experimental data from the literature are consistent and show that the traditional OLR target values may be too conservative. The modeling results suggest that designing codigester experiments with higher loadings would be worthwhile as it would provide the opportunity to assess how much additional waste could be treated and how much extra energy could be produced for existing digester capacity.

Figure 7 also shows that the traditional target values for the C/N ratio guidelines have limited correlation with predicted stability. Many of the unstable simulated digesters had influents that met the C/N ratio target value, while many stable digesters had influents outside of the recommended range (i.e., less than 10:1). The general C/N target values may be limited since it aims to maximize methane production but does not identify all C/N ratios that support stable digestion. For example, Wang et al.[137] determined the optimal C/N ratio (corresponding to maximum methane production) for the batch codigestion of dairy manure, chicken manure, and wheat straw to be in the relatively narrow range of 25:1 to 30:1. They also found that a wider C/N range of 20:1 to 35:1 resulted in methane production and COD removal, which shows that recommending design C/N ratios too narrowly exclude C/N ratios that can achieve stable digestion. Also, typical WWTP sludge digesters are operated with C/N influent ratios of 6:1 to 16:1[36]. These influents result in stable sludge digestion even though the C/N ratio is not in the target design range. Since the currently recommended C/N ratio range is so wide, inclusive of some unstable and exclusive of some stable conditions, this design guideline used in isolation may be too limited to inform the design of stable codigestion systems.

The traditional values for these two design guidelines may be limited since the combined target values do not predict the observed stability of typical WWTP sludge digestion (OLRs of 1-5 kg $COD/m^3/d$[163] and C/N ratios of 6:1 to 16:1[36]) or the predicted stability of the modeled influents. Even though that target values seem to provide limited process stability insight, using these two

parameters (and axes) to describe codigestion influent has the potential to predict digester stability. The stability index modeling approach can be used to generate new target values for OLR and C/N ratio design guidelines that are specific to different codigestion influents. These guidelines can be used to guide subsequent experimentation needed to validate modeling predictions and subsequently to reduce cost, time, and failures probabilities during full-scale design.

### 4.4.4 Modeling the codigestion of WWTP sludge with organic wastes highlights the potential for using the stability index to establish design criteria for codigestion.

To identify waste streams and waste combinations that result in stable codigestion, the stability index was used to evaluate the addition of organic wastes to WWTP sludge digesters. For this evaluation, the representative sludge influent characteristics were based on waste characterization for primary sludge[250] and waste activated sludge[205] (Table 21 in Appendix B). The sludge waste stream contribution corresponded to an OLR of 2 kg COD/m$^3$/d and consisted of equal volumes of primary and waste activated sludges, which was based on typical design criteria for mesophilic sludge digestion[163]. To simulate multiple waste combinations, a diverse array of additional waste streams were combined with the sludge waste stream to predict codigestion performance for 10,000 different codigestate waste streams. Each codigestate waste stream had loadings of carbohydrates, lipids, and proteins that were each within a range of 0 to 8 kg COD/m$^3$/d, inert loadings within 0 to 20 percent of the codigestate waste stream's total OLR, and total OLRs between 0 to 8 kg COD/m$^3$/d. All of the modeled influent compositions corresponded to a consistent OLR of sludge of 2 kg COD/m$^3$/d, which was combined with a codigestate waste stream for a total influent OLR between 2 and 11 kg COD/m$^3$/d.

The simulated digestion for the lowest OLR of 2 kg COD/m$^3$/d (i.e., only the sludge waste stream) resulted in a highly stable digester with a stability index rating of 0.96. Figure 8 shows the relationship between digester stability and influent composition for the addition of a codigestate waste stream to a sludge digester with an OLR of 2 kg COD/m$^3$/d. In this figure, the influent compositions were described using the loadings of the codigestate waste streams only since the sludge OLR was consistent for all influents. The relationship between these codigestion influent compositions and stability is similar to the relationship between stability and particulate influents (i.e., Figure 6). One main similarity is that there is a strong relationship between digester stability

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and the protein loadings of the codigestate waste stream. However, this is one a notable difference: stable digestion is expected at low codigestate protein loadings (e.g., less than 1 kg COD/m$^3$/d). This is because the sludge waste stream (not shown in figure) is a large source of nitrogen. The sludge supply of nitrogen is both in the form of organic and inorganic nitrogen. Both forms of nitrogen were included when characterizing the sludge waste stream since researchers have found that the total nitrogen contribution of inorganic nitrogen is about 18-29% for primary sludge[250,251] and 2% for waste activated sludge[205,252]. The minimum codigestate protein loading required to avoid nitrogen-limited was based on the supply of nitrogen from the sludge waste stream. Both inorganic nitrogen and organic nitrogen have a strong influence on digester stability, and the inclusion of all wastes' nitrogen contributions will likely improve digester stability predictions.

Sections I and II of Figure 8c-d highlight two main codigestate waste stream characteristics that exhibit high stability: high loadings of carbohydrates and high loadings of lipids. For each codigestate characteristic, the potential for stable codigestion was examined by analyzing an example. The high-lipid codigestate example was the codigestion of WWTP sludge with restaurant grease trap waste[146]. The digesters at the Oceanside Water Pollution Control Plant in San Francisco, CA, codigested primary and waste activated sludges with grease trap waste from local food service establishments. The grease trap waste was loading at 0.4 to 1.9 kg COD/m$^3$/d. It was observed that the addition of this codigestate improved digester performance, as shown by an increase in the biogas methane content, methane generation, and volatile solids reduction. This example of a high-lipid codigestion influent shows the potential of stable codigestion of WWTP sludge with high-lipid waste, as suggested by Section I of Figure 8c.
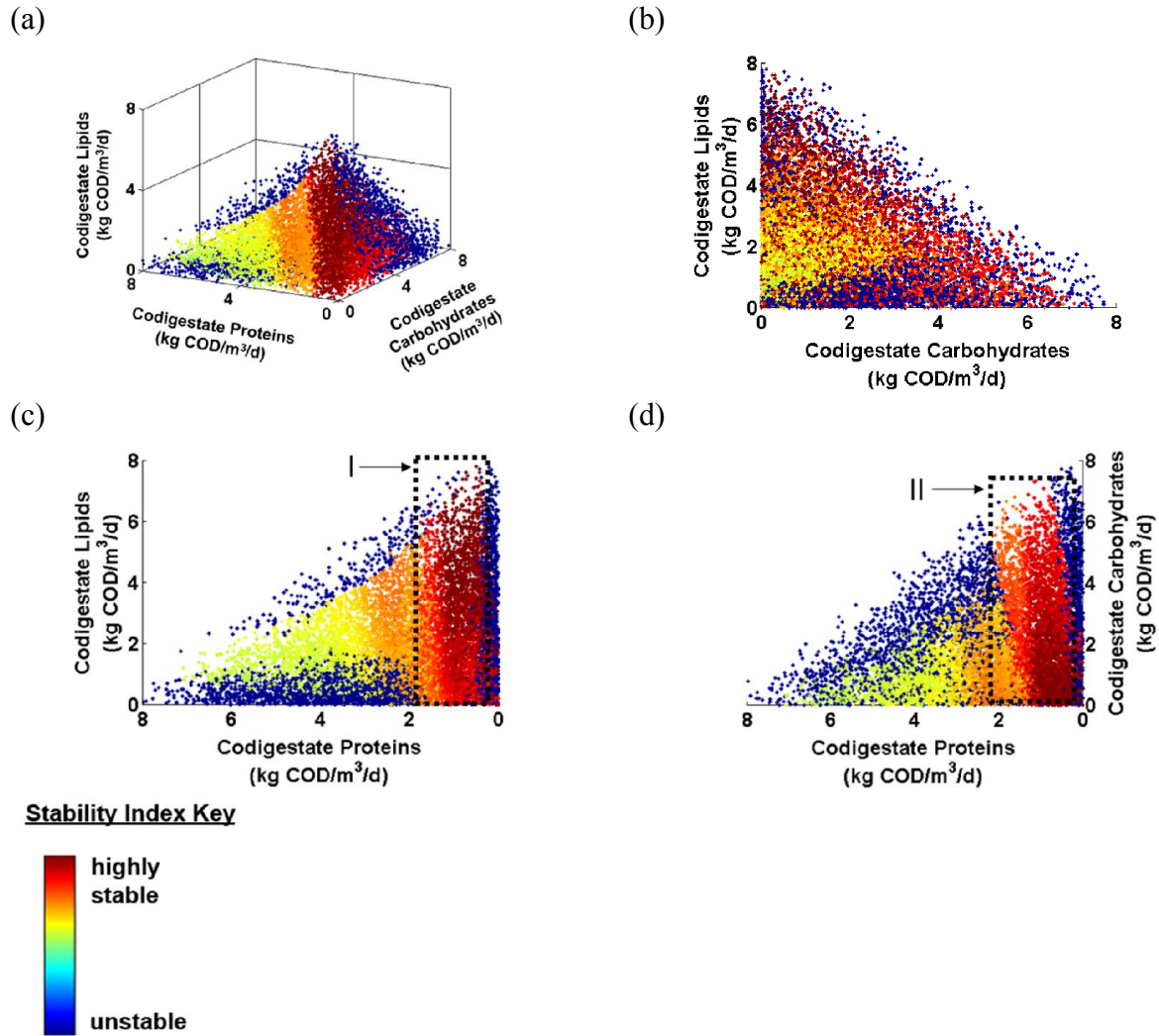
**Figure 8. Stability estimates for influents that combined a waste stream of 2 kg COD/m³/d loading of WWTP sludge with multiple codigestate waste streams that consisted of carbohydrates, proteins, and lipids loadings ranging from 0-8 kg COD/m³/d and inert loadings of 20% of the total codigestate waste stream COD. The total OLR for the combined waste streams was 2-11 kg COD/m³/d. Only the loadings of the codigestate waste streams are shown.**

A high-carbohydrate codigestate example was the codigestion of potato processing waste with WWTP sludge[35]. A lab-scale reactor was operated to replicate the full-scale digestion of WWTP primary and waste activated sludges with potato processing waste. The potato processing waste consisted of mostly carbohydrates, and the OLR of this waste was around 2.8 kg COD/m³/d (assuming a conversion of 1.6 g COD/g volatile solids[30]). The researchers concluded that the

combined waste stream resulted in stable digestion and found that loadings of the carbohydrate-rich waste could be increased to around 5 kg COD/$m^3$/d. This example shows the potential for the stable codigestion of WWTP sludge with carbohydrate-rich wastes, as suggested by Section II of Figure 8d.

Comparing the modeling results with these examples also suggests that detailed waste characterization data are needed to predict stability. The modeling results show that insufficient influent nitrogen could result in process instability, which is particularly important for influents with high levels of lipids and of carbohydrates. Although highly stable digestion is predicted with these types of influents, there is a clear stability boundary related to nitrogen-limited growth that shows the importance of knowing influent nitrogen loadings. Detailed information about influent and waste nitrogen loadings will improve the selection of waste combinations for stable codigestion. Also, understanding the digester's stability boundaries could provide valuable insight when working with highly variable wastes. Both Sections I and II of Figure 8 show that there is a large range of codigestate compositions that can result in stable codigestion with WWTP sludge. The stability index modeling approach can be used to evaluate the impact of a waste with variable composition by determining if its composition range falls within the digester's stable operating bounds. Coupling modeling results with experimentation can help determine how to control a digester's influent so that the composition and loading fluctuations are within the expected stable region. While the modeling results may suggest a range of waste compositions can result in stable operation, steady state simulations cannot address the impact of influent fluctuations over time on digester stability. Experimentation and an understanding of the microbial community is needed to understand the digester's ability to be resilient or resist perturbations[177] as well as to acclimate to high concentrations of inhibitory compounds, such as ammonia-nitrogen[247]. Overall, using the stability index coupled with an anaerobic digestion biological process model supports a design-oriented analysis of anaerobic codigestion.

## 4.5 Conclusions

By coupling this study's establishment of a stability index with anaerobic digestion process modeling, this work improves our understanding of codigestion from the functional perspective. This research also provides a modeling tool to guide codigestion experimentation and inform

performance optimization to reliably increase resource recovery from waste. Evaluating the stability index and elucidating the link between codigestion influent composition and process stability found that:

- Influent nitrogen strongly influences digester stability. The characterization of waste, especially inorganic and organic nitrogen content, is important for understanding and modeling codigestion stability.
- Traditional influent digestion design values have limited application to codigestion. The stability index provides a mechanism for updating target digestion design values for codigestion.
- There is potential for the stable codigestion of WWTP sludge with carbohydrate-rich or lipid-rich waste streams. The examination of full-scale digester studies provides examples of stable codigestion with these types of influents and demonstrates the potential of the stability index to predict stable digestion and select appropriate waste combinations.

## 4.6 Acknowledgements

# Chapter 5.
# Engineering Significance

This dissertation aimed to advance the sustainability of WWTPs by developing decision support tools for two complex WWTP issues: improving effluent quality and reducing net energy use. Two case studies were selected to evaluate these complex issues. Chapter 3's unused medication disposal case study resulted in a decision support tool that can inform public policy and disposal implementation options. Chapter 4's codigestion stability case study resulted in a decision support tool that can direct experimentation, inform sustainability assessments, expand and expedite field implementation, and help optimize digestion systems. Ultimately, a sustainability decision framework (Figure 1) was developed for and shaped by these two case studies. In this chapter, the potential impact and use of these two decision support tools and case studies are discussed. Finally, the impact of the sustainability decision framework is presented.

## 5.1 Unused Medication Disposal Case Study

For the unused medication disposal case study in Chapter 3, the decision-support tool has the opportunity to impact both U.S. legislation and future research. Specifically, the model we built to assess and compare unused medication disposal options is currently being used to understand the tradeoffs and limitations of disposal options. For example, the University of Michigan has used this tool and results from it to inform its policy and recommendation for its campus Safe Drug Disposal Program. While other programs in Michigan support only take-back disposal, the University of Michigan's program also supports the trash disposal of unused medications as a result of this study. In addition, this study is being included in discussions about and to oppose a California legislative proposal to establish a statewide take-back disposal program[65]. By informing the decisions of these two organizations, this study has helped to support unused medication disposal systems that balance the tradeoffs between environmental impacts (e.g., increasing

greenhouse gas emissions to reduce pharmaceutical emissions) while still being safe and cost-effective.

The unused pharmaceutical disposal model could also be used to help prioritize future legislative and research efforts to reduce environmental emissions of pharmaceuticals. For example, the model can be updated as research in this area progresses (e.g., with pharmaceutical removal predictions during wastewater treatment based on new technologies) in order to re-evaluate and re-design disposal options. Also, the model can be used to help determine the importance of different sources of pharmaceutical emissions and identify other pharmaceutical life cycle stages (e.g., pharmaceutical design to reduce excretion, pharmaceutical dispensing practices to reduce accumulation of unused medications, etc.) that should also be evaluated to reduce the environmental impact over the entire life cycle of a pharmaceutical. The model is available free of charge as a spreadsheet at: [http://deepblue.lib.umich.edu/handle/2027.42/91619](http://deepblue.lib.umich.edu/handle/2027.42/91619).

## 5.2 Anaerobic Codigestion Stability Case Study

The stability index tool developed in Chapter 4 provides a mechanism to generate data needed to support the design of stable codigestion systems. While anaerobic digestion technology and full-scale codigestion systems exist, there is still uncertainty regarding the process stability implications of codigestion. The tool specifically allows for a quick evaluation of available, characterized wastes to identify the best wastes to mix and to help design influent loadings. Use of this tool advances knowledge about operating stable codigestion systems and provides the confidence needed to implement codigestion over a wider range of applications by linking stability assessments with biological process models. Overall, by relying less on trial-and-error based experimental approaches and their concomitant resource requirements, this tool can help expedite the implementation of codigestion systems and allow more energy to be recovered from waste. In particular, the stability index supports the immediate implementation of sustainable technologies in the field, especially for WWTPs that have excess digester capacity.

The stability index tool can also be used to optimize digestion systems and design sustainable codigestion systems. In the case of codigestion, the tool can be used to maximize energy production without compromising process stability. Further, the stability index concept can be

adapted and applied to other biological systems to understand how they function and to support the optimization of their performance. Using the sustainability decision framework, the stability index can be combined with environmental and cost data to formulate a set of sustainability criteria for designing and operating a stable codigestion facility with minimal costs and environmental impacts.

## 5.3  Sustainability Decision Framework

This dissertation also aimed to advance the science of sustainability decision-making in the context of wastewater treatment. It increased the ability of the field to engage in sustainability decision-making by developing an assessment framework that extended LCA methodology to include the four pillars of sustainability--environment, economy, society, and function. The resulting sustainability decision framework (Figure 1) was developed through the process of evaluating the case studies in Chapters 3 and 4.  In particular, these case studies highlighted the need for a decision framework to incorporate uncertainty assessments and the four pillars of sustainability.

Chapter 3's unused medication disposal case study established the need to integrate life cycle thinking with economic and social considerations as well as the need to assess the impact of uncertainty on modeling results. The quantification and comparison of environmental emissions from three common disposal options highlighted the important tradeoff between the emission of pharmaceuticals and all other types of environmental emissions. In light of this environmental tradeoff, social and cost considerations were also necessary to understand the broader implications of the different disposal options. Once data for all three pillars of sustainability were considered, by integrating safety consideration (e.g., tendency for home storage), consumer participation, and disposal system costs with Chapter 3's environmental data, the most sustainable disposal option became apparent.

In addition, this case study showed the ability of uncertainty assessments to increase confidence in modeling results. This study's assessment of uncertainty highlighted the importance of evaluating assumptions in the face of incomplete information. Specifically, the uncertainty assessment emphasized the differences in environmental impacts between the disposal options (e.g., take-back disposal had significantly greater environmental emissions than the other disposal

options even when uncertainty was considered). Ultimately, the uncertainty assessment added certainty to the results and environmental performance trends.

Chapter 4's codigestion stability case study demonstrated the need to explicitly include the assessment of treatment performance during sustainability assessments. For example, digester failure or the need for corrective actions due to digester instability are resource intensive, so they merit careful consideration, especially during environmental and economic assessments. By evaluating codigestion from the functional perspective, insight is gained to support the comprehensive design of sustainable codigestion systems. This case study illustrated the value of adding "function" as a fourth pillar to sustainability, since a process or practice cannot be sustainable if it does not function.

Both case studies contributed to the development of this dissertation's sustainability decision framework. The framework provides a tool that engineers can use to gain a critical understanding of and set sustainability criteria for the design and implementation of waste management systems. It also has the potential for engaging engineers in public policy-making and has already proven useful for informing policy in the case of unused medication disposal. Further, this decision framework can be applied to other contemporary and complex issues in environmental engineering.

# Appendices

# Appendix A

## Supporting Information for Chapter 3

Although this paper focuses on disposal options for unused pharmaceuticals, we also note that system improvements are needed to reduce the occurrence and impacts of active pharmaceutical ingredient (API) release to the environment from all sources. First, human and animal excretion is considered a major source of environmental API emissions,[55] and therefore pharmaceuticals should be designed to minimize excretion.[131] Second, the adoption of a waste prevention mentality by pharmaceutical companies, prescribing entities, and consumers would reduce the accumulation of unused pharmaceuticals. Third, pharmaceutical design should prioritize API susceptibility to waste treatment so once APIs are excreted or directly disposed of by toilet/trash, they are easily rendered harmless at end-of-life.[131] Until such developments occur, the immediate disposal of all unused pharmaceuticals in the trash will significantly reduce human health risks associated with pharmaceuticals accumulated in households, decrease current releases of APIs to the environment, and avoid significant increases in most other non-API emissions.

## I.  Disposal Options

This section outlines the main assumptions and sources of data used for each modeling disposal option, and it follows the layout of the freely accessible spreadsheet (available at deepblue.lib.umich.edu; file name "Unused Pharmaceutical Disposal"), which contains all values and calculations used in this study (see Figure 9). Figure 10 shows the waste treatment systems employed by each disposal option for both APIs and associated packaging.
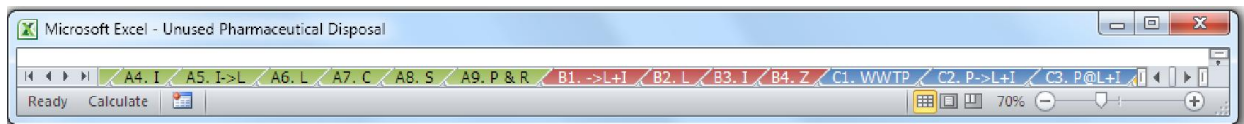
**Figure 9. Spreadsheet "Unused Pharmaceutical Disposal" with worksheet labels that match the headings in this section.**
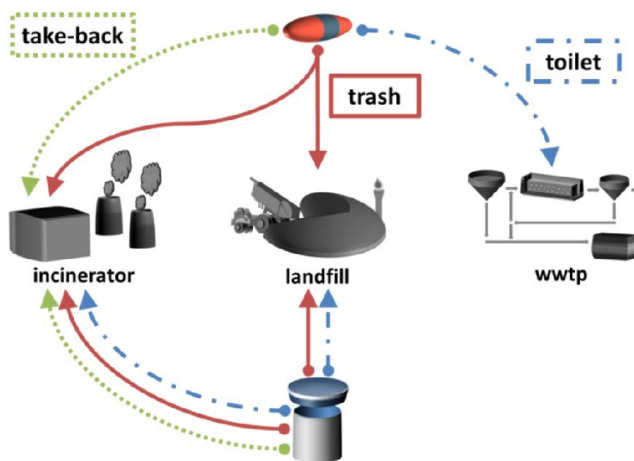


**Figure 10. Waste treatment systems employed by each disposal option to manage the APIs from unused pharmaceuticals and associated packaging.**
(wwtp = wastewater treatment plant)

## A. Take-back
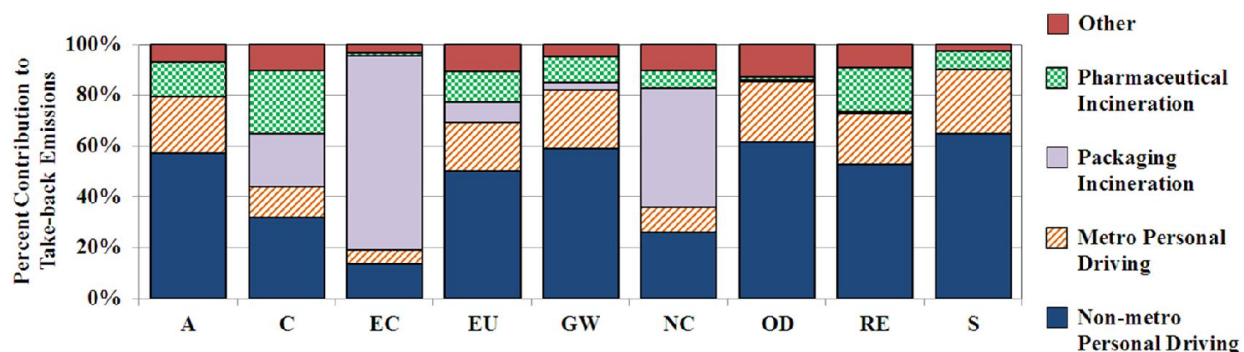


**Figure 11**. **The contribution to overall take-back emissions by the materials and processes required by take-back disposal**.
Abbreviations: A=acidification (H+ moles equivalent); C=carcinogenics (benzene equivalent); EC=ecotoxicity (kg 2,4-D equivalent); EU=eutrophication (kg nitrogen equivalent); GW=global warming (kg $CO_2$ equivalent); NC=non carcinogenics (toluene equivalent); OD=ozone depletion (kg CFC-11 equivalent); RE=respiratory effects (kg $PM_{2.5}$ equivalent); and S=smog (kg $NO_X$ equivalent).

### A1. Personal driving to pharmacy
- Average distances from 50 random residences to the closest pharmacy are determined for nine cities that represent the U.S.'s nine rural-urban continuum codes[222] (referred to as rurality categories)
- U.S. population (300 million) is distributed among these codes according to U.S. Census Bureau data[222]
- Types of personal vehicle are based on the 2008 vehicle market share,[253] and all were assumed to use gasoline only
- Uncertainty parameters are used to calculate total personal driving emissions (Table 14):
  - Number of annual trips by each participant (#1)
  - Percentage of trips that are single-purpose vs. multi-purpose (#2)
  - Percentage of miles allocated to take-back during a multi-purpose trip (#3)
  - Vehicle fuel economy (#54 and #55)

### A2. Hauling from pharmacy to secure storage facility
- Number of collection boxes is based on the mass of pharmaceutical waste returned and an assumed box capacity (Table 14 #4)
- Number of trips is based on the total mass hauled (pharmaceuticals, packaging, and boxes[254]) and a 21-ton[255] truck load
- For each rurality category, Google Maps™ mapping service is used to estimate the average driving distance from a pharmacy to a secure storage facility

### A3. Hauling from secure storage facility to incinerator
- Number of trips is based on a 21-ton truck load[255] and the total mass hauled
- Driving distances from a secure storage facility to the closest commercial hazardous waste incinerator[225] are estimated using Google Maps™ for two major pharmacy chains

### A4. Incineration of APIs, packaging, & cardboard boxes
- Emissions from incinerating packaging and cardboard boxes are from life cycle inventory (LCI) databases (see Table 8)
- APIs' emissions are specific to each representative API (see Section 3C and Table 11)
- Energy recovery during hazardous waste incineration is based on the mass percent combusted with energy recovery (Table 14 #6) and the assumed energy density of 0.535 MWh electricity/ton combusted[215]

### A5. Hauling API ash from incinerator to landfill
- Distances and the number of trips are based on the mass of ash generated (Table 14 #7), a 21-ton truck load,[255] and the average driving distance from a commercial hazardous waste incinerator[225] to the closest commercial hazardous waste landfill[226]

### A6. Landfilling of API ash
- Emissions from general landfill processes are from LCI databases (see Table 8)

### A7. Cardboard box production
- Emissions from cardboard production are from LCI databases (see Table 8)

## A8. Steel bin production

- Total steel mass is based on a steel bin's mass[224] and the number of U.S. pharmacies[256] (assuming each would have one steel bin if take-back is implemented nationally)
- Emissions from steel production are from LCI databases (see Table 8)
  - 60% by basic oxygen furnace and 40% by electric arc furnace[257]
  - A steel bin's lifetime is assumed to be 40 years, consistent with other LCI data and approaches[258,259]

## A9. Building infrastructure (pharmacy and secure storage facility)

- Dimensions of a steel bin[224] were used to calculate the floor area required at a pharmacy
- Floor area required at a secure storage facility is based on 3 months of storage capacity, cardboard box dimensions,[254] and assuming that boxes were stacked 25 high

## B. Trash
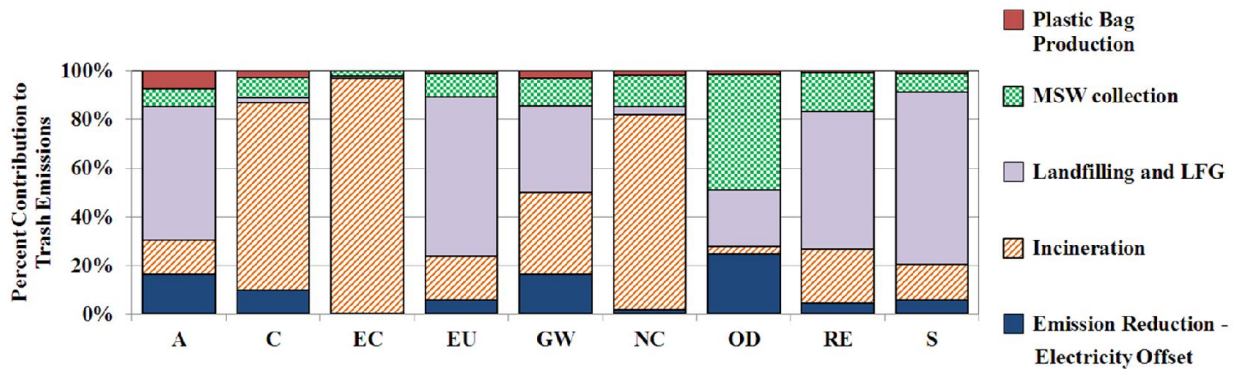


**Figure 12**. **The contribution to overall trash emissions by the materials and processes required by trash disposal**.
Note: recovery energy during incineration reduces emissions by offsetting electricity production.

### B1. Trash collection and hauling APIs, packaging, and plastic bags with municipal solid waste (MSW)

- Trashed APIs, packaging, and plastic bags are landfilled (81%) and combusted with energy recovery (19%)[210]
- MSW collection assumed a 7-ton garbage truck load[255] and distances for both collection (Table 14 #10) and hauling to an incinerator (Table 14 #9) or a landfill (Table 14 #8)

### B2. Landfilling of APIs, packaging, and plastic bags with MSW

- Emissions from general landfill processes are from LCI databases (see Table 8)
- API landfill gas (LFG) emissions are specific to each representative API (see Section 2B and Table 10)
  - LFG has 3 fates: flared (28%), combusted for energy recovery (31%), or directly emitted (41%)[217] (Table 14 #15)
- API contributions to leachate are specific to each representative API (see Section 2B)
  - Leachate is sent via piping (no energy required) to a wastewater treatment plant (WWTP)

### B3. Incineration of APIs, packaging, and plastic bags with MSW

- Emissions from packaging and plastic bags are from LCI databases (see Table 8)
- APIs' emissions are specific to each representative API (see Section 3C and Table 11)
- API ash from an MSW incinerator is hauled to a MSW landfill
  - Distance is estimated using an uncertainty parameter (Table 14 #11)
  - Number of trips is based on a 21-ton truck load[255] and total mass hauled
- Energy recovery assumes 0.535 MWh electricity/ton MSW combusted[215]

### B4. Plastic bag production (LDPE film)

- The mass of low-density polyethylene (LDPE) needed plastic bags is based on the mass of a "sandwich-sized" bag
- Emissions from LDPE production are from LCI databases (see Table 8)

## C. Toilet
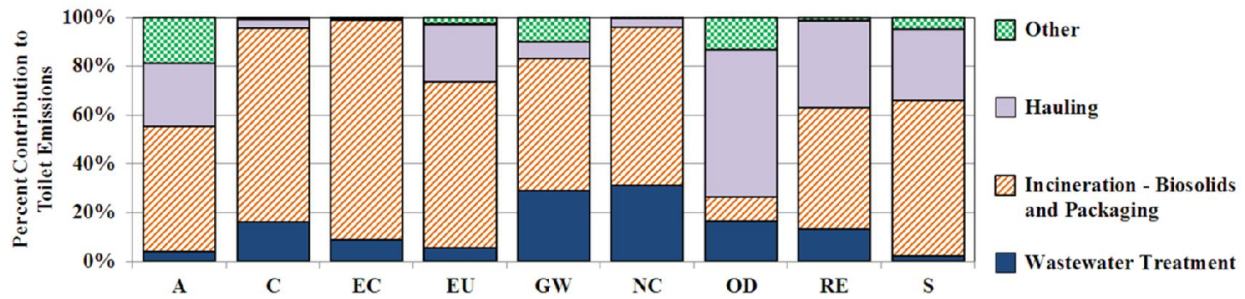


**Figure 13**. The contribution to overall toilet emissions by the materials and processes required by toilet disposal.

### C1. Wastewater treatment of flushed APIs

- Amount of wastewater generated is based on typical volume of U.S. toilets (1.6 gallons/flush) and the number of times a person flushed APIs in a year (Table 14 #19)
- Amount of electricity used to treat this wastewater based on typical values (Table 14 #20)
  - Emissions from electricity are from LCI databases (see Table 8)
- API emissions and fate specific to each API (see Section 3A)
  - APIs sorbed to the biosolids and biosolids formed by the assimilation of API carbon are incinerated, landfilled, or land applied[221]
    - Incinerated biosolids and sorbed APIs generate emissions specific to the compound: air emissions (with flue gas scrubbing) and ash generation (with final disposal in a landfill)
    - Landfilled biosolids and sorbed APIs are retained in a landfill
    - Land applied biosolids have no further emissions
    - Land applied sorbed APIs are counted as API emissions

### C2. Hauling packaging waste with MSW to a landfill and an incinerator

- Trashed packaging are combusted with energy recovery (19%) or landfilled (81%)[210]
- MSW collection assumed a 7-ton garbage truck load,[255] a collection route distance (Table 14 #10), hauling distance to a landfill (Table 14 #8), and hauling distance to an incinerator (Table 14 #9)

### C3. Packaging at landfill and incinerator

- Emissions from landfilling and incinerating packaging are from LCI databases (see Table 8)

# II.   Non-API Emissions

### A.  Life Cycle Inventory

Table 8 lists the unit processes considered in the life cycle inventory. Emissions are from three life cycle inventory databases: Ecoinvent,[124] Franklin USA 1998,[125] and U.S. LCI.[126] It also includes any modifications made to the original database's emissions (e.g. when U.S. data are substituted for European data).

**Table 8. Unit processes in the life cycle inventory.**

| Unit Process | Source(s) of Emissions Data |
|---|---|
| **Electricity generation and distribution (U.S. average)** | Franklin USA 1998 |
| **Building infrastructure (30% wood and 70% steel)** | Ecoinvent |
| **Landfill process and infrastructure** | Ecoinvent modified with U.S. electricity from Franklin USA 1998 database |
| **Landfill emissions, compound specific (no APIs)** | Ecoinvent |
| **Incineration process and infrastructure** | Ecoinvent |
| **WWTP infrastructure** | Ecoinvent |
| **Cardboard production** | Franklin USA 1998 |
| **Steel production (basic oxygen and electric air furnace)** | Franklin USA 1998 |
| **Plastic bag production (low-density polyethylene film)** | Franklin USA 1998 |
| **Gasoline production** | Franklin USA 1998 |
| **Personal vehicle (car) emissions and infrastructure** | Ecoinvent modified with emissions and fuel economy from EPA[260] |
| **Personal vehicle (light truck) emissions and infrastructure** | Ecoinvent modified with emissions and fuel economy from EPA[260] |
| **Diesel production** | Franklin USA 1998 |
| **Garbage truck emissions and infrastructure (full, load 7 tons)** | Ecoinvent and modified with fuel economy from U.S. LCI |
| **Garbage truck emissions and infrastructure (empty)** | Ecoinvent modified with fuel economy from U.S. LCI and emissions from EPA[261] and Federal Highway Administration[262] |
| **Semi-trailer truck emissions and infrastructure (full, load 21 tons)** | Ecoinvent modified with emissions and fuel economy from U.S. LCI |
| **Semi-trailer truck emissions and infrastructure (empty)** | Ecoinvent modified with fuel economy from U.S. LCI and emissions from EPA[261] and Federal Highway Administration[262] |

# III. API Fates and Emissions

Table 9 shows the nine representative APIs used in this study and their physicochemical properties (octanol/water partition coefficient, acid dissociation constant, and Henry's law constant).

**Table 9. API identification and physicochemical properties.** *(n/a is not available)*

| API | CAS# | log $K_{ow}$ (citation) | p$K_a$ (citation) | $K_h$[99] (atm-L/mol) |
|---|---|---|---|---|
| Acetaminophen | 103-90-2 | 0.46 [99] | 9.4 [99] | $6.42 \cdot 10^{-10}$ |
| Aspirin | 50-78-2 | 1.2 [99] | 3.5 [99] | $1.30 \cdot 10^{-6}$ |
| Vitamin E | 59-02-9 | 12 [99] | 11 [100] | *n/a* |
| Prednisone | 53-03-2 | 1.5 [99] | 12 [100] | $2.83 \cdot 10^{-7}$ |
| Ibuprofen | 15687-27-1 | 4.0 [99] | 4.9 [99] | $1.50 \cdot 10^{-4}$ |
| Warfarin | 81-81-2 | 2.6 [99] | 5.1 [99] | $2.77 \cdot 10^{-6}$ |
| Topiramate | 97240-79-4 | 1.3 [101] | 9.2 [100] | *n/a* |
| Etodolac | 41340-25-4 | 3.9 [99] | 4.7 [98] | *n/a* |
| Gabapentin | 60142-96-3 | -1.1 [99] | 4.7 [100] | $1.81 \cdot 10^{-7}$ |

Further Description of Table 1 in the manuscript
Values presented in Table 1 are calculated in the following manner (all percentages are bounded by 0% and 100%).

The WWTP overall percent removal values from the literature indicate the mass removal of an API from the liquid phase. The minimum WWTP overall removal is the cited literature value minus 25%, and the maximum value is the cited literature value plus 25%.

The WWTP percent sorption values are calculated using one-parameter linear free energy relationships and each API's physicochemical properties (as described below in Section 2A). The minimum values are calculated using minimum WWTP overall removal values and a mixed liquor suspended solids (MLSS) concentration of 1 g/L. The maximum sorption values use the maximum WWTP overall removal values an MLSS concentration of 4 g/L.

The WWTP percent biotransformation values are calculated as the difference between the overall percent removal and the percent removal by sorption. The minimum value subtracts the maximum sorption value from the minimum overall removal value. The maximum value subtracts the minimum sorption value from the maximum overall removal value.

The landfill biotransformation values are calculated by applying the anaerobic biotransformation potential parameter ($N_{LF/WWTP}$) to the average WWTP percent biotransformation values. The average WWTP percent biotransformation values are calculated using the literature WWTP overall removal values and the percent sorption values calculated with a MLSS concentration of 2.5 g/L. The minimum landfill biotransformation values are calculated when $N_{LF/WWTP}$=0. The maximum values are calculated when $N_{LF/WWTP}$=1.

Landfill sorption values are calculated with op-LFERs. The minimum values are estimated by subtracting 25% from the sorption values calculated when $N_{LF/WWTP}$=1, the open leachate collection and removal system (LCRS) flowrate=10000 L/ha/d, the closed LCRS=1000 L/ha/d, and the landfill height=10m. The maximum values are calculated by adding 25% to the sorption values calculated when $N_{LF/WWTP}$=0, the open LCRS=100 L/ha/d, the closed LCRS=10 L/ha/d, and the landfill is 50 meters high.

### A. Wastewater Treatment Plant

Figures S6 shows how API removal from the liquid phase of wastewater at a WWTP is modeled. Each API's physicochemical properties (Table 9) and values for the overall percent removal during wastewater treatment (Table 1) are used to calculate the percent of an API's total mass that will both sorb and be biotransformed.
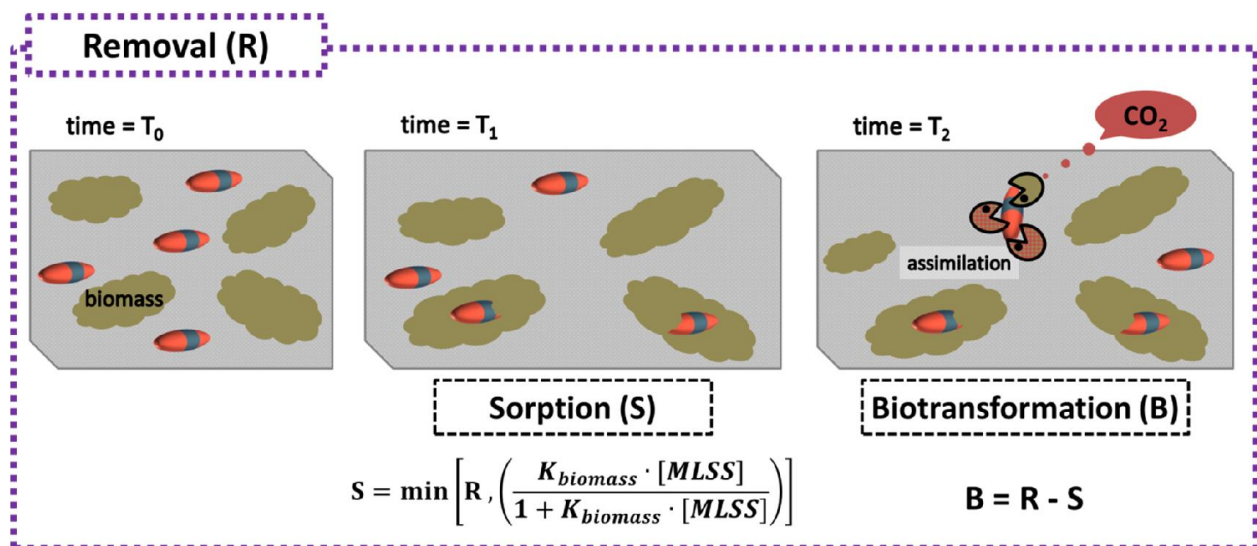


**Figure 14**. Removal is the overall WWTP percent removal for an API (R); sorption is to biomass (S); biotransformation is aerobic (B).

$$S = \min\left[R,\left(\frac{K_{biomass} \cdot [MLSS]}{1 + K_{biomass} \cdot [MLSS]}\right)\right]$$

$$B = R - S$$

For sorption calculations, we assume that sorbed APIs are not bioavailable. Desorption is not included in these calculations but is later incorporated with uncertainty parameters used for the final sorption estimations (see Table 14 #31-39). Each API's biomass/wastewater partition coefficient ($K_{biomass}$) is estimated using its octanol/water partition coefficient($K_{OW}$), its acid dissociation constant ($pK_a$), a wastewater pH of 7.5,[107] and the op-LFER given below:[58]

$$K_{biomass}\left(\frac{liter}{gram}\right) = \frac{\left(\frac{mass\ of\ API\ sorbed}{mass\ of\ biomass}\right)}{\left(\frac{mass\ of\ API\ in\ liquid\ phase}{volume\ of\ solution}\right)} \tag{S1}$$

$$= 10^{m \cdot \log\left(\frac{K_{ow}}{1+10 \cdot |pH-pK_a|}\right)+b}$$

Where coefficients m and b are based on an API's functional groups.

The API mass percentage that sorbs to biomass is calculated using typical MLSS concentrations (Table 14 #40) and each API's $K_{biomass}$ coefficient with the following equation:[43]

$$\%\ Sorbed = \frac{C_{sorbed}}{C_{soluble}+C_{sorbed}} = \frac{K_{biomass} \cdot [MLSS]}{1 + K_{biomass} \cdot [MLSS]} \tag{S2}$$

Where $C_{sorbed}$ is the concentration of API sorbed and $C_{soluble}$ is the concentration of API in the liquid phase.

For the final calculation of percent removal by sorption, the percentage was bound by an API's overall WWTP percent removal as shown by the following equation:

$$(\%\ Removed\ by\ Sorption)$$
$$= \min\left[(Overall\ \%\ Removal), \left(\frac{K_{biomass} \cdot [MLSS]}{1 + K_{biomass} \cdot [MLSS]}\right)\right] \tag{S3}$$

The remaining overall percent removal is assumed to be by biotransformation, shown here:

$$\text{(\% Removed by biotransformation)}$$
$$= \text{(Overall WWTP \% Removal)} - \text{(\% Removed by Sorption)} \qquad \textbf{(S4)}$$

Due to the lack of information about API transformation products (TPs), we assume that all aerobically biotransformed APIs are mineralized or assimilated into biomass. The mass of API carbon assimilated is based on observed yields for heterotrophs (Table 14 #17).

## B. Landfill

Figure 15 shows the approach used to calculate the mass of APIs that anaerobically biotransform, sorb to MSW, or leave a landfill in the leachate.
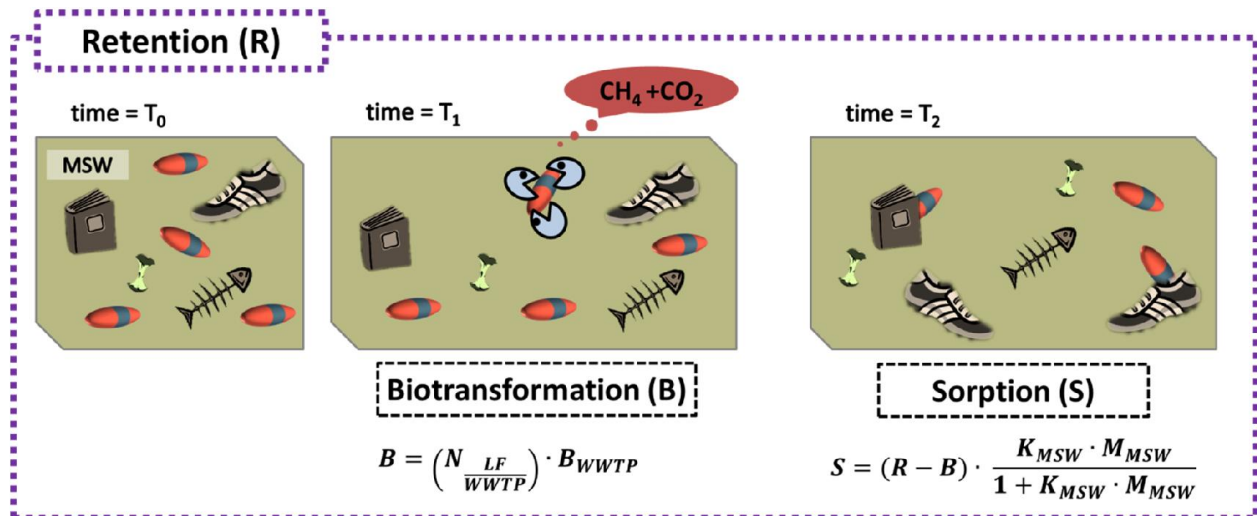


**Figure 15**. Landfill retention (R) is the total API mass that does not enter the leachate. It is the sum of APIs that are anaerobically biotransformed (B) and sorbed to MSW (S). Anaerobic biotransformation is based on the extent of aerobic biotransformation during wastewater treatment (BWWTP).

The mass of API expected to anaerobically biotransform in a landfill is calculated using each API's aerobic wastewater biotransformation percentage with the following equation:

$$(\% \text{ Biotransformation in a Landfill}) = \left( N_{\frac{LF}{WWTP}} \right) \cdot (\% \text{ Biotransformation in a WWTP}) \quad \text{(S5)}$$

Where the following is an uncertainty parameter (Table 14 #41):

$$N_{\frac{LF}{WWTP}} = \frac{extent\ of\ landfill\ anaerobic\ biotransformation}{extent\ of\ wastewater\ aerobic\ biotransformation} = [0,1]$$

Similar to wastewater treatment, due to the lack of information about API TPs, we assume that all anaerobically biotransformed APIs form LFG. Since all non-API emissions will be lower than baseline emissions if landfilled APIs do not produce LFG, the recommendation of trash disposal will not change if future research shows that LFG production from APIs is minimal. The LFG composition[216] is based on each API's chemical formula and the assumption that 45% of carbon forms $CO_2$ and 55% forms $CH_4$[212] (see Table 10). Combusted LFG (31% for energy recovery and 28% flared[217]) is assumed to be completely oxidized.

**Table 10. Landfill gas composition (mole compound/mole API) before flaring.**

| API | $CO_2$ | $CH_4$ | $H_2$ | $H_2O$ | $N_2$ | $H_2S$ |
|---|---|---|---|---|---|---|
| Acetaminophen | 3.6 | 4.4 | 0.0 | 0.0 | 0.5 | 0.0 |
| Aspirin | 4.1 | 5.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Vitamin E | 13 | 16.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Prednisone | 9.5 | 12 | 0.0 | 0.0 | 0.0 | 0.0 |
| Ibuprofen | 5.9 | 7.2 | 0.0 | 0.0 | 0.0 | 0.0 |
| Warfarin | 8.6 | 11 | 0.0 | 0.0 | 0.0 | 0.0 |
| Topiramate | 5.4 | 6.6 | 0.0 | 0.0 | 0.5 | 1.0 |
| Etodolac | 7.7 | 9.4 | 0.0 | 0.0 | 0.5 | 0.0 |
| Gabapentin | 4.1 | 5.0 | 0.0 | 0.0 | 0.5 | 0.0 |

Sorption estimations are based on a MSW/leachate partition coefficient ($K_{MSW}$) for each API. These coefficients are calculated with the following op-LFER that describes the sorption of organic compounds to MSW:[220]

$$K_{MSW}\left(\frac{liter}{gram}\right) = \frac{\left(\dfrac{mass\ of\ API\ sorbed}{net\ mass\ of\ MSW}\right)}{\left(\dfrac{mass\ of\ API\ in\ liquid\ phase}{volume\ of\ leachate}\right)} = 10^{0.816 \cdot \log(K_{ow}) + 0.166} \tag{S6}$$

The net mass of MSW and volume of leachate and are calculated using average values for landfills in the U.S.[212,216,255,263] The net mass of MSW is calculated yearly over a 40 year period, which is the length of time that the leachate from a landfill is collected, removed, and treated.[212] A landfill was assumed to accept MSW for 20 years.[212] MSW degradation is calculated using the U.S. Environmental Protection Agency's (EPA) model[216] with average parameter values.[212] Leachate volume is calculated by estimating a typical height for landfills (Table 14 #44) and modeling a landfill to have one cell, 2-8 acres in size,[263] open at a time. An open cell is assumed to receive waste and be compacted and covered daily. After 2 years,[263] an open cell is assumed to be full and is closed by placing a permanent cover on top. Open and closed cells have different leachate collection and recovery system (LCRS) rates due to the differences in each cover's permeability,[255] and these values are estimated with uncertainty parameters (Table 14 #42 and #43).

The API mass percentage that sorbs to MSW is calculated using the net mass of MSW in a landfill ($M_{MSW}$) and the API's $K_{MSW}$ coefficient with the following equation:

$$(\%\ Sorbed) = \frac{C_{sorbed}}{C_{leachate} + C_{sorbed}} = \frac{K_{MSW} \cdot [M_{MSW}]}{1 + K_{MSW} \cdot [M_{MSW}]} \tag{S7}$$

Finally, the API mass that does not biotransform or sorb is removed with the landfill leachate and sent to a WWTP for treatment.

## C. Incinerator

Table 11 shows the air emissions assumed to result from API incineration before flue gas treatment. Before release to the atmosphere, a wet scrubber removes 90% of the sulfur dioxide mass[213,214] and a catalytic converter removes 90% of the nitrogen dioxide mass[214] from the flue gas. The percentage of API mass that becomes incineration ash is based on average values for MSW (Table 14 #7). Incineration ash does not contain APIs and is hauled to a landfill.

**Table 11. Emissions from the complete oxidation of APIs (incineration) based on each compound's chemical formula.**

|  | $CO_2$ (kg/kg API) | $H_2O$ (kg/kg API) | $NO_2$ (kg/kg API) | $SO_2$ (kg/kg API) |
|---|---|---|---|---|
| **Acetaminophen** | 2.0 | 0.4 | 0.3 | 0.0 |
| **Aspirin** | 1.9 | 0.3 | 0.0 | 0.0 |
| **Vitamin E** | 2.5 | 0.9 | 0.0 | 0.0 |
| **Prednisone** | 2.2 | 0.6 | 0.0 | 0.0 |
| **Ibuprofen** | 2.4 | 0.7 | 0.0 | 0.0 |
| **Warfarin** | 2.3 | 0.4 | 0.0 | 0.0 |
| **Topiramate** | 1.3 | 0.5 | 0.1 | 0.2 |
| **Etodolac** | 2.2 | 0.6 | 0.1 | 0.0 |
| **Gabapentin** | 2.0 | 0.8 | 0.2 | 0.0 |

# IV. Emissions Results

Table 12 describes the scenarios discussed in the research article. Table 13 lists emission magnitudes for all disposal scenarios. Table 14 describes all uncertainty parameters. Table 15 provides result from the sensitivity analysis.

**Table 12**. **Disposal scenarios with participation rates and adjustments made to uncertainty parameters**.

| Disposal Scenarios | Participation | | | | Uncertainty Parameters | |
| --- | --- | --- | --- | --- | --- | --- |
| | Take-back (metro) | Take-back (non-metro) | Trash | Toilet | % Single purpose trips for take-back | % Miles allocated to take-back during multi-purpose trips |
| Baseline | 0% | 0% | 60% | 40% | 0-100% | 0-100% |
| Toilet | 0% | 0% | 0% | 100% | | |
| Trash | 0% | 0% | 100% | 0% | | |
| Take-back | 100% | 100% | 0% | 0% | | |
| "Best-Case" Take-Back (I) | 50% | 0% | 0% | 0% | 0% | 0% |
| "Best-Case" Take-Back (II) | 50% | 0% | 10% | 7% | | |
| 43% Take-back | 43% | 43% | 57% | 0% | | |
| 43% Take-back without driving | 43% | 43% | 57% | 0% | 0% | 0% |

## A) Emissions Magnitudes
### Table 13. The mean as well as the 10th, 25th, 50th, 75th, and 90th percentile values from each scenario's Monte Carlo simulation.

| | | A | C | EC | EU | GW | NC | OD | RE | S | API |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Baseline** | Mean | 5.3E+07 | 7.2E+04 | 1.9E+08 | 3.7E+04 | 8.4E+07 | 6.8E+08 | 4.7E-01 | 1.5E+05 | 7.6E+05 | 2.3E+07 |
| | 10% | 1.2E+07 | 6.2E+04 | 1.3E+08 | 1.5E+04 | 6.0E+07 | 5.2E+08 | 3.0E-01 | 2.5E+04 | 2.5E+05 | 2.1E+07 |
| | 25% | 2.0E+07 | 6.6E+04 | 1.5E+08 | 1.9E+04 | 6.8E+07 | 5.8E+08 | 3.8E-01 | 3.5E+04 | 3.5E+05 | 2.2E+07 |
| | 50% | 3.9E+07 | 7.1E+04 | 1.8E+08 | 3.1E+04 | 7.9E+07 | 6.7E+08 | 4.7E-01 | 6.7E+04 | 6.1E+05 | 2.3E+07 |
| | 75% | 7.1E+07 | 7.6E+04 | 2.2E+08 | 5.0E+04 | 9.6E+07 | 7.6E+08 | 5.6E-01 | 2.0E+05 | 1.0E+06 | 2.4E+07 |
| | 90% | 1.2E+08 | 8.2E+04 | 2.4E+08 | 7.1E+04 | 1.2E+08 | 8.6E+08 | 6.4E-01 | 4.1E+05 | 1.5E+06 | 2.5E+07 |
| **Toilet** | Mean | 4.5E+06 | 7.4E+04 | 2.0E+08 | 8.2E+03 | 7.3E+07 | 8.1E+08 | 2.6E-01 | 1.1E+04 | 1.2E+05 | 5.3E+07 |
| | 10% | 2.9E+06 | 6.0E+04 | 1.4E+08 | 6.4E+03 | 5.8E+07 | 5.5E+08 | 1.9E-01 | 7.7E+03 | 8.2E+04 | 4.9E+07 |
| | 25% | 3.6E+06 | 6.5E+04 | 1.6E+08 | 7.2E+03 | 6.4E+07 | 6.3E+08 | 2.2E-01 | 9.0E+03 | 9.7E+04 | 5.1E+07 |
| | 50% | 4.5E+06 | 7.1E+04 | 2.0E+08 | 8.2E+03 | 7.3E+07 | 7.4E+08 | 2.5E-01 | 1.1E+04 | 1.2E+05 | 5.3E+07 |
| | 75% | 5.3E+06 | 7.9E+04 | 2.3E+08 | 9.2E+03 | 8.2E+07 | 8.9E+08 | 3.0E-01 | 1.2E+04 | 1.3E+05 | 5.5E+07 |
| | 90% | 6.0E+06 | 9.4E+04 | 2.5E+08 | 1.0E+04 | 9.0E+07 | 1.2E+09 | 3.5E-01 | 1.4E+04 | 1.5E+05 | 5.7E+07 |
| **Trash** | Mean | 8.5E+07 | 7.0E+04 | 1.8E+08 | 5.7E+04 | 9.2E+07 | 6.0E+08 | 6.1E-01 | 2.4E+05 | 1.2E+06 | 2.7E+06 |
| | 10% | 1.7E+07 | 6.1E+04 | 1.3E+08 | 1.9E+04 | 5.4E+07 | 4.7E+08 | 3.6E-01 | 3.4E+04 | 3.3E+05 | 1.1E+06 |
| | 25% | 3.0E+07 | 6.5E+04 | 1.4E+08 | 2.7E+04 | 6.4E+07 | 5.2E+08 | 4.7E-01 | 5.1E+04 | 5.0E+05 | 1.8E+06 |
| | 50% | 6.2E+07 | 7.0E+04 | 1.8E+08 | 4.6E+04 | 8.1E+07 | 5.9E+08 | 6.1E-01 | 1.0E+05 | 9.4E+05 | 2.6E+06 |
| | 75% | 1.2E+08 | 7.5E+04 | 2.1E+08 | 7.8E+04 | 1.1E+08 | 6.8E+08 | 7.4E-01 | 3.3E+05 | 1.7E+06 | 3.5E+06 |
| | 90% | 1.9E+08 | 7.9E+04 | 2.3E+08 | 1.1E+05 | 1.4E+08 | 7.3E+08 | 8.6E-01 | 6.7E+05 | 2.4E+06 | 4.4E+06 |
| **Take-back** | Mean | 3.5E+08 | 9.0E+05 | 1.2E+09 | 3.7E+05 | 1.9E+09 | 5.8E+09 | 9.5E+00 | 4.6E+05 | 1.4E+07 | 0.0E+00 |
| | 10% | 1.5E+08 | 6.1E+05 | 8.6E+08 | 1.8E+05 | 7.8E+08 | 4.1E+09 | 3.6E+00 | 2.1E+05 | 4.7E+06 | 0.0E+00 |
| | 25% | 2.1E+08 | 7.0E+05 | 1.0E+09 | 2.4E+05 | 1.1E+09 | 4.8E+09 | 5.5E+00 | 2.9E+05 | 7.5E+06 | 0.0E+00 |
| | 50% | 3.3E+08 | 8.6E+05 | 1.2E+09 | 3.5E+05 | 1.8E+09 | 5.7E+09 | 8.8E+00 | 4.3E+05 | 1.3E+07 | 0.0E+00 |
| | 75% | 4.8E+08 | 1.1E+06 | 1.4E+09 | 4.8E+05 | 2.6E+09 | 6.8E+09 | 1.3E+01 | 6.0E+05 | 1.9E+07 | 0.0E+00 |
| | 90% | 6.0E+08 | 1.2E+06 | 1.5E+09 | 5.9E+05 | 3.3E+09 | 7.7E+09 | 1.7E+01 | 7.4E+05 | 2.5E+07 | 0.0E+00 |

**Table 13 Continued.**

| | | A | C | EC | EU | GW | NC | OD | RE | S | API |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **"Best Case" Take-back** | Mean | 1.1E+08 | 5.0E+05 | 8.7E+08 | 1.4E+05 | 5.3E+08 | 3.7E+09 | 2.6E+00 | 1.8E+05 | 3.0E+06 | 4.0E+06 |
| | 10% | 7.1E+07 | 4.3E+05 | 6.3E+08 | 1.0E+05 | 3.5E+08 | 3.0E+09 | 1.6E+00 | 1.2E+05 | 1.6E+06 | 3.6E+06 |
| | 25% | 8.4E+07 | 4.6E+05 | 7.2E+08 | 1.2E+05 | 4.1E+08 | 3.3E+09 | 2.0E+00 | 1.4E+05 | 2.0E+06 | 3.8E+06 |
| | 50% | 1.1E+08 | 5.0E+05 | 8.7E+08 | 1.3E+05 | 5.1E+08 | 3.7E+09 | 2.5E+00 | 1.6E+05 | 2.8E+06 | 3.9E+06 |
| | 75% | 1.3E+08 | 5.4E+05 | 1.0E+09 | 1.6E+05 | 6.3E+08 | 4.1E+09 | 3.2E+00 | 2.1E+05 | 3.8E+06 | 4.1E+06 |
| | 90% | 1.6E+08 | 5.8E+05 | 1.1E+09 | 1.8E+05 | 7.4E+08 | 4.4E+09 | 3.7E+00 | 2.6E+05 | 4.6E+06 | 4.3E+06 |
| **43% Take-back** | Mean | 2.0E+08 | 4.3E+05 | 6.2E+08 | 1.9E+05 | 8.8E+08 | 2.9E+09 | 4.6E+00 | 3.3E+05 | 6.6E+06 | 1.5E+06 |
| | 10% | 9.8E+07 | 3.1E+05 | 4.5E+08 | 1.1E+05 | 3.9E+08 | 2.2E+09 | 2.0E+00 | 1.5E+05 | 2.6E+06 | 6.4E+05 |
| | 25% | 1.4E+08 | 3.5E+05 | 5.2E+08 | 1.4E+05 | 5.4E+08 | 2.5E+09 | 2.8E+00 | 2.0E+05 | 3.9E+06 | 1.0E+06 |
| | 50% | 1.9E+08 | 4.2E+05 | 6.2E+08 | 1.8E+05 | 8.3E+08 | 2.9E+09 | 4.3E+00 | 2.9E+05 | 6.2E+06 | 1.5E+06 |
| | 75% | 2.6E+08 | 5.1E+05 | 7.1E+08 | 2.4E+05 | 1.2E+09 | 3.3E+09 | 6.1E+00 | 4.1E+05 | 8.9E+06 | 2.0E+06 |
| | 90% | 3.1E+08 | 5.8E+05 | 7.8E+08 | 2.9E+05 | 1.5E+09 | 3.8E+09 | 7.6E+00 | 5.9E+05 | 1.1E+07 | 2.5E+06 |
| **43% Take-back Without driving** | Mean | 8.0E+07 | 2.7E+05 | 5.2E+08 | 8.2E+04 | 2.0E+08 | 2.0E+09 | 1.1E+00 | 1.9E+05 | 1.3E+06 | 1.5E+06 |
| | 10% | 4.1E+07 | 2.4E+05 | 3.7E+08 | 5.9E+04 | 1.7E+08 | 1.7E+09 | 8.6E-01 | 7.4E+04 | 7.7E+05 | 6.4E+05 |
| | 25% | 4.9E+07 | 2.5E+05 | 4.3E+08 | 6.5E+04 | 1.9E+08 | 1.8E+09 | 9.5E-01 | 8.4E+04 | 8.7E+05 | 1.0E+06 |
| | 50% | 6.7E+07 | 2.7E+05 | 5.2E+08 | 7.6E+04 | 2.0E+08 | 2.0E+09 | 1.1E+00 | 1.1E+05 | 1.1E+06 | 1.5E+06 |
| | 75% | 9.8E+07 | 2.8E+05 | 6.2E+08 | 9.5E+04 | 2.2E+08 | 2.3E+09 | 1.2E+00 | 2.4E+05 | 1.5E+06 | 2.0E+06 |
| | 90% | 1.4E+08 | 2.9E+05 | 6.8E+08 | 1.1E+05 | 2.4E+08 | 2.4E+09 | 1.3E+00 | 4.3E+05 | 2.0E+06 | 2.5E+06 |

Abbreviations: A=acidification (H+ moles equivalent); C=carcinogenics (benzene equivalent); EC=ecotoxicity (kg 2,4-D equivalent); EU=eutrophication (kg nitrogen equivalent); GW=global warming (kg $CO_2$ equivalent); NC=non carcinogenics (toluene equivalent); OD=ozone depletion (kg CFC-11 equivalent); RE=respiratory effects (kg $PM_{2.5}$ equivalent); and S=smog (kg $NO_X$ equivalent).

## B) Uncertainty Parameters

**Table 14**. **The ranges and data sources of each uncertainty parameter.**

Notes: OM=order of magnitude. All distances are one-way.

| # | Uncertainty Parameters | Units | Low (L) Value | High (H) Value | Justification | Notes |
|---|---|---|---|---|---|---|
| 1 | Number of trips to pharmacy | trips/cap/yr | 1 | 12 | L=once a year H=once a month | Number of trips an individual makes to return unused medications |
| 2 | Single-purpose trips | % trips | 0% | 100% | L,H=total possible range | Percentage of trips with the sole purpose of returning medication |
| 3 | Take-back mile allocation for multi-purpose trips | % distance | 0% | 100% | L,H=total possible range | Percentage of a multi-purpose trip's distance allocated to take-back |
| 4 | Mass of full cardboard box | lbs | 20 | 40 | L=minus 10 lbs H=plus 10 lbs | Assumed 20-gallon box can hold 30 lbs (twice mass for 10-gallon box[254]). |
| 5 | Mass packaging per pharmaceutical | % mass | 10% | 20% | L,H=cited range | Pilot take-back program found 10-20% of the collected waste was packaging[67] |
| 6 | Take-back incineration energy recovery | % mass | 0% | 100% | L,H=total possible range | Energy recovered during the incineration of hazardous waste |
| 7 | Incineration ash from APIs and biosolids | % mass | 2% | 20% | L=PE[259] H= MSW[259] | Ash produced estimated as similar to polyethylene (PE) and average MSW |
| 8 | City to landfill | miles | 10 | 100 | L=half of [264] H=OM more than L | EPA WARM default distance (20 miles)[264] |
| 9 | City to incinerator | miles | 10 | 100 | L=half of [264] H=OM more than L | EPA WARM default distance (20 miles)[264] |
| 10 | Garbage truck collection route distance | miles | 10 | 100 | L=half of [264] H=OM more than L | Assumed similar to WARM default distance |
| 11 | Incinerator to landfill | miles | 10 | 100 | L=half of [264] H=OM more than L | Assumed similar to WARM default distance |
| 12 | WWTP to incinerator | miles | 10 | 100 | L=half of [264] H=OM more than L | Assumed similar to WARM default distance |
| 13 | WWTP to landfill | miles | 10 | 100 | L=half of [264] H=OM more than L | Assumed similar to WARM default distance |

**Table 14. Continued.**

| # | Uncertainty Parameters | Units | L | H | Justification | Notes |
|---|---|---|---|---|---|---|
| 14 | WWTP to land application site | miles | 10 | 100 | L=a third of [265] H=OM more than L | A WWTP to a land application site is usually more than 30 miles[265] |
| 15 | Landfill gas (LFG) recovered for energy | % mass | 31% | 100% | L=[217] H=maximum | Currently 28% flaring, 31% energy recovery, and 41% direct emissions[217] |
| 16 | Landfill gas energy content | kWh/short ton MSW | 139 | 362 | L,H=cited range | Energy content of landfill gas when combusted for electricity generation[266] |
| 17 | Aerobic, heterotroph observed yield | g cell COD/ g API COD | 0.0 | 0.71 | L=no growth H=[163] | Typical activated sludge kinetic coefficients for heterotrophic bacteria[163] |
| 18 | Mass of pharmaceutical per bag (trash) | lbs | 0.06 | 0.7 | L=once a month H=once a year | Each person has 0.7 lbs of drugs thrown away at least once a year |
| 19 | Mass of pharmaceutical per flush (toilet) | kg/flush | 0.027 | 0.32 | L=once a month H=once a year | Each person has 0.32 kg of drugs flushed away at least once a year |
| 20 | Wastewater treatment energy demand | kWh per $10^6$ gallons | 530 | 1100 | L=1200*(1-0.56) H=2500*(1-0.56) | WWTPs use 1200-2500 kWh/Mgal and 56% is for aeration[11] |
| 21 | WWTP biogenic electricity sources | % kWh | 0% | 100% | L,H=total possible range | Energy recovered from waste (with biogenic carbon emissions) |
| 22-30 | WWTP removal factors for all 9 APIs | % mass | -25% | +25% | L,H=(% overall removal) ±25% | Factor applied to WWTP overall percent removal values[92–98] |
| 31-39 | WWTP sorption factors for all 9 APIs | % mass | -25% | +25% | L,H=(% sorption) ±25% | Factor applied to the calculated percent sorption values |
| 40 | MLSS | g/L | 1.0 | 4.0 | L=conventional H=complete mix | Mixed liquor suspended solids (MLSS) typical concentrations and ranges[163] |
| 41 | $N_{LF/WWTP}$ | *dimension-less* | 0.0 | 1.0 | L,H=total possible range | Ratio of anaerobic to aerobic percent biotransformation |
| 42 | Daily cover LCRS | L/ha/d | 100 | 10000 | L=OM less than [212] H=OM more than [212] | Leachate collection and recovery systems (LCRS) rate for open landfill[212] |
| 43 | Final cover LCRS | L/ha/d | 10 | 1000 | L=OM less than [212] H=OM more than [212] | LCRS rate for closed landfill[212] |
| 44 | Final landfill height | m | 10 | 50 | L=shortest H=tallest | Total height for several landfills in the U.S.[212] |

**Table 14. Continued.**

| # | Uncertainty Parameters | Units | L | H | Justification | Notes |
|---|---|---|---|---|---|---|
| 45-53 | Landfill sorption factor – for all 9 APIs | % mass | -25% | +25% | L,H=(% sorption) ±25% | Factor applied to the calculated percent sorption |
| 54 | Car mileage | miles per gal gas | 19.4 | 26.9 | L=([260])*90% H=([260])*125% | Average car mileage,[260] assumed fuel economy will improve |
| 55 | Light truck mileage | miles per gal gas | 15.5 | 21.5 | L=([260])*90% H=([260])*125% | Average car mileage,[260] assumed fuel economy will improve |
| 56 | Combination Truck Mileage | miles per gal diesel | 4.9 | 6.8 | L=([267])*90% H=([267])*125% | Average combination truck mileage,[267] assumed fuel economy will improve |
| 57 | Single-Unit Truck mileage | miles per gal diesel | 4.0 | 5.5 | L=([268])*90% H=([268])*125% | Average mileage for garbage trucks less than 40,000 lbs (empty),[268] assumed fuel economy will improve |
| 58 | Garbage Truck mileage | kg diesel per tkm | 0.25 | 0.37 | L=([124])*75% H=([124])*110% | Average mileage for garbage trucks,[124] and assumed lower fuel economy due to stops during collection route |
| 59 | API Chemistry – Carbon | mole carbon | -10 | 10 | L, H= average of range (8-29 moles C) | Number of moles added during API oxidation and reduction calculations (impacting non-API emissions) |
| 60 | API Chemistry – Hydrogen | mole hydrogen | -20 | 20 | L, H= average of range (8-50 moles H) | Number of moles added during API oxidation and reduction calculations (impacting non-API emissions) |
| 61 | API Chemistry – Nitrogen | mole nitrogen | -1 | 1 | L, H= average of range (0-1 moles N) | Number of moles added during API oxidation and reduction calculations (impacting non-API emissions) |
| 62 | API Chemistry – Sulfur | mole sulfur | -1 | 1 | L, H= average of range (0-1 moles S) | Number of moles added during API oxidation and reduction calculations (impacting non-API emissions) |

### C) Sensitivity Analysis

The model results are sensitive to only 3 of the 62 uncertainty parameters (see Table 15): 1) the mass of packaging per mass of API; 2) the number of trips to a pharmacy in one year by each take-back participant; and 3) the heterotrophic yield observed when APIs are the substrate. While the disposal options are sensitive to these parameters, the emission trends of each disposal scenario is unaffected by uncertainty in these parameters.

**Table 15. Emissions categories sensitive (|ρ|>0.8) to a particular uncertainty parameter. Negative and positive signs indicate correlation. Disposal scenarios include: baseline, 100% Trash, 100% Toilet, and 100% Take-back.\***

| | | Uncertainty Parameters | | |
|---|---|---|---|---|
| | | **Mass of packaging per mass API** | **Heterotroph Observed yield** | **Number of trips to pharmacy** |
| **Emissions Categories** | **Acidification** | | | +Take-back |
| | **Carcinogenics** | | | +Take-back |
| | **Ecotoxicity** | +All (baseline, toilet, trash, Take-back) | | |
| | **Eutrophication** | | | +Take-back |
| | **Global Warming** | | -Toilet | +Take-back |
| | **Non carcinogenics** | +Trash | | |
| | **Ozone depletion** | | | +Take-back |
| | **Respiratory effects** | | | +Take-back |
| | **Smog** | | | +Take-back |
| | **API Emissions** | | | |

\*An example for the 100% take-back disposal scenario ("Take-back") that shows how to interpret this table: As the value of the uncertainty parameter "number of trips to a pharmacy" increases, so do acidification emissions because the acidification emissions category is strongly and positively correlated with the number of trips.

# Appendix B

# Supporting Information for Chapter 4

Supporting Information for Chapter 4.

**Stability Indicator Rating Examples**

The following is an example of calculating an indicator rating for pH. A stable pH range is assumed to be $6.1 - 8.3$. A digester with a pH of 6.2 was assigned a pH indicator rating of 0.50, since the digester pH was within the stable range but close to the stability boundary (i.e., within 5% of the minimum value). A digester with a pH of 7 was assigned a pH indicator rating of 1.0 since the pH was in the middle of the stability range (i.e., not within 10% of the maximum or minimum value). These pH indicator ratings convey that a digester with a pH of 7 is more stable than a digester with a pH of 6.2.

The following is an example of assigning indicator ratings for a biogas composition, which only has a minimum stability bound of 55% methane. A digester with a biogas composition of 58% methane is assigned an indicator rating of 0.75, since the biogas composition was assumed to be stable since it was greater than the minimum value but close to the stability boundary (i.e., within 10% of the minimum value). A digester with a biogas composition of 65% methane was assigned an indicator rating of 1.0 because this percentage was much greater than the minimum value (i.e., greater than 110% of the minimum value).

**Uncertainty and Sensitivity Assessment of Stability Indicator Values**

The impact of the stability threshold values for each stability indicator on the stability index rating was tested. Each applicable maximum and minimum value for all stability indicators was assigned an upper and lower value based on data from the literature (Table 16). The Monte Carlo method was applied using uniform probability distributions for each stability threshold value range to randomly generate 1,000 sets of stability threshold values (i.e., maximum and minimum values). All 10,000 model outputs (corresponding to the 10,000 influent compositions) were evaluated using each set of stability bounds to calculate the overall stability index rating (i.e., each influent composition had 1,000 potentially different stability index ratings). For each influent composition, the resulting stability index ratings for all 1,000 outputs (one for each set of stability threshold values) were compared in order to determine the sensitivity of the overall stability index rating to each indicator's uncertain maximum and/or minimum value. The stability index rating was defined as "sensitive" to an uncertain stability bound if the resulting correlation coefficient was significant ($p<0.05$) and strong ($|\rho|>=0.8$), and if the correlation was found for more than 1% all influents modeled. Any identified sensitivity may warrant further testing of the accuracy of the stability threshold value, especially when applied to model outputs.

An assessment of indicator stability bound uncertainty showed that the stability index was robust to this uncertainty (data not shown). In other words, slight changes in the stability indicators' maximum and minimum values are not expected to change the overall stability index rating. The stability index rating was found to be sensitive (the correlation coefficient showed a significant ($p$-value $<0.5$) and strong ($|\rho|>=0.8$) correlation) to the minimum biogas methane composition percentage and to the maximum alkalinity concentration for only 5% and 2% of the influents modeled, respectively. This result suggests that further examination of the minimum biogas methane composition and the maximum alkalinity concentration expected in stable digesters may be needed.

**Table 16. Ranges for stability indicator bounds that are inclusive of uncertainty.**

| Indicator | Units | Minimum | | | | Maximum | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | lower limit | basis | upper limit | basis | lower limit | Basis | upper limit | basis |
| **pH** | (--) | 6.1 [167] | | 6.8 [168] | | 7.4 [168] | | 8.3 [167] | |
| **Alkalinity** | mg CaCO$_3$/L | 2,000* [39,163] | | 2,000 [39,163] | | 5,000 [163] | | 20,000 [35] | |
| **Free ammonia** | mg NH$_3$-N/L | n/a | | | | 150 [31] | | 500 [163] | |
| **Biogas Composition** | Methane Volume/ Biogas Volume (%) | 55 [167] | | 60 [163] | | n/a | | | |
| **COD removal** | % (1-CODout/CODin) | 45 [169] | | 60 [163] | | n/a | | | |
| **Acetate** | mg COD as acetate/L | n/a | | | | 640 [167] | | 850 [170] | |
| **VFA to Alkalinity Ratio** | mg acetate equivalent/ (mg CaCO$_3$ equivalent) | 0 | Absolute minimum | 0.1 [160] | | 0.35 [160] | | 1.0 [36] | |
| **Long Chain Fatty Acids** | mg COD as LCFA/L | n/a | | | | 1,400 [172] | | 6,000 [171] | |
| **VFA$_{C2-C5}$** | mg COD as C2-C5 VFAs/L | n/a | | | | 2,000 [173] | | 3,700 [167] | |
| **Ammonium** | mg NH$_4^+$-N/L | n/a | | | | 5,000 [31] | | 8,000 [163] | |

*The minimum value was found to be consistent in the literature.

**Evaluations of the Benefit of using Multiple Stability Indicators**

The correlation between biogas methane composition values and other model outputs was evaluated (Table 17). The modeling results indicate that VFAs, alkalinity, and ammonium concentrations were not strongly correlated with percent methane biogas composition. Given these results and given that previous work has determined that these parameters are important for effectively assessing stability[28,159–161], we concluded that methane production alone cannot represent overall stability. Thus, developing a stability index that incorporates information from multiple ADM1 outputs to assess overall process stability is important.

**Table 17. Significant Pearson correlation coefficients between percent methane biogas composition and selected ADM1 outputs, which are expected to be related to process stability.**

| ADM1 Outputs | Units | Significant Pearson Correlation Coefficients with Percent Methane |
|---|---|---|
| pH | (--) | 0.93 |
| Alkalinity | mg $CaCO_3$/L | -0.21 |
| Free ammonia | mg $NH_3$-N/L | 0.79 |
| COD removal | (%) | 0.97 |
| Acetate | mg COD as acetate/L | 0.05 |
| VFA to Alkalinity Ratio | mg acetate equivalent/ (mg $CaCO_3$ equivalent) | 0.31 |
| Long Chain Fatty Acids | mg COD as LCFA/L | -0.74 |
| $VFA_{C2-C5}$ | mg COD as C2-C5 VFAs/L | -0.35 |
| Ammonium | mg $NH_4^+$-N/L | 0.47 |

**Table 18. Comparison of total biomass concentrations between stable and unstable digesters for the modeled particulate influents.**

| Description | | Unstable Digesters with Low Protein Loadings | Stable Digesters | Highly Stable Digesters |
|---|---|---|---|---|
| Total Biomass Concentrations (mg COD/L) | Mean | 9.65E+01 | 1.48E+04 | 1.36E+04 |
| | Minimum | 9.65E-35 | 1.89E+03 | 2.43E+03 |
| | 10th percentile | 1.96E-34 | 9.18E+03 | 7.63E+03 |
| | 25th percentile | 2.27E-34 | 1.22E+04 | 1.07E+04 |
| | Mode | 3.00E-34 | 1.52E+04 | 1.39E+04 |
| | 75th percentile | 4.62E-34 | 1.77E+04 | 1.68E+04 |
| | 90th percentile | 1.05E-33 | 1.95E+04 | 1.90E+04 |
| | Maximum | 2.75E+03 | 2.30E+04 | 2.27E+04 |
| Stability Index Rating | | 0.0 | >0.5 | >=0.9 |
| Protein Loading | | <1 kg COD/m$^3$/d | any | any |
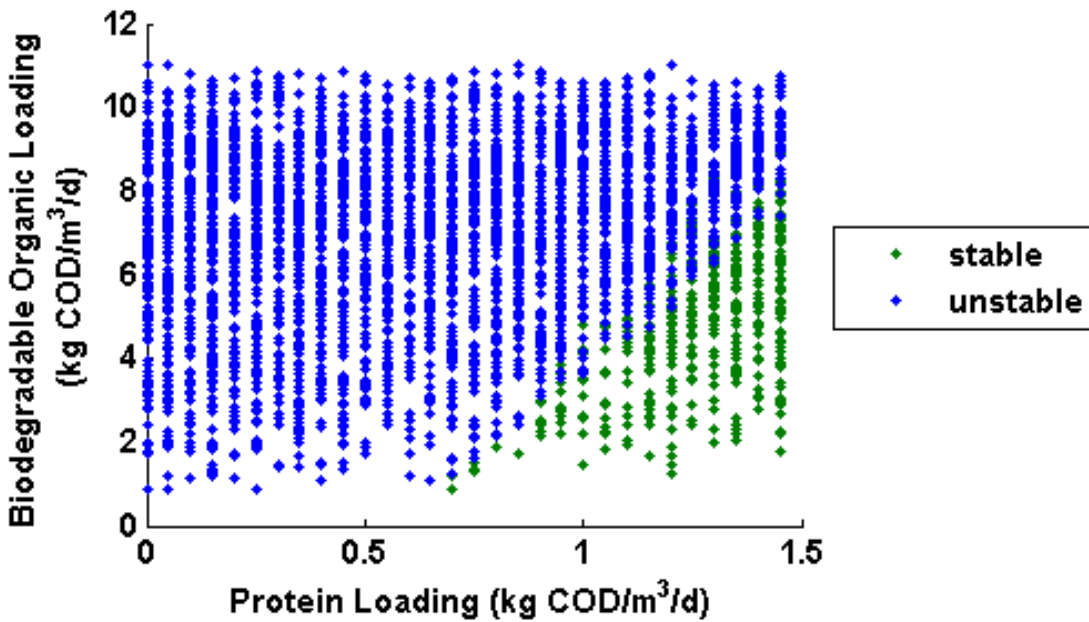


**Figure 16. The relationship between total OLR and protein loading for influent compositions resulting in unstable digestion that was assumed to be due to nitrogen-limited growth.**

**Table 19. Comparison of acetate concentrations between stable and unstable digesters for the modeled particulate influents.**

| Description | | Unstable Digesters with High Protein Loadings | Stable Digesters | Highly Stable Digesters |
|---|---|---|---|---|
| **Acetate (mg/L)** | Mean | 3.18E+04 | 6.91E+03 | 1.42E+02 |
| | Minimum | 8.69E+02 | 5.17E+01 | 5.17E+01 |
| | 10th percentile | 9.87E+03 | 5.90E+01 | 5.49E+01 |
| | 25th percentile | 1.82E+04 | 9.39E+01 | 6.13E+01 |
| | Mode | 3.06E+04 | 5.03E+02 | 9.23E+01 |
| | 75th percentile | 4.34E+04 | 1.14E+04 | 1.62E+02 |
| | 90th percentile | 5.61E+04 | 2.35E+04 | 2.98E+02 |
| | Maximum | 9.27E+04 | 5.42E+04 | 7.86E+02 |
| **Stability Index Rating** | | 0.0 | >0.5 | >=0.9 |
| **Protein Loading** | | >3 kg COD/m3/d | any | any |

**Table 20. Comparison of free ammonia concentrations between stable and unstable digesters for the modeled particulate influents**

| Description | | Unstable Digesters with High Protein Loadings | Stable Digesters | Highly Stable Digesters |
|---|---|---|---|---|
| **Free Ammonia (mg NH3-N/L)** | Mean | 8.02E+01 | 7.36E+01 | 3.49E+01 |
| | Minimum | 6.34E-02 | 3.35E+00 | 3.35E+00 |
| | 10th percentile | 5.08E-01 | 1.10E+01 | 8.45E+00 |
| | 25th percentile | 1.21E+00 | 2.95E+01 | 1.41E+01 |
| | Mode | 1.14E+02 | 8.70E+01 | 3.00E+01 |
| | 75th percentile | 1.16E+02 | 1.16E+02 | 5.12E+01 |
| | 90th percentile | 1.16E+02 | 1.16E+02 | 7.23E+01 |
| | Maximum | 1.17E+02 | 1.17E+02 | 9.67E+01 |
| **Stability Index Rating** | | 0.0 | >0.5 | >=0.9 |
| **Protein Loading** | | >3 kg COD/m3/d | any | any |
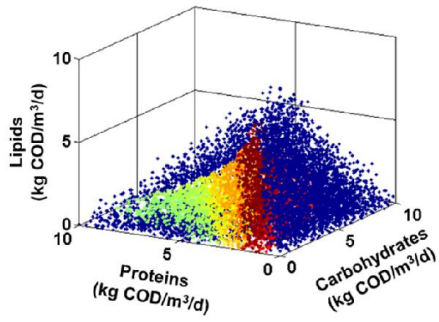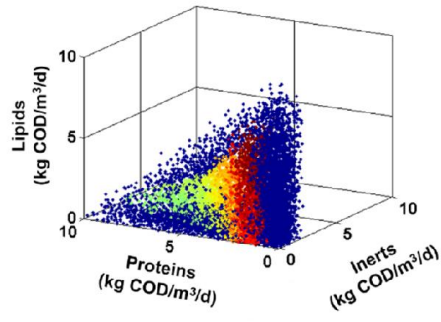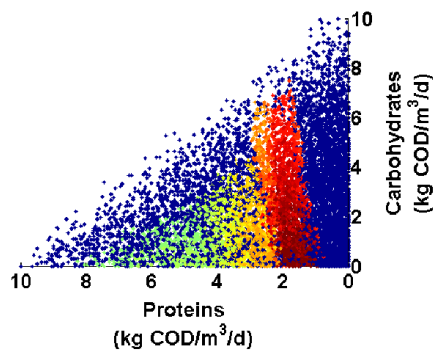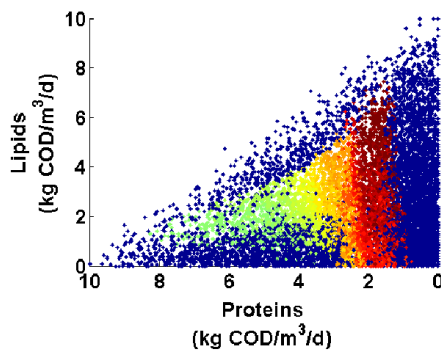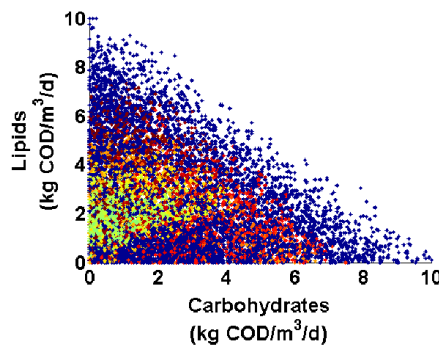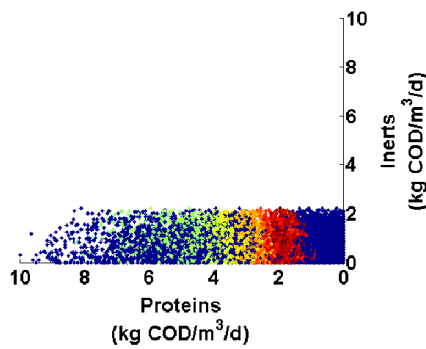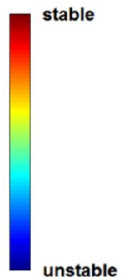
**Figure 17. Stability estimates for particulate influents after modifying the stability index calculation by removing "COD removal" as a stability indicator.**

**Table 21. Characterization of Primary and Waste Activated Sludges as ADM1 inputs.**

| | S_su | S_aa | S_fa | S_ac | S_IN | X_ch | X_pr | X_li | X_I | Citation |
|---|---|---|---|---|---|---|---|---|---|---|
| | kgCOD/ m³ | kgCOD/ m³ | kgCOD/ m³ | kgCOD/ m³ | kmoleN /m³ | kgCOD/ m³ | kgCOD/ m³ | kgCOD/ m³ | kgCOD/ m³ | |
| Primary Sludge | 0.14 | 0.39 | 3.4 | 2.1 | 0.0099 | 10 | 4.9 | 5.3 | | 250 |
| Waste Activated Sludge | 4.8 | 7.9 | 2.8 | 0.00 | 0.0072 | 7.2 | 12 | 4.8 | 35 | 205 |
| Primary and Waste Activated Sludges | 2.5 | 4.2 | 3.1 | 1.0 | 0.0085 | 8.8 | 8.4 | 5.1 | 18 | equal volumes |

**Table 22. Values and uncertainty ranges used when assessing the impact of biochemical parameter uncertainty.**

| Study | Y_su kgCOD/ kg COD | Y_aa kgCOD/ kg COD | Y_fa kgCOD/ kg COD | Y_c4 kgCOD/ kg COD | Y_pro kgCOD/ kg COD | Y_ac kgCOD/ kg COD | Y_h2 kgCOD/ kg COD | k_dis 1/d | k_hyd_ch 1/d |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 1.00E-01 | 8.00E-02 | 6.00E-02 | 6.00E-02 | 4.00E-02 | 5.00E-02 | 6.00E-02 | 5.00E-01 | 1.00E+01 |
| 2 | | | | | | | | | |
| 3 | | | | | | | | | |
| 4 | | | | | | | | 1.00E+00 | |
| 5 | 1.00E-01 | 8.00E-02 | 6.00E-02 | 4.00E-02 | 4.00E-02 | 7.00E-02 | 6.00E-02 | 5.00E-01 | 1.00E+01 |
| 6 | 1.21E-01 | 1.24E-01 | 4.40E-02 | 3.00E-02 | 4.30E-02 | 4.40E-02 | 1.13E-01 | 3.89E-01 | |
| 7 | | | | | | | | | 0.53, 1.1 |
| 8 | | | | | | | | 1.60E-01 | 1.2, 0.95 |
| 9 | | | | | | | | | |
| 10 | 1.50E-01 | 1.50E-01 | 4.50E-02 | | 5.00E-02 | 2.50E-02 | 4.50E-02 | 2.50E-01 | |
| 11 | | | | | | | | 5.50E-01 | |
| 12 | | | | | | | | 1.00E-01 | |
| 13 | | | | | | | | 9.92E-01 | |
| 14 | | | | | | | | | 0.28, 0.19, 0.13 |
| 15 | | | | | | | | | |
| 16 | | | | | | | | | |
| 17 | | | | | | | | 1.00E-01 | |
| **Minimum** | **1.00E-01** | **8.00E-02** | **4.40E-02** | **3.00E-02** | **4.00E-02** | **2.50E-02** | **4.50E-02** | **1.00E-01** | **1.34E-01** |
| **Maximum** | **1.50E-01** | **1.50E-01** | **6.00E-02** | **6.00E-02** | **5.00E-02** | **7.00E-02** | **1.13E-01** | **1.00E+00** | **1.00E+01** |

**Table 22. Continued.**

| Study | k_hyd_pr | k_hyd_li | k_m_su | K_S_su | pH_LL_acidacet | k_m_aa | K_S_aa | k_m_fa | K_S_fa |
|---|---|---|---|---|---|---|---|---|---|
| | 1/d | 1/d | kgCOD/ kgCOD/d | kgCOD/ m3 | -- | kgCOD/ kgCOD/d | kgCOD/ m3 | kgCOD/ kgCOD/d | kgCOD/ m3 |
| 1 | 1.00E+01 | 1.00E+01 | 3.00E+01 | 5.00E-01 | 4.00E+00 | 5.00E+01 | 3.00E-01 | 6.00E+00 | 4.00E-01 |
| 2 | | | | | | | | | |
| 3 | | | | | | | | | |
| 4 | | | | | | | | | |
| 5 | 1.00E+01 | 1.00E+01 | 3.00E+01 | 5.00E-01 | | 5.00E+01 | 3.00E-01 | 6.00E+00 | 4.00E-01 |
| 6 | 1.44E+00 | | | | | | | | |
| 7 | 0.65, 1.0 | | | | | | | | |
| 8 | 1.30E+00 | 7.60E-01 | | | | | | | |
| 9 | | 6.30E-01 | | | | | | | 1.02E-01 |
| 10 | | | 2.67E+01 | 5.00E-02 | 5.00E+00 | 2.67E+01 | 5.00E-02 | 1.33E+01 | 1.00E+00 |
| 11 | | | | | | | | | |
| 12 | | | | | | | | | |
| 13 | | | | | | | | | |
| 14 | 0.68, 0.35, 0.24 | | | 2.07E-01 | | | 2.03E-01 | | |
| 15 | 2.93E+00 | | | | | | | | |
| 16 | | | | | | | | 1.6, 2.0, 1.9, 8.1, 6.4 | 1.2, 4.1, 2.9, 9.2, 3.3 |
| 17 | | | | | | | | | |
| Minimum | 2.42E-01 | 6.30E-01 | 2.67E+01 | 5.00E-02 | 4.00E+00 | 2.67E+01 | 5.00E-02 | 1.59E+00 | 1.02E-01 |
| Maximum | 1.00E+01 | 1.00E+01 | 3.00E+01 | 5.00E-01 | 5.00E+00 | 5.00E+01 | 3.00E-01 | 1.33E+01 | 9.19E+00 |

**Table 22. Continued.**

| Study | K_Ih2_fa kgCOD/ m3 | k_m_c4 kgCOD/ kgCOD/d | k_m_pro kgCOD/ kgCOD/d | K_S_pro kgCOD/ m3 | K_Ih2_pro kgCOD/ m3 | k_m_ac kgCOD/ kgCOD/d | K_S_ac kgCOD/ m3 | K_I_nh3 kmol N/ m3 | pH_LL_ac -- |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 5.00E-06 | 2.00E+01 | 1.30E+01 | 1.00E-01 | 3.50E-06 | 8.00E+00 | 1.50E-01 | 1.80E-03 | 6.00E+00 |
| 2 | | | | | | | | | |
| 3 | | | | | | 9.00E+00 | 6.50E-01 | 2.80E-03 | |
| 4 | | | 9.00E+00 | 2.00E-01 | | 9.00E+00 | | 1.10E-03 | |
| 5 | | 1.00E+01 | 1.10E+01 | 1.00E-01 | | 8.00E+00 | 1.00E-01 | | |
| 6 | | 8.00E+00 | 3.26E+01 | | | 1.97E+01 | 7.89E-01 | | |
| 7 | | | | | | | | | |
| 8 | | | | | | | | | |
| 9 | | | | | | | | | |
| 10 | 3.00E-06 | | 1.20E+01 | 2.00E-02 | 1.00E-06 | 1.48E+01 | 4.00E-02 | 1.21E-03 | 6.30E+00 |
| 11 | | | | | | | | | |
| 12 | | | | | | | | | |
| 13 | | | | | | | | | |
| 14 | | | | | | | | | |
| 15 | | | | 1.97E-02 | | | 4.10E-02 | | |
| 16 | | | | | | | | | |
| 17 | | | | | | | | | |
| **Minimum** | **3.00E-06** | **8.00E+00** | **9.00E+00** | **1.97E-02** | **1.00E-06** | **8.00E+00** | **4.00E-02** | **1.10E-03** | **6.00E+00** |
| **Maximum** | **5.00E-06** | **2.00E+01** | **3.26E+01** | **2.00E-01** | **3.50E-06** | **1.97E+01** | **7.89E-01** | **2.80E-03** | **6.30E+00** |

**Table 22. Continued.**

| Study | k_m_h2 kgCOD/ kgCOD/d | K_S_h2 kgCOD/ m3 | k_dec_Xsu 1/d | k_dec_Xaa 1/d | k_dec_Xfa 1/d | k_dec_Xc4 1/d | k_dec_Xpro 1/d | k_dec_Xac 1/d | k_dec_Xh2 1/d |
|---|---|---|---|---|---|---|---|---|---|
| 1 | 3.50E+01 | 7.00E-06 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.0E-02 | 2.0E-02 |
| 2 | | | | | | | | | |
| 3 | | | | | | | | | |
| 4 | | | | | | | | | |
| 5 | 3.50E+01 | 7.00E-06 | 4.00E-02 | 4.00E-02 | 4.00E-02 | 4.00E-02 | 4.00E-02 | 2.0E-02 | 2.0E-02 |
| 6 | 9.47E+01 | 4.87E-04 | 4.00E-02 | 4.00E-02 | 4.00E-02 | 4.00E-02 | 1.49E-01 | 1.9E-01 | 5.7E-01 |
| 7 | | | | | | | | | |
| 8 | | | | | | | | | |
| 9 | | | | | | | | | |
| 10 | 4.44E+01 | | 8.00E-01 | 8.00E-01 | 6.00E-02 | | 6.00E-02 | 5.0E-02 | 3.0E-01 |
| 11 | | | | | | | | | |
| 12 | | | | | | | | | |
| 13 | | | | | | | | | |
| 14 | | | | | | | | | |
| 15 | | | | | | | | | |
| 16 | | | | | | | | | |
| 17 | | | | | | | | | |
| Minimum | 3.50E+01 | 7.00E-06 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.00E-02 | 2.00E-02 |
| Maximum | 9.47E+01 | 4.87E-04 | 8.00E-01 | 8.00E-01 | 6.00E-02 | 4.00E-02 | 1.49E-01 | 1.87E-01 | 5.74E-01 |

**Table 22. Continued.**

| Study | Citation |
|---|---|
| 1 | [132] |
| 2 | [228] |
| 3 | [203] |
| 4 | [245] |
| 5 | [244] |
| 6 | [234] |
| 7 | [235] |
| 8 | [197] |
| 9 | [236] |
| 10 | [195] |
| 11 | [238] |
| 12 | [239] |
| 13 | [240] |
| 14 | [241] |
| 15 | [242] |
| 16 | [243] |
| 17 | [246] |

**Stability Index Calculation Code (written in MATLAB language)**

%This function determines if a digester is stable or unstable based on stability index calculations

%13 values are output for the FOSsum array for each influent: 12 stability indicator ratings and one weighted average (stability indicator rating)

%overall stability = FOSsum(:,13) from 0 (unstable) to 1 (highly stable)

function[FOSsum, FOSsum_max, FOSsum_work]=fun_FOSsum_calc(effluent, FOSmin, FOSmax, tier) %output = FOSsum array; input = 1 effluent set

%Stability Indicators values loaded from spreadsheet (values available in Table 5)

%FOSsum calculation values

```matlab
numInd = length(FOSmin);

numrating=10;

%Indicator ranges and what percent of range is used for spectrum/bounds

range1=0.05;   %5percent

range2=0.1;    %10percent

%Array of all sub-ranges

subrange=zeros(numrating,numInd);

subrange(1,:)=FOSmin*(1-range2);

subrange(2,:)=FOSmin*(1-range1);

subrange(3,:)=FOSmin;

subrange(4,:)=FOSmin*(1+range1);

subrange(5,:)=FOSmin*(1+range2);

subrange(6,:)=FOSmax*(1-range2);

subrange(7,:)=FOSmax*(1-range1);

subrange(8,:)=FOSmax;

subrange(9,:)=FOSmax*(1+range1);

subrange(10,:)=FOSmax*(1+range2);

%Array of all ratings; Points designating how close value is to the center of the FOS range

in_center=1.0;
```

in_range2=0.75;        %within in x% (range2=x%) of boundary line or "2steps away from the min or max"

in_range1=0.5;

out_range1=0.25;

out_range2=0.0;

out=0;

rating=[out_range2 out_range1 in_range1 in_range2 in_center in_range2 in_range1 out_range1 out_range2 out]';

rating=repmat(rating, numrating, numInd);

%Adjustments for stability indicators that only have a minimum or maximum value

for i=1:numInd

  if FOSmax(1,i)==0%Minimum only - set higher values equal to min+FOSrange*range1 (greater than this then "center" rating)

    subrange(6:10,i)=subrange(5,i);

    rating(6:10,i)=rating(5,i);

  end

  if FOSmin(1,i)==0 %Maximum only; "center" rating if less than max-FOSrang*range2

    subrange(1:5,i)=FOSmin(i);

    rating(1:5,i)=in_center;

  end

end

%Tier weights

weights=ones(1,numInd); %tier1 weight = 1

weights(find(tier==2)) = .5; %tier2 weight

weights(find(tier==3)) = 0; %tier3 weight

FOSsum_work = 0.5; %min value of a working digester

FOSsum_max = 1;

% FOSsum_min=0;

%Indicators

indicators = [effluent(62) effluent(27) effluent(58:59) effluent(64) effluent(65) effluent(61) effluent(55:57) effluent(60) effluent(66)];

%Working definition

FOSsum = zeros(1,numInd+1);

for j=1:numInd

    %Determine rating for each indicator

    for i=1:5

        if indicators(j) >= subrange(i,j) %if value = min; then rating is in_range1

            FOSsum(j)=rating(i,j);

        end

    end

    for i=6:10 %logic switches when evaluating upper bounds

```
        if indicators(j) > subrange(i,j) %if value = max; then rating is in_range1

            FOSsum(j)=rating(i,j);

        end

    end

end

FOSsum(numInd+1)  =  sum(FOSsum(1,1:numInd).*weights)/sum(weights);  %weighted  and
normalized sum

num_outbounds=length( find(  FOSsum(find(tier==1)) <0.5));

numInd_weight = length(find(tier==1));

if num_outbounds>=numInd_weight/2; %at least half of tier 1 need to be within bound or it is
completely unstable

    FOSsum(numInd+1)=0;

end

end
```

# References

# References

(1)     ASCE. *Report Card for America's Infrastructure*; 2009.

(2)     MacMillan, L. Those Snowy Slopes , Sprayed With Wastewater. *New Yrok Times*, 2012.

(3)     Høibye, L.; Clauson-Kaas, J.; Wenzel, H.; Larsen, H. F.; Jacobsen, B. N.; Dalgaard, O. Sustainability assessment of advanced wastewater treatment technologies. *Water Sci. Technol.* **2008**, *58*, 963–968.

(4)     Wenzel, H.; Larsen, H. F.; Clauson-Kaas, J.; Høibye, L.; Jacobsen, B. N. Weighing environmental advantages and disadvantages of advanced wastewater treatment of micro-pollutants using environmental life cycle assessment. *Water Sci. Technol.* **2008**, *57*, 27–32.

(5)     Guest, J. S.; Skerlos, S. J.; Barnard, J. L.; Beck, M. B.; Daigger, G. T.; Hilger, H.; Jackson, S. J.; Karvazy, K.; Kelly, L.; Macpherson, L.; et al. A new planning and design paradigm to achieve sustainable resource recovery from wastewater. *Environ. Sci. Technol.* **2009**, *43*, 6126–6130.

(6)     Verstraete, W.; Van de Caveye, P.; Diamantis, V. Maximum use of resources present in domestic "used water."*Bioresour. Technol.* **2009**, *100*, 5537–5545.

(7)     Zeeman, G.; Kujawa, K.; de Mes, T.; Hernandez, L.; de Graaff, M.; Abu-Ghunmi, L.; Mels, A.; Meulman, B.; Temmink, H.; Buisman, C.; et al. Anaerobic treatment as a core technology for energy, nutrients and water recovery from source-separated domestic waste(water). *Water Sci. Technol.* **2008**, *57*, 1207–1212.

(8)     McCarty, P. L. The development of anaerobic treatment and its future. *Water Sci. Technol.* **2001**, *44*, 149–156.

(9)     Cakir, F. Y.; Stenstrom, M. K. Greenhouse gas production: a comparison between aerobic and anaerobic wastewater treatment technology. *Water Res.* **2005**, *39*, 4197–4203.

(10)    Pasqualino, J. C.; Meneses, M.; Abella, M.; Castells, F. LCA as a decision support Tool for the Environmental Improvement of the Operation of a Municipal Wastewater Treatment Plant. *Environ. Sci. Technol.* **2009**, *43*, 3300–3307.

(11)    Gallego, A.; Hospido, A.; Moreira, M. T.; Feijoo, G. Environmental performance of wastewater treatment plants for small populations. *Resour. Conserv. Recycl.* **2008**, *52*, 931–940.

(12)    Corominas, L.; Foley, J.; Guest, J. S.; Hospido, A.; Larsen, H. F.; Morera, S.; Shaw, A. Life cycle assessment applied to wastewater treatment: state of the art. *Water Res.* **2013**, *47*, 5480–5492.

(13)    Roeleveld, P. J.; Klapwijk, A.; Eggels, P. G.; Rulkens, W. H.; van Starkenburg, W. Sustainability of municipal waste water treatment. *Water Sci. Technol.* **1997**, *35*, 221–228.

(14)    Jeswani, H. K.; Azapagic, A.; Schepelmann, P.; Ritthoff, M. Options for broadening and deepening the LCA approaches. *J. Clean. Prod.* **2010**, *18*, 120–127.

(15)    Muñoz, I.; José Gómez, M.; Molina-Díaz, A.; Huijbregts, M. a J.; Fernández-Alba, A. R.; García-Calvo, E. Ranking potential impacts of priority and emerging pollutants in urban wastewater through life cycle impact assessment. *Chemosphere* **2008**, *74*, 37–44.

(16)    Carlsson, C.; Johansson, A.-K.; Alvan, G.; Bergman, K.; Kühler, T. Are pharmaceuticals potent environmental pollutants? Part I: environmental risk assessments of selected active pharmaceutical ingredients. *Sci. Total Environ.* **2006**, *364*, 67 – 87.

(17)    Kärrman, E. Strategies towards sustainable wastewater management. *Urban water* **2001**, *3*, 63–72.

(18)    Glassmeyer, S. T.; Hinchey, E. K.; Boehme, S. E.; Daughton, C. G.; Ruhoy, I. S.; Conerly, O.; Daniels, R. L.; Lauer, L.; McCarthy, M.; Nettesheim, T. G.; et al. Disposal practices for unwanted residential medications in the United States. *Environ. Int.* **2009**, *35*, 566–572.

(19)    Kuspis, D. A.; Krenzelok, E. P. What happens to expired medications? A survey of community medication disposal. *Vet. Hum. Toxicol.* **1996**, *38*, 48–50.

(20)    McCarty, P. L.; Bae, J.; Kim, J. Domestic wastewater treatment as a net energy producer--can this be achieved? *Environ. Sci. Technol.* **2011**, *45*, 7100–7106.

(21)    Gray, D. M. D.; Suto, P.; Peck, C. *Anaerobic Digestion of Food Waste*; Oakland, CA, 2008.

(22)    Cook, S. M.; Guest, J. S.; Skerlos, S. J.; Love, N. G. Environmental characteristics of different energy recovery systems managing sewage sludge and food waste. In *12th International Water Association Sludge Conference –Sustainable Management & Technologies of Sludges*; Harbin, China, 2009.

(23)    Peters, G. M.; Rowley, H. V. Environmental comparison of biosolids management systems using life cycle assessment. *Environ. Sci. Technol.* **2009**, *43*, 2674–2679.

(24)    Rankin, M. J.; Trabold, T. A.; Blythe, R. Environmental impact asessment of an anaerobic codigesiton system in western new york state. In *Proceedings of the ASME 2013 7th International Conference on Energy Sustainability*; Minneapolis, MN, 2013; pp. 1–8.

(25)    Pavan, P.; Bolzonella, D.; Battistoni, E.; Cecchi, F. Anaerobic co-digestion of sludge with other organic wastes in small wastewater treatment plants: an economic considerations evaluation. *Water Sci. Technol.* **2007**, *56*, 45–53.

(26)    Edelmann, W.; Baier, U.; Engeli, H. Environmental aspects of the anaerobic digestion of the organic fraction of municipal solid wastes and of solid agricultural wastes. *Water Sci. Technol.* **2005**, *52*, 203–208.

(27)    Hess, J.; Bernard, O. Advanced dynamical risk analysis for monitoring anaerobic digestion process. *Biotechnol. Prog.* **2009**, *25*, 643–653.

(28)    Dong, F.; Zhao, Q.-B.; Li, W.-W.; Sheng, G.-P.; Zhao, J.-B.; Tang, Y.; Yu, H.-Q.; Kubota, K.; Li, Y.-Y.; Harada, H. Novel online monitoring and alert system for anaerobic digestion reactors. *Environ. Sci. Technol.* **2011**, *45*, 9093–9100.

(29)    Parry, D. *Co-Digestion of Organic Waste Products with Wastewater Solids*; Alexandria, VA, 2010.

(30)    Labatut, R. A.; Angenent, L. T.; Scott, N. R. Biochemical methane potential and biodegradability of complex organic substrates. *Bioresour. Technol.* **2011**, *102*, 2255–2264.

(31)    Parameswaran, P.; Rittmann, B. E. Feasibility of anaerobic co-digestion of pig waste and paper sludge. *Bioresour. Technol.* **2012**, *124*, 163–168.

(32)    Ferrer, P.; Cambra-López, M.; Cerisuelo, A.; Peñaranda, D. S.; Moset, V. The use of agricultural substrates to improve methane yield in anaerobic co-digestion with pig slurry: Effect of substrate type and inclusion level. *Waste Manag.* **2014**, *34*, 196–203.

(33)    Safferman, S. I.; Faivor, L. L.; Wall, D. M. Anaerobic Digestion Biogas Potential in Support of Renewable Energy Development. In *Proceedings of the Water Environment Federation Technical Exhibition and Conference (WEFTEC)*; Orlando, FL, 2009; Vol. 2009, pp. 4796–4809.

(34)    Fernández, A.; Sánchez, A.; Font, X. Anaerobic co-digestion of a simulated organic fraction of municipal solid wastes and fats of animal and vegetable origin. *Biochem. Eng. J.* **2005**, *26*, 22–28.

(35)    Murto, M.; Björnsson, L.; Mattiasson, B. Impact of food industrial waste on anaerobic co-digestion of sewage sludge and pig manure. *J. Environ. Manage.* **2004**, *70*, 101–107.

(36)    Stroot, P. G.; McMahon, K. D.; Mackie, R. I.; Raskin, L. Anaerobic codigestion of municipal solid waste and biosolids under various mixing conditions--I. Digester performance. *Water Res.* **2001**, *35*, 1804–1816.

(37)    Callaghan, F.; Wase, D. Continuous co-digestion of cattle slurry with fruit and vegetable wastes and chicken manure. *Biomass and Bioenergy* **2002**, *27*, 71–77.

(38) Zaher, U.; Li, R.; Jeppsson, U.; Steyer, J. P.; Chen, S. GISCOD: general integrated solid waste co-digestion model. *Water Res.* **2009**, *43*, 2717–2727.

(39) Alvarez, J. a; Otero, L.; Lema, J. M. A methodology for optimising feed composition for anaerobic co-digestion of agro-industrial wastes. *Bioresour. Technol.* **2010**, *101*, 1153–1158.

(40) Balmant, W.; Oliveira, B. H.; Mitchell, D. a.; Vargas, J. V. C.; Ordonez, J. C. Optimal operating conditions for maximum biogas production in anaerobic bioreactors. *Appl. Therm. Eng.* **2014**, *62*, 197–206.

(41) International Organization for Standardization. *Environmental management - life cycle assessment - principles and framework (ISO 14040)*; Geneva, Switzerland, 1997.

(42) Gros, M.; Petrović, M.; Ginebreda, A.; Barceló, D. Removal of pharmaceuticals during wastewater treatment and environmental risk assessment using hazard indexes. *Environ. Int.* **2010**, *36*, 15–26.

(43) Ternes, T. A.; Joss, A.; Siegrist, H. Scrutinizing pharmaceuticals and personal care products in wastewater treatment. *Environ. Sci. Technol.* **2004**, *38*, 392A–399A.

(44) Kolpin, D. W.; Furlong, E. T.; Meyer, M. T.; Thurman, E. M.; Zaugg, S. D.; Barber, L. B.; Buxton, H. T. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999-2000: a national reconnaissance. *Environ. Sci. Technol.* **2002**, *36*, 1202–1211.

(45) Kinney, C. A.; Furlong, E. T.; Zaugg, S. D.; Burkhardt, M. R.; Werner, S. L.; Cahill, J. D.; Jorgensen, G. R. Survey of organic wastewater contaminants in biosolids destined for land application. *Environ. Sci. Technol.* **2006**, *40*, 7207–7215.

(46) Benotti, M. J.; Trenholm, R. A.; Vanderford, B. J.; Holady, J. C.; Stanford, B. D.; Snyder, S. A. Pharmaceuticals and endocrine disrupting compounds in U.S. drinking water. *Environ. Sci. Technol.* **2009**, *43*, 597–603.

(47) Hospido, A.; Carballa, M.; Moreira, M.; Omil, F.; Lema, J. M.; Feijoo, G. Environmental assessment of anaerobically digested sludge reuse in agriculture: Potential impacts of emerging micropollutants. *Water Res.* **2010**, *44*, 3225–3233.

(48) Kidd, K. A.; Blanchfield, P. J.; Mills, K. H.; Palace, V. P.; Evans, R. E.; Lazorchak, J. M.; Flick, R. W. Collapse of a fish population after exposure to a synthetic estrogen. *Proc. Natl. Acad. Sci. U. S. A.* **2007**, *104*, 8897–8901.

(49) Pomati, F.; Castiglioni, S.; Zuccato, E.; Fanelli, R.; Vigetti, D.; Rossetti, C.; Calamari, D. Effects of a complex mixture of therapeutic drugs at environmental levels on human embryonic cells. *Environ. Sci. Technol.* **2006**, *40*, 2442–2447.

(50) Escher, B. I.; Fenner, K. Recent advances in environmental risk assessment of transformation products. *Environ. Sci. Technol.* **2011**, *45*, 3835–3847.

(51) Joss, A.; Siegrist, H.; Ternes, T. A. Are we about to upgrade wastewater treatment for removing organic micropollutants? *Water Sci. Technol.* **2008**, *57*, 251–255.

(52) Kümmerer, K. *Pharmaceuticals in the Environment: Sources, Fate, Effects and Risks*; Kümmerer, K., Ed.; 3rd ed.; Springer: Hiedelberg, 2004.

(53) Daughton, C. G.; Ruhoy, I. S. Environmental footprint of pharmaceuticals: The significance of factors beyond direct excretion to sewers. *Environ. Toxicol. Chem.* **2009**, *28*, 2495–2521.

(54) Ruhoy, I. S.; Daughton, C. G. Types and quantities of leftover drugs entering the environment via disposal to sewage-Revealed by coroner records. *Sci. Total Environ.* **2007**, *388*, 137–148.

(55) Ruhoy, I. S.; Daughton, C. G. Beyond the medicine cabinet: An analysis of where and why medications accumulate. *Environ. Int.* **2008**, *34*, 1157–1169.

(56) Daughton, C. G.; Ruhoy, I. S. Green pharmacy and pharmEcovigilance: prescribing and the planet. *Expert Rev. Clin. Pharmacol.* **2011**, *4*, 211–232.

(57) Daughton, C. G. Comment on "life cycle comparison of environmental emissions from three disposal options for unused pharmaceuticals."*Environ. Sci. Technol.* **2012**, *46*, 8519–8520.

(58) Cunningham, V. L. Special characteristics of pharmaceuticals related to environmental fate. In *Pharmaceuticals in the Environment: Sources, Fate, Effects and Risks*; Kümmerer, K., Ed.; Springer: New York, NY, 2004; p. 13.

(59) Daughton, C. G. *Drugs and the Environment: Stewardship & Sustainability*; Las Vegas, NV, 2010; p. 196.

(60) U.S. Food and Drug Administration. *Disposal of unused medicines: What you should know*; Silver Spring, MD, 2012.

(61) *Secure and responsible drug disposal act of 2010*; 111th Congress of the United States of America, 2010; p. Public Law 111–273.

(62) *Safe Pharmaceutical Disposal Act*; People of the State of Illinois, 2010; p. Public Act 096–0221.

(63) Garcia, M. Quinn signs bill in support of drop boxes for unused pills: Initiative aimed at keeping medicines out of water supply. *Chicago Tribune*, 2011.

(64) LaFontaine; Denby; Franz; Lori; Lyons; Price; Haveman; Lauwers; Graves; Schor; et al. *Enrolled House Bill No. 5005*; State of Michigan (97th Legislature): Lansing, MI, 2014.

(65)   Lauer, G. Drug Take-Back Bill Faces Opposition. *California Healthline*, 2014.

(66)   Grasso, C.; The Pharmaceuticals from Households: A Return Mechanism (PH:ARM) Pilot. *Secure medicine return in Washington state*; Seattle, WA, 2010; p. 60.

(67)   Johnson, E.; The Pharmaceuticals from Households: A Return Mechanism (PH:ARM) Pilot. *Disposal of medications from residential consumers: issues, barriers and opportunities*; WA, 2007; p. 35.

(68)   Hubbard, M. *Oregon pharmaceutical take back stakeholder group final report*; OR, 2007.

(69)   Teleosis Institute. *Green pharmacy program: helping communities safely dispose of unused medicine*; Berkeley, CA, 2008.

(70)   Kaye, L.; Critteden, J.; Gressit, S. *Reducing prescription drug misuse through the use of a citizen mail-back program in Maine -- Safe medicine disposal for ME: A handbook and summary report*; Bangor, Maine, 2010; p. 219.

(71)   Take Back Your Meds http://www.takebackyourmeds.org/what-you-can-do/other-take-back-locations (accessed May 7, 2014).

(72)   eWashtenaw.                                          www.dontflushdrugs.com! http://www.ewashtenaw.org/government/departments/environmental_health/recycling_home_toxics/medication_disposal/frontpage (accessed May 7, 2014).

(73)   Great Lakes Clean Water Organization. Yellow Jug Old Drugs Program.

(74)   California Department of Resources Recycling and Recovery. Facility Information Toolbox: Disposal Directory for Sharps and Medication http://www.calrecycle.ca.gov/FacIT/Facility/Search.aspx (accessed May 7, 2014).

(75)   National Community Pharmacists Association (NCPA) Foundation. DisposeMyMeds http://www.disposemymeds.org/ (accessed May 7, 2014).

(76)   U.S. Drug Enforcement Administration. National Take-Back Initiative http://www.deadiversion.usdoj.gov/drug_disposal/takeback/ (accessed May 5, 2014).

(77)   DEA Public Affairs. Americans Turn Out in Droves for DEA's Seventh National Prescription Drug Take-Back Day. *Headquaters News*, 2013.

(78)   Kroger. Medication Disposal: How to Dispose of Medications Properly https://www.kroger.com/topic/medication-disposal (accessed May 7, 2014).

(79)   Chain Drug Review. Rite Aid facilitates safe drug disposal, 2011.

(80)   CVS. CVS/pharmacy Offers Customers Safe Disposal Program for Medications. *2011*.

(81)     Walgreens. Walgreens Pharmacy Resources: Biohazard Disposal http://www.walgreens.com/topic/help/pharmacyhelp/pharmacy_reference_help_main.jsp# BiohazardDisposal (accessed May 7, 2014).

(82)     Walgreens. Walgreens Safe Medication Disposal Program http://www.walgreens.com/topic/sr/sr_safe_medication_disposal.jsp (accessed May 7, 2014).

(83)     Washtenaw Community Health Organization. Big Red Barrel Project http://redbarrelinitiative.wix.com/onlinetoolkit (accessed May 7, 2014).

(84)     Titz, A.; Döll, P. Actor modelling and its contribution to the development of integrative strategies for management of pharmaceuticals in drinking water. *Soc. Sci. Med.* **2009**, *68*, 672–681.

(85)     Tong, A. Y. C.; Peake, B. M.; Braund, R. Disposal practices for unused medications around the world. *Environ. Int.* **2011**, *37*, 292–298.

(86)     Kotchen, M.; Kallaos, J.; Wheeler, K.; Wong, C.; Zahller, M. Pharmaceuticals in wastewater: behavior, preferences, and willingness to pay for a disposal program. *J. Environ. Manage.* **2009**, *90*, 1476–1482.

(87)     Daughton, C. G. Cradle-to-cradle stewardship of drugs for minimizing their environmental disposition while promoting human health. II. Drug disposal, waste reduction, and future directions. *Environ. Health Perspect.* **2003**, *111*, 775–785.

(88)     Kasprzyk-Hordern, B.; Dinsdale, R. M.; Guwy, A. J. The removal of pharmaceuticals, personal care products, endocrine disruptors and illicit drugs during wastewater treatment and its impact on the quality of receiving waters. *Water Res.* **2009**, *43*, 363–380.

(89)     Khan, S. J.; Ongerth, J. E. Modelling of pharmaceutical residues in Australian sewage by quantities of use and fugacity calculations. *Chemosphere* **2004**, *54*, 355–367.

(90)     Jones, O. A. H.; Voulvoulis, N.; Lester, J. N. Aquatic environmental assessment of the top 25 English prescription pharmaceuticals. *Water Res.* **2002**, *36*, 5013–5022.

(91)     Stadler, L.; Ernstoff, A.; Aga, D.; Love, N. G. Micropollutant fate in wastewater treatment: Redefining "removal."*Environ. Sci. Technol.* **2012**, 10485–10486.

(92)     Drewes, J. E.; Sedlak, D.; Snyder, S.; Dickenson, E.; WateRuse Foundation. *Development of indicators and surrogates for chemical contaminant removal during wastewater treatment and reclamation*; Alexandria, VA, 2008.

(93)     Onesis, K. M.; Yu, J. T.; Bouwer, E. J. Biodegradation and removal of pharmaceuticals and personal care products in treatment systems: a review. *Biodegradation* **2009**, *20*, 441–466.

(94) Eganhouse, R. P.; Kaplan, I. R. a-Tocopheryl acetate as an indicator of municipal waste contamination in the environment. *Environ. Sci. Technol.* **1985**, *19*, 282–285.

(95) Harrold, K. H.; Yee, D.; Sedlak, M.; Klosterhaus, S.; Davis, J. A.; Woudneh, M.; Riley, P.; San Francisco Estuary Institute. *Pharmaceuticals and personal care products in wastewater treatment plant influent and effluent and surface waters of lower South San Francisco Bay*; Oakland, CA, 2009.

(96) Chang, H.; Hu, J.; Shao, B. Occurrence of natural and synthetic glucocorticoids in sewage treatment plants and receiving river waters. *Environ. Sci. Technol.* **2007**, *41*, 3462–3468.

(97) POSEIDON. *Assessment of technologies for the removal of pharmaceuticals and personal care products in sewage and drinking water facilities to improve the indirect potable water reuse*; Wiesbaden, Germany, 2006; p. 61.

(98) Wishart, D. S.; Knox, C.; Guo, A. C.; Cheng, D.; Shrivastava, S.; Tzur, D.; Gautam, B.; Hassanali, M. DrugBank: a knowledgebase for drugs, drug actions and drug targets. *Nucleic Acids Res.* **2008**, *36*, D901–D906.

(99) U.S. National Library of Medicine. *ChemIDplus advanced*; National Institutes of Health: Bethesda, MD, 2009.

(100) ACD/Labs. *Advanced chemistry development software*; Toronto, Canada, 2010; Vol. 8.14.

(101) Tetko, I. V; Gasteiger, J.; Todeschini, R.; Mauri, A.; Livingstone, D.; Ertl, P.; Palyulin, V. A.; Radchenko, E. V; Zefirov, N. S.; Makarenko, A. S.; et al. Virtual computational chemistry laboratory-design and description. *J. Comput. Aided. Mol. Des.* **2005**, *19*, 453–463.

(102) Nguyen, T. H.; Goss, K.-U.; Ball, W. P. Polyparameter linear free energy relationships for estimating the equilibrium partition of organic compounds between water and the natural organic matter in soils and sediments. *Environ. Sci. Technol.* **2005**, *39*, 913–924.

(103) Goss, K.-U.; Schwarzenbach, R. P. Linear free energy relationships used to evaluate equilibrium partitioning of organic compounds. *Environ. Sci. Technol.* **2001**, *35*, 1–9.

(104) Zukowska, B.; Breivik, K.; Wania, F. Evaluating the environmental fate of pharmaceuticals using a level III model based on poly-parameter linear free energy relationships. *Sci. Total Environ.* **2006**, *359*, 177 – 187.

(105) Tülp, H. C.; Goss, K.-U.; Schwarzenbach, R. P.; Fenner, K. Experimental determination of LSER parameters for a set of 76 diverse pesticides and pharmaceuticals. *Environ. Sci. Technol.* **2008**, *42*, 2034–2040.

(106) Chong, N.-M.; Tsai, S.-C.; Le, T. N. The biomass yielding process of xenobiotic degradation. *Bioresour. Technol.* **2010**, *101*, 4337–4342.

(107) Ternes, T. A.; Herrmann, N.; Bonerz, M.; Knacker, T.; Siegrist, H.; Joss, A. A rapid method to measure the solid-water distribution coefficient (Kd) for pharmaceuticals and musk fragrances in sewage sludge. *Water Res.* **2004**, *38*, 4075–4084.

(108) Helbling, D. E.; Hollender, J.; Kohler, H.-P. E.; Singer, H.; Fenner, K. High-throughput identification of microbial transformation products of organic micropollutants. *Environ. Sci. Technol.* **2010**, *44*, 6621–6627.

(109) Fatta-Kassinos, D.; Vasquez, M. I.; Kümmerer, K. Transformation products of pharmaceuticals in surface waters and wastewater formed during photolysis and advanced oxidation processes - degradation, elucidation of byproducts and assessment of their biological potency. *Chemosphere* **2011**, *85*, 693–709.

(110) Quintana, J. B.; Weiss, S.; Reemtsma, T. Pathways and metabolites of microbial degradation of selected acidic pharmaceutical and their occurrence in municipal wastewater treated by a membrane bioreactor. *Water Res.* **2005**, *39*, 2654–2664.

(111) Radjenović, J.; Petrović, M.; Barceló, D. Complementary mass spectrometry and bioassays for evaluating pharmaceutical-transformation products in treatment of drinking water and wastewater. *TrAC Trends Anal. Chem.* **2009**, *28*, 562–580.

(112) Kümmerer, K.; Al-Ahmad, A.; Mersch-sundermann, V. Biodegradability of some antibiotics, elimination of the genotoxicity and a ection of wastewater bacteria in a simple test. *Chemosphere* **2000**, *40*, 701–710.

(113) Al-Ahmad, A.; Daschner, F.; Kümmerer, K. Biodegradability of cefotiam, ciprofloxacin, meropenem, penicillin G, and sulfamethoxazole and inhibition of waste water bacteria. *Arch. Environ. …* **1999**, *163*, 158–163.

(114) Musson, S. E.; Campo, P.; Tolaymat, T.; Suidan, M.; Townsend, T. G. Assessment of the anaerobic degradation of six active pharmaceutical ingredients. *Sci. Total Environ.* **2010**, *408*, 2068–2074.

(115) Carballa, M.; Omil, F.; Ternes, T.; Lema, J. M. Fate of pharmaceutical and personal care products (PPCPs) during anaerobic digestion of sewage sludge. *Water Res.* **2007**, *41*, 2139–2150.

(116) Helbling, D.; Johnson, D.; Honti, M.; Fenner, K. Micropollutant biotransformation kinetics associate with WWTP process parameters and microbial community characteristics. *Environ. Sci. …* **2012**, *46*, 10579–10588.

(117) Joss, A.; Zabczynski, S.; Göbel, A.; Hoffmann, B.; Löffler, D.; McArdell, C. S.; Ternes, T. A.; Thomsen, A.; Siegrist, H. Biological degradation of pharmaceuticals in municipal wastewater treatment: proposing a classification scheme. *Water Res.* **2006**, *40*, 1686–1696.

(118) OECD. *OECD Guildlines for the Testing of Chemicals*; Paris, France, 2006; pp. 1–49.

(119) EPA. *BIOWIN Model*; Washington, D.C., 2012.

(120) Tran, N. H.; Urase, T.; Ngo, H. H.; Hu, J.; Ong, S. L. Insight into metabolic and cometabolic activities of autotrophic and heterotrophic microorganisms in the biodegradation of emerging trace organic contaminants. *Bioresour. Technol.* **2013**, *146*, 721–731.

(121) Li, D.; Alidina, M.; Drewes, J. E. Role of primary substrate composition on microbial community structure and function and trace organic chemical attenuation in managed aquifer recharge systems. *Appl. Microbiol. Biotechnol.* **2014**.

(122) Tan, D. T.; Arnold, W. a; Novak, P. J. Impact of organic carbon on the biodegradation of estrone in mixed culture systems. *Environ. Sci. Technol.* **2013**, *47*, 12359–12365.

(123) Alidina, M.; Li, D.; Drewes, J. E. Investigating the role for adaptation of the microbial community to transform trace organic chemicals during managed aquifer recharge. *Water Res.* **2014**, *56*, 172–180.

(124) Swiss Centre for Life Cycle Inventories. *Ecoinvent database*; Düebendorf, Switzerland, 2007; Vol. 2.0.

(125) Franklin Associates. *Franklin US life cycle inventory 98 library*; Prairie Village, Kansas, 1998.

(126) National Renewable Energy Laboratory. *U.S. life cycle inventory database*; Golden, CO, 2008; Vol. 1.6.0.

(127) U.S. Environmental Protection Agency. *Tool for the reduction and assessment of chemical and other environmental impacts (TRACI)*; Cincinnati, OH, 2010; Vol. 2.0.

(128) Wernet, G.; Conradt, S.; Isenring, H. P.; Jiménez-González, C.; Hungerbühler, K. Life cycle assessment of fine chemical production: a case study of pharmaceutical synthesis. *Int. J. Life Cycle Assess.* **2010**, *15*, 294–303.

(129) Jiménez-González, C.; Curzons, A. D.; Constable, D. J. C.; Cunningham, V. L. Cradle-to-gate life cycle inventory and assessment of pharmaceutical compounds. *Int. J. Life Cycle Assess.* **2004**, *9*, 114–121.

(130) Northeast Recycling Council Inc. *Managing pharmaceutical waste: best management practices for plastic medication containers from consumers*; Brattleboro, VT, 2005.

(131) Daughton, C. G. Cradle-to-cradle stewardship of drugs for minimizing their environmental disposition while promoting human health. I. Rationale for and avenues toward a green pharmacy. *Environ. Health Perspect.* **2003**, *111*, 757–774.

(132) Batstone, D. J.; Keller, J.; Angelidaki, I.; Kalyuzhnyi, S. V.; Pavlostathis, S. G.; Rozzi, A.; Sanders, W. T. M.; Siegrist, H.; Vavilin, V. A. *IWA Anaerobic Digestion Model No. 1*

*(ADM1)*; IWA Scient.; IWA Publishing: London, 2002; p. IWA Scientific and Technical Report No. 13.

(133) Werner, J. J.; Knights, D.; Garcia, M. L.; Scalfone, N. B.; Smith, S.; Yarasheski, K.; Cummings, T. A.; Beers, A. R.; Knight, R.; Angenent, L. T. Bacterial community structures are unique and resilient in full-scale bioenergy systems. *Proc. Natl. Acad. Sci. U. S. A.* **2011**, *108*, 4158–4163.

(134) Hansen, K. H.; Ahring, B. K.; Raskin, L. Quantification of syntrophic fatty acid-beta-oxidizing bacteria in a mesophilic biogas reactor by oligonucleotide probe hybridization. *Appl. Environ. Microbiol.* **1999**, *65*, 4767–4774.

(135) Stams, a J.; Van Dijk, J. B.; Dijkema, C.; Plugge, C. M. Growth of syntrophic propionate-oxidizing bacteria with fumarate in the absence of methanogenic bacteria. *Appl. Environ. Microbiol.* **1993**, *59*, 1114–1119.

(136) Kim, H. W.; Han, S. K.; Shin, H. S. Anaerobic co-digestion of sewage sludge and food waste using temperature-phased anaerobic digestion process. *Water Sci. Technol.* **2004**, *50*, 107–114.

(137) Wang, X.; Yang, G.; Feng, Y.; Ren, G.; Han, X. Optimizing feeding composition and carbon-nitrogen ratios for improved methane yield during anaerobic co-digestion of dairy, chicken manure and wheat straw. *Bioresour. Technol.* **2012**, *120*, 78–83.

(138) Astals, S.; Nolla-Ardèvol, V.; Mata-Alvarez, J. Anaerobic co-digestion of pig manure and crude glycerol at mesophilic conditions: biogas and digestate. *Bioresour. Technol.* **2012**, *110*, 63–70.

(139) Hublin, A.; Zelic, B. Modelling of the whey and cow manure co-digestion process. *Waste Manag. Res.* **2013**, *31*, 353–360.

(140) Gelegenis, J.; Georgakakis, D.; Angelidaki, I.; Mavris, V. Optimization of biogas production by co-digesting whey with diluted poultry manure. *Renew. Energy* **2007**, *32*, 2147–2160.

(141) Astals, S.; Ariso, M.; Galí, A.; Mata-Alvarez, J. Co-digestion of pig manure and glycerine: experimental and modelling study. *J. Environ. Manage.* **2011**, *92*, 1091–1096.

(142) Gavala, H. N.; Skiadas, I. V; Bozinis, N. A.; Lyberatos, G. Anaerobic Codigestion of Agricultural Industries' Wastewaters. *Water Sci. Technol.* **1996**, *34*, 67–75.

(143) Derbal, K.; Bencheikh-lehocine, M.; Cecchi, F.; Meniai, A.-H.; Pavan, P. Application of the IWA ADM1 model to simulate anaerobic co-digestion of organic waste with waste activated sludge in mesophilic condition. *Bioresour. Technol.* **2009**, *100*, 1539–1543.

(144) Sosnowski, P.; Wieczorek, A.; Ledakowicz, S. Anaerobic co-digestion of sewage sludge and organic fraction of municipal solid wastes. *Adv. Environ. Res.* **2003**, *7*, 609–616.

(145) Bolzonella, D.; Battistoni, P.; Susini, C.; Cecchi, F. Anaerobic codigestion of waste activated sludge and OFMSW: the experiences of Viareggio and Treviso plants (Italy). *Water Sci. Technol.* **2006**, *53*, 203.

(146) Miot, A.; Guevarra, K.; Ajedegba, J.; Jones, B. M.; Ving, K.; Jolis, D. Restaurant Trap Waste Characterization and Full Scale FOG Co-Digestion at the San Francisco Oceanside Plant. In *Proceedings of the Water Environment Federation Technical Exhibition and Conference (WEFTEC)*; Chicago, IL, 2013; pp. 1–18.

(147) Siles López, J. A.; Martín Santos, M. D. L. A.; Chica Pérez, A. F.; Martín Martín, A. Anaerobic digestion of glycerol derived from biodiesel manufacturing. *Bioresour. Technol.* **2009**, *100*, 5609–5615.

(148) Park, N. D.; Thring, R. W.; Garton, R. P.; Rutherford, M. P.; Helle, S. S. Increased biogas production in a wastewater treatment plant by anaerobic co-digestion of fruit and vegetable waste and sewer sludge - a full scale study. *Water Sci. Technol.* **2011**, *64*, 1851–1856.

(149) Rizk, M. C.; Tavares, G. Anaerobic co-digestion of fruit and vegetable waste and sewage sludge. *Int. J. Chem. React. Eng.* **2007**, *5*.

(150) Gomez, X.; Cuetos, M.; Cara, J.; Moran, A.; Garcia, A. Anaerobic co-digestion of primary sludge and the fruit and vegetable fraction of the municipal solid wastes: conditions for mixing and evaluation of the organic loading rate. *Renew. Energy* **2006**, *31*, 2017–2024.

(151) Björnsson, L.; Murto, M.; Mattiasson, B. Evaluation of parameters for monitoring an anaerobic co-digestion process. *Appl. Microbiol. Biotechnol.* **2000**, *54*, 844–849.

(152) Kim, J. K.; Oh, B. R.; Chun, Y. N.; Kim, S. W. Effects of temperature and hydraulic retention time on anaerobic digestion of food waste. *J. Biosci. Bioeng.* **2006**, *102*, 328–332.

(153) Alatriste-Mondragon, F.; Samar, P.; Cox, H. H. J.; Ahring, B. K.; Iranpour, R. Anaerobic codigestion of municipal, farm, and industrial organic wastes: a survey of recent literature. *Water Environ. Res.* **2006**, *78*, 607–636.

(154) Mata-Alvarez, J.; Dosta, J.; Macé, S.; Astals, S. Codigestion of solid wastes: a review of its uses and perspectives including modeling. *Crit. Rev. Biotechnol.* **2011**, *31*, 99–111.

(155) Converti, A.; Drago, F.; Ghiazza, G.; Borghi, M. Del. Co-digestion of Municipal Sewage Sludges and Prehydrolysed Woody Agricultural Wastes. *J. Chem. Technol. Biotechnol.* **1997**, *69*, 231–239.

(156) Wan, C.; Zhou, Q.; Fu, G.; Li, Y. Semi-continuous anaerobic co-digestion of thickened waste activated sludge and fat, oil and grease. *Waste Manag.* **2011**, *31*, 1752–1758.

(157) Long, J. H.; Aziz, T. N.; Reyes, F. L. D. L.; Ducoste, J. J. Anaerobic co-digestion of fat, oil, and grease (FOG): A review of gas production and process limitations. *Process Saf. Environ. Prot.* **2012**, *90*, 231–245.

(158) Nielsen, H. B.; Angelidaki, I. Codigestion of manure and industrial organic waste at centralized biogas plants: process imbalances and limitations. *Water Sci. Technol.* **2008**, *58*, 1521–1528.

(159) Schoen, M. a; Sperl, D.; Gadermaier, M.; Goberna, M.; Franke-Whittle, I.; Insam, H.; Ablinger, J.; Wett, B. Population dynamics at digester overload conditions. *Bioresour. Technol.* **2009**, *100*, 5648–5655.

(160) Switzenbaum, M. S.; Giraldo-Gomez, E.; Hickey, R. F. Monitoring of the anaerobic methane fermentation process. *Enzyme Microb. Technol.* **1990**, *12*, 722–730.

(161) Boe, K.; Batstone, D. J.; Steyer, J. P.; Angelidaki, I. State indicators for monitoring the anaerobic digestion process. *Water Res.* **2010**, *44*, 5973–5980.

(162) Moletta, R.; Escoffier, Y.; Ehlinger, F.; Coudert, J.-P.; Leyris, J.-P. On-line automatic control system for monitoring an anaerobic fluidized-bed reactor: response to organic overload. *Water Sci. Technol.* **1994**, *30*, 11–20.

(163) Tchobanoglous, G.; Burton, F. L.; Stensel, H. D. *Wastewater Engineering: Treatment and Reuse*; 4th ed.; McGraw Hill: New York, NY, 2003.

(164) Bouallagui, H.; Torrijos, M.; Godon, J. J.; Moletta, R.; Cheikh, R. Ben; Touhami, Y.; Delgenes, J. P.; Hamdi, M. Two-phases anaerobic digestion of fruit and vegetable wastes: bioreactors performance. *Biochem. Eng. J.* **2004**, *21*, 193–197.

(165) Donoso-Bravo, A.; Ruiz-Filippi, G.; Chamy, R. Anaerobic Treatment of Low-Strength Wastewater With a High Fraction of Particulate Matter in an Unconventional Two-Phase ASBRs System. *Biochem. Eng. J.* **2009**, *43*, 297–302.

(166) Speece, R. E. *Biotechnology and Odor/Corrosion Control for Municipalities and Industries*; Archae Press: Nashville, TN, 2008.

(167) Ferrer, I.; Vázquez, F.; Font, X. Long term operation of a thermophilic anaerobic reactor: process stability and efficiency at decreasing sludge retention time. *Bioresour. Technol.* **2010**, *101*, 2972–2980.

(168) Grady, C. P. L.; Daigger, G. T.; Love, N. G.; Filipe, C. D. M. *Biological Wastewater Treatment*; 3rd ed.; IWA Publishing and Alliance House: London, 2011.

(169) Hampton Roads Sanitation District. *Yearly Plants Operation Report*; 2013.

(170) Hill, D.; Cobb, S.; Bolte, J. Using volatile fatty acid relationships to predict anaerobic digester failure. *Trans. ASABE* **1987**, *30*, 496–501.

(171) Neves, L.; Oliveira, R.; Alves, M. M. Fate of LCFA in the co-digestion of cow manure, food waste and discontinuous addition of oil. *Water Res.* **2009**, *43*, 5142–5150.

(172) Salminen, E.; Rintala, J.; Lokshina, L. Y.; Vavilin, V. A. Anaerobic batch degradation of solid poultry slaughterhouse waste. *Water Sci. Technol.* **2000**, *41*, 33–41.

(173) Kusowski, J.; Graziano, S. J.; Desing, W. E.; Whitlock, D. D.; Sigmund, T. W.; Zitomer, D. H. Anaerobic Co-digestion Planning and Research for New Water (Green Bay, WI). In *Proceedings of the Water Environment Federation Technical Exhibition and Conference (WEFTEC)*; Chicago, IL, 2013; pp. 1–20.

(174) Kaspar, H. F.; Wuhrmann, K. Product inhibition in sludge digestion. *Microb. Ecol.* **1977**, *4*, 241–248.

(175) Nielsen, H.; Uellendahl, H.; Ahring, B. Regulation and optimization of the biogas process: Propionate as a key parameter. *Biomass and Bioenergy* **2007**, *31*, 820–830.

(176) Renard, P.; Dochain, D.; Bastin, G.; Naveau, H.; Nyns, E. J. Adaptive control of anaerobic digestion processes-a pilot-scale application. *Biotechnol. Bioeng.* **1988**, *31*, 287–294.

(177) McMahon, K. D.; Stroot, P. G.; Mackie, R. I.; Raskin, L. Anaerobic codigestion of municipal solid waste and biosolids under various mixing conditions--II: Microbial population dynamics. *Water Res.* **2001**, *35*, 1817–1827.

(178) McCarthy, P. Anaerobic waste treatment fundamentals. Part III: toxic materials and their control. *Public Work.* **1964**, *95*, 91–94.

(179) Angelidaki, I.; Ahring, B. K. Effects of free long-chain fatty acids on thermophilic anaerobic digestion. *Appl. Microbiol. Biotechnol.* **1992**, *37*, 808–812.

(180) Palatsi, J.; Laureni, M.; Andrés, M. V; Flotats, X.; Nielsen, H. B.; Angelidaki, I. Strategies for recovering inhibition caused by long chain fatty acids on anaerobic thermophilic biogas reactors. *Bioresour. Technol.* **2009**, *100*, 4588–4596.

(181) Hanaki, K.; Matsuo, T.; Nagase, M. Mechanism of inhibition caused by long-  chain fatty acids in anaerobic digestion process. *Biotechnol. Bioeng.* **1981**, *XXIII*, 1591–1610.

(182) Roy, F.; Albagnac, G.; Samain, E. Influence of calcium addition on growth of highly purified syntrophic cultures degrading long-chain Fatty acids. *Appl. Environ. Microbiol.* **1985**, *49*, 702–705.

(183) Braun, R.; Huber, P.; Meyrath, J. Ammonia toxicity in liquid piggery manure digestion. *Biotechnol. Lett.* **1981**, *3*, 159–164.

(184) Palatsi, J.; Illa, J.; Prenafeta-Boldú, F. X.; Laureni, M.; Fernandez, B.; Angelidaki, I.; Flotats, X. Long-chain fatty acids inhibition and adaptation process in anaerobic thermophilic digestion: batch tests, microbial community structure and mathematical modelling. *Bioresour. Technol.* **2010**, *101*, 2243–2251.

(185) Mosey, F. E. Mathematical Modelling of the anaerobic digesiton process: regulatory mechanisms for the formation of short-chain volatile acids from glucose. *Water Sci. Technol.* **1983**, *15*, 209–232.

(186) Podruzny, M. F.; van den Berg, L. Development of a computer control system for anaerobic methane-producing reactors. *Biotechnol. Bioeng.* **1984**, *26*, 392–393.

(187) Steyer, J. P.; Buffière, P.; Rolland, D.; Moletta, R. Advanced control of anaerobic digestion processes through disturbances monitoring. *Water Res.* **1999**, *33*, 2059–2068.

(188) Andrews, J.; Graef, S. Dynamic modeling and simulation of the anaerobic digestion process. In *Anaerobic Biological Treatment Processes*; Pohland, F., Ed.; American Chemical Society: Washington, D.C., 1970; pp. 126–162.

(189) Hill, D. T. A Comprehensive Dynamic Model for Animal Waste Methanogenesis. *Trans. ASAE* **1982**, *25*, 1374–1380.

(190) Pavlostathis, S. G.; Gossett, J. M. A kinetic model for anaerobic digestion of biological sludge. *Biotechnol. Bioeng.* **1986**, *28*, 1519–1530.

(191) Costello, D.; Greenfield, P.; Lee, P. Dynamic modelling of a single-stage high-rate anaerobic reactor—I. Model derivation. *Water Res.* **1991**, *25*, 847–858.

(192) Angelidaki, I. A mathematical model for dynamic simulation of anaerobic digestion of complex substrates: focusing on ammonia inhibition. *Biotechnol. Bioeng.* **1993**, *42*, 159–166.

(193) Batstone, D. J.; Keller, J.; Newell, R. B.; Newland, M. Modelling anaerobic degradation of complex wastewater. I: model development. *Bioresour. Technol.* **2000**, *75*, 67–74.

(194) Siegrist, H.; Renggli, D.; Gujer, W. Mathematical modelling of anaerobic mesophilic sewage sludge treatment. *Water Sci. Technol.* **1993**, *27*, 25–36.

(195) Siegrist, H.; Vogt, D.; Garcia-Heras, J. L.; Gujer, W. Mathematical model for meso- and thermophilic anaerobic sewage sludge digestion. *Environ. Sci. Technol.* **2002**, *36*, 1113–1123.

(196) Angelidaki, I.; Ellegaard, L.; Ahring, B. K. A comprehensive model of anaerobic bioconversion of complex substrates to biogas. *Biotechnol. Bioeng.* **1999**, *63*, 363–372.

(197) Shimizu, T.; Kudo, K.; Nasu, Y. Anaerobic waste-activated sludge digestion--A bioconversion mechanism and kinetic model. *Biotechnol. Bioeng.* **1993**, *41*, 1082–1091.

(198) Christ, O.; Wilderer, P. a; Angerhöfer, R.; Faulstich, M. Mathematical modeling of the hydrolysis of anaerobic processes. *Water Sci. Technol.* **2000**, *41*, 61–65.

(199) Tomei, M. C.; Braguglia, C. M.; Cento, G.; Mininni, G. Modeling of anaerobic digestion of sludge. *Crit. Rev. Environ. Sci. Technol.* **2009**, *39,* 1003–1051.

(200) Batstone, D. J.; Keller, J.; Steyer, J. P. A review of ADM1 extensions, applications, and analysis: 2002–2005. *Water Sci. Technol.* **2006**, *54*, 1.

(201) Nopens, I.; Batstone, D. J.; Copp, J. B.; Jeppsson, U.; Volcke, E.; Alex, J.; Vanrolleghem, P. a. An ASM/ADM model interface for dynamic plant-wide simulation. *Water Res.* **2009**, *43*, 1913–1923.

(202) Young, M. N.; Marcus, A. K.; Rittmann, B. E. A Combined Activated Sludge Anaerobic Digestion Model (CASADM) to understand the role of anaerobic sludge recycling in wastewater treatment plant performance. *Bioresour. Technol.* **2013**, *136*, 196–204.

(203) Boubaker, F.; Ridha, B. C. Modelling of the mesophilic anaerobic co-digestion of olive mill wastewater with olive mill solid waste using anaerobic digestion model No. 1 (ADM1). *Bioresour. Technol.* **2008**, *99*, 6565–6577.

(204) Vavilin, V. A.; Fernandez, B.; Palatsi, J.; Flotats, X. Hydrolysis kinetics in anaerobic degradation of particulate organic material: an overview. *Waste Manag.* **2008**, *28*, 939–951.

(205) Girault, R.; Bridoux, G.; Nauleau, F.; Poullain, C.; Buffet, J.; Steyer, J. P.; Sadowski, A. G.; Béline, F. A waste characterisation procedure for ADM1 implementation based on degradation kinetics. *Water Res.* **2012**, *46*, 4099–4110.

(206) Zaher, U.; Buffiere, P.; Steyer, J.-P.; Chen, S. A Procedure to Estimate Proximate Analysis of Mixed Organic Wastes. *Water Environ. Res.* **2009**, *81*, 407–415.

(207) Huete, E.; de Gracia, M.; Ayesa, E.; Garcia-Heras, J. L. ADM1-based methodology for the characterisation of the influent sludge in anaerobic reactors. *Water Sci. Technol.* **2006**, *54,* 157.

(208) Galí, a; Benabdallah, T.; Astals, S.; Mata-Alvarez, J. Modified version of ADM1 model for agro-waste application. *Bioresour. Technol.* **2009**, *100*, 2783–2790.

(209) National Community Pharmacists Association (NCPA). *800 community pharmacies launch initiative for safe disposal of unused patient medications*; Alexandria, VA, 2010.

(210) U.S. Environmental Protection Agency. *Municipal solid waste generation, recycling, and disposal in the United States: facts and figures for 2008*; Washington, D.C., 2008.

(211) Kandris, K.; Pantazidou, M. Landfill base liners: assessment of material equivalency and impact to groundwater. *Geotech. Geol. Eng.* **2011**, *30*, 27–44.

(212) Ecobalance Inc. *Life cycle inventory of a modern municipal solid waste landfill*; Bethesda, MD, 1999.

(213) Goemans, M.; Clarysse, P.; Joannès, J.; De Clercq, P.; Lenaerts, S.; Matthys, K.; Boels, K. Catalytic NOx reduction with simultaneous dioxin and furan oxidation. *Chemosphere* **2004**, *54*, 489–497.

(214) Alvim-Ferraz, M. C. M.; Afonso, S. A. V. Incineration of different types of medical wastes: emission factors for gaseous emissions. *Atmos. Environ.* **2003**, *37*, 5415–5422.

(215) U.S. Environmental Protection Agency. *Electricity from municipal solid waste*; Washington, D.C., 2010.

(216) Qian, X.; Koerner, R. M.; Gray, D. H. *Geotechnical Aspects of Landfill Design and Construction*; Prentice Hall: Upper Saddle River, New Jersey, 2002.

(217) U.S. Environmental Protection Agency. *Solid waste management and greenhouse gases: a life-cycle assessment of emissions and sinks*; Washington, D.C., 2006.

(218) U.S. Environmental Protection Agency. *Assessment and recommendations for improving the performance of waste containment systems*; Cincinnati, OH, 2002.

(219) Reinhart, D. R.; Pohland, F. G.; Gould, J. P.; Cross, W. H. The fate of selected organic pollutants codisposed with municipal refuse. *Res. J. Water Pollut. Control Fed.* **1991**, *63*, 780–788.

(220) Reinhart, D. R.; Gould, J. P.; Cross, W. H.; Pohland, F. G. Sorptive behavior of selected organic pollutants codisposed in a municipal landfill. In *Emerging Technologies in Hazardous Waste Management*; Washington, D.C., 1990.

(221) U.S. Environmental Protection Agency. *Biosolids generation, use, and disposal in the United States*; Washington, D.C., 1999.

(222) U.S. Department of Agriculture. *Measuring rurality: rural-urban continuum codes*; Washington, D.C., 2003.

(223) U.S. Census Bureau. *Annual population estimates: 2009 population estimates*; Washington, D.C., 2009.

(224) dVault Companies Inc. *Floor express - drug disposal collection unit*; Englewood, CO, 2010.

(225) Environmental Health & Safety Online. *Commercial hazardous waste incinerators*; Atlanta, GA, 2009.

(226) Environmental Health & Safety Online. *Commercial hazardous waste landfills*; Atlanta, GA, 2009.

(227) U.S. Environmental Protection Agency. *Clean energy: air emissions*; Washington, D.C., 2007.

(228) Rosén, C.; Jeppsson, U. *Aspects on ADM1 Implementation within the BSM2 Framework*; Lund, Sweden, 2006.

(229) C. Zickefoose, R. B. H. *Anaerobic sludge digestion: operations manual*; Washington, D.C., 1976.

(230) Henze, M.; Grady, C. P. L., J.; Gujer, W.; Marais, G. v. R.; Matsuo, T. *Activated Sludge Model No.1 (ASM1)*; London, 1987; p. IWA Scientific and Technical Report No. 1.

(231) O'Rourke, J. T. Kinetics of Anaerobic Treatment at Reduced Termperatures, Standford University, 1968.

(232) Garcia-Heras, J. L. Reactor sizing, process kinetics and modelling of anaerobic digestion of complex wastes. In *Biomethanization of the organic fraction of municipal solid wastes*; Mata-Alvarez, J., Ed.; IWA Publishing, 2003; pp. 21–62.

(233) Zhang, B.; Fowler, K. R.; Grace, M. D.; Mondal, S.; Grimberg, S. J. Optimization of Anaerobic Digestion Model No. 1 (ADM1): Simulation of Dairy Manure Digestion. In *American Society of Agricultural and Biological Engineers Annual International Meeting*; Reno, NV, 2009.

(234) Brouwer, A. F. The Dynamic Anaerobic Reactor & Integrated Energy System (DARIES) Model, Clarkson University, 2010, pp. 1–143.

(235) Elbeshbishy, E.; Nakhla, G. Batch anaerobic co-digestion of proteins and carbohydrates. *Bioresour. Technol.* **2012**, *116*, 170–178.

(236) Masse, L.; Massé, D. I.; Kennedy, K. J.; Chou, S. P. Neutral fat hydrolysis and long-chain fatty acid oxidation during anaerobic digestion of slaughterhouse wastewater. *Biotechnol. Bioeng.* **2002**, *79*, 43–52.

(237) Lokshina, L. Y.; Vavilin, V. A.; Salminen, E.; Rintala, J. Modeling of anaerobic degradation of solid slaughterhouse waste. *Appl. Biochem. Biotechnol.* **2003**, *109*, 15–32.

(238) Vavilin, V. A.; Lokshina, L. Y.; Jokela, J. P. Y.; Rintala, J. Modeling solid waste decomposition. *Bioresour. Technol.* **2004**, *94*, 69–81.

(239) Vavilin, V. A.; Angelidaki, I. Anaerobic degradation of solid material: importance of initiation centers for methanogenesis, mixing intensity, and 2D distributed model. *Biotechnol. Bioeng.* **2005**, *89*, 113–122.

(240) Ristow, N. E.; Sötemann, S. W.; Wentzel, M. C.; Loewenthal, R. E.; Ekama, G. a. The effects of hydraulic retention time and feed COD concentration on the rate of hydrolysis of primary sewage sludge under methanogenic conditions. *Water Sci. Technol.* **2006**, *54*, 91–100.

(241) Gavala, H. N.; Skiadas, I. V.; Lyberatos, G. On the performance of a centralised digestion facility receiving seasonal agroindustrial wastewaters. *Water Sci. Technol.* **1999**, *40*, 339–346.

(242) Gavala, H. N.; Lyberatos, G. Influence of anaerobic culture acclimation on the degradation kinetics of various substrates. *Biotechnol. Bioeng.* **2001**, *74*, 181–195.

(243) Novak, J. T.; Carlson, D. A. The kinetics of anaerobic long chain fatty acid degradation. *J. Water Pollut. Control Fed.* **1970**, *42*, 1932–1943.

(244) Page, D. I.; Hickey, K. L.; Narula, R.; Main, a L.; Grimberg, S. J. Modeling anaerobic digestion of dairy manure using the IWA Anaerobic Digestion Model no. 1 (ADM1). *Water Sci. Technol.* **2008**, *58*, 689–695.

(245) Blumensaat, F.; Keller, J. Modelling of two-stage anaerobic digestion using the IWA Anaerobic Digestion Model No. 1 (ADM1). *Water Res.* **2005**, *39*, 171–183.

(246) Vavilin, V. A.; Lokshina, L. Y.; Rytov, S. V.; Kotsyurbenko, O. R.; Nozhevnikova, A. N.; Parshina, S. N. Modelling methanogenesis during anaerobic conversion of complex organic matter at low temperatures. *Water Sci. Technol.* **1997**, *36*, 531–538.

(247) Angenent, L. T.; Sung, S.; Raskin, L. Methanogenic population dynamics during startup of a full-scale anaerobic sequencing batch reactor treating swine waste. *Water Res.* **2002**, *36*, 4648–4654.

(248) McMahon, K. D.; Zheng, D.; Stams, A. J. M.; Mackie, R. I.; Raskin, L. Microbial population dynamics during start-up and overload conditions of anaerobic digesters treating municipal solid waste and sewage sludge. *Biotechnol. Bioeng.* **2004**, *87*, 823–834.

(249) Heo, N. H.; Park, S. C.; Kang, H. Effects of Mixture Ratio and Hydraulic Retention Time on Single-Stage Anaerobic Co-digestion of Food Waste and Waste Activated Sludge. *J. Environ. Sci. Heal. Part A* **2004**, *39*, 1739–1756.

(250) Miron, Y.; Zeeman, G.; Lier, J. Van; Lettinga, G. The role of sludge retention time in the hydrolysis and acidification of lipids, carbohydrates and proteins during digestion of primary sludge in CSTR systems. *Water Res.* **2000**, *34*.

(251) Banerjee, A.; Elefsiniotis, P.; Tuhtar, D. The effect of addition of potato-processing wastewater on the acidogenesis of primary sludge under varied hydraulic retention time and temperature. *J. Biotechnol.* **1999**, *72*, 203–212.

(252) Bougrier, C.; Carrère, H.; Delgenès, J. P. Solubilisation of waste-activated sludge by ultrasonic treatment. *Chem. Eng. J.* **2005**, *106*, 163–169.

(253) U.S. Environmental Protection Agency. *Light-duty automotive technology and fuel economy trends: 1975 through 2008*; Washington, D.C., 2008; p. 96.

(254) Medical Arts Press. *TakeAway^{TM} mail-away medication disposal container*; Minneapolis, MN, 2010.

(255) U.S. Environmental Protection Agency. *Waste transfer stations: a manual for decision-making*; Washington, D.C., 2002.

(256) Levinson, D.; Krizek, K. J. *Planning for Place and Plexus: Metropolitan Land Use and Transport*; Routledge: New York, NY, 2008.

(257) Athena Sustainable Materials Institute. *Cradle-to-gate life cycle inventory: Canadian and US steel production by mill type*; Ottawa, Canada, 2002.

(258) Swiss Centre for Life Cycle Inventories. *Life cycle inventories of waste treatment services: landfills, underground deposits, landfarming*; Düebendorf, Switzerland, 2009; Vol. 14.

(259) Swiss Centre for Life Cycle Inventories. *Life cycle inventories of waste treatment services: waste incineration*; St. Gallen, Switzerland, 2009; Vol. 14.

(260) U.S. Environmental Protection Agency. *Average annual emissions and fuel consumption for passenger cars and light trucks*; Washington, D.C., 2000.

(261) U.S. Environmental Protection Agency. *Average carbon dioxide emissions resulting from gasoline and diesel fuel*; Washington, D.C., 2005.

(262) U.S. Federal Highway Administration. *Assessing the effects of freight movement on air quality at the national and regional level*; Washington, D.C., 2005.

(263) Sharma, H. D.; Reddy, K. R. *Geoenvironmental Engineering: Site Remediation, Waste Containment, and Emerging Waste Management Technologies*; John Wiley & Sons, Inc.: Hoboken, New Jersey, 2004.

(264) U.S. Environmental Protection Agency. *WAste Reduction Model (WARM) aggregate*; Washington, D.C., 2009; Vol. 10.

(265) McFarland, M. J. *Biosolids Engineering*; McGraw-Hill: New York, NY, 2001.

(266) Aumonier, S. The greenhouse gas consequences of waste management - identifying preferred options. *Energy Convers. Manag.* **1996**, *37*, 1117–1122.

(267) U.S. Federal Highway Administration. *Highway statistics*; Washington, D.C., 2008.

(268)  Argonne National Laboratory. *Estimation of fuel use by idling commercial trucks*; Argonne, Illinois, 2006.