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An Integrated Environmental Analysis Framework for Multi-Functional Urban Food Production Utilizing Nutrient Recycling from Organic Waste Streams

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AN INTEGRATED ENVIRONMENTAL ANALYSIS FRAMEWORK FOR MULTI-FUNCTIONAL URBAN FOOD PRODUCTION UTILIZING NUTRIENT RECYCLING FROM ORGANIC WASTE STREAMS

by

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ABSTRACT

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Increasing enthusiasm for local food, including urban agriculture, has piqued research interest in the tenets underlying perceived benefits of localizing food production. This study develops and demonstrates the application of a comprehensive framework for the life cycle environmental assessment of the utilization of urban organic wastes in urban agriculture, specifically fruit and vegetable production. Results indicate that this full “urban nutrient cycle” may have significant environmental benefits in terms of land area requirements, water use, wastewater generation, nutrient recovery, environmental contamination and green infrastructure potential, compared to more conventional methods of waste processing and food production. Urban intensive food production using soil amendments produced from locally-sourced organic wastes in Northern and Eastern U.S. cities could meet up to 70% of current vegetable and 17% of current fruit consumption needs. Urban food production at this level would require 2,000 - 4,000 hectares for a population of one million, and has significant green infrastructure potential. Potential water savings from urban production are in the range of 10 - 17% of the urban area’s annual domestic use, and this “virtual water” can offset irrigation water

use in more arid production areas. Optimizing resource recovery by separating sources of organic wastes results in 1-2% lower wastewater generation and up to 44% greater phosphorus recovery compared to current baseline methods. Source separation also reduces contaminant types and levels. Overall, energy and emissions benefits of urban nutrient recycling and food production are in the range of 1-2% of the city's annual totals. The benefits of shorter transportation loops for both organic wastes and food are negligible. The lifecycle environmental impacts of alternative methods of food waste processing and reuse vary depending on policies at the local, state, and federal level. This research suggests how the LCA framework can inform policy analysis. Policies for waste processing, urban agriculture, and green infrastructure affect the relative environmental performance of different approaches to managing food waste. Evidence-based policy utilizing the framework developed here may outperform conventional approaches on a number of sustainability metrics. The framework can be applied to inform location-specific policy regarding food waste processing and urban food production.

Keywords: urban agriculture, nutrient cycle, LCA, phosphorus recovery, green infrastructure, urban soil, compost, biosolids, food-water nexus, environmental policy, virtual water

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LIST OF ABBREVIATIONS

AAPFCO	Association of American Plant Food Control Officials
AD	Anaerobic Digestion
BTU	British Thermal Unit
C	Carbon
CAR	Climate Action Reserve
CCME	Canadian Council of Ministers of the Environment
CFR	Code of Federal Regulations
CO_2	Carbon Dioxide
CoEAT	Co-Digestion Economic Analysis Tool
EPA	United States Environmental Protection Agency
ERS	Economic Research Service of the USDA
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GHG	Greenhouse Gas
GI	Green Infrastructure
ha	hectare
K	Potassium
kg	Kilogram
km	Kilometer
LCA	Life Cycle Assessment
L	Liter

m^3	Cubic Meter
mg	milligram
MJ	Million Joules
MSW	Municipal Solid Waste
$MTCO_2$-e	Million (metric) Tons Carbon Dioxide Equivalent
N	Nitrogen
N_2O	Nitrous Oxide
NASS	National Agricultural Statistics Service of the USDA
NOAA	National Oceanic and Atmospheric Administration
NOP	National Organic Program
OM	Organic Matter
P	Phosphorus
PCB	Polychlorinated Biphenyl
PCPP	Personal Care and Pharmaceutical Product
PWEP	Percent Water Extractable Phosphorus
SOM	Soil Organic Matter
STA	Seal of Testing Approval (U.S. Composting Council)
TNSSS	Targeted National Sewerage Sludge Survey
TSP	Triple Super Phosphate
UK	United Kingdom
UNCTAD	United Nations Conference on Trade and Development
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency

USGS	United States Geological Survey
WARM	Waste Reduction Model (EPA)
WFRP	Whole Farm Revenue Protection
WWTP	Wastewater Treatment Plant
XRF	X-ray fluorescence

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Introduction

Urban food production, as epitomized by the “local food” movement in the U.S., is implicitly or explicitly promoted as a more sustainable option than conventional, mostly rural, agricultural production or large-scale organic production. Sustainability, in turn, encompasses economic, social and environmental components. Studies have provided evidence that localized food production can have economic advantages compared to conventional agriculture in providing employment and contributing to local economies (Low & Vogel, 2011). Researchers have also documented many of the social benefits of urban food production (Lovell et al., 2010). However, the environmental sustainability of urban food production has not been well-studied using comprehensive life cycle assessments (LCAs). On the contrary, recent studies have called into question one of the major assumptions of localized food production, the widespread idea that “food miles” is a significant indicator of environmental sustainability (Edwards-Jones et al., 2008; Garnett, 2011; Weber & Matthews, 2008). In addition, although closing urban “nutrient loops” is widely advocated in the literature (Mihelcic et al., 2011; Takata et al., 2012), the true potential of urban nutrient cycling to provide a portion of food needs for urban populations, as well as urban ecosystem benefits (such as stormwater mitigation), is another current research gap. Some LCAs have addressed individual components of urban plant nutrient flows, such as urban phosphorus (P) (Baker, 2011) and nitrogen (N) (Forkes, 2007) fluxes. However, few or none comprehensively address all of the critical plant nutrient components for both urban food production and urban ecosystem services in the U.S. context, namely phosphorus, nitrogen and organic

matter (carbon).

Currently, approximately 31% of food is wasted in the U.S. at the consumer and retail levels, and over 50% is disposed of in landfills (Buzby et al., 2014). Food waste is the largest component of landfilled municipal waste, where it produces methane, a potent greenhouse gas, and leachate that poses risks of groundwater contamination. Another 30% of food waste is conveyed to wastewater treatment plants, where nutrients are only partially removed before effluents are discharged to surface waters, contributing to nutrient pollution. Another 13% of food waste is incinerated, with most of the resulting ash disposed of in landfills, while just 5% of food waste is currently recycled via composting (EPA, 2015). In comparison, 60% of yard waste is currently composted on average; while some states still allow landfilling of yard waste, roughly half have banned the practice (EPA, 2015).

The environmental sustainability of urban food production is likely to be greater if it utilizes nutrients and organic matter from urban waste streams, such as food waste and yard wastes. These organic waste streams have the potential to convey harmful nutrient and carbon loads to urban surface waters and groundwater. They require processing and management measures, including wastewater treatment and landfilling of wastes, to avoid or mitigate negative ecosystem consequences. These processes may lower nutrient quality by adding contaminants or may sequester and discard valuable nutrients such as phosphorus. While P is abundant in the environment, global reserves of phosphate rock are finite and P scarcity will become a critical driver of urban nutrient

recycling in the future (Cordell et al., 2011).

This study advances a framework for the comprehensive environmental assessment of the complete urban nutrient cycle, from food and yard waste streams to food production (fruits and vegetables) and ecosystem services, including stormwater mitigation. The framework will allow assessment of the environmental effects of recovering plant nutrients from urban organic waste streams and recycling those nutrients for urban agricultural production. It addresses nutrient quantity and quality from the food waste and yard waste streams of an urban population, as well as the resulting food production potential and land requirements based on recycled nutrient inputs. Phosphorus recoverability is characterized for various combinations of nutrient recycling processes. The framework also examines energy balances and greenhouse gas (GHG) emissions from waste recovery/recycling processes and food production, including transportation for each stage. It addresses the potential for GHG mitigation through carbon sequestration in urban soils used for food production. In addition, the framework facilitates the environmental assessment of urban crop land in terms of stormwater capture and runoff potential, as well as soil erosion and nutrient re-deposition. Water use and wastewater generation throughout the entire recovery, reuse and food production cycle are addressed.

Arguably, the currently dominant food production and distribution system in the U.S. (and elsewhere) suffers from an acute lack of life cycle perspective. Humans have altered the ”natural,” or background flows of N, P, and C considerably by producing food

largely in rural areas and exporting it to urban areas, where its use and disposal have become significant environmental problems. Urban areas have become nutrient sinks, where excess N and P contribute to surface water and groundwater contamination, leading to algal growth and impaired drinking water (Drechsel et al., 2007). Significant quantities of nutrients are exported to distant locations when waste effluent is discharged to rivers. Even in presumably closed systems, such as traditional livestock-crop farms, export of nutrients is inevitable as long as food is being exported from the farm (Schröder et al., 2011).

The present study addresses environmental factors only, but urban organics recycling for urban food production has the potential to provide local jobs and retain more food dollars locally. Local fruit and vegetable production also has the potential to improve public health through better nutrition, along with the oft-cited social benefits of community gardening. These economic and social benefits figure prominently in food policy discussions, and can be significant drivers of the increasing interest in urban food production.

The results of the framework analysis can inform local and regional policy development for organics processing and food production. The potential scale of urban food production is an important policy question, and the framework provides both general scale estimates and the means to determine more precise, context-specific estimates of scale. Across the range of environmental effects considered here, the framework results provide general estimates, or baseline information, that is useful for more generally-

focused policy. The framework approach and specific values calculated here, along with the external models used in the framework, can be used to inform more specific and place-based policy.

Three scenarios for urban organic waste processing are modeled, based on combinations of the most widely used current practices. Practices include landfilling, processing via wastewater treatment plant (WWTP), and anaerobic digestion and composting of source-separated solid waste streams. Anaerobic digestion is included because it is an emerging technology for processing urban food wastes, and as modeled here, works in conjunction with the more common practice of aerobic composting of food and yard waste. Anaerobic digestion may be more suitable for processing of high strength food wastes in proximity to residential and public land uses. Its controlled conditions facilitate capture of noxious odors and can eliminate pest issues sometimes found in open windrow composting. Additional modeling characterizes the ecosystem performance of urban production of fruits and vegetables in comparison with other forms of production, including conventional production and large-scale organic production. The scenarios and modeling produce first approximations and broad estimates of the potential benefits of urban organics recycling under different technological management processes and practices.

Life Cycle Assessment (LCA)

Environmental LCA is designed to account for a comprehensive and cumulative set of

environmental impacts from all stages of a product's life cycle. It encompasses raw material extraction, processing, use, maintenance, and eventual disposal or reuse, and often encompasses transportation of materials as well (EPA, 2006). LCA attempts to account for relevant material and energy inputs as well as environmental releases, and the potential environmental impacts of those inputs and releases.

LCA helps to reveal the true environmental costs and benefits of a process or product and assists in focusing on policies that can internalize significant environmental externalities. With this information, managers and policy-makers can make better decisions to achieve sustainable outcomes. As noted, many existing studies have utilized LCA concepts and methodology to examine the component parts of the framework being developed here for nutrient recycling and urban food production. This study develops a full LCA by linking and combining the various components of nutrient recycling and food production, including multi-functional elements such as green infrastructure potential. Multi-functional landscapes are “landscapes that provide a range of beneficial functions across production, ecological, and cultural dimensions, considering the needs and preferences of the owners and users.” (Lovell et al., 2010)

In addition to the more traditional components of LCA, such as energy and material flows and balances, climate change awareness has highlighted the importance of carbon emissions. Like N and P, which can function as both environmental pollutants and valuable agricultural resources, carbon is both a potent atmospheric pollutant and an essential soil component. LCA for nutrient cycling is complicated by the fact that

these three components can function as both useful products and harmful emissions, depending on where they end up in the system, both temporally and spatially. For that reason, it is useful to focus on these components as the fundamental units of analysis. Their presence in surface water and groundwater, locked away in a landfill (N, P, and C), or in the atmosphere (N and C) are environmental liabilities. On the other hand, their presence in soils, in the right ratios and quantities, make food production possible. Therefore, the LCA assessment here does not proceed from a clearly defined starting point, such as material extraction, and end up at a well-defined end-of-life or disposal point. Instead, once the critical unit of analysis is identified, the study can proceed in both directions (Bernstad & la Cour Jansen, 2011; Butler & Hooper, 2010). This is especially appropriate because the nutrient cycle (the system under assessment) is a looped or circular system. For the assessment of nutrient recycling for urban food production, plant nutrients and soil constituents (organic matter or carbon) are the critical units of LCA analysis, and that is the approach adopted here.

An important aspect of LCA is establishing realistic and useful system boundaries. Boundaries that are too constrained risk excluding significant impacts, while boundaries that are too broad may make assessments unmanageable, highly resource-intensive, and include too many insignificant factors. The LCA system boundaries here include cycling materials (i.e., food and organic wastes and their constituents), and related operational functions for processing wastes and growing food. They encompass the range of inputs, outputs, and emissions addressed in the framework components. These include nutrients, contaminants, energy, GHG emissions, water use, and

wastewater generation. Ecosystem (surface) water quantity and quality are included as well, from both green infrastructure and nutrient pollution perspectives. The analytical spatial boundary coincides with the administrative (political) unit for the urban population where food is consumed and where organic waste is produced, plus any portion of included or downstream watersheds. The spatial boundary also encompasses land outside the city where food is grown or organic wastes are processed, as well as a transportation component for inputs and outputs. However, the LCA excludes environmental factors related to the construction, maintenance, and decommissioning/disposal of infrastructure components, such as wastewater treatment plants, anaerobic digesters and transportation vehicles (except as noted).

The current extent of urban food production in the U.S. is inherently difficult to quantify, due to its distributed and multi-scalar nature and the difficulty of defining precise spatial boundaries. Food and agriculture policies are becoming more common in the comprehensive plans, sustainability plans, and zoning codes of U.S. cities (Hendrickson & Porth, 2012). A multi-functional life cycle approach will help both researchers and policy-makers address questions of the potential environmental effects of urban food production, and allow a limited comparison to both rural organic and conventional food production. The nexus of urban food production and urban wastewater/organic solid waste processing is a promising avenue for multi-functional research, because it integrates waste recycling, energy & emissions, agricultural productivity, land-use, and plant, food, and environmental contaminants.

The potential scale of urban food production is an important policy question, with a host of related components. How much food could be produced on a sustainable basis from a city's organic waste streams? How much land and plant nutrients might be required? Are soil amendments produced from urban wastes likely to improve or degrade urban soils, and are amended urban soils suitable for producing food? Do some methods of organic waste processing produce better soil amendments than others?

In addition to the above, urban nutrient loops have some more generalized environmental impacts. What are the impacts on water, energy, and greenhouse gas emissions of urban crop production and alternative methods of waste processing? Can urban cropland mitigate urban stormwater runoff?

This research advances a framework that can function as both a policy tool and a research tool for urban food production, particularly as it relates to urban nutrient cycling. The framework identifies components that are significant to environmental sustainability and quantifies environmental effects through scenario and sensitivity modeling. The findings can help to inform policy-making regarding the appropriate levels and processes for nutrient recovery from urban waste streams, as well as the environmental effects (positive and negative) of multi-functional urban food production. In addition, the research provides valuable information for the effective management of organic waste streams across a range of realistic scenarios.

Research Objectives

Identify and develop the components of a useful environmental LCA framework for multi-functional urban food (fruit and vegetable) production.

Apply the framework to make limited environmental comparisons among alternative waste processing and food production systems. Alternative food production systems include conventional production (synthetic fertilizer inputs and mechanization), large-scale organic production (mechanized), and urban intensive production with nutrient recycling (smaller scale and not mechanized).

Apply the framework for scenario analysis, to quantify the environmental performance of different processes, practices and methods for urban nutrient recycling.

Evaluate the effects of urban organics recycling and food production in the wider social-ecological system, in comparison with the impacts of other sectors such as domestic energy and water use, and GHG emissions.

Identify significant policy drivers related to urban organic waste processing and food production. Assess policy in relation to the framework and research. Provide examples of suggested policy modifications.

The research addresses the full urban nutrient-cycling loop, from organic waste streams to food production on soils amended with recycled organics and encompasses

environmental multi-functionality, which includes green infrastructure. It considers the potential for urban agriculture to act as a source or sink of nutrients and carbon to the environment, its potential to absorb and retain stormwater, and its potential to improve or degrade native urban soils.

Potential environmental components for food production and nutrient recycling from urban organic wastes Include:

- recycled nutrient quantity and quality for agricultural and GI applications
- phosphorus (a finite elemental resource) conservation and recycling
- food production capacity from recycled nutrients
- energy use and production (processing, transportation)
- greenhouse gas (GHG) emissions balances
    - net emissions from organic waste processing and food production
    - carbon sequestration potential of amended soils
- water use and wastewater generation
- stormwater, contaminant, and nutrient runoff to urban surface waters
- land use (area requirements)

Hypothetical scenario analyses employ the framework to model a likely range of effects for a typical city of one million people, using different combinations of processing methods for recycling nutrients for food production. Scenarios are based on current organic waste processing and urban food production practices in US cities. The

scenarios can help to identify key drivers of LCA outcomes.

The research also identifies major policy drivers and related factors that shape nutrient processing and urban food production policy in the U.S. context and discusses policy choices that may be informed by the framework and the scenario analyses.

Methods & Data

The study synthesizes data and values from the literature and applies a number of existing models and new calculations to form a comprehensive framework comprising the most significant life cycle indicators for urban food production from recycled nutrients. On a meta-level, the research methodology consists of linking existing data and models that separately address individual components of the framework[1]. Specific methodology is addressed in more detail for each framework component. Organic waste and nutrient flows, as well as the resulting soil and food production potential, are characterized for a hypothetical metropolitan area with a population of one million.

Modeling, based on this hypothetical urban area, examines three scenarios for processing urban organic wastes. Serial modeling utilizes the outputs of prior analyses as inputs for subsequent discrete models and analysis stages. Scenario evaluation under the framework provides results for each component, with discussion of tradeoffs

[1] LCA modeling utilizes a variety of approaches appropriate to each metric which reflect average or typical U.S. values and parameters. No independent model, such as GaBi, is employed here as a general check on results.

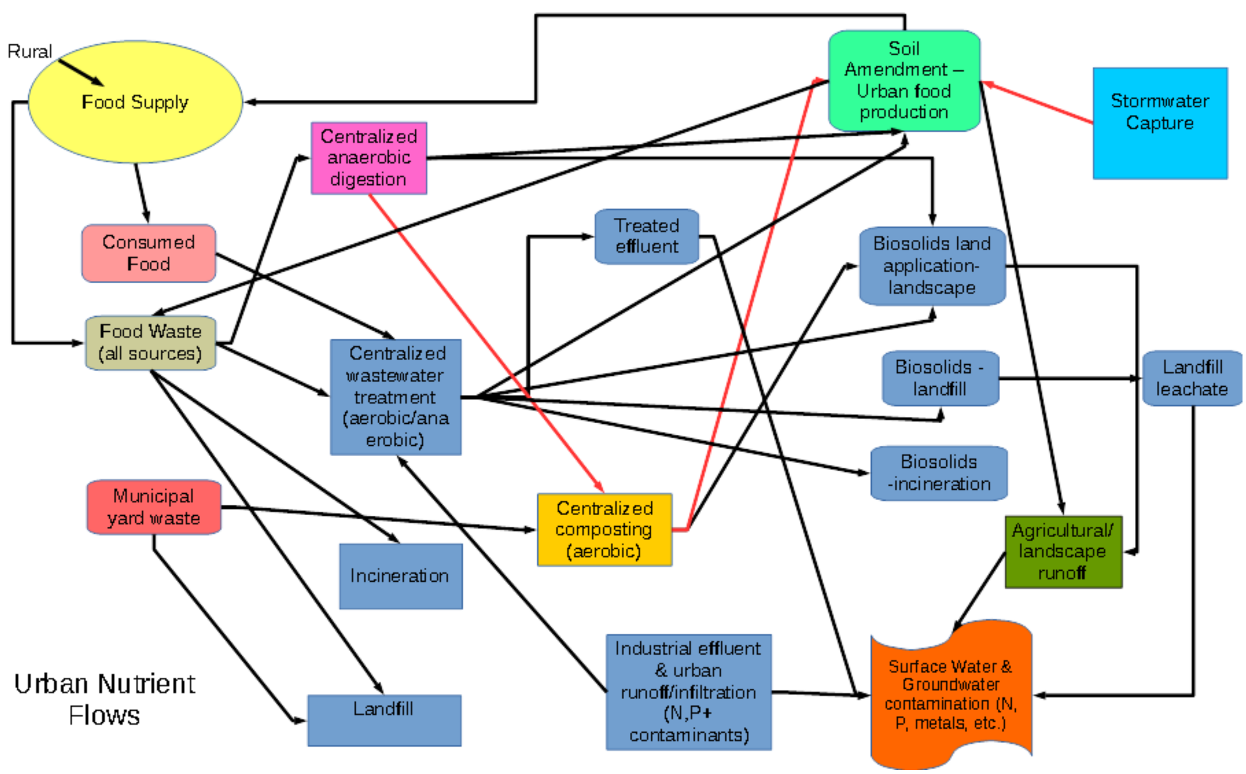
and implications.

Three specific scenarios are modeled: an anaerobic digestion process, as well as waste processing through a wastewater treatment plant with input from in-sink disposal units. These processes are compared to each other and to the common practice of landfilling of unprocessed organic wastes.

Biosolids or Sewage Sludge? The terms "biosolids" and "sewage sludge" both refer to the high solids output of wastewater treatment processes, excluding the effluent discharge. Biosolids are sludges that meet USEPA metals and pathogen standards for land application. Both terms are distinct from "digestate," which is the output of anaerobic processing of organic wastes. Depending on the specific anaerobic process and feedstock, digestate may be relatively liquid or solid in form. Although WWTPs may employ anaerobic stages, "biosolids" or "sewage sludge" are used exclusively here to designate WWTP solids output.

Figure 1 represents urban nutrient flows graphically.

Figure 1- Urban Nutrient Flows



The framework addresses a subset of these components, in order to characterize the environmental effects of urban organic waste processing and food production.

As the schematic suggests, outputs from some processes become inputs for other processes, and (semi) closed nutrient loops are formed when food production, consumption, and organics processing occur in close proximity.

LCA Framework Components

Many individual components of the framework are relatively well-developed from other contexts and can be readily adapted and synthesized to characterize multifunctional nutrient recycling for urban food production. In other cases, components are underdeveloped, and the present research will identify those gaps and provide recommendations for further development. Each component is addressed below, with literature sources for data, and the methods and models that are used in the framework.

Plant Nutrients From Organic Wastes – Quantity/Quality

This component addresses the quantity and quality of soil amendments (plant nutrients) that the city of one million can potentially produce from organic wastes annually. Food waste, yard waste and sewage (human excreta) are quantified, and contaminant types and levels are quantified by example. The discussion considers contaminant sources and their significance for urban fruit and vegetable production.

Since plant nutrients are both inputs (soil amendments) and outputs (from food and yard waste) in the urban agricultural context, these are the functional units of analysis for the study. In urban nutrient cycling, the quantity and quality of urban nutrients, including P, N, and carbon (C), are the primary units of interest and serve to link organic wastes with food production. (Carbon comprises 58% of the organic matter (OM) found in organic waste and soils). Waste processing methods, as well as urban soil and food production, are assessed in terms of the production and use of these plant nutrients. Although potassium (K) is an important plant macro-nutrient, it is not considered in the present study. K is not typically an environmentally harmful pollutant, and it is relatively simple to amend K-deficient soils.

Contaminant levels are a significant concern in soil amendments. These concerns may be especially acute in densely populated urban contexts where soils may have some level of existing contamination with the potential for human contact. The U.S. EPA focuses on metals and pathogen contamination levels in biosolids (CFR 40 part 503 rules). These standards (Exceptional Quality level) have been adopted by the U.S. Composting Council, a national industry organization. While biosolids and composts may have similar metals levels (Brinton, 2000), biosolids are much more likely to contain “emerging contaminants” such as personal care and pharmaceutical products (PCPPs) and legacy contaminants like PCBs. On the other hand, composts made from yard wastes may include pesticide residues. These may also be present in biosolids where WWTPs receive stormwater runoff (combined sewers).

Methods and Sources

The following studies provide data to calculate the raw inputs and processed outputs of organic matter in urban waste flows, both solid and wastewater, under various scenarios**.**

- Quantity: Buzby et al. (2014); Cordell et al*.* (2009); CAR (2009); Brown et al. 2011), Baker (2011), Fissore (2011), EPA (2006); EPA (2015); Bellevi (2002) Bernstad & LaCour Jansen 2012; EPA CoEAT 2010; Jeavons (2012); Cornell Univ. (2014)
- Quality: Brobst (2016); Carballa et al. 2011; Morra et al. (2010); Dimambro et al. (2007); Koenig (2014); Zhang et al. (2011); Brinton (2000); EPA (2009b); EPA (2011) Schröder et al. (2011); Brown et al. (2015); Clark et al. (2006)

Organic Waste Quantity

Food Waste

Conversion Factors

1 lb. = 0.454 kg

A city of a given size in the U.S. will produce a certain quantity of food wastes, excreta, and yard wastes. Once processed using a variety of methods, the components may be suitable for use as soil amendments that enable crop production. Some waste processing methods, such as landfilling or incineration, do not readily allow for the recovery and beneficial reuse of plant nutrients. The first step in a life cycle assessment is to quantify the nutrients that can potentially be recovered from urban waste streams of the hypothetical city. Nutrient quantities, in turn, will determine the amount of food

that can be produced and the land area required to produce that quantity of food.

Using 2010 data, the Economic Research Service of the USDA estimated that annual food loss in the US was 429 pounds (195 kg) of food per capita, which amounts to 31% of the nation's food supply (Buzby et al., 2014). Importantly, the study only looked at food losses at the retail (10% of overall production) and consumer levels (21%). The study did not address farm level and farm-to-retail losses. Because retail and consumer level losses occur mainly within municipal boundaries (as opposed to far flung farms or along supply chains, for example), the USDA study provides a means to calculate the size of the food waste streams for the hypothetical city. A city of one million inhabitants will generate 195 million kg of food waste annually at the consumer (111 million kg) and retail (84 million kg) levels.

Limitations

The food waste estimates do not account for commercial food processing waste, such as from produce wholesalers or commercial food production facilities, that may be available in a given city. In that sense, the estimates are likely to understate food waste in many urban centers. According to EPA’s CoEAT (2010) model, the average amount of annual food waste for a city of one million is in the range of 255 million kg, accounting for all sources. However, many food wholesalers and processors divert food wastes for animal feed (a higher use), so some of this waste stream is likely not available for processing into soil amendments. (These and other “generating establishments” can be addressed on a custom basis for a given city if data are available).

In addition, since the estimate only includes food waste based on city population at the consumer and retail levels, it may understate the amount of food wastes available in large cities and regional centers. Regional urban centers may have greater daytime populations due to commuter influx, and typically draw significant numbers of visitors for sporting and other events. The food waste estimates do not account for meals consumed by in-commuting workers and visitors. On the other hand, “bedroom” communities with significant out-commuting may have lower levels of per-capita and total food wastes. In both cases, the actual levels of food wastes available for urban soil amendment and food production are likely to reflect actual food availability in a given city, accounting for commuter (in or out) and visitor demand. Therefore, food availability and food waste both track actual food consumption in the city, so urban food production potential will also track the actual need. Urban food production capacity depends on adequate quantities of food waste that can be turned into soil amendments, while consumption drives food availability needs from urban production. However, the land area requirement for urban food production (addressed later) will be context-dependent. Regional centers will require more cropland per capita, because, in effect, they have large (transient) populations consuming food.

Organics Processing

According to EPA (2015), the most common methods of food waste processing are landfilling, incineration, processing via sewer/WWTP, composting, and anaerobic digestion (direct, not via WWTP). Consumed food is the appropriate input for the

wastewater processing pathway. In US cities, food that is consumed is ultimately disposed of through centralized wastewater treatment processes. Since 31% of the food entering cities is wasted, it follows that 69%, or 434 kg/capita/year is consumed and excreta is processed at WWTPs. Therefore, the hypothetical city generates somewhat less than 434 million kg per year of excreta. It is important to note that mass balances of human excreta and WWTP influent waste streams will not correspond one-to-one with food consumption inputs. Human water intake and dilution from other sources (e.g., flushing water, toilet paper, etc.) in sewage collection and processing stages, as well as human energy derived from food consumption are the main reasons. In similar fashion, WWTP biosolids outputs have often been dried through various processes and may have lost (carbon) mass as a result of aerobic and anaerobic digestion, so corresponding WWTP output mass balances are unlikely to correspond with human food inputs (excreta) to the WWTP. Cities may also be "sinks" for excreta from meals consumed by in-commuters and visitors (see discussion above for food waste). Average annual per capita biosolids mass is used here as a proxy for food waste processed via WWTP. Since food waste is inseparable from other WWTP inputs once commingled, its proportionate contribution to biosolids outputs is not calculated here.

Yard Waste

Yard trimmings are another significant source of municipal organic waste. According to EPA (2006), yard trimmings contribute approximately 216 kg/capita annually to municipal solid waste (MSW) streams. Thus, the hypothetical city will generate approximately 216 million kg of yard trimmings annually. Although the composition of

this waste varies geographically and seasonally, EPA has adopted a standard composition of 50% grass clippings, 25% leaves, and 25% brush and branches. While a minority of states still allow landfilling of yard trimmings, many have banned the practice and have turned to composting or direct land application of shredded trimmings. The present study considers only composting as the method for yard waste processing.

Organic Waste Quality

Nutrient Levels

The levels and combinations of plant nutrients in soils and soil amendments are critical for food productivity in all systems. They are especially important for urban food production, due to its (presumed) high yield intensity and unique soils, which may be compacted, nutrient-poor, or contain contaminants. This section addresses the quantities of nutrients that exist in urban organic waste streams and waste-stream derived soil amendments.

Food waste contains 31% dry matter by weight, while the corresponding figures for yard wastes are 18%, 74%, 85%, and 30% for grass clippings, leaves, shrub trimmings, and tree trimmings, respectively (Cornell Univ., 2014). Except for grass clippings, these comprise the inputs for composting for production of soil amendments. Processing of food and yard waste incurs some losses of C and N mass, and reduces initial C:N ratios. Aerobically digested compost typically loses carbon mass (≈ 50%) and nitrogen mass (≈ 30%) as gases (Bellevi, 2002). (Open windrow composting, a common form of production, emits these GHGs to the atmosphere.) Phosphorus is conserved, while

overall mass is reduced by 36% (Bellevi), and volume decreases by 70% (Jeavons, 2012). Anaerobic digestion of food waste produces CH_4 (methane, or natural gas) from roughly 65 % of the C, with possible conservation of more nitrogen in the digestate (Bernstad & laCour Jansen, 2012; EPA CoEAT, 2010). Methane can produce energy, it can be flared (burned in open air without energy recovery), or lost to the atmosphere as a greenhouse gas. Table 1 lists estimates of annual quantities of food and yard wastes, including nutrient levels on a dry weight basis, for a city of one million people. This study assumes that all methods and combinations of methods for processing organic wastes yield roughly equivalent levels of soil amendments, those these may vary in relative nutrient composition depending on the processing method (see discussion below).

Average C and N concentrations in food waste on a dry weight basis are 36% and 2.4% respectively (Cornell Univ., 2014), and these values are used in the present study. The average P content in foods is 0.2% (Davidson et al. 2011, in Bernstad & laCour Jansen, 2012) to 0.3% (Ervin et al., 2004 in Fissore, 2011). According to the Food and Agriculture Organization of the United Nations, lawn clippings and leaves contain 0.30% P on a dry weight basis (FAO, 2004). The present study adopts 0.3% P levels for both food and yard wastes.

Data from a number of controlled studies provide average values used for modeling the C, N, and P concentrations (as % of dry weight) in composts derived from combined yard and food wastes. The average N level in Morra et al. (2010) in aerobic compost produced from a combination of food waste and yard waste was 1.8%, which falls within

the ranges in Dimambro et al. (2007). Koenig (2014) found a P concentration of 0.29% for digestate from “dry” anaerobic digestion of combined food and yard waste that was subsequently composted. (While C and N levels are reduced in both AD and aerobic composting digestion processes, P is conserved). This study will adopt 0.3% P and 1.8% N levels in finished composts (including composted anaerobic digestates) for modeling purposes.

Biosolids produced at WWTPs may have different nutrient levels depending on the combinations of treatment processes utilized. Average levels on a dry weight basis are 32% C, with N content of 4% and P content of 2.2% (Bob Brobst, US EPA Region 8, personal communication, March 25, 2016), and these averages are used for the biosolids modeling.

The C content of composts is typically specified in a ratio with N. Recommended C:N ratios for finished composts typically range from 15:1 – 20:1, because higher ratios may immobilize soil N, making it unavailable to plants (Dimambro et al., 2007). Morra et al. (2010) document an average C:N ratio of 17:1 over a multi-year study utilizing a mix of food and yard wastes. The present study assumes a C:N ratio range of 15:1 – 20:1 in finished digestates and composts.

Table 1- Annual Organic Waste and Nutrient Quantities for a City of One Million

Source	**Wet Weight Kg**	**Dry Weight kg**	**%C**	**C Flux to Organic Waste Stream kg**		**%N**	**N Flux to Organic Waste Stream kg**	**%P**	**P Flux to Organic Waste Stream kg**
Food Waste	195 million	60.5 million	36%	21.8 million		2.4	1.5 million	0.3%	0.18 million
Grass Clippings[a]	108 million	19.4 million	58%	11.3 million		3.4%	0.7 million	0.3%	.058 million
Leaves (avg. wet/dry)	54 million	39.7 million	49%	19.5 million		0.9%	0.4 million	0.3%	0.12 million
Shrub Trimmings	27 million	12.7 million	53%	6.7 million		1.0%	0.13 million	0.3%	0.038 million
Tree Trimmings	27 million	8.1 million	48%	3.9 million		3.1%	0.25 million	0.3%	0.024 million

Source: food and yard waste elemental concentrations and compost parameters from Cornell University compost calculator (2014) http://compost.css.cornell.edu/download.html
a - grass clippings are included to show nutrient levels, but are not composted – see sidebar

Mulch or Bag?
According to EPA, grass clippings comprise 50% of urban yard wastes – they would seem to be an abundant source of organic material for composts. However, clippings are best left on the turf for a number of reasons. 1. Clippings contain valuable nutrients that are ideal for growing more grass, and can substitute for synthetic fertilizers. 2. By covering soils, clippings can help to reduce water evaporation and reduce weed pressures. 3. Like food waste, clippings have a low C:N ratio, so clippings are not a useful addition to food composting given urban supplies of C and N. 4. Clippings are more likely to contain unwanted residues, due to pesticide applications and greater exposure to ground-settling contaminants from adjacent land uses. Therefore, clippings are not included in the compost modeling.

Scenario modeling assumes that 60% of food wastes in the city are recovered and recycled, combined with 50% of yard wastes, which includes 100% of leaves, and shrub and tree trimmings, but excludes grass clippings. Table 2 lists total annual mass of food and yard wastes, and C and N quantities in AD (food) and compost (food and yard waste) inputs, as well as estimates of resulting annual soil amendment production

potential.

In comparison, as noted above, average nutrient levels in WWTP biosolids are 32% C, 4% N, and 2.2% P. While C levels are similar to those in food wastes, N levels are 65% greater and P levels are nearly an order of magnitude greater. On average, across all combinations of processing methods, biosolids have an average C:N ratio of 8:1. These levels and ratios may also vary widely depending on specific WWTP processes and post-processing of the biosolids, such as drying or composting.

Table 2 -Annual Soil Amendment Potential (Dry Weight Basis) from 60% Food Waste and 50% Yard Waste (excluding grass clippings) – sums of masses from Table 1

	Total Mass kg	Carbon[a] kg	Nitrogen kg	C:N Ratio
AD and Compost Inputs	96.8 million	43.2 million	1.68 million	26:1
Soil Amendment Outputs	71 - 77 million[2]	17.7 – 23.6 million[b]	1.18 million	15:1 – 20:1

a - carbon comprises 58% of organic matter (OM) in soils and composts. Compost OM levels modeled here range from 43 – 53%.
b - Range of modeled average C loss in composting and AD (alone and combined) is 45-59%

Biosolids are produced from wastewater, and consist primarily of the residuals of human excreta from consumed food. In 1998, U.S. WWTPs produced 6.9 million (short) tons of biosolids on a dry weight basis (EPA, 1999). With a 1998 population of 270,248,003 (https://www.census.gov/population/estimates/nation/popclockest.txt), and a conversion factor of 907.2 kg/short ton, per capita annual biosolids production is 23.2 kg.

$$6.9 \text{ million tons X} \frac{907.2 \frac{\text{kg}}{\text{ton}}}{270{,}248{,}003 \text{ population}} = 23.2 \frac{\text{kg}}{\text{capita}}$$

This is within the ranges specified in Gomez, et al. (2010) and Rose, et al. (2015).

Table 3 - Annual Biosolids Production (Dry Weight Basis) for a City of One Million

	Total Mass kg	Carbon kg	Nitrogen kg	C:N Ratio
WWTP Biosolids from Excreta	23.2 million	7.3 million	0.9 million	8:1

Sources: total mass calculated from EPA (1999). C and N masses calculated from biosolids elemental concentrations from Brobst (personal communication 3/25/2016) – EPA DRAFT averages for all WWTP processes from 2006-2007 TNSSS

Results and Discussion

From Tables 2 and 3, it is clear that the city's organic waste sources have the potential to produce significant quantities of soil amendments and biosolids. Modeled quantities of food wastes and yard wastes (excluding grass clippings) create a compost mixture with a 30:1 C:N ratio, which is ideal for reaching the temperatures (131-139ºF, or 55-59ºC) required to meet EPA pathogen standards for high quality compost.

Nutrient ratios are important for plant growth. The 1.8:1 N:P ratio typical of biosolids may eventually result in high P accumulation in soils with regular applications designed to meet plant N requirements. To increase N, intensive production methods utilize N-fixing cover crops, which are grown in the off season and subsequently utilized as green manure or composted. However, animal manure applications, which have similar N:P ratios as biosolids, result in P overloading in large scale organic systems over time (Schröder et al., 2011). Excessive buildup can exacerbate P leaching and runoff. Composts have a much better average N:P ratio of 5:1, which results in less P buildup over time, and cover crops provide additional N.

In addition, levels of soil organic matter affect plant P use efficiency. Schröder et al.

(2011) cite an example in which P efficiency grew by a factor of nearly three, due solely to improvements in soil structure, when soil organic matter was increased from 1.5 to 2.4%. The foregoing factors suggest that greater P use efficiency from increasing organic soil matter content via biosolids may cause soil P overloading with repeated applications. This is less likely than when using mineral P fertilizers, but is a concern where soil matter may be subject to runoff erosion.

Carbon to nitrogen ratios affect the plant availability of N over time. Higher ratios (over 20:1) immobilize organic N in soils, while lower C:N ratios (15:1-20:1) typical of composts allow a measured mineralization of N, making it gradually available to plants. The 8:1 ratio of raw biosolids may make N too readily available, and may tend to increase N losses to the atmosphere, all else being equal. It is important to note that different combinations of processes at WWTPs, both for wastewater treatment and sludge handling, will create biosolids with differing nutrient characteristics. In addition, biosolids may be composted, or dried and pelletized, which will affect carbon and organics contents, as well as nutrient ratios. While the resulting nutrient quality of biosolids is context-dependent, the foregoing baseline analysis provides guidance on the potential differences between source separation and WWTP processing of food wastes for soil amendment.

There are alternative forms of organics processing, such as vermi-composting (using worms), or pyrolysis (high temperature combustion in the absence of oxygen). According to Lleó et al. (2013), vermicomposting creates a more nutrient-rich blend than

aerobic composting, with lower GHG emissions during processing. These and other alternatives are not considered in the scenario analyses.

Contaminants

Because the soil amendments considered here are utilized for food production, the levels and types of contaminants they may contain are important considerations. As with nutrient levels and balances, contamination will vary by the types of recovery and processing methods employed for food wastes.

In the U.S., compost quality is regulated by states, and at the national level by the U.S. Composting Council. Both generally follow the federal EPA CFR 40 part 503 Rule for biosolids (from sewage sludge). The U.S. Composting Council industry group has adopted the part 503 standards for compost, which address (9) heavy metals and pathogen levels only. The allowable levels of heavy metals in the U.S. are much higher than EU standards and individual Western European country standards (Brinton, 2000). The focus on heavy metals is "probably … a combined result of the well-established toxicity models existing for these compounds and their bioaccumulative character" (Carballa et al., 2011). Centralized waste processing methods are required to meet pathogen standards, so pathogen levels are not considered in the present study. However, unregulated home composting may result in pathogen levels that exceed regulatory standards, through insufficient temperatures and/or anaerobic conditions.

A hypothesis considered here is that compost made from food and yard waste

feedstocks may have lower levels of heavy metals than biosolids from WWTPs, since WWTPs typically have a wider range of inputs, including commercial and industrial sources, as well as urban runoff and infiltration. However, there is limited evidence for this hypothesis. According to Brinton (2000), compost made from source-separated organics can have heavy metals concentrations similar to levels in biosolids from WWTPs. In contrast, metals levels in compost made from mixed (not source-separated) solid waste are, on average, 4 times greater than levels in either composts or WWTP biosolids (Brinton). Fig.1 provides mean metals concentrations for biosolids and U.S. agricultural soils, as well as CFR 503 limits for biosolids. In addition, fig. 1 shows examples of metals levels from recent tests of a highly processed biosolids fertilizer (Milorganite®), a source-separated compost from Milwaukee, soil samples in Philadelphia, and kitchen food waste. Except for the WWTP biosolids means and agricultural soils, these results presented for illustration purposes; they are not national or local means.

U.S. data regarding average compost contaminant levels are not readily available; the listed concentrations are typical values from a composting operation in the Milwaukee (WI) region that handles urban waste streams. The U.S. Composting Council administers voluntary industry standards for compost. The Composting Council's Seal of Testing Assurance (STA) certification program awards the seal if the compost does not exceed any of the 503 rule limits for heavy metals, but the organization does not compile records of this information (Al Rattie, personal communication, August 24, 2015). STA certified commercial compost producers and WWTPs that produce biosolids

for land application will provide this information on request. The parity for some heavy metals between source-separated compost and WWTP biosolids may be due, in part, to the ubiquity of heavy metals in the environment, but this is a current research gap.

The European standards for compost are similar to the typical metals concentrations found in soils, while the U.S. limits are much higher (Brinton, 2000). The current Part 503 rules were promulgated in 1993, when average metals levels in biosolids were much higher. In the intervening years, industrial and other source control measures have resulted in lower levels for many of the metals. With the exception of copper and zinc, which come largely from plumbing systems, biosolids metals contents are now close to background environmental levels (Bob Brobst – U.S. EPA – personal communication, March 25, 2016). Rainwater captured for urban crop and compost production may also contain high levels of zinc from roofing and cladding materials, and also from galvanized conveyance infrastructure. These sources must be accounted for in designing rainwater capture and catchment systems for food production, and should be addressed as a matter of municipal policy for the long-term sustainability of urban food production.

It is important to note that the metals levels in biosolids, composts, and soils can vary greatly based on a number of factors. Of the metals, lead is perhaps the most-studied soil contaminant due to its former use in gasoline and paints, along with its significant deleterious effects on human health. Lead levels in urban soils vary greatly by location, but are generally highest in older cities where leaded gasoline and lead paint were used

over long periods. Studies cited in Brown et al. (2015) measured a range of 12-5210 mg/kg in total lead levels in urban soils, and many urban soils have lead levels above the 400 mg/kg EPA level of concern for direct exposure in residential soils (Clark et al. 2006). Brown et al. found a 58-305 mg/kg range of lead levels in surface soils in a single 30X30 meter plot in Kansas City, MO, which confirms that lead levels can be highly variable even over short distances. The bioavailability of lead via consumption of produce grown in urban soils is typically very low (Brown et al., 2015; Clark et al. 2006), and the overall risks across the range of contaminants typically found in urban soils is also generally very low (EPA, 2011). However, the precautionary principle suggests that achieving lower levels of contaminants is preferable. Likewise, soil amendments that decrease, rather than increase, soil contaminants are best over the long term.

Metals levels in biosolids and (presumably) composts may exhibit a wide range of variation. For example, the TNSSS found maximum lead levels in biosolids exceeded the mean by a factor of five. Results were similar for other metals, in which maximums exceeded mean levels by factors of 3 to11. Areas that were settled after national lead bans in paint, gasoline, and plumbing are likely to have lower lead levels than older cities. Very recently settled areas in which plastic plumbing pipes predominate may also have lower levels of copper and zinc in biosolids (although perhaps higher levels of other contaminants leached from the plastics). The foregoing underscores the need to research the contaminant levels in soils and soil amendments in the specific locations where urban food production is planned. However, as of 2010, few local jurisdictions required soil testing for urban agriculture (EPA, 2011).

As metals levels in biosolids have receded over time due to better source controls such as industrial on-site pretreatment, wastewater streams now contain increasing amounts and numbers of other contaminants as well. These include several classes of chemicals, in addition to “emerging contaminants” like pharmaceuticals, steroids, and hormones. The EPA's (2009b) Targeted National Sewage Sludge Survey (TNSSS) identified 145 contaminants of potential concern and sampled 74 large WWTPs in 35 states to determine their prevalence. An overwhelming majority of the analytes was detected in the sludge at a majority of the WWTPs. Almost all of the 28 metals and most of the chemicals were detected at a vast majority of plants. Results were somewhat more mixed for steroids and hormones, with the greatest detection variability for pharmaceuticals.

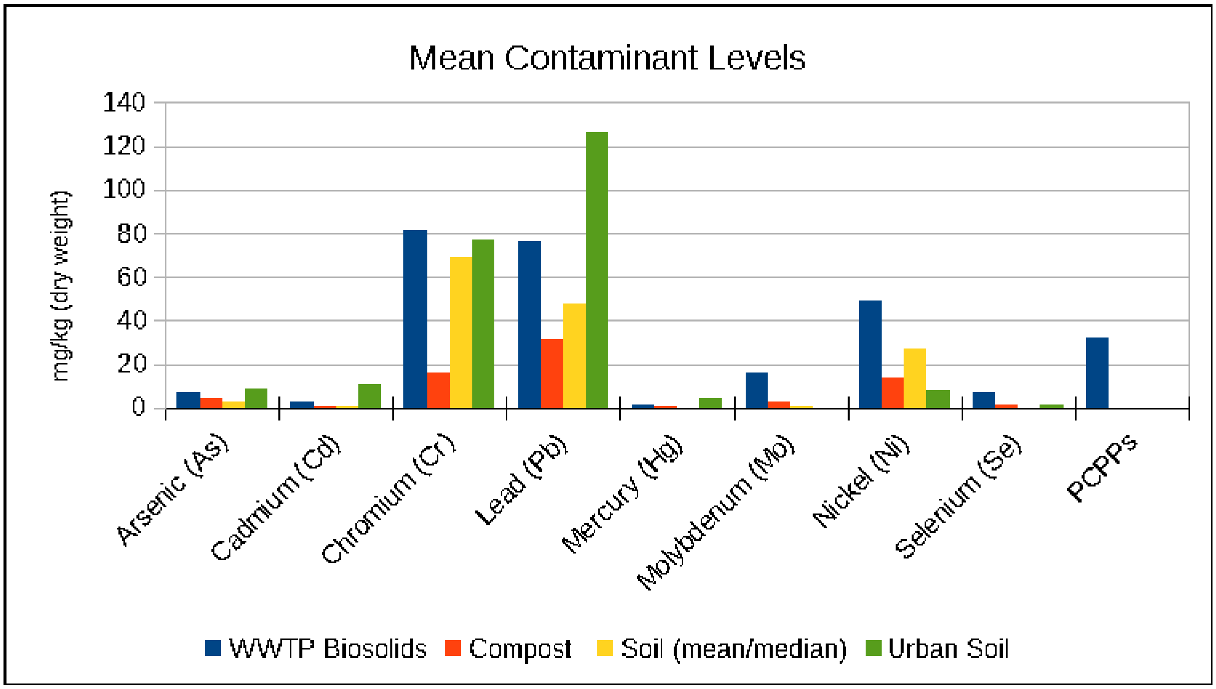
Figure 2 – Metals Limits and Levels for Biosolids and Compost

Regulatory (Federal)		**Environmental Levels - Examples**					
Contaminant	**EPA 503 Exceptional Quality biosolids limits mg/kg**	**TNSSS WWTP Mean mg/kg[a]**	**Milorganite (Biosolids) mg/kg[b]**	**Soil Median[c] mg/kg**	**Compost Mean mg/kg[d]**	**Urban Soil Mean mg/kg[e]**	**Kitchen Food Waste[f] mg/kg**
Arsenic (As)	41	7.1	8.1	2.7	4.6	9	*
Cadmium (Cd)	39	2.7	1.7	0.3	<0.5	11	1
Chromium (Cr)	N/A	81.5	217	69	16	77	3
Copper (Cu)	1500	558.1	254	21.6	33	51	31
Lead (Pb)	300	76.6	49	48	31.3	126	4

Mercury (Hg)	17	1.2	0.4	0.2	<0.6	4	2
Molybdenum (Mo)	40	16.3	10	0.9	2.6	0	
Nickel (Ni)	420	48.9	37	27	13.7	8	2
Selenium (Se)	36	7.1	1.5	0	<1.1	1	*
Zinc (Zn)	2800	993.7	498	153	112	165	76
PCPPs	N/A	30-35[g]	2[h]	*	*	*	*

a – TNSSS (2009) - 2006-2007 sampling
b – Milwaukee Metropolitan Sewerage District (2013)
c - Brobst, R.B. (2011)
d – representative STA test results for Milwaukee (WI) mixed food/yard waste compost
e - Kondo et al. (2016) – Philadelphia (PA) soils
f - Zhang et al. (2007)
g – McClellan & Halden (2010) - from 2001 TNSSS data
h - Snyder (2013) - triclosan only
* - no data, assumed zero or negligible

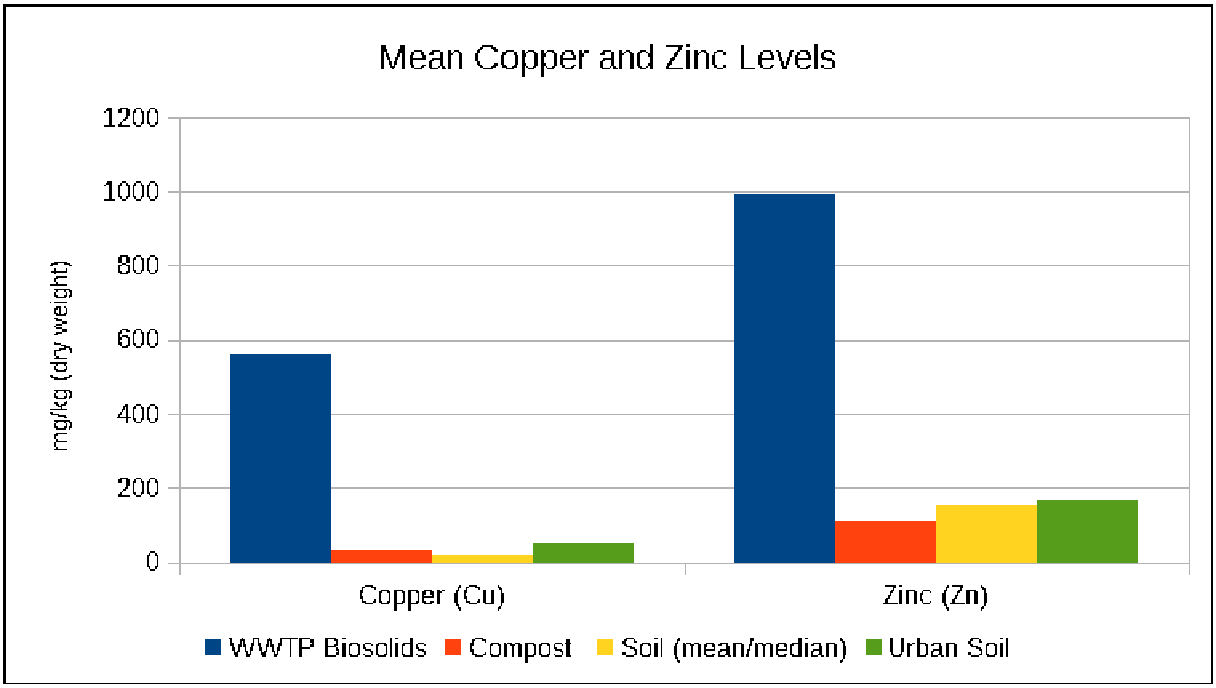
Figure 3 - Mean Contaminant Levels for Soils and Soil Amendments - Example Comparison (sources as listed in Fig. 2)



Results and Discussion

In these examples, food wastes are the “cleanest” source of urban organic wastes in terms of metals contamination, followed by source-separated compost. Biosolids are on average third best in terms of metals contamination, with very high levels of copper and zinc compared to the soils and compost included here. Food wastes also have lower levels of regulated metals than average soils metals levels, with the exception of cadmium and copper. This suggests that soil amendments made from food wastes may help to improve contaminated urban soils. On the other hand, the table shows that biosolids, as well as composts that include yard wastes, may exceed average soils metals levels. Therefore, land application of biosolids and mixed-source composts may have the potential to degrade native soils. Some researchers speculate that metals buildup in soils is naturally mitigated or that plant uptake eventually reaches a plateau, but studies are inconclusive or contradictory (Lu et al., 2012). EPA (2011b) suggests that both plant uptake and direct human intake of metals will be a small fraction of the contaminant levels in urban soils, and within safe health limits, if farmers and consumers follow good handling practices.

Figure 4 - Mean Copper and Zinc Levels in Selected Sources



Sources from Figure. 2

The rough similarity of metals contaminant levels in biosolids and compost is intriguing. Except for copper and zinc, which largely enter WWTPs from plumbing systems, metals in the compost from mixed food and yard wastes are generally on the same order of magnitude as the average levels in biosolids. Since food wastes are relatively free of metals, metals contamination is presumed to come from yard wastes. If yard waste metals levels are due to uptake from urban soils and environments, reductions may be difficult. On the other hand, attention to sourcing, collection, and processing methods may improve the purity of composts that contain yard waste. Examples include lawn pickup rather than street pickup, and research on possible contamination introduced via chipping and shredding machinery. Finally, the potential for compost and specially selected plantings to mitigate soil pollutants and plant uptake of contaminants may

outweigh their contributions to soil metals contamination. Zaccone et al. (2010) found that conventionally-grown semolina wheat had significantly higher levels of cadmium and chromium (but lower levels of nickel) than organically grown wheat in the same soils. This was despite significantly higher heavy metal inputs in the organic system. Accumulation will also vary by crop type and cultivar.

Cadmium, in particular, may be an issue in conventionally fertilized systems. Because it tends to be found in phosphate rock, it may be present at relatively high levels in p-containing fertilizers, leading some governments to establish cadmium limits for fertilizers (Roberts, 2014). The recent push among states to restrict the use of fertilizers derived from phosphate rock for residential use will presumably result in lower cadmium levels in municipal composts and urban crops produced from them. In addition, a number of studies suggest that organic crops may have lower cadmium levels than conventional systems, (Zaccone et al., 2010; Baranski et al., 2014) leading to a further advantage for (essentially organic) urban production. On the other hand, cadmium levels in fertilizers may pose little human health risk (Roberts, 2014).

There are currently few federal limits or standards in the U.S. for the other metals or classes of contaminants in WWTP biosolids, except for certain legacy contaminants such as PCBs. McClellan & Halden (2010) estimate that average combined levels of 72 pharmaceuticals and personal care products (PCPPs) in biosolids are on the order of 30-35 mg/kg, with the antibacterial compounds triclocarban and triclosan accounting for 65% of the mass. Potential ingestion pathways include consumption of food crops that

have taken up contaminants or direct exposure from soils amended with biosolids. In a similar vein, it is plausible that the pharmaceuticals, steroids and hormones, for which no limits currently exist, may have detrimental effects on plants grown for food, either directly or by affecting the complex web of soil bacteria, fungi, and species such as earthworms (i.e., the soil microbiome). Researchers have already demonstrated deleterious effects of emerging contaminants on aquatic food chains (Blair, 2015). Others have found that PCPPs may be persistent in biosolids-amended soils for many years (Walters et al., 2010). Canadian researchers have documented the degradation of certain emerging contaminants via composting of biosolids (CCME, 2010), and heat treatment may degrade some classes of compounds. Degradation is often compound-specific. However, given the uncertainty regarding the long-term and potentially cumulative soil effects of PCPP contaminants, keeping PCPPs out of soil amendments via source separation is arguably the most reliable option.

Source separation of organics, by avoiding the mixing of sewage and food wastes, is likely the most reliable option for reducing or avoiding these substances in soil amendments. Composts almost certainly contain lower levels of these new classes of contaminants than WWTP biosolids. The pathways of introduction to wastewater streams via human excretion, bathing, and direct disposal into sewerage systems are absent in the case of composts produced from food and yard wastes. The levels and potential effects of these contaminants on soil health, food crops, and human health are current research gaps, and beyond the scope of the present study. However, the precautionary principle would seem to give the advantage to source-separated solids

(AD digestates and composts - with careful sourcing of yard wastes) over WWTP biosolids. USDA national organic standards currently prohibit the use of WWTP biosolids in certified organic crop production. From a policy standpoint, the fact that many urban soils have historical contamination argues for the cleanest possible amendments, especially where food production or human contact with soil are concerned.

Conservation of Phosphorus – a Finite Resource

Phosphorus is a finite resource, one that is critical for food production, among other uses. It is also an environmental pollutant which can cause excessive algae growth (eutrophication) in surface waters. Eutrophication can create public health risks from human contact with algae, including ingestion of potable water. Algae can clog water intakes and create aesthetic issues, with potential economic consequences for treatment plants and regions that rely on tourism. Excessive algae can also degrade aquatic systems and harm aquatic life. While P is the “limiting nutrient” (its absence curtailing excessive algal growth) in many surface water systems, N may be the limiting nutrient in some contexts. N fluxes to the environment from food and yard wastes are highly variable depending on processing or disposal method(s), and are outside the scope of the present study.

Conventional agriculture relies on mined phosphorus as a fertilizer component. Because phosphorus is an element, it cannot be synthesized or replaced with another plant

nutrient. While P is relatively abundant in global ecosystems, known reserves of concentrated phosphate rock are finite and limited. Mined (mineral) phosphorus is found in quantity in just a few countries worldwide. Morocco (85%), China, the U.S., and Jordan account for 96% of known phosphate rock reserves (Cordell et al., 2011). As reserves are drawn down, the P fraction of phosphate rock is declining and becoming harder to extract, in processes that create radioactive and other waste byproducts. P recovery from organic waste streams is arguably a more sustainable option than reliance on mined P, and has the advantage of reducing eutrophication potential. Eventually, P recovery will become a necessity unless new phosphate rock reserves are discovered. This section quantifies the P content of urban organic waste streams. It also addresses the recoverability of P and the resulting quality from the processing methods considered here.

Since reserves of mineral P are finite, P scarcity will become a stronger driver of urban nutrient recycling in the future (Cordell et al., 2011). According to the USDA (2014), currently, over 50% of U.S. food waste is disposed of in ways that make nutrient recovery impossible. Since P conservation, on the one hand, and P pollution of (freshwater) surface waters, on the other, are significant concerns, it is important to quantify P mass balances in organic waste streams.

Methods and Sources

- Using data from the literature, this study quantifies the P content of urban waste streams and total fluxes from all streams for a city of one million. It ranks the

various alternatives of organics waste processing in terms of P recoverability (quantity and quality).

P in Waste Streams

On a global per capita basis, humans consume (in food) and excrete on average 1.2 g/day of P, with a range of 0.3 to 0.6 kg/year; the higher levels are due to higher meat consumption (Cordell et al., 2009). (Humans excrete virtually 100% of the P that is consumed in food). In the U.S., P intake per capita is 1.5 g/day (Dawson & Hilton, 2011).

$$1.5\,\mathrm{gcd}\,\mathrm{P}\ \mathrm{X}\ 365\,\frac{\mathrm{days}}{\mathrm{year}}\,\mathrm{X}\ 1{,}000{,}000\ \mathrm{population}{=}547{,}000$$

Therefore, average annual P excretion is roughly 0.55 kg/capita, or 550,000 kg for a city of 1 million. Summing the P fluxes for both food waste (derived from USDA food waste estimates – see Nutrient Quantity section above) and consumed food, total annual P flux to waste streams for the city is 726,000 kg. This is reasonably close to Fissore. et al. (2011) estimates of total annual P flux from food of 0.81 kg/capita, or 810,000 kg for a population of 1 million.

Adding the figure for yard wastes, annual P fluxes to waste streams are in the range of 1 million kg/year, or 1 kg/capita/year. For comparison, annual P fluxes in wastewater effluent from a WWTP (Milwaukee Metropolitan Sewerage District) with high P removal rates (87%) adjusted for a service area of one million population are approximately 90,000 kg (Kort, 2014). Typical WWTP removal rates will vary with the wastewater treatment method(s) and number of stages employed; Baker (2011) documented an

average 56% removal rate (year 2000) for WWTPs in the Minneapolis-St. Paul (MN) region. Table 4 gives estimated P fluxes from multiple sources for the city of one million; the estimate for WWTP effluent includes P from all influent streams, not just from food waste.

Table 4 - Annual Estimated P Flux - Organic Waste and WWTP Effluent (from food waste) for a City of One Million

Source	**Kg/year**
P in Food Waste	176,000
P in Excreta	550,000
P in Yard Trimmings	270,600
Total	**997,000**
WWTP Effluent (assumes 87% P removal rate)	90,000

From Table 4, it is apparent that the P contained in food and yard waste streams is comparable to wastewater effluent levels. Since P is conserved in AD and composting processes, soil amendments from urban food and yard wastes can contain significant quantities of P, compared to overall annual urban fluxes. If not well-managed from an erosion standpoint, amended soils have the potential to release this P into the environment, primarily via runoff to surface waters.

P Recoverability

Landfilling of organic waste sequesters 100% of P, and makes it unavailable for beneficial use, so it is the worst option from a P recovery standpoint. If leaching occurs, the leached P will end up as a water pollutant and can only be (partially) recovered when the leachate is treated. Landfill leachate treatment is required in the U.S., but

standards vary globally. Recovery of P from the incineration ash of organic wastes is a topic for ongoing research. While P can be recovered from the residue, incineration “results in phosphorus-containing material of the lowest agricultural quality. Incineration removes nitrogen and carbon from biowaste, resulting in an unknown recycling potential...” (Kalmykova et al, 2012).

A variety of processes are used to recover P from wastewater streams, including combinations of anaerobic and aerobic digestion, precipitation with metallic salts, and newer processes such as struvite (magnesium ammonium phosphate) extraction. The biological digestion processes have the advantage of producing a form of P that is highly beneficial for plant growth and less susceptible to groundwater leaching and runoff transport than mineral P (Chinault & O’Connor, 2008). One significant drawback of P recovery at WWTPs is the potential for adulteration by the various contaminants that EPA found to be common in sewage sludge (see discussion above). The actual levels of most of these contaminants in recovered P from precipitation and crystallization processes, and their potential effects on food crops, remain topics for further research. However, contaminant levels are likely to be similar to levels detected in WWTP effluents and biosolids.

Due to the concerns noted above, biological processing of source-separated organic wastes is likely to be the best option for P recovery for agricultural use. Anaerobic digestion and aerobic digestion (composting), produce P ideally suited for plant growth, while source separation minimizes the pathways for potential contamination compared

to WWTP processing. If yard waste is a component of the compost, one potential source of contamination is pesticide residues from the yard waste. However, this path is not unique to source-separated solids processing; runoff and infiltration can transport pesticides to WWTP influent streams as well.

Table 5 - P Recoverability From Various Organics Processing Methods – Quantity and Quality

Process (Source)	**Recovery Potential**	**Quality (contaminant load)**[b]	**Overall P Recoverability Rank**
Landfilling (Food Waste or Sludge)	0%	N/A	5
Land Application (Biosolids)	56% -87%[a]	Worst	4
Incineration (Food Waste)	100%	Unknown	[d]
Incineration (Sludge)	56-87%[a]	Unknown	[d]
Anaerobic Digestion (Food Waste)	100%	Best	1 (tie)
Composting (Food Waste)	100%	Best	1 (tie)
Composting (Yard Waste)	100%	Fair	2
Composting (Biosolids)	56-87%[a]	Poor[c]	3

a - some P is inevitably lost to WWTP effluent streams (this assumes effluent is not used for crop irrigation). Range calculated from Baker (2011) and Milwaukee Metropolitan Sewerage District (2013)
b – see nutrient quality section for contamination discussion
c – composting can reduce some chemical and emerging contaminants in biosolids/sludge
d – not enough information to rank

Results and Discussion

The analysis shows that the quantities of P in food wastes and yard wastes form a significant proportion of total urban P fluxes. Recovering this P is important both to conserve this critical resource and to help prevent eutrophication of surface waters.

Determining recoverability involves both quantitative and qualitative dimensions, i.e., the proportion of P that can be recovered from organic waste streams and its suitability for beneficial use in crop production. Organics processing methods vary widely along these dimensions, with landfilling and incineration generally performing the worst. The former sequesters 100% of P, while the latter produces ash with a potentially wide range of nutrient and contaminant levels.

Recovery of P via WWTP necessarily results in some losses to the environment in effluent, as well as contamination issues from mixed wastewater influent streams discussed previously. Therefore, WWTP processing ranks higher than landfilling or incineration on P recoverability metrics, because it enables greater recovery compared to the former while potentially producing a higher quality soil amendment than the latter. While technologies that enable higher levels of P removal from wastewater are under development, they are not yet widely employed due to higher costs and capital expenditures required for implementation. These technologies are not considered in the present study. Source separated processing of food and yards wastes provides the best overall P recovery. It preserves 100% of the P contained in the original materials, with the lowest potential for levels and types of contaminants in the resulting soil amendments.

As noted above, P can be a significant ecosystem pollutant. While landfilling and processing via WWTP both lose at least some P to the environment, soil amendments created from source separated processing, incineration ash, or biosolids all may release

P into the environment if they are washed into surface waters. This points up the importance of appropriate site management for both urban soil amendment processing and food production. The modeling in the present study assumes that no leaching occurs during soil amendment production employing practices such as pit and enclosed (in-vessel or covered windrow) composting. In the same vein, urban cropland can be sited and managed so that runoff potential is minimized.

Food Production Capacity and Comparisons Among Alternative Systems

The availability of adequate land area and sufficient quantities of quality soil amendments are the driving factors for assessing the feasibility of urban food production. This section focuses on the potential for urban fruit and vegetable production with comparisons to conventional and large-scale organic systems. Production potential estimates depend on yields per unit area and per unit soil amendment, but yields are not well documented in the academic literature. Several academic sources provide yield data for a number of crops, while a number of alternative sources provide a range of less carefully documented yield estimates for a wider range of crops. The present study assumes that yields obtained in season-extending hoop houses are representative of potential yields in urban agriculture. In contrast to intensive commercial greenhouse production, this study assumes that urban production does not rely on external energy inputs such as heat, artificial lighting, or CO_2 supplementation. Urban intensive production may employ “passive” sources, as when urban farmers co-locate composting and crop production to capture the heat and CO_2 released by aerobic fermentation.

Currently, urban agricultural production is heavily weighted toward fruits and vegetables due to their relatively high market values and adaptability to relatively small-scale and non-mechanized production methods. Perishability is another factor favoring localized production in these categories. In addition, transportation for fruits and vegetables comprises a much larger share of lifecycle (including production activities) GHG emissions (18%) compared to the 11% average across all food categories (Weber & Matthews, 2008). This makes localization of fruit and vegetable production potentially more beneficial from an environmental standpoint than for other crops.

Fruit and Vegetable Availability and Consumption

The Economic Research Service (ERS) of the USDA tracks per capita fruit and vegetable consumption in the U.S. and per capita supply (farm weight or primary availability). For the period 2004-2013, average daily per capita consumption of fresh vegetables was 0.11 kg, out of total daily vegetable consumption of 0.2 kg (ERS, 2015b). Total daily vegetable consumption provided 128 calories and was the equivalent of 1.7 servings. Daily per capita fruit consumption was 0.15 kg, including .06 kg of fresh fruit. Fruit consumption provided 81 daily calories per capita, and amounted to 0.8 daily servings. While the total calories from fruit and vegetable consumption are less than 10% of total calorie intake, these foods are not primarily consumed for their caloric value. Rather, fruits and vegetables provide a range of essential nutrients, including vitamins and polyphenols, that are not found in other foods. Dietary

improvements through increased fruit and vegetable availability are among the public health benefits often claimed for urban agriculture.

On an annual basis, per capita vegetable availability at the farm level for 2004-2013 was 181 kg (87 kg fresh), and fruit availability was 116 kg (58 kg fresh) (ERS 2015). Losses occurring along the supply chain and at the consumer level, including spoilage, inedible and uneaten portions mean that consumed weight is much lower than farm level availability. Assuming equal supply chain losses among alternative methods of production, farm level availability (primary weight) is the appropriate standard for determining the production potential of urban agriculture and for comparisons with conventional and large scale organic production.

Table 6 – Fruits and Vegetables: Annual Availability for the City of One Million and Daily Per Capita Consumption and Nutrition

	Annual Farm Weight Availability kg	Annual Urban Consumption kg	Per capita Daily consumption kg	Per capita Daily calories	Per Capita Daily servings
Total Vegetables	181 million	73 million	0.20	128	1.7
Vegetables (fresh portion)	87 million	40 million	0.11	49	0.9
Total Fruit	116 million	55 million	.15	81	0.8
Fruit (fresh portion)	58 million	22 million	.06	36	0.4

Sources: 2015 ERS food availability and loss-adjusted availability reports

Land Area Estimates

The National Agricultural Statistics Service (NASS) provides national level yield data for

conventional production for all crops, including the fruits and vegetables that can be grown in USDA plant hardiness zones 5-6. The Northern and Eastern industrial cities with surpluses of vacant land and concentrations of urban agriculture are largely found in these zones. The present study estimates cropland area requirements for conventional and large-scale organic production of fruits and vegetables for a city of one million. Similar studies that consider cropland area under both conventional and intensive forms of production for various levels of urban consumption provide a range of estimates that are used in the sensitivity analysis. Additional studies of documented intensive production yields across a limited range of crops provide a check on the yield estimates assumed in the city level studies.

Methods and Sources

- Yields per unit land area: the 2014 Agricultural Statistics Report published by NASS provides national (conventional production) yield data for vegetables and melons for the period 2004-2013, while NASS triennial Non-Citrus Fruit and Nut Summaries provide yield data for fruits for 2004-2013. All yields are converted to kg/ha.

Conversion Factors:

1 cwt (hundredweight) = 50.8023 kg

1 pound = 0.454 kg

1 acre = 0.409 ha

1 (short) ton = 907.185 kg

1 ft^2 = 0.0929 m^2

- The guide to Michigan produce published by Michigan State University (Colasanti et al., 2013) provides seasonal availability for fruits and vegetables grown in Michigan, which is utilized here as a reasonable proxy for USDA zones 5-6. It includes field-fresh, extended season, and storage seasonality. The combination of all three results in the greatest annual production potential, and corresponds with techniques common in intensive urban agriculture across cities, e.g., season extension via hoop house and greenhouses. (Will Allen of Growing Power 9/24/14 – Milwaukee School of Engineering presentation). Urban production modeled here uses values combining field-fresh and extended seasons, as well as storage for the 30 vegetables and 14 fruits included in the analysis[2]. This provides the most accurate picture of true production capability, in contrast to studies that do not consider seasonality. While virtually all vegetables in the ERS availability reports can be grown in zones 5-6, about half of the fruit varieties, including citrus and tropical types, cannot. The present study accounts for this factor.
- Yields per unit land area calculated from the NASS reports, annual per capita availability and seasonality values for each fruit and vegetable are used to calculate both urban production potential and land area needed under conventional yields for the city of one million.

[2] Asparagus, Bell Pepper, Broccoli, Brussel Sprouts, Cabbage, Carrot, Cauliflower, Celery, Collard Greens, Corn – Sweet, Cucumber, Eggplant, Escarole/Endive, Garlic, Kale, Lettuce – Head, Lettuce – Romaine & Leaf, Lima Bean, Mushroom, Mustard Greens, Onion, Potato, Pumpkin, Radish, Snap (Green) Bean, Spinach, Squash, Sweet Potato, Tomato, Turnip Greens, Apples, Apricots, Blueberries, Melons (Cantaloupe & Watermelon), Cherries (Sweet & Tart), Grapes, Nectarines, Peaches, Pears, Plums (& Prunes), Red Raspberries, Strawberries

Production potential kg/year (vegetables, fruit similar):

veg_x availability: annual per capita fresh weight production at the farm level to meet 100% of consumption needs

veg_x seasonality: annual potential production/utilization capacity in USDA zones 5-6, based on combined months of field production, season extension production, and storage. This period is divided by 12 (months) to arrive at an annual proportion.

$$\left(veg_1 \text{ availability} \frac{kg}{year} \text{ X } 1{,}000{,}000 \text{ population}\right) \text{X } \left(veg_1 \text{ seasonality} \frac{months}{12}\right) + \ldots$$

$$+ \left(veg_n \text{ availability kg X } 1{,}000{,}000 \text{ population}\right) \text{X } \left(veg_n \text{ seasonality} \frac{months}{12}\right) =$$

$$\text{annual production potential} \frac{kg}{year}$$

Cropland area requirement ha (vegetables, fruit similar):

$$\frac{\left(veg_1 \text{production potential kg}\right)}{\left(veg_1 \text{yield} \frac{kg}{ha}\right)} + \ldots + \frac{\left(veg_n \text{production potential kg}\right)}{\left(veg_n \text{yield} \frac{kg}{ha}\right)}$$

$$= \text{cropland area required ha}$$

Results and Discussion

Tables 5 and 6 summarize the results of the land area estimates and fruit and vegetable production potential of the present study and several comparable studies. Colasanti & Hamm (2010) estimated the produce production potential and cropland area requirements for Detroit’s population of 834,557 (2006) under various combinations of yield and production levels. Grewal & Grewal (2012) estimated the urban production capacity and cropland area requirements of Cleveland, OH, with a 2009 population of

431,363. All results listed here are normalized for a city population of one million.

Martellozzo et al. (2014) use global data (FAO) to estimate the amount of urban land required to meet 100% of urban vegetable consumption. This study estimates that U.S. cities would need 8.76% of urban land to produce enough vegetables for urban populations, assuming conventional U.S. yields (J-S Landry, personal communication, May 16, 2016). The U.S. city set included in the study had an average population density of 18.8 people/ha (J-S Landry, personal communication, June 6, 2016). Adjusting for the U.S. average 10 people/ha urban population density adopted in the present study results in an urban cropland requirement of 4.7% of the overall urban land area.

$$\left(\frac{8.76\%_{\text{croplandarea}}}{18.8_{\text{popdensity}}}\right) \text{X } 10.0_{\text{popdensity}} = 4.7\%$$

Urban land area calculation:

Conversion Factors

1 mile^2 = 260 ha

1 ha = 10,000 m^2

Given an average urban population density in the U.S. of 2,534 people per square mile (US Census 2010[3]), the city of one million occupies 395 square miles, or roughly 100,000 ha.

[3] Average population density of the nearly 500 largest “urbanized areas” (population > 50,000) as defined by the U.S. Census. Urbanized areas include both central cities and their surrounding areas. Population density generally tends to be higher with greater population, but there is wide variation among areas.

$$1{,}000{,}000 \text{ people}/\ 2{,}534\,\frac{\text{people}}{\text{mile}^2} = 394.6 \text{ mile}^2$$

$$395 \text{ mile}^2 \text{ X } 260\,\frac{\text{mile}^2}{\text{ha}} = 102{,}605 \text{ ha}$$

Assuming U.S. average urban population density, the land area required for urban vegetable production to meet 100% of consumption is 4.7% of 100,000 ha, or 4,700 ha.

It is important to note that population density is just one factor affecting the feasibility of urban crop production. Urban areas that support high population densities "vertically" (e.g., apartment and condominium towers) may contain significant amounts of vacant land. Conversely, areas with more single family and single story residences may have a relative lack of contiguous open land.

Table 7 - Land Area Estimates for Fruit and Vegetable Production for a City of One Million

Source	**Production Levels**	**Fresh US Conven-tional Yield ha**	**Fresh + Processed Conven-tional Yield ha**	**Fresh Intensive Urban Yield ha**	**Fresh + Processed Intensive Urban Yield ha**
Present Study	70% vegetables 34% fruit[1,2]	2,053	3,988	-	-
Colasanti & Hamm (2010)	76% vegetables 42% fruit[1]	1,748	3,495[3]	275-1012	550-2024[3]
Grewal & Grewal (2012)	100% vegetables 100% fruit[1]	7,692[4]	16,422[4]	6212[4,5]	11,802[4,5]
Martel-lozzo et al. (2014)	100% vegetables	-	4,700[6]	-	-

1 - includes only fruits grown in Michigan (Colasanti & Hamm) or Ohio (Grewal & Grewal)
2 - 17% including full consumption (citrus and tropical fruits) needs
3 - calculated from comments in study
4 - includes dry beans & peas
5 - intensive vegetable production and conventional fruit production

Seufert et al. (2012) provide data to convert conventional yields to large-scale organic production yields. For vegetables, large-scale organic yields per land area are 67% of conventional yields. For fruits, there are no statistically significant yield differences between organic and conventional yields. Another meta-study by Badgley et al. (2007) estimates that organic production yields are approximately 93% and 83% of conventional yields for fruits and vegetables, respectively. Ponisio et al. (2014), find very slightly reduced yields for organic production across a range of crops in developed countries (and slightly greater yields in developing countries).

Adjusted for the seasonally limited (see above) production potential determined in the present study under conventional yields, data in Seufert et al. result in the following high-range land area requirements for large-scale organic production:

Fresh Consumption:

$$1678\ ha_{vegetable}/0.67+375\ ha_{fruit}= 2{,}879\ ha$$

Fresh + Processed Consumption:

$$3051\ ha_{vegetable}/0.67+937\ ha_{fruit} = 5{,}491\ ha$$

Estimates from Ponisio et al (2014) result in low-range land area estimates that are essentially identical to those under conventional production.

Table 8 - Large-Scale Organic Production Land Area Estimates

Source	Production Levels	Fresh Consumption Large Scale Organic Yield ha	Fresh + Processed Consumption Large Scale Organic Yield ha
Present Study	70% vegetables 34% fruit[1]	2000 - 2,879	4,000 - 5,491

1 - 17% if citrus and tropical production needs are included

Urban Cropland Area Estimates

For conventional yields, results from the present study are comparable to the estimates of Colasanti & Hamm (2010), which used older ERS food loss estimates and average consumption levels from the period 1996-2006. Updated (2011) ERS food loss estimates used in the present study are higher for many fruits and vegetables (Muth et al., 2011). This may partially account for the slightly lower production potential and somewhat higher land area estimates in the present study. Results from the present study are also comparable to estimates in Martellozzo et al. (2014) for 100% vegetable production, but excluding fruit.

The much higher (≈ 4x) land area estimates in Grewal & Grewal (2012) reflect their adoption of 100% production levels (meeting all consumption needs); the high land area required to meet 100% of fruit production for the varieties grown in Ohio; and their inclusion of dry beans and peas for processed consumption. Beans and peas are more akin to "calorie crops," such as cereal grains, and are not included in the present study (or in the other studies referenced here). Fruit production accounts for 78% of total land area for fruit and vegetable production in Grewal & Grewal. Adjusting for the averages of vegetable and fruit potential found in the present study (70% and 34%, respectively) and Colasanti & Hamm (2010) (76% and 42%, respectively) would reduce the land area requirements in Grewal & Grewal significantly.

Urban food production utilizing high-organics soils and season-extending technologies such as greenhouses can be considerably less land-intensive than conventional food

production. There are a number of different sources for estimating intensive yields. Colasanti & Hamm (2010) utilize low- and high- “biointensive” yields from Jeavons (2012). The biointensive methods detailed by Jeavons (2012) are a set of relatively high-yield (compared to typical large-scale yields) production methods suitable for food production in urban contexts. The lower yielding version is applicable to beginning producers and/or production in new areas, while the higher-yielding version is achievable by experienced farmers in areas with longer production histories. Averaged over all vegetables, Jeavons’ low-biointensive and high-biointensive yields are roughly 125% and 525% compared to conventional yields, respectively. For fruits, yields under low-biointensive and high-biointensive production are 80% and 200% of conventional yields, respectively.

Grewal & Grewal (2012) estimate intensive vegetable yields using data from a range of studies and find that overall intensive production yields for vegetables are 240% of conventional yields. A number of other studies find urban agricultural yield levels across a range of vegetables that are somewhere between low-biointensive and high-biointensive yields, e.g., Vitiello & Nairn’s 2008 study of Philadelphia’s community gardens. Grewal & Grewal (2012) found average lower-than-conventional yields across Cleveland’s community gardens, underscoring the potential variability of yields in community gardens. However, the documented examples of higher production levels validate that intensive production levels are possible.

The intensive production yields in the studies above come from “grey literature,” which

is generally not from academic, peer-reviewed sources. There is a general lack of peer-reviewed yield data for urban agriculture across a range of crops (Martellozzo et al., 2014). However, peer-reviewed literature and university extension publications provide documented and estimated intensive production yields for select fruits and vegetables. For example, Morra et al. (2010) documented average hoop house tomato yields at a test site in Italy of 111,000 – 119,000 kg/ha over a four year trial. A high tunnel (hoop house) production manual produced by Iowa State University Extension estimated tomato production at 88,000 kg/ha (Everhart et al., 2010). These values are roughly three times the levels of conventional U.S. yields and two times Jeavons' low-biointensive levels for tomatoes. For fruits, the Iowa State publication estimates raspberry yields at over 25,000 kg/ha, compared to roughly 6,000 kg/ha for both conventional field and low-biointensive production.

The range of land requirement estimates in these studies is vast, ranging from 550 to 11,802 ha under intensive production. However, the foregoing analysis suggests a number of important criteria for determining a useful and realistic range for urban cropland requirements. The first is working within the biophysical constraints imposed by climate. The Cleveland study demonstrates that trying to meet 100% of produce (and especially fruit) consumption requirements raises land area requirements dramatically and disproportionately. Therefore, the present study assumes that meeting 70-76% of vegetable and 34% -42% of fruit consumption requirements (or 17% if citrus and tropical fruit consumption is included) is a reasonable goal for urban production in USDA zones 5-6.

Second, while production for just fresh produce consumption requires approximately half the land for production for fresh and processed consumption, the latter is more likely to be sustainable from an economic standpoint, so production for fresh and processed consumption needs are assumed in the present study. Production of value-added food products can result in greater income than fresh sales alone, and can provide greater profit margins to producers. Value-added products also often have extended shelf-life, which can help to counteract the seasonal nature of fresh-only farm income. Finally, the yield comparisons (albeit limited across crop types) among the values used in the urban production studies considered here and academic sources suggest that many of the grey literature values are within a reasonable range. In particular, yields in the academic studies fall between the low-biointensive and high-biointensive yields from Jeavons (2012). Therefore, the upper range estimate of roughly 2,000 ha from Colasanti & Hamm (2010), which is based on low-biointensive yields, is a conservative and reasonable low-end estimate of cropland area needed for both fresh and processed produce consumption.

Conventional production yields represent a reasonable low-range estimate for urban production yields, and, consequently, comprise the high end of land area requirements for urban production assumed here. Indeed, if urban production land requirements exceeded conventional rural production land requirements, that may in fact be an argument against urban production from a strictly environmental standpoint. Land area requirements under conventional production in the studies considered here fall within a

fairly narrow range, not including the outlier (Grewal & Grewal, 2012), which assumes 100% fruit and dry bean and pea production. Results from the present study for conventional yields lie between results for the other two non-outliers, and are based on the latest ERS and NASS data. Therefore, the conventional production estimated land requirement of approximately 4,000 ha, as determined in the present study, forms the upper bound of urban cropland area. Cropland required to meet 70-76% of the city's vegetable consumption and 34% (or 17%) of its fruit consumption ranges from 2,000 – 4,000 ha. This range almost certainly overestimates land requirements because it assumes that land is used for just one crop per year. In practice, annual rotations of up to three crops are likely because rotations can increase income, improve soils, and reduce pest and disease pressures (Magdoff & van Es, 2009). One example is lettuce in the spring, tomatoes in the summer, and a legume crop over the fall and winter (Morra et al., 2010).

The range of 2,000 – 4,000 ha is used for testing the framework and general feasibility analysis of urban food production. Actual yields, and thus land requirements, are affected by a host of contextual factors, including soil type and fertility, farming practices, cultivar types, and others. These estimates are not meant for use in production planning, which would require, at a minimum, specific data on all of the factors above. Outputs from land area estimates (including resulting compost depth/volume for urban intensive production) are also inputs for other model sections, including green infrastructure potential and soil water-holding capacity.

Visualizing Hectares
A hectare is an SI unit of land equivalent to 10,000 m^2. It is also equivalent to 2.47 acres. For non-farmers, some urban context may help. A hectare is approximately the size of an average urban block, to the centerlines of the surrounding streets. Block sizes vary +/- 40% among large cities – Sacramento (CA) blocks are right on the money, while Portland (OR) blocks run small and Houston (TX) blocks run large. Another visualization aid comes from the world of sports – a standard football field is approximately 0.9 acre, so the area of three football fields is slightly larger than a hectare.

Soil Amendment Requirements – Initial and Annual

Sustainable crop production requires long-term maintenance of soil fertility, without a buildup of excess nutrients. Nitrogen and phosphorus levels, in particular, that exceed plant needs may be susceptible to leaching and runoff. Long-term soil degradation, a serious problem in conventional agriculture, is a combination of reduced organic matter along with a buildup of excess N and P (Magdoff & van Es, 2009). Topsoil erosion and soil compaction also contribute to soil degradation. A number of studies suggest a range of parameters for achieving intensive fruit and vegetable yields while maintaining long-term soil fertility, and these are applicable to urban production with season extension via hoop houses.

Soil Fertility – Initial Soil Amendment

Urban soils are likely to have lower fertility than required for intensive production, especially if the organics- (and contaminant) rich topsoil layer is removed. Therefore, initial soil amendments may be required to achieve a target level of soil organic content. In addition, annual applications of soil amendment make intensive yields possible, while maintaining target soil fertility levels. In practice, “soil building” to reach maximum urban vegetable and fruit production potential may be phased over time, as cropland is brought into production and soil fertility is gradually increased. The following analysis

demonstrates a range of overall soil amendment needs, but “initial” soil building need not occur before food production can begin.

Methods and Sources

- Initial and annual soil amendment requirements – dry matter basis:
    - Model inputs include mass and volume of processed urban organic wastes, with N, P and organic matter (OM) levels, as well as initial OM levels of existing soils. Outputs include required mixing ratios of soil and amendments.
    - Agricultural soil organic matter (SOM) ranges from 1-6% on a dry matter basis (Magdoff & van Es, 2009). The present study assumes that intensive yields require an SOM level near the top of this range. It sets a target of 5% SOM. Morra et al. (2010) demonstrate intensive yields while maintaining 4.4 % SOM (2.6% carbon) over a four year trial under annual three-crop rotations (lettuce, tomato, snap bean). Jeavons (2012) suggests that intensive yields require 4-6% SOM. Heavily depleted soils may contain as little as 0.5% SOM in the top 30 cm (0.30m) root zone (Magdoff & van Es 2009), and that is the lower limit of the initial value for urban soils assumed here. The upper limit of initial values is set at (still marginal) value of 1.5%. (SOM is not the same as the soil carbon level: C comprises 58% of SOM, so SOM is always greater than C).

To calculate soil amendment volumes, the cropland required area (in m^2) is first multiplied by the 0.30 m standard rooting depth commonly used in soil analysis.

Cropland range required for the city of one million:

$$2{,}000\text{-}4{,}000 \text{ ha } X \ 10{,}000 \frac{\text{m}}{\text{ha}} = 20{,}000{,}000 - 40{,}000{,}000 \text{ m}^2$$

Rooting depth (soil + soil amendment) volume:

$$20{,}000{,}000 - 40{,}000{,}000\ m^2\ \underline{X\ 0.30\ m =}\ 6{,}000{,}000 - 12{,}000{,}000\ m^3$$

Bulk density (BD) is a measure of the dry mass-to-volume ratio of soils in g/cm^3 (t/m^3). While bulk density for composts is usually given on a wet weight basis, the present study adopts dry weight BD as the measure for both soils and composts to facilitate calculations. Bulk density is affected by both inherent soil properties such as the density of the mineral portion and organic matter content, and by contextual factors such as degree of compaction. According to the USDA's Natural Resource Conservation Service (NRCS, 2008), bulk densities for crop growth range from < 1.1 – 1.47 g/cm^3 (clay soil) to < 1.60 – 1.80 g/cm^3 (sandy soil). Silty (loam) soils fall between these values.

For calculating required soil amendment mass, the SOM and BD ranges of native urban soils, and soil amendments from urban organic wastes, are characterized as follows:

Native urban soil organic matter: 0.5%, (0.005 g/cm^3) – 1.5%, (0.015 g/cm^3)

Compost average organic matter: 48%, or 0.48 g/cm^3

Final SOM (amended soil) target: 5% (.05 g/cm^3)

Native urban soil bulk densities (BD): 1.3 g/cm^3 (clay) – 2.1 g/cm^3 (sandy).

Soils with higher bulk densities that cannot be lowered via deep tilling (subsoiling) may not be suitable for crop production.

Soil amendment (compost) values: BD = 0.5 g/cm^3

To meet SOM .05 target for initial soil SOM of 0.5% and BD of 1.3:

$$0.005 \times 1.3 \text{ (soil mass)} + .48\,a \text{ (amendment mass)} = 0.05(1.3+a)$$

$$0.0065+0.48\ a= 0.065+(0.05\ a)$$

$$0.43\ a=.0715$$

$$a= 0.17$$

Final mix = 83% soil + 17% amendment

Final bulk density: 1.3 BD soil X 0.83+0.5 BD amendment X 0.17= 1.16

Initial soil amendment required for 6 x 10^6 m^3 cropland (intensive yield assumption), assuming initial soil bulk density (BD) of 1.3, and soil amendment BD of 0.5 g/cm^3 (500 kg/m^3), with a final calculated BD of 1.0:

$$\left(6 \text{ X } 10^6\right) \text{X } 0.17 \text{ X } 1160 \frac{kg}{m^3} = 1.2 \text{ X } 10^9\text{, or 1.2 billion kg}$$

For 12 X 10^6 m^3 (conventional yield assumption), the initial soil amendment required:

$$\left(12 \text{ X } 10^6\right) \text{X } 0.17 \text{ X } 1160 \frac{kg}{m^3} = 2.42 \text{ X } 10^9\text{, or 2.4 billion kg}$$

For soil with an initial SOM of 0.5% and BD of 2.1 g/cm^3 (highly compacted):

0.005 X 2.1 (soil mass)+ .48 X a (amendment mass)= 0.05(2.1+a)

$$0.011+0.48\ a = 0.105+(0.05 \text{ X } a)$$

$$0.48\ a- 0.05\ a = 0.105-0.011$$

$$0.43\ a= 0.094$$

$$a= 0.22$$

Final mix = 78% soil+ 22% amendment

Final bulk density: 2.1 BD soil X 0.78+0.50 BD amendment X 0.22= 1.75

Initial soil amendment required for 6 x 10^6 m^3 cropland (intensive yield assumption), assuming initial soil bulk density (BD) of 1.3, and soil amendment BD of 0.5 g/cm^3 (500 kg/m^3), with a final calculated BD of 1.35:

$$(6 \text{ X } 10^6)\text{X } 0.22 \text{ X } 1350\,\frac{\text{kg}}{\text{m}^3} = 1.8 \text{ X } 10^9\text{, or 1.8 billion kg}$$

For 12 X 10^6 m^3 (conventional yield assumption) initial soil amendment requirement:

$$(12 \text{ X } 10^6)\text{X } 0.22 \text{ X } 1350\,\frac{\text{kg}}{\text{m}^3} = 3.6 \text{ X } 10^9\text{, or 3.6 billion kg}$$

For soil with an initial SOM of 1.5% and BD of 1.6 g/cm^3:

$$0.015 \text{ X } 1.6 \text{ (soil mass)} + .48\text{ a (amendment mass)} = 0.05(1.6+\text{a})$$

$$0.024+0.48\text{ a} = 0.08+(0.05\text{ a})$$

$$0.43\text{ a}=.056$$

$$\text{a} = 0.13$$

Final mix = 87% soil+ 13% amendment

Final bulk density: 1.6 BD soil X 0.87+0.5 BD amendment X 0.13= 1.46

Initial soil amendment required for 6 x 10^6 m^3 cropland (intensive yield assumption)

$$(6 \text{ X } 10^6)\text{X } 0.13 \text{ X } 1460\,\frac{\text{kg}}{\text{m}^3} = 1.1 \text{ X } 10^9\text{, or 1.1 billion kg}$$

For 12 X 10^6 m^3 (conventional yield assumption) initial soil amendment requirement:

$$(12 \text{ X } 10^6)\text{X } 0.13 \text{ X } 1460\,\frac{\text{kg}}{\text{m}^3} = 2.2 \text{ X } 10^9\text{, or 2.2 billion kg}$$

The analysis above characterizes the amounts of soil amendment needed to bring compacted and marginally fertile urban soils into suitable condition for sustainable and intensive vegetable and fruit production. A realistic range of soil bulk densities and organic matter levels, coupled with land area estimates from the previous section suggests that 1.1 – 3.6 billion kg is a likely range of initial soil amendment.

Soil Fertility and Yields – Annual Soil Amendment

In addition to the amendment required for initial soil-building, annual application of soil amendment supports intensive yields and maintains soil fertility over time. Several sources suggest typical ranges of annual soil amendments that will meet these goals. Morra et al. (2010) evaluated 15 t/ha, 30 t/ha, or 45 t/ha (dry weight) annual compost application derived from 50% food waste and 50% yard waste to soils with 4.4% SOM. The compost (1.8% N, 51% OM, 30% C) was utilized in a controlled hoop house trial in a Mediterranean climate (USDA zone 8-9 equivalent). Over the four year trial, tomato yields under the 15 t/ha scenario were 111,000 kg/ha, or three times U.S. average conventional yields of 37,000 kg/ha. Compared to 15 t/ha, tomato yields increased 7.5% in the 30 t/ha scenario, but just 5% with a compost application of 45 t/ha. This suggests a non-linear relationship between compost levels and yields, with peak yield/compost application rates somewhere between 15 t/ha and 30 t/ha, at least for tomatoes under the test conditions. Compared to the 15t/ha application rate, lettuce yields increased 25% under both the 30 t/ha and 45 t/ha scenarios, but yields were just 45 – 70% of U.S. conventional yields. Snap bean yields were comparable to U.S. conventional yields, and did not vary significantly among application scenarios. Lettuce and snap beans were grown in an annual rotation with tomatoes, so all three shared the same cropland area over a twelve month period. In this sense, annual yields per unit land area were much higher. Together, these results suggest that yields do not necessarily increase with increases in annual compost application, and that yield responses are crop dependent.

Turning to soil health, the Morra et al. (2010) study found that soil organic carbon levels

remained stable over the four-year trial period under 15 t/ha annual compost application rates, and increased somewhat at the higher application rates. Soil respiration and soil carbon losses (via respiration) both increased at higher application rates. The 15 t/ha application rate maintained the most stable soil conditions, lowest overall carbon losses, and highest yields per unit of compost applied.

Everhart et al. (2010) provide data on compost requirements based on crop N requirements for high tunnel (hoop house) tomato production at Iowa State University (USDA zones 4-5). They suggest that 322 gallons (984 lbs. dry weight) of compost derived from bedded cattle manure (1.4% N, 14:1 C:N ratio) is sufficient for 2,016 ft^2 of hoop house production area.

Conversion factors:

1 kg = 2.2 lbs.

1 ha = 107,637 ft^2

Recommended annual compost application rate (dry weight basis):

$$\frac{984\,\frac{\text{lb}}{2{,}160}\,\text{ft}^2}{2.2\ \text{lb/kg}} \times\ 107{,}637\,\frac{\text{ft}^2}{\text{ha}} = 22{,}288\ \text{kg/ha}$$

This is roughly in line with the results in Morra et al. (2010), and amounts to 0.07 m (0.29 inch) depth over the entire cropland surface area. The guidance in Everhart et al. assumes that only 20% of N is plant available in the first growing season, and estimates compost needs accordingly. (This rate of application likely overestimates the amount of compost needed for urban production, because the N content of manure-derived

compost is slightly lower, at 1.4%, than food/yard waste compost). The expected tomato yield is given as 5,200 lbs/2,160 ft^2:

$$\frac{5{,}200\,\dfrac{\text{lb}}{2{,}880}\,\text{ft}^2}{2.2\ \text{lb/kg}}\text{X}\ \ 107{,}637\,\frac{\text{ft}^2}{\text{ha}}=88{,}388\ \text{kg/ha}$$

This yield is over twice the average U.S. commercial yield of 37,000 kg/ha.

The Iowa State Extension study estimates hoop house raspberry yield at 1,440 lbs/2,700 ft^2:

$$\frac{1{,}400\,\dfrac{\text{lb}}{2{,}700}\,\text{ft}^2}{2.2\ \text{lb/kg}}\text{X}\ \ 107{,}637\,\frac{\text{ft}^2}{\text{ha}}=25{,}369\ \text{kg/ha}$$

This yield is over 3.5 times the average U.S. conventional field yield of 6,920 kg/ha.

Iowa State compost application recommendations are roughly in line with the results in Morra, et al. This is equivalent to 0.007 m (0.29 inch) depth on a wet weight basis over the entire cropland surface area. Jeavons (2012) recommends 0.25 – 0.5 inches of compost application annually for the range of low to high biointensive yields. Results are summarized in Table 9.

Table 9 - Estimates of Annual Soil Amendment Requirements for Sustainable Intensive Production

Source	Compost (dry weight) kg/ha	Compost Depth (wet weight) m/ha	Crops	Notes
Morra et al. (2010)	15,000	-	Tomato, Lettuce, Snap Bean	Maintains SOC levels
Everhart et al. (2010)	22,288	0.007	Tomato	Based on crop N need – manure compost N content is lower than food-yard waste compost
Jeavons (2012)	-	0.006 - 0.012	Average for all fruits and vegetables	Range of low- and high biointensive yields

The data in Table 9 suggest that annual soil amendment requirements range from 15,000 kg/ha to 25,000 kg/ha. These levels maintain SOM and support annual yield levels under intensive production, including multiple annual crops in rotation. Table 10 lists land area and annual soil amendment requirements for the city of one million, along with annual soil amendment production potential.

Results from Morra et al. (2010) suggest a mass balance method to estimate annual soil amendment needs. At the 15,000 kg/ha annual compost application rate, average plant dry matter mass over the study period was 10.2 t/ha (tomato), 1.05 t/ha (lettuce), and 3.75 t/ha (estimated mean for snap bean).

Converting to kg and summing:

$$10{,}200\ kg_{tomato} + 1{,}050\ kg_{lettuce} + 3{,}750\ kg_{snap\ bean} = 15{,}000\ kg$$

The plant dry matter content exactly balances the dry matter content of the annual compost application in this study. Doubling the compost application to 30,000 kg resulted in plant dry matter yield increases of just 2% for tomatoes and 5% for lettuce (snap bean data not provided). These results imply that optimal annual compost applications result in an equivalent plant dry matter yield, once adequate SOM is achieved and maintained. Belitz & Grosch (1987) and Kirk et al. (1991) provide average dry matter content for the range of fruits and vegetables considered for urban production in the present study. Calculated annual per-capita dry matter content (fresh farm weight) for fruits is 4.3 kg, and 14.7 kg for vegetables. For the city of one million, this amounts to 19 million kg of produce on a dry matter basis annually. If soil dry matter amendment requirements match the dry matter content of crops, this suggests that 19 million kg of soil amendment are needed on an annual basis (see Appendix A).

Combining the land area and soil amendment estimates per unit area (low range estimate):

$$15{,}000\frac{kg}{ha}\ X\ 2{,}000\ ha = 30\ million\ kg$$

Combining the land area and soil amendment estimates per unit area (high range estimate):

$$22{,}288\frac{kg}{ha}\ X\ 4{,}000\ ha = 89{,}152\ million\ kg$$

The analysis above represents a "shorthand" method of mass balance accounting, and does not consider all the modes of action for carbon cycling in agro-ecosystems, such

as plant CO_2 fixation. It may be sufficient for establishing a data point (in this case, the low-range estimate) for annual compost needs to meet a target production levels. Tools such as Agro-IBIS allow for more sophisticated modeling of a range of nutrient cycling processes.

Table 10 - Land Area and Soil Amendment Quantities for a City of One Million

Production Mode	**Land Area (Ha)**	**Productive Soil Volume m^3**	**Initial Soil Building Amendment (dry weight) kg**	**Annual Land Application (dry weight) kg**	**Annual Soil Amendment Production Potential (dry weight) kg**
Urban Biointensive	2,000 – 4,000	6-12 billion	1.1 billion – 3.6 billion	19[a] – 89 million	71 – 77 million
Conventional Rural	4,000				
Large-scale organic	4,000-5,500				

a- based on fruit and vegetable dry matter mass balance method

Results and Discussion

The overall analysis suggests that cities could support fruit and vegetable production at significant levels, from both land requirement and soil amendment perspectives. Land requirement would be lower than either conventional or large-scale organics production requirements. Yields in urban food production are not yet well-studied or documented, but the methods adopted here suggest means to estimate plausible yields based on a convergence of evidence from varied sources. The upper ranges listed for initial and annual soil amendment underscore the importance of achieving higher-than-conventional yields in urban production. The high-end estimates for both land area and annual soil amendment requirements are based on conventional yields, and results

suggest that the city's annual soil amendment production potential would not be sufficient to meet target fruit and vegetable production needs at these lower yield levels. In addition, high yields confer a range of potential additional benefits, including higher economic returns and lower land costs per unit production. The high end estimates for soil amendments may be reduced under the assumption that lower yield levels would also require lower annual soil amendment application per unit production and unit area.

At lower range estimates of annual soil amendment needs, production potential more than covers annual requirements at full targeted food production levels, with additional capacity to build soils over time. Utilizing WWTP biosolids could accelerate soil building by increasing the overall amount of soil amendments available. Aerobic composting of the biosolids would likely create the most beneficial form of soil amendment, albeit at the risk of introducing more and different types of contaminants compared to source separated organics-derived soil amendments. As noted previously, nutrient balances in biosolids may also be less advantageous compared to SSO amendments. This may be acceptable for building soil organics levels in the short term, followed by annual additions of "cleaner" SSO amendments to maintain fertility and yields over the long term.

It is important to note that, in practice, potential soil amendment quantities may be lower than estimated above. A proportion of biosolids, digestates, and composts may be unusable due to contamination of some batches. In these cases, the resulting material may have to be landfilled because it does not meet contaminant limits. SSO processing

is not a panacea, but it may have less potential for contamination than the relatively accessible sewer systems that feed WWTPs. Contamination considerations highlight the importance of knowing the provenance and full custody chain of organic wastes destined for soil amendments and food production.

Energy and GHG Emissions Balances

Waste processing, food production, and transportation (all phases of processing and production)

Energy use and greenhouse gas (GHG) emissions are potentially significant aspects of urban organic waste processing and food production. For example, the U.S. EPA (2016) estimates that agricultural production accounted for 8.4% of the nation's total GHG emissions in 2014. This share was equivalent to 572 million megatons CO_2 equivalent ($MTCO_2$-e). It can be challenging to generalize about the energy and emissions performance of organics waste processing, due to the range of combinations of processes and feedstocks that are available. Recent meta-studies by Yoshida (2013) and Morris et al. (2013) allude to the difficulty of reaching generalizable conclusions because of differences in methodologies and system boundaries across waste processing studies. For example, co-processing food wastes through anaerobic digestion at wastewater treatment plants may entail a combination of aerobic, anaerobic and biosolids processing methods. EPA has developed a Waste Reduction Model (WARM) that reduces these uncertainties by standardizing many of the assumptions involved. In addition to customizable input parameters, WARM also includes default national averages for many parameters, and these are used in the present study.

Similarly, the energy and emission performance of food production systems can be difficult to quantify due to the complexity of the systems involved (Nesheim et al., 2015). The primary emissions components of crop-based agricultural systems are nitrogen compounds; agriculture accounts for nearly 80% of the country's N_2O emissions (EPA, 2016). Nitrogen emissions are highly context dependent, based on field tillage and fertilizer application practices. N may also be applied in a highly volatile gas form. Although the N in composts and biosolids may be less concentrated and labile than N in mineral or gas fertilizers, this advantage may be lost through inappropriate urban field practices or over-application of soil amendments. Anaerobic digestion, in particular, can create highly volatile forms of nitrogen. The variability of field and application practices, and the resulting agricultural N_2O emissions, are outside the scope of the present study, except for anaerobic processing (see below).

In a comprehensive study of GHG emissions in the food system, including transportation of inputs and outputs, Weber & Matthews (2008) demonstrate that production of animal products (meat, poultry, dairy, fish, and eggs) is by far the largest contributor to GHG emissions. In their study, fruit and vegetable production (including transportation) accounted for approximately 10% of food-related emissions. Given that agriculture accounts for 8% of overall emissions, this equates to just 0.8% of overall emissions from all sectors. Therefore, the contribution of urban food production is likely to play a very small role in the overall emissions picture.

Existing models and data from the literature are linked here to estimate life cycle

emissions of GHGs across a range of waste processing and food production systems, including transportation of inputs and outputs. The WARM model used here for estimating GHG emissions from processing of organic wastes includes N_2O emissions for anaerobic processing and land application of the resulting digestate.

Methods and Sources

- Organic waste processing: the EPA WARM model calculates energy and GHG emissions balances for processing via landfilling and centralized composting methods, including the energy use for transportation of organic wastes. WARM also accounts for soil C storage and synthetic (mineral) fertilizer substitution. The EPA CoEAT models energy data for WWTP (anaerobic) co-processing.
- Urban food production: data from the Congressional Research Service (Schnepf, 2004) details energy use in U.S. agriculture. Pearson (2007) and Gomiero et al., (2008) provide estimates of energy differences between conventional and organic production. EPA (2016) gives an estimate of GHG emissions form the agricultural sector. Weber & Matthews (2008) provide information on food system GHG emissions, including the proportion for fruit and vegetable production and transportation-related emissions. Garnett (2008), Foster et al. (2006) and Smith et al. (2005) provide emissions estimates from the European (UK) context. Hendrickson (1994) summarizes food system energy use by category in the U.S. context.

Energy production and use have several facets that must be integrated to form a true

picture of life cycle net energy balances and comparisons between alternate methods of both organic waste processing and food production. The various methods used in organic waste processing, along with the potential for carbon sequestration in soils and soil amendments, may sometimes result in the net creation of energy and net mitigation of GHGs. Some methods of organic waste processing, such as anaerobic digestion, have the potential to generate electric power, produce heat, and natural gas (biogas) for use as a fuel. A proportion of the methane from landfills may also be captured for the same energy-generating uses. These processing methods may therefore create energy as a byproduct. Other processing methods, such as commercial scale windrow composting, may utilize energy for turning piles, aerating piles, and irrigation to maintain optimum moisture levels. The result is net energy use. The energy used for transporting wastes from generation site to processing site and finished soil amendments to point of use is also a component of the net energy balance for the various processes. Net energy creation and GHG mitigation are indicated by negative values in the tables, while net energy use and GHG emissions are indicated by positive values in the tables.

The transportation component of organic waste processing is accounted for in existing models. EPA's Waste Reduction Model (WARM ver. 13) calculates average energy use for common transport distances for organics disposal methods, including landfilling and composting. This study uses U.S. averages from EPA's WARM model for waste transport distances (20 mi./32 km). The present study estimates that (shorter) nutrient and production loops for urban waste and agriculture are five miles (8 km). Table 11 shows the WARM results for net energy and GHG emissions for various methods of

processing 100% of the city's food and yard wastes.

Conversion Factors

1 kg = 0.0011 short ton (WARM input unit)

1 BTU = 0.0011 MJ

Table 11 – Annual Energy and GHG Balances for Organic Waste Processing Methods for a City of One Million

Processing/Production Method/Practice (Transport Distance)	**Food Waste Million kg/year[a]**	**Energy Use (negative) Million BTU**	**Energy Use (negative) Million MJ**	**GHG Balance (Negative) $MTCO_2$-e**
Landfilling (32 km)	195	72,127	79	152,919
Composting (32 km)	195	125,268	138	(32,734)
Urban Composting (8 km)	195	119,123	131	(33,186)
Incineration With Energy Recovery (32 km)	195	(445,184)	(490)	(25,861)
Anaerobic Digestion	195	(837,508)[b]	(921)[b]	(48,750)[c]
WWTP (anaerobic digestion)	195	(837,568)[b]	(921)[b]	(48,750)[c]
Processing/Production Method/Practice (Transport Distance)	**Yard Waste Million kg/year[a]**	**Energy Use (negative) Million Btu**	**Energy Use (negative) Million MJ**	**GHG Balance (Negative) $MTCO_2$-e**
Composting (32 km)	216	138,758	152	(36,259)
Urban Composting (8 km)	216	131,074	144	(35,423)

Sources: EPA WARM model unless indicated
a – present study
b – EPA CoEAT
c - Morris et al., 2013

Food production and transportation also consume energy and generate GHG emissions. Studies suggest that organic systems, especially non-mechanized small-scale production, can be far more energy efficient than conventional production. One study that compared energy ratios (energy outputs divided by inputs) of conventional and organic systems in the UK found that organic systems (4.09:1 ratio) are almost 30 times more energy efficient than conventional systems (0.14:1 ratio) (Pearson, 2007). Gomiero et al. (2008) examine a host of studies and conclude that organic systems are

15-70% more energy efficient per unit land area than conventional production across a range of crop and dairy production.

Overall energy use for agricultural production in the U.S. (2002) is roughly 1.7 quadrillion BTU, or 1.9 trillion MJ, which is less than 2% of overall energy use in the U.S. (Schnepf, 2004). Fruit and vegetable production accounts for roughly 16% of total agricultural energy use, or 300 billion MJ annually (Schnepf). With a U.S. population of 288 million (2002), this is equivalent to 944,000 BTU (1041 MJ) annually on a per capita basis, or roughly 1 billion MJ for the city of one million.

Hendrickson (1994) cites a number of studies on energy and food and estimates that overall energy use in the food system is 15.6% of total U.S. energy use. Of that proportion, food production utilizes 17.5% and transportation accounts for 11%. Food preparation at home and restaurants (40.8%) and packaging (28%) account for the largest shares of energy use, while meat production is much more energy intensive than production of fruits and vegetables.

Assuming 16% energy use share as noted in Schnepf (2004) for fruit and vegetable production:

$$0.16_{\text{f \& v}} \times 0.175_{\text{production share}} \times 0.156_{\text{food system share}} = 0.004$$

By this measure, fruit and vegetable production accounts for just 0.4% of energy use in the U.S.

Transportation of fruits and vegetables:

$$0.16_{\text{f \& v}} \times 0.11_{\text{transportation share}} \times 0.156_{\text{overall food system share}} = 0.003$$

Transportation of fruits and vegetables accounts for roughly 0.3% of U.S. energy use.

In 2014, estimated GHG emissions for the U.S. agricultural sector were 574.1 $MTCO_2e$ (EPA, 2016). Agricultural emissions are primarily N_2O and CH_4 (methane), while urban emissions contain more CO_2, but these can be standardized using the CO_2 equivalents. On a per capita basis, and with a U.S. 2014 population of 318.9 million, agricultural emissions are equivalent to 1.8 $MTCO_2e$ for the city of one million. Weber & Matthews (2008) analyze energy and GHG emissions for both production and transportation of food, and find that the contribution of fruit and vegetable production and related transportation is approximately 10% of the total emissions from overall food production and transportation. The contribution of urban production of fruits and vegetables to GHG emissions for the city would therefore amount to roughly 0.18 $MTCO_2e$ (10% of 1.8 $MTCO_2e$) annually.

Results and Discussion

From Table 11, it is clear that organic waste processing methods vary over a wide range in their energy and emissions performance. The annual energy difference between the most energy intensive process (composting) and the lowest (anaerobic digestion and WWTP – which have net negative energy use) is nearly one billion BTU (over 1,000 MJ) annually for the city of one million Annual levels of domestic (residential) and overall energy use provide context. According to the U.S. Energy Information Administration, annual per capita residential energy use in 2010 (heating fuel and electric) was 71 million BTU, or 78,000 MJ. A city of one million uses 71 trillion

BTU (78 billion MJ) for household heating and electricity.

http://www.eia.gov/todayinenergy/detail.cfm?id=3590.

Potential proportion of domestic energy use offset by optimal organic waste processing:

$$\frac{\text{1 billion MJ}}{\text{78 billion MJ}} = .013\text{, or the energy use of 13,000 residents.}$$

In the context of total energy consumption, the potential offset is even lower. Total annual per capita energy use (2011) was 313 million BTU; one million people use 313 trillion Btu (297 billion MJ). http://www.eia.gov/tools/faqs/faq.cfm?id=85&t=1.

Potential proportion of overall energy use offset by optimal organic waste processing:

$$\frac{\text{1 billion MJ}}{\text{313 billion MJ}} = .003\text{, or the energy use of 3,000 residents.}$$

Under the assumption that energy use for urban intensive food production could be significantly lower than for conventional production, as some studies of organic systems suggest, the maximum reduction (to zero energy use) would be close to one billion BTU (1.1 million MJ), which is the estimate for conventional production of fruits and vegetables. This would approximately double the potential offsets noted above for organic waste processing.

The outcomes for GHG emissions are similar to those for energy. In the U.S., per capita GHG emissions averaged 21 $MTCO_2$-e annually (2011-2015), or 21 million $MTCO^2$e for the city of one million.

http://data.worldbank.org/indicator/EN.ATM.CO2E.PC/countries/1W?display=default.

The difference between the worst-performing food waste option from an emissions standpoint (landfilling) and the best (AD and WWTP – negative emissions) is approximately 200,000 $MTCO_2$-e.

Potential proportion of GHG emissions offset by optimal organic waste processing:

$$\frac{200{,}000 \text{ MTCO2e}}{21 \text{ million MTCO2e}} = .01\text{, or 1\% of the city's annual emissions}$$

This equals the annual emissions of 10,000 residents. Emissions from fruit and vegetable production are negligible. Similarly, as Table 11 indicates, emissions reductions from the modeling of shorter transportation distances for organics processing (5 mi. v. 20 mi. for urban and peri-urban composting, respectively) are also negligible.

Research from the UK confirms the relatively low importance of food transportation emissions compared to food production GHG emissions; the latter are 7.5% of all emissions (Garnett, 2008), similar to the U.S. figure of 8.4%. Foster et al. (2006) estimate that GHG emissions due to fruit and vegetable transportation contribute less than 1% of overall emissions in the UK on a life cycle basis. According to Smith, et al. (2005) - food transport (all types) accounts for just 1.8% of annual UK GHG emissions, and Garnett's estimates range from 1.5% to 2.3%. Both studies argue that GHG emissions are highly dependent on transport mode and efficiency, not just distance. For example, large truck transport from far-flung producers or centralized distribution nodes may produce less emissions than local transport via light truck or car, especially where the latter are not loaded to capacity (Smith et al.).

From the foregoing analysis, it is clear that organics waste processing methods, on the one hand, and urban food production, on the other, do not have a large effect on energy or GHG balances in the wider social-ecological context. Food waste processing and fruit

and vegetable production decisions would best be made according to other more compelling environmental, economic, or social contextual factors. Energy and emissions benefits are best sought in other areas, such as home energy use and certain transportation sectors.

Water Use and Wastewater Generation

Organic Waste Processing and Food Production

Water use is a critical aspect of both food production and organic waste processing. The agricultural sector accounts for the greatest share of consumptive water use globally. Consumption occurs through evapotranspiration and incorporation into the agricultural product. In the U.S., agricultural irrigation accounts for 31% of total freshwater withdrawals (2005) and agricultural irrigation for 80%-90% of consumptive water use (Schaible & Aillery, 2012). Power plant cooling accounted for approximately 45% of freshwater withdrawals in 2010 (Maupin et al., 2014), but the majority of cooling water (98%) is returned to its source and is therefore not considered consumptive use (Schaible & Aillery).

Despite the high yields of small-scale intensive production, several factors mitigate water use. In addition, there is evidence that organic systems may perform better in droughts compared to conventional production (Rodale, 2014). Because intensive small scale crop production as modeled here is essentially organic, greater drought resistance is likely. Organics waste processing also utilizes water and generate wastewater, and there are large differences among water use and wastewater generation among the

various processes considered here. Similar to the energy and emissions analysis, this framework component quantifies the water use of alternative food production systems in the wider social-ecological context.

Methods and Sources

- Schaible & Aillery (2012) and Maupin et al. (2014) provide data on irrigation and overall water use in the U.S. Johnson & Cody (2015) provides crop-specific irrigation data for California.
- Rodale (2014) provides information on large-scale organic systems in terms of drought performance. Gomiero et al. (2008) and Pimentel et al. (2005) provide data on large scale organic production.
- Morra et al. (2010) provide average irrigation levels over a four year intensive production (hoop house) trial for three crops. Jeavons (2012) provides water use data for "biointensive" production. Jeavons also alludes to the theoretical underpinnings of water consumption under close plant spacing.
- Data from Morris et al. (2013), and BioFerm (2009) allow a quantification of water use and wastewater generation for several organics waste processing methods. Diggelman (1998) and Thomas (2011) provide data on water use and wastewater generation at WWTPs

Irrigation - Water for Food

The food-water nexus is one of the most critical aspects of social ecological systems. Irrigation enables a significant proportion of fruit and vegetable production in the U.S.,

and much of it from water-scarce Western states. Irrigation is used on 70% of conventional vegetable acreage and 80% of orchard acreage. Western states accounted for 83% of water use for irrigation in 2010 (Maupin et al., 2014). As with yields, data on water use for intensive hoop house production utilizing organic soil amendments is scarce. In addition, water use is dependent on crop type, soils, climate, and weather during the growing season, and is therefore highly context-dependent in practice. However, several sources provide evidence that allows rough estimates of water requirements and comparisons among systems. This section gives estimates of absolute and relative water use for a city of one million, as well as contextual comparisons with water use in wider social-ecosystem contexts.

Conversion Factors

1 acre-foot = 1233 m^3

2.47 acres = 1 hectare

1 gallon = 0.00378 m^3

In 2010, the average rate of water application for irrigation in the U.S. was 2.1 acre-feet (Maupin et al., 2014). This is equivalent to the application rates that Schaible and Aillery (2012) estimated for agricultural production in the Western U.S. for 2008. This rate is within the irrigation ranges for California, one of the nation's top producers of fruits and vegetables. There, vegetables require 1.7-2.8 acre-feet, depending on variety, while irrigation rates for fruits range from 0.6 acre-feet for berries to 2.7 acre-feet for orchards (Johnson & Cody, 2015). This likely overestimates irrigation requirements for Northern and Eastern U.S. growing regions, with shorter growing seasons and lower evapotranspiration. Thus, it represents an upper bound for irrigation needs for cities in

those regions. Consequently, assuming that urban irrigation needs are equivalent to needs in Western states likely understates the potential water savings in irrigated urban production compared to Western states production.

The average of 2.1 acre-feet for conventional production is equivalent to 6,396 m^3/ha.

$$2.1 \text{ acre feet X } \left(1233 \frac{m^3}{\text{acre foot}}\right) \text{X } 2.47 \frac{\text{acres}}{\text{ha}} = 6{,}396 \text{ m}^3/\text{ha}$$

As noted above, an average of 75% of conventional fruit and vegetable production is under irrigation, and conventional production for the city of one million would require an estimated 4,000 ha.

$$0.75 \text{ X } 4{,}000 \text{ ha X } 6396 \frac{m^3}{\text{ha}} = 19{,}188{,}000 \text{ m}^3$$

Irrigation required to meet the city’s annual fruit and vegetable consumption under conventional production is estimated at 19.2 million m^3 annually.

Morra et al. (2010) provide irrigation averages over a four year period for intensive production of annual rotations of tomatoes, snap beans and lettuce. The total annual irrigation level was 4250 m^3/ha. For tomatoes, irrigation was 1100 m^3/ha for production of 111,000 kg/ha, compared to at 36,907 kg/ha for conventional yields. The comparative water use per unit of production between conventional and intensive systems is calculated as follows:

Tomatoes - conventional production: $\frac{6{,}396 \text{ m}^3}{36{,}907 \text{ kg}} = 0.17 \text{ m}^3/\text{kg}$

Tomatoes - intensive production: $\frac{1{,}100 \text{ m}^3}{111{,}000 \text{ kg}} = 0.01 \text{ m}^3/\text{kg}$

$$\frac{.01}{0.17} = .06$$

For tomatoes in the Morra et al. trial, intensive hoop house production used just 6% of

the water required for conventional field production per unit.

Snap Beans - conventional production: $\frac{6{,}396\ m^3}{6{,}163\ kg} = 1.0\ m^3/kg$

Snap Beans - intensive production: $\frac{1{,}650\ m^3}{7{,}100\ kg} = 0.23\ m^3/kg$

$$\frac{.23}{1.0} = .23$$

For snap beans in the Morra et al. trial, intensive hoop house production used 23% of the water required for conventional field production per unit.

Lettuce (average of leaf and head varieties) - conventional production:

$$\frac{6{,}396\ m^3}{40{,}629\ kg} = 0.16\ m^3/kg$$

Lettuce - intensive production: $\frac{1{,}500\ m^3}{22{,}450\ kg} = 0.07\ m^3/kg$

$$\frac{.07}{0.16} = .44$$

For lettuce in the Morra et al. trial, intensive hoop house production used 44% of the water required for conventional field production per unit.

Annual per capita consumption of tomatoes in the U.S. is 3.8 kg, while consumption of snap beans and lettuce is 0.3 kg and 4.4 kg, respectively. Water use per unit production on a weighted average annual per capita consumption basis is calculated as follows:

$$\frac{(3.8\text{ kg tomato X .06})+(0.3\text{ kg snap bean X 0.23})+(4.4\text{ kg lettuce X 0.44})}{(3.8\text{ kg}+0.3\text{ kg}+4.4\text{ kg})} = 0.26$$

For the range of representative vegetables in the Morra et al. study, overall water use

under intensive production is 26% compared to conventional production, for a water savings of 74%.

Converting to annual requirements in m^3:

$$0.26 \text{ X } 19.2 \text{ m}^3 = 5 \text{ million m}^3$$

Under this scenario, annual intensive produce production for the city of one million would require 5 million m^3.

The intensive production methods advocated by Ecology Action require 12.5% (high-biointensive yields) to 25% (low-biointensive yields) of the water used in conventional production (Jeavons, 2012). Increasing soil organic matter from 0.5% to 2% reduces plant transpiration by up to 75%, and closely-spaced plantings in intensive systems reduce soil evaporation by up to 63% (Jeavons). The enclosed production environments of greenhouses and hoop houses will also reduce evaporative losses under many conditions.

Converting to annual requirements in m^3:

$$0.125_{\text{high biointensive}} \text{ X } 19.2 \text{ m}^3 = 2.4 \text{ million m}^3$$

$$0.25_{\text{low biointensive}} \text{ X } 19.2 \text{ m}^3 = 4.8 \text{ million m}^3$$

Under these assumptions, annual intensive production for the city of one million would require 2.4 – 4.8 million m^3.

Water requirements for large-scale organic production are likely to be somewhere between intensive and conventional production, and are not calculated here. Gomiero et al. (2011) note that the water holding potential of organic soils can be 100% greater

than soils under conventional production, but this does not readily translate into water use estimates. As noted above, organic and biointensive systems may have significant advantages during droughts. Pimentel et al. (2005) cite long-term studies of organic cropping systems, in which corn yields were 28 – 34% higher than conventional yields in dry years.

Table 12 provides estimates of water requirements under a range of assumptions for alternative production methods to meet the targets for the city's annual fruit and vegetable consumption.

Table 12 - Water Requirements for Alternative Vegetable Production Modes for the City of One Million

Source	**Production Mode**	**Water Requirement m^3**
Present Study	Conventional Rural	19.2 million
Morra, et al., 2010 (calculated above)	Intensive Hoop House	5 million
Jeavons, 2012 (calculated above)	High and Low Biointensive	2.4 - 4.8 million

Results and Discussion

The information in Table 12 indicates that intensive urban production, as defined in the present study, is likely to require less water per unit of production than conventional fruit and vegetable production. The difference in water use between intensive and conventional production can be compared to annual domestic (residential) use, as well as overall water use in the wider context. The annual estimated water savings of 14.2 – 16.8 million m^3 under biointensive production is equivalent to 3.7 – 4.4 billion gallons

annually for a city of 1 million, or 3,700 - 4,400 gallons/capita. For comparison, current USGS estimates of residential water use/capita range from 29,000 – 37,000 gallons (110-140 m^3) annually http://water.usgs.gov/edu/qa-home-percapita.html. Urban fruit and vegetable production has the potential to save the equivalent of 10-15% of annual domestic water use. However, domestic supply accounts for less than 10% of water withdrawals in the U.S. Overall annual per capita water withdrawals for all uses in 2010 were 1,568 m^3 (Maupin et al., 2014), so annual water savings from urban production would amount to about 1%.

Where precipitation is at least 20” during the growing season, crops are unlikely to need irrigation (Maupin et al., 2014). In this case, rainfall can supply all of the water needed for urban intensive production of fruits and vegetables. This would result in further consumptive water savings. Intensive (and organic) systems may perform significantly better than conventional production without needing irrigation during periods of low precipitation due to greater levels of soil organic matter (Brown et al., 2011). In addition, urban production may be coupled with rain catchment devices, which store water to supply crops needs during periods of inadequate precipitation. Finally, where urban cropland is designed to function as green infrastructure and receives water from a larger surrounding area, growing season precipitation requirements may be as low as 4” (see GI discussion).

Virtual Water

“Virtual water” is the water embedded in the production of agricultural products (some

have extended the concept to cover the production of other products as well). Introduced by Tony Allan in the early 1990s, the term provides a way to conceptualize the water used to produce food that is grown in one area and exported to another. Virtual water is not the actual water contained in the exported food, but rather, the precipitation or irrigation water needed to produce the food. Since urban agricultural production offsets production in other areas, either nearby or far-flung, it can have virtual water effects. For example, in urban areas where precipitation is plentiful, virtual rainwater may replace virtual irrigation water from a production area where irrigation is required to grow fruits and vegetables. This is, in fact, what occurs when urban produce grown in a Northern or Eastern city replaces fruit and vegetable imports from an arid Western state. This is likely the most common scenario, and highlights the potential for local production to ease water scarcity in other regions of the country or world.

Allan (2003) originally conceived the term to convey how food imports from water rich areas could address water scarcity in areas where food production requires inordinate water resources. In terms of water use, the former have a comparative advantage in food production. As noted, in the urban production scenario modeled here, green water may suffice to produce target levels of fruits and vegetables; if production required extensive use of potable municipal supply, energy (for treatment) considerations would come into play, and the comparative advantage may be reduced or eliminated. Comparative advantage also involves factors such as temperature (growing degree days) and opportunity costs for urban land. As the present analysis suggests, there are natural climatic limits to fruit and vegetable production in the modeled region; other

regions will be able to produce a greater or lesser proportion depending on their climates. Further context-specific analysis would be needed to determine the types and magnitudes of the tradeoffs necessary to produce more food locally. Given the range of comparative advantage considerations, most urban areas are likely better off importing a large proportion of their food; local production is not automatically preferable from a policy standpoint. These considerations are national and even global in nature. Agricultural policy could encourage appropriate levels of local, urban production to reduce the negative effects of virtual water transfers from arid to water rich regions. Similarly, the water savings potential of intensive urban production compared to more conventional forms of rural production could also reduce water stress within arid regions.

Water for Organic Wastes

Conversion Factor

1 liter = .001 m^3

Organic waste processing methods vary greatly in their water use and wastewater generation. In the context of the framework, processing methods that use less water and generate less wastewater perform better environmentally, all else being equal. This component addresses water use and wastewater generation for the processing methods considered in the scenario analysis, as well as alternatives.

According to the 2013 meta-review by Morris et al., WWTPs processing of food waste (via in-sink disposers) results in both the greatest water use and wastewater volumes.

Each kg of food waste requires approximately 12 liters of (potable) water for grinding, conveyance, and treatment at a centralized WWTP (Diggelman, 1998; Thomas, 2011). By definition, this is also the quantity of additional wastewater generated by the sink disposal-to-WWTP processing method. Separated solid waste collection with centralized composting utilizes 0.31 l/kg (Diggelman, 1998). Lundie & Peters (2005) estimates are in similar ranges (12.4 l/kg for disposers, 0.1 l/kg for compost), using similar LCA system boundaries. The present study assumes this water is sourced from a potable supply in the urban context. Centralized aerobic composting (AC) may use water to keep piles moist to create optimal conditions for aerobic digestion. The assumption in the present study is that aerobic composting does not generate wastewater under appropriate siting and management practices. In practice, leaching may occur if the process is not sited and managed appropriately, but this is outside the scope of the present study.

There is a range of water requirements for AD processing. Some systems require no additional processing water beyond the 70% moisture found in food waste, on average (BioFerm, 2009). “Wet” systems generally require 85% moisture, and so require addition of an equivalent amount of water (mass basis) for processing. For this study, AD systems are assumed to generate no wastewater because the moisture remaining in the digestate is used in subsequent stages (land application or aerobic composting). Landfilling and incineration result in water use similar to composting (Diggelman, 1998). Leachate (wastewater) is a problem for many landfills. Treatment of leachate, as required in the U.S., is outside the system boundaries considered here.

To calculate water use estimates:

$$\text{Organic Waste Input}\frac{\text{kg}}{\text{year}} \text{ X Process Water}_n \text{ Requirement}\frac{\text{m}^3}{\text{kg}} =$$

$$\text{Annual Process Water Requirement}$$

The calculation is identical for wastewater generation estimates, substituting wastewater generation rates for process water requirements. In contrast to the calculations for soil amendments, the inputs here include all of the city's annual yard and food wastes, in order to provide the widest range of potential estimates.

Table 13- Water and Wastewater for Organics Waste Processing for a City of One Million

Organic Waste Process (Source)	**Process Water m^3/kg**	**Wastewater Generation m^3/kg**	**Waste Input Kg/year[c]**	**Annual Water Use m^3**	**Annual Wastewater Generation m^3**
WWTP (Food Waste)	0.012	0.012	195 million	2.3 million	2.3 million
Compost (Food Waste)	0.00031	0[a]	195 million	60,450	0
Compost (Yard Trimmings)	0.00031	0[a]	216 million	66,960	0
Anaerobic Digestion (Food Waste)	0 – 0.001	0[a]	195 million	0-195,000	0
Landfilling (Food Waste)	0.00037	0[b]	195 million	72,000	0
Incineration (Food Waste)	0.00034	0	195 million	66,000	0

Source: Diggelman (1998) - includes water use for capital equipment manufacture
a assumes no water use under appropriate site and process management
b leaching may occur, but is outside the system boundaries of the present study
c amounts are total annual food waste and yard waste available in the city

Results and Discussion

Table 13 lists estimates of water use and wastewater generation for various methods of

food and yard waste processing. Even accounting for potentially unrealistic levels of organic wastes recovery and input, water use for food waste processing and yard waste composting is not significant compared to urban agricultural requirements. Even the relatively high WWTP consumption amounts to less than 2% of the city's annual residential water use. However, water consumption may be a consideration for site-specific decentralized organics waste processing, since it is required for some processes and the only source may be the potable supply.

The Water – Energy Nexus

It is important to note that indirect energy use and greenhouse gas emissions related to water use will vary significantly across production systems. For example, urban food production is the only system that is likely to use potable water to any significant degree. Potable supply may have significant “embedded” energy and emissions compared to many agricultural supplies, assuming equivalent pumping and conveyance costs. For this reason, the use of potable supply for urban food production would tend to decrease any energy and emissions advantages of urban production. On the other hand, if conveyance distances for large scale production are large, urban production may have an energy and emissions advantage. By the same token, there is also significant embedded, or virtual, water in the fossil energy used in large scale food production, so under sufficiently broad system boundaries, this would have to be taken into account. Both embedded water and energy are outside the scope of the present study.

Green Infrastructure Potential of Urban Cropland

Water Infiltration, Retention, and Runoff

An important aspect of urban food production is its potential to mitigate, or exacerbate, urban stormwater and nutrient flows. Although vegetable and fruit production was common in U.S. cities until the mid-twentieth century (e.g., WWII-era Victory Gardens), urban food production in metropolitan areas is currently a unique land use, subject to a variety of zoning codes and ordinances. In the years since the original Victory Gardens, awareness of the significance of urban nutrient pollution has grown. At the same time, a proliferation of impervious surfaces, combined with the greater precipitation variability wrought by climate change, have brought urban stormwater issues to the forefront. A

number of studies have considered the possibility that urban cropland can serve as green infrastructure, but there is a dearth of information on its performance potential (Freshwater Society, 2013; Lovell & Taylor, 2013).

- Like its rural counterpart, urban agricultural land is part of the environment. It has the potential to provide a range of ecosystem services, such as stormwater capture, retention, and infiltration. Urban cropland can perform similar functions to rain gardens and bioswales if designed and managed with green infrastructure goals in mind. However, like rural agriculture, urban food production also has the potential to contribute to nutrient loading of surface waters and groundwater. For example, some of the practices common in urban agriculture, such as siting production on impervious surfaces, or otherwise disconnecting it from the natural hydrology in the area, may facilitate nutrient and stormwater runoff in severe storm events. This framework component quantifies some of the potential ecosystem benefits and drawbacks of urban food production from recycled organic wastes. Here, the GI potential of biointensive urban production is assessed in relation to the city's spatial footprint, based on urban cropland area and average urban land area for a city of one million.

Methods and Sources

- Selbig & Balster (2010) provide data on contributing to receiving area ratios for rain gardens, and Steinke et al. (2008) characterize plant type performance.
- Water retention of conventional and large scale organic production are calculated from research by Pimentel et al. (2005) and Gomiero et al. (2011).

- Data from the U.S. Census Bureau for average U.S. urban population density (2010) allows the calculation of urban land area for the city of one million.

Conversion Factors

$1 \text{ mile}^2 = 260 \text{ ha}$

$1 \text{ ha} = 10{,}000 \text{ m}^2$

1" = .0254 m

Using rain garden design principles, urban cropland can capture and infiltrate stormwater, which, in turn, can provide irrigation for urban crops. The lower bulk density and higher organic matter levels of urban cropland increase both infiltration rates and water holding capacity compared to low-fertility or compacted native urban soils. According to Selbig & Balster (2010), rain gardens in both sand and clay soils (Midwest - USDA Zone 5) infiltrated virtually all of the precipitation, which exceeded 40" annually in three of the five study years. The rain gardens, constructed at 6" (0.15m) below the surrounding surface, collected roof runoff from contributing areas that were five times greater in area, a ratio of 5:1. The study estimated that prairie-vegetated gardens in clay soils could store 2.91 inches of precipitation, equivalent to 739 m^3/ha.

$$2.91\text{inches} \times 0.0254\,\frac{\text{inches}}{\text{m}} \times 10{,}000\,\frac{\text{m2}}{\text{ha}} = 739\text{m}^3/\text{ha}$$

This corresponds with the 2005 Pimentel et al. (2005) estimate of 816 m^3/ha storage in the root zone for large scale organic crop production. This suggests that root zone storage for urban cropland may be similar to large-scale organic storage. By comparison, the corresponding water-holding estimate for conventional agricultural production is roughly 408 m^3/ha, or 50% of the capacity of organic and urban cropland

(Gomiero et al., 2011).

Given an average urban population density in the U.S. of 2,534 people per square mile (US Census 2010), the city of one million occupies 395 square miles, or 100,000 ha. (See footnote 3 above for land area considerations).

$$1{,}000{,}000 \text{ people}/\ 2{,}534\,\frac{\text{people}}{\text{mile}^2} = 394.6 \text{ mile}^2$$

$$395 \text{ mile}^2 \text{ X } 260\,\frac{\text{mile}^2}{\text{ha}} = 102{,}605 \text{ ha}$$

Urban fruit and vegetable production, as already noted, would cover up to 2,000 – 4,000 ha of urban land area. At a 5:1 contributing to receiving area ratio, 2,000 – 4,000 ha of cropland could mitigate the stormwater runoff from 12,000 – 24,000 ha of impervious surface (including cropland area), or 12 – 24% of the total urban land area at the population density considered here. This suggests significant potential in the ability of urban cropland to mitigate stormwater on a city scale. The appropriate range of contributing to receiving area will vary considerably according to local and site-specific conditions, such as precipitation, soil type, and depth to subsoil. For example, sandy soils will infiltrate water more rapidly than clay soils. Under some conditions, the ratio could exceed 5:1, so the potential land area mitigated could be even greater. Under conditions of high annual precipitation and low infiltration potential, an urban garden may only infiltrate its own area. In addition, crop plants may not achieve quite the same performance as deep-rooted native drought-resistant species. As a result, the stormwater mitigation area may just equal the urban garden land area, for a low-range estimate of 2,000 – 4,000 ha.

Berms are the most important performance factor in buffer strip and rain garden design, while vegetation type is not critical to overall infiltration performance (Steinke et al., 2008). Constructing berms at the perimeter of the garden help to create the topographical depression that detains water. Berms also reduce the possibility of water and nutrient runoff from heavy or prolonged precipitation events.

Table 14 - Potential Urban Cropland Stormwater Mitigation for the City of One Million

Land Use/Land Cover	**Cropland Area ha**	**Urban Land Area ha**	**Total Area Mitigated ha**	**Total Area Mitigated %[a]**
Urban Cropland	2,000 – 4,000	100,000	2,000 – 24,000+	2 – 24+

a - at average U.S. urban population density

Results and Discussion

The foregoing suggests that urban cropland can function as green infrastructure if it is designed with GI in mind and connected to the local hydrology. Urban food production could mitigate stormwater flows from 2 – 24% of the urban land area (including rooftops), and perhaps more under favorable conditions. This percentage will vary according to urban population density. The mitigation potential is likely similar to conventional rain gardens, and far exceeds the potential capacity of GI features such as rain barrels. However, water quality for food production is a critical issue. Contaminated runoff from streets and parking lots would generally not be suitable for agricultural production, so these would need to be mitigated by more traditional rain gardens and bioswales.

Growing food in greenhouses and hoop houses is the primary means of season extension for urban production. It is important to note that these structures can add to

the city's impervious surface area, negating GI functionality (assuming no other contributing land area), unless there is some means of water capture from the structure's roof area. Options include retractable roofs for greenhouses, and gutter systems that direct runoff onto the growing surface. In addition, engineered rainwater catchment and storage systems like cisterns and rain barrels can both extend GI functionality and provide water when it is needed by crops. Where urban cropland is receiving runoff from nearby land or structures, some means of conveyance will also be necessary. To function reliably as GI, plots must be constructed 6" (0.15 m) below the surrounding surface or within 0.15 m berms (Selbig & Balster, 2010; Steinke et al., 2008). For the same reason, urban cropland must be level.

The literature comparisons suggest that large-scale organic crop production could mitigate stormwater flows as well as urban cropland from a water holding capacity, and the greater land area required (4,000 - 5,500 ha, compared to 2,000 ha urban intensive) would increase the overall potential substantially. Conventional cropland would roughly equal the urban cropland potential; half the water-holding capacity and double the land area (4,000 ha) of urban intensive (2,000 ha). However, it is unlikely that the large-scale options would be strategically located to mitigate stormwater flows entering the city. In addition, it would be difficult to engineer large-scale production to mitigate stormwater flows from a topographical standpoint; runoff from large-scale agricultural production is a well-documented issue.

Nutrient Runoff and Infiltration/Contaminant Mitigation

Urban compost and biosolids soil amendments have the potential to both mitigate and contribute to soil and groundwater contamination. There is ongoing research interest in quantifying this potential for soil amendments in the GI context of rain gardens, bioswales and the like. However, compared to conventional agricultural production, nutrient leaching is of far less concern in intensive production. There is evidence that recycled nutrients are much less prone to leaching than conventional fertilizers. Leytem & Bjorneberg (2009), found that compost amendments to soils resulted in significantly lower total P and soluble P runoff concentrations compared to soils amended with manure or conventional fertilizers. According to Carpenter et al. (1998), infiltration and leaching are significant means of moving N and P to hydrological systems; in conventional agriculture, from 10-40% of applied N is exported in this manner in loamy or clay soils, while the figure for sandy soils is 25-80%.

While nutrient laden runoff is a significant issue in conventional production, it is difficult to isolate the contributions of conventional vegetable cropland from overall agricultural production. P is generally the nutrient of greatest concern in freshwater lake contexts, because it is the limiting nutrient for eutrophication. In freshwater streams, N may be a limiting nutrient. Chinault & O'Connor (2008) compared the P leaching potential of fertilizers derived from biosolids to standard fertilizer (triple super phosphate or TSP). A key measure of P leaching potential (PWEP) for most of the biosolids was one to two orders of magnitude below the TSP value. This suggests that organic fertilizers produced through similar AD and aerobic processes will also display very low leaching

potentials compared to conventional food production using TSP. Some states, like WI, exempt biosolids fertilizers from their fertilizer P bans.

N leaching is largely governed by the C:N ratio in the soil. Recommended C:N ratios for finished composts are designed to balance long-term and short-term plant availability of N through a controlled mineralization process. In contrast, the inorganic (mineralized) N in synthetic fertilizers is both immediately available to plants and highly leachable. Organic systems generally leach less N to groundwater than conventional systems; studies cited in Gomiero et al (2011) indicate that N leaching might be 4.4 – 5.6 times lower in organic systems. If urban cropland is designed to function as GI, there should rarely be any water, soil or nutrient export via runoff, so that is another factor in favor of urban production. However, there is an important exception, covered below.

The P contained in composts is the potential range of annual compost production listed in table 2 multiplied by 0.3% P content.

71-77 million kg compost X 0.003 P=213,000-231,000 kg P

The P contained in biosolids is the estimate of annual biosolids production listed in table 3 multiplied by an average 2.2% P content from Brobst (2016). The relative P content of biosolids compared to effluent will vary greatly depending on WWTP P removal rates; average estimates are for order of magnitude comparisons.

23.2 million kg biosolids X 0.022 P=510,400 kg P

Table 15- Potential P Release from Soil Amendments and Annual WWTP Effluent Release

Source	**Kg/year**
P in Compost	213,000 – 231,000
P in Biosolids (assuming average WWTP P removal in 2006)	510,400
WWTP Effluent (87% - 56% removal range)[a]	90,000 - 305,000

a - adapted from Kort (2014)

Results and Discussion

As shown in Table 15, the P contained in biosolids and composts is generally greater than the annual P released in WWTP effluents. Since P loads from effluents are known environmental concerns, this suggests that the pollution potential of soil amendments is even more significant should the P they contain be released to surface waters.

Unfortunately, the same qualities that confer high performance to soils from a green infrastructure standpoint, such as high organic matter content and lower bulk density, also make urban intensive cropland highly water-erodible compared to turfgrass and (no-till) conventional production methods. Urban cropland soils that leach during storms or that are washed away in floods would release significant amounts of P to the environment. In addition, they would release organic matter and N, which can also degrade aquatic ecosystems. Increasing use of GI had led to research interest in the leaching potential from soil amendments used in bioswales and rain gardens. For

example, Mullane et al. (2015) found significant nutrient leaching potential from bioswales, while Mendry (2013) cites a number of codes that limit compost content in bioswales to reduce leaching.

According to Pfeiffer et al. (2013), it is common for cities to require raised beds, or clay or geo-textile barriers to prevent uptake or migration of existing soil contaminants. In addition, some urban farmers plant in soil placed on impervious surfaces such as asphalt, for the same reasons. These and similar practices disconnect urban agricultural production from the surrounding hydrology, and may reduce their GI potential. Disconnected urban agricultural sites are capable of capturing and filtering stormwater, contaminants, and nutrients up to a point. However, when their water holding capacity is exceeded, they may act as a source of both runoff and nutrients to the ecosystem rather than a sink.

Policies to prevent urban soils from eroding and releasing P (and N, C, and organic matter) are important to protect urban waters. Judging from the common siting standards and practices surveyed in Pfeiffer et al. (2013), this concept needs more policy attention if urban food production is to become a permanent land use. From a pollution perspective, the benefits of permanent and appropriate siting may advance the case for urban agriculture and address current land tenure barriers for urban farmers. This is a rich area for further research and policy discussion.

Contaminant Mitigation

Over time, the organic matter and biological activity in composts and biosolids can also help to break down a range of contaminants in urban soils and reduce the bioavailability of certain metals. For example, Charlesworth et al. (2012) found that mixed compost can degrade 65% - 80% of motor oil added to soils. Defoe et al. (2014) tested biosolids and compost amended soils and documented lead and arsenic bioavailability reductions of 10-50% and 12-25%, respectively. Beyond this brief mention, the potential for urban agriculture to mitigate soil contaminants is outside the scope of the present study.

Framework Summary and Discussion

The framework developed here serves to provide both a structure and some first level estimates and value ranges for the environmental assessment of urban nutrient processing for multi-functional urban food production. Results suggest that a significant portion of the city's vegetable, and to a lesser extent, fruit consumption, can be provided by local resources that might otherwise end up as urban environmental (primarily water) pollution. Results show that a system of urban nutrient cycling and food production at the scale modeled here can have benefits in terms of green infrastructure, with the potential to mitigate stormwater for a significant percentage of the urban land area. Stormwater mitigation from urban cropland can be significant. Compared to conventional food production, urban intensive agriculture with recycled nutrients can also result in water savings, in the range of 10 - 17% of the city's domestic use, which may be significant in water-scarce areas. Land area requirements are also less than conventional production and large scale organic production, by 50% and up to 64%, respectively. Recycled nutrients also perform much better in terms of runoff potential

compared to conventional fertilizers, with the caveat that if urban soils leach or are washed away, this advantage largely disappears. In the case of P, the runoff advantage can be one or two orders of magnitude better due to the relatively low water-soluble fraction. Nutrient recycling also conserves P, a valuable and finite resource that is critical for food production.

While energy use and GHG emissions are also lower for the urban system compared to conventional production, these results are perhaps less compelling in the wider social-ecological context. Both energy and emissions advantages of the complete organic waste and food production cycle are in the range of 1-2% of domestic and overall energy and emissions impacts. While there is keen public interest in energy and emissions for food production, e.g., the “food-miles” concept and producing energy from food waste processing, this study finds that these have fairly small effects in the urban social-ecological context.

The framework results provide first-level estimates of many of the relevant values and indicators for urban nutrient-food systems, but the present study is not designed to provide precise values. Many of the values are highly context dependent, and will vary accordingly. The framework is a roadmap that researchers can employ to study aspects of the urban nutrient-food-environment nexus in more detail, with more precision, or adapted to specific contexts. It is also a tool for public officials and managers, offering both the structure and metrics to inform policy decisions and suggest policy directions.

Scenario Analysis – Employing the Framework

The framework can be used to model a range of assumptions about organic waste processing and food production, including sensitivity analysis to determine which factors are most important. Here, three realistic scenarios for urban waste processing are analyzed using the framework and estimated values from the present study, along with the external models used in the framework. For example, water use and wastewater generation are estimated using values from Diggelman (1998), and energy and GHG emissions estimates are calculated from WARM, CoEAT and Morris et al. (2013).

Modeling evaluates and compares three scenarios of processing methods for the food and yard waste for an urban population of one million. The first is the current practice, or baseline scenario (EPA, 2015 – data from 2013), in which specific portions of organic wastes are either landfilled (53%), incinerated (13%), processed via wastewater treatment (29%), or composted (5%). It is important to note that these averages mask some distinctions. In real cities, either landfilling or incineration will be an option for organic waste processing, but typically not both. This is a limitation of the hypothetical scenario analysis presented here, in that the baseline will not reflect any city's actual situation. However, this approach does support the overall generalizability of the results for the "city type" considered here.

The second scenario is sewer-centric, in which a greater portion of food wastes are diverted to the WWTP via the sewer system (60% in total), instead of landfilled or

combusted. The second scenario models an initiative currently advocated at the federal level by EPA (CoEAT). This co-digestion policy option is also being advanced in some municipalities (e.g., Milwaukee, Philadelphia), because it uses existing wastewater treatment infrastructure and excess capacity to generate energy and fuel in addition to soil amendments. In this scenario, the biosolids resulting from the additional food waste to WWTP are dedicated entirely to land application for urban food production.

The third scenario focuses on source-separated organics (SSO). It captures a greater portion of food waste for direct anaerobic/aerobic processing, with yard waste in the aerobic phase. This SSO (source separated organics) scenario includes the anaerobic digestion of a high proportion (60%) of food waste. Anaerobic digestion allows processing of high strength food wastes under controlled conditions, including the capture and treatment of potentially noxious odors. It is potentially more suitable for decentralized food waste processing in proximity to residential and public land uses.

SSO is the (hypothesized) optimum scenario utilizing currently available technologies and realistic organic waste capture rates. Capturing 60% of food waste is at the high end of demonstrated diversion in the U.S. (City of Seattle, 2014). SSO combined with anaerobic digestion shows the most favorable net energy and GHG emissions balances (Morris et al., 2013). Metals and other forms of contamination should also be lowest under SSO, since food waste is not co-mingled with other wastes, as it is in the Sewer scenario. Results would likely be very similar for direct centralized aerobic composting of mixed food and yard wastes (no AD stage), with somewhat worse energy and GHG

emissions for the composting only option.

Assumptions for all scenarios:

- incineration ash is landfilled – no P or soil amendment recovery
- WWTP P recovery efficiency is 87%
- 50% of yard wastes are composted (100% of leaves and shrub/tree trimmings)
  - model input is 108 million kg wet weight
  - grass clippings are not included in the scenario inputs, as they are excluded from the framework (see sidebar in Nutrient Quantity section)

Table 16 - Three Organics Processing Scenarios

Organics Processing Scenario	**Organics Processing Method(s)**	**Rationale**
Baseline	60% households w/kitchen disposals x ≈ 50% capture rate for food wastes = 29-30% domestic food waste processed at WWTPs, (Brown et al. 2009) 56% landfill, 13% incineration U.S. averages (2013) for MSW processing (EPA, 2015).	Current (2013) practices.
Sewer Food waste processed through WWTP	60% of residential and retail food waste conveyed, via in-sink disposal units to WWTP and co-digested with wastewater.	Policy option being advanced at the federal level (EPA CoEAT) and in some municipalities.
SSO (Source separated organics) for food waste	60% of residential and retail food waste captured and processed via centralized anaerobic digestion, with yard waste for aerobic composting	Hypothesized optimum scenario that is plausibly achievable in the near term.

Input quantities for a city of one million are based on the estimates given in the text of

the present study. Results are calculated from the values for each framework component, as listed in the corresponding section of the text. Annual waste inputs for the baseline are calculated from current shares for each processing method, based on national averages and scaled for a population of one million. Modified inputs for the SSO and sewer scenarios are taken proportionately from the landfill and combustion inputs of the baseline.

Figure 5 - Food Waste Processing Scenarios

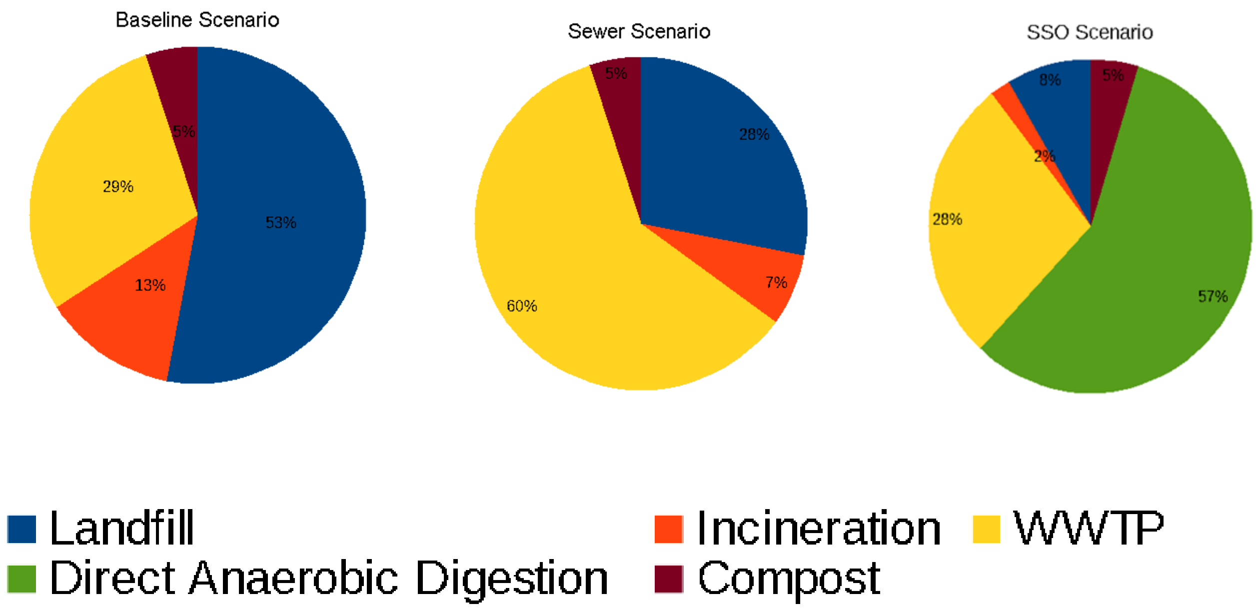


Table 17 – Scenario Organic Waste Processing Inputs by Percentage

Scenario	Process Inputs				
	Landfill	**Combustion – Incineration w/energy recovery**	**WWTP**	**SSO Anaerobic Digestion (non-WWTP)**	**Compost**
Baseline - Food Waste	53%	13%	29%	0	5%
Sewer - Food Waste	28%	7%	60%	0	5%
SSO - Food Waste	9%	2%	29%	60%	60% (post-AD)

Table 18 - Scenario Organic Food Waste Inputs – Wet Weight Basis

Scenario	Annual Process Inputs - million kg wet weight				
	Landfill	**Combustion – Incineration w/energy recovery**	**WWTP**	**SSO Anaerobic Digestion (non-WWTP)**	**Compost**
Baseline	103.0	25.2	56.6	0*	9.8
Sewer	54.8	13.5	117	0	9.8
SSO	17.2	4.2	56.6	117	117 (post-AD)

Scenario Results and Discussion

Scenario results are presented in Table 19. In terms of total input mass recovered, Baseline performs the worst, sequestering 129 million kg (wet weight basis) of the original 303 million kg inputs. Recovery in the Sewer scenario is 235 million kg, while

SSO delivers the best performance, with 282 million kg recovery. P recovery and loss show similar patterns, and all three scenarios result in net energy gains. Net GHG emissions are negative (sequestration) for Sewer and SSO, with some emissions in Baseline. Water use and wastewater generation are similar for Baseline and SSO, and approximately double for Sewer.

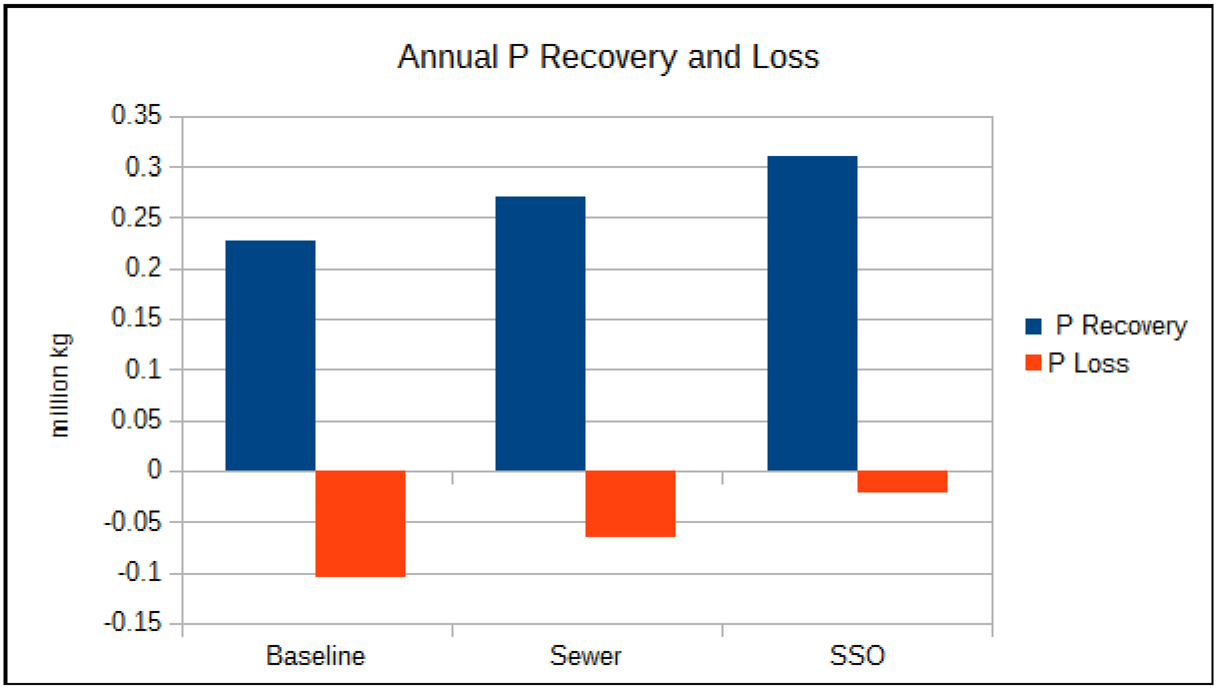
Table 19 - Scenario Results

Scenario	**Resource Recovery (wet wt.) million kg[a]**	**P Recovery million kg**	**P Loss million kg**	**Energy million MJ (negative)**	**Emissions $MTCO_2$-e (negative)**	**Water Use million m^3**	**Wastewater Generation million m^3**
Baseline	174	0.227	0.105	(181)	0.04	0.79	0.68
Sewer	235	0.270	0.065	(418)	(0.01)	1.50	1.40
SSO	282	0.310	0.022	(586)	(0.07)	0.82	0.68

a - original inputs = 303 million kg wet wt. (195 million kg food waste and 108 million kg yard waste)

Figure 6 shows P recovery and loss for each scenario. The SSO scenario performs best in terms of both phosphorus recovered for beneficial use and the lowest amount of phosphorus lost to the environment. The least phosphorus is recovered under the baseline scenario, with the largest portion sequestered directly in landfills and via landfilled incineration ash.

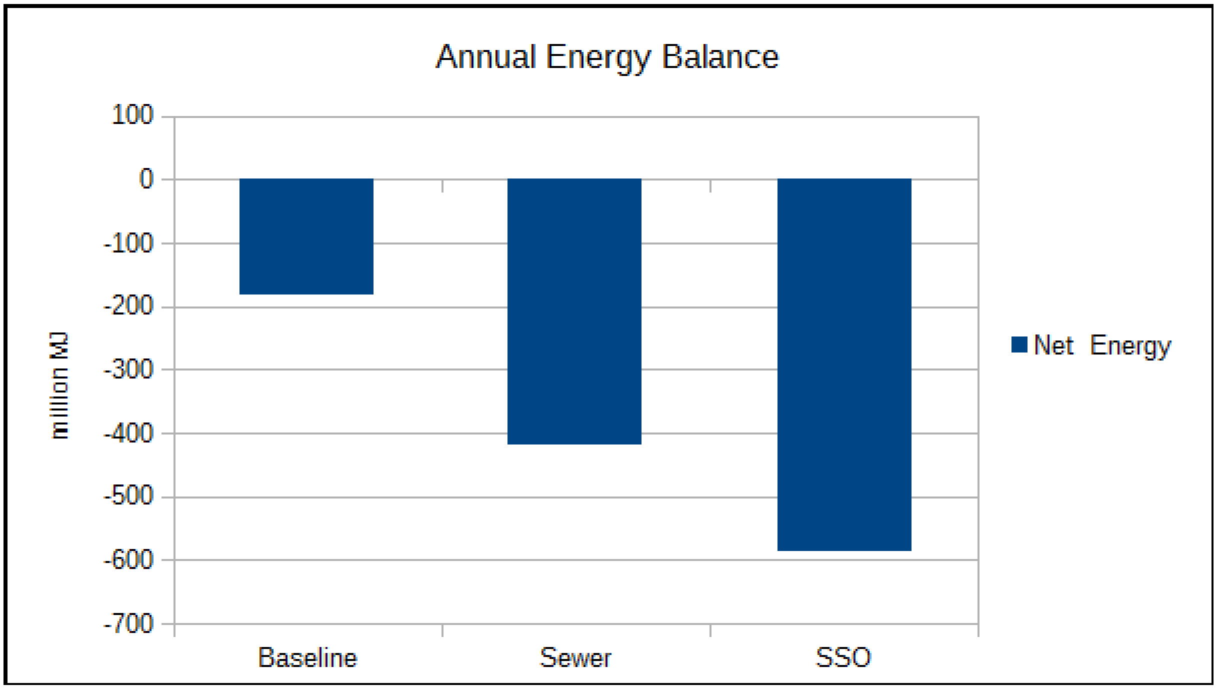
Figure 6 - Phosphorus Recovery and Loss to the Environment



Baseline losses of approximately 100,000 kg are comparable to annual environmental discharges in WWTP effluent, assuming 87% WWTP P capture (from all influent sources). The sewer scenario increases phosphorus recovery compared to the baseline, due to less terrestrial sequestration. However, a higher proportion (≈50%) of the phosphorus lost to the environment in the sewer scenario ends up in WWTP effluent, where it presents more eutrophication potential compared to landfilling. The sewer scenario, because of the increased food waste processed via WWTP, increases annual P discharges to waterways by approximately 65,000 kg, or 70% greater compared to the baseline. Advanced tertiary treatment at WWTPs to remove a greater proportion of P may narrow this gap. The SSO scenario reduces terrestrial P sequestration in landfill and incineration, but does not affect WWTP discharge of P. The result is an annual additional recovery of 83,000 kg of P.

Figure 7 shows net energy balance for the scenarios. Perhaps surprisingly, all three scenarios result in a net energy gain. This is because food waste is high in energy potential and all three scenarios have mechanisms to extract at least some of that energy.

Figure 7 - Net Energy Balance



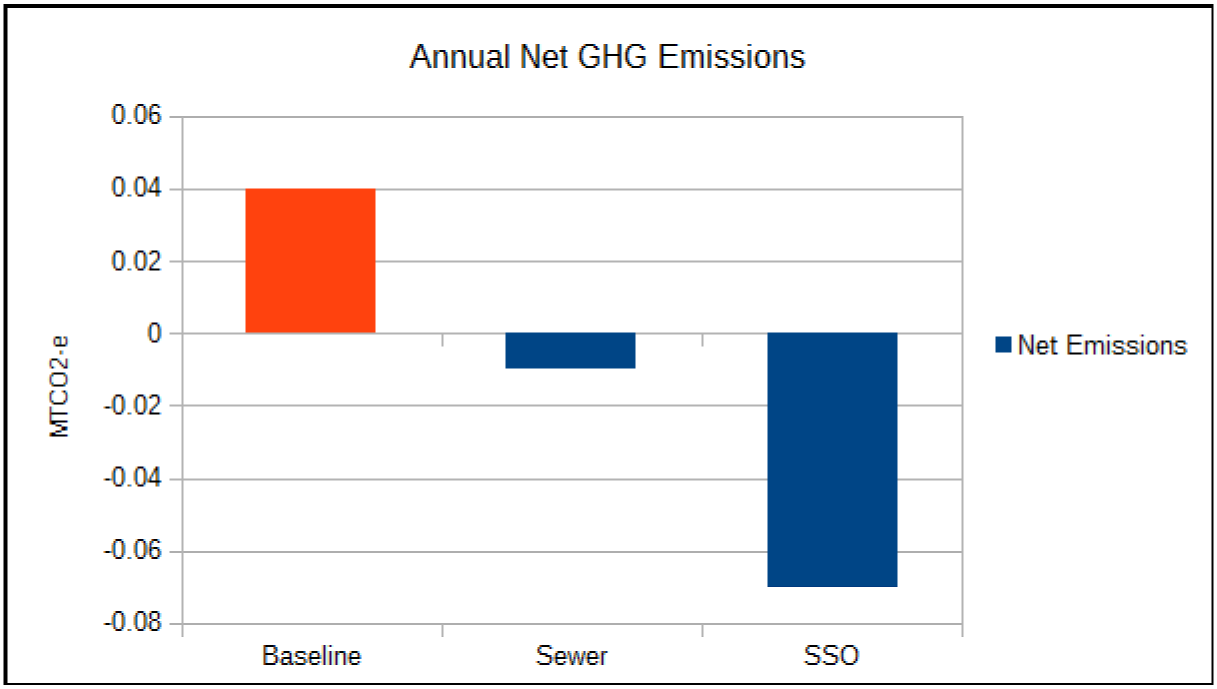
All three scenarios employ landfill gas recovery and incineration with energy recovery to varying extents. However, on average, landfill gas recovery captures just 40% of the methane produced over the landfill's lifetime (EPA, 2015b). Therefore, scenarios that rely less on landfilling are likely to perform better from an energy standpoint. The energy recovery from the sewer and SSO anaerobic digestion processes is typically much more efficient. The process energy used for anaerobic digestion may be as low as 5% of the energy recovered from the food waste (Bioferm, 2009). This allows for the beneficial use of 95% of the recovered energy.

SSO performs best from an energy standpoint, while Baseline performs the worst. The performance difference between SSO and baseline is 405 million MJ annually. In the larger perspective, this is equivalent to the annual domestic energy use of just five city residents. This suggest that organics processing decisions should not rest on energy considerations alone. However, there may be other benefits that support various processing methods. The EPA CoEAT initiative, for example, stresses improved WWTP nutrient removal efficiencies through co-digestion of food wastes. These benefits must be weighed against any drawbacks of co-mingling organic wastes, as discussed in the nutrient quality section.

(The net energy gain modeled in the scenarios does not imply that municipal solid waste disposal and wastewater treatment are net energy-positive. The scenarios model food waste and yard waste processing only, while solid wastes and wastewater are comprised of significant additional low-energy materials, which typically require net energy to process).

Figure 8 shows net GHG emissions, where lower (and negative) net balances indicate better performance.

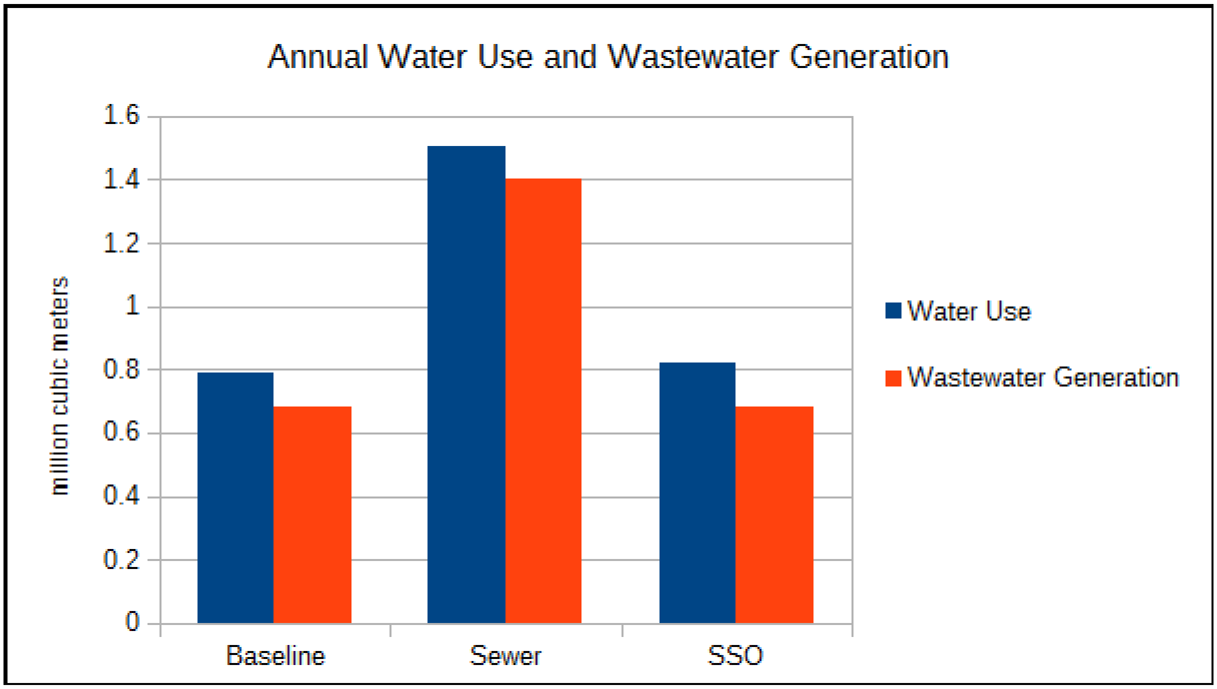
Figure 8 - Net GHG Emissions



SSO results in the lowest (net negative) emissions, Sewer is essentially GHG neutral, and Baseline creates net emissions. As in the energy analysis, higher proportions of landfill and incineration in a scenario account for higher emissions. As noted above in the discussion for these framework components, these energy and emissions values are relatively small compared to other sectors in the city. The difference between SSO and Baseline is less than 120,000 $MTCO_2$-e, which is equivalent to the annual domestic GHG emissions of just over 5,000 city residents, or 0.5% of the city's population. Similar to energy balances, GHG avoidance is unlikely to be a compelling environmental reason to choose one organics processing method over another. However, there may be salient social benefits in highlighting emissions performance, given public interest in climate change.

Figure 9 shows annual water use and wastewater generation for the scenarios. The baseline and SSO scenarios perform similarly in terms of water use and wastewater generation, because there is an identical amount of food waste processed through WWTP in these two scenarios.

Figure 9 - Water Use and Wastewater Generation



Processing greater amounts of food waste through WWTP in the sewer scenario results in higher levels of water use and wastewater generation compared to the other scenarios. The additional 68-70,000 m^3 of water use and 80 million m^3 of wastewater generation amount to less than 1% of the city's annual domestic totals.

Scenario Rankings

Overall, the SSO scenario performs best across the majority of indicators, as hypothesized. It conserves the greatest amount of organics with the lowest levels of contaminants per unit of recovered nutrients. While landfilling and combustion create

overall lower levels of contaminant loads (modeling assumes no leachate), they also sequester significant amount of nutrients. SSO metals levels are similar to Sewer levels, but biosolids contain a host of emerging contaminants such as PCPPs, which are (assumed) not found in SSO food or yard waste. The main barrier to SSO implementation is currently a lack of infrastructure, though urban SSO and AD is increasing. In addition, small and relatively inexpensive AD units designed for processing food waste have recently come onto the market in the U.S. (e.g., units from Impact Bioenergy). These developments indicate increasing interest in the SSO option.

The Sewer scenario performs second best overall, especially on the important organics recovery and P recovery components. Contaminant levels are higher than the other options, but that is partially due to a decision to model emerging contaminants similarly to metals, due to a lack of knowledge about their long term effects on soils and food crops. Water use is the highest among the scenarios, and this is the only option that generates wastewater for organics processing. These may be important considerations in some contexts, but not in others. One advantage of the Sewer scenario is the ubiquity of existing infrastructure in large urban areas. WWTPs that have excess AD capacity can bring in more food wastes, increasing overall nutrient removal efficiencies for food wastes and excreta (not modeled here) in some cases (EPA CoEAT, 2010).

The Baseline scenario performed the worst across most of the important metrics. Baseline was equal to or better than other scenarios in terms of water use, wastewater generation, and contaminants, mostly due to significantly lower organics recovery and P

recovery rates. Essentially, landfilling and combustion sequester useful resources along with contaminants. As more jurisdictions realize the downsides of this sequestration, policies banning landfilling are becoming more common (e.g., MA, TX and others).

The analysis demonstrates application of the framework to realistic scenarios. While the framework identifies those metrics that are likely to be most important overall, their relative significance will vary by context. For example, water scarcity in a given city will increase the importance of the water use component relative to other metrics.

Policy Analysis

Potential Policy and Management Applications of the Framework and Findings

Nutrient and Contaminant Policies

Various federal, state, local, and industry standards regulate urban organic waste processing and food production. For example, regulations govern the use of aerobic and anaerobic digestates, WWTP biosolids, and conventional fertilizers for agricultural production. Contaminant levels in these materials, including heavy metals, legacy compounds, and emerging contaminants are significant from soil, crop, and human health perspectives. This study has highlighted regulated (metals) contaminants and shows that biosolids and composts may have similar levels. Both are significantly lower than EPA CFR part 503 standards for the cleanest class of biosolids for land application. These regulations have also become the de facto standards for composts. However, these (1993) standards may be outdated, as metals levels have been steadily

declining at WWTPs due to better source controls. The standards allow metals levels far in excess of average native soils and urban soils levels, sometimes by an order of magnitude. This suggests that the current policy may be too lenient, although the practical significance of revised standards may be limited, unless the allowed levels are lower than current levels found in biosolids.

One area of possible policy reform concerns national organic production standards, which prohibit the use of biosolids on organically certified crops, but encourage the use of composts (and manures). Seemingly, the similarity between biosolids and composts would mitigate against this distinction. (There are initiatives to allow WWTP effluent irrigation in organic production, which would seem to have the same contamination drawbacks as biosolids).

On the other hand, part 503 standards do not address the majority of the 145 chemicals and compounds of interest found in the latest (2009) TNSSS. There is a general lack of research on the effects that these contaminants may have on soils or crops. There is research that examines the degree of degradation of some of these compounds in WWTP processing and biosolids composting (CCME, 2010; Xia et al., 2005). While the degree of degradation can be substantial for certain classes of compounds, it is difficult to know with any certainty how the combinations of chemicals might interact and how their metabolites might affect soil amendments. Research has documented effects on aquatic organisms at very low concentrations of some emerging contaminants (Blair, 2015). Effects on soil biota may be similar.

This study has posited that composts have much lower levels of non-metals contaminants than biosolids, due to the lack of transmission pathways. However, actual levels and effects remain a research gap. This gap may persist, as soils tend to respond slowly to inputs and only long-term soil tests are likely to provide answers. Given this uncertainty, the (USDA) National Organic Program's (NOP) prohibition of biosolids may be a prudent standard, based on the precautionary principle.

A similar precautionary principle operates at the municipal level. As previously noted, in many jurisdictions urban food production is disconnected from urban soils due to public health and liability concerns about metals uptake (Pfeiffer et al., 2013). These blanket restrictions are counter to science- and evidence-based approaches, and ignore the significant variability found across urban soils due to historic uses (Halloran & Magid, 2011). While soil testing can be cost-prohibitive, tools such as X-ray fluorescence (XRF) technology allows rapid soil contaminant assays of large areas, for further testing of potential hotspots (Clark et al., 2006; Kondo et al., 2016). The foregoing suggests that municipal precaution may be misplaced. Testing of both soils and soil amendments for contaminants, as well as avoiding contamination through source separation, may be more effective precautions. EPA (2011) identifies acceptable soil conditions for urban food production, given low edible-portion plant uptake levels of contaminants under many scenarios.

The current study examines the idea that urban food production can function as green

infrastructure if it is physically connected to existing soils and hydrology. Both cropland infiltration and crop evapotranspiration can reduce water and nutrient runoff, and the benefits can be substantial in dense urban areas. However, urban agriculture that is temporary in nature cannot reliably function as GI. Policy reform that addresses urban cropland as a permanent use can also enable GI functionality. Food production also requires water of a certain quality, but not potable quality. Local regulation can govern stormwater practices where urban cropland is collecting runoff. For example, codes might require first-flush diverters to reduce contaminant levels where roof runoff is used for crop irrigation.

Contaminant levels for products marketed as fertilizers are regulated by the states, and many have adopted the heavy metal standards promulgated by the Association of American Plant Food Control Officials. Allowable contaminant levels are tied to the guaranteed P (P2O5) levels in the fertilizer product (AAPFCO, 2015). For some contaminants, these standards are roughly in accordance with the EPA part 503 rule, and for others, differ widely from the rule. Nutrient levels are regulated for fertilizer products only, in the form of minimum guaranteed levels for N, P, and K. Nutrient levels in composts and biosolids products are not required to be listed unless marketed as fertilizers or under certain voluntary industry programs such as STA.

The wide variation in contaminant standards among compost, biosolids, and fertilizer products, as well as variations in nutrient levels and reporting requirements among these products, suggest a number of knowledge and policy gaps for agricultural

production. The current study suggests that policy can be much more effective when it is based on sound science and adequate relevant information. The framework is one means of deciding what is relevant and also what remains to be known.

Infrastructure Path Dependence

Infrastructure path dependence is an important policy consideration for organic waste processing. Where infrastructure is costly, long-lived, or physically difficult to replace, it may limit transitions to newer technology or better practices. In these cases, society remains tied to older, often sub-optimal ways of doing things. Path dependence can inhibit innovation and new, often disruptive, technologies. The existence of large-scale and long-lived infrastructure, such as wastewater treatment plants, landfills, and incinerators, profoundly affect urban organic waste processing decisions (Forkes, 2007; USDA, 2014).

Federal, state, and local policies that encourage the use of existing infrastructure for organic waste processing, through facilities upgrades or operational changes, may sometimes be appropriate. Alternatively, such policies may be encouraging the use of unsuitable and unsustainable organics processing methods, and may have the effect of extending the lifespan of outdated technologies that should be replaced instead. Villeras et al. (2008) make the case that source separation (greywater, urine, and feces) can help to mitigate issues of emerging contaminants and antibiotic resistance in wastewater and biosolids. This kind of source separation could facilitate the adoption of new types of mostly decentralized infrastructure, in opposition to centralized

infrastructure path dependence. Since urban food production is at its core a decentralizing practice, synergies with decentralized organics waste processing may occur. Some developed countries in the Western world are already on the path to source separation of excreta. A recent review of 38 urine-separation projects in 7 European countries found high levels of acceptance among users, both for the domestic technology (75-85%), and for the idea of consuming food grown from urine-fertilization (Lienert & Larsen, 2010).

One example of a policy that may encourage path dependency is EPA's COEAT program. It encourages WWTPs to increase energy production by co-digesting food wastes as part of the wastewater treatment process (EPA, 2010). While energy output is increased, the resulting digestate may have higher contaminant levels than source separated organics, as argued in the present study.

In addition, some WWTPs expend significant (and potentially offsetting) energy for processing and drying larger sludge volumes. The present study models a co-digestion process (food waste from in-sink disposal units processed via WWTP) in terms of energy, water and emissions. In some cases, co-digestion may be a sustainable option. The present study highlights that fact that policies must be designed to account for the appropriate contextual variables under different scenarios.

Additional examples of policies that may be informed by the present study include household organic waste regulations and incentives; kitchen sink food disposal

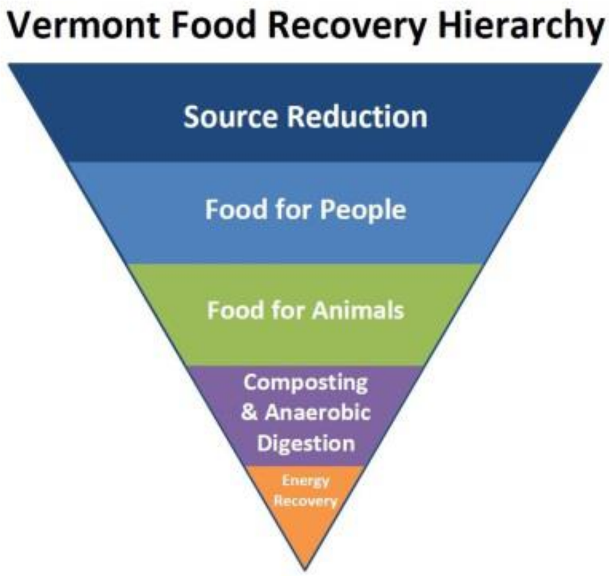
incentives; state fertilizer phosphorus regulations; and green infrastructure incentives, among others. This suggests a modification to EPA's food recovery hierarchy (below) to include more nuanced range of processing options that recognize the value of clean nutrients. For example, the food recovery hierarchy ranks energy recovery via anaerobic digestion (Industrial Use) higher than creating soil amendment (Composting).

Figure 10 - EPA Food Recovery Hierarchy

Source: www.epa.gov/sustainable-management-food/food-recovery-hierarchy

However, the results in the present study suggest that there may be greater value in creating quality, low-contaminant soil amendment if the alternative is higher contaminant soil amendment via WWTP with energy recovery. There is also no mention of source separation in the hierarchy, or the idea that digested wastes can be used as soil amendment, either directly, or subsequently composted (as modeled in the present study). The framework developed here is one place to start. Vermont’s version of the hierarchy more closely reflects this modified approach to food recovery.

Figure 11 - Vermont Food Recovery Hierarchy



Source: www.anr.state.vt.us/.../solid/urs/images/VT_FRHierarchy.pdf

Economic and Social Drivers

The present study focuses explicitly on environmental analyses, and advocates for science-driven policy. However, policy is often driven more substantially by economic and social factors. The framework hints at some of these, such as energy consumption (and costs) in wider contexts. The framework allows the determination of environmental factors that may be relevant in economic or social spheres. This study argues that, in general, the energy and emissions benefits from optimal organics waste processing and local food production are relatively slight. However, energy and emissions are often very socially and economically relevant, and cost–benefit analysis does not always (perhaps rarely) consider larger societal impacts. For example, the idea of generating power from wastes may convince a WWTP to purchase an anaerobic digester. This may increase local alternatives for waste processing and provide energy independence for the

WWTP. It also may increase the competition for waste streams, and shut out the producers of cleaner soil amendments. On the other hand, a municipally owned digester with may increase the availability of locally-sourced soil amendments, especially if it has the power to enforce separation of organics at the source.

Energy production is an important research topic, and new forms of power generation from wastes, such as microbial fuels cells, are on the horizon. These kinds of research and innovation may encourage societal interest in organics processing in their own right, even though their ultimate potential is yet to be determined. The present study includes anaerobic digestion in the scenario analysis because it is a novel (at least in the U.S) and emerging technology for processing food wastes, one that has energy and emissions implications. These factors almost certainly account for some of the current interest in anaerobic digestion. The capital costs of AD systems are generally much higher than aerobic composting systems, and these costs may prove prohibitive for smaller communities.

Economic considerations often drive policy, and one key to improving decision-making is a true accounting of costs and benefits. Better policy can result from accounting for externalities that may not currently be valued. One case study in Cleveland estimated that the monetized GI benefits of urban agriculture could outweigh the food production value by a factor of two to one, and that their combined annual value could exceed $250,000/ha (Shammin & Auch, n.d. in Freshwater Society, 2013). Economic innovations such as GI credits and stormwater utility credits may be applicable to urban

food production in some contexts. In addition, urban farmers may have the incentive to better maintain organic matter levels and soil bulk density (via regular deep tilling) on income-producing cropland compared to other forms of passive GI. These will help to ensure its infiltration performance over time. In contrast, once other forms of GI are installed, owners may be reluctant to invest in the regular maintenance needed to maintain effectiveness (Nader Jaber, City of Milwaukee, personal communication, June 14,2016). Of course, regular maintenance of urban cropland depends on stable land tenure.

Virtual water credits trading, similar in concept to carbon trading, is another way to internalize some of the externalities of food trade (Ravi Shankar & Jayasri, 2015). Credits may make urban food production at the levels suggested in the present study more tenable from an economic standpoint, while being a viable way to address droughts in increasingly water stressed regions of the country where fruits and vegetables are produced. Trade in virtual water may turn out to be complex to administer, but perhaps no more complex than the current system of agricultural and irrigation subsidies. Farming is an inherently risky business, due to vagaries of weather and markets. Until recently, farms producing a diversified mix of non-commodity crops had few means of insuring against losses. With the introduction of UDSA-sponsored Whole Farm Revenue Protection (WFRP), these farms now have an insurance mechanism comparable to the programs for commodity producers. Presumably, urban farms will be eligible for WFRP, which may temper some of the potential economic considerations of growing food in cities. Urban farmers are also likely to favor high-

yielding and high value crops to achieve economic viability. Consequently, they may produce the full range of fruits and vegetables considered in the present study (and other studies) only with the kinds of subsidies that currently exist for a host of commodity crops. These types of policy advancements, in turn, may facilitate longer-term views of urban food production potential, and lead to land tenure reforms that codify food production as a permanent land use.

The social benefits of gardening are well documented, and these considerations are often prominent in discussions regarding urban food production. Because it is largely enclosed and hidden from view, the intensive urban production modeled here might not meet the traditional idea of a garden as community space. This is an important consideration in determining how to balance social and production efficiency needs. Similarly, organics processing and food production in dense urban contexts may pose aesthetic challenges, including unpleasant odors. The choice of organics processing method(s) and soil amendment application practices may hinge on these kinds of nuisance considerations, and these practices are often regulated by local codes. Policies that encourage backyard composting, largely to raise social awareness of recycling and food waste, for example, can in fact have detrimental environmental consequences. For example, where moisture levels are too high, anaerobic digestion may occur, creating nuisance odors and releasing methane, a more powerful GHG than the carbon dioxide released by aerobic composting (Lundie & Peters, 2005). In contrast, large-scale centralized and commercial composting is likely to be better managed.

There are social and economic considerations for the collection and conveyance of organic wastes. It may be difficult to achieve high capture rate for food and yard wastes, especially from residential sources, due to social resistance or path dependence. High capture rates may require novel policies and enforcement mechanisms, as well as investments in new technology, such as food waste bins that can be picked up by automated arms on garbage trucks.

In summary, the examples above illustrate some of the range of social and economic considerations that may drive urban organics processing and agricultural policy. While the framework here provides a means of environmental analysis, policy results will always be subject to social and economic considerations. These three aspects are core to the predominant ideas of sustainability.

Data Needs for Policy Development

As noted above, the framework can be a useful tool for policy-makers to help determine the feasibility and likely environmental effects of urban organics recycling and urban food production across a range of metrics. Policy analysis can be refined and enhanced with the availability of context-specific information for a given urban area. A number of informational metrics are critical to refine the broad estimates given here for place-specific analysis.

- Population density and an inventory of additional food waste volumes in a given locale are key to determining the feasibility of providing a specific proportion of

the city’s produce needs, from both land area and organic waste inputs perspectives.

    - Population density also determines the relative capability of urban cropland to provide GI ecosystem services (see summary discussion below).
    - Actual food consumption within the city may vary from modeled or standard values based on averages because cities may serve greater or lesser numbers than their resident populations (see food waste discussion). Data on WWTP influent excreta volumes provide information on volumes of consumed foods, from which estimated food waste volumes may be calculated with more precision.
    - Data on yard waste volumes (climate dependent) and other carbon sources, such as sawdust, that may be available will affect the amount of soil amendments that can be produced in a given locale.

- An inventory of existing waste processing infrastructure and its capacities will help in determining the feasibility of alternative scenarios. Examples include WWTP process type(s) and capacities, and the existing means for municipal solid waste disposal, including information on conveyance infrastructure, such as sewer and trucking capacities. WWTP information can be used with the CoEAT (EPA) modeling tool to determine feasibility of organic waste processing via WWTP. EPA’s WARM tool can be used to model alternative organics processing methods in terms of energy and GHG emissions balances.

    - Where composts and biosolids are already being produced, data on their typical nutrient and contaminant levels will inform decisions on their suitability for urban food production. The framework presented here provides examples of these kinds of analyses and comparisons.
- Vacant land area and a realistic assessment of likely alternative uses are key to the long term feasibility and political desirability of urban food production. Land availability is the overarching consideration for urban soil amendment and food production. The amount of land available both drives and limits production capacity. Urban food production and composting are likely to be deployed at the scales modeled here only when these land uses can compete among likely alternatives. (More space efficient production alternatives, such as hydroponics and vertical gardening are outside the scope of the present study, but could be accommodated using the framework approach. Potential caveats are presented in the conclusion below.)
    - Location and proximity of vacant land to areas where there is adequate sun exposure and clean runoff from adjacent features (either natural or man-made).
- Local soil type(s) and the condition of urban soils on vacant land, from contamination and nutrient level perspectives, allows the determination of the feasibility of producing food on native soils, rather than in isolated systems. Soil type data allow for location-specific determination of the likely performance of GI, using models such as SUSTAIN or SWWM (EPA).
    - Data on soil conditions, combined with information on the quality of

locally-produced soil amendments also permit the determination of whether soil amendments are likely to enhance or degrade native soils.

- USDA plant hardiness zone and annual precipitation, combined with local produce availability information, if available, allow the calculation of seasonally-limited food production proportions. These set an upper limit on annual production potential. Cities with more favorable growing conditions than the climate modeled here will be able to meet a greater proportion of produce needs locally, and vice versa. On the other hand, lack of precipitation may be an issue in some areas, requiring irrigation or much larger runoff contributing areas and water storage infrastructure.
- Organics waste processing shares by treatment type in a given city allow the calculation of water use and wastewater values for organics waste given in the framework discussion, and based on Diggelman (1998).
- Where urban gardens are sited on impervious surfaces or in raised beds, an estimate of their soil volumes and densities can be used with the framework values given here to determine the potential for P (and other nutrients) runoff.

Given the availability of appropriate data, context specific information can be used to estimate the environmental impacts at a greater level of precision than presented here. However, due to the variability of natural systems and current research gaps, greater precision may be an elusive goal in some cases. The framework and scenario values presented here can be scaled to some extent based on population and population density to arrive at general approximations, but they are not designed to inform specific

and detailed policy or planning analyses.

Summary Discussion

The environmental framework developed here extends existing research on urban organic waste recycling and urban food production of fruits and vegetables. It addresses these processes together and separately across a range of environmental metrics, and assesses their performance within the wider urban environmental context of a city of one million people. This research quantifies, as a first order approximation, the environmental effects of closing the “nutrient loop” between urban organic wastes and food production. Some environmental impacts are larger than others in the overall urban social-ecological context, while their importance is at least partially determined through social and economic lenses.

This study finds that intensive urban production uses less land, water, and energy than larger scale methods of food production, both conventional and organic. Greenhouse gas emissions are lower for urban intensive production. Compared to common forms of food waste processing, including landfilling and incineration, urban nutrient recycling also demonstrates environmental benefits in terms of water use, wastewater generation, energy, and emissions. Taken together, the combination of urban nutrient recycling with urban food production result in increased environmental performance in all of these categories.

- Land use savings ranges from 0 - 64%, or up to 3,500 ha compared to conventional or organic large-scale production. This may be important in the

wider social-ecological context. It is of critical importance at the global scale, as the detrimental effects of converting grassland and forests to cropland are well-documented.

- Combined water savings from organics waste processing and urban food production could be in the range of 13-15% of the urban population's annual residential water use. Shifting food production from water-stressed areas to urban areas with adequate or excess precipitation could enhance national water security.
    - Wastewater savings from organics source separation and diversion from WWTP processing are less significant, at under 2% of the city's annual influent volume.
- The overall energy and emissions benefits of the urban nutrient recycling for fruit and vegetable production are slight, equivalent to less than 2% of the city's annual figures.
    - Transportation contributes a negligible amount of energy and emissions savings for both food production (food-miles) and nutrient processing (nutrient-miles). Energy use in production processes predominates in both cases.

This research develops data that advance a range of concepts for nutrient, quality, nutrient conservation, and pollution potential for urban food production and various processes of urban nutrient recycling.

- Source separation regimes perform better in terms of nutrient recovery compared

to WWTP, which in turn outperforms landfilling. Incineration allows some nutrient recovery, but its potential is highly variable depending on the specific process(es) involved.

- The phosphorus recovery potential for source separated organic waste processing is 13-44% greater than other pathways.
- Phosphorus runoff potential of recycled nutrients is 1-2 orders of magnitude lower than conventional fertilizer, while plant uptake efficiency may be significantly better.
- The total phosphorus mass contained in soils amended with recycled nutrients is significant, over twice that contained in the annual wastewater effluent for the city.
    - Where urban cropland soils may be prone to leaching and flood erosion, significant nutrient pollution potential exists.
- Nutrient contamination with heavy metals is similar in magnitude for both wastewater-derived biosolids and compost produced from food and yard wastes.
    - In both cases, metals levels may be higher than levels in urban soils.
    - Emerging contaminants are found at levels in biosolids on a par with metals levels. Their long-term effects on soils, crops, and human health is a current research gap.

Scenario analysis suggests that source separated nutrient processing is superior across the range of environmental metrics, followed by WWTP processing. Currently dominant disposal methods, which include landfilling and incineration, perform worst. Results are

significant in the wider urban context for water use, nutrient quantity and quality, and P conservation. Energy and emissions benefits are smaller in magnitude.

The study finds that urban cropland could function as green infrastructure, absorbing stormwater and collecting runoff from impervious surfaces.

- The 2,000 – 4,000 ha of cropland needed to produce 70% and 17% of urban vegetable and fruit needs, respectively, for the city of one million could absorb and infiltrate stormwater from a significant proportion of the total urban land area.
    - At the average U.S. urban population density, as considered here, urban agriculture could mitigate stormwater flows from 2 – 24% of urban land area, depending on soil type, annual precipitation, and design goals
    - In urban areas with higher population densities, this proportion will increase, as more urban cropland would be required, combined with a smaller overall urban land area (but land availability might be an issue). The opposite is true for cities with lower population densities.

Where urban vacant land is relatively plentiful, the case for growing food and providing green infrastructure will be stronger. Since alternative competing uses are less likely in these contexts, the opportunity costs are relatively low. Examples include the de-industrializing city-type that serves as the model in the present study. Where vacant land is scarce, compost and food production land uses may be less compelling, and “higher and better” uses may hold sway. On the other hand, the need for stormwater mitigation may be so great in some areas and at some sites that GI, including urban agriculture, is seen as the highest and best use of the land.

The research framework and scenario analyses suggest a number of environmental benefits of multi-functional urban nutrient recycling and food production, and identify a number of research and policy gaps. The environmental benefits are found not primarily in the commonly advanced areas of energy and emissions, but rather in water use, nutrient conservation and quality, and green infrastructure potential. Further research can employ the framework to address economic and social metrics, and context specific analyses can produce more precise estimates of environmental impacts.

Conclusion

The framework, and its application, comprise conceptual and practical tools, and provide ranges and values for the environmental analysis of urban nutrient recovery for urban food production. Necessarily, the research has been reductionist, to enable the science that can inform policy. Ultimately, though, nature tends to work holistically, and this is certainly true for food webs. Soils are among the most complex ecosystems on the planet, and it may be that we will never have the understanding that we think we need. Crops have been growing in soils for hundreds of millions of years, long before Liebig invented N-P-K reductionism in the 1800's, and Haber-Bosch enabled the agricultural “revolution” of the mid-20th century. Alternative means of urban food production, including aquaponics, aquaculture, and hydroponics, may deviate more or less from the production methods found in nature. The framework can be adapted to cover these alternatives, but the precautionary principle suggests that waste processing and food production processes that conform with nature are likely best over the long

term.

It is likely that natural methods of soil-building and food production will stand the test of time. This study suggests that “biomimetic” methods of organics processing, soil-building and food growing show the most positive effects on the environment. Anaerobic and aerobic digestion are natural processes, and the C:N:P ratio of composts is closer to the needs of soils and crops. Urban production from recovered nutrients, as modeled here, is essentially an organic method, whether or not it is USDA certified as such. Studies of food quality have documented significantly greater phytonutrient content for organically grown crops (Baranski et al., 2014). Other studies consider evidence and mechanisms for apparent declines in food nutrients since the agricultural revolution of the mid-20th century, when synthetic fertilizers became common (Davis, 2009).

These findings suggest that urban food production from recycled nutrients could increase the food nutrient value of crops, with concomitant public health benefits. Maybe it really is all about the soil.

Future Research

The findings here suggest a number of avenues for future research in the areas of water use, soils and runoff, economic considerations, and impacts of the various forms of urban waste processing and food production.

The potential water savings realized from growing food in the city are greatest when precipitation provides all of the crop water needed over extended growing seasons. Water savings are reduced if crops require water from the municipal supply to supplement rainfall, which also results in energy use and GHG emissions for treatment and conveyance. Research that broadens the systems boundaries considered here could produce estimates of these various impacts. Broadening the system boundaries even further would enable a comparison between urban irrigation and irrigation in the areas where the city's supply of fruits and vegetables is currently grown. For example, treatment to potable standards requires more energy than water fit for agricultural use, but conveyance impacts may be greater in rural agricultural contexts. A comparison of the water, energy, and emissions trade-offs between these modes of production would enrich the knowledge base to inform policy and further quantify the effects of urban food production. In addition, research is needed to advance rainwater capture and storage technology for cold climates, to ensure both water quality and timing of availability for crop needs.

The present study suggests that urban soils can be improved through the use of food and yard waste derived soil amendments, and also perhaps by the application of biosolids. By the same token, soils may also be degraded by these amendments, and urban agricultural runoff may pose a risk of polluting urban waters via nutrient runoff. Studies that quantify the long-term effects that these various soil amendments have on soil quality can inform municipal policy, which is currently largely based on technological performance standards (e.g., raised beds) and questionable assumptions about the

levels and effects of soil contaminants. Research needs also include rapid and inexpensive soil and amendment testing methods tailored for the high variability found in urban soils. By the same token, site-based research on nutrient releases from urban cropland under a range of weather and management conditions is needed to develop both best practices and science based policy for urban food production.

Urban food production needs to be economically viable over the long term to reach its full potential scale of production, as considered here. Yields are critical in achieving this scale, and are also critical in achieving economic viability for urban farmers. There is a dearth of rigorous and well-documented yield data for urban agriculture. Research that quantifies both the inputs and outputs of urban food production (including costs and revenues) over the long term can help in assessing its economic viability. Policy research that considers a range of tools, such as GI credits, whole farm insurance, virtual water trading, and organic certification can help to identify and optimize revenue streams for urban farms. Marketing research can identify novel and emerging approaches, such as restaurant supported agriculture, to create stable markets for urban production.

Finally, methods and types of urban food production vary across a wide range, from backyard gardens to the intensive urban production modeled here. Similarly, backyard composting may vary significantly from centralized commercial or municipal composting and anaerobic digestion. The LCAs for water, nutrient, energy and emissions impacts for this range of practices are current research gaps. Research that covers this range

can help to illuminate their different impacts, as well as their significance in social-ecological. It is important to distinguish among the various forms of urban production; some of them have the potential to produce significant quantities of food and human nutrition, while others may serve a range of social and community needs.

Research on urban agriculture is still in its infancy, and there is still much to learn. The LCA framework developed here can inform the direction and shape of future research on a range of related topics.

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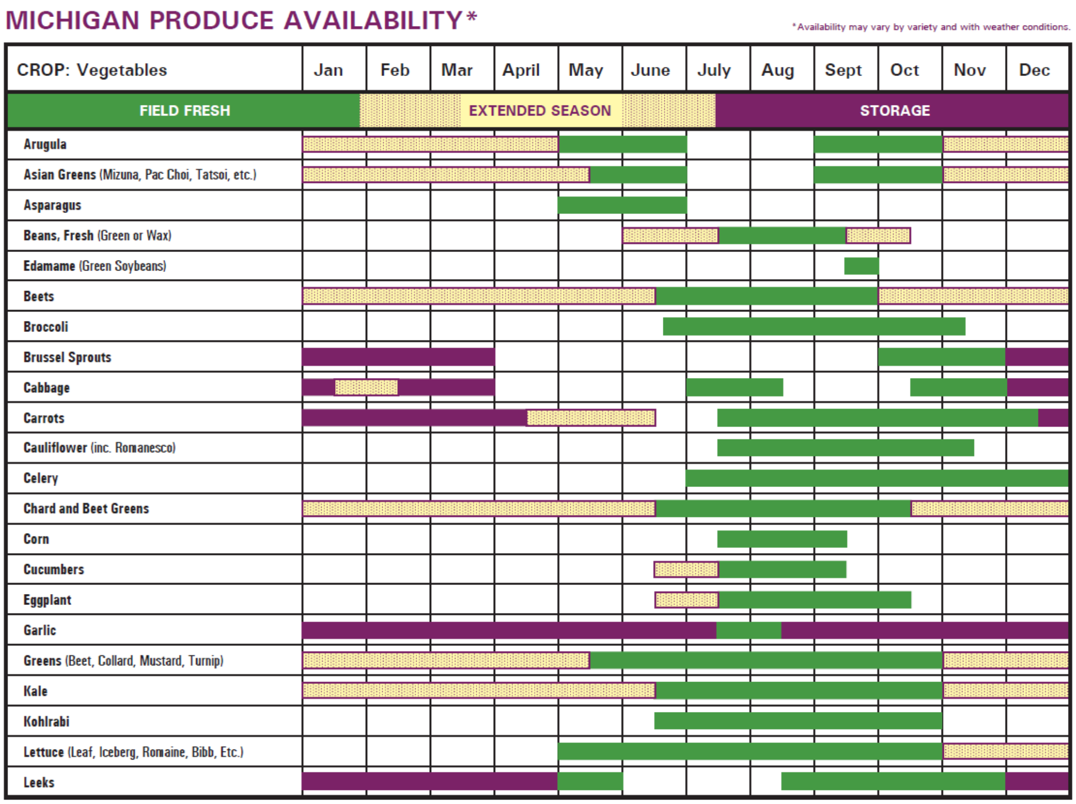
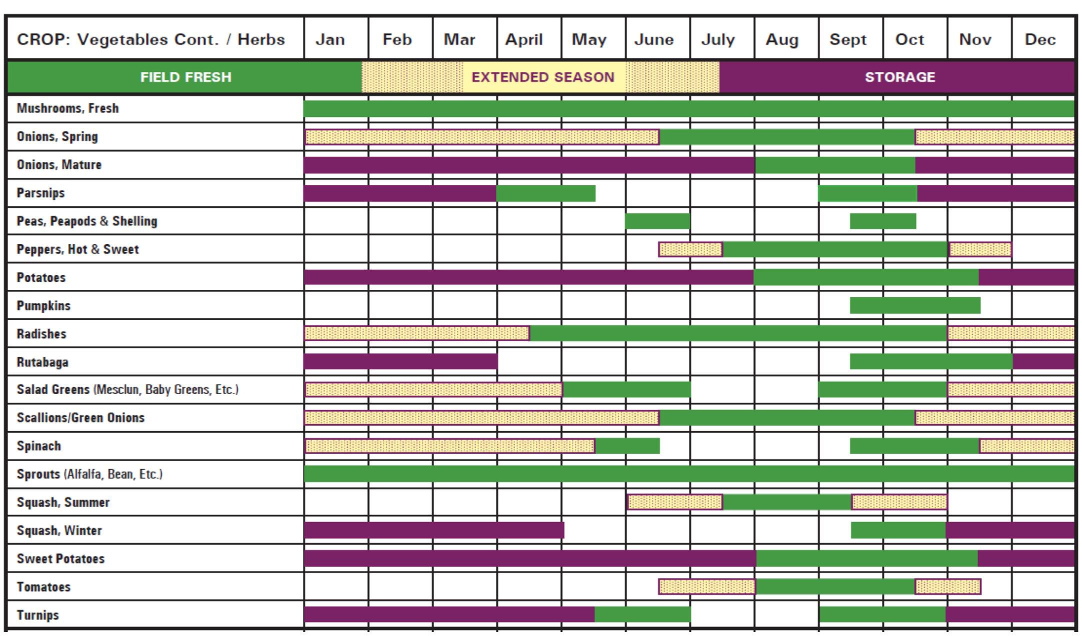
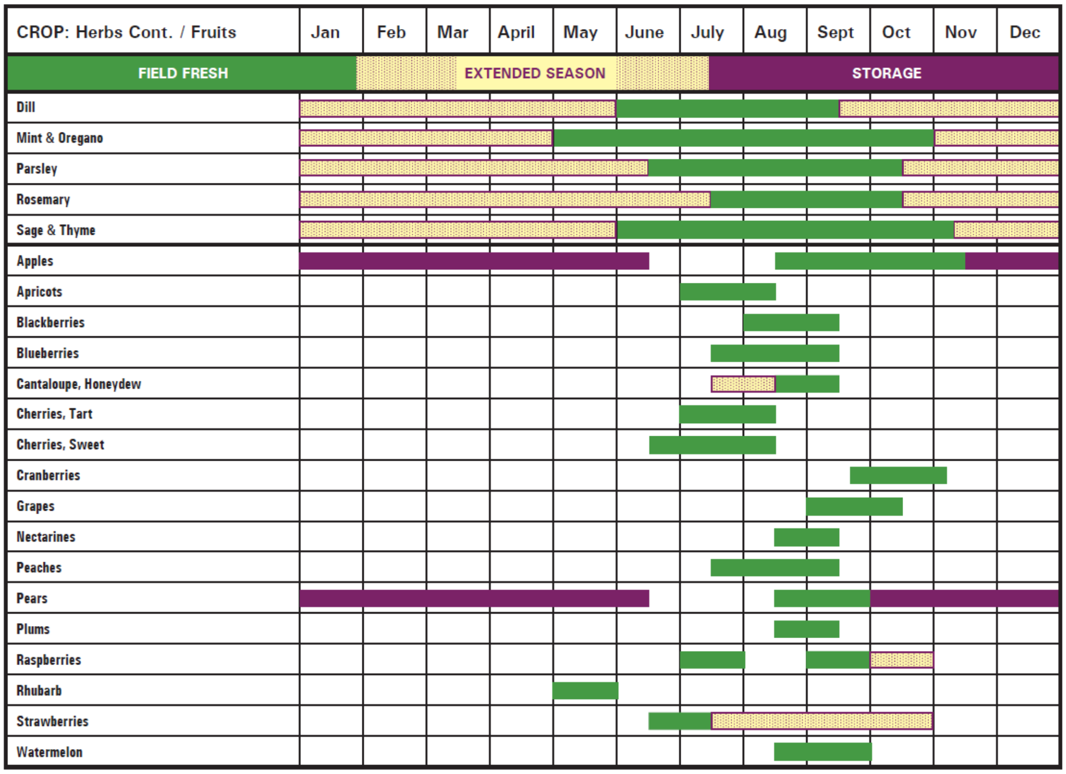
APPENDIX A: Consumption - Land Area - Yields - Dry Matter Calculations

Produce (varieties from ERA Fresh availability)	Michigan Seasonal (climate-limited) Availability Proportion	Hectares – US Average Conventional Production Yields (seasonal availability)	Hectares @ US Average Conventional Yields (100% availability)	100% Farm Fresh Weight Availability kg/capita	Michigan Farm Fresh Weight Seasonal Availability kg/capita	Proportion of Availability That Could be Locally Sourced	National Average Conventional Yield 2004-2013 kg/ha	Jeavons Yields low/med/high – pounds/100 square feet	Jeavons Low Biointensive lb/100 ft2	Jeavons Low Biointensive kg/ha	Jeavons High Biointensive lb/100 ft2	Jeavons Low Biointensive kg/ha	Morra Yields (15/30/45 t dry weight compost/ha) – t/ha	Morra Yields (15/30/45 t dry weight compost/ha) – kg/ha	Iowa State kg/ha	Belitz & Grosch, Kirk Dry Matter Proportion	Dry Matter kg
Lettuce – Romaine & Leaf	0.67	96.3	143.7	5.12	3.43		35,463	135/202/540	135	65,974	540	263,897	20.2/24.7/24.7	20,200		0.051	0.17478
Lima Bean*	0.38	0.6	1.6	0.01	0.00		6,163	50/150/350	50	24,435	350	171,044				0.099	0.000376
Mushroom	1	4	4.0	1.18	1.18		292,827	N/A		0		0				0.1	0.1184
Mustard Greens*	1	5.2	5.2	0.19	0.19		35,463	100/225/270	100	48,870	270	131,948				0.173	0.032005
Onion	1	150.7	150.7	9.14	9.14		60,770	100/200/540	100	48,870	540	263,897				0.109	0.996587
Potato	1	426.5	426.5	17.20	17.20		40,635	100/200/780	100	48,870	780	381,184				0.222	3.817512
Pumpkin	0.17	11.9	70.0	2.09	0.36		29,237	48/96/191	48	23,457	191	93,341				0.087	0.030941
Radish*	1	5.2	5.2	0.21	0.21		40,688	100/200/540	100	48,870	540	263,897				0.056	0.011928
Snap (Green) Bean	0.38	51.6	135.8	0.84	0.32		6,163	30/72/108	30	14,661	108	52,779	6.3	6,300		0.099	0.031676
Spinach	0.75	29.1	38.8	0.82	0.62		21,265	50/100/225	50	24,435	225	109,957				0.093	0.057335
Squash	0.92	91.5	99.5	1.98	1.83		19,910	75/150/307	75	36,652	307	150,030				0.087	0.158799
Sweet Potato	1	105.7	105.7	2.61	2.61		24,567	82/164/492	82	40,073	492	240,439				0.308	0.805112
Tomato	0.42	102.7	244.5	9.07	3.81		36,907	100/194/418	100	48,870	418	204,276	111/119.4/116.6	111,000	88,305	0.065	0.247529
Turnip Greens*	1	4.9	4.9	0.17	0.17		35,463	100/200/360	100	48,870	360	175,931				0.173	0.029756
Fresh Vegetable Totals		**1678**	**2846**	**86**	**60**	**0.70**	**39,936**			**39,022**		**163,404**				**Fresh**	**8.327885**
Fresh + Processed		**3051**	**5174**	**156**	**109**										**Fresh + Processed**		**15.1**
* Yield/ha estimated from																	0
similar vegetables																	0
																	0
Fruit																	0
All Citrus (2013)	0			10.71	0.00												0
All Tropical (2013)	0			20.37	0.00												0
Apples	0.83	206	248.2	7.60	6.31		30,798	50/75/100	50	24,435	100	48,870				0.16	1.00928
Apricots	0.13	0.6	4.6	0.05	0.01		12,041	25/50/100	25	12,217	100	48,870				0.14	0.00091
Blueberries	0.17	10.5	61.8	0.41	0.07		6,344	19/37/75	19	9,285	75	36,652				0.14	0.00971
Cantaloupe + Watermelon	0.15	47.4	316.0	11.43	1.71		35,199	50/72/145	50	24,435	145	70,861				0.06	0.102879
Cherries Sweet + Tart	0.15	9.9	66.0	0.54	0.08		8,007	17/34/51	17	8,308	51	24,924				0.17	0.013668
Grapes	0.13	26.1	200.8	3.60	0.47		17,288	45/67/90	45	21,991	90	43,983				0.17	0.079472
Nectarines	0.08	0.3	3.8	0.07	0.01		17,115	40/60/80	40	19,548	80	39,096				0.13	0.000686
Peaches	0.17	14	82.4	1.66	0.28		19,847	60/90/120	60	29,322	120	58,644				0.13	0.036774
Pears	0.83	31.5	38.0	1.38	1.15		36,919	36/72/108	36	17,593	108	52,779				0.175	0.20059
Plums + Prunes	0.08	3.3	41.3	0.40	0.03		10,634	19/38/57	19	9,285	57	27,856				0.14	0.004446
Red Raspberries	0.25	3.7	14.8	0.10	0.03		6,920	12/18/24	12	5,864	24	11,729			25,408	0.14	0.00357
Strawberries	0.38	21.3	56.1	3.07	1.17		53,822	40/80/160	40	19,548	160	78,192				0.1	0.116622
Fresh Fruit Totals		**375**	**1133**	**61**	**11**	**0.18**	**21,245**			**16,819**		**45,205**				**Fresh**	**1.578608**
Fresh + Processed Fruit		**937**	**2834**	**153**	**28**	**0.18**							**Fresh + Processed (non-citrus, non-tropical)**				**3.9**
Fresh Totals		**2053**	**3979**	**147**	**71**	**0.48**	**30,590**			**27,151**							
Fresh + ProcessedTotals		**3988**	**8008**	**310**	**137**												

APPENDIX B: Michigan Produce Availability







Source: COLASANTI ET AL. 2013

APPENDIX C: Cornell University Compost Feedstock Nutrient and Moisture Content

Material	Type of value	% N (dry weight)	C:N ratio (weight to weight)	Moisture content % (wet weight)
Municipal wastes				
Garbage (food waste)	Typical	2.4	15	69
Grass clippings	Average	3.4	17	82
Grass clippings, Loose	Typical	3.4	17	82
Grass clippings, Compacted	Typical	3.4	17	82
Leaves	Average	0.9	54	38
Leaves, Loose and dry	Typical	0.9	54	15
Leaves, Compacted and moist	Typical	0.9	54	38
Shrub trimmings	Typical	1	53	15
Tree trimmings	Typical	3.1	16	70

http://compost.css.cornell.edu/OnFarmHandbook/apa.taba1.html

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Master of Urban Planning, Master of Public Administration – *University of Wisconsin-Milwaukee, 2012***.**

Water Sustainability Short Course, *Ohio State University, 2012.*

Urban Planning Certificate, *University of Wisconsin-Milwaukee, 2010.*

BA, Philosophy and Political Science, *University of Minnesota, Minneapolis, MN, 1993.*

Dissertation Title: An Integrated Environmental Analysis Framework for Multi-Functional Urban Food Production Utilizing Nutrient Recycling from Organic Waste Streams

Publications

Brault, Erin, et al., *Water on TaP: Water Centric Cities*. UW-M School of Architecture and Urban Planning, 2012. (co-author)

S.B. White, et al. *Water Markets of the United States and the World: A Strategic Analysis for the Milwaukee Water Council.* Milwaukee: UW-M Center for Workforce Development, with funding from the US Economic Development Administration, 2010. (co-author)

S.B. White, et al. *The Contribution of the Bioscience Industry to the Wisconsin Economy.* Bioforward, Madison, WI, 2010. (data analyst and contributing author)

Dearlove, Andrea, and Mary McIntyre, eds. *Red Hot Red, the Wizard of Waukesha* in *In My Neighborhood: Celebrating Wisconsin Cities.* 1000 Friends of Wisconsin. Madison, WI, 2000. (contributing author)

Presentations and Awards

American Planning Association – Wisconsin. APA-WI 2013 Student Project Award for *Water on TaP.*

National Cooperative Grocers Association Conference, St. Paul, MN – 2013. Presentation on board-management strategic alignment in cooperative organizations.

Milwaukee Water Summit, Milwaukee, WI – 2012. National conference presentation of technical, policy, and economic aspects of decentralized urban water frameworks.

UW-M School of Architecture and Urban Planning, Milwaukee, WI – 2010
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Graduate Fieldwork and Research Experience

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