Surveying the Environmental Footprint of Urban Food Consumption

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Summary

Assessments of urban metabolism (UM) are well situated to identify the scale, components, and direction of urban and energy flows in cities and have been instrumental in benchmarking and monitoring the key levers of urban environmental pressure, such as transport, space conditioning, and electricity. Hitherto, urban food consumption has garnered scant attention both in UM accounting (typically lumped with "biomass") and on the urban policy agenda, despite its relevance to local and global environmental pressures. With future growth expected in urban population and wealth, an accounting of the environmental footprint from urban food demand ("foodprint") is necessary. This article reviews 43 UM assessments including 100 cities, and a total of 132 foodprints in terms of mass, carbon footprint, and ecological footprint and situates it relative to other significant environmental drivers (transport, energy, and so on) The foodprint was typically the third largest source of mass flows (average is 0.8 tonnes per capita per annum) and carbon footprint (average is 2.1 tonnes carbon dioxide equivalents per capita per annum) in the reviewed cities, whereas it was generally the largest driver of urban ecological footprints (average is 1.2 global hectares per capita per annum), with large deviations based on wealth, culture, and urban form. Meat and dairy are the primary drivers of both global warming and ecological footprint impacts, with little relationship between their consumption and city wealth. The foodprint is primarily linear in form, producing significant organic exhaust from the urban system that has a strong, positive correlation to wealth. Though much of the foodprint is embodied within imported foodstuffs, cities can still implement design and policy interventions, such as improved nutrient recycling and food waste avoidance, to redress the foodprint.

Introduction

Modern cities neither supply their bulk resource needs nor have the capacity to assimilate their wastes within their borders (Hodson et al. 2012; Chrysoulakis et al. 2013), which given the predominance of urban economies characterized by linear flows (material needs imported, waste produced exported) (Barles 2007; Swaney et al. 2011), has left them physically reliant on their hinterlands and beyond (Rees and Wackernagel 2008). Because cities now accommodate the bulk of humanity and economic activity, they exercise environmental pressures at a global scale through impacts embedded within supporting supply chains and waste management conduits (Weisz and Steinberger 2010; Goldstein et al. 2013; Grubler et al. 2012).

Through the maelstrom of global trade, urban food consumption exerts pressures in terms of greenhouse gases (GHGs) (Dias et al. 2014; IPCC 2014a), land occupation (Moore et al. 2013; Warren-Rhodes and Koenig 2001; WWF 2013; Foley et al. 2011), resource exhaustion (Cribb 2010; FAO 2006), biodiversity loss (Jansson 2013), and a host of other impacts at global as well as regional scales (Heller and Keoleian 2003; Gliessman 2015). It is estimated that the global food system

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causes, directly and indirectly, between 20% and 50% of total anthropogenic environmental pressures (Roy et al. 2012; Notarnicola et al. 2012; McLaren 2010), with the majority attributable to the demands of cities by virtue of their population and wealth. The environmental impacts resulting from a city's food demands have been termed by some its "foodprint" (Billen et al. 2008; Chatzimpiros and Barles 2013), a phrase that will be adopted here. The urban *foodprint* is a term used to capture the various elements of diverse resource consumption and environmental impacts associated with the production, processing, distribution, and waste generation of food demanded by urban residents. The foodprint may be measured in a variety of ways and include units of mass, embodied carbon, ecological footprint (EF), nutrient flows, or other relevant indicators.

Despite the strong link between food and the environment, urban foodprints have been largely absent in urban environmental policy, excepting the drive to reduce the distance from farm to city ("food miles")(Hara et al. 2013; Edwards-Jones et al. 2008; Born and Purcell 2006). A recent analysis of climate-change initiatives in 12 key areas by 59 cities ranked "food and agriculture" the third least addressed issue in terms of the number of policy interventions (C40 2014). Castán Broto and Bulkeley's review of climate-change mitigation interventions in 100 cities does not even contain the word "food" (Castán Broto and Bulkeley 2013). The environmental integrity of the food system is viewed by most urban dwellers (and policy makers) as operating independently of urban built form and therefore only tangentially affected by urban environmental policies (Brunori and Di Iacovo 2014) and, consequentially, receives limited attention from urban decision makers (Grewal and Grewal 2012). This rift is the outcome of fossil-fuel–based agriculture and transportation systems that have shifted food production well beyond municipal borders since industrialization, effectively obscuring urbanites from much of the land-use conversion (LUC), climate-change impacts, biodiversity losses, eutrophication, and nonrenewable resource exhaustion that stem from urban food demands (Cribb 2010; Marx 1976), though cities do deal with food waste (and will have to contend with future climate-change impacts). This rift is further intensified by the expansion of urban areas into urban agriculturally productive urban hinterlands that could provide local food to cities (Seto et al. 2011).

The low prioritization of foodprints on the urban agenda represents a lost opportunity to address significant urban environmental pressures as cities continue to grow in size and wealth (Kennedy et al. 2014a) and adopt more environmentally intensive diets predicated on increased animal product consumption (Tilman and Clark 2014). An accounting of the scale and nature of the foodprint is required to highlight the need to explore potential urban design and policy interventions to tackle it at the city level. Currently, a knowledge gap persists given that only a handful of studies of urban nutrient flows have directly addressed the issue (e.g., Færge et al. [2001], Forkes [2007], or Kennedy et al.'s [2007] grazing of the subject in their review of urban material and energy flows). Moreover, though overviews exist for other important urban pressures, such as building energy (Grubler et al. 2012; Steemers 2003), transport energy (Grubler et al. 2012; Kenworthy and Laube 1996), and water use (Darrel Jenerette and Larsen 2006), but urban food has not received congruent treatment. Thus, the motivation for cities to properly acknowledge, and consequently mitigate, their foodprints is diminished.

Though a gap is present in this sphere of urban sustainability research, much work has been done to document the foodprint of urban systems. For decades, environmental scientists have been documenting the energy and material metabolism of cities (Kennedy et al. 2007). Of the dozens of studies of cities, many have included food, yielding considerable data on individual urban areas, but this piecemeal manner of quantifying the foodprint on a study-by-study basis has not coalesced into a cohesive conversation about this important driver of urban environmental burdens. A survey of this body of literature is an ideal starting point from which to begin this dialogue. Through a comprehensive literature review, this article consolidates the results of urban foodprints to develop a broader narrative surrounding the environmental impacts of food consumption in cities. Through this synthesis, we will sketch how urban food demands translate to environmental impacts and highlight future challenges in managing and reducing the urban foodprint.

Quantifying Urban Foodprints: Review Methodology

Providing a synopsis of the urban foodprint requires a methodology to measure urban food flows and, potentially, the embodied environmental burdens of upstream production. The field of industrial ecology (IE) is well situated to address this need, with its focus on the scale, nature, and interconnections of material and energy exchanges between different sociotechnical systems and the environment (Ferrão and Fernández 2013). It is from this discipline that the *urban metabolism* (UM) concept arose (Kennedy et al. 2007).

UM applies IE principles to the geographical region (city, conurbation, and commutershed), accounting for selected material and energy exchanges (Kennedy et al. 2014b) and, occasionally, using network analysis, between sub-urban systems (e.g., heavy industry and waste management) (Li et al. 2012). Since Wolman's (1965) seminal publication, the material flow analysis (MFA), mass-based framework has been complimented by other methodologies. Carbon footprinting (CF) (Ramaswami et al. 2011) and water footprinting (Vanham and Bidoglio 2014) account for UM-related GHG emissions and embodied water flows, respectively, whereas EF quantifies the bioproductive area underpinning consumption and sequestration of carbon dioxide (CO_2) (Wackernagel 1998). Emergy accounts for embodied energy in UM flows (Stanhill 1977), whereas the life cycle assessment (LCA) tool estimates the environmental impact potentials of UM in a broad range of indicators throughout the supply and waste management chains (Goldstein et al. 2013).

This review is focused on MFA, CF, and EF assessments of the foodprint because these assessment methods are the most represented in the literature. The MFA studies were not limited to complete accounts of all major UM flows, but also include substance flow analyses of nitrogen or phosphorous through urban systems, if urban food needs were also included. Each of the three methods has its strengths and weaknesses, complimenting one another to provide a balanced perspective of the foodprint. Urban-scale MFA accounts for physical flows through cities, avoiding the uncertainties of abstracting out to other indicators further along the environmental cause-effect chain. Conversely, the scale of mass flows says little about the environmental impacts embodied within mass, though it can highlight deleterious exchanges between sociotechnical systems and the ecosphere. CF provides both an indication of an actions contribution to society's largest environmental challenge, whereas it is also easily understood within policy, economic, and public spheres; however, as a single indicator, it can ignore other potentially negative environmental impacts ('burden shifting'). EF quantifies the amount of global average bioproductive land and sea commandeered by humanity, providing an indication of "ecological overshoot" and encroachment on animal habitats. However, EF is limited in the variety of waste flows it captures (only CO_2) and that it is usually based on land-use data at national levels, ignoring the considerable heterogeneity of bioproductivity within countries. Table 1 outlines the essential properties of these indicators as they pertain to the foodprint.

Identification of Studies

The review began by isolating comprehensive literature reviews of UM studies. For UM, Decker and colleagues' (2000), Kennedy and colleagues' (2007, 2011), Zhang's (2013), and Stewart and colleagues' (2014) all provide good lists of essential UM studies at their respective publishing dates. Private and public databases were also utilized to find material within the review scope. Though the focus was on peer-reviewed material, other gray literature document types were considered for inclusion (e.g., theses, reports, and so on). Strategic key terms related to UM (e.g. "urban metabolism," "urban substance flow analysis," "urban ecological footprint") were used to probe 15 databases (e.g., ISI Web of Science, Google Scholar, Oxford Journals, science.gov, Technical University of Denmark, Scopus, and so on).

Urban Metabolism Studies Included

A total of 206 texts on UM were found. This number was reduced to the pertinent literature through a number of limiting criteria: (1) food flows were included in the study; (2) the foodprint was separately presented or disaggregated using minimal manipulation (reducing risk of error and/or misinterpretation); (3) a demand-side urban foodprint was calculated related to urban food *demands* (the sum of food consumed and wasted) not urban food *production* (e.g., scope 1 and 2 CFs); and (4) literature was published in or translated to English. Moreover, primarily qualitative historical narratives or highly speculative forecasts were excluded. With all criteria applied, 43 studies were reviewed, covering 100 cities, sometimes over multiple years or UM types within the same year, resulting in approximately 132 foodprints. Figure 1 shows the geographical distribution of the foodprints considered, whereas tables S1 to S3 in the supporting information available on the Journal's website provides an overview of where they are used in the meta-analysis.

Some data pruning was performed before the analysis of the foodprints. Li and colleagues' (2013) CF of Macao from 2005 to 2009 was taken as the average foodprint over the study period to avoid the biasing effect of including five nearly identical data points. Similarly, the results for Rosado and colleagues' (2014) and Niza and colleagues' (2009) MFA of Lisbon from 2003 to 2009 were also averaged because of the similarity of their methods (regional trade balance) and findings. Calcott and Bull's (2007) EF study of UK cities accounted for 60 of the foodprints and was taken here as the average for those cities in the study for which city-level gross domestic product (GDP) data were available (see table S6 in the supporting information on the Web). For the four studies for which averages were taken, no large changes in consumptive patterns or foodprints were noted for those assessments (over years or between cities), making the means fair representations of their respective studies. Aside from these exceptions, no manipulations of the original data were performed.

Despite efforts to maintain consistency between studies, discrepancies were unavoidable. The inclusion of tourist and/or commuter activities in the studies was not universal. Differences in study scope between "household" (residents) and "city-wide" (residents and businesses) were also noted, whereby the urban foodprint was underestimated in studies where the scope of urban metabolic activities beyond the household boundary were excluded. System boundaries were also occasionally misaligned for CF and EF studies, whereby impacts from cooking and food waste were typically, but not always, unaccounted. Last, the different methodologies outlined in table 1 were encountered for all the three indicators.

Tables S1 to S3 in the supporting information on the Web provides an overview of the included studies their data sources and methodologies. Organization for Economic and Cooperative Development (OECD) Statistics (2015) provided much of the GDP data that were used in the analysis, but where these were lacking, tables S4 to S6 in the supporting information on the Web outline estimation methods.

Results: The Urban Foodprint

Figure 2a displays the percentage contribution of the foodprint to the reviewed cities aggregate metabolisms for the reviewed assessments. Figure 2b presents a histogram of the foodprint ranks in comparison to other commonly accounted urban metabolic flows, such as the consumption of transport fuels, building energy, aggregates, and metallic minerals. The mode

Table 1 Properties of the study categories considered in the review

Study category	*Indicator*	*Method*	*Relation to the foodprint*
Material flow analysis (MFA)	Per capita annual mass of food demanded by a city (t/cap/a)	Household: statistics of per capita food demands at city, regional, or national resolution Trade: balances of imported and exported foodstuffs at city, regional, or national level	Strengths: • Measures the amount of environmentally intensive foods demanded • Can map food waste and nutrient flows in urban systems Shortcomings: • Ignores environmental impacts embodied in food products
Carbon footprint (CF)	Per capita embodied CO_2 equivalents in annual food demanded by a city (t CO_2 eq/cap/a)	Process based: summing of emissions from processes (farming, transport, etc.) along supply chain Input-output (I-O): coupling of local food expenditures with environmentally extended I-O tables to capture direct and intersectoral GHG flows	Strengths: • Quantifies GHG emissions embodied in food and identifies burdensome dietary choices Shortcomings: • Land-use changes (LUCs) and farm-related land management strategies (e.g., tilling) typically not included in CF studies • Focus on single indicator ignores other food-related impacts (eutrophication, soil degradation, etc.)
Ecological footprint (EF)	Per capita global average bioproductive land requirements to support annual food demands (gha/cap/a)	Component: summing of land-use requirements from processes (farming, transport, etc.) along supply chain Compound: coupling of local food expenditures with environmentally extended I-O tables to capture direct and intersectoral land demands	Strengths: • Links foodprint to Earth's biocapacity and potential encroachment on habitat from dietary choices Shortcomings: • Single indicator • Accounts for single waste flow (CO_2) ignoring other GHGs and important food system waste streams • Land-based indicator biased toward agriculture, potentially inflating foodprint relative to other UM drivers

Note: t/cap/a = tonnes per capita per annum; CO_2 = carbon dioxide; t CO_2-eq/cap/a = tonnes carbon dioxide equivalent per capita per annum; gha/cap/a = global hectares per capita per annum; GHG = greenhouse gas; UM = urban metabolism.

of the foodprint's rank as a contributor to the cities' environmental impacts are first for 62% of the EF studies and third for more than 50% of the CF and MFA studies. It is natural that the foodprint tends to dominate EF studies, a consequence of the method's focus on land use, where agriculture is a dominating activity, whereas its CF and MFA pressures are significant, but less intense. Food production is actually estimated to contribute 24% to 50% of global GHG emissions (IPCC 2014b; Schmidt and Merciai 2014), which hints that the reviewed foodprints may be underestimated given that most of the observed carbon foodprints fall below this range. Looking at the CF methods in table S5 in the supporting information on the Web, we find that none of the CF studies included GHG emissions related to LUC (e.g., shifting from forest to pasture releasing carbon stored in biomass) or tilling (activating bacteria, which produces CO_2 and nitrous oxide). GHG emissions data on the latter are scarce, but estimates of LUC ranges from 6% to 20% of global CO_2 emissions (Hörtenhuber et al. 2014; Garnett 2010), providing evidence that more inclusive CF methodologies might elevate the importance of the foodprint in a city's overall GHG burdens. The foodprint ranks lower in the MFA studies because transport fuels and construction materials flows are much greater. Irrespective of assessment method, the foodprint is generally an important driver of urban environmental impacts.

Figure 1 Locations, years, and study category (MFA, CF, EF) for the included foodprints. MFA = material flow analysis; CF = carbon footprint; EF = ecological footprint.

[After the initial online publication of this article, several of the values used in the article were discovered to be in error and corrected. The mass foodprint value for Toronto was corrected from 0.6 t/cap/a to 1.1 t/cap/a in the underlying data. This value has been corrected by repositioning the datapoint representing Toronto in figure 3a and by correcting the resulting R^2 value showing the correlation between wealth and the rise in food demand to 0.37, both in the text and in figure 3a. Similarly, the value for the Dehli carbon foodprint was corrected in the underlying data from 0.16 t CO_2-eq/cap/a to 0.3 t CO_2-eq/cap/a. Correcting this error resulted in the study average changing to 2.1 t CO_2-eq/cap/a, as reflected in the summary, text, and figure 3b. Additionally, the maximum value for the Lisbon foodprint was corrected to 2.1 t/cap/a.]

Figure 3a shows a scatter plot of mass foodprints (determined by MFA) versus per capita GDP, with detailed data in table S5 in the supporting information on the Web. The average per capita annual mass foodprint for the studies is approximately 0.8±0.3 tonnes per annum (t/a) (tonnes per capita per annum [t/cap/a]—where tonne refers to metric tons, as will be the case for all other uses in the article). Wealth affects a rise in food demand, echoing others' findings (Cirera and Masset 2010) supported by the moderate correlation ($R^2 = 0.37$). The study average and almost all of the case cities are above global per capita (0.5 t/a), implying that continued economic growth and urbanization may intensify global bulk food demands. However, it is clear that food demands cannot grow ceaselessly with income after nutritional needs have been met, which means that a logarithmic relationship between mass foodprint and wealth might also be expected, potentially explaining some of the weak correlation here. A modest difference was observed between OECD and non-OECD cities, where a number of the former lie above the study average. The daily per capita food consumption in the OECD cities is 2.5 kilograms (kg), greater than the amount of food a human can realistically consume on a daily basis (Barles 2009), hinting at excessive demand and food generation, particularly with increased incomes.

Paris's foodprint represented 36% of total regional material consumption given that it is a dense, mature city with high non-durable goods consumption, whereas Limerick's foodprint was only 4% owing to a metabolism defined by large construction aggregate additions to stock. The largest mass foodprints (Paris, 1.8 t/cap/a; Lisbon, 1.4 to 2.1 t/cap/a) utilized urban-level trade statistics to generate a more inclusive assessment (Barles 2009; Rosado et al. 2014; Niza et al. 2009), as opposed to foodprints calculated from household consumption data or national-level food availability balances (e.g., FAOSTAT), which may underestimate the gravitational pull of resources to cities or domestic purchasing power inequalities. Moreover, the Lisbon study also included biomass imported into the metropolitan area for feed, certainly playing an important role in the elevated numbers. The significant error bars around the Lisbon study also show how food demands can fluctuate across years. Nonetheless, the Paris and Lisbon studies suggest that a number

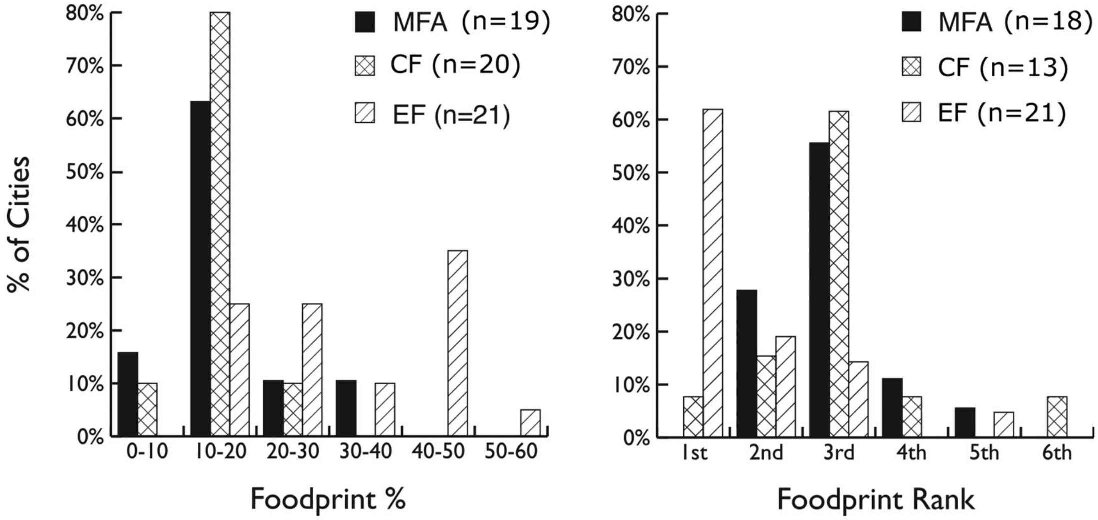

Figure 2 Importance of the foodprint in the urban metabolic profile of the reviewed cities: (a) percentage of cities with foodprint impacts as a distinctive fraction of total impacts and (b) histogram of foodprint's rank compared to other main urban metabolic categories (e.g., transport, building energy, and so on) as a contributor to gross urban environmental pressures measured through MFA, EF, or CF. Ignores studies solely studying food. Sample sizes disagree for CF and MFA because some studies did not disaggregate total impacts into categories in a way that would support ranking. See tables S1 to S3 in the supporting information on the Web for clarification. MFA = material flow analysis; CF = carbon footprint; EF = ecological footprint. [After the initial online publication of this article, an error in the counting of cities included in the foodprint percentage and foodprint rank calculations was corrected, specifically for those cities coming from MFAs and CFs. This has been corrected here in figure 2, parts (a) and (b).]

of cities may have much higher mass foodprints than indicated in figure 3a.

Figure 3b shows carbon foodprint as a function of per capita GDP (details in table S6 in the supporting information on the Web). Average per capita annual carbon foodprint was 2.1 ± 1.1 t CO_2 equivalents (CO_2–eq)/cap/a, representing a carbon intensity of 2.6 t CO_2-eq/t urban food demand. Similar to the MFA assessment, a modest relationship is observed between income and carbon foodprint ($R^2 = 0.30$). Though the non-OECD countries generally perform lower, this is not always a result of economic necessity. For instance, despite its wealth, Macao has markedly lower bovine product intake (Macao 2005–2009 average; beef, 13 kg/cap/a; dairy, 49.9 kg/cap/a) relative to similarly wealthy populations (U.S. 2005–2009 average; beef, 41 kg/cap/a; dairy, 135 kg/cap/a) (FAO 2014). These differences strongly affect the carbon foodprint given that bovine products have large embodied GHG emissions (FAO 2006). Conversely, London's and Cardiff's carbon foodprints were low for their relative wealth (0.9 and 1.1 t CO_2-eq/cap/a, respectively), though these foodprints are likely underestimated considering recent findings that peg the average UK resident's carbon foodprint at 2.7 t CO_2-eq/cap/a (Berners-Lee et al. 2012). Macao's development is divergent from the findings of longitudinal studies at the global level that have found shifts in diets from traditional food systems toward highly processed foods and increased meat intake (Tilman and Clark 2014; Monteiro and Cannon 2012). Figure 4 corroborates this finding by removing the outlier Macao, providing a strong positive correlation between the carbon foodprint and GDP at the urban level ($R^2 = 0.65$). This finding, combined with the fact that the CF models in the reviewed foodprints ignore LUC and tilling-related GHGs, means not only that the CF plays a larger role in a cities embodied GHG emissions than is currently acknowledged, and that these emissions are poised to grow lockstep with economic development in many countries. Geography should not be discounted, given that cities located in regions with longer growing seasons or highly productive agricultural lands might be able to locally supply more of their nutritional needs, thereby reducing food-miles and embodied energy, though the sample size precludes an analysis of this.

Ecological foodprint as a function of per capita GDP is shown in figure 3c. Average per capita annual ecological foodprint is 1.2 global hectares (gha)/cap/a, with an eco-efficiency of 1.5 gha/t urban food demand. The scatter plot was found to best fit a logarithmic curve ($R^2 = 0.35$), with EF quickly growing with income and then leveling off above US$10,000. Moreover, even though the study average GDP was more than 2.5 times the global average, the global and study averages were comparable (0.9 and 1.2 gha/cap/a, respectively), showing that economic development quickly leads to demands for higher-quality protein from animal products with large land-use needs for feed and grazing, but that these demands saturate at modest income levels. This is in agreement with the United Nations Environment Program (UNEP 2012) work showing that per capita meat consumption follows a logarithmic trend that saturates around US$10,000 for national populations. The modest correlation also means that other factors contribute to the EF. Comparative regional market advantage can make environmentally

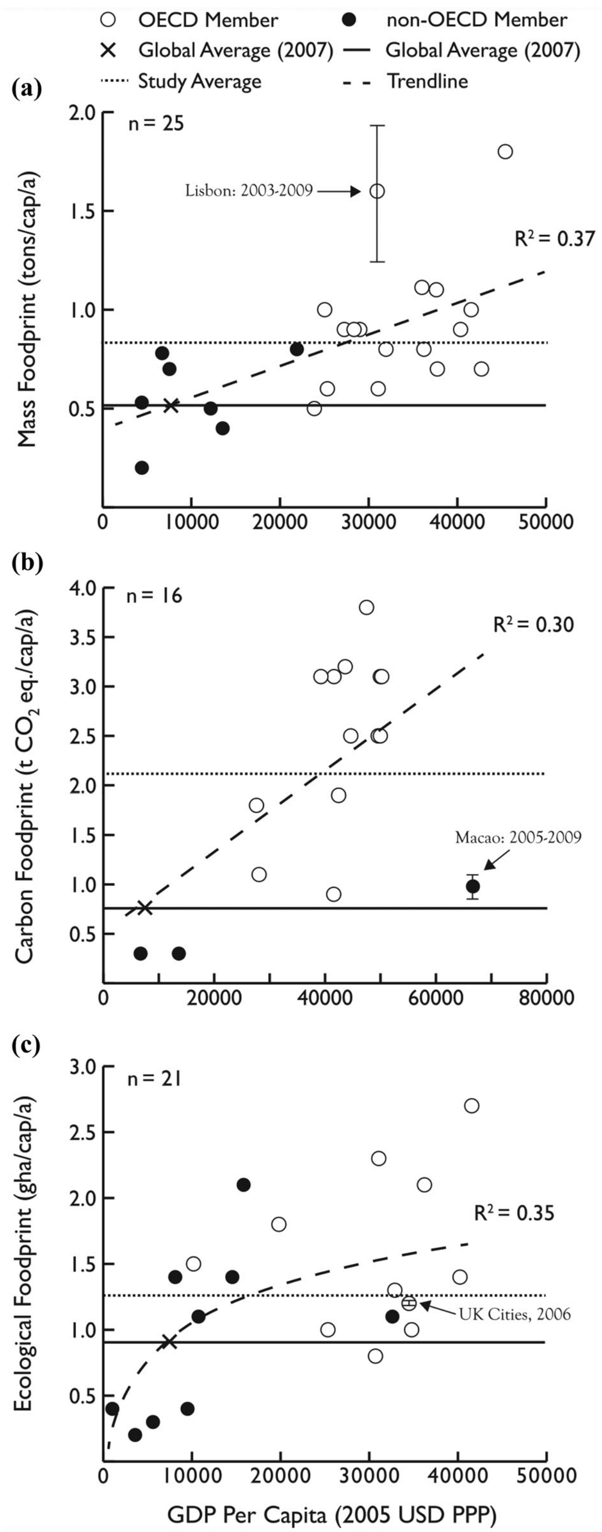

Figure 3 Urban foodprint vs. GDP per capita with foodprint in terms of: (a) mass; (b) carbon footprint; and (c) ecological footprint. Sample size disagrees with figure 2 because additional studies that only included food flows are now included. GDP = gross domestic product. [After the initial online publication of this article, wording in this figure caption was reversed, so that panel (b) is now correctly labelled as relating to carbon footprint, and panel (c) as relating to ecological footprint.]

burdensome foodstuffs affordable to less-wealthy urban consumers (Popkin 2006; Darmon and Drewnowski 2008), such as the cheap beef abundant in South America, which fuels that large EF of Sao Paulo (WWF 2012). In close to 50% of the cities, EF foodprints accounted for 20% to 30% of the overall EF of the cities, with foodprints approaching 50% of total EF burdens for multiple cities. In some unique instances, the EF foodprint played a minor role in the overall UM foodprint, for instance, in Shenyang, China, and Kawasaki, Japan, where the majority of both cities' EFs originate from industrial energy consumption (Geng et al. 2014).

Discussion

The importance of the foodprints in the total environmental impacts of the reviewed cities warrants a deeper look. This section highlights study shortcomings that must be kept in mind in interpreting the results, identifies foodstuffs that strongly influence the foodprint, how the consumption of these evolves with the economic development of cities, and how the design of urban systems can exacerbate foodprints.

Review Shortcomings

This review has relied on a number of disparate studies to assemble an overview of the urban foodprint, with these supporting studies using equally distinct methodologies within assessment study categories (e.g., input output [I-O] vs. process), entity accounted (household vs. city), and data sources (national, regional, or city). This is an obstacle when trying to compare across studies and make inferences on the influence of economic development on the foodprint, because it is hard to disentangle where differences between cities arise because of methodological bias or lifestyle drivers. As such, the correlations of the scatter plots were tested against the influence of these different modeling choices to understand how they affected the results.

Figures S1 and S2 in the supporting information on the Web test the effect of the application of I-O and process-based methodologies on the carbon and ecological foodprints, respectively (not applicable to the included mass foodprints). The I-O method shows a tendency to be higher than process-based carbon foodprint methods for cities of high incomes (no low-income I-O foodprints were available for comparison), a consequence of the recursive GHG flows between sectors captured by the method. Ecological foodprints were insensitive to the different methods. Figures S3 to S5 in the supporting information on the Web show that some methodological bias is present for carbon and mass, but not ecological foodprints, when the unit of analysis is shifted from the household to the city. Household-level studies showed lower impacts compared to the city-level assessments at comparable income brackets, demonstrating that food consumption outside of the house needs to be accounted to accurately reflect urban food pressures. Figures S6 to S8 in the supporting information on the Web show the effects of different data sources on the results, with little discernable difference between city, regional, or national data, except in the Paris and

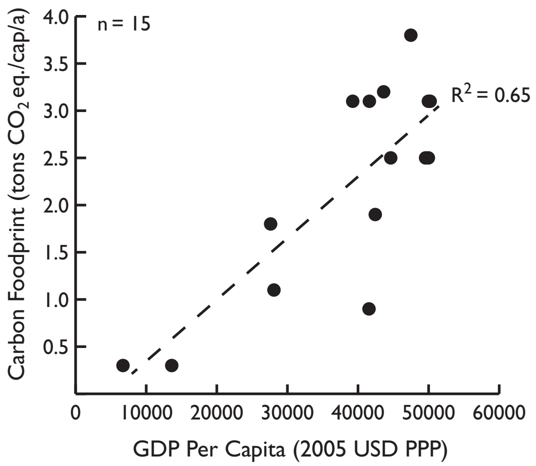

Figure 4 CF vs. GDP with Macao removed from the data set. CF = carbon footprint; GDP = gross domestic product.

Lisbon studies, which had noticeably higher mass foodprints. Most important, the observed trends in the results remained robust, though income ranges of foodprints within some of the methodologies were not broad enough to test correlations between foodprint and wealth.

In terms of the effect of scope, documenting the foodprint was not the goal of many of the studies, causing some aspects of the foodprint to be excluded or conflated with other impacts. Some of the reviewed foodprints allocated energy used in preparation (Wu et al. 2012), and the waste management burdens (collection, processing, and disposal) to building and transport energy segments of the UM studies, increasing those drivers, while diminishing the foodprint. This misallocation is noteworthy given that studies have found that household-side food preparation *can* (contingent on food and preparation method) represent a significant share of a food product's life cycle primary energy demands, and ergo, its environmental burdens (Muñoz et al. 2010; Davis et al. 2010).

A couple of caveats should also be kept in mind when reading the results. Calculating per capita GDP at the city level is a complex exercise with numerous assumptions that can also ignore economic disparities within city regions. Nonetheless, the GDPs here can be broadly interpreted as the purchasing power of the average residents in the cities included. Last, that the majority of foodprints included represent middle- and high-income cities, which may skew the observations upward and make statements about foodprints in the Global South difficult to extract from the data. More foodprints from lower-income cities would strengthen the observations from made here.

Foodprint Drivers

Much like their citizens, each city has a unique foodprint. Notwithstanding, a clear connection between increasing animal product consumption and foodprint was observed, with this trend being ubiquitous across UM methods. Authors of the Cardiff and London carbon foodprints identified dairy and meat products as large contributors to overall CF (Best Foot Forward Ltd. 2002; WWF 2005). The other CF studies did not describe foodprint contributors, either by agricultural source or supply-chain process. The exception was Wu and colleagues' (2012) study of Beijing household food consumption, which identified food preparation as the largest contributor to the foodprint (60%), likely owing to Beijing's fossil-fuel–dominated energy production. Goldstein and colleagues' (2013) UM-LCA study found that air transport of seafood was an important factor in the GHG foodprint of Hong Kong residents. UM studies neglected to mention GHG impacts from deforestation, enteric methane generation, or long-distance refrigerated transport, though these impacts can be considerable (Foley et al. 2011; Born and Purcell 2006).

With the EF studies, animal products feature prominently because of their grazing territory and arable land requirements. In Belfast, meat and dairy accounted for over two thirds of the foodprint (Walsh et al. 2006). A study of Beijing found that the pork consumption was the origin of 65% of the household urban foodprint, increasing to 70% for wealthier households (Zhang et al. 2012). In the London EF study, meat and milk were, respectively, responsible for 28% and 12% of the total foodprint (itself 41% of the city's total EF), with additional significant impacts from other dairy products (Best Foot Forward Ltd. 2002). Beef production requires direct land occupation for feed production, and often, grazing, and indirect land to offset methane production from cattle and deforestation, making it the agricultural product with the highest unit EF (though it would be larger if EF accounted for soil erosion, which reduces the land productivity). This causes high-beef-consuming cities to have corresponding EF foodprints. Sao Paulo residents, with a propensity for beef consumption, had a similar per capita foodprint to citizens from the UK studies, despite the average Brazilian's comparatively lower impacts in many other respects (WWF 2012). Where longitudinal studies of a single city were performed, it was found that the share of these burdensome foods were only increasing (Warren-Rhodes and Koenig 2001; Sahely et al. 2003; Alfonso Piña and Pardo Martínez 2014; Wang et al. 2013), excepting Macao (Li et al. 2013). This was true for advanced and emerging economy cities alike, keeping with global trends of urbanization, economic development, and the shift toward processed, high-energy-density foodstuffs (Popkin 2006; Tilman and Clark 2014).

Foodprint Form

MFA and nutrient balance literature (see table S4 in the supporting information on the Web) revealed a linear foodprint, in line with the general observations of UM studies and other socioeconomic systems (Barles 2010; Huang and Hsu 2003; Ferrão and Fernández 2013). This linearity is defined by the importation of food from beyond the urban boundaries, its ingestion by inhabitants, and the solid and liquid waste (digested and discarded food) sent to repositories typically beyond municipal limits. This contrasts with a natural ecosystem's cyclical metabolism, where material and energy exchanges between

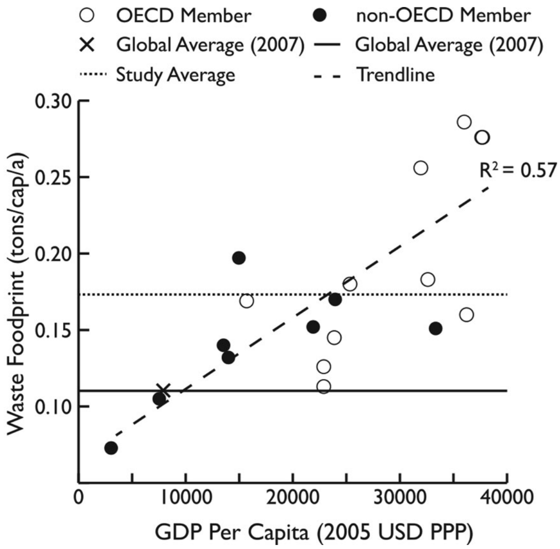

Figure 5 Per capita waste foodprint in tons/annum (t/cap/a) as a function of per capita income. t/cap/a = tonnes per capita per annum. Waste Foodprint is the mass of per capita food waste from a city's metabolism.

components are symbiotic (one subsystem's effluent is another's feedstock), mitigating the concept of "waste," avoiding long-term buildup of noxious substances (Korhonen 2001).

Linear metabolism was observed in the majority of studies, as communicated by the significant solid waste flows destined for city landfills, with biomass being a weighty portion of this. Figure 5 outlines per capita food waste found in the reviewed literature, with all of the data points except two based from urban-level waste statistics. Codoban and Kennedy (2008) found that 44% of food imported into Toronto in 2000 households did not actually nourish residents. With the inclusion of commercial activities on a city-wide level, the percentage of total food sent to landfill were 19%, 20%, 26%, and 31% (0.2, 0.2, 0.3, and 0.2 t/cap/a) in Hong Kong, Vancouver, Toronto, and Limerick, respectively (Warren-Rhodes and Koenig 2001; Moore et al. 2013; Forkes 2007; Walsh et al. 2006). Food waste from the study cities as well as additional urban waste studies cited in UM literature (see table S7 in the supporting information on the Web) were plotted against wealth showing significant positive correlation ($R^2 = 0.57$), which has also been observed for waste in general at the global scale (Hoornweg et al. 2015) and urban food waste (Adhikari et al. 2006). Global per capita food waste over the processing, distribution, and consumption stages was approximately 0.1 t/cap/a (FAO 2013), lower than the 0.2 t/cap/a average food waste for the reviewed cities, which, ostensibly, covers a consumption waste and a portion from processing and distribution. The FAO number is likely overestimated compared to the UM studies, given that significant food processing and distribution (and related waste generation) occurs outside cities. Thus, cities as accumulators of wealth also appear to become centers of excess consumption with economic development, though future research is need to understand whether the organic waste in cities is comprised of high-impact food (meat and dairy) let alone edible food. Even the relatively middle-income city of Bogota relegated 140 kg/a/capita of food to landfills (Alfonso Piña and Pardo Martínez 2014), elevated well above the global average.

Food waste is not only an issue because of the embodied environmental impacts in discarded edibles, but also because organic waste not recycled within the economy escalates nutrient removal and soil degradation at farms, increasing the reliance on fossil-fuel– and mineral-based fertilizers to maintain yields (Jones et al. 2013) and further perturbing global nutrient cycles (Steffen et al. 2015). Another concern are the methane emissions from urban food waste, which are set to grow under current management scenarios leave food to anaerobically degrade in landfills (Adhikari et al. 2006). Highly developed cities with their advanced infrastructures can collect and control their food waste, but despite a renaissance in organic waste diversion, the efficiency of such systems has been mixed (Slater and Frederickson 2001). For instance, Toronto's household compost collection captured only 4.7% of nitrogen, failing to include businesses or the apartments that make up a large portion of the housing stock (Forkes 2007), whereas Paris' food waste was relegated primarily to toxic incinerator fly and bottom ashes, precluding recovery (Barles 2009). Where waste collection infrastructure is lacking, nutrient recycling is not only limited, but also a potential contributor to nutrient-driven algal blooms, as witnessed in the waterways of Bangkok (Færge et al. 2001). Solid food waste has also posed a challenge in cities in the emerging economies, where rotting food has been known to pile in the streets, causing both a nuisance and public health hazard (Hazra and Goel 2009; Hasan and Mulamoottil 1994).

The reviewed cities showed the same pattern in their handling of liquid waste from households and businesses, also an readily accessible source of nutrients (Forkes 2007). Toronto was capturing approximately 90% of digested nitrogen at the wastewater treatment plant, but this was redirected back to landfills owing to public health concerns (IBID). Stockholm more successfully pelletizes sewage sludge to make fertilizer, recycling 60% of phosphorous contained in imported food (Burstrom et al. 1998), a more common practice in Europe. In cities lacking infrastructure, significant household wastewater flows were sent directly to local water bodies harming the ecosystem, as was the case in Bangkok (Færge et al. 2001), Beijing, and Cape Town (Goldstein et al. 2013). Since the 1940s, human waste from cities has been one of the dominant sources of nutrient discharge to global surface waters (Morée et al. 2013).

Urban Design and Policy Interventions

The clear trend of urban foodprints dominated by animal products is a challenge for policy makers trying to affect sustainable urban development. Moreover, the relation between economic growth and the increased consumption of these compounds the complexity of the issue. Having cities intervene in what is largely a matter of personal preference, cultural practice, and politics is likely a political nonstarter in most societies owing to the paternalistic undertones of such tactics. New York City's foray into behaviorally inspired regulation that banned oversized soft drinks in hopes of combating obesity in the city was both publically abhorred and ruled unlawful (Galle 2014), though the city has made strides in reducing food packaging waste (Stringer 2015). A more tractable aspect of behavior to address is edible food waste generation, either through awareness campaigns, organic waste fraction disposal fees, or legislation that curtails food waste generation at commercial operations, such as France's law forcing supermarkets to donate edible food waste to charities or sell it for biofuel production (Chrisafis 2015).

Though, admittedly, cities have limited influence over the types of foods imported or personal waste production, design interventions are still available at the urban level to redress the linear nature of the foodprint. Intercepting the nutrients contained in solid food waste and wastewater for reuse in the agricultural system before they are sent to the landfill or surface waters provides double dividends of reducing eutrophication and avoiding the production agricultural inputs reliant on non-renewable resources (fossil fuels and mineral phosphorous) that are likely to see a 60% increase in demand over coming decades (Tilman et al. 2011).

Historical cities are instructive in this regard through their circular metabolisms that coupled nutrient recycling with food production. In nineteenth century century Paris, latrine residues and horse manure were used as inputs to an extensive horticulture system that produced leafy greens in excess of local needs (Barles 2007). More recently, 1970s Hong Kong pig farming in the territory had a mutualistic relationship with local produce production within the city limits, whereby pigs consumed food waste, while producing high-quality manure and protein (Warren-Rhodes and Koenig 2001). In present-day African cities, low-tech, informal nutrient recycling systems are commonly employed to combine sewage with urban food production, but improper pathogen eradication remains a threat to viability (Srikanth and Naik 2004; Qadir et al. 2010). A more sustainable solution has been found in Kolkota, India, where, for over 100 years, a 3,000-hectare //ha wetlands has processed 550,000 cubic meters of the city's raw sewage daily, simultaneously producing 16% of the city's fish needs and fertilizer for fields, demonstrating ecologically sensitive use of landscape as infrastructure (Newman and Jennings 2008).

Because of the risk of pathogens in nutrients mined from human waste, a multiforked set of solutions to the linear foodprint is required. This is already present in the way that a number of cities apply nutrients in wastewater sludge to fields producing feed crops for livestock, as opposed to crops for direct human consumption (Miljøministreriet 2005). Nutrients collected at wastewater plants are also entrained with heavy metals and other pollutants from industrial wastewater and surface water runoff, portending the need to separate nutrient-rich human waste streams (or effluent from food processing plants) before the wastewater treatment plant (Forman 2014). A potentially effective strategy is the point source collection of bulk of nutrients expelled by humans using urine diversion toilets (IBID, Baccini and Brunner 2014); however, the large sunk costs, slow replacements rates, and centralized structures of urban wastewater collection and treatment systems means that this type of intervention will be difficult in cities with mature wastewater handling infrastructure. Source segregated urban food waste is pathogen free when correctly cured and is thus better suited for human food production. The generation of compost from organic waste both recycles nutrients and enriches soil with organic carbon; however, concerns about toxic metals concentrations remain a challenge (Hargreaves et al. 2008). Composting must also overcome public resistance to sorting and separating food waste and the aversion of municipalities to its perceived higher costs over landfilling (Decker et al. 2000), putting compost at a disadvantage even in developed cities with sufficient technical capacity.

Regardless of the design interventions employed, it is essential that the foodprint be understood from a system-wide perspective. Reducing urban foodprints by moving toward cyclical UM most avoids the pitfalls of focusing on single waste streams, given that this increases the potential for ignoring key food-related flows and reduces the environmental efficacy of these strategies (Kalmykova et al. 2012). Further, cyclical UM remains a challenge given that nutrients embedded in food imports represent a fraction of the nutrients used in production, owing to the fact that swathes are lost in agricultural runoff and microbial action (Baccini and Brunner 2014; Gliessman 2015), necessitating actions at the urban scale and beyond to redress nutrient losses. It should also be noted that cyclical UM schemes need not "close the loop" by coupling with food production near cities (hypothetically, nutrients could be captured in cities and sold on the global market), but such programs have the added benefit of reducing the significant distance that food travels to urban markets (Born and Purcell 2006). Metson and colleagues (2012) documented the symbiotic relationships between the urban dairies in the Phoenix Metropolitan Area and alfalfa farmers, which used waste from the dairies and biosolids from treated wastewater to recycle phosphorous.

Urban Development as a Foodprint Driver

From the data obtained from the literature review, there seems to be a tenable linkage between economic activity and the mass, carbon, and ecological foodprints, as well as the food waste generation. Owing to the higher per capita economic activity in cities, the average urbanite is likely to have more income to spend on food than their rural counterpart, supporting the assertion that cities eat better than the countryside (Hoornweg et al. 2012). The OECD estimates that the share of global GDP from agriculture will continue to decrease, along with crop prices, which would act to decrease the cost of food to many urbanites (OECD and FAO 2015), hinting at further divergence of purchasing power between rural and urban inhabitants. Combining cheaper food with the superlinear economic growth related to urbanization (Bettencourt and West 2010), it seems possible that bulk food demands may also follow a suite as rural populations continue to migrate into cities. Kennedy and colleagues' (2015) review of megacities has already revealed this superlinear scaling in the metabolism of certain metabolic flows (waste, gasoline, and electricity), and future research should explore whether the urban foodprint shares this property.

Urbanization also affects consumption patterns and household food management practices. Figure 3a and 3C shows that the ecological foodprint increases at a quicker pace with wealth than the mass foodprint, as evidenced by the former's logarithmic correlation to GDP. This could indicate that beyond once nutritional demands are met, the increase in the environmental burden from food consumption is not caused by bulk, but by shifts toward foods with higher land use and embodied energy demands. Additionally, as figure 5 revealed, increasing wealth is coupled with a surge in food waste. That is, the increase in the environmental burden observed for increasing GDP is most likely caused by household food management practices and shifting consumption patterns toward expensive food items with larger environmental burdens.

Linkages between economic development and increasing intake of high-burden foods by others support this (Tilman and Clark 2014). Recent United Nations (UN) reports also show that food waste in wealthy nations originates largely at the consumer end (FAO 2013). This evokes an accelerating pattern: As incomes rise, people tend to consume more environmentally burdensome foods, but at the same time consume less of the total food they purchase. Looking deeper into global food waste data, disposal rates of edible food by consumers in wealthy countries are 19%, 8%, 26%, 31%, and 32% for meat, dairy, fruits and vegetables, cereals, and roots and tubers, respectively (IBID). Fruits, vegetables, grains, and tubers are most commonly castaway at the household level; exactly the foods that studies have shown to be more easily accessible in wealthy areas of U.S. cities (Shove and Walker 2010; Algert et al. 2006; Gordon et al. 2011). Wealth is not the sole reason that consumers discard fruits, vegetables, and grains (education, storage options, and other factors are important), but the fact that these foods are more available might promote excessive purchasing by wealthy urbanites.

Last, the spatial characteristic of urban development has an effect on the foodprint, given that low-density growth potentially consumes productive agricultural land at the peri urban fringe. This type of development reduces local capacity for food production, locking residents into increased consumption of food transported over long distances.

Conclusions

Through an assemblage of earlier quantifications of UM, this review demonstrates that environmental impacts from urban food demands are not only nontrivial, but sometimes the largest contributor to a city's environmental loading. In light of this, researchers and cities should be compelled to further develop methods and better quantify the urban foodprint. Such a task is easier said than done, considering the complexities of the food system and its many interfaces with other systems of production and consumption. Notwithstanding these challenges, it is clear that future assessments should leverage multimetric approaches to gauge environmental impacts, given that differences between the three examined metrics in this study mirror the fact that they are linked to different drivers.

The main drivers of urban foodprints are animal-based food products. Consumption of these, and resultantly foodprints, generally increase with comingled urbanization and economic development, though a number of other important factors assert influence (cultural preferences, lower prices, and so on). The UM was also found to be linear in form with low production of food within cities and usually marginal recycling of nutrients in food and human waste back to the agricultural system. Moreover, where proper waste management facilities are lacking, the foodprint can manifest within urban regions in the form of nutrient fed algal blooms that damage local aquatic life. Thus, the foodprint is a multiscale issue exerting pressure at the city level and beyond.

Given the numerous challenges facing the long-term sustainability of the global food system in the coming decades both in terms of resource availability (land, fossil fuels) and minimizing the collateral environmental damage of agricultural production (biodiversity loss, eutrophication), it is essential for cities to evaluate how they can actively contribute to positive change. Given that the food choices of urbanites largely influence the food-related environmental impacts of a city, combating it at the city level requires urban design interventions that redirect the current linear UM to better recycle valuable nutrients and organic carbon within the agricultural system, both locally and abroad. Though many cities already do this to some capacity, there is room for improvement through expanded organic waste diversion and human waste management schemes that reduce the spread of pathogens and toxic chemicals. Behavioral changes should also be explored even if limited in purview. Attacking edible food waste through awareness campaigns and user fees to discourage generation reaps double dividends of landfill diversion and circumventing the environmental loading embodied within food production.

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Supporting Information S1: This supporting information includes tables that identify the material flow analysis (MFA), carbon footprint (CF), ecological footprint (EF), and urban waste food studies that were included in the article's meta-analysis. It also provides some analysis of the parameters of the cited studies.