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Assessing the edible city: Environmental implications of urban agriculture in the Northeast United States

Goldstein, Benjamin Paul; Birkved, Morten; Fernández, John; Hauschild, Michael Zwicky

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Assessing the edible city

Environmental implications of urban agriculture in the Northeast United States

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DTU Management Engineering

Benjamin Paul Goldstein
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DTU Management Engineering
Department of Management Engineering

Preface

The document in your hands is the culmination of three years of work. Work that started out under a different title, but which has eventually nestled comfortably under the name 'Assessing the edible city: environmental implications of urban agriculture in the Northeast United States'. This dissertaion is submitted in partial fulfillment of the requirements for the degree Doctor of Philosophy in Management Engineering, as prescribed by the conferring institute, The Technical University of Denmark. Support and guidance throughout the course of this project were provided by my principal supervisor Associate Professor Morten Birkved, and co-supervisors Prof. Michael Hauschild and Prof John Fernández, Massachusetts Institute of Technology.

The majority of this work was carried out at the Division for Quantitative Sustainability Assessment of the Department of Management Engineering at The Technical University of Denmark between December 2013 and November 2016. Approximately quarter of that time was spent at the Massachusetts Institute of Technology's Department of Architecture, Building Technology Program under the supervision of Prof John Fernández.

The scaffolding of this project are the six articles contained as appendices to the main report, four of which have been published, and two of which have been submitted at the time of writing. The articles are as follows:

1. Goldstein, B., Birkved, M., Fernández, J., Hauschild, M. Surveying the Environmental Footprint of Urban Food Consumption. *J. Ind. Ecol.* 2016.

2. Goldstein, B., Hauschild, M., Fernández, J., Birkved, M. Urban versus conventional agriculture, taxonomy of resource profiles: a review. *Agron. Sustain. Dev.* 2016.

3. Goldstein, B., Hauschild, M., Fernández, J., Birkved, M. Testing the environmental performance of urban agriculture as a food supply in northern climates. *J. Clean. Prod.* 2016, 135, 984–994.

4. Goldstein, B. Hauschild, M., Fernández, J., Birkved, M. Contributions of local agriculture to urban sustainability in the Northeast United States. *Env. Sci. & Tech.* 2017, 51, 7340-7349

5. Goldstein, B., Hansen, S. F., Gjerris, M., Laurent, A, Birkved, M. Ethical aspects of life cycle assessments of diets. *Food Policy* 2016, 59, 139–151.

6. Goldstein, B., Moses, R., Sammons, N., Birkved, M. Potential to curb the environmental burdens of American beef consumption using a novel plant based beef substitute. *PLoS One* 2017. Submitted.

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Summary

One of the pivotal environmental challenges in the coming decades will be feeding an increasingly wealthy and populated planet in a sustainable manner. As industrialization and concomitant urbanization affects hitherto peripheral economies, much of this challenge will depend on the ability to support the nutritional demands of a global urban population in a fashion aligned with the biophysical capacity of the planet. Amongst the myriad of solutions proposed to guide humanity towards more environmentally sustainable food system, co-locating food production and consumption in cities is an area that has seen significant action in research, design and practice. In the Northeast United States, where per capita diets are amongst the most environmentally intensive globally, there is a growing interest in local food production as a way to reduce the ecological burdens of food demand. Urban farms and pro-urban agriculture planning agendas are proliferating throughout many of the region's cities, typically with urban agriculture's environmental sustainability evoked to varying degrees in support of these initiatives. However, environmental appraisals comparing urban and rural food production are scarce in existing literature, leaving a number of lingering questions surrounding urban agriculture's environmental performance. In a Northern context, it remains to be seen whether the benefits of reducing distance from farm to fork are outweighed by the energy demanded by year-round growing systems. Even if urban agriculture does provide leaner resource intensities at the farm scale, do these add up to meaningful shifts in a city's environmental footprint at the urban scale? The aim of this project was to begin removing these uncertainties using the Northeast United States as a case study, since cities within that region have some of the most vibrant and well-supported urban farming communities in the Global North. This report is comprised of six chapters that probe and add to our current understanding of urban food systems.

Chapter one traces the historical development of the physical and psychological rift between city dweller and farm. This rift obscures urbanites from the ecological deterioration that results from their nutritional needs,

which have come to pose challenges at the global level in terms of greenhouse gas emissions, land occupation, water use and toxic chemical loading. Recent years have seen some acknowledgment of these challenges, and many municipal governments, urban designers and citizens are championing local food production as a means to 'green' urban food supply networks. I then define urban agriculture as "local food production woven within the urban fabric utilizing pre-existing material and energy flows" and the edible city as "the proliferation of urban agriculture throughout a city to the extent that it becomes an integral part of that city, both in form and function". I speculate that there is a lack of knowledge at present to support the recent promotion of the practice in cities in the Northeast United States, particularly given the focus of previous urban agriculture research in milder climes. Lastly, the City of Boston, Massachusetts is introduced as a case city, due to its built form and climate, which are both relatively representative of many cities within the Northeast Region.

Chapter two fills a key gap in the urban sustainability literature: the lack of comprehensive review of environmental burdens from urban food consumption. I use existing literature in the field of industrial ecology (the study of material, energy and environmental performance of systems) to illustrate the importance of urban food demands as a driver of a city's overall environmental performance. My review looks at studies that used material flow analysis (accounting of mass flows through a city), carbon footprint (greenhouse gases from city activities) and ecological footprint (amount of land occupied in and out of city to support city activities) to assess urban food systems. I find that the environmental footprint from urban food consumption ('the foodprint') is often on par or greater than many other more recognized drivers of urban environmental performance, such as transport, building heating and water. Moreover, there is a tendency for these impacts to increase in concert with per capita income, portending larger food-related environmental impacts into the future as the world's urban population continues to grow in numbers and wealth. Of concern is the robust correlation between urban food waste and wealth, both because the nutrients within the waste are often not recovered and the environmental burdens embodied in producing food that does not nourish urbanites. Lastly, my perusal of urban sustainability literature finds that even though the urban foodprint has seen low priority on the sustainability agendas of many cities, the past few years have seen cities making the connection between food supply webs and their environmental performance, including specific calls to improve the latter using local production.

Chapter three tests the environmental performance of urban agriculture in Boston. It starts with a review of existing literature that has addressed the environmental dimensions of urban food production. I find that despite some evidence of urban agriculture's superior environmental performance over conventional production in mild climates, support in many sustainability dimensions are wanting. In particular, an assumption that local food is somehow inherently sustainable is a common theme in much of the writing. Moreover, there are a number of claims regarding positive interactions between urban farms and the surrounding environment (rainwater capture, organic waste recycling, building energy use reduction) that are grounded in common sense, but lacking empirical support. To overcome these data gaps, I develop a basic taxonomy of the four different urban farm types that exist using their siting within the built form (ground or roof) and status of the grow space (conditioned and non-conditioned). I then use primary data from six farms in Boston and New York City to quantitatively compare the environmental performance of urban and conventional production. I find that the environmental savings of reducing transport distance from farm to city can easily be negated by energy inputs to some urban farm types. Simple open-air farming in cities can have lower global warming impacts than conventional counterparts, but at the expense of land occupation and water consumption. Even when urban farming does provide environmental benefits, these pale in comparison to the application of urban space for solar electricity production.

Chapter four takes the results from the individual farms and scales them up to the city level to gauge the impacts of the edible city on Boston's baseline foodprint. I combine census data, nutritional surveys and the EXIOBASE 2.2 model of environmental burdens from production (including international trade) to build environmental foodprints of the city's 560 neighborhoods in terms of land occupation and greenhouse gas emissions. The average Bostonian causes the equivalent of 1.2 metric tons of carbon dioxide emissions and 1 hectare land occupation in relation to their food needs. Fully employing urban agriculture to combat these pressures results in a 5% reduction in food-related greenhouse gas emissions and a 1% increase in food-related land occupation. Interactions with the urban environment are also lackluster, mitigating 2% of the city's surface runoff and absorbing less than 10% of household organic solid waste. More substantial are the practices potential dietary contributions, where urban farming could provide a significant proportion of the city's vegetable nutritional needs. Market value of urban farming in the Boston could be as much as $160 million, generating revenue in some of the city's poorest areas. The chapter closes with thoughts about balancing the marginal environmental gains of urban agriculture against its larger social and nutritional benefits.

Chapter five looks at complimentary methods to the edible city that could reduce the urban foodprint. Specifically, I focus in on the fact that most of the foodprint emanates from the animal-sourced products consumed by city-dwellers, and that these are not tackled by urban farming. I build environmental models of the average American, vegetarian and vegan diets to see how much lifestyle changes could affect Boston's foodprint. Switching to vegetarian and vegan diets result in environmental benefits that are significantly greater than those provided with urban agriculture. For instance, a move to vegetarian and vegan diets reduce the city's carbon foodprint by 32% and 67%, respectively. Because it is not realistic to expect the entire city to adopt a meat-free lifestyle, I also look at the ability of a novel plant-based meat substitute (the 'Impossible Burger' from Impossible Food) to reduce foodprint. I find that substituting 10% of annual ground beef intake with the plant based burger provides the same environmental benefits of a fully edible city.

Chapter six concludes with a synthesis of the results and the short discussion of ideas for future research directions in quantifying the foodprint and better assessing the environmental dimensions of urban food production. I close this report with some thoughts on the necessity of quantitative sustainability models to avoid poor urban design solutions to environmental challenges and the importance of urban agriculture's social benefits as these might ultimately be the most justified grounds for its proliferation in the American Northeast.

Resume

En af de centrale, miljømæssige udfordringer i de kommende årtier vil være at ernære en stadigt mere velhavende og befolket planet på en bæredygtig måde. Da industrialiseringen og simultan urbanisering påvirker selv hidtil perifere økonomier, vil udkommet af denne udfordring hovedsageligt afhænge af evnen til at understøtte de ernæringsmæssige krav fra en global bybefolkning på en måde, som er afpasset til den biofysiske kapacitet af planeten. Blandt de utallige løsninger, der foreslås til at bringe menneskeheden i retning af et mere miljømæssigt bæredygtigt fødevaresystem, er samlokalisering af fødevareproduktion og forbrug i byerne, som er et område, der har set en betydelig indsats, hvad angår forskning, design og praksis. I det nordøstlige USA, hvor kostvanerne målt per indbygger er blandt de mest miljøbelastende globalt set, er der en stigende interesse for lokal fødevareproduktion som et redskab til at reducere de økologiske byrder, som efterspørgslen på fødevarer medfører. Urbane landbrug og byplanlægning der tilgodeser urbane landbrug, er tiltag som i stigende grad finder anvendelse i mange af regionens byer, typisk med byernes miljømæssige bæredygtighed som argument for introduktion af disse initiativer. Men miljøvurderinger som sammenligner fødevareproduktion i byer og landdistrikter, er sjældent forekommende i den eksisterende litteratur, hvilket efterlader en række grundlæggende, ubesvarede spørgsmål omkring miljøgevinsterne ved urbant landbrug. I en nordlig sammenhæng er det stadig uvist, om fordelene ved at reducere den fysiske afstand fra jord til bord opvejes af den energi, der kræves til kontinuerlig landbrugsproduktion i byer året rundt. Selv hvis urbane landbrug reducerer landbrugsproduktionens ressourceforbrug, er det stadig et spørgsmål, om disse fødevareproduktionstiltag leder til ønskværdige forskydninger i byers miljømæssige fodaftryk? Formålet med dette projekt er at reducere usikkerheden relateret til de miljømæssige påvirkninger forbundet med urbant landbrug. Til dette formål bruges det nordøstlige USA som case, fordi byerne inden for dette område har nogle af de mest levende og velunderstøttede, urbane landbrugssamfund i det globale nord. Denne rapport består af seks kapitler, der hver især sonderer urbant landbrug og derved

bidrager til vores nuværende forståelse af byers fødevaresystemer.

Kapitel 1 klarlægger den historiske udvikling af den fysiske og psykiske kløft mellem byboer og gård. Denne kløft slører for byboerne den økologiske forringelse, som er et resultat af deres ernæringsmæssige behov og som er blevet en udfordring på globalt plan med hensyn til udledning af drivhusgasser, arealanvendelse, vandforbrug og udledning af giftige kemiske stoffer. De seneste år er disse udfordringer blevet mere generelt erkendt, og mange kommunale beslutningstagere, urbane designere og borgere har sluttet sig til kampen for den lokale fødevareproduktion, da denne produktionsform opfattes som et middel til "grønne" urbane fødevareforsyningsnetværk. I den efterfølgende tekst definerer jeg urbant landbrug som "den lokale fødevareproduktion vævet ind i bystrukturer og som udnytter eksisterende materiale og energistrømme" og den spiselige by som "spredning af byernes landbrug i hele byen, i det omfang spredningen resulterer i integration af landbrug som en del af byen, både hvad angår form og funktion". Jeg foreslår derefter, at der på nuværende tidspunkt er en udpræget mangel på viden, som kan understøtte den seneste fremme af landbrugspraksis i byer i det nordøstlige USA, specielt da tidligere forskning inden for urbant landbrug fokuserer på urban landbrugsproduktion under mildere himmelstrøg. Slutteligt bliver byen Boston, Massachusetts, introduceret i dette kapitel som en case-by på grund af byens tæthed og klima, som begge regnes for forholdsvis repræsentative for mange af de byer i den nordøstlige region.

Kapitel 2 udfylder et vigtigt hul i litteraturen omkring bæredygtigheden af byer: manglen på omfattende gennemgang af miljøbelastningen fra byers fødevareforbrug. Hertil benytter jeg eksisterende litteratur inden for industriel økologi (studiet af materiale-, energi- og miljø-performance af systemer) for at illustrere betydningen af byernes fødevareefterspørgsel som en drivkraft for en bys samlede miljø-performance. Min litteraturgennemgang dækker relevante vurderinger af byer, der benytter massestrømsanalyser (dvs. sammenstillinger af massestrømme gennem en by), carbon footprints (sammenstillinger af drivhusgasudledninger som resultat af byens aktiviteter) og økologiske fodaftryk (dvs. arealet af jord udnyttet inden- og udenfor byen for at imødekomme byens funktioner og aktiviteter). Efter en gennemgang af 132 byer når jeg frem til, at det miljømæssige fodaftryk fra byernes fødevareforbrug (herefter benævnt "foodprint" eller på dansk "fødevarefodaftryk") ofte er på samme niveau eller større end mange andre aktiviteter, der ofte opfattes som mere direkte koblet til byers miljø-performance, såsom transport, opvarmning af bygninger og vandforbrug. Desuden er der en tendens til, at disse fødevarerelaterede påvirkninger øges i takt med byers rigdom. Dette indikerer større miljøpåvirkninger fra fødevarerelaterede aktiviteter i fremtiden som resultat af, at verdens bybefolkning fortsætter med at vokse i antal og rigdom. Af konkret bekymring er den robuste korrelation mellem fødevarespild i byer og rigdom, både fordi næringsstoffer i affald ofte ikke inddrives, og på grund af de miljømæssige byrder relateret til at producere fødevarer, der ikke ender med at ernære den endelige forbruger. Slutteligt indikerer min gennemgang af litteraturen omkring bæredygtighed i byerne, at selv om det urbane foodprint har opnået lav prioritet på dagsordenen for mange byer med fokus på bæredygtighed, har der over de sidste par år udkrystalliseret sig byer, som helt klart prioriterer sammenhængen mellem fødevareforsyningsnetværk og byernes miljø-performance, og som præsenterer specifikke tiltag til at afbøde yderligere forringelse af byers miljø-performance ved hjælp af lokal fødevareproduktion.

Kapitel 3 undersøger miljø-performance af urbane landbrug i Boston. Indledningsvist præsenteres en gennemgang af eksisterende litteratur, der har behandlet de miljømæssige dimensioner af urban fødevareproduktion. Jeg finder ved denne litteraturgennemgang, at på trods af sporadiske beviser for urbant landbrugs overlegne miljø-performance relativt til konventionel landbrugsproduktion i milde klimaer, er konkrete beviser for urbant landbrugs overlegne miljø-performance mangelfulde. Især én antagelse om "at lokalt producerede fødevarer i sagens natur er bæredygtige" fremstår som et fælles tema i meget af den gennemgåede litteratur. Derudover er der en række påstande om positive interaktioner mellem byernes landbrug og det omgivende miljø (regnvandsopsamling, genbrug af organisk affald, reduktion af byggeriets energiforbrug), som primært er funderet i sund fornuft, men som mangler empirisk fundament. For at kompensere for de manglende data relaterende til urbant landbrugs miljø-performance og struktur herpå, har jeg udviklet en grundlæggende taksonomi bestående af fire forskellige urbane landbrugstyper, der er baseret på landbrugets fysiske placering i det urbane system (dvs. placering på jorden eller tagkonstruktion) og vækstbetingelserne (konditioneret og ikke-konditioneret). Jeg bruger derefter primære data fra seks landbrug i Boston og New York til at bygge modeller, der kvantitativt kan sammenligne miljø-performance for urbane og konventionelle landbrugsproduktioner. Jeg finder, at de miljømæssige besparelser, der opnås ved at reducere transportafstand fra jord til byen let kan blive modvirket af energitilførsel til f.eks. konditionerede urbane landbrugstyper. Simple friluftslandbrug i byerne kan have lavere bidrag til den globale opvarmning end konventionelle landbrugsformer, dog på bekostning af arealanvendelse og vandforbrug. Selv når urbant landbrug giver miljømæssige fordele, blegner disse i sammenligning med anvendelsen af byrum til elproduktion baseret på f.eks. solceller. Kapitel 4 skalerer resultaterne fra de enkelte landbrug op til byniveau med henblik på at kvantificere virkningerne af den spiselige by på Bostons baseline foodprint. Jeg kombinerer folketællingsdata relaterende til Bostons demografi, ernæringsmæssige undersøgelser og EXIOBASE 2.2 modellen for miljøbelastningen fra produktion (herunder international handel) til at bygge det miljømæssige foodprint for Bostons 560 kvarterer i form af arealanvendelse og udledningen af drivhusgasser. Den gennemsnitlige beboer i Boston har en årlig miljøpåvirkning udelukkende relaterende til deres ernæringsbehov, der svarer til udledningen af 1,2 tons kuldioxid og 1 hektar arealanvendelse. Såfremt man skifter til urbant landbrug for at kompensere for disse påvirkninger resulterer dette i undseelige reduktioner i drivhusgasemissionerne (2,5%), mens man, hvad arealanvendelsen for fødevareproduktionen angår, vil forøge denne med 1%. Interaktionerne med bymiljøet er også overskuelige med et potentiale til at afbøde 2% af byens overfladeafstrømning og absorbere mindre end 10% af husholdningernes organiske, faste affald. Mere væsentlig er det potentielle kostbidrag som urbant landbrug kunne give i form af en betydelig procentdel af byens vegetabilske fødevarebehov. Markedsværdien af landbrug i Boston kan være så høj som $ 160.000.000, hvilket vil resultere i indtægter i nogle af byens fattigste områder. Kapitlet slutter med en afvejning af de marginale miljømæssige gevinster i byernes landbrug mod potentielt større sociale og ernæringsmæssige fordele.

Kapitel 5 fokuserer på andre tilgange til en spiselig by, som kan reducere det urbane foodprint. Konkret fokuserer jeg på det faktum, at hovedfraktionen af urbane foodprint hidrører produkter med animalsk oprindelse, og at dette faktum formentlig ikke ændres ved en overgang til urbant landbrug. Jeg konstruerer derfor miljømæssige modeller af den gennemsnitlige amerikanske kost, en vegetarisk og en vegansk kost for at se, hvordan disse livsstilsændringer potentielt kan påvirke Bostons foodprint. Skift til vegetarisk og vegansk kost resulterer i miljømæssige fordele, der er betydeligt større, end hvad kan opnås med urbant landbrug. For eksempel vil et skifte til vegetarisk og vegansk kost reducere byens carbon foodprint med hhv. 32% og 67%. Da det ikke er realistisk at forvente, at hele byen vil tilpasse sig en kødfri ernæringsstrategi, ser jeg også på foodprint reduktionspotentialet ved introduktion af en ny plantebaseret køderstatning (den såkaldte ”Impossible Burger” fra Impossible Food). Selv ved en udskiftning på blot 10% af hakket oksekød med et plantebaseret produkt, som ligner og smager som hakket oksekød, vil dette på årsbasis medføre påvirkningsreduktioner svarende til fordelene ved en fuldt spiselig by.

Kapitel 6 afslutter afhandlingen med en syntese af resultaterne og en kort diskussion af potentielle fremtidige forskningsretninger i form af kvantificering af foodprints samt bedre kvantificering af de miljømæssige dimensioner for urban fødevareproduktion. Jeg afslutter denne rapport med nogle tanker om betydningen af at anvende kvantitative bæredygtighedsmodeller til at imødegå dårlige urbane designløsninger samt betydningen af byernes agrikulturelle, sociale ydelser, der i sidste ende kan være den mest berettigede grund til det urbane landbrugs yderligere spredning i det amerikanske nordøst.

Contents

Abbreviations

BI - building-integrated
C - conditioned
CC - climate change
CF - carbon footprint
CO_2e - carbon dioxide equivalents
EF - ecological footprint
GB - ground-based
GHG - greenhouse gas
gha - global hectares
GMP - gross metropolitan product
GWP - global warming potential
HVAC - heating, ventilation and air-conditioning
ICLEI - International Council for Local Environmental Initiatives
ILCD - International Reference on Life Cycle Data
IO-LCA - input-ouput life cycle assessment
IPCC - Intergovernmental Panel on Climate Change
kcal - kilo calories
LAFA - loss adjusted food availability
LCA - life cycle assessment
LU - land use
MFA - material flow analysis
MRIO - multi-region-input-output
NC - non-conditioned
NHANES - National Health and Nutrition Examination Survey
OECD - Organization of Economic Cooperation and Development
PBB - plant based burger
UA - urban agriculture
UM - urban metabolism
US - United States
USDA - United States Department of Agriculture
WRD - water resource depletion

Chapter 1

Introduction and motivations

1.1 From forager to farmer to "foodie"

It is oft repeated that in 2008 human civilization passed an important milestone: for the first time in history more than half of humanity lived in what could be described as cities [57]. It has taken a long time to get to what has been gilded by some scholars 'the urban century'[23].

Most of humanity's ≈100,000 years have been a placeless, nomadic enterprise of subsistence living supported by hunting wild animals and foraging wild plants [30]. Man was limited to what nature's bosom provided, and was by default, forced to live within the local environments ability to replenish and supply resources, its 'carrying capacity'. The material and energy appropriations from the environment needed to maintain these nascent societies - their "social metabolism" - was limited to immediate survival, with little long-term storage [14]. The guesstimated 4 million foragers roamed the planet; the vicissitudes of nature precluding permanent settlements [40].

The first agricultural (r)evolution in the 'fertile crescent'of the Nile Delta around 10,000 BC boded the end of the forager. By taming plant and animal, humans had reliable food sources for the the first time reliable. Instead of foraging and hunting on the margins of survival, humans had more than they needed. Much more, in fact: per capita social metabolism grew tenfold [14]. Stable food supplies supported the first permanent settlements and over a 6,000 year period, the majority of humans became farmers while the foraging way of life withered [40]. This newfound stability and affluence allowed portions of the population to expand their activities beyond those directly related to bare subsistence, undergirding the development of increasingly complex social hierarchies, divisions of labor, art and technological innovations. It was around this time that the first cities began to appear in Asia Minor [36, 43]. The earth's population began to grow modestly (≈24,000/year) [40], which combined with the larger social metabolism to make man's environmental footprint felt in marginal ways - clearing vegetation, salting soils and extinguishing fauna. Notwithstanding, agrarian societies were primarily limited to energy from the Earth's solar budget, and hence, functioned within the planet's carrying capacity.

For thousands of years, cities housed the minority of humanity not involved in extractive work. This began to change in the latter half of the 18th century with the liberation of markets and new energy sources. The invention of the combustion engine in 1790 allowed humans to efficiently convert hitherto unavailable solar energy stored in fossil fuels to motion [48]. Combined with the disruptive forces of capitalism, fossil fuels led to a shift in the economy from countryside to city, as factories sprung up to make new goods for a diversifying economy. People followed suit. The process of urbanization took hold as people abandoned the country for the city in search of employment. Urbanization was (and is) also propelled by a second agricultural revolution, whereby advances in soil chemistry, consolidation of small landholdings and farming mechanization made large fractions of the rural workforce redundant [33, 12, 45]. Harvesting stored solar energy provided a tenfold increase in per capita energy use compared to previous agrarian lifestyles, and a new social metabolic regime emerged: the industrial societies [14]. It was the intertwined processes of urbanization and industrialization that buttressed the emergence of the urban epoch we inhabit today.

Our relationship with our food supplies have altered with every social metabolic bifurcation. The foragers were connected to their local ecosystems as they required intimate knowledge of plant species and seasonal shifts to follow edible plants and track prey. Farmers in agrarian societies were also closely tied to their food, as they farmed the soil under their feet for survival. Only with the advent of a predominantly urban society did the connection between producer and consumer become severed. Food became an abstraction, produced 'out there' in the hinterlands and imported to the city. The end result is that most humans living in the wealthier societies where urbanization first took hold are, in a *very* loose sense of the term, "foodies" - voracious, hedonistic consumers of food - lacking the faintest knowledge of their food's origins.

1.2 The "metabolic rift"

In his writings Karl Marx coined a term that aptly describes the transition from forager to urban foodie: *the metabolic rift* [33, 16]. Although the metabolic rift can be read from numerous angles (see McClintock [33]) its essence is the dislocation of people from the natural resources that support their daily sustenance and provide inputs to industry, as they are forced by capitalism from working the land to working in factories in cities [37, 33]. An outcome of this is alienation of the urbanite from the natural world and imagining of the self and urban society in contrast to nature [33].

Marx also pointed out the ecological impacts of the rift, using nutrient flows as an example to show how the cleavage of production and consumption areas led to resource degradation at the former and pollution at the latter [33]. Marx was acutely aware of the problem of nutrient stripping of agricultural soils, noting that "For a century and a half...England has indirectly exported the soil of Ireland" [16]. Marx was also perturbed by the lack of recycling of these resources, lamenting that "In London...they can do nothing better with the excrement produced by 4.5 million people than pollute the Thames with it" [16].

Today about one third of the world's labor force is involved in agriculture

[45], and hence, the remaining two thirds live in some state of cleavage from the planet that sustains them. Another outcome of the rift and the industrialization of agriculture has been the loss of farming as a communal body of knowledge passed down through generations, and the evaporation of farming and stewarding the land as a culture (hence 'agri' + 'culture') [42, 29].

1.3 Industrialization and the environment

So what is the metabolic rift shielding urbanites from? For one, the extractive regions where non-renewable mineral and fossil fuel resources are mined. With the concomitant drop in mortality rates and growth in food supplies, humanity has ballooned from one to seven billion since the cusp of the industrial revolution [40]. Supporting this population requires increasing amounts of these resources, the extraction of which exacts large-scale scarring of the landscape and intrusions into natural habitats that are hidden by global trade networks [46, 12]. Importantly, the fossil-fuel combustion buttressing the industrial age releases atmospheric warming gases that are now understood to risk the stability of geochemical cycles. The affects of the warming atmosphere are currently being experienced most acutely by those living on the margins in rural societies, though predicted levels of sea rise will make this a concern for coastal cities in the future [25]. In short, the resource demands and pollution of a largely urban, industrial society are compromising the planet's long term ability to support our current numbers at present living standards [52]

Agriculture might be the driver of global environmental change most obscured from city-dwellers. The contribution of agriculture and deforestation to climate change is estimated at 25% of total anthropogenic emissions [26]. The vast majority of human appropriated land is used for agriculture, amounting to 12% and 26% of ice-free land for crops and grazing, respectively [15]. Water use is also overwhelmingly driven by the needs of food production, accounting for over 90% of anthropogenic withdrawals [24]. The nutrient stripping and pollution deplored by Marx is now at such a scale that it has severely altered bio-geochemical cycles [52] and contributed to the deterioration of water quality in many watersheds and river basins [53, 13, 22]. Outside of the city, the link between current modes of food production and environmental degradation are conspicuous, while urbanites remain insulated due to geographic separation between city and supply region [47].

1.4 Food and the city

The rift has not always been so pronounced between city, food production and environmental burdens. Providing a stable food supply was typically under the aegis of the local government in many cities into modern times [27, 33]. Up until the turn of the 20th century food production was substantial in many of today's capitals of cosmopolitan urban living. Paris was once renowned for its copious green-vegetable production using a system predicated on the recycling of 'night soil' (sewage), even exporting across The Channel to England [5, 3]. Boston Common, a verdant park on the outskirts of Boston's downtown, was exactly what its namesake evokes, a shared space where residents of the incipient city could send their cows to pasture as late as the mid 19th century[31]. Many cities in the Global South still maintain such ties, with active urban farming and nutrient recycling schemes [39].

In the United States (US) Northeast, farming within the city has waxed and waned with economic necessity or times of war. 40% of food in the US was produced by 20 million urban "victory gardens" in 1944. Each economic downturn has seen an uptake in urban food production as urbanites stave off food insecurity, the latest round being the "great recession" of 2008 which saw Michelle Obama planting vegetables in the White House garden [33]. Despite these scattered 'green thumb moments', food production was largely driven from the regions cities in the previous century by modern urban planning practices focused on the strict separation of land uses and sanitizing the city [33, 29]. Urban food production in the region is also usually expelled as more profitable land uses emerge [33].

1.5 Connecting city with consumption

An important concept in understanding urban-rural linkages is the concept of 'teleconnections' as articulated by Seto and colleagues as "the virtual shrinking of distances between places, strengthening connectivity between distant locations, and growing separation between places of consumption and production..."[46, p. 7687]. Through the application of this concept to urban systems, researchers hope to strengthen our understanding of how urban consumption regimes translate into deforestation, pollution and water scarcity in disparate supply regions, including impacts from urban food demands [47].

The food-water-energy nexus is another powerful concept framing the current thrust of urban-food systems research [61]. 'Nexus' refers to the property of inter-connectedness between seemingly weakly related or unrelated system components. For instance, modern sewage systems require large amounts of potable water, act as the primary nutrient sink (and a potential source) for agricultural systems, and entail energy use for processing [61].

Equally salient to these theoretical developments are concrete actions by city governments. Numerous cities in the Global North have been moving towards urban food policies aimed at promoting healthy, stable and sustainable diets [35]. Late 2015 saw the signing of the Milan Urban Food Policy Pact at the Universal Expo in Milan, an agreement between 132 cities (combined population > 460 million residents) recognizing the important role of municipal governments in shaping healthy eating patterns of residents and ensuring sustainable food supplies into the future [9]. This pact recommends actions in a number of realms:

- Ensuring and enabling environment for effective actions
- Sustainable diets and nutrition
- Social and economic equity
- Food supply and distribution
- Food waste
- Food production

The last point explicitly says that cities should "Promote and strengthen urban and peri-urban food production..." with a number of subsequent proposals related to this and the proliferation of "short food chains" to reduce the environmental burden of feeding cities [9]. Some cities have presaged this pact through their own local studies and ordinances supporting urban food production [8, 1, 33, 35].

This movement towards farming in cities dovetails nicely with recent urban design trends that wed farm and city such as 'biophilic cities'[55] and urban agriculture[49, 11]. Hypothetical design proposals are buttressed by real change on the ground through variegated urban farming schemes cropping up in many wealthy cities that shunned the practice a generation ago [34].

Combining these trends in academia, urban planning and design with the ever present work of ecologically concerned citizens on the ground, a narrative coalesces espousing the need for cities to repair the metabolic rift and mitigate their food related environmental impacts. A vital aspect of this is the use local production, chiefly, urban agriculture (UA). Of key importance is that this movement shares support amongst citizens and municipal governments, as opposed to many previous pushes for UA that saw antagonisms between farmers and local governments (excepting the extraneous situations in Section 1.4).

Figure 1.1: The different shapes and sizes of urban agriculture (Author's own images)

1.6 Defining urban agriculture and the edible city

UA is a term that has seen increasing use by urbanists and urbanites in the past decade, despite a lack of formal definition. At its core it is a practice carried out by city-dwellers, within cities, of producing food. Through a series of workshops Vejre and colleagues identified multiple definitions based on one's focus [60]:

- Spatial - based on location in space, in relation to urban area.
- Functional - based on activities performed, regardless of size, ownership, barring activities beyond the immediate hinterlands.
- Externalities - based on the basket of goods provided by the farm and their marketing towards urbanites.
- Market - division between farms operating for local urban and global markets.

Out of these framings I blend spatial and functional approaches in my definition of UA, since they align with the urban design and engineering orientation of this dissertation. More concretely, I borrow from Koc and colleague's groundbreaking UA research to define UA as '*the production of food in and adjacent to cities, leveraging pre-existing urban material and energy flows as farming inputs*'[28]. I also limit this definition to exclude farming beyond the urban boundary, as this is peri-UA from a strictly spatial perspective. In this definition I capture the essential characteristics relevant to this work: siting within the city and interactions with a host city's energy and mass flows. As will become clear over the course of this report, UA comes in a variety of forms, and in line with this, my applied definition is broad enough to encompasses them all (see Figure 1.1).

Others have envisioned the scaling up of UA in cities, where food production becomes a ubiquitous aspect of the urban built form, enmeshed within the very essence of a city's fabric. Such visions have been called *continuously productive urban landscapes* or *the edible city*; where food production becomes an integral part of a city, both in form and function [10, 6]. Hypothetical assessments have shown that the edible city could produce a significant portion of a city's plant-based food requirements [32, 38, 35] (see Figure 1.2). Much like UA, many urban designers and policy makers believe that the edible city will play an important role in a sustainable urban future [11, 49, 35].

1.6.1 Environmental dimensions of UA

UA and the edible city have emerged partially as responses to the metabolic rift and to the ecological challenges facing current food systems. Edible cities have been championed due to a number of perceived environmental benefits:

- Shorter transport distances and related greenhouse gas emissions.
- Improved production efficiencies compared to conventional farming.
- Reduced wastage and packaging.
- Positive interactions with the host city's material and energy flows.

The predominant discourse surrounding the environmental impacts of urban food production has been pro-UA. It has only been in the past few years that researchers have seriously started investigating the environmental implications of urban food production [34, 50, 7]. A reoccurring trope within UA literature is a focus on transport related impacts and the inherent belief by many UA champions that reducing 'food-miles' is the most important environmental challenge facing urban food supply networks, despite ample evidence to the contrary [7, 17]. Moreover, the singular focus on food-miles ignores the risk that urban farming might in fact provide disservices to the local environment [41].

Figure 1.2: Visions of the edible city in New York City (left)[49] and Boston (right)[2]

Reviews of UA literature have emphasized the need for increased research into the environmental dimensions of the edible city. Recent assessment have shown that in limited contexts, UA can produce food with lower embodied environmental burdens than conventional agriculture [21, 44]. Evidence at the scale of the edible city is scant, but an assessment of Lisbon, PT hinted at environmental benefits at this scale [4].

Although a step in the right direction, these nascent assessments of UA's sustainability are wanting. Firstly, there is the challenge of scale. Where primary data have been collected to assess UA's environmentally, they have only covered single farms, missing the opportunity to see what a scaled-up, edible city might look like. Moreover, these studies tend to focus on single types of UA out of the multitude that exist. When the edible city has been assessed, researchers used data from conventional farms as a proxy for UA, eschewing potentially significant differences between the production methods. Finally, although not a shortcoming of there previous work, there has been the focus on UA in sub-tropical or Mediterranean climes that are amenable to year-round production and low-energy demands for farming, limiting the applicability of the results to other climes.

1.7 Research objectives

The overarching aim of this project was to provide a clear picture of the environmental performance of urban food production beyond the topic's treatment to date. This included the use of primary data from multiple types of urban farms and the scaling up of that data to the city level to better understand the edible city from an environmental angle. My geographic scope was the US Northeast for reasons explained in Section 1.8.

In terms of explicit research questions, three related themes covering six questions were intended to be tackled at the project outset:

1. Environmental impacts of urban food consumption.
   (a) How large are the environmental impacts of urban food consumption?
   (b) What level of coverage is given to food related impacts by city governments?
2. Urban agriculture.
   (a) What is the current understanding of the environmental performance of UA?
   (b) What types of UA exist and can they be categorized based on material and energy regimes?

(c) What is the comparative environmental performance of urban and conventional agriculture supplying food to the US Northeast?

3. To what extent would an edible city change its baseline food related environmental burdens?

An additional research question arose out of my investigation of research question 3:

4. What can be done to compliment the edible cities in the Northeast US to further reduce their food related environmental pressures?

1.8 Why the American Northeast?

One of the first places to experience the metabolic rift was the American Northeast, particularly the Northeast Megalopolis along the coast [16, 33]. The region was a first mover in changing to a manufacturing economy. Westward colonization revealed incredibly fertile soils that combined with an expanding railroad network to bring the markets of the large cities of the northeast within range of Midwest farmers [29]. As a result, agriculture largely died out as a way of life for many in the Northeast, and the population became predominantly urban by the end of the 1800s [54, 29]. The divorce of city and supply-region has only become increasingly pronounced. There now exist whole neighborhoods in the region's cities that lack easy access to many of the fresh fruits and vegetables that were once grown on the same soil where those cities now stand [35, 20]. Marx's observations surrounding nutrients remain prescient, with many cities in the US Northeast exhausting the bulk of their imported nutrients to landfills and sewage treatment plants with little recovery and recycling [18].

The region is also an interesting case due to its enthusiasm for urban food production. This enthusiasm translates into one of the most diverse and dynamic UA scenes in the Global North, where urban farming innovations are born and advanced UA techniques tested. The region's cities have taken an active hand in promoting UA [8, 1, 19] and are therefore well suited to benefit from their work here, hopefully providing a practical audience for this study. The climate within the region, though varied, supports similar types of crops (see Figure 1.3) and exhibits pronounced warm summers and cold winters. As a result, operating conditions and inputs for farms *roughly* are comparable throughout the region's cities, expanding the applicability of this report beyond a single city.

Lastly, the per capita food-borne environmental impacts of the United States are amongst the largest on the planet [56, 51]. This makes the region's cities well suited to test the efficacy of UA as a green design intervention.

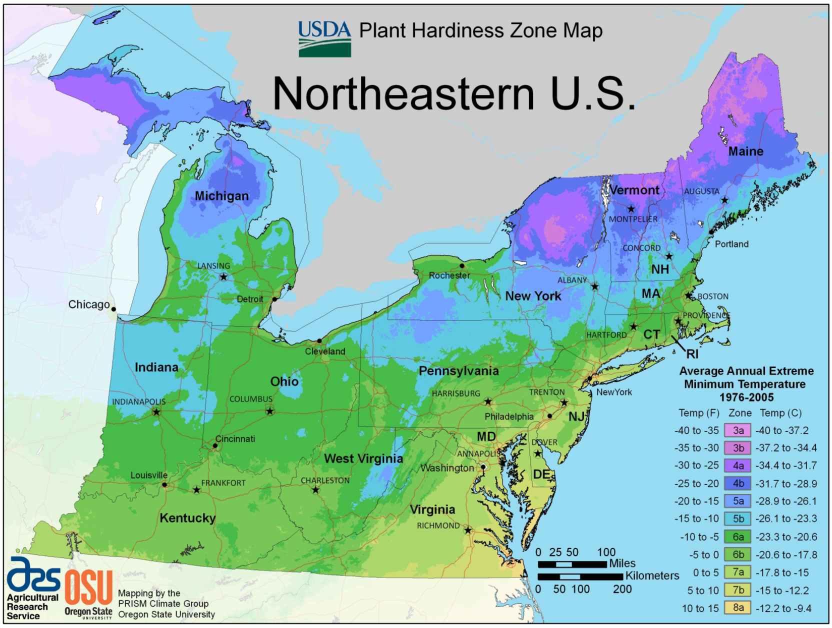

Figure 1.3: USDA Northeast Region and plant hardiness map. Source: USDA [59]

1.8.1 Boston as a case city

I used Boston, Massachusetts as a case city for this study. I chose Boston because of the municipal governments active support for UA, as enshrined in recently passed ordinances [8]. This city of roughly 630,000 (≈6,000,000 in the metropolitan region) [58] also typifies the built-form of many cities of the region in that it has a dense core and reducing population density as one traverses the urban transect [31]. Lastly, it sits near the middle of the United States Department of Agriculture (USDA) plant hardiness zone map for the region (see Figure 1.3), providing an average climate for the assessed geographic scope, particularly the Northeast Megalopolis.

1.9 Dissertation outline

The remainder of this dissertation is comprised of a five chapters, each supported by a published article or submitted manuscript, the exception being the final summary chapter. **Chapter 2** focuses on research theme one: the scale of environmental impacts of urban food demands and food on the urban sustainability agenda, supported by article 1. **Chapter 3** focuses on research questions two: classifying different UA types and quantifying their effectiveness as an alternative food supply, supported by articles 2 and 3. **Chapter 4** focuses on research question three: the environmental performance of the edible city, supported by article 4. **Chapter 5** explores question 4, looking at compliments to the edible city, supported by articles 5 and 6. **Chapter 6** synthesizes the findings and proposes future research needs.

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Chapter 2

Quantifying and addressing the urban ‘foodprint’

2.1 Chapter overview

This chapter is a response to the two questions under the first research theme: What is the scale of the environmental burdens from urban food consumption? What priority do municipal governments place on food related impacts? I start with a definition of the 'food-print' and proceed into the research questions in sequence. In this section I expand beyond my stated geographic scope, as my goal in this chapter is to understand the general discourse surrounding food consumption and urban environmental sustainability. Limiting this section to the Northeast US would also quickly exhaust the stock of relevant studies, handicapping my review. Notwithstanding, I conclude this chapter by placing the findings within the project's geographic context. This chapter is a companion to Article 1 which can be found in Appendix A of this report.

2.2 Quantifying the urban 'foodprint'

2.2.1 Defining the urban 'foodprint'

The "food-print" (here 'foodprint') was initially coined by researchers investigating nutrient flows related to Paris's food intake [7, 11]. It is a portmanteau of 'food'and 'footprint'; encompassing the notion of food related environmental pressures. Here I define the 'foodprint' as *the environmental burdens related to the provision of food for a city's residents and guests.*

2.2.2 Quantifying the urban foodprint with 'industrial ecology'

In quantifying the urban foodprint I pull from the field of industrial ecology. Industrial ecology is multidisciplinary field of study that can be defined in a rather open-ended manner as "a metaphor for looking at our civilization" [53, p. 3] that helps us align our long-term modes of resource use, production and consumption with the planet's capacity to support our civilization given continued economic, cultural and technological evolution [27].

Industrial ecology is unique in that it is normative, endeavoring to provide tools that help industry benchmark and reduce pollution from production and transition to circular material regimes that more efficiently use available resources (akin to nested material and energy networks observed in natural systems, hence the ecological metaphor). The toolbox of industrial ecology is primarily quantitative in nature, including varied methods such as material flow analysis, life cycle assessment and exergy analysis. The field spills over into other disciplines, employing pieces of network theory, graph theory, systems theory, transitions theory, ecology and economics to model products, consumers, industries, territories and other systems in terms of their form, function and environmental performance [1].

2.2.3 'Urban metabolism' as a metaphor

One popular focus of industrial ecology research is cities, as they are humanity's hotspots of consumption and pollution generation [61, 28]. Most of this work has fallen under the umbrella term 'urban metabolism' (herein 'UM'), itself a metaphor loaded with ontologies (fundamental truths and properties) based on the term's user [40, 9]. In speaking of UM, industrial ecologists evoke the flow and accumulation of materials and energy in cities and the application of the field's tools for quantitative analysis. Urban ecologists apply the term to mean the interconnections and feedback mechanisms between different sub-systems that govern material and energy fluxes in a city, moving beyond the black-box models employed by industrial ecologists. Lastly, political ecologists look at urban space as a socio-natural hybrid, concerned with the social and political processes that lead to the creation and ossification of the 'created landscape'that is the city, with a sub-set of this researchers focused on Marx's metabolic rift [40, 2].

In my study I took the industrial ecologist's view of UM, as my project concerned with the scale of the urban foodprint, not the inter-workings of Boston's food system nor identifying the actors and processes that construct and reinforce these systems. More concretely, I used Kennedy's definition of UM as "the sum total of technical and socioeconomic processes that occur in cities, resulting in growth, production of energy, and elimination of waste" [36, p. 44].

UM is a metaphor for a phenomenon (the material and energy regimes of cities) and not itself method, as it is occasionally misconstrued. Numerous methods have been applied to quantify this phenomenon [61], and I focused on the three most prolific to date: material flow analysis (MFA), carbon footprint (CF) and ecological footprint (EF). These methods also have individual strengths, compensating for the others weaknesses to provide a somewhat balanced analysis of the urban foodprint. MFA describes the form of the foodprint in terms of pathways and fates of food and food waste.CF covers the foodprint's contribution to climate change and EF looks at the foodprints land occupation. A fourth popular method is eMergy, but I chose to exclude it as it is poorly suited to account for environmental impacts from solar derived resources, and by consequence food.

Material flow analysis

MFA is built around the 1st law of thermodynamics, in that at its core it is a general mass balance of a material (e.g. steel) or substance (or element) (e.g. iron) through industrial or socioeconomic process [3]. It is formalized through a number of metrics that describe the material handling regime of a bounded system: *domestic extraction* (materials brought under human control within the system), *imports/exports* (materials crossing the system's boundary), *net additions to stock* (balancing terms of mass balance) and *domestic material consumption* (apparent consumption as domestic extraction plus imports minus exports) amongst others [3].

When following strict MFA methodology, studies of UM tend to focus on calculating domestic material consumption. Occasionally, total material requirements are estimated, which is domestic material consumption plus the 'hidden flows' that arise during raw material extraction (e.g. overburden removed during mining). Sometimes MFA based UM accounts study a single substance such as a nutrient [35, 19] or metal [3], in which case one is more likely to see the use of strict mass-balance principles, including chemical stoichiometric equalities.

In reality, researchers rarely follow strict mass balance methodologies (usually due to data and computational challenges), and are more apt to perform a straightforward accounting of material consumption in terms of mass per annum as imported into the system, occasionally normalized to city population. The list of materials studied typically includes food, water, construction aggregates, fossil fuels, plastics, metals and wood [17, 61]. Electricity consumption is also commonly included [17]. These studies tend to either employ bottom up methods (scaling up individual activities to the city level, such as housing construction starts times the amount of concrete per house) [49] or trade data to balance material flows across a political boundary [48].

One strength of MFA is its lack of abstraction and clarity in describing fates of materials manipulated by human. Framing this in the context of my study, this included the disposal pathways of nutrients from food imported into the city, identifying where to best collect these resources for reuse. Conversely, MFA is at a disadvantage to quantify the environmental burdens embodied within imported goods or from pollution within a city [13].

Carbon footprint

CF is a means to estimate the emissions of climate change inducing greenhouse gases (GHGs) that result from the delivery of a product or service [33]. CF methodology has three scopes of assessment: scope 1 (direct emissions from fossil fuel combustion by the study system), scope 2 (indirect emissions from electricity consumption by the study system) and scope 3 (emissions embodied within the good imported into the city) [33]. To provide a clear metric, CF converts different GHGs from their mass to the mass of carbon dioxide that induces an equivalent radiative forcing, expressed as carbon dioxide equivalents (CO_2e)[33].

The dominant methods for calculating CF are bottom-up (or process based) and top-down. Bottom-up looks at the individual activities required to provide a good or service and multiplies the amount of activity (e.g. kilometers driven by a specific truck) by a carbon intensity for that activity (e.g. CO_2e/km driven), summing the emissions from all activities to provide a life-cycle accounting of embodied GHG emissions [33]. When investigating UM, this typically means determining apparent masses consumed using MFA and applying GHG intensities for those goods, calculating transport related GHG combustion in the city and combining electricity consumption with carbon intensities for the underlying grid [45, 12]. Top-down CFs rely on economic supply-use tables to capture interdependencies between economic sectors, which when combined with total sectoral estimates of GHG emissions can be permutated to estimate embodied GHG emissions per unit economic output [34]. The final economic demand of the residents is then used to ascribe the GHGs of production to the city [34]. Occasionally both methods are applied in a hybrid manner [46].

A strength of CF methodology is that it moves beyond mass accounting to communicate a city's potential contributions in terms of changes to the state of the environment. At the same time, the focus on a single metric introduces the risk of promoting solutions that lead to unforeseen, deleterious impacts in other domains (so called 'burden shifting') [29].

Ecological footprint

EF methodology, although not explicitly a product of industrial ecology (it was developed by landscape architects), approximates resource draws to support consumption activities, and has seen wide application in the field. EF studies provide a weighted measures of land occupation. Weighting is based on the ability of a piece of land to convert solar energy to biomass ('primary production'), with areas of high productivity (e.g. rainforests) weighted higher than those of low production (e.g. arctic tundra). Different land types are converted into "global average hectares (gha)" based on their ratio of production over a year relative to a piece of land producing biomass at the global average (total global annual primary productivity divided by total land area). The resulting metric estimates the share of the Earth's annual capacity for producing renewable resources and absorbing pollution appropriated by an activity [8, 23].

In the same manner this method can be scaled to a city using both the bottom-up and top-down methods. The bottom-up methodology uses a bundle of goods consumed at the city scale and individual EF intensities for those activities to estimate the gross impacts from a population [38]. The top-down method combines national economic accounts and land use data to estimate EF, using expenditures to ascribe the area of land appropriated to the final consumer [6].

The strength of the EF method is that by relating an activity's impact to a constrained resource, land, a measure of carrying capacity appropriation is provided. If humanity's activities occupy more land than is available over the long term, say by harvesting more timber in a year than can be grown in the next year, then the planet's carrying capacity has been short-circuited [8, 23]. EF methodology is powerful in its communicative capacity, but its validity has been questioned. Firstly, it is argued that EF models do not actually account for the Earth's innate ability to regenerate natural resources, but are more representative of man's ability to boost biomass production using other non-sustainable means, chiefly mineral fertilizers [8, 23]. Second, EF methodology only includes carbon dioxide as a waste stream to be remedied [8, 23]. Notwithstanding these challenges, I included EF studies here since they link urban food consumption to the hinterlands and embodied resource draws.

2.2.4 Foodprint analysis

In my review I looked at three different metrics:

- The rank of a city's foodprint relative to other important metabolic drivers (transport, construction materials, energy, water provision).
- The foodprint as a percentage of total impacts for a given metric.
- The foodprint as a function of per capita gross metropolitan product (GMP), to look for patterns between wealth and foodprint scale.

Foodprints included

In populating a list of candidate studies for this review I used online literature repositories including but not limited to DTU's own library system, ISI Web of Science, Google Scholar and Scopus. My target studies were any MFA, CF or EF studies of UM. Searches were performed throughout 2014 using relevant search terms such as 'urban metabolism', 'urban carbon footprint', 'urban ecological footprint' and the like. Using this method I found 206 initial studies. By limiting my review to those that explicitly accounted for food in a transparent manner and excluding historical reconstructions, relevant studies were reduced to 43. Accounting for studies that covered multiple years of the same city and/or multiple cities, a total of 132 foodprints were reviewed. Figure 2.1 maps the included cities.

2.2.5 Material flow analysis findings

Twenty-five MFA foodprints were analyzed. To compare across assessments, all results were converted to tons of annual, apparent per capita food consumption (tons/cap/a). Relative to other primary UM drivers (transport, building energy, etc.), food was most commonly the third largest, accounting for 10-20% of material consumption (see Figures 2a-b in Paper 1), often overshadowed by fossil fuels and construction materials. Exceptions were cities with extremely static built forms (e.g. Paris), where food was more conspicuous.

When wealth was considered, the relationship between per capita GMP and food was moderate (R^2=0.34), with a slight upward trend as shown in Figure 2.2a. This could very well be a result of the self-limiting nature of food demand, since unlike other goods, a person can only consume so much food. Once nutritional demands are satiated, food expenditures appeared to start flattening. Evidence of this can be seen in US nutritional surveys that have found little difference in the total amounts of food consumed between low and high income Americans [39]. This is the basis behind Frederich Engel's law: above a certain income threshold, the share of household expenditures on food decreases [14]. Notwithstanding, a statistical difference at the 95% confidence level (p=0.0272) was found

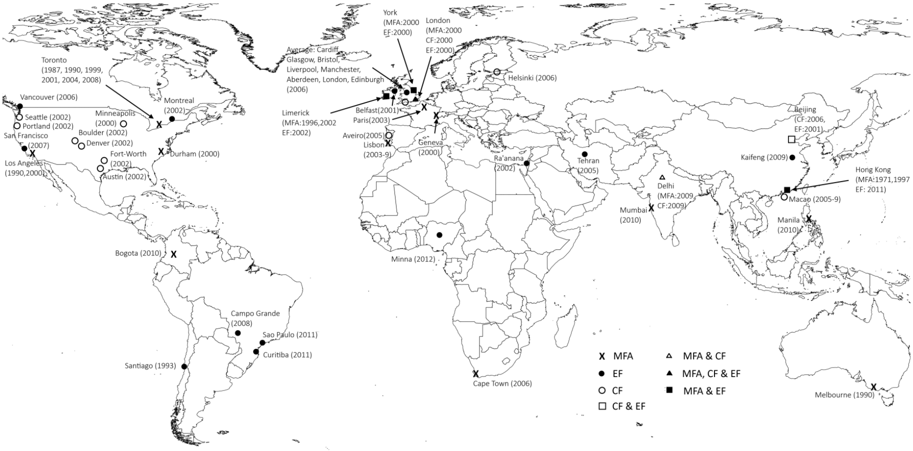

Figure 2.1: Cities included with the year of analysis and methodology.

between the mean foodprint of cities from Organization of Cooperation and Economic Development (OECD) member countries (0.93±0.33 ton/cap/a) and non-member countries (0.60±0.15 tons/cap/a) (boxplots in Figure 2.2b).

Even though the past 20 years of global development has shifted wealth generation away from the OECD countries and the distinction between non- and member countries is shrinking [18], remaining gaps between these groups hint at the scale of increase of food demands if the aspirational consumers in non-member countries start earning near OECD levels. These findings align with other estimates of increasing per capita global food demands in the coming decades if economic development continues apace [21]. Moreover, the mean of all included foodprints was greater than the global average, hinting at potential links between the urbanization process, increased purchasing power and growing food demand [51].

Some of the MFA studies tracked the fate of food flows demanded by their respective case cities, providing a glimpse into the foodprint form. Earlier work revealed predominantly linear foodprints, typified by food importing and waste exporting, with little recovery and recycling of waste in most cities. Some cities appeared to be wasting over 40% of their food stock[22], with wastage rates between 20% and 30% the norm (see Article 1 for further details). Figure 5 of Article 1 plots food waste and per capita GMP, displaying a robust relationship between the variables (R^2=0.57). This is an unsettling pattern because food waste in the upper income countries is largely due to over-production and unnecessary purchasing, combined with refrigerated storage capacity [30], as opposed to unreliable supply chains in poorer countries [20]. In terms of percentages, both high and low income countries waste approximately 1/3 of edible food, the difference being where along the lifecycle the food spoils [20].

One could argue that it is preferable that food waste is generated in densely populated cities, as this eases collection and further processing, but this overlooks a couple of key points. First, collecting and re-processing the food waste can only recoup a portion of the resources used during production and does not redress the pollution emitted on the farm and during processing [3, 24]. Second, shifts towards meaningful circular urban metabolic profiles should focus on the urban sewage streams that contain the bulk of nutrients exit-

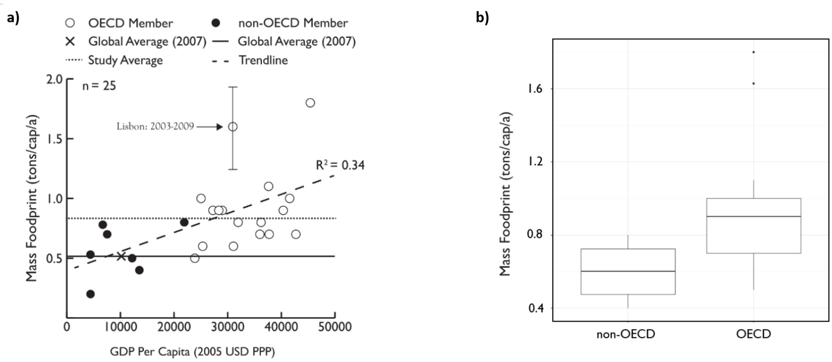

Figure 2.2: (a) MFA 'foodprints'versus per capita income in cities. (b) Boxplots of per capita food consumption in different cities based on OECD affiliation

ing the city [22, 35].

2.2.6 Carbon foodprint findings

Fifteen carbon footprints were assessed. In a similar fashion to the MFA studies, CF was typically the 3rd largest driver of GHG emissions from the cities, contributing 10-20% of city-wide impacts (see Figure 2 in Article 1 for further details). Article 1 plots GMP against CF in full detail, including cities in non- and OECD member countries. Figure 2.3 displays a refined version of the original GMP against CF plot, dropping the OECD dichotomy (too few non-OECD cities) and removing the outlier Macao (see Article 1 for reasoning). There was a robust correlation between CF foodprint and wealth (R^2=0.65). Studies comparing food-borne CFs between nations find comparable patterns, a result of shifting towards GHG intensive animal-sourced foods with increasing income [56, 55]. This is not surprising, as it has long been observed that one of the first lifestyle shifts to occur when one moves out of abject poverty is a increased animal proteins consumption [43].

The CF impacts of this shift cannot be overstated. For instance the GHG emissions imparted in delivery 1 kg of protein in the form of beef range between 60-640 kg CO_2e compared to under 20 kg CO_2e for vegetal proteins [41]. Although other livestock products are less GHG intensive, it is beef that has seen the largest uptake in the developing economies in recent decades [56].

The mean CF foodprint for all included studies was well above the global average in 2007 (2.0±1.1 vs 0.75 metric tons CO_2e/cap/a). Maintaining or reducing global agricultural GHG emissions will be a challenge if this is any indication of the scale of future foodprints of currently urbanizing societies. Shifting from vegetal- to animal-sourced foods is not in itself environmentally unsustainable; switching to lower GHG intensity meats (e.g chicken) can actually improve dietary quality while avoiding profligate GHG emissions. It is the trend of increased beef intake in particular that is challenging due to methane emissions from enteric fermentation and deforestation for feed crops and grazing [32].

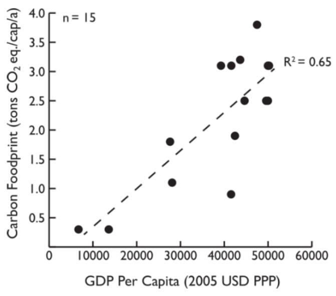

Figure 2.3: CF in tons CO_2e/cap/a plotted against per capita GMP

2.2.7 Ecological foodprint findings

Twenty-one EF foodprints were reviewed. Food was the largest driver of EF impacts in over 60% of the cities assessed, where it contributed more than 40% of the total urban EF for more than 40% of the reviewed cities (see Figure 2 in Article 1 for more detail). The conspicuous role of food in driving city-wide EF profiles was natural given that agriculture dominates human occupation of land globally [21]. Again, the primary driver of the urban EF foodprint were animal-sourced foods, most notably beef, which is well documented to require the largest land use for production due to its feed and grazing demands [41, 21].

Plotting EF foodprint against wealth for the cities in Figure 2.4a revealed a modest correlation (R^2=0.35). This correlation was more logarithmic than linear, agreeing with the earlier discussion surrounding saturating demand beyond a certain income level and Engel's law. The difference between the study average and global average was not as pronounced as in the MFA studies. When comparing non- and OECD member cities, a borderline statistical difference (p=0.0471) was found between the groups' means (see Figure 2.4b). One reason for the narrowing of the gap could be the inclusion of several South American cities in my review, which had elevated beef consumption relative to cities of comparable income. These exceptional cities highlight that foodprint scale is an outcome of a factors beyond wealth, including cultural proclivities, comparative advantages in production and access to cheap food, both imported and local.

2.3 City governments and food sustainability

As my analysis showed, food consumption plays an important role in a city's metabolism and consequently, its CF, EF, and to a lesser extent, mass flows. However, given the metabolic rift between city supply region, the environmental fallout of this consumption remains obscured from city-centric planners, politicians, designers, residents and other actors that influence urban sustainability. Other environmental challenges related to transport, building energy and water are more easily grasped as they viscerally impact city-dwellers, and are consequently well represented when cities discuss 'greening'. Some have even gone as far to call agriculture 'climate change's forgotten sector', due to the relatively low number of climate change mitigation initiatives targeting the agricultural sector [4].

In this section I try to see if the prominence of the urban foodprint in the broader, urban sustainability agenda accords with its effects on urban environmental performance. By *urban sustainability agenda* I refer to the discussions and concrete actions made explicitly by cities under the auspices of understanding and reducing their environmental impacts, be they multi-city agreements, climate change plans, municipal bylaws or the like.

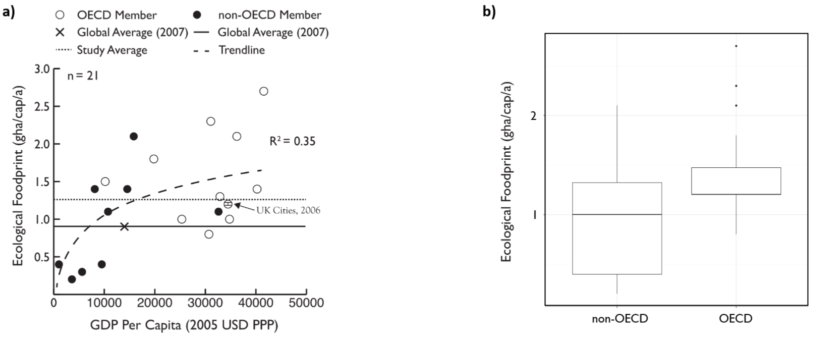

Figure 2.4: (a) Ecological foodprint versus income. (b) Boxplot of ecological foodprint based on OECD membership

Instead of performing a formal review of environmental plans from all available cities, I opted for a cursory analysis that consolidated earlier meta-analyses of such documents by others. I strengthened this by looking at larger multi-city sustainability frameworks and partnerships, and global sustainability 'gray literature'. This was in no way a surrogate for a full literature review, but it hopefully provided a useful glimpse of the degree to which food and environmental sustainability have been connected in the policy arena.

2.3.1 Literature reviews of urban climate change plans

Climate change mitigation and adaptation plans have been the most pervasive type of environmental plan published by cities in recent years, and hence, I focused on these when synthesizing the earlier literature. To date three comprehensive literature reviews have been performed of urban climate change plans, each outlined below.

The earliest review, from 2001, looked at pre-millennial climate change plans in US cities [5] for the 79 US members of the International Council for Environmental Initiatives (ICLEI), which as the name suggests is a coalition of cities dedicated to local action on global environmental issues. The authors review of climate change mitigation actions by membership cities identified neither food nor agriculture as concerted areas of climate change mitigation activity.

The next review from 2008 revisited the US ICLEI cities and their "first generation" climate change plans [59]. Aside from the author's own conclusion that the "proposed measures are inadequate" and "implementation is a problem"[59, p. 487], the document failed to identify concrete plans or actions to combat the climate change impacts of urban food consumption in US cities.

A final review from 2013 took a more international scope, looking at climate plans and experiments in 100 cities around the globe [10]. Like earlier climate change plans reviews, the main targets for GHG mitigation were mobility and building energy. The words 'food' and 'agriculture' are absent from the review.

2.3.2 Review of 'gray' literature

5th Intergovernmental Panel on Climate Change Report

The most recent Intergovernmental Panel on Climate Change (IPCC) report is a good barometer of the predominant discourse surrounding climate change at the global level. The IPCC 5$^{\text{th}}$ report contains a number of notable chapters that illuminate the degree that the urban foodprint has been linked to global warming.

The second working group's report concerns impacts, adaptation and vulnerability, with dedicated chapters on agriculturally related activities [44] and urban settlements [47]. The chapter on agriculture discusses cities tangentially with a few scattered references of the links between urbanization, shifting diets and rising food insecurity [44, p. 502]. The report's urban settlements chapter was much more explicit in both outlining the links of urbanization and dietary transitions that exacerbate the urban foodprint [47, p. 568] and introduces urban design interventions such as urban agriculture (UA) to help cities adapt to a future climate, though primarily from a food security angle [47, p. 560, 562, 568, 591].

The third working group's report addresses climate change mitigation, also containing dedicated agriculture [52] and human settlement [50] chapters. The former chapter only mentions cities in their capacity to infringe on peri-urban agricultural land as they expand. The human settlements chapter is light on links between food and urban climate change performance, but does contain a review of urban climate change plans. The authors found that although UA was included in the climate change mitigation plans of over twenty 20 cities, it was overshadowed by transport, waste, building energy demand and other urban climate change drivers [50, Fig. 12.22].

UN-Habitat World Cities Report 2016

The United Nations Human Settlements Programme releases 'state of the world's cities'biennially and in 2016 released a maiden report covering the evolution of urbanization and urban sustainability over a 20 year period, with chapter five focusing on environmental sustainability [57]. The report mentions the need for local governments to play a role in shaping sustainable food systems, including protecting agricultural land at the city fringe and integrating 'green' infrastructure into land use planning schemes. Moreover, this report advocates building stronger 'urban-nature' relationships in cities, tacitly invoking the metabolic rift.

Milan Urban Food Policy Pact

As mentioned in Chapter 1, the Milan Urban Food Policy Pact emerged out of the Milan Universal Exposition of 2015 [16] as a voluntary agreement between 132 cities around the world. It recognizes the connectivity of urban food consumption and global environmental challenges and is a strong proponent in the ability of cities to influence their food supply networks in positive ways. This includes the proliferation of UA and shorter supply chains as a means to reduce food waste and the urban foodprint [16].

Sustainable Food Cities Network

This network of over 50 cities in the United Kingdom committed to playing an active role in a future food systems that are healthier and more environmentally sustainable. An explicit part of their agenda is the reduction of ecological footprints from urban food consumption [54].

2.3.3 Findings

The agenda surrounding contemporary urban sustainability appears to be in a state of constant evolution. Reviews of climate change mitigation strategies in cities from 1990s up until 2013 revealed little acknowledgment of food as an important driver of urban environmental performance. Climate adaptation schemes in the assessed cities revolved around energy and transport planning, and less so the impacts embodied within imported goods. However, the past few years have seen a shift towards increased awareness and willingness to engage in the complex challenges surrounding urbanization, food consumption and the state of the earth system. The IPCC 5^{th} report does this by linking urbanization and changing diets, and whilst it does promote UA to strengthen food security, it stops short of promoting urban design or local solutions to affect environmental performance. More recent agreements between groups of cities directly link their food consumption with a host of environmental challenges in their supply regions and to larger global environmental issues such as global warming. Food is high on the agenda in participating cities and urban planning and design are proposed as remedies to unsustainable food supply networks. Although too early to tell, these observations portend a larger role of food in future urban climate change and environmental plans.

2.4 Chapter conclusions

My analysis of UM in three metrics suggests that the foodprint is an important driver of total urban environmental impacts, competing for top spot with transport and building-energy related pressures in most of the reviewed cities. It appears to be primarily linear in form, typified by large amount of food imported and the exhaust from food consumption exported beyond city boundaries. The linear metabolic profiles of cities is challenging as it precludes the collection of entrained nutrients in urban food waste, though this resource stream is secondary to sewage in its ability to contribute to a more circular metabolism. There is a tendency for the foodprint to grow with wealth, but a number of other factors are likely required to more fully explain the evolution of a city's foodprint. Article 1 and its related appendices fleshes out this analysis further.

Although long acknowledged in academic literature as a key aspect of a city's sustainability, movement by city governments has lagged. Up until the past few years, scant attention was given to the foodprint by cities in their sustainability plans. Recent developments such as the Milan Urban Food Policy Pact and the Sustainable Food Cities Network hint that the foodprint might start playing a more prominent role in more cities' environmental considerations.

Framing these results within the Northeast US, the average wealth of residents in my study region is well above the income threshold highlighted earlier, implying large foodprints for the region's residents. Assuming national estimates are representative of regional food-borne GHGs, the CF foodprint for the Northeast US would between of 2-3 metric tons CO_2e/capita/a [31, 34, 58]. This is certainly above the global average and likely in the same area as the largest foodprints unearthed by my review, which just so happened to be other US cities. Looking at the regional food waste outlook, a report on organic solid in the region found that many cities, including New York City, Boston and Philadelphia were making concerted efforts to push for increased circularity and resource recovery, though most projects were either in pilot stages or only relevant to commercial waste generators [25].

Lastly, a read of the GHG emissions reduction plans from the larger cities in the region showed that neither New York City [42], Philadelphia [37] nor the District of Columbia [26] address food related emissions, with my observations supported by a recent World Wildlife Federation report on climate change plans in US cities [60]. Of the action plans perused, only the City of Boston's explicitly mentions food sustainability and UA as a means to mitigate food-related GHG emissions [15]. It should be noted that New York City, Pittsburgh and Baltimore are all members of the Milan Urban Food Policy Pact [16]. It is likely that these municipal governments are cognizant of the intersection of sustainability and food, foreshadowing the emergence of the foodprint in their future environmental literature and actions.

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Chapter 3

The environmental performance of urban agriculture

3.1 Chapter overview

This chapter is a response to the second research theme: What is known of urban agriculture's (UA) environmental performance? What types of UA exist and can they be categorized based on material and energy regime? What is the comparative environmental impacts of urban and conventional agriculture supplying food to cities on the US Northeast? My point of departure is a review the literature surrounding UA's environmental dimensions. I then browse UA existing taxonomies and follow with my own attempt to provide an alternative UA systematics suitable for environmental assessments. I follow up with an application of this systematics to a comparative life cycle assessment (LCA) of urban farms in Boston and New York City. This chapter is a companion to Articles 2 and 3, which can be found in the appendices of this report.

3.2 Review of urban agriculture's environmental dimensions

It is worthwhile revisiting the definition of UA used here before delving into a detailed discussion of its merits and challenges. Here I borrow heavily from Koc et al's work [38], defining UA as *the production of food in and adjacent to cities, leveraging pre-existing urban material and energy flows as production factors.*

As I showed in Chapters 1 and 2, food demand is a key driver of a city's environmental burdens in a variety of dimensions. Importantly, cities themselves are awakening to the connections between their food consumption and many global, regional and local environmental challenges. One response has been the promotion of UA by governments, academics, urban designers and citizens alike [51, 66, 6]. Numerous reasons are cited to support increased food production in cities.

Here I looked at the various arguments given in literature to justify UA. The panoply of claims were grouped under four umbrella terms: *urban symbiosis, supply-network efficiencies, in-situ environmental gains and ex-situ environmental gains.* This was simplified from the original five terms used in Article 2, as hindsight allowed me to see that 'building energy' is an aspect of urban symbiosis. Relevant literature was perused between December 2013 to October 2015 to build the list of claims and look for supporting evidence (electronic repositories included Google Scholar, Scopus, ISI Web of Science, DTU Library, etc.) Here I present an abridged discussion of the current knowledge of UA's environmental performance, as this work is detailed further in Article 2.

3.2.1 Urban symbiosis

Urban symbiosis can be understood as the mutualisms between a farm and host city's material and energy systems afforded by the co-locating of food production in an urban setting. The supposed outcomes of this coupling of farm and city are reduced inputs for farming, attenuation of local pollution loading and resource recycling amongst others.

Building energy was one area where a number of claims were found in literature. UA is supposed to passively reduce building energy demands by increasing roof albedo (light reflection) [54], providing evaporative cooling of a building's micro-climate [54] and insulating the host building [65]. Models of vegetated roofs supported these claims with 41% heating load [10] and 23% cooling load reductions [4], though field trials revealed more modest decreases (5-10%) [40]. Active benefits of coupling a buildings heating-ventilation and air conditioning (HVAC) system with rooftop greenhouses have also been surmised [1, 58], with a 79% cooling load reduction modeled in a Mediterranean case [8].

Ample evidence was found supporting *nutrient capture and recycling* of a host city's organic solid waste, gray water (shower, sink) or black water (sewage) for fertilizer. Sewage application on farms in 19th century Paris [3] and contemporary African [59] and Asian [36] urban farms have been observed. Solid waste derived compost is commonly used on Cuban [28], British [14] and Parisian [21] urban farms. The benefits of alternative nutrient sources/waste attenuation should of course be weighed against the risk of introducing pathogens and heavy metals to produce.

The ability for *rainwater harvesting* to reduce farm irrigation needs while attenuating surface water runoff was also encountered in the literature [24, 11]. Support for this included rainwater harvesting by greenhouses in New York City [47] and Barcelona, ES [61], although the impacts of such exercises on city-wide hydrology remain untested. Moreover, the potential disservices of nutrient and pollutant laden runoff have seen limited study to date [50, 41]. Concerns regarding entrainment of airborne pollution in rainwater should also be considered in relation to this mutualism [75].

Excess building *heat utilization* has also been proposed [1]. Energy modeling of a Mediterranean rooftop greenhouse revealed poor alignment between periods of excess building heat and greenhouse heating needs, precluding this benefit in that case [8]. In contrast the owners of a rooftop greenhouse in Montreal, CA claimed to capture 50% of greenhouse heating needs from their host building [42].

3.2.2 Supply-network efficiency

Supply-network efficiency covers those claims related to UA's supposed ability to provide food with a leaner material and energy regime relative to conventional farming. Two main topics arise, the reduction of distance from farm to fork and UA's superior yields.

Ubiquitous in pro-UA literature was the emphasis on *reducing distance from farm to fork ("food-miles")* by co-locating production and consumption [6, 1, 66]. Whilst logical if UA products are consumed within the producing region, this overestimates the contribution of transport to the total environmental burdens of food production for *most* foods, as this is typically under 10% [20, 72]. Perhaps more important is the ability for this co-location to reduce wastage along the supply chain [60, 61]. Distribution losses are more of a challenge in the Global South [16] in contrast to wealthier societies where wastage is predominantly an issue during retailing and final consumption [27, 25, 71, 16]. Hypothetical UA work has assumed reduced edible losses along the supply chain [60, 5], though actual accounting exercises have yet to be done.

Another recurring claim surrounding UA supply-networks was UA's *superior yields* over conventional farming [65, 11], primarily based around assumptions of advanced hydroponic applications in UA. A study of Japanese "plant factory" utilizing hydroponics and automation revealed significantly higher yields than conventional counterparts [63, 12], but such operations are not constrained to urban areas and can sited in rural settings. Moreover, the benefits of reduced direct, land occupation should be weighed against profligate energy consumption and related greenhouse gas (GHG) emissions [63].

3.2.3 In-situ environmental benefits

In-situ environmental benefits are the improvements to a host city's local environment from UA, including increased biodiversity, attenuated heat island effect, soil quality upgrading and cleaner air.

Improved biodiversity is an ancillary benefit of UA as it provides a potential refuge for wildlife, particularly pollinators [24, 37]. Vegetated roofs planting multiple crops demonstrated improved flora diversity and increased pollinator presence [29, 48], although mono-cropping and/or pesticide use might have the opposite effect on urban biodiversity [56].

Attenuating the urban heat island effect (the combined increase in heat absorption by black surfaces in cities) was a common claim found in literature, since the additive improvements to building micro-climate (see Section 3.2.1) might lead to city-wide temperature reductions during summer in an edible city scenario [51, 74]. Models of potential vegetated roof proliferation in New York City and Toronto, CA supported this claim with potential summer, peak temperature reductions estimated to be $\approx 2°$[1, 4].

Improvements in soil quality have been seen in UA operations, whereby repeated compost applications led to a stabler soil structure, higher organic carbon content and greater nutrient absorption [14]. *Local air quality benefits* around urban forests supported this claim [33], although potential disservices of toxin releases by stressed crops have not been studied in urban environments [50].

3.2.4 Ex-situ environmental benefits

Ex-situ benefits refer to claims surrounding the ability for UA to improve environmental conditions in supply-regions and beyond, including carbon sequestration [64, 11], carbon footprint minimization [64], smaller ecological footprints [58] and improved biodiversity [58].

Claims by UA champions surrounding *carbon sequestration* appear optimistic. Firstly, meaningful carbon uptake results from significant long-term additions to biomass and soil, which are largely absent in annual crops grown and consumed within short time frames. Assessments of urban tree canopies in Toronto, CA [35] and Salt Lake City, US [50] showed uptakes of <1% of their the cities' annual CO_2 output, hinting at the limited capacity for UA to directly affect a city's carbon balance.

Carbon footprint minimization is a supposed consequence of the supply-network efficiencies and urban symbioses of UA. Results have generally supported this claim; urban rooftop tomatoes in Barcelona, ES [61] and local farms producing lettuce and chicken in Sydney, AU [57] had lower carbon intensities than their conventional counterparts. Hypothetical exercises at that city-level revealed potential reductions of embodied food-related GHGs in Lisbon, PT (10%) [5] and London, UK (1%) [39].

Shrinking ecological footprints and *improved soil quality* from UA are predicated on the assumption that UA will either help halt deforestation or allow farmland to return to its natural state. Such assumptions are difficult to justify in light of changing dietary patterns, increasing affluence and limited available land for agricultural expansion, portending either a stabilization or increase in cultivated land into the foreseeable future [19, 70, 55]. Nonetheless, large scale UA could help a city reduce ecological burdens if UA turns out to require less land, both directly and indirectly, than conventional produce, but no support for this was found in my review.

3.2.5 Findings

Evidence in support of UA as an environmental improvement over conventional food supply networks appeared mixed. I classified literature support for these claims as *well supported*, *preliminary* and *conjectural*:

- **Well supported:** urban stormwater management, building energy use reductions, local biodiversity improvements, nutrient capture, carbon footprint reduction
- **Preliminary:** improved air quality, carbon sequestration, urban heat island attenuation
- **Conjectural:** ecological footprint reduction, soil upgrading and biodiversity gains in supply-region, higher yields

The uncertainty of this picture is enhanced by the fact that most assessments of UA either assessed a single metric in isolation, making it difficult to account for unintended environmental trade-offs from such limited optimizations. Moreover, the number of studies on operating urban farms was limited to three at the time of review [57, 61, 63]. Lastly, assessments were on isolated types of UA, ignoring the wide variety of farms in operation and their potential to have disparate environmental impacts. My findings are supported by earlier reviews which identified uncertainty surrounding UA's environmental effectiveness as one of the largest data gaps in current UA research [66, 51].

3.3 Varieties of Urban Agriculture

Farming in and around cities has occurred for centuries all over the world [58, 65, 64]. Currently the balance of activity is in the Global South where UA is a prominent, if unstable, source of about 15% of global food production [49]. Wealthier countries moved away from UA with the advent of modern city planning and its penchant for separated land uses, though scattered moments of popularity in recent times can be found [44].

Over time UA has evolved into a number of forms and practices. Moreover, within a single UA form, different methods and inputs are demanded by local circumstances such as climate or ambient pollution. These considerations are relevant here as they dictate a farm's material and energy regime, and the resultant embodied environmental impacts of its products [26].

In my goal of assessing the contributions of UA to a city's environmental performance, I decided to use a schema to segment UA types based on their divergent material and energy needs. This was proposed in order to advance this study beyond the previous literature which has tended to lump many different types of farming practices under the general term "urban agriculture". To avoid redundant work, I reviewed existing literature to see if such an UA systematics already existed.

Study	UA Characteristics	UA Archetypes	Typology Focus
[64]	location, product destination, production scale/technology	(i) subsistence home production (ii) farm-type commercial production (iii) multi-cropped 'rurban' system	socioeconomic
[13]	farmer gender, marital status, formal education level, contribution to salary, crops, planted area, experience	(i) commercial gardening (ii) commercial gardening and livestock (iii) commercial livestock plus subsistence field cropping (iv) commercial gardening plus semi commercial cropping (v) commercial field cropping (vi) commercial gardening	socioeconomic
[32]	location, crops	(i) High and low density home gardening (ii) livestock production (iii) community gardens (iv) open space production (v) peri-urban production	technological
[46]	location, market, size, crops, intensification, gender	(i) home subsistence (ii) family type commercial (iii) entrepreneurs (iv) multi-cropping peri-urban farmers	socioeconomic
[22]	production type, location	(i) intensive gardening (ii) hydroponic rooftop gardening (iii) occupied lots (iv) vacant lots (v) rooftop farms (vi) conventional	technological
[30]	land tenure, end consumer	(i) urban homesteads (ii) community gardens (iii) urban farms	socioeconomic
[45]	location, scale, products	(i) peri-urban (ii) intra-urban (iii) rural	spatial
[2]	location	(i) peri-urban (ii) vacant/open-space (iii) household	spatial
[1]	location, morphology	(i) ground level farm (ii) rooftop farm (iii) greenhouse	technological
[66]	technology	(i) edible green walls (ii) vertical greenhouse (iii) vertical farm (iv) rooftop farm (v) rooftop greenhouse (vi) indoor farm	technological
[49]	location, technological level, gender, location	(i) small-scale private (ii) small-scale commercial (iii) large-scale commercial (iv) non-specialized farming	socioeconomic
[43]	agricultural method	(i) conventional (ii) low-biointensive (iii) high-biointensive	cultivation style
[69]	species diversity, end consumer, tenure	(i) field crops (ii) orchard (iii) micro-orchard (iv) diverse garden	socioeconomic

Table 3.1: Sample of existing UA typologies to date

3.3.1 Previous systematics of UA

A quick review of UA literature revealed that a significant number of UA systematics have been devised over the past two decades of UA research. Table 3.1 outlines the basic elements of previous systematics and the types of UA identified. The list is by no means exhaustive, but is meant to show the perspectives that have been used to classify UA to date.

As Table 3.1 shows, there has been a fair amount of work to date towards classifying UA. Most of the classifications fell into themes that I loosely described as technological (tools used in production), socioeconomic (aims and beneficiaries), spatial (location of UA) and cultivation style (organic, conventional, etc.) Out of these themes, technologically focused taxonomies were most relevant to my study, since different production modes such as greenhouses and open lots will have divergent material and energy needs, as opposed to, say, two greenhouses with different ownership structures. However, existing UA systematics lack two key properties: completeness and exclusivity. Completeness refers to the ability to cover all possible of UA forms. For instance, Specht et al.'s [66] taxonomy includes numerous building integrated farming methods, but does not account for farms on the ground. Exclusivity means that each UA form is unique and cannot be considered a permutation of another archetype. Ackermans's [1] taxonomy treats greenhouses and rooftop farms as separate types, even though greenhouses are found on roofs.

3.3.2 Environmentally relevant UA Systematics

Since none of the systematics found at the time of my review met my needs, I developed my own taxonomy that is sensitive to farming energy and material intensities and meets the completeness and exclusivity requirements outlined above.

The primary purpose of this systematics was to capture the essential material and energy aspects of different UA archetypes. The systematics also aimed to differentiate between the level and types of interactions a farm can have with a city's metabolism. Instead of trying to account for every unique permutation of UA, I opted for an approach that covered broad classes of the phenomenon. The outcome was a heuristic tool to support quick analysis of predicted UA environmental performance given two essential urban farm attributes: *siting* and *space-conditioning.*

Siting is binary in UA: ground-based (GB) or building-integrated (BI). This covers almost all types of urban farms (exceedingly rare floating farms [47] excluded) including those situated inside buildings (e.g plant factories).

Siting is critical because it has an important influence how a farm interacts with a city's metabolism. BI farms can be directly coupled with a building's energy and plumbing systems to utilize resources, while GB farms can have higher capacity for solid waste assimilation as they are not limited by structural concerns. Siting also impacts capital requirements, such as structural buttressing for rooftop farms or fencing for GB operations.

Space-conditioning is also binary: conditioned (C) or non-conditioned (NC). Conditioning influences the operational inputs of a farm; heating fuels to maintain greenhouse temperature in the winter or pesticide applications to combat pests on outdoor farms. Conditioned farms also negate or minimize dispersive losses (irrigation runoff, soil losses) that affect non-conditioned forms. Obvious implications of conditioning on capital requirements include the need for structural and glazing components, HVAC equipment and the like when constructing a conditioned a space. From my review of literature and visits to urban farms the C/NC distinction is also complete in describing UA types currently operating.

Combining these two attributes produced four distinct UA forms: GB-NC, GB-C, BI-NC and BI-C. In the following sections I will discuss there predicted energy and material regimes in three realms for the four UA types: operating characteristics, capital inputs and potential for urban symbiosis. Within each of these three realms are a number of indicators. The predicted characteristics of the UA forms for each are shown in Figure 3.1a-c. I will not belabor this text with a detailed description of each indicator since Article 2 explains the typology development and indicators. Article 2 also presents a cursory analysis of the typology's performance using previous environmental assessments of UA. Observations regarding the four systems were gleaned from existing UA literature and site visits to urban farms in Boston and New York City throughout 2015.

Ground-based, non-conditioned UA

GB-NC is arguably the simplest of the UA forms, typified by planting directly in topsoil of a site or raised containers (see Figure 3.2a). A combination of low yields and potential for high ambient losses of some inputs (water, fertilizers) are expected to lead to significant operational inputs per mass food delivered. Advantages are that GB-NC systems tend to require minimal capital inputs (e.g. fencing, irrigation lines, wheelbarrows, small unheated greenhouses, etc.) and do not use energy for heating. Urban symbiosis potential appears to be medium since GB-NC operations can easily accept solid waste and capture rainwater, but cannot directly interact with a buildings energy system nor tap effortlessly into generated waste flows. Environmental impacts for this UA types are expected to stem from the operational inputs.

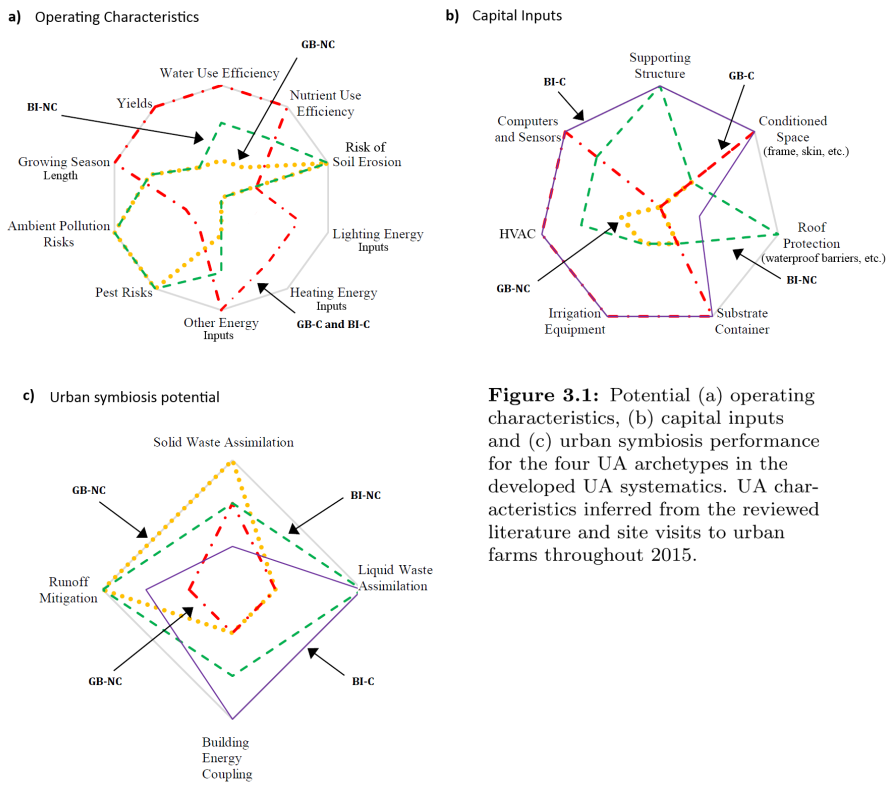

Figure 3.1: Potential (a) operating characteristics, (b) capital inputs and (c) urban symbiosis performance for the four UA archetypes in the developed UA systematics. UA characteristics inferred from the reviewed literature and site visits to urban farms throughout 2015.

Ground-based, conditioned UA

GB-C contrasts with GB-NC in that by moving farming activities into a controlled environment, water and nutrients are contained and less-prone to losses. Optimized conditions will likely lead to high yields and year-round growing. A downside of this is the risk of significant inputs of heating energy in colder climates and auxiliary energy for mechanical equipment (pumps, louvers, etc.) Capital inputs are greater for this farm type including HVAC equipment and structural components. Automated control systems are becoming increasingly common for these types of operations. Potential for urban symbiosis is the lowest amongst the four UA forms. Compost usage is limited due to the hydroponic nature of most conditioned operations. Conditioned spaces also do not attenuate runoff and are limited in their capacity

Figure 3.2: (a) GB-NC farm in Boston, (b) GB-C farm in Boston, (c) BI-NC farm in New York City and (d) BI-C farm in New York City

to capture rainwater to whatever storage space can be found on site (see the black water tank in the back corner of Figure 3.2b). Lastly, being isolated from the surrounding buildings hinders material and energy exchanges with the built environment. Environmental impacts for this form are expected to center around those embodied within capital and energy inputs, particularly heating inputs in cold climes.

Building-integrate, non-conditioned UA

BI-NC urban farms, similarly to GB-NC farms, are expected to have lower yields and higher losses of water and fertilizer inputs, although there is possibility for dispersed losses to be captured at the building edge. Capital inputs are expected to be medium to high, since BI-NC operations require green roof components (plastic barriers between soil and roof, edging, grow medium, fencing, etc.), mechanical and electronic equipment (pumps, sensors, computer), and conceivably, structural support to carry the added weight of the farm. Urban symbiosis potential appears to be high, since this type can accept solid waste, attenuate rainfall and couple directly with a building's energy and material fluxes. The balance of environmental burdens of growing on this type of farm are expected to lean more towards capital than operating inputs.

Building-integrated, conditioned UA

The BI-C form is expected to be very efficient in using water and nutrient flows, because of the contained nature of the growing environment, high yields and the propensity for hydroponic cultivation, which allows for recycling of these inputs. Like the GB-C, the need for stable temperatures in colder climates portends high energy related inputs. Capital requirements are also expected to be greater than the other UA forms, including all of the inputs of a GB-C farm plus the potential for structural buttressing to the host building. The trend in BI-C operations is towards increasing 'datafication' and computer monitoring of growing conditions (see http://openag.media.mit.edu or www.farmedhere.com), meaning that computer components may take on a larger importance in capital inputs in the future. Urban symbiosis appears to be medium since BI-C forms naturally interact well with the host building (including active coupling between growing environment and building HVAC). Conversely, BI-C farms are not ideally suited to accept compost nor rainwater.

3.4 Environmental performance of UA in the Northeast US

To date the majority of environmental assessments of UA have centered around a single UA form and have been performed in temperate climate zones (Barcelona, ES and Sydney, AU). I built on these foundational studies and modeled six urban farms in Boston and New York City. I compared the environmental impacts between the identified UA forms and tested assertions surrounding UA's environmental superiority over conventional farming. I reached into the industrial ecology toolbox to perform my analysis, using life cycle assessment (LCA) to appraise the relative environmental impacts of vegetables produced using the different systems. Article 3 was the outcome of this portion of the project.

3.4.1 Methodology

LCA is a continually evolving methodology to estimate the environmental implications over the life-cycle of a provided good or service. The 'life cycle' aspect references the different stages common to most economic activities: raw material extraction and forming, production and assembly, distribution, use and end-of-life (final disposal or recycling) [18]. By accounting for resource draws and pollution emissions over all (or some) of these stages, a holistic estimates of environmental burdens and resource depletion are provided for a defined product or service. Although methodological details are constantly in flux, the basic LCA framework is stable and codified through international standards [31] and best-practice documents [15].

Two main streams of LCA exist: process-based and input-output. I employed process-based LCA to compare the UA systems and all following discussion in Section 3.4.1 refers to this method (see Chapter 4 for an explanation of input-output LCA methodology). Process-based LCA divides the life-cycle of a product into distinct steps of 'processes'. Resource inputs

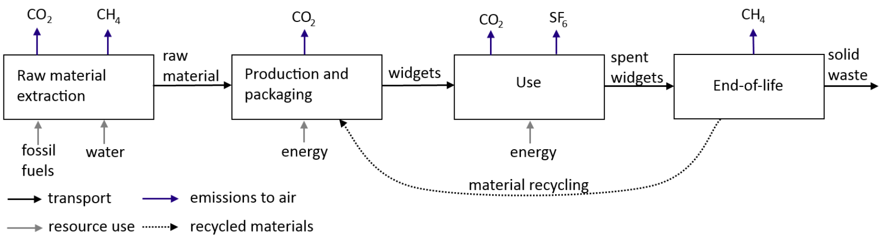

Figure 3.3: Life cycle of a generic good. In an LCA the amounts of different substances emitted to air would be converted to CO_2 equivalents and summed over for all processes. Likewise with the resource inputs which would be converted to metrics for water use, non-renewable resource depletion, etc.

and chemical emissions for each process are inventoried, and like emissions or inputs are converted to a common proxy related to different environmental/resource challenges. For example, all GHGs are converted to CO_2 equivalents to account for global warming potential. By summing these proxies over all steps, estimates of a product's complete environmental impacts or resource depletion are provided [18]. These indicators are called midpoint impact potentials, since they communicate a value of *estimated* environmental loading. Further manipulations can be performed to move to endpoint impact potential that from loading to damage (e.g. lost healthy years of living in the human population, species extinction, etc.) By performing these steps on different systems that provide the same product or service (e.g. provide 1 kg of protein to the US market using beef or tofu), termed the *functional unit*, a snapshot of comparative environmental performance of the systems is provided. Figure 3.3 provides a system diagram for a generic product.

Like any modeling framework, in simplifying the physical world LCA comes with a number of caveats. Firstly, summing impacts across all products ignores the spatially explicit nature of chemical pollution and the varied capacities of receiving environments to assimilate different substances. This is not so crucial in modeling global pollution challenges like global warming, but it is important when looking at agriculturally related pollution such as nutrient runoff and biocide application [26]. A similar challenge exists in modeling agriculturally related land occupation and water consumption [26].

Goal and scope

The goal of this assessment was to provide comparative environmental footprints of six UA systems and two conventional production systems from farm to point of purchase. This was analogous to drawing the system's boundaries around the two leftmost boxes and arrows in Figure 3.3. Although this assessment did not account for food preparation, cooking and disposal, by assuming that these practices are identical for the food produced in all systems, I could ignore the related environmental burdens. System modeling was performed in SimaPro 8.0.2 using the ecoinvent 3.1 database (most recent version at time of modeling). Ecoinvent 3.1 includes inventories of emissions and resource draws for a number of background process (energy provisions, fertilizer application, irrigation) that can be combined to develop complete models of product-service systems.

The unit of comparison employed here was 1 kilogram of fresh vegetables delivered to the point of purchase in Boston. Two vegetables were studied here: tomatoes and lettuce. These vegetables are chosen as they are both the most consumed in the US diet [27] and have the 2nd (tomato) and 3rd (lettuce) largest harvested areas in the US [17].

As mentioned earlier, LCA is an evolving field with a constantly shifting state-of-the-art. Most germane to my study were the methods of converting from raw pollution to midpoint impacts, since I only modeled to this step in the environmental cause-effect chain (see Article 5 for a discussion of difference between midpoint and endpoint and subjectivity in LCA). Competing chemical fate and impact modeling schools exist, each with their own sets of indicators and conversion factors to move from raw pollutant releases or resource draws to midpoint metrics [23]. I modeled midpoint impacts in six areas where food systems make significant contributions globally [68]: climate change (CC), freshwater ecotoxicity, marine eutrophication, water resource depletion (WRD), land use (LU) and mineral, fossil and non-renewable resource depletion (RD). I used the ILCD method to convert from elementary flows of pollutants and resources to midpoint indicators, since it includes the most advanced publicly advanced models for this purpose at present [23].

Consequential LCA method was applied in this study. This method accounts for expected changes at the market level based on 'marginal producers' available to supply the next unit of demand for a good. For instance, demanding palm oil at the market is modeled as production in Thailand and Malaysia since these are the two suppliers expanding their output to meet growing global demand, instead of a mix of historical palm-oil producers [73]. Moreover, in consequential modeling, system expansion is used to account for processes that produce multiple goods. Taking UA as an example, a BI farm would provide energy savings to the host building. In this study, the impacts from the energy that would have been demanded by the building in the absence of the farm were subtracted from the BI system. The other option would have been to divide the impacts of running the farm between food production and the service of insulating the host building using economic values or some other allocation key [15].

ID	Area (m^2)	Technology	Grow season	Profit?	Crops(s)
GB-NC1	560	low-tech field	Apr to Oct	no	tomato, arugula
GB-NC2	1269	average field	Apr to Oct	no	tomato, lettuce
GB-C1	558	soil media in heated greenhouse	Year round	no	tomato, salad greens
GB-C2	30	automated hydroponics	Year round	yes	tomato
BI-NC	1469	soil media	Apr to Oct	yes	tomato, lettuce
BI-C	3493	heated hydroponic greenhouse	Year round	yes	tomato

Table 3.2: Attributes of the farms included in this LCA

Case farms

All four UA types from my taxonomy were included. Data on material and energy consumption were collected during the 2015 growing season. Where data were lacking, the most reasonable estimation methods were used. These data included structural buttressing for BI farms, energy savings to buildings and water consumption for some of the farms. I was not able to secure a BI-C operation for this study (fiduciary responsibilities, labor burdens). As such, a hypothetical BI-C farm was used here based on an operating farm in Montreal, CA. Full details on the material and energy inventories for the studied farms can be found in the appendices of Article 3. The single farm operating in New York City (GB-NC2) was assumed to be operating in Boston, since the climates and growing seasons are similar.

Conventional agriculture

Primary data was not collected for the conventional tomato and lettuce systems. Instead the ‘tomatoGLO’ and ‘lettuce GLO’ processes were taken from the ecoinvent 3.1 database, which modeled European greenhouse production. Although representative in terms of the technologies used in North American production, these processes likely overestimate heating inputs compared to US counterparts using similar technologies or field tomatoes. Nonetheless, these proxies provide results in line with other LCAs of these products for land [70] and GHGs [27]. Final transport distances were taken from earlier work [53] and edible losses during the retail phase assumed to be 11% [71].

3.4.2 Results

Here I focus on CC, WRD and LU as they adequately describe the most important trends between the systems. CC was measured using the IPCC 2013 methodology outlined in Chapter 2. WRD used the Swiss Ecoscarcity method, which measures withdrawn water against local scarcity [23]. LU was accounted as the measure of fertile agricultural and forest occupied converted

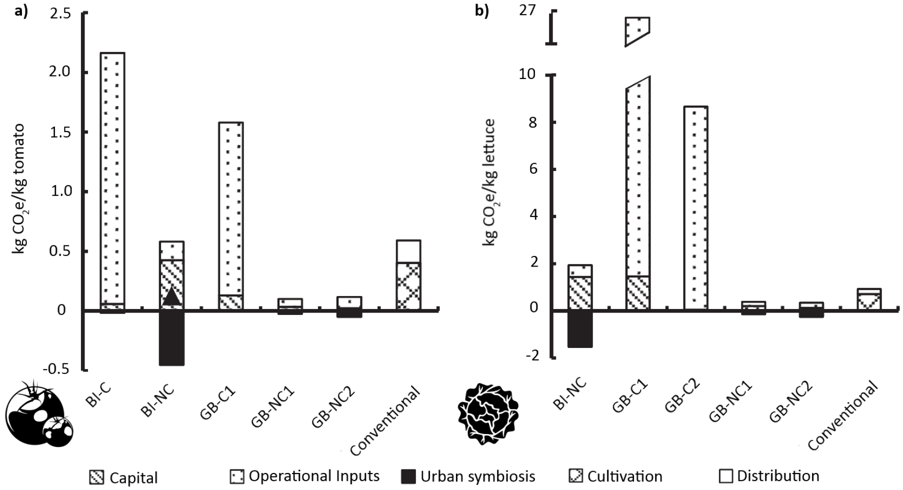

Figure 3.4: Potential CC impacts in terms of kg CO_2e for UA and conventional (a) tomatoes and (b) lettuce. Triangle identifies the total CC impacts for the BI-NC farm. Tomato icon source: The Noun Project - Zoran Djordjijevic. Lettuce icon source: The Noun Project: mmejules.

to organic carbon equivalents [23]. A short synopsis of the other indicators concludes this section to present any other interesting findings. The full LCA including all six metrics is available in Article 3. Figures 3.4-6 show the comparative life cycle impacts from the different production systems. UA results are broken down into the three areas covered by my UA taxonomy (see Figure 3.1). Results for conventional counterparts are broken down into cultivation and distribution impacts, as this was the amount of detailed afforded by the supporting ecoinvent processes.

Climate change

Figure 3.4 presents CC results. Not surprisingly the energy inputs for conditioned systems dominated predicted CC impacts, elevating both the tomato and lettuce producing systems well above other UA types, and importantly, higher than conventional methods. The latter fact is notable since the models of the conventional counterparts are likely elevated. Strikingly, the GB-C1 produced lettuce with a similar CC potential to steak [70, 27]. This was a combination of low yields and high energy inputs, hinting that if growers are going to proceed with conditioned UA in northern climes, choosing high producing crops will at least attenuate CC burdens.

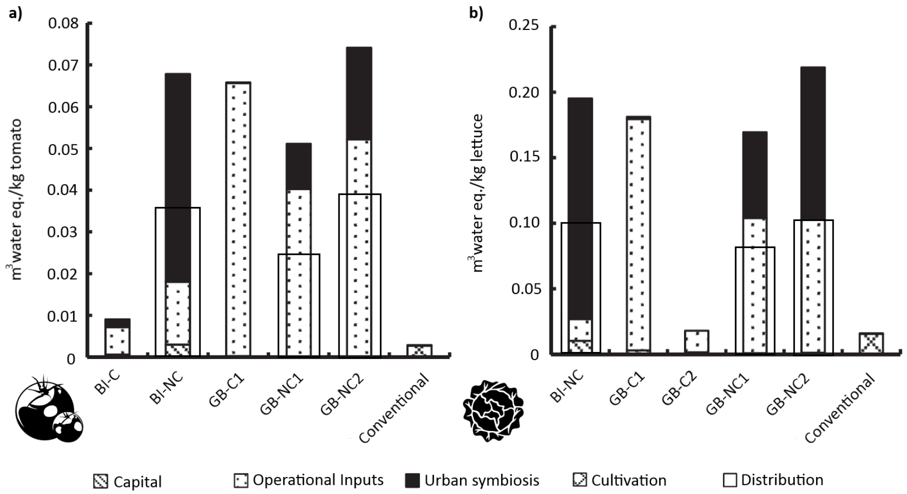

Figure 3.5: Estimated WRD impacts in terms of cubic meter equivalents (m^3 eq.) for UA and conventional (a) tomatoes and (b) lettuce. Black boxes outline the predicted range of values if runoff returned to the surrounding environment are discounted from the predicted impacts.

Results suggest that NC forms provide leaner production pathways than conventional methods. For the BI-NC, the appreciable embodied CC impacts in the steel needed for structural strengthening of the roof were tempered by the heating savings to the supporting building. None of the other farms benefited to an appreciable degree from their interactions with the surrounding material and energy flows. The GB-NC operations had low CC burdens due to the low capital and operational inputs.

It is also key to note that the benefits of eliminating 'food miles' were completely wiped out by heating costs, belying the conventional wisdom of many UA champions [6]. Choosing appropriate crops and UA forms for the local climate are ways to avoid these pitfalls as the NC farms demonstrate here.

Water resource depletion

Figure 3.5 focuses on predicted WRD impacts. Here the conditioned spaces tended to have an advantage in their ability to contain and efficiently use irrigation inputs. Both the BI-C and GB-C2 were the only systems with comparable footprints to conventional agriculture. GB-C1 is an anomaly for conditioned farms since it is used soil as a growing medium, and despite the sup-

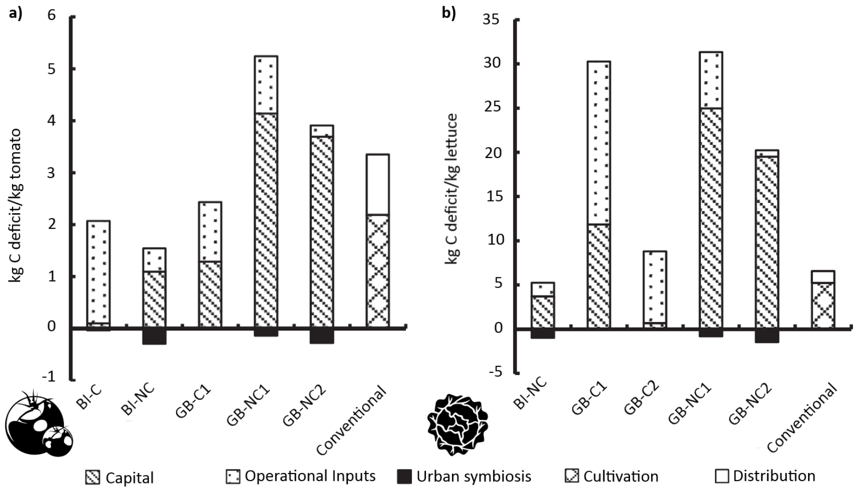

Figure 3.6: Predicted LU impacts in terms of kilograms carbon deficit (kg C deficit) for UA and conventional (a) tomatoes and (b) lettuce.

posedly efficient drip irrigation system, the farm's meager yields elevated its WRD above its conventional counterpart.

Non-conditioned farms appear hampered by high ambient losses yielding WRD results that are several factors larger than conventional produce. The large impacts from urban symbiosis were due to the harvesting of rainwater. The reason that these impacts were modeled as positive is that water incorporated into the plant instead of reaching the local environment can exacerbate local scarcity and degrade ecosystem functionality. In reality, some of the water is leaching off the site and reaching the local environment (≈50%) and hence I have used black boxes in Figures 3.5a-b to outline an alternative range of WRD results when ambient losses reach the ecosystem. Even with this generous reading of the results, predicted WRD for these farms would remain greater than conventional produce. Increased WRD impacts should be weighed against the attenuation of stormwater sewer loading, which might help Boston avoid the sewer overflow events that occasionally release raw sewage into local waterways [7].

Land use

None of the urban farms performed markedly better than conventional production, especially considering the uncertainties inherent in LCAs (a factor of two difference is warranted with this indicator before conclusions can be made). Even when UA was not directly occupying land for cultivation, the systems had indirect land use greater than or equal to conventional production. For the BI-NC this was due to the embodied land use in the extraction of iron ores and conversion to structural steel. The BI-C was burdened by from fossil fuel extraction for the electrical grid. Fossil fuel extraction was also the driver of the operational land use from both GB-C farms. The low yields of both GB-NC farms meant that their direct land occupation is equal (tomato) or greater than (lettuce) conventional production.

Other metrics

Of note were the low predicted impacts of all urban farms for toxicity and eutrophication impacts when growing tomatoes. This was a result of the very low artificial fertilizer application rates and near-ubiquitous eschewing of biocides by all the farms. Resource depletion was driven by the farm capital and heating inputs, though these only elevated the urban farms above conventional agriculture in a couple of instances (BI-NC tomato and lettuce, GB-C1 lettuce).

Alignment with UA taxonomy predictions

It is worthwhile to ground truth the proposed UA taxonomy in relation to the LCA findings, both to strengthen the taxonomy and to be able to provide assurance to potential practitioners of its dependability. It should be noted that this is not an exercise in post-rationalization since the taxonomy (Article 2) was published 6 months prior to the LCA results (Article 3).

An obvious but important observation is that no two UA types performed the same across all indicators, lowering the chances that I proposed redundant UA forms. Conversely, some of the farms within the same UA form showed divergent performance in some indicators (GB-C1 and 2), hinting that the taxonomy might require expansion to include UA sub-types (GB-C soil and GB-C soilless). A cautionary note surrounding these claims is the small sample size of this study, a problem that will hopefully be solved as the number of LCAs of urban farms grows.

Looking at the *operational inputs* the predicted results tend to hold up to what was observed. My conditioned cases were burdened by heating energy, but efficient in terms of water use. The outlier here is GB-C1, which in addition to high heat inputs, also had elevated water use. This was a result of the low yields which may be atypical for other conditioned farms that are operating for financial gain. The non-conditioned farms mirrored each other with their high water use, although the taxonomy was incorrect in assuming higher water use efficiency for the BI-NC form. A deviation was the BI-NC's

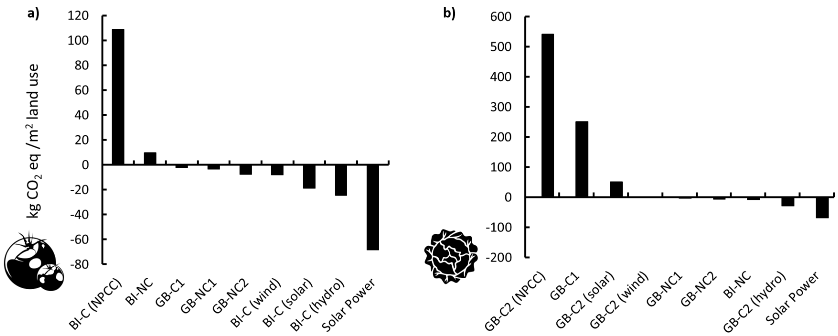

Figure 3.7: Marginal shift in Boston's climate change performance per square meter land use for (a) tomatoes and (b) lettuce. Terms in brackets outline the background electrical source. NPCC is the Northeastern Power Coordinating Council, the current grid supplying Boston.

soil erosion, which was actually higher than its ground-based counterpart due to losses from wind.

Capital inputs lined up quite well with Figure 3.1b, but the observed share of burdens resulting from capital for the conditioned systems was lower than expected. Moreover, though the taxonomy foresaw the important of capital in the BI-NC form, it was surprising to see steel buttressing play such a prominent role in the impacts. The near absence of capital in the GB-NC impacts agreed with the taxonomy.

Finally, given that *urban symbiosis* played a minimal role in the results it was difficult to compare with taxonomy predictions. Runoff capture was the highest for the non-conditioned farms, aligning with the taxonomy. The BI-NC was the most prolific in its coupling with the city, netting considerable energy savings to the host building and assimilating a fair amount of solid waste (redressing soil erosion), which also dovetailed with my initial appraisal of the systems. The BI-C also aligned with the taxonomy in that it captured half of its heating demand by coupling directly with the host buildings HVAC system. Liquid waste assimilation remains untested at present, so no comparisons can be gleaned in that direction.

3.4.3 Alternatives to UA in the city

UA appears to occasionally provide environmental gains to the host city over conventional agriculture, but often with trade offs between metrics. In Article 3, I tested the effects of changing the background electricity grid for the BI-C and GB-C2 farms (see Article 3, Table 5) to see if these high production systems could provide sustainable produce in the future. When powered by wind, solar or hydro, these UA forms performed better than conventional agriculture by some metrics and produced in significant quantities. Despite these positive signs, it is important to test the ability of UA to compete against alternative land applications that could also bring environmental benefits to the city. I compared the marginal shift in Boston's CC impacts per square meter urban land used for solar energy generation (substituting for the current grid) or UA (BI-C and GB-C2 substituting for tomatoes and lettuce, respectively). Appendix I of Article 3 details the assumptions in calculating the benefits of the solar panels and alternative energy grids.

Figures 3.7a-b compare the marginal climate change shifts with the different land uses. Although both the BI-C and GB-C2 appeared to provide some benefits to Boston's environmental profile when utilizing low carbon intensity electrical sources, their gains paled in comparison to installing solar panels on the same area.

3.4.4 Assessment shortcomings

A number of shortcomings of this assessment should be mentioned to allow for a balanced interpretation of the findings. One challenge was that the results were based off a single growing season. A longitudinal study would have captured both good and bad harvests, while also testing the evolution in resource intensities as younger farms matured. Such a data set would have also provided the opportunity for me to perform Monte-Carlo simulations of the LCA results given uncertainties on yields and farm inputs. In a similar vein, more case farms would have helped me identify outlier farms or better identify shortcomings with my taxonomy.

An important consideration is the risk for UA to be impacted by toxic chemicals, either taken up from the soil or aerial deposition [75, 62]. Near-field chemical exposure in consumer products, including foods, is a developing area of LCA [34], which could help strengthen future LCAs of UA. Boston, like many US cities, has a long history of soil contamination from metals, degreasers and hydrocarbons [9]. Elevated lead levels have been observed in vegetables unwittingly cultivated in contaminated soil in Boston [67], and this should be considered when appraising UA's application in the region. Northeastern cities are acutely aware of this challenge and have mandated soil testing for commercial urban farms in hopes of avoiding contaminated produce [52, 9]. Raised bed farming and only using compost from trusted sources are two ways to reduce the risk of farming in contaminated soil. Moreover, pollutants tend to accumulate best in certain parts of the plants (roots and sometimes tops), meaning that the edible portions of some plants can still be consumed as long as the offending portions are removed and residual soil is washed off during preparation [75].

Lastly, I did not consider social and economic benefits in this assessment. This is key since three of the included case farms were operating as community programs to provide food security, foster nutritional literacy and strengthen communities at the time of assessment (GB-C1, GB-NC1 and GB-NC2). These intangible benefits to the city are not captured in CC or LU indicators. Meager performance by non-profit farms in environmental terms should be situated relative to the stated objectives of those farm when assessing their success or failure. Contrasting not-for-profit farms to market-bound conventional operations might have also been a poor appraisal of UA's environmental promise. However, my GB-NC cases are representative of the majority of farms currently producing in my study region. As such, it might be reasonable to extrapolate my results to hypothetical scenarios of greater UA penetration in the Northeast US, since GB-NC operations appear to be the most likely UA form that would crop up given current conditions.

3.5 Chapter Conclusions

It appears that there are a lot of expectations surrounding the environmental performance of UA. From my review and earlier work by others, a definite gap exists between the claims and knowledge surrounding UA's environmental services and disservices [66, 51]. The dominant narrative in UA literature is an inherent belief in the sustainability of local food. Looking deeper into the literature revealed pro-UA claims in a multitude of dimensions: *urban symbiosis, supply-network efficiencies, in-situ environmental gains and ex-situ environmental gains*, with varying degrees of support for them.

To better quantify the environmental outcomes of UA and test the claims of others I developed a simple taxonomy to act as a heuristic for urban designers and to aid in a more structured analysis of the variety of forms of urban farming. Applying LCA to six farms in Boston and New York City showed that the environmental dimensions of the practice are more complex than has been stated in much of the literature, since the ability to produce more sustainable food than conventional farming varied with UA form. In the Northeast US, impacts for the high-tech, conditioned UA forms were larger than those of conventional farms, due to heating inputs. This contrasted with earlier studies in temperate climates [60, 61], underscoring the need to consider the climactic context when discussing UA. Low-tech UA had diminished CC potential, but at the expense of the LU and WRD metrics. The LCA also emphasized the pitfalls of equating local production with environmental sustainability, since the fuel savings from obviated transport were eclipsed by the heating impacts for conditioned farms. Even when UA was found to have advantages over conventional supply networks, these results should be placed in relation to the larger CC reductions provided by solar energy production on an equivalent area.

My findings suggest that UA interacts positively with Boston's metabolism, supporting assertions about the co-location benefits of urban farming. Nonetheless, the influence of these material and energy exchanges on the results for most farms tended to be weak. An exception to this was the coupling of farm and building energy system for the BI cases. UA interactions with the hydrological system may also be important, but my results pose more questions than answers, hinting at the need for future explorations of this potential nexus.

This work has only explored individual farms. A lingering question is what UA's environmental contributions might be when scaled up in a Northeast US city, both in terms of substituting conventional produce and affecting the pre-existing urban metabolism. The next chapter will address these unknowns, hypothesizing Boston as an 'edible city' and applying industrial ecology tools to test the environmental outcomes in my study region.

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Chapter 4

Assessment of the 'edible city'

4.1 Chapter overview

This chapter responds to research question 3: to what degree can the edible city affect the environmental foodprint of a city? The chapter is split into three parts. The first concerns estimating the baseline foodprint for my case city Boston. The second portion estimates the available space for UA in Boston. The final section takes the LCAs of UA from the previous chapter and scales them up to the city level to test the environmental impacts of the edible city. The companion publication for this chapter is Article 4 and its appendices. **Note that background EXIOBASE v3.2 data used at the time of the dissertation publication were flawed and have since been updated. Please see online version of article for correct results**.

4.1.1 Previous assessments of the edible city

To date only two studies have explicitly tested the impacts of UA above the farm scale: Kulak et al.'s study of a London, UK neighborhood [23] and Benis and Ferrão's assessment of Lisbon, PT [1]. Both methods relied on estimates of UA's environmental performance instead of using primary data. Moreover, baseline foodprints were assumed to be equal to the national average. This study differed from earlier work by (i) using actual data on urban farms, (ii) combining nutritional survey and demographics data to provide baseline foodprints at the sub-urban level, and (iii) including UA interactions with Boston's energy and material metabolism.

4.2 Quantifying Boston's foodprint

As outlined implicitly in Chapter 2, two methods are typically used to estimate a city's environmental burdens: top-down, input output and process-based, bottom-up models. Here I applied the top-down, input-output LCA (IO-LCA) method in estimating the foodprint of Boston. IO-LCA was chosen here because of its improved completeness over process-based LCA [34, 17]. Process-based-LCA has been applied at the urban scale [12], but the high level of detail it provides is not necessarily a benefit when looking at a city or neighborhood composed of aggregated consumers.

The foundation of IO-LCA are supply-use tables linking economic activities of different sectors within an economy. Given a single economy this can be represented as a matrix, A, where each entry in this matrix, $a_{i,j}$, is the monetary units demanded by sector j from sector i per unit output j. The total output from the economy, X, is then represented as combination of inter-sectoral exchanges to meet final demand, $X \cdot A$, and the final demand itself, Y. This can be represented as equation (4.1):

$$X = X \cdot A + Y \tag{4.1}$$

Re-arranging the above and solving for X yields (4.2), where I is the identity matrix of diagonal ones:

$$X = [I - A]^{-1} \cdot Y \tag{4.2}$$

The emissions (or resource use) from each sector i per unit output, r_i, are calculated as the total emissions from the sector, R_i, divided by that sector's total output x_i:

$$r_i = \frac{R_i}{x_i} \tag{4.3}$$

Thus a vector containing the emissions per unit output for all sectors, R, can be multiplied by total demand to determine total economy-wide pollution, B:

$$B = R \cdot [I - A]^{-1} \cdot Y \tag{4.4}$$

As the matrices are linear in nature, this framework can be used to estimate incremental emissions, Δb, resulting from incremental changes in final demand, Δy:

$$\Delta b = R \cdot [I - A]^{-1} \cdot \Delta y \tag{4.5}$$

Equations 4.1-5 describe the steps to develop an IO model for a single economy. It is conceptually simple (but computationally taxing) to extend this model to include multiple trading economies as (4.6) [39, 34, 17], given that dissimilarities between accounting methods for the different economies can be reconciled:

$$\begin{pmatrix} B_{11} & \cdots & B_{1n} \\ \vdots & \ddots & \vdots \\ B_{n1} & \cdots & B_{nn} \end{pmatrix} = \begin{pmatrix} R_1 & \cdots & 0 \\ \vdots & \ddots & \vdots \\ 0 & \cdots & R_n \end{pmatrix} \cdot \begin{pmatrix} I - A_{11} & \cdots & -A_{1n} \\ \vdots & \ddots & \vdots \\ -A_{n1} & \cdots & I - A_{nn} \end{pmatrix}^{-1} \cdot \begin{pmatrix} Y_{11} & \cdots & Y_{1n} \\ \vdots & \ddots & \vdots \\ Y_{n1} & \cdots & Y_{nn} \end{pmatrix} \tag{4.6}$$

B_{ij} are the emissions resulting from goods produced in economy j to satisfy the final demand of economy i, Y_{ij}. R_i are the emissions per unit output for the sectors in economy i. A_{ij} are the interindustry dependencies of economy i for products from economy j.

Over the past two decades numerous multi-region (MRIO) models based on this structure have been developed. The general trend in recent years has been convergence of results between the competing models [30]. Given their relatively equal performance, I chose the EXIOBASE 2.2 MRIO-LCA model, which includes 43 trading regions accounting for ≈90% of global economic activity in the year 2007. EXIOBASE 2.2 was used due to its high level of product disaggregation (over 200 unique industries) [52].

My application of EXIOBASE 2.2 was limited to its global warming potential (GWP) and land use extensions (R matrices). The GWP extension includes CO_2, CH_4, N_2O and SF_6 emissions, converted to CO_2 equivalents (CO_2e) using the IPCC 2013 method. The land use extension is unweighted so that a m^2 occupied in the amazon is considered equivalent to a m^2 occupied in the northern boreal forest.

4.2.1 Overcoming challenges in linking IO to food demands

A challenge in using IO-LCA to assess the foodprint is that in IO models the final demand vector (Y) and all of the interindustry dependencies (A) are in monetary terms. Determining the marginal impacts from economic activity requires a shift in final demand expressed in dollars. This intimates that a doubling of food expenditures results in a doubling of food production and resultant environmental burdens or resource use. There exists no evidence of a strictly linear relationship between expenditures and the volume of food purchased. Looking at nutritional surveys of the US shows that low income residents consume similar masses of food to high income residents [26] despite a clear incongruence between household food expenditures for the two groups [43]. The plateauing of the foodprint found in Chapter 2 belies this assumption further.

A different currency is needed to describe Boston's food demands. Calories are a suitable proxy for this currency, since they are common for all food items. Moreover, satisfying calorific needs is a common goal of all consumers, in contrast to intakes of macro- and micro-nutrients, which deviate considerably between different demographics [4]. That is, regardless of economic level or race, two consumers with a similar mass and physical activity will consume comparable amounts of calories per day [4, 26]. This logic has been applied successfully in a similar manner to calculate foodprints in studies of US households [20, 19]. I took this simplification and combined it with the EXIOBASE model to develop Boston's foodprint.

4.2.2 Linking EXIOBASE and the National Health and Nutrition Examination Survey

The Center for Disease Prevention and Control's National Health and Nutrition Examination Survey (NHANES) generates usual daily intakes of well-defined food groups (e.g. grains, dairy, dark green vegetables, etc.) for different demographics (sex, age, race) [26, 4]. The United States Department of Agriculture's (USDA) loss adjusted food availability (LAFA) data tracks food available to end consumers and lost throughout the supply network for ≈250 individual foods in the US [49].

Foods monitored in the LAFA project include data in kilo calories (kcal) and "serving equivalents". Serving equivalents come from the USDA dietary guidelines, which recommend the number of servings per day to consume within each food group. The unit allows foods to be understood in terms of their contributions to those recommended intakes. For instance, in the protein food group, both 31 grams of beef and 38 grams of lamb provide a single "ounce equivalent" of meat, and contribute the same nutritional value to the consumer [50]. NHANES usual daily intakes are also expressed in serving equivalents.

Using LAFA data I calculated the average kcal per serving in each food group. These conversion factors were weighted averages of individual foods within each group based on kcal *available* at the US market, accounting for losses between producer and plate. Scaling up from per capita calorific availability to total kcal available on the US market, and equating these with the carbon and land impacts calculated by EXIOBASE 2.2 for total US final demand, I developed embodied carbon and land impacts per kcal brought to market. Embodied impacts per kcal were combined with the conversion factors between kcal and serving equivalent to get the embodied impacts per serving. Multiplying these factors by the usual daily intakes from NHANES, I estimated the impact profiles for different demographics based on age and sex. Figure 4.1 outlines this work flow. Detailed explanations and calculations are in the supplementary information of Article 4, including descriptions of additional steps (linking food groups to products in EXIOBASE 2.2, determining transport impacts, etc.)

Foodprints at the sub-urban scale

I supplemented the food-borne environmental impact profiles of different demographics with population data to estimate Boston's foodprint. The 2010 US census provided population estimates of different demographic groups at the 'block-group' level, which are the most granular level available to the general public (typically 600 to 3000 residents). Wedding these two data

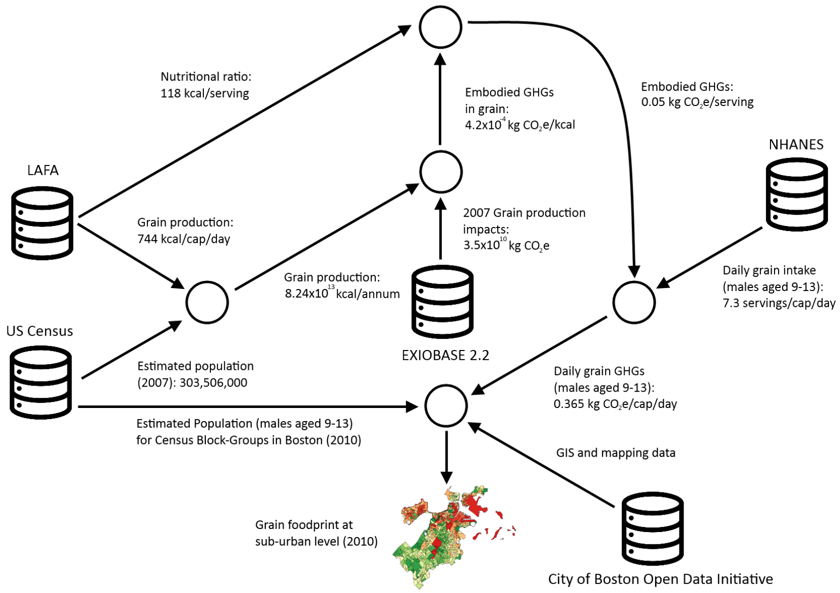

Figure 4.1: Work flow followed in linking USDA LAFA, EXIOBASE, NHANES, US Census and Boston Open Data Initiative data to develop foodprints. Grain consumption by adolescent boys used as an example. Database icon source: The Noun Project - Shmidt Sergey.

sources resulted in foodprints at the block-group level for Boston in the year 2010, including breakdowns of foodprint components. This model focused on the City of Boston, and did not include the bordering municipalities that comprise the Boston Statistical Metropolitan Area. The workflow in Figure 4.1 includes these final two steps and illustrates the cartographic output of my model.

4.2.3 Boston's baseline carbon and land foodprints

Figures 4.2a-b show the baseline average foodprint for the entirety of Boston in terms of (a) GWP and (b) land occupation for the year 2010. Figure 4.2c shows the average block-group carbon foodprint for Boston's 560 block-groups.

Carbon foodprint

Boston's baseline carbon foodprint in 2010 averaged 1179±20 kg CO_2e/cap/a, although block-group averages ranged between 1122 to 1342 kg CO_2e/cap/a. The percentage of adults had a strong impact on a block-group's burdens, particularly adult males, as their larger calorific needs elevated carbon foodprints above the average Bostonian. More youthful block-groups sat at the lower end of the carbon foodprint spectrum.

Meat and dairy were the major drivers of Boston's foodprint, accounting for more than 40% of total impacts, aligning with the urban foodprints reviewed in Chapter 2 and earlier assessments of the US diet's carbon impacts [16, 20, 51, 38, 37]. Conspicuously, beverage impacts contributed 16% to the total, which follows directly from the EXIOBASE sector 'Beverages' itself accounting for 19% of total impacts when the unaltered MRIO model was run at the national level. This finding agrees with Jones and Kammen's IO-LCA based work on US households [20], but deviates from the process-based work on the US diet where beverages are not explicitly included [16]. Until process-based LCA work including beverages of the US diet is performed, it is difficult to conclude on whether this is an indication of IO-LCA's superior coverage over process-based methods or a modeling aberration.

In terms of scale, the foodprint is smaller than other studies that have modeled average US carbon food impacts, which is surprising considering the completeness espoused by IO-LCA proponents. Heller and Keoleian's use of process-based LCA to gauge GHG emissions from US food consumption pegged average impacts at around 2000 kg CO_2e/cap/a with a possible range of 912 to 3358 kg CO_2e/cap/a [16]. Jones and Kammen estimated the foodprint of the average US citizen at approximately 3000 kg CO_2e/cap/a [20] using IO-LCA (Carnegie-Mellon 2002 US model: www.eiolca.net), agreeing with similar work of Weber and Matthew's [51]. Although my estimate fell within the lower range of possibilities estimated by Heller and Keoleian, it was markedly below earlier IO work. Discussions with the EXIOBASE developers at the Norwegian University of Science and Technology confirmed that my manipulations of the EXIOBASE 2.2 model were correct (published studies by this group calculated average US food impacts at 1200 kg CO_2e/cap/a[18]), and hence the gap between my work and the other studies lay within the database itself. This is a finding that warrants some reflexive thinking by IO-LCA model developers and users, as it does not jibe with the usual narrative of inventory completeness surrounding the work.

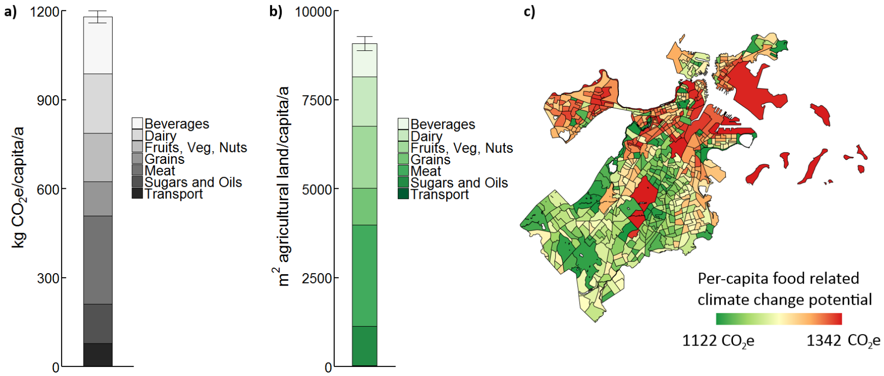

Figure 4.2: Baseline average foodprint for the entirety of Boston in terms of (a) climate change in CO_2e/cap/a and (b) land occupation in m^2/cap/a. (c) shows the average block-group carbon foodprint for Boston's 560 block-groups

Land use

Embodied land use in Boston's food consumption averaged 9077±198 m^2/cap/a, with range of 8600 to 10000 m^2/cap/a found amongst block-group averages. Demographic trends mirror those seen with the carbon foodprint.

Impacts emanated from animal-sourced foods (≈50%). Findings were on the lower end of predicted meat and dairy land use contributions relative to Peters et al's estimate of 75% [33] or Eshel and colleagues' finding of 1 ha/a/cap for meat and dairy alone [9], both for the average US consumer. This could have been an outcome of the large impacts of the 'General food nec' sector in the EXIOBASE model, which was allocated amongst multiple food categories when calculating per kcal embodied impacts (see Article 4 supporting information). This division might have taken some of the land use impacts for animal-sourced foods that were bundled within this catch-all product category - it includes frozen and highly processed foods of all types - and allocated an unnecessarily large quota of land occupation to vegetal foods. Unfortunately, given the high degree of product aggregation in IO models, this type of mis-allocation was unavoidable using my method.

Model performance

It is worth stepping back from the numbers to consider the outcomes of coupling NHANES and EXIOBASE 2.2 to model urban foodprints. Aside from the apparent underestimation of food-related climate change impacts by EXIOBASE 2.2, the general conclusions appear to be defensible in that the sources of impacts aligned with patterns seen in other diet studies. Challenges exist in allocating the impacts of the highly aggregated food products in EXIOBASE between different food groups in NHANES. Different decomposition methods could include weighting of the calorific contributions of food groups (e.g. beef calories get a higher percentage of 'General food nec' impacts than their straight proportion of total calories), but this veers in to a subjective space that could further compromise the results.

It is also worth considering whether the data gathering, parsing and programming work necessitated in building the model actually resulted in a better estimate of the urban foodprint. The tight spread around the mean of both the carbon and land foodprints suggests that one can get away with using average consumption data to model a city's food consumption, be they national or at a more granular level. A caveat is that this simplification is likely only reasonable for societies where the majority of residents lie above or below the level of wealth where the foodprint plateaus. Studies in highly 'mixed' societies should aim to stratify the city population into subpopulations so contrasting consumption profiles can be captured, as has been done for Beijing, CN [53].

4.3 Capacity for UA in Boston

No formal method exists for determining the amount of space available for UA in a city. In determining space on the ground, some researchers have used simple visual inspection of aerial photography to estimate latent UA area [29]. Others have taken a more advance approach, utilizing machine-learning algorithms to teach a computer to identify potential area from aerial photographs [36]. Estimating available UA space on buildings has seen even less study. Most researchers circumnavigate the challenge by assuming all buildings can support UA [14, 32].

I estimated ground UA area in Boston using a method that fell in the middle range of the technological complexity of earlier UA estimates. I employed what I call 'subtractive' and 'additive' approaches to arrive at lower and upper bounds of UA amenable land in the city. For building UA space, a combination of datasets were used to estimate available roof area given a number of constraints.

4.3.1 Additive estimate of UA ground area

The additive approach started from a total of 0 acres UA space and *added* UA amenable space to arrive at an estimate for the city. Data retrieved from the City of Boston Open Data Initiative [6] allowed me to analyze over 165,000 individual parcels from the city's 2016 Tax Assessment. The areas of

Figure 4.3: Illustrating additive(+) and subtractive(-) methods for estimating UA space in Boston

those properties with land use classification codes suggestive of UA potential were then added to the running total of UA space for Boston. Acceptable land uses were typically various marginal land applications (e.g. vacant residential, commercial, industrial, etc.) that were devoid of structures. Table 35 in the supplementary information of Article 4 outlines these in detail. Parking lots were also accounted as available UA space, the ramifications of which are considered in the final UA optimization model. Other types of marginal land outside the tax assessment data as determined by the Massachusetts 2005 land use map were also included (e.g. cropland, pasture, transitional, etc.) Figure 4.3 illustrates the method.

Some data manipulation was necessary, including the removal of double-counted parking lots that occupy the same tract of land (common for apartments) and removing pieces of land with slopes above 10°. Overlapping areas from the various data sources were disaggregated to avoid double counting (see Figure 7 of Article 4's supplementary information).

The additive method should be taken as the lower limit of UA available space, since there is certainly more 'open' land in the city than this method can identify. That being said, because I chose to ignore building shading and contamination, this is an upper limit on this estimate.

Available UA Space

Figure 4.4a presents the additive estimate of UA space. 2000 acres for UA were found in the city using the additive method. Parking lots accounted for 191 acres of UA land, and hence, the impacts of their inclusion should be marginal relative to total results.

4.3.2 Subtractive estimate of UA ground area

The subtractive approach started with the assumption that the entire land area of Boston was suitable for UA and then proceeded to *subtract* those areas with poor UA adaptability, chipping away until a final estimate was arrived at. Infrastructure (e.g. roads, sidewalks, airports) and buildings were by default subtracted from Boston's total area. Protected open spaces (permanent and temporary), parks and sports fields, as well as cemeteries were also removed from the city's footprint. Land with slopes greater than 10° were also excluded. The full list of UA incompatible land types and their data sources is available in Table 38 of Article 4's supporting information. Intersecting pieces of land were handled in the same manner as the additive method to avoid double counting. Figure 4.3 illustrates the method.

This method provided an upper limit on the land theoretically available for UA, in that it represented the area that could be cultivated without disrupting Boston's current spatial configuration. Moreover, this method ignored shading effects from buildings, and therefore provided an absolute upper limit on UA space. Another detail is that contaminated land was not accounted here, although this would not necessarily preclude any of the estimated space, but necessitate remediation or farming methods that avoid

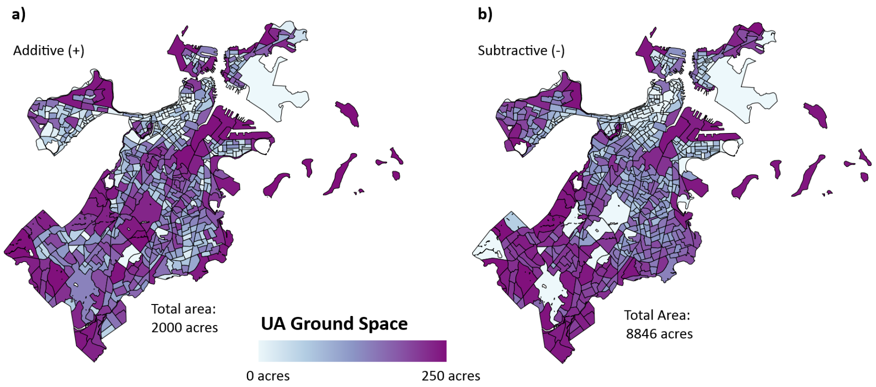

Figure 4.4: Estimated of ground space for UA in Boston using (a) additive methods and (b) subtractive methods.

contact with native soils.

Available UA Space

Total UA area was estimated at 8846 acres (1 acre ≈ 4047 m^2) in the city (≈29% of total city land), about four times greater than the additive estimate. Figure 4.4 shows how this land is dispersed amongst the city's block-groups. Naturally, block-groups outside of the denser, older parts of the city tended to have greater UA capacity, but some exceptions were present around the city's port. Conversely, some of the less-dense periphery block-groups had low UA space estimates due to significant park and protected green space coverage.

4.3.3 Building capacity for UA

The suitability of a building for UA is dependent on a myriad of factors: roof pitch, structural stability, historical considerations and the like. Unfortunately, such a dataset containing all of this information does not exist and so a roundabout method was employed to estimate Boston's UA roof space. Firstly, all buildings in historical preservation areas were excluded from hosting UA, since major alterations to building exteriors are not permitted in those neighborhoods. Essential transport infrastructure such as airport buildings were also precluded. Three criteria were used to determine the potential for UA on the remaining buildings: building age, roof pitch and height.

Building age is relevant since there is ostensibly some relation between a building's structural attributes and its age, given the gradual establishment and maturation of building codes throughout the 20th century. Since these codes resulted in the introduction of standards dictating minimum capacities for snow and wind loads, it can be reasonably assumed that older buildings are less able to support urban farms (19th century warehouses and factories may be notable exceptions to this). Instead of picking an arbitrary construction year as a cutoff, I modeled cutoff years ranging from 1900 to 2000 in decade intervals to see how this choice affected the results.

Roof pitch is an obvious design constraint, as slanted roofs largely preclude farming due to access limitations and challenges with growing media stability (edible walls are not considered in this study). As such, only flat roofed buildings were considered for UA here. Height is another natural limiting factor in the application of UA. Farming on buildings above a certain height may be unsafe (or simply uncomfortable) and/or complicated by unstable growing media from intense winds. An analysis of Boston's built stock found that over 90% of the buildings are less than 30 meters tall (see Figure 7 in Article 4's supplementary information), and thus, this was taken as a threshold for the maximum acceptable building height.

Developing dataset of Boston's buildings

In order to apply this criteria to Boston's built stock, data for all of the city's buildings were needed. To develop this dataset I adapted the method of Devila et al. [5] to develop a clean, consistent geospatial dataset of Boston's built stock that included roof pitch, height, age and historical designation. The entire process is detailed fully in the supplementary information of Article 4, but a quick overview is in order. First, the aforementioned tax data were cleaned to remove multiple entries for the same building and then linked with spatially explicit tax assessment parcel maps from the city's open data initiative [6]. The resulting data set contained mappable building ages, heights, roof types, use (residential or commercial) and heating/cooling system information. This dataset was then spatially joined with Boston's building dataset using the Geographic Information System software QGIS 2.4.0 and cleaned of unsuitable building types (e.g. ruins, foundations, infrastructure, etc.) or entries lacking essential information. The end result was a dataset of approximately 75,000 buildings in Boston that could be assessed for UA suitability.

Only a proportion of the entries in the tax assessment data included information on roof type. Where roof data was absent, a roof type was assigned probabilistically based on known information about Boston's residential (21.3% flat roofs) and commercial (25.2% flat roofs) building stocks.

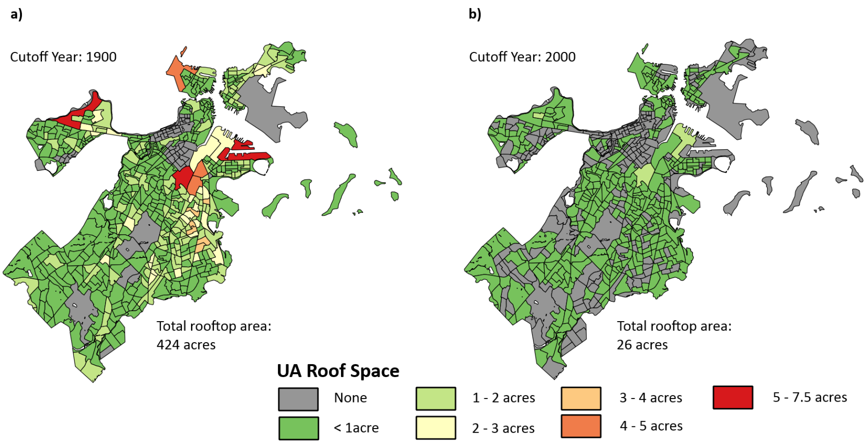

Figure 4.5: Estimated building space with a building construction cutoff years of (a) 1900 and (b) 2000 over 100 Monte-Carlo trials.

These probabilities taken from the US commercial building energy survey [47] or tax assessment data. Because of these uncertainties, 100 runs for each building age cutoff were completed with the roof type assigned to candidate buildings when they lacked roof data.

Available rooftop UA space

Figure 4.5a-b show average UA space over 100 Monte-Carlo runs for all of Boston's block-groups given building construction cutoffs of 1900 and 2000, respectively (results for all years are in Figure 12 of Article 4's supplementary information). Building space decreased from 424±8 acres with a cutoff year of 1900 to 26±3 acres with a 2000 limit. The decrease was concentrated in the older parts of the city, which change from red to green between Figures 4a and b. Block-groups with newer building stocks remained relatively stable in terms of rooftop space over the different cutoffs. At most, rooftop UA accounted for 20% of Boston's total UA space.

4.4 Assessing the edible city

The LCA of urban farms in Chapter 3 provided the foundation of my edible city model. Although only tomatoes and lettuce were assessed in that chapter, my collaborations with the farms produced primary data on fifteen different vegetables grown in Boston and New York City. In scaling up to the

Vegetable	UA Type	Yield (kg/m^2 UA)	Marginal climate change shift (CO_2e/m^2 UA)	Marginal land use shift (m^2/m^2 UA)
Beet	GB-NC	2.26	-0.57	0.70
Bell pepper	GB-NC	2.30	-2.29	1.84
Bell pepper	BI-NC	2.44	-0.15	-0.65
Cabbage	GB-NC	4.70	0.43	0.36
Carrot	BI-NC	1.59	1.47	0.16
Carrot	GB-NC	1.63	-0.51	1.16
Collard greens	GB-NC	0.41	0.10	1.14
Cucumber	BI-NC	5.28	-1.11	-1.73
Cucumber	GB-NC	3.34	-2.46	-0.01
Eggplant	GB-NC	2.27	-4.04	2.24
Green beans	GB-NC	4.27	-3.06	-4.36
Green beans	BI-NC	1.12	1.57	-1.56
Kale	BI-NC	1.26	1-57	-0.35
Kale	GB-NC	4.72	-4.24	-1.96
Lettuce	GB-NC	0.80	-0.90	2.61
Lettuce	BI-NC	0.80	0.49	0.14
Radish	BI-NC	3.11	0.45	-3.29
Scallion	BI-NC	0.93	2.22	0.43
Scallion	GB-NC	0.76	-0.25	1.42
Squash	GB-NC	2.54	-0.08	-0.01
Tomato	GB-NC	2.94	-2.50	0.80
Tomato	BI-NC	4.70	-1.61	-0.99
Turnip	BI-NC	1.84	1.63	-0.86
Turnip	GB-NC	3.50	-1.39	-1.17

Table 4.1: LCA results for different vegetables grown in Boston and New York City using ground-based (GB) and building-integrated (BI), non-conditioned (NC) UA forms. BI-NC results shown below do not include any interactions with the host building's energy system.

city I included all of these vegetables, since it is not reasonable to expect a city to either sustain itself nor significantly reduce its environmental impacts by only producing tomatoes and lettuce. Table 4.1 outlines the attributes of these vegetables, including yield and farm type. The table also presents the marginal shift in Boston's carbon and land foodprints per square meter UA space employed to grow each vegetable (see Article 4's supplementary information for detailed calculation method and data for conventional counterparts).

I only included the GB-NC and BI-NC UA forms (see Chapter 3 for descriptions) for multiple reasons. Firstly, these were the only two forms that were modeled to have environmental benefits over conventional production modes. Including the other UA forms would have been counterproductive since the edible city's negative impacts would have been all but guaranteed. Including the poorer performers might have provided a 'worst case scenario' for the edible city, but this would have been a misrepresentation of future UA scenarios given the constant innovations in urban farming and the predicted evolution of the region's electricity grid. The UA types included here were also, anecdotally, the predominant UA forms found in my study region, and hence, a better barometer of what the edible city could resemble in the short-term. At the same time, I have included individual vegetables with marginal increases in burdens when substituting for conventional agriculture, since this captured some of UA's unintended adverse effects. This also simulated a scenario where my 'farmers' were free to choose their crops within the limited set of UA forms I provided. For a further discussion of the challenges of using these data, please see Article 4.

4.4.1 Optimizing UA

In quantifying the edible city's environmental promise I applied 'greedy' algorithms to optimize separately for GWP and land use reductions. The algorithm were greedy in that they attempt to maximize the volume produced of those vegetables that have the largest marginal environmental benefits. Although greedy algorithms are not guaranteed to find the global optimum solution, it can identify local optimums and give a reasonable estimate of the global optimum [25]. In the simplest case, the greedy algorithm would result in the production of one vegetable, that with the largest environmental reductions, until all UA space was occupied in the city. In order to avoid this unrealistic scenario, constraints needed to be introduced. Here, the global constraint was that edible city was not allowed to produce a vegetable in volumes greater than it is currently being consumed in Boston. This provided an upper bound for all of the included vegetables and assured that my simulation was grounded in reality.

Figure 4.6 shows the structure of the program's optimization algorithm. In short, the program cycled through each block-group in Boston checking for unused UA space. Block-groups with free UA space were then tested to see if their needs for all vegetables in Table 4.1 were met. This local constraint ensured that the global constraint - city-wide production $\leq$ city-wide demands - was not violated. Block-groups that had space, but no longer needed vegetables were set aside. Block-groups with unmet demands attempted to produce the best available vegetable using their available space in 100 m^2 chunks, until UA space was exhausted or block-group vegetable demands were met. Once all block-groups were cycled through, if city-wide demands remained unsatisfied, block-groups with extra capacity produced (while optimizing for GWP or land use) until UA space was exhausted or city-wide vegetable demands met. Supplementary information of Article 4 describes the algorithm further and calculations undergirding my estimates of city-wide vegetable demands.

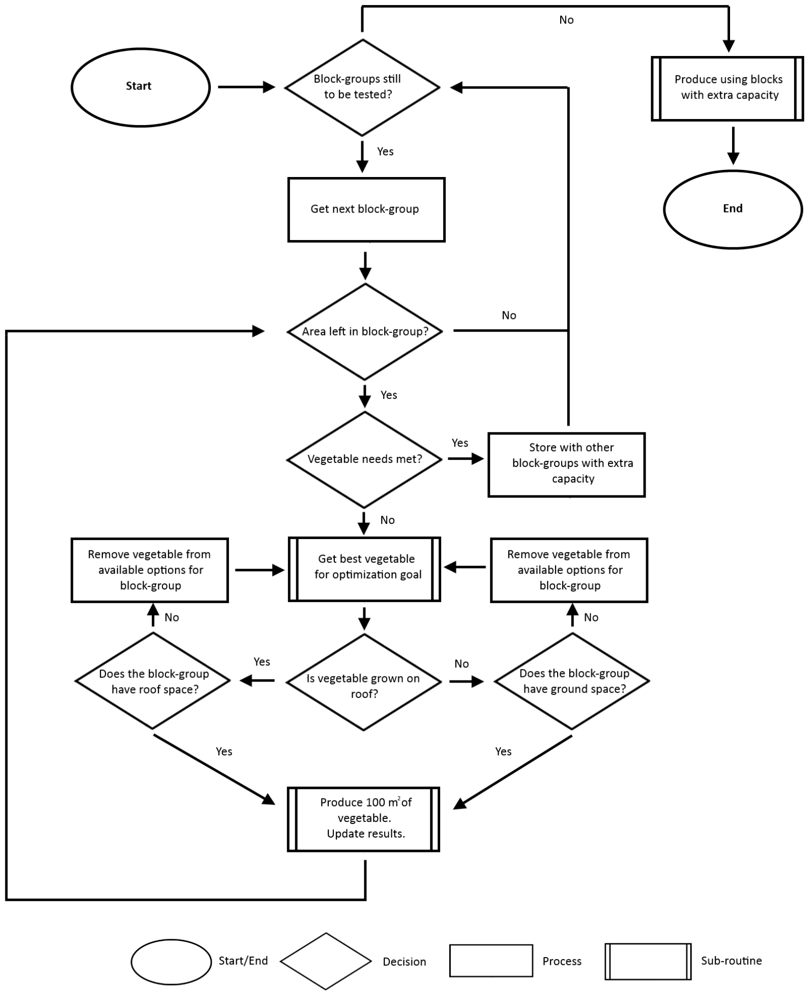

Figure 4.6: Optimization method employed in assessing the edible city

The optimization program was written in Python 2.7 and implemented throughout the fall of 2016. The program was run for both subtractive and additive UA space estimate methods. For each ground-UA space method 100 round Monte-Carlo simulations for each building age cutoff were also performed. Given the two indicators and two space estimation methods, four scenarios were run. Within each scenario building space was estimated for the different cutoff years as outlined above in Section 4.3.3. Table 4.2 below describes the four scenarios.

4.4.2 Interactions with Boston's metabolism

I accounted for three types of interaction between UA and Boston: building energy, waste assimilation and runoff avoidance. Building energy was calculated for each individual building based on the current heating system (natural gas, heating oil, electric, propane), insulation level (good, adequate, poor, none) and presence of air conditioning. Building data were taken from tax assessment data, and where lacking, stochastically assigned in a similar manner to roof type, using data for New England from the residential [48, 46] and commercial [48] building energy consumption surveys. Heating and cooling intensities were also retrieved from the building energy consumption surveys. Estimated energy savings from UA placement were taken from La Roche and Bernardion's field trials of green roofs in different US climate zones [24].

Waste assimilation capacity was taken directly from primary observations of compost uptake at the case farms [2]. Avoided runoff was estimated as the amount of impermeable space converted to UA, taken here as rooftops and parking, times the 15 year average annual rainfall [3]. Upper [31] and lower runoff [40] attenuation factors were used to test a range of results. the supplementary information of Article 4 further details the accounting of all three urban metabolic interactions.

Scenario	Scenario Details	Construction Cutoff Years
GWP(+)	Optimization for GWP reduction using additive method to estimate UA space	1900, 1910, 1920, 1930, 1940, 1950, 1960, 1970, 1980, 1990, 2000
GWP(-)	Optimization for GWP reduction using subtractive method to estimate UA space	1900, 1910, 1920, 1930, 1940, 1950, 1960, 1970, 1980, 1990, 2000
Land(+)	Optimization for land use reduction using additive method to estimate UA space	1900, 1910, 1920, 1930, 1940, 1950, 1960, 1970, 1980, 1990, 2000
Land (-)	Optimization for GWP reduction using subtractive method to estimate UA space	1900, 1910, 1920, 1930, 1940, 1950, 1960, 1970, 1980, 1990, 2000

Table 4.2: Four scenarios tested during the optimization of the edible city.

4.4.3 Results

In the following sections, all additive scenarios included parking as latent UA space. Tables 52, 56 and 60 in Article 4's supplementary information present additive scenarios excluding parking.

GWP and land use results

Figure 4.7a-b present the results for the four optimization scenarios averaged over all construction cutoff years (all tons are metric, in both text and figures). Error bars represent the standard error over all trials for each optimization. The GWP(+) optimization reduced Boston's foodprint by ≈18000 tons CO_2e/year or approximately 2.5% of the total carbon foodprint of the city (17% of fruits and vegetable impacts). This was 20% greater than the reductions from the Land(+) optimization. UA actually resulted in a net increase of Boston's land foodprint, although optimizing the Land(+) scenario was able to attenuate these by 40% compared to the GWP(+) trials. Relative to the baseline land foodprint presented in Section 4.2.3, this was less than 1%. The increase was a result of the mass application of GB-NC across the city and its tendency to produce in low yields. In terms of meeting Boston's vegetable demands, neither of the additive scenarios reached this goal.

The GWP(-) and Land(-) programs converged towards the same GWP and land use shifts, since they both met Boston's demands for the produced vegetables. The minor differences arose from the slightly different paths the algorithms traversed to meet the city-wide constraint (slightly more vegetables on the roof in the Land(+) scenario). Food-borne GWP impacts shrank by 24000 tons CO_2e/year (3.4% of foodprint) while land increased by 6400 acres/a (<1% total land foodprint). In the subtractive scenarios, about 50% of Boston's total UA space was utilized, intimating that an even larger volume of the city's nutrition could be procured locally. It should be kept in mind that this was only a fraction of Boston's vegetable demands (32% by mass and 24% by calories), and therefore, results do not necessarily predict that Boston can achieve complete vegetable self-sufficiency.

Across all scenarios, UA appeared to change baseline food related GWP and land use by marginal amounts in Boston. My findings agree with earlier studies of London [23] and Lisbon [1] that predicted baseline carbon foodprint reductions of less than 10% from UA. The increased land use observed in Boston contradict the findings of the Lisbon study [1]. Although there are a number of factors that could explain this difference, one key aspect might be the the reliance on conventional agriculture life-cycle impacts to develop proxies for UA performance in Lisbon. If I had made the same simplification,

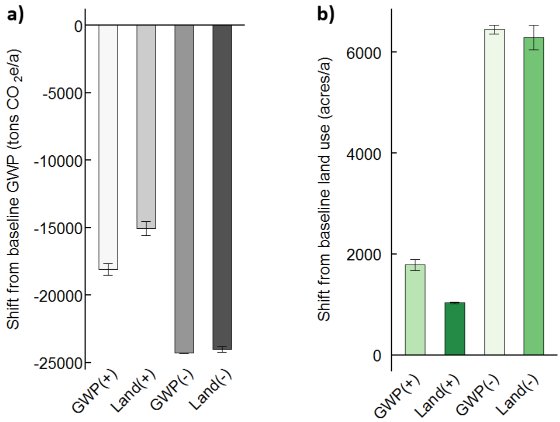

Figure 4.7: Predicted (a) GWP and (b) land use shifts of Boston's foodprint for all optimization scenarios. Each optimization is averaged over all construction cutoff years (100 runs per year). See Table 4.2 for scenario descriptions. Error bars show standard error around the mean.

my land use results would have also suggested that the edible city in Boston reduced land occupation. Here I have shown that simply removing transport burdens and assuming no wastage in distribution may not be an accurate substitute for farm level data, since the modes of production between urban and conventional agriculture practices can diverge markedly.

Urban metabolic interactions

Figures 4.8a-b show the interactions of UA with Boston's material and energy metabolism. Only the 1900 and 2000 construction cutoff years are shown as they demonstrate the key impacts of changing this parameter (full results in Figure 13 of Article 4's extra material). The combined results for the GWP(+) and Land(+) trials are shown for runoff retention as their effects on the city were nearly identical. Similarly, Figure 4.6b combines the GWP and land for the additive and subtractive scenarios.

Building energy was subsumed within the results of Figure 4.7a contributing 3200 CO_2e/year of the foodprint shift for the 1900 cutoff year in the GWP(+) and Land(+) scenarios. At its greatest, building energy reductions accounted for 19% of the total GWP shift, bottoming out at 1% when the rooftop area was at its minimum (year 2000). For the GWP(-) and Land(-) scenarios, building energy contributed less than 5% to total GWP savings. In terms of contributions to reducing Boston's total building energy use and related GHG impacts, the savings were in the single digits [5].

Runoff retention was also closely related to building space, since buildings represented the majority of area converted from impermeable surfaces. UA appeared to reduce surface runoff by 2.0 Mm^3, which accounted for less than

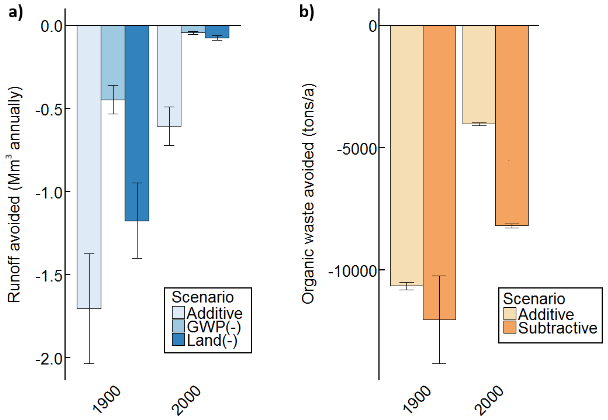

Figure 4.8: Predicted (a) runoff reduction and (b) waste assimilation of UA in Boston using years 1900 and 2000 as construction cutoffs. GWP(+) and Land(+) were averaged for runoff reduction. The 'additive' and 'subtractive' scenarios represent the averages of the GWP(+)/Land(+) and GWP(-)/Land(-) scenarios, respectively. Error bars represent the standard error over 100 runs. See Table 4.2 for scenario descriptions.

2% of the city's annual total (see Article 4 for estimate). Maximization of the runoff reduction occurred during the GWP(+) and Land(+) scenarios for cutoff year 1900, since these employed the greatest roof area. Waste assimilation was highest for the GWP(-) and Land(-) trials, hitting its peak of 12000 tons solid waste/annum when the building cutoff year was 1900, and dropping to about 4000 tons solid waste/annum for the GWP(+) and Land(+) scenarios for year 2000. At most, UA waste assimilation accounted for under 10% of Boston's total organic solid waste production [27].

The degree to which UA can shift the energy and material metabolism of an edible city appears meager in the context of baseline material and energy flows. The fundamental nature of the claims by UA proponents remain unchallenged, but my models do hint that the contribution of UA to more sustainable urban metabolisms is limited using the UA forms I modeled. For instance, even in capturing one tenth of organic solid waste flows in Boston, this would still likely represent little more than a couple percent of total nutrient fluxes through the city, since the primary mode of exhaust is human waste [11, 10, 21]. Even my own work in Article 3 that highlighted the importance of building energy interactions for the BI-NC farm appeared insignificant when scaled up to the context of a large, modern city's total energy demands.

4.4.4 Beyond environmental motives

The edible city appears to have limited ability to reduce the GWP and land foodprints of its residents or shift the baseline material and energy flows of the city. Notwithstanding, UA may be able to contribute in other manners to a more sustainable urban future, albeit in different realms. To explore this, I made a cursory analysis of how the edible city would influence its nutritional intake and generate income for UA practitioners, as these are two motives often invoked in promoting UA in the Northeast US [13].

Nutritional contribution

In appraising UA's nutritional aspects I considered UA's ability to help Boston meet the USDA recommended dietary intakes for dark green, red and orange and other vegetables. USDA guidelines outline suggested consumption in these three areas for different ages and levels of physical activity [45]. Starchy vegetable and legume needs were excluded, as none of the farms produced these during the 2015 growing season. The vegetables were classified as follows:

- **Dark green:** collard greens, kale
- **Red and orange:** bell pepper, carrot, tomato
- **Other:** beet, cabbage, cucumber, eggplant, green beans, lettuce, radish, scallion, squash

The optimization algorithm was altered to change the boundary condition from current vegetable *demands* to vegetable *needs*. Demographics data were combined with the recommended USDA guidelines to estimate Boston's nutritional needs for the three vegetable sub-groups. See the supplementary information of Article 4 for the calculation of nutritional needs and a richer discussion of the alterations to the optimization script. Two nutrition scenarios were modeled: Nutrition(+) and Nutrition(-) for the additive and subtractive UA area estimation methods, respectively. Once again, construction cutoff years between 1900 and 2000 were tested.

Economic contribution

The potential market value of vegetables was chosen to gauge the economic contributions of UA. This was taken as the production volume of a given product times its market price [41, 42].

I considered both intra-city trade and exporting. Intra-city trade occurred when block-groups with extra capacity sold to those lacking self sufficiency. Exporting modeled the production of food for sale outside the city, a practice

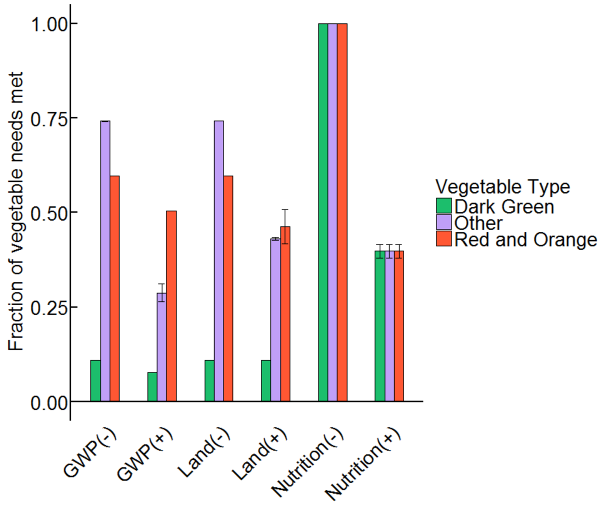

Figure 4.9: The nutritional contribution of the edible city given different optimization and UA space estimation scenarios averaged over all construction cutoff years. Error bars represent the standard error over all years and runs within each scenario. See Table 4.2 for scenario descriptions.

modeled to occur on excess land after Boston had satisfied its own vegetable or nutrition needs. See Article 4's supplementary material to better understand how the optimization script was adapted to monitor the market value production. Parking was excluded from the simulations using additive UA space estimates, since this is already a revenue generating land application.

Results

Figure 4.9 displays the edible city's nutritional contributions for all all construction cutoff years within each scenario. Firstly, when optimized for land and GHGs, the edible city supplied other and red and orange nutritional needs to a greater degree than the dark green needs, a reflection of the citizens natural proclivity for those vegetable types. When producing to meet vegetable demands in the GWP(-) and Land(-) runs, UA can meet 1/2 and 3/4 of red and orange and other vegetable guidelines, respectively. The Nutritional(+) scenario increased the dark green consumption at the expense of red/orange and other vegetables, providing approximately 40% of nutritional guidelines for all three. The edible city met USDA guidelines for all three vegetable types using 64% of UA space in the Nutritional(-) scenario.

Figure 4.10a shows the market value of UA in Boston when limited to intra-city transactions. The upper limit was around $50 million (all figures in US dollars) in the Nutrition(-) scenario, as the number of block-groups with deficits was at the maximum. Of interest is that the GWP(+) and Land(+)

scenarios provided the larger market values than their subtractive counterparts, a result of the higher number of block-groups that could not meet their own demands.

The results for an exporting edible city are shown in Figure 4.10b. The market value in these scenarios hovered around \$160 million, with the proportion exported varying depending on whether the city was growing to meet its current vegetable demands ($\approx$90%) or nutritional recommendations ($\approx$67%). Maximum market value coincided with Boston satiating its nutritional needs for the three vegetable groups, a side-effect of the large volume of high-yield dark green vegetables needed to meet USDA guidelines, which led to the largest yields of all the scenarios.

Situating the market value in terms of the Boston metropolitan area, \$160 million amounts to less than 1% of the regions gross economic activity [44]. Whilst insubstantial in this context, the ability of UA to provide revenue and employment in some of Boston's poorer areas is certainly noteworthy as shown in Figure 4.10c where UA value in each block-group is overlaid with the city's poverty rates. There already exists evidence of UA in the Boston neighborhood of Dorchester being used to generate jobs for lower income residents, both through direct sales at farmers markets and community supported agriculture (monthly/seasonal vegetable deliveries to subscribers) [22]. It has been estimated that a \$5-10 investment (*discounting the land value*) in UA in New York City can result in \$500-700 in fruits and vegetables [35]. Kaufman and Bailkey discussed the challenges of entrepreneurial UA, citing lack of institutional support in the US, unstable land tenure and low economic returns [22]. Although UA is increasingly supported and recognized by cities in the Northeast US, obstacles persist in finding permanent spaces for UA and consistently turning a profit with the practice. Moreover, city bureaucrats in the US tend to see UA as a transitional land use, preferring that vacant lots be developed into uses that generate more profitable tax receipts for the city [22]. Figure 4.11 presents how UA can quickly appear and disappear in a dynamic urban context.

4.4.5 Beyond quantifying UA

I have tried my best to outline in quantitative terms the performance of UA in environmental, and to a limited extent, nutritional and economic spheres. It is, however, essential to acknowledge that UA performs many intangible benefits that cannot be captured in my calculus. These benefits might represent more defensible justifications for implementing and promoting UA than those I have focused on in my review in Chapter 3 or tested in this chapter.

To a certain degree, I have already captured one of the intangibles: nutrition and physical health. However, simply counting nutritional units fails

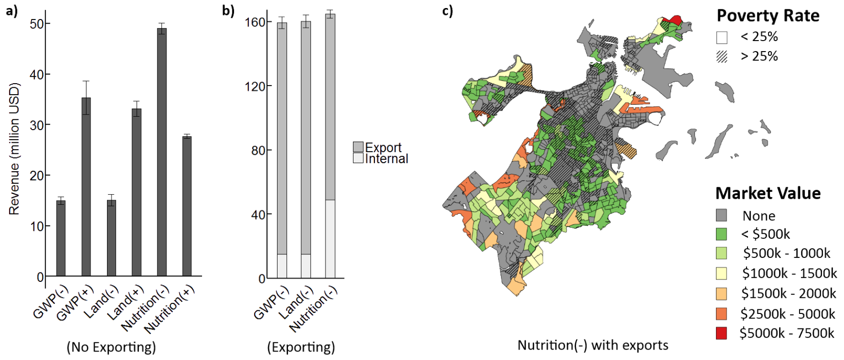

Figure 4.10: Market value of UA when (a) restricted to intra-city trading and (b) exporting to the Boston conurbation. (c) The market value of UA in different block-groups in the Nutrition(-) scenario.

to recognize the richer contribution that UA can make to the health of a community. Participation in UA is linked to sustained increases in fruit and vegetable consumption for urban farmers and non-farming members of their households [8]. Notably, much of the UA practiced in the US is in lower income neighborhoods that are disproportionately lacking in affordable and convenient access to fresh fruits and vegetables ('food deserts') [7, 8]. The ability of UA to play a role in redressing unequal food distribution and combat food insecurity in these neighborhoods should be emphasized given the elevated rates of diabetes and other lifestyle related diseases in the US's poor urban areas [7]. Another ancillary health benefit is the exercise that practitioners get while tending to their farms/gardens [8].

Psychological wellbeing of urbanites is another known benefit of urban farming. Numerous studies have found that urban farmers report lower levels of stress and dis-ease while farming, and that these positive impacts followed them off the farm into other aspects of their lives [15]. Urban farms and community gardens are also often used as spaces for families to spend leisure time and relax together [8]. Equally important are the reported connections that urbanites make to nature while farming [15] and increased agricultural literacy amongst youths participating in UA programs [8]. Perhaps these simple connections between food and nature can be built upon to help city-dwellers understand the teleconnections between cities and their supply regions. McClintock and colleagues have already advocated that UA could be a means to close the metabolic rift, galvanizing a more ecologically engaged urban population [28].

Crime prevention has also been linked to the presence of urban farms,

Figure 4.11: The transitory nature of vacant urban land in Newark, NJ. Moments of urban agriculture/gardening in 1985 and 1995. (photo credit: Camilo José Vergara)

with numerous anecdotal reports of positive changes to the surrounding area's character after the establishment of communal urban farming activities [8]. UA is also practiced explicitly as a youth education exercise in many instances [8], with the added benefit of providing social activities and alternative spaces in crime-plagued areas. Connected to this is the fact that many urban farms are communal spaces that encourage interaction between participants [8]. It is this type of contact that is the glue of neighborhoods, transforming strangers living in proximity to a community with shared hopes and ambitions, without which, stability and safety are difficult to maintain.

Finally, the aesthetic benefits of UA are also an area where quantitative tools are at a disadvantage to make judgments. Studies have shown that many UA practitioners and neighbors of urban farms preferred the farm to what was in its place beforehand (typically an empty lot) [8, 15]. UA's ability to secure open space in growing cities is also of note, provided much need verdant, moments in juxtaposition to buildings and infrastructure [8]. However, as Figure 4.11 reveals, these spaces require formalization and protection if they are to become permanent components of the urban fabric.

4.5 Chapter conclusions

Here I have attempted to look at some aspects of what I and others have called 'the edible city'. In my assessment I have tested the ability of scaled-up UA to affect carbon and land foodprints, finding that the contributions of UA in reducing these is meager using those UA practices that currently have some environmental advantages over conventional agriculture. Interactions with Boston's building energy, surface runoff and organic waste were also little more than superficial. Thus, the only conclusion one can draw from this work is that UA does not, in its current form, make a significant impact on the city of Boston, and by extension, similar cities in the Northeast US. A main reason for this is that by substituting vegetables (and ostensibly fruits), UA does not tackle the animal-sourced foods that are the primary drivers of the urban foodprint. Fruits and vegetables combined account for less than 14% and 20% of climate change and land use occupation burdens, respectively, limiting the scope of potential positive changes that UA can make to urban supply networks. As such, other means of reducing the foodprint should be explored.

At the same time, UA holds benefits in multiple other dimensions. Nutritional contributions could be significant to the residents of an edible city, particularly those in food deserts. Moreover, there appears to be a limited argument in favor of UA. The true strengths of UA may lie on the social side, as extensive research has revealed that urban farming can have positive impacts on physical health, psychological wellbeing, crime levels, community cohesiveness and neighborhood aesthetics. Lastly, if UA can in some small way bridge the psychological gap between urbanites and their supply regions, as has been suggested by some, then it might have the positive side-effect of a more ecologically minded population.

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Chapter 5

Complimenting the ’edible city’

5.1 Chapter Overview

Chapters 3 and 4 found that urban agriculture (UA) and the 'edible city' appear to provide limited environmental benefits in Boston given current UA practices in the region. My results are likely transferable to many other cities in the Northeast US with comparable climes and built forms. This chapter is a response to those findings, asking an additional research question: what types of activities can compliment the edible city to lead to a more environmentally sustainable city? I start by revisiting the main drivers of the foodprint. Afterwards I explore the power of dietary changes to reduce Boston's urban foodprint. I then look at the ability of a novel, plant-based protein substitute to shift the foodprint. I close with a short discussion of the role that city governments can play in addressing the foodprint by stewarding the behavior of their residents. This chapter is supported by Articles 5 and 6, both of which can be found in the appendices.

5.2 Revisiting foodprint drivers

My review of previous urban foodprint estimates showed that animal-sourced foods are, without exception, the largest components of a city's foodprint. This corroborates with my own modeling of Boston's where meat and dairy consumption played the largest role in the greenhouse gas (GHG) and land foodprints of the city. Jones and Kammen's work on US households aligns with my findings, where they estimate that meat and dairy account for ≈50% of food related carbon emissions [22]. Similarly, Heller and Keoleian found that meat and dairy generated 60% of average US dietary GHGs [17]. UA diet related land use is also disproportionately a result of meat and dairy, accounting for approximately 3/4 by recent estimates [31, 7].

At the global scale, agriculture and related land use change account for a quarter of total anthropogenic GHG emissions [20], themselves primarily stemming from meat and dairy [20, 30, 39]. The reasons for this are manifold, including inefficient conversions from feed to animal mass, enteric fermentation by ruminants, manure management practices and deforestation for feed and grazing [20, 39]. Agricultural occupies nearly 40% of ice-free land area, with the bulk of this allocated to livestock grazing (40% of global cereal production is also destined for animal feed) [11]. Global water appropriation is primarily driven by agriculture, with the largest proportion of this going to animal feed crops and direct consumption on livestock farms [27, 19]. Lastly, livestock is associated with nutrient runoff from excrement that is having deleterious environmental impacts on local watersheds, soils and river basins [13, 7, 37].

With the bulk of food-borne environmental impacts beyond the grasp of solutions such as reducing 'food miles'or leaner production networks, my findings of the edible city's meager environmental contributions appear reasonable.

5.3 Mitigating the urban foodprint with alternative diets

Humans are blessed with a remarkably robust digestive tract that can extract nutrients and energy from a plethora of foods. As a result humans can survive on diets with little or no animal product intake. Numerous studies in the past fifteen years have shown that these types of diets are markedly leaner in their environmental burdens compared to typical diets in wealthy nations that lean heavily on animal products [39, 11]. A recent systematic review of over 60 studies found nearly unanimous agreement that switching from baseline diets to those with reduced animal-sourced intake netted median reductions of 20-30% GHG, land use and water impacts, although estimated shifts greater than 50% were common in all three indicators [1]. Please see Table 1 in Article 5 for my own short synopsis of such studies.

An intense area of study has been the shift to diets with no meat consumption (vegetarian) or animal products (vegan) [35]. Switches from baseline diets in wealthy countries to vegan were predicted to provide median reductions of 45% and 55% for GHG and land use, respectively [1]. In the same manner, vegetarian diets might result in reductions of 31%, 51% and 37% for GHGs, land use and water, respectively [1]. In a US context, shifts from average US food demands to vegetarian and vegan were estimated to reduce dietary GHGs by 33% and 53%, respectively [17], and vegetarian diets could attenuate the US diet's water footprint by 52% [33].

Given the above, it is worthwhile to estimate the degree that dietary shifts could reduce Boston's foodprint. Rather than extrapolating from previous work, I performed my own analysis and applied it to the city's population. I did this so I could model the diets at a detailed enough level to test the impacts of substituting individual foods with novel protein sources.

5.3.1 Method

I combined process-based with input-output (IO) life-cycle assessment and built a hybrid-LCA model of the US diet. I chose this LCA method due to the high level of detail that it afforded and the ability to include impacts from processing beyond the farm gate. I modeled three diets here: the mean

US (MUD), vegetarian (VEG) and vegan (VGN) diets. I used the MUD instead of Boston specific numbers due to the proximity of the city and national means found in the previous chapter. Moreover, intakes of the most environmentally intensive foods have been observed to be similar across income and ethnic categories [23], supported by the tight spread I found around the Boston's mean.

The functional unit - the basis of comparison between diets - was taken as *the fulfillment of the food demands of the respective diets over the course of a year*. This functional unit was left purposefully open-ended since the intakes of calories, masses and nutrients in the diets deviated from each other, as will be explained below. The scope of the assessment was from farm to factory gate, including the environmental impacts of post-harvest processing, but excluding final distribution and preparation. Impact categories covered were climate change (IPCC 2013 method) and area of organic land occupied (Impact 2002+ method) [21]. The land use impact assessment method was used at the behest of our industrial collaborator, and hence, the deviation from the unweighted land use methods applied in Chapters 3 and 4.

Hybrid LCA

Hybrid LCA combines the best aspects of process-LCA (detail) and IO-LCA (completeness). The foundation of this assessment was a process-based LCA that inventoried on-farm resource draws and emissions. The majority of the life cycle inventories were taken from earlier LCAs of individual food products. Ecoinvent 3.2 database background processes were then combined to model individual foods, or when available and appropriate in the database, provide complete LCAs of food products.

Environmental burdens from processing of the food beyond the farm gate were estimated with the Carnegie-Mellon IO-LCA database. This was a tiered hybrid approach, where the process- and IO-LCA results were separately calculated and then summed at the end. For instance, to determine the impacts of processed beef involved taking the impacts from livestock rearing on the farm as determined from the process-LCA and adding these to the impacts from the 'Animal (except poultry) slaughtering, rendering and processing' sector in the IO database. To avoid double counting of the upstream impacts of livestock rearing included in the IO-LCA, I removed inputs from the agricultural sectors in the supply-use table (A). The IO manipulations were then performed using the altered matrices and a final demand vector, Y, of 1 USD for the 'Animal (except poultry) slaughtering, rendering and processing' sector to estimate the impacts from a single unit off processing. Equations 5.1-6 outline these steps

For a hypothetical economy that only contains 'steel', 'electricity', 'farming' and 'slaughter', final demands for all sectors are set to 0 except 'slaughtering' which is set to 1 so that Y_1 is given by Equation 5.1:

$$Y_1 = \begin{pmatrix} y_{steel} \\ y_{farming} \\ y_{electricity} \\ y_{slaughter} \end{pmatrix} = \begin{pmatrix} 0 \\ 0 \\ 0 \\ 1 \end{pmatrix} \tag{5.1}$$

X_1 then corresponds to the market-wide reaction to a single USD of final demand from the 'slaughter' sector:

$$X_1 = [I - A]^{-1} \cdot Y_1 \tag{5.2}$$

To exclude impacts from infrastructure during slaughtering, one modifies the interindustry dependency matrix, A, where each entry, $a_{i,j}$, represents the final demand from sector i per unit output sector j:

$$A = \begin{pmatrix} a_{steel,steel} & a_{steel,farming} & a_{steel,electricity} & a_{steel,slaughter} \\ a_{farming,steel} & a_{farming,farming} & a_{farming,electricity} & a_{farming,slaughter} \\ a_{electricity,steel} & a_{electricity,farming} & a_{electricity,electricity} & a_{electricity,slaughter} \\ a_{slaughter,steel} & a_{slaughter,farming} & a_{slaughter,electricity} & a_{slaughter,slaughter} \end{pmatrix} \tag{5.3}$$

The necessary modification is to set $a_{steel,slaughter}$ to 0 so that the direct demands for 'steel' by the 'slaughter' sector are excluded.

$$A^* = \begin{pmatrix} a_{steel,steel} & a_{steel,farming} & a_{steel,electricity} & 0 \\ a_{farming,steel} & a_{farming,farming} & a_{farming,electricity} & a_{farming,slaughter} \\ a_{electricity,steel} & a_{electricity,farming} & a_{electricity,electricity} & a_{electricity,slaughter} \\ a_{slaughter,steel} & a_{slaughter,farming} & a_{slaughter,electricity} & a_{slaughter,slaughter} \end{pmatrix} \tag{5.4}$$

In a similar manner, the inputs from 'farming' are also removed to avoid the double counting of those impacts covered by the process-LCA portion of the assessment:

$$A^* = \begin{pmatrix} a_{steel,steel} & a_{steel,farming} & a_{steel,electricity} & 0 \\ a_{farming,steel} & a_{farming,farming} & a_{farming,electricity} & 0 \\ a_{electricity,steel} & a_{electricity,farming} & a_{electricity,electricity} & a_{electricity,slaughter} \\ a_{slaughter,steel} & a_{slaughter,farming} & a_{slaughter,electricity} & a_{slaughter,slaughter} \end{pmatrix} \tag{5.5}$$

The end result is an equation of the form:

$$X_1^* = [I - A^*]^{-1} \cdot Y_1 \tag{5.6}$$

Equation 5.6 can then be combined with the pollution intensity vector (see Equation 4.4) to estimate impacts from sectors that directly react to the demand for a single unit final demand for 'slaughter'.

In my model, only the inputs related to energy and chemical inputs were kept in the supporting IO-LCA processes, to the effect that infrastructure impacts were excluded since these are typically not relevant for high-volume food processing operations.

A link needed to be made between the IO model which was expressed in monetary units and the process-LCA model which used physical units. Thus the total economic output from relevant sectors were divided by the total physical production of the US economy to provide the dollars per mass produced. For instance the total value of the 'Poultry processing' sector in 2002 (the base year for the IO model) was $4.52 \mathrm{x} 10^{10}$ 2002 USD [42], which when divided by a poultry output of $2.36 \mathrm{x} 10^{10}$ kg for 2002 [40] provided a ratio of 1.92 2002 USD/kg output.

The supplementary information (S3) from Article 6 outlines the hybrid LCA methodology, including the IO sectors included, the calculation of the conversion factors and the inventories for the process based models.

Diets

The MUD was taken from the United States Department of Agriculture (USDA) loss-adjusted food availability (LAFA) numbers for the year 2010 [43]. Losses during the retail and consumption stages were included, so total production upstream of the final consumer were accounted. The LAFA data includes apparent consumption of over 250 individual food items, many of which are not consumed in substantial volumes by Americans. To lower the modeling burdens, those food items that contributed less than 1% to the nutritional intake of a given food group were excluded from the assessment. The food groups are listed in the LAFA data and are commensurate with those used in Chapter 4 in the development of the Boston foodprint model.

VEG and VGN diets were taken from 2010 USDA dietary guidelines, as the 2015 guidelines do not contain specific recommendations for a vegan diet [41]. The USDA guidelines provided recommended intakes in terms of servings of broad food groups (e.g. dark green vegetables). Consumption of vegetables within each food group were taken as their final share in the consumption of the MUD. Thus, if the 10% of the MUD's dark green vegetable intake came from spinach, the same was assumed for VEG and VGN dark green vegetable intakes. This resulted in VEG and VGN diets that were sensitive to US preferences. The VEG and VGN were scaled to the 2000 kcal/day, as this aligned with actual calorific intakes observed for US vegetarians [16]. Although this was less than the MUD LAFA values ($\approx$2400 kcal/day) it represented a more realistic facsimile of US meat-free diets than if they had been aligned with the MUD's energy consumption.

Two methods were used to model ground beef intake, as this was the focus of the second part of the study. An upper limit of ground beef intake used industry data on the percentage of ground beef on the market combined with a low estimate on the carcass yield from live cow to final product. The lower limit used US nutritional survey data which gave a lower estimate of consumption and a more positive estimate of carcass yield. The full details of the diet development and estimates of ground beef intake are in the supplementary material (S1 and S3) of Article 6.

5.3.2 Results

Figure 5.1a shows the result for climate change impact for the three diets. Estimated annual climate change impacts of the MUD were 2032±32 kg CO_2e/a, with the variance a result of the different ground beef estimates. This was significantly higher than my estimate from the previous chapter for Boston using EXIOBASE 2.2, but in line with other process-LCA work on the US diet [17]. Switching from the MUD to the VEG and VGN reduced these impacts by 32% and 67%, respectively. Meat and dairy accounted for over 60% of total climate change impacts, up from the ≈40% in the previous chapter, but still in agreement on the main drivers of GHG impacts. To reiterate my observation from the previous chapter, the low estimate from the EXIOBASE is embedded within the database itself, and should be reason to pause for concern for practitioners employing EXIOBASE. One explanation for this could the exclusion of land use change from the current EXIOBASE inventories, although these would not be large enough to account for the discrepancy between the hybrid and IO assessment [10, 20].

Land occupation for the MUD was 4165±17 m^2/a as shown in Figure 5.1b. The hybrid LCA predicted 50% lower land occupation than my previous chapter and top-down estimates by others [31]. This could be a result of the top-down IO methods inclusion of ultra low intensity grazing schemes (see Eshel at al. [7]). Here I estimated that over 80% of land use was attributable to animal products, higher than the 50% from the previous chapter, and more in line with previous work on the US diet [31]. Switching from the MUD to the VEG and VGN diets resulted in land use reductions of 70% and 79%, respectively. These findings were the high end of results by others, but not unreasonable [1]. See Article 4's supplementary information (S4) for a complete breakdown of the LCA. A quick check using an unweighted land use indicator (ReCiPe midpoint [14]) reinforced the findings with the MUD requiring 3947 ±16 m^2/a and the VEG and VGN reducing this by 73% and 84%, respectively.

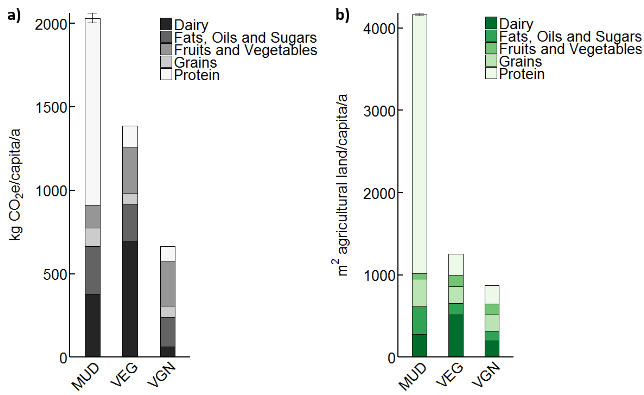

Figure 5.1: (a) climate change and (c) land occupation for the mean US (MUD), vegetarian (VEG) and vegan (VGN) diets. Error bars around the MUD show the range of results given different ground beef demands.

The results were only at individual scale. Figure 5.2 shows the result of a city-wide switch from the MUD to the VEG and VGN, respectively (all results in text and figures expressed in metric tons). In comparison to the GWP(-) scenario, switching diets led to 16 and 35 times the predicted GHG reductions. Curtailing the MUD's meat impacts by only 2.6% would have the same benefits as peppering Boston with UA in all of the city's interstitial spaces. Even a modest switch to 'meatless Mondays' would likely provide greater gains then UA. Comparisons with land are not necessary since UA actually caused increased land use.

In addition to the significant reductions in climate change and land use impacts through less intensive food choices, other environmental improvements are possible. Article 5 explores dietary shifts in a Danish context (similar to the US in general consumption patterns) using more indicators. There I found that impacts from nutrient runoff were also reduced with lower animal product intakes although uncertainties remained in some other realms (toxicity, ozone depletion, non-renewable resource scarcity). Article 6 also includes an assessment of water use, finding significant 70% and 75% reductions in shifting to the VEG and VGN patterns.

The benefits of such hypothetical dietary shifts are well known. Espousing the benefits of such shifts is easier than actually getting people to trade beef for beans. The consumption of meat is wrapped up in a whole host of psychological, social and hedonic factors that normalize the practice in our daily lives [6, 34, 32, 24]. Piazza and colleagues summed the rationalization of meat eating up as "the 4 Ns": normal, natural, necessary and nice. When directly confronted with facts about linkages between climate change and meat consumption, the common response is typically skepticism or a cognitive dissonance between stated meat and actual intake [24]. Other avenues to

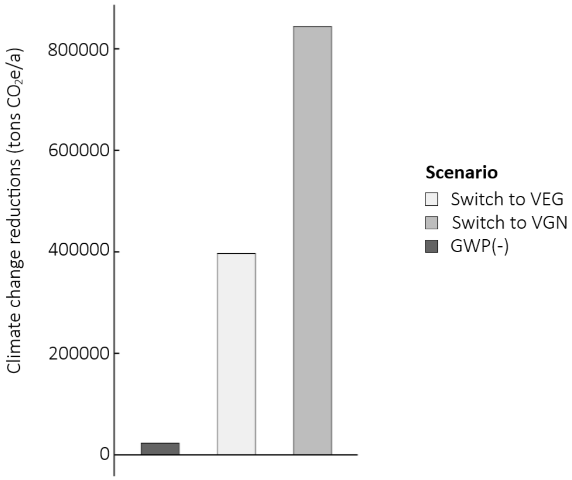

Figure 5.2: Potential shifts in Boston's foodprint by means of the edible city (GHG optimized, subtractive UA space estimate) and by shifting diets. Results are in metric tons.

help reduce the foodprint, aside from waiting for people to change their diets, should be explored.

5.4 Tackling the foodprint with a novel protein substitute

An alternative to radical alterations of the Bostonian diet might be the substitution of meat for some of the novel protein sources that have appeared on the market in recent years. These deviate from typical meat substitutes (e.g. tofu, seitan, etc.) in that they mimic the gustatory and cooking experience of meat consumption. This is different than taking tofu and forming it into the shape of a burger and putting some grill marks on the sides. New generation meat substitutes combine advanced bioengineering and food science to make plant-based burgers that bleed when cooked or grow meat in labs. Production methods vary, including 'cellular agriculture', yeast culture and 'bioprinting', but the products are similar in that they hold the potential to circumnavigate some of the psychological hangups that have limited the adoption of earlier meat substitutes by omnivorous.

5.4.1 Beef as a hotspot in the MUD

Of all of the foods consumed in the MUD, beef imparted the largest environmental burdens by a wide margin. In the above results beef accounted for 40% of total climate change impacts. Two thirds of all land impacts also stemmed from beef. There is ample literature support for these findings [7, 37, 30, 13, 8, 13] including my own work in Article 5 [15]. The reasons for beef's environmental intensity are manifold: ill-suited diets for ruminants and the related methane emissions from enteric fermentation, poor conversion rates from feed to flesh, deforestation for grazing and feed, as well as lax excrement management [13].

Despite declining beef consumption in the US in recent decades, beef remains a key component of the MUD [44]. Nonetheless, the apparent elasticity of beef intake bodes well for proponents of novel protein substitutes, as there might be segments of the population willing to shift from beef to other protein sources.

Impossible Foods

One of the new generation meat substitutes is Impossible Foods out of California. Impossible Foods has recently developed a plant based burger (PBB) that is intended to mimic and replace meat on the market. A modified yeast culture is utilized to produce leghemoglobin from vegetal inputs, a protein found in beef that gives it its distinctive olfactory qualities. The result is a plant sourced ground beef substitute that smells and tastes like beef, and even bleeds in a similar manner to beef. The main ingredients of the PBB are wheat protein, potato protein and coconut oil, resulting in a resource intensity of less than one quarter of that required for the same mass of ground beef [36].

I utilized primary data from Impossible Food's PBB pilot-scale production facility (136 kg/day) provided by the company. I used this data to test the impacts of the products diffusion into the MUD, VEG and VGN and to understand these shifts at Boston's scale.

5.4.2 Method

The hybrid-LCA used above was augmented to include the PBB in the three diets. PBB impacts were calculated using process-based LCA from farm to factory gate, with infrastructure at the PBB production facility ignored. To validate the PBB numbers, independent checks were performed by Berkeley Labs and the independent LCA consultant Quantis.

Penetration rates in the MUD, VEG and VGN of 10%, 25% and 50% were tested. The PBB replaced ground beef on a 1:1 mass basis in the MUD, since the nutritional quality is comparable with the exception of increased sodium levels and reduced cholesterol (see Article 6 supplementary information (S2) for more details). For the VEG and VGN, the PBB substituted for

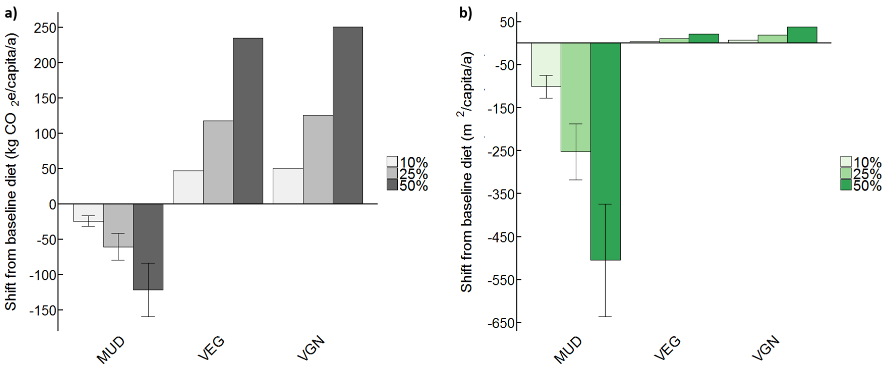

Figure 5.3: (a) climate change and (b) land occupation shifts with penetration of PBB in the the mean US (MUD), vegetarian (VEG) and vegan (VGN) diets

total protein intake, spread amongst the different recommended protein constituents outlined in the USDA guidelines. Supplementary information (S1) of Article 6 fully outlines the altered diets.

5.4.3 Results

Figures 5.3a-b display the change in impacts from the MUD, VEG and VGN at increasing diffusion rates of the PBB for climate change and land use. A 10% replacement rate reduced climate change impacts by approximately 25 kg CO_2e/a, similar to those provided by to the GWP(-) and Land(-) simulations (≈30 kg CO_2e/a). However, the 25% and 50% scenarios provided markedly greater reductions over UA. Land use impacts were reduced by 2-12% compared to baseline MUD levels, a clear improvement over the land use increases brought on by UA in Boston. Of interest was the PBB's ability to exacerbate the VEG and VGN climate change and land use impacts. However, given that less than 3% of the US population ascribes to a strict meat-free diet [5], the chance of a net increase in food related burdens in the US are low (see Article 6 for a richer discussion and analysis).

Figure 5.4 presents predicted reductions in climate change impacts for the City of Boston for different diffusion rates of the PBB into the MUD. Increases for the VEG and VGN were considered negligible and ignored here. At the 10% PBB substitution rate, UA provided a slight advantage in climate change reduction over the best UA scenario. At higher PBB intakes, estimated reductions were two to four times greater than the GWP(-) scenario.

The marginal foodprint improvements or disservices (land use) provided by UA should be considered by urban designers given these findings. If a switch to a plant-based meat substitute for a meager proportion of the diet can exact the same environmental benefits as the edible city, then the suitability of UA in Northeast US urban spaces needs to be revisited by those who are promoting UA as a means to reduce the foodprint. Moreover, the results here are for PBB at the pilot scale. Significant improvements can be expected due to mixing and heat transfer efficiencies at industrial scale, similar to what has been witnessed in bio-energy and materials [4].

Figure 5.5 shows the climate change impacts of the PBB against other protein sources per kilogram protein delivered to the consumer. Not only was PBB equal to or significantly better than animal-protein sources at present, but the predicted final climate change burdens after improvements with upscaling (shown by the black box around the PBB column), could make the PBB similar to eggs or insect protein in the future. Furthermore, reducing the impact of animal-sourced foods is biologically constrained (feed and water needs, digestion byproducts, etc.), and hence, these foods will not be able to evolve in the same manner as the PBB.

5.5 Study challenges

Aside from the typical data and modeling challenges associated with all LCA studies a few thoughts about the application of LCA to diets are warranted. Firstly, setting a proper functional unit where all three diets provide the same nutritional content is nearly impossible. Heller and Keoleian found that most practitioners rely on calories to set equivalent diets, but this ignores micro- and macro-nutrient content [18]. They went on to suggest the application of nutritional indices that consider calories and other nutritional components. However, one shortcoming of these indices is that they only consider

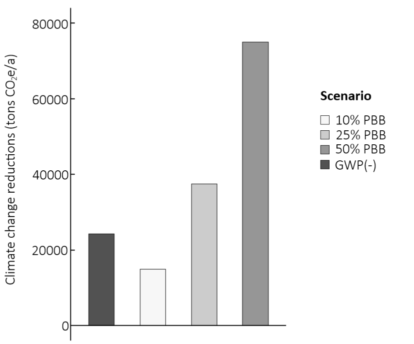

Figure 5.4: Potential shifts in Boston's foodprint by means of the edible city (GHG optimized, subtractive UA space estimate) and increasing intake of the plant based burger. Results are in metric tons.

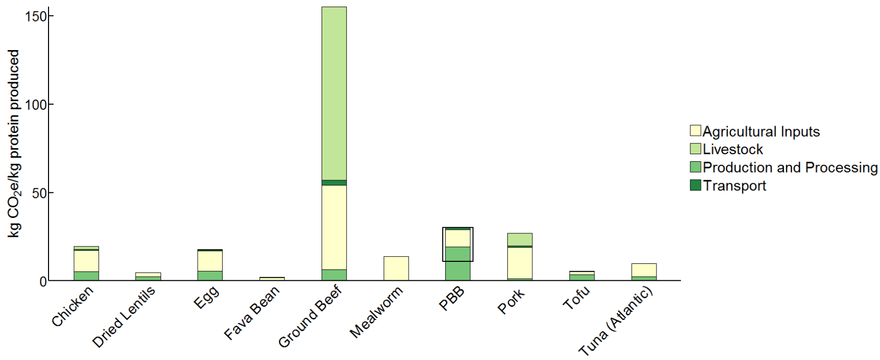

Figure 5.5: Climate change impacts per kilogram protein delivered to consumer for typical protein sources.

the presence or absence of various nutrients and lack any weighting of the relative importance of different components within the diet. In this study, I avoided this challenge to an extent by using the USDA guidelines which should provide roughly equivalent nutrient intakes for the VEG and VGN, but discrepancies are unavoidable.

Article 5 explores numerous additional challenges in using LCA, including the choice of impact assessment methodology, system boundaries and scope. One interesting observation in the LCA of the Danish diet was the challenges related to including agrichemicals in LCA assessments. In that study, the VGN diet ended up having the largest toxicity impacts since soybean feed for livestock was credited for avoided palm oil production. The life cycle impact assessment method I used in that assessment (ReCiPe [14]) included palm related pesticides, but not those used in soybean cultivation, resulting in a large negative toxicity result for the soybean feed needed to support the omnivorous diet. The occurrence of these types of challenges in LCA are one of the reasons that I limited the bulk of my work on the urban foodprint to climate change and land use, since these are two of the areas with more robust accounting methods and straightforward readings of the results.

5.6 Cities affecting diets

The extent that cities can directly influence the diets of its residents is limited in a free society. New York City's ban on extra-large soda servings in a bid to combat diabetes, aside from smacking of paternalism and being deeply unpopular, was eventually ruled unlawful for the state supreme court

[12]. Notwithstanding, opportunities to use less invasive means to change diets abound insofar as government run meal programs for inner-city children could be designed to minimize or exclude meat. The dietary guidelines used in this study are another example of such maneuvers, with explicit guidelines for Mediterranean (low-meat), vegetarian and vegan diets. The USDA guidelines also outlines the types of shifts needed to move from the MUD to a more balanced diet and includes instructions for influential actors (school principals, cafeteria workers, community leaders) on how to steward the public towards meaningful, long lasting dietary changes [28].

Dietary guidelines tend to be driven by public health concerns. Fortunately, this dovetails nicely with environmental goals, since healthier diets prescribe lower red meat and dairy intake [39, 1]. China's new diet guidelines aim to reduce meat intake by 50% by the year 2030 for health reasons, which would also reduce GHG emissions by 10^9 tons annually. By invoking health as a reason to change diets as opposed to ecological concerns, cities could reap double dividends of lower public-health burdens and a reduced foodprint. Studies have found that health is the most cited motivation for eschewing meat [35]. Cities interested in reducing their foodprint would be wise to take note of these findings in order to maximize the likelihood of galvanizing meaningful behavioral shifts in their populations.

Recently, cities have begun wading into the environmental impacts of meat. Numerous cities have signed up to the "Meatless Monday" initiative, promising to provide vegetarian menus at city-run facilities and promote the practice amongst the general population in a bid to reduce their foodprints [26, 38, 2]. A more ambitious example comes from the Italian city of Turin, where the mayor recently advocated that the city go vegan to save the environment [25, 2]. What such a scenario actually entails remains to be seen, but the broaching of a dialog surrounding the ecological responsibility of urbanites in wealthy, high meat consuming countries is a positive sign [29, 9]. What has previously been a "the forgotten climate change sector" by cities is now coming to the fore [3].

5.7 Chapter Conclusions

The edible city is not the only option for a city trying to reduce its foodprint. As shown here, changing diets to common non-meat patterns is predicted to meaningfully reduce the foodprint in multiple dimensions. Dietary shifts need not be so drastic as completely eschewing meat, since marginal decreases in total meat or adoption of novel protein sources could provide equal or greater foodprint reductions than the edible city.

Controlling the diet of residents is clearly not under the aegis of the cities in my study region. However, a city need not rule by fiat to reduce the demand for animal-sourced foods in its borders. Softer methods such as "Meat Free Mondays" or promoting healthy eating patterns that also align with environmental goals can have positive impacts on a city's foodprint while respecting the rights and freedoms of its dwellers.

Implementing the edible city and influencing diets are not mutually exclusive initiatives. An edible city with easy access to fruits and vegetables could very well be ideally situated to promote reduced meat or meat-free diets. However, given UA's low environmental returns per area cultivated, urban designers should keep in mind alternative uses for urban spaces if a primary design motive is environmental sustainability.

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Chapter 6

Conclusions

6.1 Summarizing the project outcomes

Chapters 2-5 attempted to address the questions that were laid out in the introduction. The general goal was to provide a glimpse of what the edible city's environmental performance might look like in the Northeast US. Although I have parceled the problem up into smaller, more digestible pieces, I hope that the common thread remained clear and that any detours were both necessary and kept to a minimum. Here I will revisit the outcomes of each research question and its sub-questions.

Research questions 1.1 and 1.2 focused on the urban foodprint. **Question 1.1** asked what was the scale of the urban environmental foodprint. Chapter two explored this with a review of available studies that quantified the foodprint using the techniques of industrial ecology. My review found that food consumption is often one of the more important drivers of a city's environmental burdens in terms of climate impacts and land use. Moreover, there is a tendency for the foodprint to increase alongside income, which foreshadows increasing food-related ecological burdens as many hitherto rural societies urbanize and become wealthier. **Question 1.2** dealt with the awareness of the foodprint by cities. From my cursory review, it seems that cities have been slow to tackle the foodprint, but the last few years have seen increasing acknowledgment of the issue and efforts to redress it. Many of the proposed solutions have been technology driven, including a push for local agriculture (UA), but as chapter 5 showed, changing diets has also recently emerged on the agenda.

Research questions 2.1-3 were centered on understanding urban agriculture (UA). **Question 2.1** asked what was known about UA's environmental characteristics. My review showed that there is a data gap in this domain and that many of the supposed benefits of UA are based on a mix of common sense and conjecture. Some evidence was found of UA having reduced embodied greenhouse gas emissions relative to conventional agriculture, but these findings were from mild climates, and were not applicable to my study region. **Question 2.2** asked what types of UA existed and if classification schemes were applicable to environmental assessments. A review of existing UA work showed that despite the plethora of attempts to document the different forms of UA, those schemes were predicated on social and economic aspects. In response I developed a simple schema that generated four UA forms based on the conditioning of grow space and interactions with the surrounding built form. **Question 2.3** tested the environmental performance of UA in the Northeast UA. Here I operationalized my UA taxonomy using primary data from urban farms in Boston and New York City, finding that conditioned UA forms are not suited for the climate given the underlying energy grid in the Northeast US. Burden shifting was found between indicators, whereby those farms with superior climate change impacts to conventional agriculture were at odds with land use and water consumption. As an application of urban land, UA paled in its climate change benefits relative to solar electricity generation.

Question 3 looked at the big picture, wondering how the edible city applied to Boston would alter the environmental performance of the city. Answering this first involved the development of a method to apply input-output life cycle assessment (LCA) to urban foodprints. Scaling up the UA assessments from chapter 3 and looking at changes to the baseline carbon and land foodprints showed that UA caused marginal benefits in the (<5% reduction) former and disservices in the latter (1% increase). Interactions between UA and the city were also found to be insubstantial, with the largest benefits being the ability to absorb 10% of Boston's household, solid organic waste. My findings suggest that at present the edible city is not well suited for the Northeast US as an exercise in environmental sustainability. Despite edible city's lackluster environmental gains, the concept does align with public health, and to a limited degree, economic motivations; filling nutritional gaps and providing a potential revenue source in my case city. All things considered, I suggest that UA be promoted largely under the auspices of educational, nutritional and community building agendas. Conversely, claims about environmental benefits in the Northeast US should be tempered until stronger evidence in their favor can be mustered.

Chapter 5 looked at how the urban foodprint could be tackled by alternative or concurrent activities to the edible city. Applying process-based LCA found that shifts to meat or animal-product free diets could result in carbon foodprint reductions that 16 to 34 times larger than those of the edible city in the US Northeast. Large savings in land use were also found. Despite these positive outcomes, changing diets on paper is a lot easier than in real life, where such diets are culturally ingrained and ossified through daily practices. An alternative is to meet people half way with products that mimic the sensory experience of preparing and eating meat. I tested the environmental impacts of one such product, Impossible Foods's Plant Based Burger, finding that the diffusion of this product at minor levels into the Bostonian diet resulted in predicted benefits equal to or beyond those of the edible city. Cities concerned about the foodprint would be wise to promote healthy eating habits as a 'sustainable trojan horse', since health is the primary motivator for reducing meat intake.

6.2 Future work

The environmental dimension of urban food consumption and the interlinkages between urbanization the foodprint are both two areas ripe for further investigation. Having toiled in the data and tried to make my own small contribution to the literature, I hope I have some insights into the direction that could help push the ball forward.

1. **Food consumption data:** Despite the minimal variation seen across demographic groups in my model of Boston's baseline foodprint, I firmly believe that future studies should still explore this avenue further. Even if the main foodprint drivers are consumed in similar levels by most demographics in the US, this does not preclude important findings at the intersection of public health and environmental studies. Unequal access to fresh fruits and vegetables, and higher prevalence to fast food consumption in poorer urban neighborhoods in the United States result in less-balanced diets and lifestyle related diseases [7]. Combining nutritional indices with LCA, as espoused by Heller Keoleian [5], would juxtapose nutritional quality and environmental impacts, showing how despite inordinate environmental burdens, such diets still fail to meet the nutritional needs of a consumer [10]. Such data can be sourced from the National Health and Nutrition Examination Survey, but as this is prone to underreporting of less healthy foods [1], primary data from field would be preferable. Another option is to mine social media for consumption trends, as has been done to estimate the nutritional content of meals using the photo sharing platform Instagram [3] and and other social media sources [8].

2. **Relating the foodprint to the built form:** Foodprints should be related to spatial configuration of the city. I made a coarse move in that direction with my mapping of the foodprint for Boston using census data. Similar methods could be employed that use the NHANES data, including ethnic breakdowns, combined with census data to develop a more nuanced picture of the foodprint. Ideally, data would be collected in neighborhoods and geo-tagged to allow for accurate cartographic representations. Maps of eating establishments and grocery stores taken from Google Maps or tax registries could be used to classify food sources (e.g fast food, fine dining, full-scale grocery store, corner store) and relate foodprint/nutritional content to proximate eating opportunities. This has already been done to a limited extent for obesity rates in Los Angeles neighborhoods [7], and expanding such work into the foodprint realm would make explicit the connections between form and foodprint that have been intimated by others [9].

3. **Better urban farm data:** Given more time I would have collected data from more farms over multiple years in order to improve the robustness of the UA assessments. Relying on data from one or two farms over a single growing season to represent the general performance of an UA form was a gross, but unfortunately, necessary simplification given budget and time constraints. Furthermore, primary data on water use efficiency, nutrient loading in runoff, agrichemical emissions, building energy interactions and crop pollutant uptake are needed to support future environmental assessments of UA. Studies would be wise to pursue partnerships with urban farms on the technological vanguard, to see if the constant improvements in the field can overcome the burdens of the background energy grid.

4. **UA Space:** Future research should also include models of potential contamination based on adjacent land uses. 3D models of shading effects would help anchor such estimates in reality.

5. **Dynamic modeling:** Harvest on urban farms comes in waves as the different crops mature. Future studies could incorporate these dynamics into the assessment, looking at the changing ability of the edible city to feed itself throughout the growing months.

6.3 Final thoughts

The original motivation of this project was not antipathy towards to the idea of the edible city, but a concern for the environmental sustainability of future cities. Given future demographic shifts, the only sustainable future for humanity will be a sustainable urban future. Great care needs to be taken in the design of new cities and evolution of existing cities so the city's latent potential as a sustainable built form is maximized and not squandered. The 20^{th} century was host to a litany of questionable urban planning ideas born into practice under the broad banner of 'modernism'. In a bid to rid the city of the squalor and vice that plagued 19^{th} cities and to make space for the automobile, many cities in the United States were sanitized of life and developed in a manner out of scale with human needs.

At the time, urban designers thought that they were the harbingers of a better urban future. One need only look at the 1939 World's Fair in New York, where Norman Geddes (and General Motors) presented the Futurama exhibition. Here they introduced to an eager public the glories of such new-fangled ideas as the limited access freeway, tower in a park and the highway rest station [4]. Under numerous names - "the radial city", "broadacre city", "the garden city" - a new urban form emerged that was sold as progress.

However, with 60 plus years of hindsight on our side, we can now see that this form of development - call it "sprawl", "suburbia", etc. - has contributed to the erosion of social ties wherever it has proliferated [6]. From an ecological perspective, 20^{th} century urban planning was responsible for the conversion of some of the United State's most fertile farmland to tract housing and a reliance on automobile transport with all its associated environmental ills [6].

Large scale socio-technological systems are 'sticky'[2]. That is, once in place, incumbent systems tend to be difficult to change, since people become normalized to their presence and perceive costs of moving away from the system as being too great. Urban designers and environmentalists have seen this with the spatial configurations that cities took on throughout the 20^{th} century: once a highway is built, it is very difficult to remove no matter the environmental or social costs of the infrastructure.

I am concerned that their might be some parallels between the unbridled enthusiasm for the edible city and the urban planning follies of the previous century. UA is espoused to be a way forward to a more harmonious relationship between city and nature, both within the city and as a result of its leaner supply chain. However, at least in a Northeast US context, the edible city is at best an infinitesimal environmental gain or at worst a step in the wrong direction environmentally. To pepper UA throughout the scarce free space available in Boston and similar cities in the name of environmental sustainability could lock those spaces into sub-optimal or even deleterious uses for decades to come. At the same time, given that some UA forms can be set up as quick temporary land uses, the edible city could be a more fluid urban form that is sensitive to evolving perspectives on the practice and needs of the host city.

Perhaps my framing has been too limited, with the outcome that I have inadvertently engaged in hyperbole. It is certainly reasonable to imagine that UA could be practiced alongside micro solar power generation or other land uses. This mixed-use scenario would let UA provide the social and health benefits that appear to be its strength, while providing space for other technologies that can make larger contributions to a city's environmental sustainability. Such a setup would also allow urbanites to engage in the act of cultivating food or simply witness this practice passively, and although maybe not completely suturing the metabolic rift, at least acting as a salve.

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Appendix A

Article 1: Surveying the Environmental Footprint of Urban Food Consumption

Surveying the environmental footprint of urban food consumption

B. Goldstein[1], M. Birkved[1], J. Fernandez[2], M. Hauschild[1]
[1]Department of Management Engineering, Technical University of Denmark
[2]Department of Architecture, Massachusetts Institute of Technology

Corresponding author: B. Goldstein, Department of Management Engineering, Technical University of Denmark, Produktionstorvet, Building 424, Kongens Lyngby, Denmark. Email: bgol@dtu.dk

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 paper 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, etc.) The foodprint was typically the 3rd largest source of mass flows (average – 0.8 ton/capita/annum) and carbon footprint (average – 1.9 tons CO2 equivalents/capita/annum) in the reviewed cities, while it was generally the largest driver of urban ecological footprints (average - 1.2 global hectares/capita/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). As 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 (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 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 which 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, 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). Broto and Bukleley's review of climate change mitigation interventions in 100 cities does not even contain the word 'food' (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, climate change impacts, biodiversity losses, eutrophication and non-renewable 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 since 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., 2007's 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 the 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 paper 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 is well situated to address this need, with its focus on the scale, nature and interconnections of material and energy exchanges between different socio-technical systems and the environment (Ferrão and Fernández 2013). It is from this discipline that the *urban metabolism* (UM) concept arose (Kennedy et al. 2007b).

UM applies industrial ecology principles to the geographic region (city, conurbation, 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 greenhouse gas (GHG) emissions and embodied water flows, respectively, while ecological footprinting (EF) quantifies the bioproductive area underpinning consumption and sequestration of CO_2 (Wackernagel 1998). Emergy accounts for embodied energy in UM flows (Stanhill 1977), while 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, as 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 each other 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 say little about the environmental impacts embodied within mass, though it can highlight deleterious exchanges between socio-technical systems and the ecosphere. CF provides both an indication of an actions contribution to society's largest environmental challenge, while 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.

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	Strengths: • Measures the amount of environmentally intensive foods demanded • Can map food waste

		Trade: balances of imported and exported foodstuffs at city, regional or national level	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 (IO): coupling of local food expenditures with environmentally extended IO tables to capture direct and inter-sectoral GHG flows	Strengths: • Quantifies GHG emissions embodied in food and identifies burdensome dietary choices Shortcomings: • Land use changes (LUC) 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 IO tables to capture direct and inter-sectoral 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 towards agriculture, potentially inflating foodprint relative to other UM drivers

Identification of Studies

The review began by isolating comprehensive literature reviews of UM studies. For UM, Decker et al.'s (2000), Kennedy et al.'s (2007b, 2011), Zhang's (2013) and Stewart et al.'s (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 grey literature document types were considered for inclusion (e.g. theses, reports, etc.) 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, etc.)

UM 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: (i) food flows were included in the study, (ii) the foodprint was separately presented or disaggregated using minimal manipulation (reducing risk of error and/or misinterpretation), (iii) 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 (iv) 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 approximately 132 foodprints. Figure 1 shows the geographic distribution of the foodprints considered, while tables S1-S3 in the supplementary material provides an overview of where they are used in the meta analysis.

Some data pruning was performed prior to the analysis of the foodprints. Li et al.'s (2013) CF of Macao from 2005-2009 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 et al.'s (2014) and Niza et al.'s (2009) MFA of Lisbon from 2003-2009 were also averaged due to 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 GDP data was available (see table S6). For the four studies for which averages were taken, no large changes in consumptive patterns or foodprints were seen 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 seen, 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. Lastly, the different methodologies outlined in table 1 were encountered for all the three indicators.

Tables S1-S3 in the supplementary material provides an overview of the included studies their data sources and methodologies. OECD Statistics (2015) provided much of the GDP data that was used in the analysis, but where these were lacking tables S4-S6 outline estimation methods.

Results - The Urban Foodprint

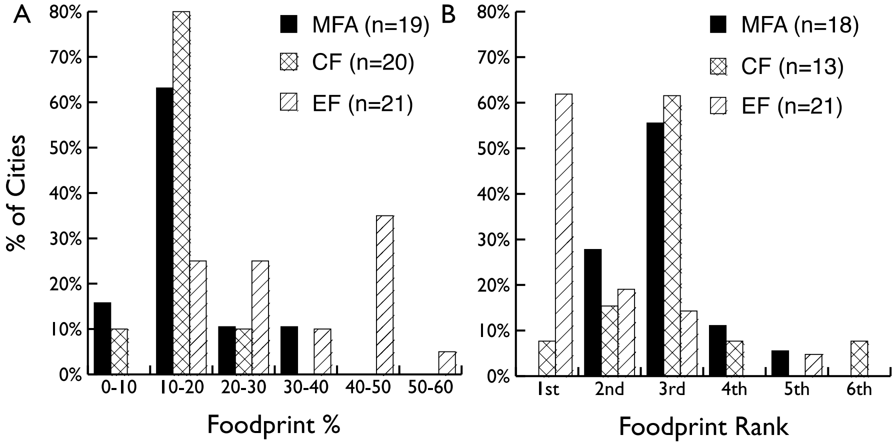

Figure 2 – Importance of the foodprint in the urban metabolic profile of the reviewed cities: a) percentage of cities with foodprint impacts as a distinctive fraction of total impacts b) Histogram of foodprint's rank compared to other main urban metabolic categories (e.g. transport, building energy, etc.) 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 supplementary material Table S1-S3 for clarification.

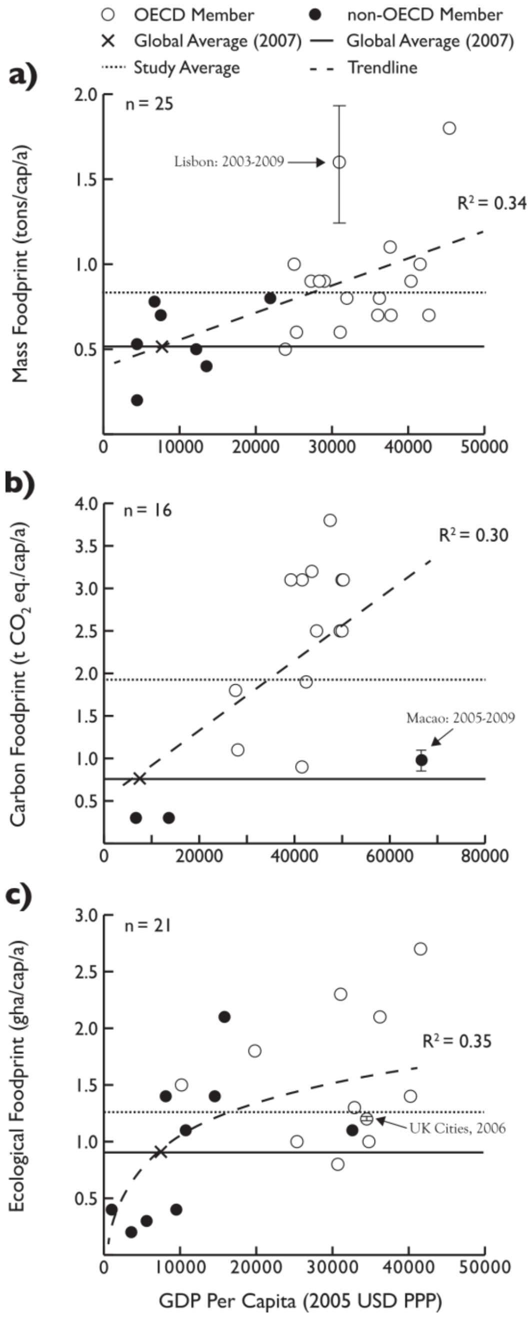

Figure 3 – The urban foodprint vs GDP per capita with foodprint in terms of: a) mass b) ecological footprint c) carbon footprint. Sample size disagrees with Figure 2 since additional studies that only included food flows are now included.

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 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, while its CF and MFA pressures are significant, but less intense. Food production is actually estimated to contribute 24-50% of global greenhouse gas emissions (IPCC 2014b; Schmidt and Merciai 2014) which hints that the reviewed foodprints may be underestimated since most of the observed carbon foodprints fall below this range. Looking at the CF methods in table S5 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 N_2O). GHG emissions data on the latter is 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 as 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 3A shows a scatter plot of mass foodprints (determined by MFA) versus per capita GDP, with detailed data in Table S5 in the supplementary material. The average per capita annual mass foodprint for the studies is approximately 0.8±0.3 ton/annum (t/cap/a – where ton 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=0.34). 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 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.

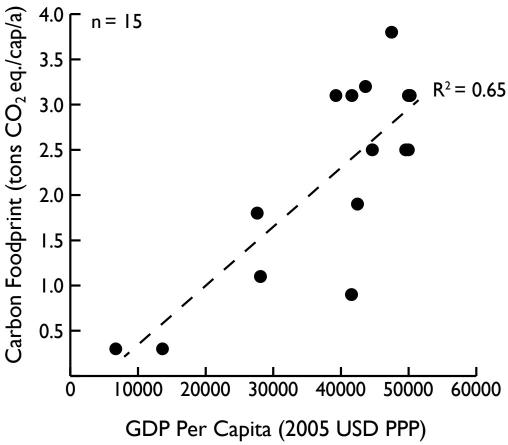

Figure 4 – CF vs. GDP with Macao removed from the data set.

Paris's foodprint represented 36% of total regional material consumption since it is a dense, mature city with high non-durable goods consumption, while Limerick's foodprint was only 4% due to a metabolism defined by large construction aggregate additions to stock. The largest mass foodprints (Paris; 1.8 t/cap/a, Lisbon; 1.4-2 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 also show how food demands can fluctuate across years. Nonetheless, the Paris and Lisbon studies suggest that a number 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 supplementary material Table S6). Average per capita annual carbon foodprint was 2.3 t CO_2 eq./cap/a, representing a carbon intensity of 2.8 t CO_2 eq./t urban food demand. Similar to the MFA assessment, a modest relationship is seen between income and carbon foodprint (R^2=0.30). Though the non-OECD countries generally perform lower, this is 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 (US 2005-2009 average – beef; 41 kg/cap/a, dairy; 135 kg/cap/a) (FAO

2014). These differences strongly affect the carbon foodprint since 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 an 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 towards 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, since 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 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 10 000 USD. 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 UNEP (2012) work showing that per capita meat consumption follows a logarithmic trend that saturates around 10 000 USD for national populations. The modest correlation also means that other factors contribute to the EF. Comparative regional market advantage can make environmentally 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 20-30% 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, CN and Kawasaki, JP, where the majority of both cities' EFs originate from industrial energy consumption (Geng et al. 2014).

Discussion

The importance of the foodprint's 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. IO 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 due to 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 S8 and S9 test the effect of the application of IO and process based methodologies on the carbon and ecological foodprints, respectively (not applicable to the included mass foodprints). The IO method shows a tendency to be higher than process-based carbon foodprint methods for cities of high incomes (no low income IO 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 S10-S12 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 S13-S15 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 Lisbon studies which had noticeably higher mass foodprints. Most importantly, 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 since 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. Lastly, that the majority of foodprints included represent middle- and high-income cities, which may skew the observations upwards 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 et al.'s (2012) study of Beijing household food consumption, which identified food preparation as the largest contributor to the foodprint (60%), likely due to Beijing's fossil fuel dominated energy production. Goldstein et al.'s (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 due to 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 towards processed, high-energy density foodstuffs (Popkin 2006; Tilman and Clark 2014).

Foodprint Form

MFA and nutrient balance literature (see supplementary material S4) revealed a linear foodprint, in line with the general observations of UM studies and other socio-economic systems (Kennedy et al. 2010; 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 components are symbiotic (one sub-system'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 in to 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

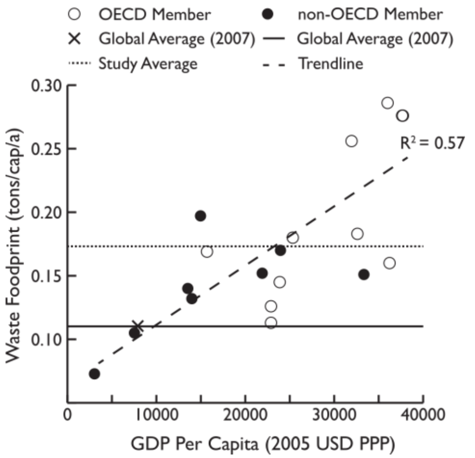

Figure 5 – Per capita waste foodprint in tons/annum (t/cap/a) as a function of per capita income

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 supplementary literature) 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 (IPCC, 2014c) 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, since 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 if 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 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 nor the apartments that make up a large portion of the housing stock (Forkes 2007), while Paris's 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 due 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. 1997); 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 non-starter in most societies due 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 19th 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 a hundred years a 3000 ha wetlands has process 550 000 m^3 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 (Carlisle 2013).

Because of the risk of pathogens in nutrients mined from human waste a multi-forked 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 towards cyclical UM most avoid the pitfalls of focusing on single waste streams, since this increases the potential for ignoring key food related flows and reduces the environmental efficacy of these strategies (Kalmykova et al. 2012). Furthermore, cyclical UM remains a challenge since nutrients embedded in food imports represent a fraction of the nutrients used in production, since 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 et al. (2012) documented the symbiotic relationships between the urban dairies in the Phoenix Metropolitan Area and alfalfa farmers which used waste from the dairies and bio-solids 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. Due 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). 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 et al.'s (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 if the urban foodprint shares this property.

Urbanization also affects consumption patterns and household food management practices. Figures 3A and 3C show 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 towards 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 seen for increasing GDP is most likely caused by household food management practices and shifting consumption patterns towards expensive food items with larger environmental burdens.

Linkages between economic development and increasing intake of high-burden foods by others support this (Tillman et al., 2014). Recent 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 US 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.

Lastly, the spatial characteristic of urban development has an effect on the foodprint, since low-density growth potentially consumes productive agricultural land at the per-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 non-trivial, 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 multi-metric approaches to gauge environmental impacts, since 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 co-mingled urbanization and economic development, though a number of other important factors assert influence (cultural preferences, lower prices, etc.) 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 multi-scale 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. Since 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.

About the authors

Benjamin Goldstein is a PhD candidate at the Technical University of Denmark in Kongens Lyngby, Denmark. Morten Birkved is an associate professor at the Technical University of Denmark in Kongens Lyngby, Denmark. John Fernandez is a Professor at the Massachusetts Institute of Technology in Cambridge, MA, USA. Michael Hauschild is a professor at the Technical University of Denmark in Kongens Lyngby, Denmark.

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Table S1 - MFA Studies						
Study	**City**	**Year**	**In Figure 2A**	**In Figure 2B**	**Figure 3A**	**Reason for Discrepancy**
Warren-Rhodes and Koenig (2001)	Hong Kong	1971	1	1	1	
Warren-Rhodes and Koenig (2001)	Hong Kong	1997	1	1	1	
Browne et al. (2009)	Limerick	1996	1	1	1	
Browne et al. (2009)	Limerick	2002	1	1	1	
Rosado et al. (2014)	Lisbon	2004	1	1	0	In average
Niza et al. (2009)	Lisbon	2003	1	1	0	In average
		2004	1	1	0	In average
		2005	1	1	0	In average
		2006	1	1	0	In average
		2007	1	1	0	In average
		2008	1	1	0	In average
		2009	1	1	0	In average
Niza et al. (2009) and Rosado (2014)	Lisbon Average	2002 - 2009	0	0	1	
Barles (2009)	Paris	2006	1	0	1	MFA was not broken down into different consumption categories
Swilling (2006)	Cape Town	2006	1	1	1	
Sahely (2003)	Toronto	1987	0	0	1	
Sahely (2003)	Toronto	1999	0	0	1	
Pina and Martinez (2013)	Bogota	2010	1	1	1	
Barrett (2002)	York	2000	1	1	1	
Emenegger (2002)	Geneva	2000	1	1	1	
Moore (2014)	Vancouver	2013	1	1	1	
Best Foot Forward (2002)	London	2000	1	1	1	
Ngo and Pataki (2008)	Los Angeles	1990	0	0	1	MFA only covered water and food
Ngo and Pataki (2008)	Los Angeles	2000	0	0	1	MFA only covered water and food
Metzger (2013)	Durham	2000	0	0	1	MFA was not broken down into different consumption categories
Forkes (2007)	Toronto	1990	0	0	1	MFA only covered food
Forkes (2007)	Toronto	2001	0	0	1	MFA only covered food
Forkes (2007)	Toronto	2004	0	0	1	MFA only covered food
Newman (2000)	Melbourne	1990	0	0	1	MFA was not broken down into different consumption

						categories
Codoban (2008)	Toronto	2008	0	0	1	MFA was not broken down into different consumption categories
Chavez (2012)	Delhi, IN	2009	0	0	1	Complete MFA not published, but food consumption given
Reddy (2013)	Mumbai, IN	2010	0	0	1	Masses of fuels not presented.
Hoornweg (2012)	Manila, PH	2010	0	0	1	Assessment not broken down into clear UM drivers
Total	-	-	19	18	25	

Table S1 - Included mass foodprint studies

Table S2 - CF Studies						
Study	**City**	**Year**	**In Figure 2A**	**In Figure 2B**	**Figure 3B**	**Reason for Discrepancy**
Ramaswami (2008)	Denver	2005	1	1	1	
Wu (2011)	Beijing	2006	1	1	1	
Heinonen (2011)	Helsinki	2006	1	0	1	Food not disaggregated into own consumption category
Chavez (2012)	Delhi	2009	1	0	1	CF was not broken down into different consumption categories
Hillman (2010)	Colorado	2000	1	1	1	
	Boulder	2000	1	1	1	
	Fort Worth	2000	1	1	1	
	Arvada	2000	1	1	1	
	Portland	2000	1	1	1	
	Seattle	2000	1	1	1	
	Minneapolis	2000	1	1	1	
	Austin	2000	1	1	1	
Dias (2014)	Aveiro	2005	1	1	1	
Cardiff Council (2001)	Cardiff	2001	1	1	1	
Best Foot Forward (2002)	London	2000	1	1	1	

Li (2013)	Macao	2005	1	0	1	CF was not broken down into different consumption categories
		2006	1	0	1	
		2007	1	0	1	
		2008	1	0	1	
		2009	1	0	1	
			20	13	20	

Table S2 - Included MFA foodprint studies

Table S3 - EF Studies						
Study	**City**	**Year**	**In Figure 2A**	**In Figure 2B**	**Figure 3C**	**Reason for Discrepancy**
Klinksy et al. (2009)	Montreal	Early 2000 s	1	1	1	
Tavallai (2009)	Tehran	2005	1	1	1	
Zhang et al. (2013)	Banqiao, Keifing City	2009	1	1	1	
	Kangping, Keifing City, CN	2009	1	1	1	
	Longcheng xiangxieli garden, Keifing City	2009	1	1	1	
Walsh et al. (2010)	Limerick	2002	1	1	1	
	Belfast	2001	1	1	1	
Wackernagel (1998)	Santiago	1993	1	1	1	
Kissinger and Haim (2008)	Ra'anana	2002	1	1	1	
Moore et al. (2013)	Vancouver	2006	1	1	1	
Hubacek et al. (2009)	Beijing	2001	1	1	1	
Razack and Ludin (2014)	Minna	2012	1	1	1	
WWF (2007)	Newport	Mid 2000 s	0	0	0	Lack of GDP data
	Plymouth	Mid 2000 s	0	0	0	Lack of GDP data
	Salisbury	Mid 2000 s	0	0	0	Lack of GDP data
	Kingston upon Hull	Mid 2000 s	0	0	0	Lack of GDP data
	Stoke on Trent	Mid 2000 s	0	0	0	Lack of GDP data
	Gloucester	Mid 2000 s	0	0	0	Lack of GDP data
	Wakefield	Mid 2000 s	0	0	0	Lack of GDP data
	Sunderland	Mid 2000 s	0	0	0	Lack of GDP data
	Truro (Carrick)	Mid 2000 s	0	0	0	Lack of GDP data
	Wolverhampton	Mid 2000 s	0	0	0	Lack of GDP data
	Salford	Mid 2000	0	0	0	Lack of GDP data

		s				
	Swansea	Mid 2000s	0	0	0	Lack of GDP data
	Coventry	Mid 2000s	0	0	0	Lack of GDP data
	Exeter	Mid 2000s	0	0	0	Lack of GDP data
	Cardiff	Mid 2000s	0	0	0	Included in average
	Glasgow	Mid 2000s	0	0	0	Included in average
	Bradford	Mid 2000s	0	0	0	Lack of GDP data
	Lincoln	Mid 2000s	0	0	0	Lack of GDP data
	Birmingham	Mid 2000s	0	0	0	Lack of GDP data
	Bristol	Mid 2000s	0	0	0	Included in average
	Liverpool	Mid 2000s	0	0	0	Included in average
	Nottingham	Mid 2000s	0	0	0	Lack of GDP data
	St Davids (Pembrokeshire)	Mid 2000s	0	0	0	Lack of GDP data
	Bangor (Gwynedd)	Mid 2000s	0	0	0	Lack of GDP data
	Worcester	Mid 2000s	0	0	0	Lack of GDP data
	Leicester	Mid 2000s	0	0	0	Lack of GDP data
	Carlisle	Mid 2000s	0	0	0	Lack of GDP data
	Derby	Mid 2000s	0	0	0	Lack of GDP data
	Sheffield	Mid 2000s	0	0	0	Lack of GDP data
	York	Mid 2000s	0	0	0	Lack of GDP data
	Leeds	Mid 2000	0	0	0	Lack of GDP data

		s				
	Dundee City	Mid 2000 s	0	0	0	Lack of GDP data
	Peterborough	Mid 2000 s	0	0	0	Lack of GDP data
	Norwich	Mid 2000 s	0	0	0	Lack of GDP data
	Preston	Mid 2000 s	0	0	0	Lack of GDP data
	Inverness	Mid 2000 s	0	0	0	Lack of GDP data
	Manchester	Mid 2000 s	0	0	0	Included in average
	Ripon	Mid 2000 s	0	0	0	Lack of GDP data
	Bath	Mid 2000 s	0	0	0	Lack of GDP data
	Weels	Mid 2000 s	0	0	0	Lack of GDP data
	Newcastle upon Tyne	Mid 2000 s	0	0	0	Lack of GDP data
	Lancaster	Mid 2000 s	0	0	0	Lack of GDP data
	Lichfield	Mid 2000 s	0	0	0	Lack of GDP data
	London	Mid 2000 s	0	0	0	Included in average
	Stirling	Mid 2000 s	0	0	0	Lack of GDP data
	Hereford	Mid 2000 s	0	0	0	Lack of GDP data
	Ely (East Cambs)	Mid 2000 s	0	0	0	Lack of GDP data
	Aberdeen	Mid 2000 s	0	0	0	Included in average
	Chester	Mid 2000 s	0	0	0	Lack of GDP data
	Edinburgh	Mid 2000 s	0	0	0	Included in average
	Portsmouth	Mid 2000	0	0	0	Lack of GDP data

		s				
	Cambridge	Mid 2000 s	0	0	0	Lack of GDP data
	Durham	Mid 2000 s	0	0	0	Lack of GDP data
	Southampton	Mid 2000 s	0	0	0	Lack of GDP data
	Oxford	Mid 2000 s	0	0	0	Lack of GDP data
	Canterbury	Mid 2000 s	0	0	0	Lack of GDP data
	Brighton and Hove	Mid 2000 s	0	0	0	Lack of GDP data
	Chichester	Mid 2000 s	0	0	0	Lack of GDP data
	St. Albans	Mid 2000 s	0	0	0	Lack of GDP data
	Winchester	Mid 2000 s	0	0	0	Lack of GDP data
	Average	Mid 2000 s	1	1	1	
WWF (2012)	Campo Grande	2008	1	1	1	
Best Foot Forward Ltd. (2002)	London	2000	1	1	1	
WWF (2012)	Sao Paulo	2011	1	1	1	
Global Footprint Network (2011)	San Francisco-Oakland-Fremont	2007	1	1	1	
WWF (2013)	Hong Kong	2011	1	1	1	
Cardiff Council (2005)	Cardiff	2001	1	1	1	
Barrett (2002)	York	2000	1	1	1	
Global Footprint Network (2010)	Curitiba	2006	1	1	1	
			21	21	21	

Table S3 - Included EF foodprint studies

Table S4 - Mass foodprints								
Study	**City**	**Year**	**Unit of Analysis**	**Food Data**	**Boundaries**	**t/cap/a (% of ciites CF) [rank as UM driver]**	**Per capita GDP (2005 USD PPP)**	**GDP Data**
				OECD				
Codoban and Kennedy (2008)	Toronto, CA	2008	Household	National statistics scaled to population	Ostensibly includes residential and commercial	1.1 (-) [-]	36004	Taken as 2005 USD PPP from OECD (2015)
Ngo and Pataki (2008)	Los Angeles, US	1990	Household	Scaled from national data (LA has similar household income to national)	Ostensibly includes residential and commercial	0.8 (-) [-]	31950	Per capita GDP taken from California Economic Forecast (2011) in 2010 USD and adjusted for inflation using World Bank (2014)
		2000				0.9 (-) [-]	40364	Taken as 2005 USD PPP from OECD (2015)
Browne et al. (2009)	Limerick, IE	1996	City	National statistics scaled to Limerick based on ratio of city to national average weekly expenditures	Residential and commercial	0.5 (5%) [3]	23872	Same as in Table 2, but adjusted for an assumed 1% growth rate
		2002				0.6 (4%) [3]	25340	See Table 2
Niza et al. (2009)	Lisbon, PT	2004	City	Metropolitan balance	Residential and commerc	2.8 (14%) [2]	30827	Taken as 2005 USD

				using trade statistics	ial			PPP from OECD (2015)
Rosado et al. (2014)	Lisbon, PT	2003	City	Metropolitan balance using trade statistics, including feed biomass for livestock.	Residential and commercial	1.3 (16%) [4]	30286	Taken as 2005 USD PPP from OECD (2015)
		2004				1.7 (16%) [3]	30827	Taken as 2005 USD PPP from OECD (2015)
		2005				1.7 (16%) [3]	30983	Taken as 2005 USD PPP from OECD (2015)
		2006				1.5 (17%) [4]	31052	Taken as 2005 USD PPP from OECD (2015)
		2007				1.8 (18%) [3]	31628	Taken as 2005 USD PPP from OECD (2015)
		2008				1.5 (19%) [2]	31508	Taken as 2005 USD PPP from OECD (2015)
		2009				1.4 (18%) [2]	30549	Taken as 2005 USD PPP from OECD (2015)
	Average					1.6 ± 0.3	30958 ± 449	
Barles (2009)	Paris, FR	2003	City	Metropolitan balance using trade statistics	Residential and commercial	1.8 (81%) [-]	45407	Taken as 2005 USD PPP from OECD (2015)
Newman (1999)	Melbourne, AU	1990	Unknown	Unknown	Unknown	1.0 (-) [-]	25037	Taken as national average from World Bank (2014)

Sahely et al. (2003)	Toronto, CA	1987	City	National statistics scaled to population	Residential and commercial	0.9 (49%) [1]	27248	Taken from article, adjusted for inflation with World Bank (2014) and converted to PPP with World Bank (2014)
		1999				0.9 (46%) [1]	28995	See above
Barrett et al. (2002)	York, UK	2000	City	Regional (Yorkshire and Humberside) food data from National Survey	Residential and commercial	0.6 (5%) [5]	31072	See Table 2
Emmenegger and Frischnecht (2003)	Geneva, CH	2000	City	Canton level statistics	Residential and commercial	0.7 (16%) [3]	37742	Same method as Lisbon, PT in 2003, but with an annual growth in GDP of 2% assumed to backcast from 2002 to 2000
Metzger (2013)	Durham-Wake-Orange Counties, US	2000	City	National statistics scaled to population	Residential and commercial	0.7 (-) [-]	42717	Taken as 2005 USD PPP from OECD (2015)
Moore et al. (2013)	Vancouver, CA	2006	Household	Scaled from national data	Residential and commercial	0.8 (25%) [3]	36224	Taken as 2005 USD PPP from OECD (2015)
Best Foot Forward Ltd. (2002)	London, UK	2000	City	City level trade statistics	Residential and commercial	1.0 (15%) [2]	41557	Taken as 2005 USD PPP from

								OECD (2015)
Forkes (2007)	Toronto, CA	1990	City	National statistics scaled to population	Residential and commercial	0.9 (-) [-]	28349	Adjusted from 1987 with a 2% assumed real growth rate
		2001				1.1 (-) [-]	37634	Taken as 2005 USD PPP from OECD (2015)
		2004				1.1 (-)] [-]	37725	Taken as 2005 USD PPP from OECD (2015)
non-OECD								
Warren-Rhodes and Koenig (2001)	Hong Kong, CN	1971	City	City level statistics	Residential and commercial	0.7 (20%) [3]	7535	GDP in 1971 taken in 2013 USD from UN Data (2015), converted to RMB from World Bank (2014), adjusted for inflation to 2005 (2014d) and converted to PPP using World Bank (2014)
		1997				0.8 (10%) [3]	21904	Taken from World Bank (2014)
Pina and Martinez (2013)	Bogota, CO	2010	City	City level statistics	Residential and commercial	0.4 (32%) [2]	13533	Taken as 2005 USD PPP from OECD (2015)

Swilling (2006)	Cape Town, ZA	2006	Household	Unknown	Unknown	0.5 (14%) [3]	12183	Per capita taken from OECD (2008) in 2006 USD PPP, converted to ZAR from World Bank (2014), adjusted for inflation from World Bank (2014), and converted to PPP using (2014b)
Chavez (2012)	Delhi, IN	2009	Household	FAO scaled to city level by population	Residential	0.8 (-) [-]	6712	Taken as 2005 USD PPP from OECD (2015)
Reddy (2013)	Mumbai, IN	2010	Household	Maharashtra state level for urban residents	Residential	0.5 (-) [-]	4413	Taken as the same as Maharashtra State per capita GDP in 2005 PPP USD from OECD (2015)
Hoornweg (2012)	Manila, PH	2010	Household	City level statistics	Residential	0.2 (-) [-]	4424	Country GDP taken in 2005 USD PPP from World Bank (2014), scaled to the región from Brooking

								s Institute (2010) and then divided by population from Hoornweg et al. (2012)
Study Average	-	-	-	-	-	1.0 ± 0.5	27988 ± 11190	-
Global Average	-	2007		-	Global average taken from Schmidtt (2014)	0.5	7585	Taken from FAO (2014) as global GDP in 2005 PPP divided by global population

Table S4 - Mass foodprints

Table S5 - CF Studies								
Study	**City**	**Year**	**Unit of Analysis**	**Food Data**	**Method and Boundaries**	**t CO2 eq/cap/a (% of ciites CF) [rank as UM driver]**	**Per capita GDP (2005 USD PPP)**	**GDP Data**
OECD								
Ramaswami et al. (2008)	Denver, USA	2005	City	County household expenditure survey	EIO model at sector level - farm to fork. No LUC	2.5 (10%) [4]	49570	Taken as 2005 USD PPP from OECD (2015)
Heinonen et al. (2011)	Helsinki, FIN	2006	Household	Regional household consumption survey	EIO model at sector level - farm to fork. No LUC	1.9 (13%) [-]	42442	Taken as 2005 USD PPP from OECD (2015)
Hillman and Ramaswami (2010)	Denver, US	Early 2000s	City	City specific consumer expenditure surveys	EIO model at sector level - farm to fork. No LUC	3.1 (13%) [3]	49967	Taken as 2005 USD PPP from OECD (2015)
	Boulder, US					3.1 (13%) [3]	50222	2002 Per capita GDP taken in 2009 USD from U.S. Department of Commerce (2014) and adjusted for inflation from World Bank (2014)
	Fort Worth, US					3.1 (17%) [3]	41609	Taken as 2005 USD PPP from OECD (2015)
	Arvada, US					2.5 (16%) [3]	49967	Assumed to be the same as Denver since

								Arvada is a wealthy portion of the Denver agglomeration.
	Portland, US					3.1 (18%) [3]	39249	Taken as 2005 USD PPP from OECD (2015)
	Seattle, US					3.8 (12%) [3]	47484	Taken as 2005 USD PPP from OECD (2015)
	Minneapolis, US					3.2 (12%) [3]	43632	Taken as 2005 USD PPP from OECD (2015)
	Austin, US					2.5 (14%) [3]	44635	Taken as 2005 USD PPP from OECD (2015)
Dias (2014)	Aveiro, PT	2005	Household	National consumer expenditure survey scaled to area based on age group	EIO model at sector level - farm to fork. No LUC	1.8 (20%) [2]	27624	Taken as the national per capita GDP from World Bank (2014) and adjusted for purchasing power with World Bank (2014)
Cardiff Council (2005)	Cardiff, UK	2001	City	Consumer expenditure data at the 'local level'	Product level embodied energy in manufacturing and transport. No LUC	1.1 (9%) [6]	28095	Taken as 2005 USD PPP from OECD (2015)
Best Foot Forward Ltd. (2002)	London, UK	2000	City	City level trade statistics	Product level embodied energy in manufacturing and transport. No LUC	0.9 (14%) [2]	41557	Taken as 2005 USD PPP from OECD (2015)
non-OECD								

Li et al. (2013)	Macao, CN	2005	City	City level statistics from Yearbook of Statistics	MRIO model (China and ROW) at sector level - farm to fork. No LUC	0.8 (12%) [-]	57088	Per capita GDP taken in 2005 USD from World Bank (2014), converted to RMB using World Bank (2014) and then converted to PPP using World Bank (2014).
		2006				0.9 (9%) [-]	63733	
		2007				1.0 (11%) [-]	70887	
		2008				1.0 (11%) [-]	71223	
		2009				1.2 (22%) [-]	70484	
		2005-2009			Average	1.0 ± 0.1	66683 ± 6195	
Wu et al. (2011)	Beijing, CN	2006	Household	City level statistics from Yearbook of Statistics	Product level process LCA - cradle to grave. No LUC	0.3 (23%) [1]	13617	Taken as 2005 USD PPP from OECD (2015)
Chavez (2012)	Delhi, IN	2009	Household	FAO scaled to city level by population	Product level LCA of unknown type	0.16 (20%) [2]	6712	Taken as 2005 USD PPP from OECD (2015)
Study Average	-	-	-	-		1.9 ± 1	45490 ± 16892	-
Global Average	-	2007		-	Global agricultural emissions from 2007 with global population from 2007 both from FAO (2014)	0.9	7585	Taken from FAO (2014) as global GDP in 2005 PPP divided by global population.

Table S5 - Carbon foodprints

Table S6 - EF Studies								
Study	**City**	**Year**	**Unit of Analysis**	**Food Data**	**Method and Boundaries**	**gha/cap/a (% of ciites CF) [rank as UM driver]**	**Per capita GDP (2005 USD PPP)**	**GDP Data**
OECD								
Klinksy et al. (2009)	Montreal, CA	Early 2000s	Household	National level per capita data scaled to region	Generic EF factor for Canada. Likely only includes direct agricultural land occuptation. Component method.	0.8 (12%) [5]	30700	Taken as 2005 USD PPP from OECD (2015)
Walsh et al. (2010)	Limerick, IE	2002	City	Average per capita Irish data from EU expenditure report	Land to sequester embodied energy and direct land use accounted. Hybrid compound - component.	1.0 (16%) [3]	25340	Per capita GDP assumed to be same as regional average. Regional GDP and population taken in from The Irish Regions Office (2014) in 2002 euros. Inflation adjusted using World Bank (2014) and convert

								ed to PPP using World Bank (2014)
	Belfast, IE	2001		Regional data for North Ireland used		1.4 (16%) [3]	40232	GDP taken from Belfast City Council (2004) 2001 euros, adjusted for inflation using World Bank (2014) and converted to PPP using World Bank (2014)
Wackernagel (1998)	Santiago, CL	1993	Household	Scaled from national data	Land to sequester embodied energy and direct land use accounted. Compound.	1.5 (55%) [1]	10186	Taken as 2005 USD PPP from OECD (2015) for year 2000 and adjusted by 2% annual growth rate to 1993 University of Ontario Institute of Technology (Unknown Date)

Kissinger and Haim (2008)	Ra'anana, IL	2002	House hold	City level consumption data	Land to sequester embodied energy and direct land use accounted. Compononent.	1.8 (45%) [1]	19807	Assumed same as Israeli. Per capita GDP taken from World Bank (2014)
Moore et al. (2013)	Vancouver, CA	2006	House hold	Scaled from national data	Land to sequester embodied energy and direct land use accounted. Component.	2.1 (45%) [1]	36224	Taken as 2005 USD PPP from OECD (2015)
WWF (2007)	Cardiff, UK	Mod 2000s	House hold	Unknown data source	Land to sequester embodied energy and direct land use accounted. Hybrid compound-component.	1.2 (23%) [1]	28095	Taken as 2005 USD PPP from OECD (2015)
	Glasgow, UK					1.1 (22%) [1]	32885	Taken as 2005 USD PPP from OECD (2015)
	Bristol, UK					1.2 (22%) [1]	34897	Taken as 2005 USD PPP from OECD (2015)
	Liverpool, UK					1.2 (23%) [1]	24974	Taken as 2005 USD PPP from OECD (2015)
	Manchester, UK					1.2 (22%) [1]	34684	Taken as 2005 USD PPP from

								OECD (2015)
	London, UK					1.3 (24%) [1]	42162	Taken as 2005 USD PPP from OECD (2015)
	Aberdeen, UK					1.2 (21%) [1]	40286	Taken as 2005 USD PPP from OECD (2015)
	Edinburgh, UK					1.2 (22%) [1]	37902	Taken as 2005 USD PPP from OECD (2015)
	Average					1.2 ± 0.05 (22) [1]	34485 ± 5835	
Best Foot Forward Ltd. (2002)	London, UK	2000	City	City level trade statistics	Product level embodied energy in manufacturing and transport. Component.	2.7 (41%) [2]	41557.04	Taken as 2005 USD PPP from OECD (2015)
Global Footprint Network (2011)	San Francisco-Oakland-Fremont, US	2007	Household	City level expenditure data	EIO model with US economy at sector level - farm to fork. Compound.	1.0 (13%) [1]	34780	GDP taken from report in 2007 USD and adjusted for inflation using World Bank (2014)
Cardiff Council (2005)	Cardiff, UK	2001	Household	Consumer expenditure data at the 'local level'	EIO model with UK economy at sector level - farm to fork.	1.3 (24%) [1]	32885	See above table

					Hybdrid compound-component.			
Barrett (2002)	York, UK	2000	Household	Regional (Yorkshire and Humberside) food data from National Survey	Embodied energy for production, transport, waste management. Sequestration of CO_2 equivalents from cattle. Direct land use. Component.	2.3 (32%) [1]	31072	2007 per capita GDP taken from OECD (2011) in 2000 USD PPP. Converted to GBP with World Bank (2014), adjusted for inflation with World Bank (2014), corrected to 2001 using an assumed 2% growth rate and converted back to PPP using (IBID)
non-OECD								
Global Footprint Network (2010)	Curitiba, BR	2006	Unknown	Unknown	Component	1.4 (41%) [1]	14555	Taken in 2007 USD from City of Curitiba (2014) and converted to Real using World Bank

								(2014), adjusted for a year of growth assuming a 3% growth rate, adjusted for inflation using World Bank (2014) and converted to PPP using World Bank (2014)
Tavallai (2009)	Tehran, IR	2005	City	Unknown	Land to sequester embodied energy and direct land use accounted. Component.	0.9 (24%) [2]	13990	Metro GDP and population taken from PWC (2009) in 2008 PPP USD, converted to Rial from World Bank (2014), adjusted for growth at assumed 1.5% rate, inflation taken from World Bank (2014) then adjusted for

								PPP using World Bank (2014)
WWF (2013)	Hong Kong, CN	2011	City	City level consumption data	Land to sequester embodied energy and direct land use accounted. Component.	1.1 (23%) [1]	32608	World Bank (2014)
WWF (2012)	Sao Paulo, BR	2011	Household	Household budget survey at the Sao Paulo State level	Land to sequester embodied energy and direct land use accounted. MRIO compound.	2.1 (49%) [1]	15832	Taken as 2005 USD PPP from OECD (2015)
WWF (2012)	Campo Grande, BR	2008	Household	Household budget survey from the State of Mato Grosso adpated to Campo Grande	Land to sequester embodied energy and direct land use accounted, based on land use per sector. Compound.	1.4 (45%) [1]	8122	Per capita GDP taken from IBGE (2014) in 2006 Real, adjusted to 2008b using 4% assumed growth rate and inflation rate taken from World Bank (2014) and then converted to PPP using World Bank (2014)

Hubacek et al. (2009)	Beijing, CN	2001	Household	Province level economic IO table	EIO model with US economy at sector level - farm to fork. Compound.	1.1 (22%) [3]	10749	Taken as 2005 USD PPP from OECD (2015) for 2003 and adjusted for 2 years growth at assumed 7%
Zhang et al. (2013)	Banqiao, Keifing City, CN	2009	Household	Neighborhood level primary data	Land to sequester embodied energy and direct land use accounted based on product level. Component.	0.2 (42%) [1]	3698	Per capita GDP taken from China Knowledge (2013) in RMB, adjusted for growth assuming 8% rate, adjusted for inflation using World Bank (2014) and converted to PPP using World Bank (2014)
	Kangping, Keifing City, CN					0.3 (28%) [2]	5612	Same as above
	Longcheng xiangxieli garden, Keifing City, CN					0.4 (16%) [2]	9508	Same as above

Razack and Ludin (2014)	Minna, NI	2012	House hold	City level primary data	Land to sequester embodied energy and direct land use accounted. Compound.	0.4 (39%) [1]	1030	Taken as national average from World Bank (2014)
Study Average	-	-	-	-	-	1.2 ± 0.6	21570 ± 13140	-
Global Average	-	2007		-	EF taken from WWF (2010). GDP per capita same as above table.	0.9	7585	Taken from FAO (2014) as global GDP in 2005 PPP divided by global population.

Table S6 - Ecological foodprints

Table S7 - Urban Food Waste							
Study	**City**	**Year**	**Unit of Analysis**	**Waste Data**	**t/cap/a**	**Per capita GDP (2005 USD PPP)**	**GDP Data**
OECD							
WWF (2013)	Hong Kong	2011	City	City MSW Stats	0.2	32608	See table S5
Kanat (2010)	Istanbul, TR	2005	City	City MSW Stats	0.2	15692	Taken as 2005 USD PPP from OECD(2015)
Codoban and Kennedy (2008)	Toronto, CA	2008	Household	City-wide municipal solid waste statistics	0.3	36004	See table S4
Browne et al. (2009)	Limerick, IE	1996	City	National waste inventory	0.1	23872	See table S4
		2002			0.2	25340	See table S4
Moore et al. (2013)	Vancouver, CA	2013	Household	Regional food waste data	0.2	36224	See table S6
Forkes (2007)	Toronto, CA	1990	City	City level statistics	0.3	31950	See table S4
		2001			0.3	37634	See table S4
		2004			0.3	37725	See table S4
Warren-Rhodes and Koenig (2001)	Hong Kong, CN	1999	City	City level statistics	0.2	21904	See table S4
		1971			0.1	7535	See table S4
Damghani (2007)	Tehran, IR	2005	City	City level statistics	0.1	13990	See Table S6
non-OECD							
Pina and Martinez (2013)	Bogota, CO	2010	City	City level statistics	0.1	13533	Taken as 2005 USD PPP from OECD(2015)
Talyan (2008)	Delhi, IN	2002	City	City level statistics	0.1	3041	Taken as 2010 rupees from Anand (2010), converted to 2005 rupees from World Bank (2014) and converted to 2005 USD PPP from World Bank (2014)
Yi (2011)	Seoul, SK	Mid 2000s	City	City level statistics	0.1	22890	Taken as 2005 USD PPP from OECD(2015)

Lee (2007)	Seoul, SK	2005	City	City level statistics	0.1	22890	Taken as 2005 USD PPP from OECD(2015)
Yoon (2002)	Tokyo, JP	1999	City	City level statistics	0.2	33339	Taken as 2005 USD PPP for year 2001 from OECD(2015) and adjusted for growth to 1999 assuming 3% growth rate.
Li (2009)	Beijing, CN	2006	City	City level statistics	0.2	15044	Taken as 2005 USD PPP from OECD(2015)
Study Average	-	-	-	-	0.2 ± 0.1	23956 ± 10766	-
Global Average	-	2007		Taken as consumer waste from FAO (2011)	0.1	7585	Taken from FAO (2014) as global GDP in 2005 PPP divided by global population.

Table S7 - Waste foodprints

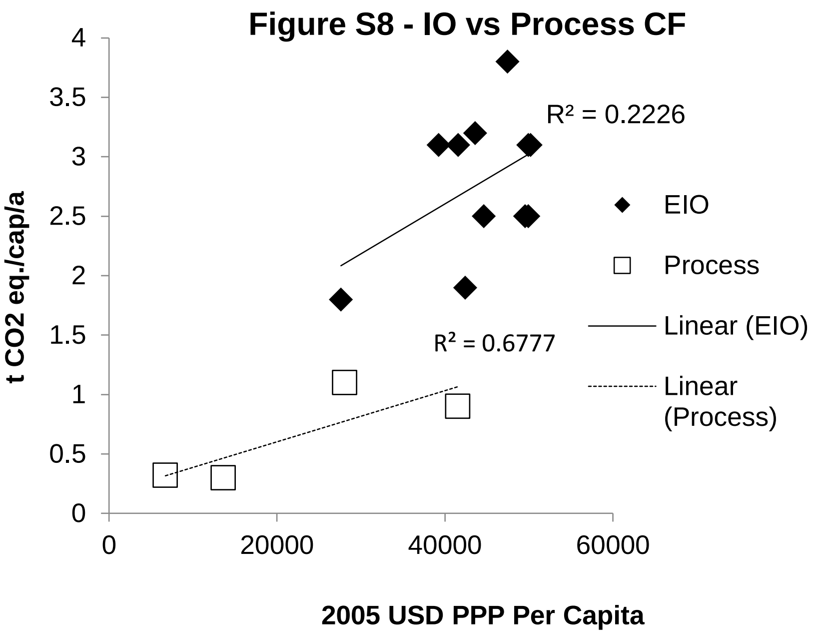


Figure S8 - IO vs Process CF
t CO2 eq./cap/a
2005 USD PPP Per Capita
4
3.5
3
2.5
2
1.5
1
0.5
0
0
20000
40000
60000
R² = 0.2226
R² = 0.6777
EIO
Process
Linear (EIO)
Linear (Process)

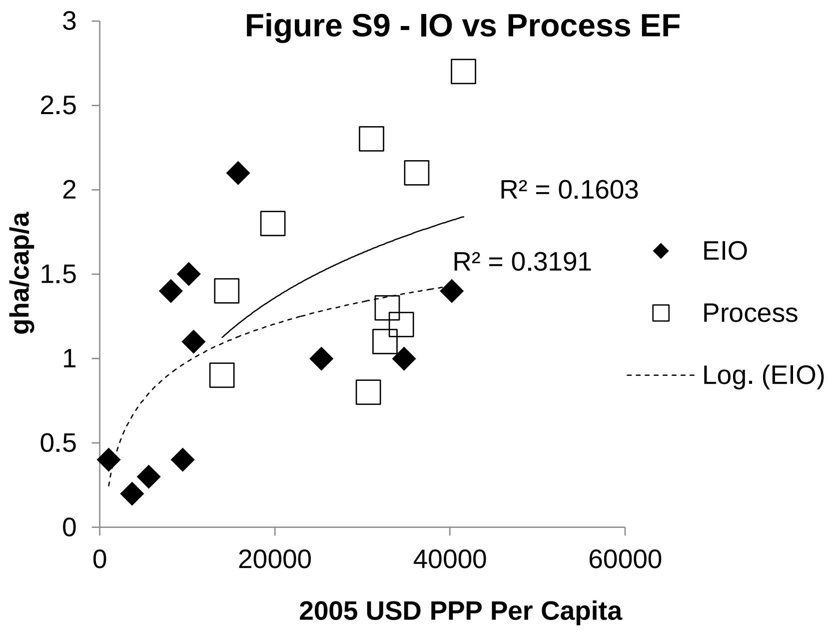


Figure S9 - IO vs Process EF
gha/cap/a
2005 USD PPP Per Capita
3
2.5
2
1.5
1
0.5
0
0
20000
40000
60000
R² = 0.1603
R² = 0.3191
EIO
Process
Log. (EIO)

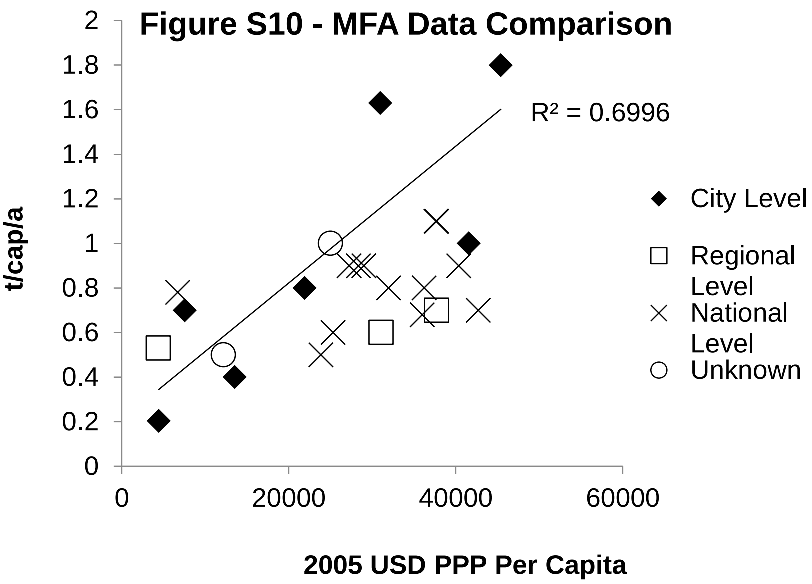


Figure S10 - MFA Data Comparison
t/cap/a
2
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
0
20000
40000
60000
2005 USD PPP Per Capita
R² = 0.6996
City Level
Regional Level
National Level
Unknown

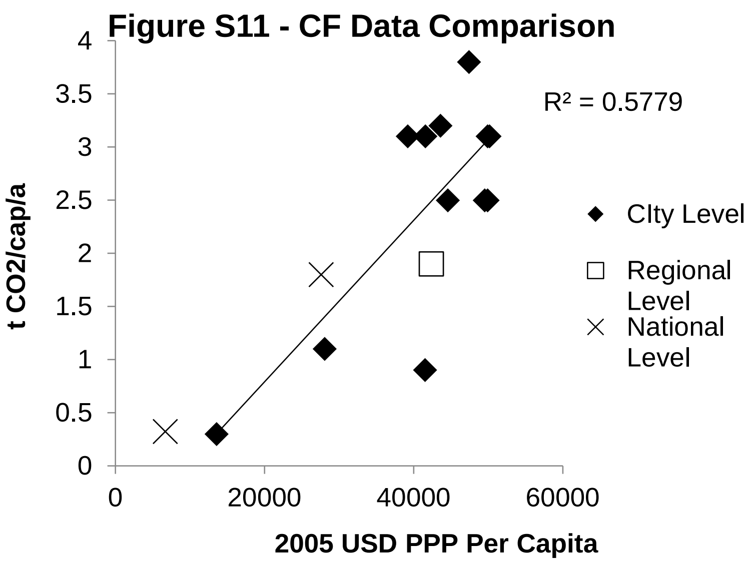


Figure S11 - CF Data Comparison
t CO2/cap/a
4
3.5
3
2.5
2
1.5
1
0.5
0
0
20000
40000
60000
2005 USD PPP Per Capita
R² = 0.5779
CIty Level
Regional Level
National Level

Figure S14 - CF Unit of Analysis

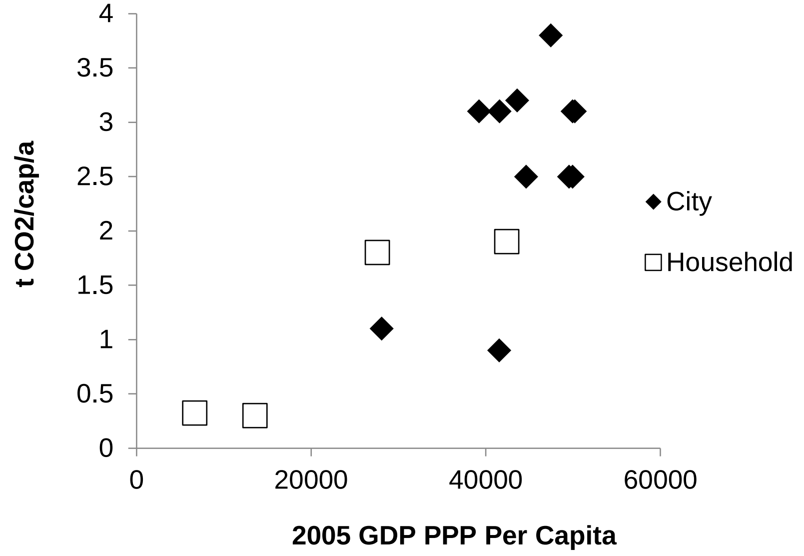
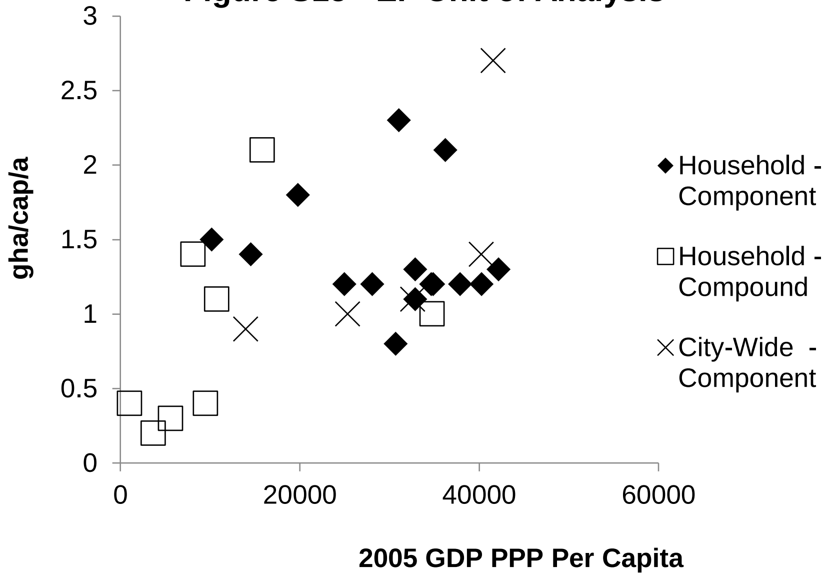


Figure S15 - EF Unit of Analysis



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Appendix B

Article 2: Urban versus conventional agriculture, taxonomy of resource profiles: a review

Urban versus conventional agriculture, taxonomy of resource profiles. A review

Benjamin Goldstein
Technical University of Denmark, Quantitative Sustainability Assessment Division
Producktionstorvet, Building 424, Room 225, Kongens Lyngby, Denmark, 2800

Michael Hauschild
Technical University of Denmark, Quantitative Sustainability Assessment Division
Producktionstorvet, Building 424, Room 209, Kongens Lyngby, Denmark, 2800

John Fernández
Massachusetts Institute of Technology, Department of Architecture
77 Massachusetts Avenue, 5-419, Cambridge, United States, 02139

Morten Birkved
Technical University of Denmark, Quantitative Sustainability Assessment Division
Producktionstorvet, Building 424, Room 216, Kongens Lyngby, Denmark, 2800

Corresponding Author: Technical University of Denmark, Quantitative Sustainability Assessment Division, Producktionstorvet, Building 424, Room 225, Kongens Lyngby, Denmark, 2800
T: +45 45 25 45 61 E-mail: bgol@dtu.dk

Urban agriculture appears to be a means to combat the environmental pressure of increasing urbanization and food demand. However, there is hitherto limited knowledge of the efficiency and scaling up of practices of urban farming. Here we review the claims on urban agriculture's comparative performance relative to conventional food production. Our main findings are: 1) benefits such as reduced embodied carbon, urban heat island reduction, and storm water mitigation, have strong support in current literature. 2) Other benefits such as food waste minimization and ecological footprint reduction require further exploration. 3) Urban agriculture benefits to both food supply chains and urban ecosystems vary considerably with system type. To facilitate the comparison of urban agriculture systems we propose a classification based on: 1) conditioning of the growing space, and 2) the level of integration with buildings. Lastly, we compare the predicted environmental performance of the four main types of urban agriculture that arise through the application of the taxonomy. The findings show how taxonomy can aid future research on the intersection of urban food production and the larger material and energy regimes of cities (the 'urban metabolism').

Keywords: urban agriculture; quantitative sustainability assessment; urban food systems; life cycle assessment; building integrated agriculture

1. Introduction

Recent scientific consensus is that humanity is skirting the planet's safe boundaries to sustainably supply resources to and assimilate society's waste (Krausmann et al. 2009; Steffen et al. 2015). As centers of population and economic activity, cities have a dominant influence on the scale and form of anthropogenic material and energy flows, consequently playing a central role in any shifts towards sustainability (Dearing et al. 2014; Pincetl et al. 2014). Hitherto, the general tenor in promoting urban sustainability has been a focus on minimizing fossil fuel intensive transport, reducing the energy consumption of buildings and shifting cities towards renewable energy sources (Grubler et al. 2012; IPCC 2014a). These transitions are all important pieces in the sustainable urban development mosaic, but they disregard one of the largest environmental pressures of cities: urban food consumption.

Supplying food to cities is one of the key contributors to greenhouse gas (GHG) emissions, biodiversity loss, water pollution, land use change, non-renewable resource exhaustion and a host of other pressing environmental challenges at the global scale (Foley et al. 2011; Tilman et al. 2011; Gliessman 2015). Therefore, transitions towards sustainable urban systems must include the mitigation of environmental impacts from urban food consumption.

Multiple angles exist to address the environmental burden of urban food demands. Internalizing environmental burdens of food production within prices using Pigovian taxes has been suggested to nudge consumers away from environmentally burdensome foods (e.g. meat and dairy) (Edjabou and Smed 2013). Others have emphasized transitions to diets that combine seasonality of local food production, selectively consume organic, and contain reduced animal protein (Saxe 2014). Another option at the demand side is to reduce food waste in cities, lowering gross urban food demands and solid waste burdens (FAO 2013). Cities have also banned certain types of food packaging (see New York City's sanction against polystyrene) to reduce the environmental impacts of the food system at this end (Stringer 2015). Conversely, supply side interventions promote eco-efficiency gains within existing production systems (e.g. reducing fertilizer per unit economic output) (Tilman et al. 2011; Davis et al. 2012).

1.1 Urban agriculture to promote environmental sustainability

While the bulk of food production is exogenous to the city, urban agriculture (UA) has been touted as a supply-side urban design intervention that can give cities agency over the environmental performance of some of their food demands (Pearson et al. 2010; Hampwaye 2013; IPCC 2014b). Though many definitions exist (see Vejre's outlining of the spatial, functional, market, and other dimensions of defining UA; Vejre 2012), the most salient features are that is consists of *food production in and around cities, weaving this practice within the urban form, such that it interacts with the host city's material and energy metabolism* (Koc et al. 1999).

Recent estimates peg urban farmers at 25-30% of global urban dwellers (Orsini et al. 2013). Most of these practitioners operate in emerging economies as an informal means of income and subsistence (Smit et al. 2001), however there has been increasing interest in intensifying and formalizing UA globally as part of a more sustainable and resilient global food system (Pearson et al. 2010). Very optimistic estimates assert that UA could supply 100% of global urban vegetable needs with 40% of urban land at current yields (Martellozzo et al. 2014), while others have estimated that UA already produces 15-20% global food (Abdulkadir et al. 2012). Shanghai and Beijing stand-out as examples in that their metropolitan regions supply the majority of their produce (Lee-Smith and Prain 2006), and Shanghai most of its dairy demands (Orsini et al. 2013). The potential in post-industrial cities is believed to be high but untapped (Grewal and Grewal 2012; Taylor and Lovell 2012; Haberman et al. 2014); rooftop UA could provide 77% of Bologna's vegetable needs (Orsini et al. 2014) or 36% of Singapore's vegetable intake (Astee and Kishnani 2010). Other assessments are less optimistic, such as Oakland's potential to supply between 0.6-1.5% of recommended vegetable needs (McClintock et al. 2013). A the institutional level, a departure from the farming-antagonistic land-use planning that dominated the 20th century (Brunori and Di Iacovo 2014) towards active promotion by cities (City of Boston 2014; Five Borough Farm 2014) punctuates a new narrative, supporting the re-integration of food production within the contemporary city.

1.2 Urban agriculture's environmental performance

Large scale implementation of UA within cities may be a vital step towards improving urban environmental performance, but many claims of UA's improved environmental sustainability relative to conventional agricultural remain premature given the paucity of field verification and quantitative sustainability assessment (QSA) of UA systems

(Pearson et al. 2010; Pataki et al. 2011; Specht et al. 2013). To date, much of the discourse around UA's environmental potential focuses on its ability to reduce distance from farm to fork, ignoring how such systems may be maladaptive when other aspects of urban food production are considered (e.g. warming greenhouses in northern climes to avoid transport from southern countries) (Born and Purcell 2006). Recent QSAs have begun to address the gaps in knowledge surrounding UA's environmental performance. Sanyé-Mengual and colleagues studies of urban rooftop greenhouses showed these systems can provide tomatoes with lower embodied environmental burdens relative to traditional supply chains (2012; 2015b). On the other hand, Kulak et al. (2013) reduced climate change impacts for UA in London relative to conventional supply chains for some produce, but not others.

These initial environmental evaluations of UA simultaneously provide answers and raise questions. Sanyé-Mengual et al.'s work on soil-less rooftop UA revealed major differences in environmental performance between the different cultivation methods (2015). If there are noticeable differences in environmental performance between cropping systems on a single rooftop, how large are the differences between UA systems with fundamentally dissimilar characteristics (integrated with buildings vs. grown on land) and are there general trends in environmental performance between these types of systems? Kulak et al. (2013) found that capital inputs for low-tech greenhouses deleteriously affected the environmental performance of UA in London. If modest inputs are influential, how do these inputs affect the performance of UA systems with markedly contrasting material needs (raised beds vs. greenhouses) and are there general performance trends between UA types? Lastly, there has been only tangential discussion about how large-scale UA would influence citywide material and energy fluxes (its 'urban metabolism') if it were scaled up within a given city (Pataki et al. 2011; Cerón-Palma 2012).

The propensity to consider single UA types out of the multitude that exist (greenhouse, raised-beds, vacant lot, etc.) overlooks the non-trivial energy and material profiles of different UA systems, leaving an incomplete picture of UA's environmental strengths and weaknesses. As a result, it remains unclear whether installing different forms of agricultural production forms *en masse* in cities across the globe constitute a net reduction in food related environmental impacts from cities, necessitating a holistic and systematic look at UA's environmental performance. If there were patterns of environmental performance for different types of UA, the development of a heuristic to support future research and urban design decisions would be of utility to academics, policy makers and UA champions alike. To date, such a system has been lacking, with previous work in cataloguing UA centered around social and economic concerns (Jacobi et al. 2000; Smit et al. 2001; Brock and Foeken 2006; Dossa et al. 2011; Orsini et al. 2013; Thomaier et al. 2015). Moreover, researchers of urban environmental performance lacking an agricultural background (e.g. industrial ecologists, urban planners, landscape urbanists, etc.) lack a tool to organize and assimilate the environmental performance of UA within their own assessments of the larger urban environment.

1.3 Study goal and scope

This article consolidates and expands on earlier reviews of UA's environmental sustainability (Pearson et al. 2010; Pataki et al. 2011; Specht et al. 2013) with an updated appraisal of the myriad environmental claims surrounding UA and their existing levels of support. We then distill these findings into an UA taxonomy based on UA system material use, energy consumption, and interaction with the built form. This taxonomy will provide an organizing framework for future QSA research and deliver clarity to non-agronomists. Through a focus on those aspects of UA most salient to researchers of urban environmental performance, the taxonomy will also act as a device to scale up from studies of environmentally sustainable at the individual UA project level to assessments of food production networks at the city-scale.

2. Environmental Performance of Urban Agriculture – Disaggregating Claims and Evidence

Using the aforementioned literature reviews as a point of departure, both peer-reviewed and grey literature were perused throughout 2014 and 2015. The reviewed material illuminated a patchwork of qualitative and quantitative environmental declarations surrounding UA. Table 1 presents these claims along with any support across five umbrella terms that encompass them: *building energy, urban symbiosis, supply-chain efficiencies, in-situ and ex-situ environmental improvements*.

2.1 Building energy

The potential benefits of UA in relation to building energy consumption are some of the best documented due to previous research on green roofing that can reasonably be extrapolated into the realm of UA. UA is posited to reduce building energy in a number of ways:

1. Passive methods - increasing building albedo (light reflection), endothermic plant/substrate evapotranspiration (Qiu et al. 2013) or improving building insulation (Smit et al. 2001);
2. Active methods – cooling building space with evaporative cooling in greenhouse (Ackerman 2012) or exchanging excess heat between building and greenhouse to reducing building energy needs (Cerón-Palma 2012).

Field trials of green roofs in Canada and China support the passive benefits along with a model of green roofs in the US (Bass and Baskaran 2003; Kokogiannakis et al. 2011; Jaffal et al. 2012). Conversely, increasing accessible roof area significantly diminished life-cycle energy savings (16% to 4%) for green-roofs in Singapore (Wong et al. 2003), a challenge considering UA operations require space for maintenance, harvesting, packaging, etc. Looking at rooftop greenhouses, a US analysis showed that potential to cool the host building is present, but in a diminished capacity compared to standard green roofs, also due to light absorbing maintenance areas (Delor 2011) with Cerón-Palma modelling annual energy savings from insulation at less than 5% in a Mediterranean climate (2012). Climate was also important, with green roofs reducing summer indoor temperatures in Stockholm and Athens, lowering winter heating load in Stockholm through insulation, but increasing winter heating load in Athens due to evapotranspiration (Jaffal et al. 2012). Moreover, these benefits to building energy diminish when well insulated buildings considered (Castleton et al. 2010; La Roche and Berardi 2014).

Less explored are UA's active benefits. Cerón-Palma (2012) integration of the energy systems of a rooftop greenhouse and an office building in a Mediterranean climate, showing that heat recovery from the greenhouse using ventilation could reduce indoor heating requirements by 79%, though it should be considered that heating requirements in the Mediterranean are already generally low.

2.2 Urban symbiosis

Urban symbiosis is UA's potential to leverage proximate urban residual material and energy fluxes as production factors, attenuating urban waste and avoiding virgin material inputs to food production. Three dominant claims emerged:

1. Waste assimilation – the use of waste with high organic carbon or nutrient content to supplement UA substrate or nutrient demands (Grewal and Grewal 2012);

2. Rainwater harvesting – reducing runoff to sewers and reducing irrigation demands (Nelkin and Caplow 2008);
3. Building Energy – utilizing excess building energy to reduce greenhouse energy inputs (Cerón-Palma 2012).

Waste assimilation is a recurring claim, since cities import large amounts of synthetically produced nutrients embedded within food that usually end up in waste streams for emission to local water bodies (Morée et al. 2013) or partial recovery during waste management (Larsen et al. 2001; Kalmykova et al. 2012). UA could act as a sink for nutrient rich grey-water (baths and sinks), black-water (toilets), and organic solid waste (household, commercial or industrial), providing the basis for a closed-loop urban food production system (Grewal and Grewal 2012).

In practice, cities have leveraged black-water for UA historically (Barles 2007) and in present day emerging economies (Qadir et al. 2010; Forman 2014), though the public health risks from pathogens and heavy metals remain high (Cofie et al. 2006; Qadir et al. 2010). Nutrients are also captured downstream at wastewater treatment plants where sewage sludge is pelletized to fertilize animal feed or energy crops (skirting the issue of direct human pathogen consumption) (Miljøministeriet, 2005), largely excluding usage in UA.

Two forms of organic solid waste are available to generate nutrient rich compost in cities; food scraps and yard detritus. Food scraps have long been utilized in UA with recent examples being in Cuba (Hernandez et al. 2014), the UK (Edmondson et al. 2014) and New York City (City of New York 2014), though policy-makers in the latter have actively fought against implementing household organic waste collection due to perceived costs (Decker et al. 2000). A rooftop, raised-bed farm in Paris used 100% local organic waste fractions (food waste, coffee grounds and mycelium, crushed wood, wood chips and potting soil) as a substrate, producing lettuce in yields comparable to commercial operations (Grard et al. 2015). Yard refuse derived compost is actively distributed to UA sites by New York City (City of New York 2014). Although composting reduces pathogen related health risks, potential contamination from heavy metals remains challenging (Hargreaves et al. 2008), while carbon-nitrogen ratios of the different waste streams must also be considered to maintain soil health and productivity (Komilis et al. 2012; Awasthi et al. 2015). The aforementioned UA project in Paris is a positive example in this regard, getting 100% of nutrient demands through a balanced waste blend, whilst producing food in line with EU pollutant regulations (Grard et al. 2015).

Other urban symbiosis potentials include rainwater harvesting and excess building energy capture. The former has been implemented (Nelkin and Caplow 2008), with over 100 operations in New York City utilizing this practice (Cohen et al. 2012), though risks exist for rain to deliver airborne contaminants acidifying the soil or depositing heavy metals (Forman 2014). Rainwater collection has also been seen in rooftop greenhouses, such as the Fertilicity project in Barcelona, ES reducing water impacts by 98% compared to a traditional tomato (Sanyé-Mengual et al. 2015b), the Arbor House in New York City (Green Home NYC 2011) and Lufa Farms in Montreal, CA (Lufa Farms 2014). Benefits of rainwater capture must be balanced against the embodied burdens additional structural buttressing, which can be significant depending on the installed system, and pumping energy requirements. Angrill et al. (2012) found that rainwater harvesting for non-potable use reduced local water demands, but had higher global warming impacts compared to municipal water supply in some instances. It remains unknown how these tradeoffs influence the overall performance of rooftop farms. Excess building energy can be used to moderate growing space temperature, therefore, it is only of utility to greenhouse systems, and though conceptually sound, lacks application. Cerón-Palma (2012) modelled using excess building energy as a means to warm a rooftop greenhouse, finding that periods of greenhouse heating demand were misaligned with periods of excess building heat over diurnal cycles, precluding use of the this energy.

2.3 Supply Chain Efficiencies

Efficient supply chains are the streamlined needs of UA compared to typical urban food supply chains. Claims in this regard appear to focus on three points:

1. Reduced 'food miles' – shorter distance between producing and consumer (Born and Purcell 2006);
2. Increased yields – improved farm performance over conventional supply chains (Despommier 2013);
3. Distribution efficiencies – reduced packaging and spoilage (Sanyé-Mengual et al. 2012).

By reducing the distance from producer to market ('food-miles') environmental sustainability claims relating to transport naturally arise, which at first glance appears defensible assuming *a priori* that food grown within a city is consumed locally. Notwithstanding, the focus on 'food-miles' may be misplaced, due to transport's relatively small environmental impacts over food supply-chains (Born and Purcell 2006; Edwards-Jones et al. 2008), except where air transport or long distance refrigerated freight occurs (FAO 2011a). A model of local vegetable production around Osaka found that local vegetable production could reduce 25% of food production energy (Hara et al. 2013), lending credence to 'food-miles'. However, energy is not a holistic indicator for environmental performance since increased impacts in other aspects of production could erase reductions in transport energy ('burden shifting').

The efficiency claim of improved yields of UA greenhouses, achieved by shielding crops from moderating variables (pests, extreme weather, etc.) is true for all greenhouses (von Zabeltitz 2011), and is not a unique benefit of UA. This claim may be justified in the context of vertical farms (stacked greenhouses), since they produce more food per unit area, such as the Mirai project in Japan which produces 10 000 lettuce heads a day with under 2500 m^2 (Dickie 2014). Vertical farms (or 'plant factories') continue to proliferate with examples in South Korean (Suwon Farm), the Netherlands (PlantLab) and the United States (Green Spirit Farms) (Marks 2014), though it remains unknown whether the increased yields offset the potentially high capital and energy requirements of these systems. At the other end of the spectrum, low-tech UA systems in sub-Saharan Africa had poor practices and profligate pesticide usage well above recommended rates, leading to yields below conventional supply-chains and increased public health risks (Perrin et al. 2015).

Lastly, analyses of rooftop greenhouses posits that UA could reduce both packaging and food waste (Sanyé-Mengual et al. 2012; Sanyé-Mengual et al. 2015b). The former is a logical consequence of lower food miles and is potentially important in reducing selected environmental impacts (IBID). Food waste is more complex since it is primarily generated at the retailer and consumer in wealthy populations, versus the at the farm or in transit in poorer countries (FAO 2011b), meaning that UA could better reduce losses in a developing context. Notwithstanding, if earlier assessments are correct, food losses might be reduced by UA relative to the conventional supply-chain.

2.4 In-Situ Environmental Improvement

In-situ environmental improvement outlines beneficial environmental amenities brought to the urban environment by UA. From the literature review, the following claims were identified:

1. Increased biodiversity (Havaligi 2011);
2. Reduced urban heat island (UHI) – lower temperatures due to increased albedo and evapotranspiration (Oberndorfer et al. 2007);
3. Reduced stormwater runoff – retention by substrate and filtering of pollutants (IBID);

4. Soil improvements – improved stability, organic carbon content (Jansson 2013);
5. Air quality – filtration of airborne pollution by plant matter (Hampwaye 2013).

Claims regarding UA's improvement of local biodiversity are supported by experiments with vegetated roofs where poly-cropping (multiple crop species) and predatory pest control (e.g ladybugs) were used (Hoffman 2007; Oberndorfer et al. 2007), but could be reversed if mono-culture cropping were implemented (Reidsma et al. 2006). UA is believed to provide refuge for keystone pollinators (e.g. bees) further enhancing urban ecosystem resilience through promoting functional diversity (IBID) and may provide green corridors for animal movement through cities when linked to larger parks systems (Forman 2014).

Two areas where the local environmental benefits of UA are well documented are UHI and urban runoff attenuation (Oberndorfer et al. 2007). UHI results from the propensity of low-albedo dark surfaces to trap solar radiation and transform it to heat, which UA mitigates by substituting these surfaces with plants that absorb sunlight for photosynthesis and provide shade (Li et al. 2014), a benefit that will reap dividends with the increasing frequency of heat-waves (Jansson 2013). UA substrates retain stormwater runoff for plant uptake or provide climate change adaptation by buffering the surges to local water systems (IBID). Moreover, ground-based UA opens a permeable hydraulic-bridge between storm-water and groundwater systems attenuating sewer systems stressed by the prevalence of impermeable surfaces in cities (Oberndorfer et al. 2007). These benefits are dependent on the UA form practiced, with shallow soil beds on green-roofs reducing the attenuation UHI and storm-water (deep substrate green roofs can become waterlogged, eliminating runoff retention) (Luckett 2009), while greenhouses without rainwater capture have little benefits towards urban runoff management. Moreover, UA in low-lying areas of the cities may be inundated with polluted runoff from adjacent impermeable surfaces (Forman 2014). A negative consequence of UA is that runoff from urban farms may contain high nutrient loads that could exceed local assimilative capacity if these systems are scaled up within cities (Emilsson et al. 2007; Li and Babcock 2014).

For the soil quality claims, UA must be planted in local soils (eliminating most greenhouses from this benefit) and the soils must avoid the contamination common in cities (Meuser 2010; Li and Babcock 2014). With these conditions met, UA may improve soil stability and fertility, contingent on harnessing ecological principles to maintain organic carbon and nutrient levels (Gliessman 2015), as demonstrated in some British allotment gardens (Edmondson et al. 2014). Lastly, air quality improvements have been seen in a number of models of green areas in cities (Yang et al. 2008; Jim and Chen 2009), though the potential for numerous plant species to emit toxic compounds when stressed (Pataki et al. 2011) requires more attention.

2.5 Ex-situ Environmental Improvement

Ex-situ environmental improvement relates to benefits conferred by UA beyond the city-region. In the reviewed material, the following claims were identified:

1. Carbon sequestration – removal and storage of CO^2 from the atmosphere (Sida 2003);
2. Reduced carbon footprint – lower embodied greenhouse gas emissions for production and distribution of food compared to conventional supply chains (IBID);
3. Reduced ecological footprint – lower agricultural land occupation for consumers (RUAF 2006);
4. Improved biodiversity – return of marginal agricultural land to nature (Knowd and Mason 2006);
5. Improved soil quality - return of marginal agricultural land to nature (Smit et al. 2001).

The first is that of the carbon sequestration, whereby UA fixates atmospheric carbon through photosynthesis. Li and Babcock's (2014) review of green roofs carbon sequestration highlighted shows the potential for this type of infrastructure to accumulate biomass. Notwithstanding the claim's veracity, UA's true contribution towards carbon sequestration may ultimately be marginal, as shown by studies of Toronto, CA (a city with considerable foliage) (Kennedy 2012) and Salt Lake City, US (Pataki et al. 2009) where the urban tree canopy sequestered <1% of urban carbon emissions.

For carbon footprint assessments of rooftop greenhouse tomato production in Barcelona, ES showed 33-62% reduction in embodied carbon impacts relative to conventional supply chains, a result of the reduced transport, packaging and predicted food distribution losses (Sanyé-Mengual et al. 2012; Sanyé-Mengual et al. 2015b). An assessment of food produced in London allotment gardens revealed significant embodied carbon reductions (25-99%) for fruits in vegetables (Kulak et al. 2013). UA performed worse for strawberries grown in low-tech greenhouses, showing that UA's benefits hinged on low material intensity methods producing local foods, or the substitution of high impact foods with UA (e.g. foods air freighted to the UK). Comparing carbon sequestration of typical urban landscaping projects (parks and forests) to reduced climate change impacts from UA, consumption of UA has a greater impact per unit-cultivated area (IBID). Research has also shown that crop choice is an important aspect of greenhouse gas emissions, with high-yield fruits and vegetables (tomatoes, eggplants) having superior performance to low-yield leafy vegetables (Sanyé-mengual et al. 2015).

The other three extended environment assertions of improving biodiversity, soil quality and the ecological footprint of cities remain difficult to prove or disprove. They appear predicated on the assumption that UA will displace farming outside of cities, allowing succession of agricultural land to mature ecosystems; a shaky contention in a globalized world with increasingly affluence, growing population (Foley et al. 2011; World Bank 2013) and limited options to expand conventional agricultural production areas (FAO 2006). However, if UA were to play a larger role in global food production, it may stymie the conversion of natural habitats and even allow for conversion of farmland back to natural ecosystems, with the added benefit of sequestering carbon within soil and mature habitat.

2.6 Urban Agriculture – Where Do We Stand?

The literature revealed a muddled picture of UA's ability to reduce the environmental impacts from urban food demands and positively contribute to the urban ecosystem. Some claims are demonstrated to varying degrees (urban storm-water management, building energy use reductions, UHI, local biodiversity, nutrient recycling and soil quality, carbon footprint reduction), others prematurely (carbon sequestration, improved yields, air quality), while a few are of more speculative nature (EF reduction, soil upgrading outside the city, biodiversity gains, avoided food waste). UA could provide some of the more conjectural benefits, but there currently remains little proof-of-concept of those gains, meaning that conclusions about UA's general environmental efficacy are *a priori*.

What is clear is that UA's capability to increase the sustainability of urban food systems is contextual, based on UA method, product and location. The case study of carbon sequestration in London, UK allotment gardens exhibited all of these traits, with changing conclusions for different UA types growing the same product, since UA type dictated the supporting infrastructure (structure, HVAC, etc.) and operating inputs (chemicals, water, energy, etc) (Kulak et al. 2013). Kulak et al. (2013) found that switching from outdoor to polytunnel strawberries reversed carbon footprint reductions over conventional production (-53% to +12% compared to base case). Interestingly, tomatoes did not show the same behavior, with significant embodied carbon reductions over conventional supply-chains using outdoor or polytunnel methods. Recent assessments of rooftop soil-less production in Barcelona, ES also showed how environmental impacts for different growing techniques can vary for production on the same roof (Sanyé-mengual et al. 2015), with soil-less production methods of leafy greens having significantly superior environmental performance compared to soil cultivated counterparts. Performance on local environmental indicators (UHI, stormwater retention, etc.) also varies according to UA scheme, highlighting that the relationship UA to the larger urban ecology also depends on the UA type employed.

At the urban scale, it remains unknown how some of the benefits and shortcomings of UA might affect the greater urban system. Nutrient runoff from UA has been studied at the individual farm level, but the effects of the aggregate runoff from urban scale UA implementation are not known. UA benefits of waste assimilation and UHI mitigation are also minimally understood at the city-level.

3. Developing a taxonomy to support the environmental assessment of UA

It has already been voiced by several researchers that further QSAs of UA are required before the environmental sustainability claims of UA champions can be verified (Ackerman 2012; Specht et al. 2013). Notwithstanding the need for more assessments, such explorations would be most effective with an organizing framework, such as a systematics of UA types based on environmental performance.

Kostrowicki started his 1977 definition of agricultural typologies with,

'An attempt at ordering the investigated facts and/or processes according to a certain system is a characteristic stage of development of any scientific discipline.' (Kostrowicki 1977)

This paper is not proposing anything as ambitious as a scientific discipline, but we do aim to provide a heuristic, in the form of a UA taxonomic scheme to order existing knowledge and future assessments of the environmental performance of UA. A taxonomic scheme (*systematics, taxonomy* and *typology* are used interchangeably hereafter*)* is a grouping of individuals in a population based on the similarity of their attributes (e.g. organic, conventional and biodynamic agriculture). This grouping does not ignore the uniqueness of the individuals (e.g. mono and poly-crop organic), but focuses on essential characteristics (e.g. organic prescribes no synthetic fertilizers or pesticides) to make a complicated reality comprehensible. It is for this reason that typologies are also hierarchical in nature, with sub-typologies belonging to higher order typologies (Kostrowicki 1977).

To date, taxonomies of UA have had a valence towards cataloguing based on socio-economic criteria (see: Brock & Foeken, 2006; Dossa, Abdulkadir, Amadou, Sangare, & Schlecht, 2011; Drechsel & Dongus, 2009; Jacobi, Amend, & Kiango, 1997; Smit et al., 2001). The social and economic aspects of UA are essential aspects of sustainability, but systematics framed around these attributes do not provide a clear picture to researchers or decision makers about the environmental performance of different UA types.

The aim of the systematics introduced here is to provide a simple, overarching scheme of the different combinations of essential attributes of UA that have important influences on the environmental dimensions of urban food production. At the base of it, the environmental performance of any production system hinges on the energy and material regime that supports the good or service it generates (Smil 2013). In agricultural it is the production factors (fertilizers, land, fossil fuel energy, pesticides, irrigation, farming structures and mechanized equipment) that influence the environmental burdens of food system (Davis et al. 2010; Roy et al. 2012; Meier and Christen 2013), and in rare instances transport (FAO 2011a). Our endeavor is to identify the broad characteristics of UA systems these capital inputs.

Considering the limited number of studies of UA's material and energy demands, a first impression of these was gathered from earlier assessments of isolated UA systems (Wong et al. 2003; Astee and Kishnani 2010; Sanyé-Mengual et al. 2012; Kulak et al. 2013; Sanyé-Mengual et al. 2015b) as well as green roof (Luckett 2009), greenhouse (von Zabeltitz 2011) and UA (Philips 2013) design books. To support the literature findings we visited UA operations and performed interviews urban farmers in Northeastern United States during the Spring and Summer of 2015. From these, we identified two organizing principles emerged that strongly influence UA energy and material regimes, forming the basis of the typology: building integration and space conditioning.

3.1 Building Integration

The first organizing principle is how physically embedded the UA form is within the built environment. Designs that leverage residual UM flows (nutrients, building heat, etc.) are at an advantage to avoid/share virgin resource inputs over less immersed UA forms. For optimal access to residual UM flows and to potentially have direct energy exchange with buildings, UA is best situated on buildings where waste flows emerge and conditioned space is able to act as a source and/or sink for energy. This is most applicable to heat, which due to its dispersive nature, requires direct coupling of the UA and building energy systems in order to share excess energy (attenuating temperatures of growing and occupied space). Moreover, attaching UA to the built environment also insulates the host building, reducing building energy consumption. This intimate coupling can also bring benefits through the circulation of CO_2 rich building exhaust into the greenhouse to promote growing (Sanyé-Mengual et al. 2014). The advantages of direct placement on buildings is less vital for nutrients, since nutrient rich waste can potentially be collected at any place between point of generation and place of disposal for application as greywater, blackwater, compost or other form, though proximity to generation points could be beneficial (IBID). Moreover, soil-based UA is best suited for urban nutrient recycling, since composting of solid organic waste the commonest recycling method is, though examples of application of waste-derived liquid growth stimulator may also be viable (Hernandez et al. 2014). Rainwater harvesting is not dependent on building integration. Because of these observations, we introduce *building integrated* and *ground-based* UA types, where the former is merged with existing building structures, while the latter occurs directly on the ground in a manner physically disconnected from surrounding buildings.

3.2 Space Conditioning

The second consideration was the degree of interaction between UA systems and ambient environment. UA systems with conditioned growing spaces (e.g. greenhouses) allow year-round operation, capture and more efficiently recycle resources, minimize weather related crop losses, and reduce pest invasion in contrast to open systems (e.g. vacant lot farming). Conversely, conditioned UA types also require large resource inputs in terms of building components, mechanical equipment and embodied energy within capital equipment. Energy for space conditioning (light and temperature) is also paramount as the environmental performance of food production systems in some indicators (fossil fuel consumption, global warming forcing) are dependent on whether the conditioned space is heated or not (Stoessel et al. 2012). Non-conditioned systems contrast with this in that they usually have higher losses of resources to the ambient environment, but are less capital intensive, and have lower direct operational energy inputs. Non-conditioned systems also have higher risks of being negatively affected by local pollution (Antisari et al. 2015) and contributing to local pollution (Emilsson et al. 2007). The result being that these two classes of UA could have markedly different environmental performance. Therefore, we introduce the notions of *conditioned* and *non-conditioned* UA, where the former is quasi-closed system and the latter exposed to the elements.

3.3 Urban Agriculture Types

Because building integration and conditioning are not mutually exclusive, we derived four overarching UA types: ground-based-non-conditioned, ground-based-conditioned, building-integrated-non-conditioned and building-integrated-conditioned. As mentioned above, the taxonomy is a simple tool for a rough organization of findings, so it does not describe the minutia of different sub-types. For instance, the building-integrated-conditioned could encompasses rooftop greenhouses and vertical farms, since they both are integrated within the built environment, have substantial capital inputs and use large amounts of operational energy, all important factors that will differentiate their resource regimes and environmental impacts from a farm on an empty lot (ground-based-non-conditioned). Figures 1a to 1d show identified UA forms. From a quick glance, it is evident that actual UA systems mirror the qualities outlined in Sections 3.1 and 3.2: conditioned spaces have high capital inputs but reduced chances of ambient resource losses, while the non-conditioned spaces are lower intensity in terms of capital inputs and operational energy, but with diminished ability to minimize resource losses.

3.4 Predicted attributes of urban agriculture Types

Figures 2a to 2c outlines a comparison of the material and energy needs of ground-based-non-conditioned, ground-based-conditioned, building-integrated-non-conditioned and building-integrated-conditioned UA types based on our cursory analysis, and table 2 provides deeper details about these properties. Indicators are grouped into three broad categories covering *operating characteristics* (efficiency of supply use, external energy inputs, potential for crop losses, yields and growing season length), *capital inputs* (typical equipment and structures), *urban symbiosis potential* (possible coupling with urban material and energy flows), and *other general traits* (economic and social motivators). This represents a very rough overview of *predicted* operating characteristics and material and energy needs of these systems given the same growing location, product and agricultural practice (organic, conventional, mono cropping, etc.) Despite the elementary nature of this assessment, it highlights some divergent environmental aspects of the systems.

3.4.1 Ground-Based-Non-Conditioned Systems

In terms of operating characteristics, low resource use efficiency and yields due to dispersive losses of inputs and the potential for crop losses, countered by low external energy inputs are expected. Capital inputs are also low (fencing, small tools, irrigation lines and sometimes low-tech greenhouses for seedlings). Kulak et al.'s (2013) work on London allotment gardens confirms that the low nutrient and water efficiency of ground-based-non-conditioned UA, but lower capital inputs counteracted this, resulting in a reduced carbon footprint over conventional supply chains. Ground-based-non-conditioned UA also show medium performance in the realm of urban symbiosis potential, whereby it can act as a significant assimilator of urban solid waste as compost, demonstrated in the UK (Edmondson et al., 2014) and Cuba (Hernandez et al., 2014), or additionally reduce stormwater runoff (Gliessman, 2015). However ground-based-non-conditioned is at a disadvantage to couple with the liquid waste or energy systems of the city, though site walkovers in NYC did demonstrate hookups with adjacent buildings to capture runoff from roofs for irrigation (see Grow NYC: http://tinyurl.com/q9cm4ba). Lastly Figure 1 also shows that the growing seasons and yields of the NC forms are less than their conditioned counterparts, which is evident when one considers that all the pictures were taken in May 2015 (except 1D which was captured in March 2015). Because of the low-tech nature and low yields of this type, it lends itself to non-profit operation (or supplemental income generation) and high levels of community engagement (nutritional education, after-school programs, etc.)

3.4.2 Ground-based-conditioned Systems

Ground-based-conditioned contrasts with ground-based-non-conditioned in almost all indicators. Containment of growing medium and recycling makes for high efficiency of water and nutrients, concomitantly reducing potential losses from pests and weather. Conversely, operating energy is much higher to run equipment (pumps, heaters, mechanical louvers). Capital inputs are also high since ground-based-conditioned requires structural components, mechanical and irrigation equipment, and increasingly common, sensors and computers. The mix of high efficiency, high-energy inputs and substantial built capital can have conflicting effects. Kulak's (2013) work in London shows that even using low-tech greenhouses without mechanical inputs or hydroponics can increase water efficiency, but the capital inputs actually caused UA strawberries to have higher embodied carbon impacts than a conventional supply-chain.

In terms of operational energy, the importance of *passive* conditioned spaces (light and heat provided by solar) and *active* (light and heat provided through fuels or electricity) on environmental performance necessitates the need for two sub-categories in within the ground-based-conditioned type: active and passive. Active types have environmental impacts driven by operational energy, in line with QSAs of buildings, since the one-time impacts of constructing durable building components diminishes compared to the perennial energy inputs over the extended lifetime of the project (Scheuer et al. 2003). In contrast, capital inputs play a stronger role in the environmental performance of passive types, because operating inputs are relatively lower. The lack of studies comparing passive and active ground-based-conditioned operations makes it difficult to conclude on the tension between operational inputs and capital inputs.

Urban symbiosis potential for ground-based-conditioned is low compared to other UA types, since the primarily hydroponic nature of greenhouses complicates organic-waste recycling, whilst their detachment from buildings makes interactions with building and energy flows difficult. Site-specific storage capacity puts a cap on rainwater capture, further constricting potential symbiotic relationships with the city. Increased capital inputs mean that these farms typically operate to generate profit (see: www.farmedhere.com or www.freightfarms.com), though non-profit projects with high levels of community engagement can also be found (see: www.thefoodproject.org/dudley-greenhouse).

3.4.3 Building-Integrated-Non-Conditioned Systems

Building-integrated-non-conditioned mirrors the ground-based-non-conditioned in that its' exposed environment lowers the efficiency of water and nutrients at the farm compared to conditioned UA, though building-integrated-non-conditioned could potentially recoup some losses at the building edge. Some building-integrated-non-conditioned systems actually utilize soil-less cultivation (perlite substrate) with high operational efficiency (Sanyé-mengual et al. 2015), though this practice is not yet pervasive in UA. Our interviews with rooftop farmers also revealed that soil erosion due to winds is a chronic issue. Looking at figure 1c we can also see that though these systems have the potential for a considerable amount of capital inputs (irrigation networks, layers to protect roofing, sensors and computers, etc.), and potentially structural buttressing.

The urban symbiosis potential of building-integrated-non-conditioned is very high as it can assimilate solid organic waste from the urban system (limited by load bearing capacity), directly affect building energy (providing insulation, increasing roof albedo and capturing residual building energy to lengthen growing season) and water systems, and mitigate stormwater runoff. The numerous examples of green roofs in Table 1 attest to this with their positive contributions to building energy consumption (Bass and Baskaran 2003), stormwater mitigation (IBID) and also urban biodiversity (Gliessman 2015). Of course, these benefits are design-dependent, whereby less-intensive installations (e.g. raised beds) would show diminished building energy synergies compared to a building with the intensive UA. Lastly, building-integrated systems (non- and conditioned) create cultivable space out of the built urban form, providing a net increase in gross agricultural area; a benefit that the GB systems cannot accrue. Much like the ground-based-conditioned type, higher capital inputs generally restrict this type of farm to for-profit operation and lower community engagement in the examples that we have found (see: www.brooklyngrange.com and www.greencitygrowers.com).

3.4.4 Building-Integrated-Conditioned Systems

Building-integrated-conditioned systems are similar to the ground-based-conditioned systems in almost all aspects. Operational characteristics for these systems are identical to ground-based-conditioned farms, with the effect that passive and active sub-types must be included under this umbrella. Capital inputs are also very similar to the ground-based-conditioned type, except that structural reinforcement of the supporting building might be necessary. Urban symbiosis potential appears to be high since building-integrated-conditioned can directly couple with the energy and water systems of its host building. Symbiosis potential is not as high as building-integrated-non-conditioned since the common usage of hydroponics (nearly ubiquitous in order to provide high enough efficiency to offset capital costs) limits waste assimilation abilities, whilst challenges to large-scale stormwater assimilation are also prevalent due to structural costs and on-site storage capacity. Sanyé-Mengual and colleagues' (2012) work on building-integrated-conditioned grown tomatoes shows that despite substantial capital inputs, these systems can have superior environmental performance over conventional methods, though this was a result of reduced packaging and distribution spoilage, and less production efficiency. Again, the high capital and operating costs of these types of operations have largely limited them to for-profit operation with limited community engagement (see: www.gothamgreens.com or www.lufa.com).

4. Applying the UA taxonomy in future assessments

The dearth of quantitative studies of UA environmental performance hampers testing of the developed taxonomy, however we apply it to Sanyé-Mengual et al.'s (Sanyé-mengual et al. 2015; Sanyé-Mengual et al. 2015b) analysis of tomatoes grown on Barcelona rooftops. System 1 is a building-integrated –non-conditioned system using raised beds with soil substrate (Sanyé-mengual et al. 2015). System 2 is a building-integrated-conditioned passive system using hydroponics. Table 3 shows that the material and energy profiles align with the predictions of the taxonomy. Capital inputs are greater per-unit output for the conditioned system, with the exception of wood, though wood has substantially lower embodied environmental burdens relative to the steel and aluminum in the conditioned system. This was echoed by the lower contribution of the cultivation system to total climate change impacts for system 1 (<10%) relative to system 2 (~30%). Operational characteristics generally agreed with the UA systematics. Lower water demands contrast with higher energy demands for the system 1 (electricity for pumps), however, against our predictions, nutrient demands were higher for the conditioned system (particularly phosphorous), though the unaccounted nutrients in the soil and compost imported to system 2 might reverse this comparison. Yields are greater for the conditioned system (~25 kg/m^2) than the non-conditioned (13-14 kg/m^2). Contributions to the climate change impacts of the two systems also agree with the systematics: system 1's impacts stemmed from operational inputs, while capital inputs had a larger influence on system 2.

The previous example shows that our system, though simple, predicted the energy and material burdens of two UA systems, although it requires further tests of its robustness. However, after future verification, this taxonomy could emerge as a simple way to gauge the efficacy of UA as an urban design intervention to mitigate the environmental burdens of urban food provision. For instance the, urban designers looking to improve a city's environmental performance with UA could use the taxonomy to understand the various tradeoffs between the systems and answer questions about the appropriateness of technologies for a clime given the operational characteristics of a proposed system. With a larger base of studies to choose from, architects and designers could look at the types of produce that would fit within a local context given a chosen UA type, such as the choice to produce fruits over leafy greens in soil-based building-integrated-conditioned systems considering the lower yields of the latter (Sanyé-mengual et al. 2015). Next steps will involve building on the nascent QSAs that have shown the benefits and occasional shortcomings of UA in the environmental arena (Sanyé-Mengual et al. 2012; Kulak et al. 2013; Sanyé-mengual et al. 2015; Sanyé-Mengual et al. 2015b). Future studies could employ the life-cycle assessment (LCA) methodology used in the aforementioned study, material input per service, material flow accounting or any other number of methods to assess the environmental sustainability of product systems.

The taxonomy aligns particularly well with LCA for a number of reasons. Firstly, LCA is a tool for comparing the environmental performance of different systems delivering a comparable function. The UA systematics here describe four UA types with markedly different attributes, facilitating LCA studies to compare four varied ways to produce food in cities that cover the broad spectrum of current UA forms. LCA is also methodologically mature; with its own international standards, a discipline specific journal, significant industry application and wide set of indicators to assess environmental performance (climate change, eutrophication, land occupation, toxicity, etc.) (Finnveden et al. 2009).

4.1 Application to studies of urban systems

For researchers in urban systems, the growing interest in the environmental aspects of urban food procurement highlights the need for a standard lexicon with which to organize dominant UA types. Moreover, the systematics underlying the vocabulary should be compatible with the perspective of urban systems researchers, that of *urban metabolism*.

Urban metabolism is the sum of material and energy produced or imported, as well as waste produced by a city in order to support its daily activities (Kennedy et al. 2007). It is a rapidly maturing area of study that continues to see growing interest from governments as a benchmarking method of urban environmental performance and methodology to quantify the environmental changes imparted by an urban design or policy decision (Kennedy et al. 2010; Clift et al. 2015). Studies of urban metabolism are typically an accounting exercise of the material and energy flows using bottom statistical data or top down national economic data, which can then be coupled with other methods to gauge urban environmental performance (Goldstein et al. 2013). This raises another benefit of LCA as an UA environmental assessment tool; it is seen as the natural choice by urban researchers to couple with studies of urban metabolism (Chester et al. 2012; Goldstein et al. 2013; Clift et al. 2015).

By using material and energy flows as an organizing principle, the UA taxonomy can be easily coordinated within metabolism assessments of neighborhoods or cities, helping understand how an up-scaled UA system would interact with this metabolism to affect urban sustainability. Urban systems researchers have already looked at food and nutrient flows through cities, but prospective urban food production has not yet to be assessed, raising questions regarding UA's potential synergies (Sanyé-Mengual et al. 2014) and antagonisms with the larger urban environment (Pataki et al. 2009). The UA operational inputs, capital inputs and urban symbiosis potential that inform the typology begin to highlight the interconnectedness of the urban system and the built environment. The indicators for solid and liquid waste assimilation align well with the numerous urban nutrient flow studies (Færge et al. 2001; Billen et al. 2008; Kalmykova et al. 2012), since the varied capacities of the UA systems to absorb these streams could cause important shifts to this metabolic aspect in an agriculturally productive city.

In general, the taxonomy would allow for a scaled up test of UA's environmental sustainability. A study could use satellite imagery and software to identify available space for ground-based UA within a case city (see Taylor and Lovell 2012). Geographic information systems software could also determine the suitability of buildings for UA incorporation based on age and design. The different material and energy flows associated with the chosen UA systems could then be framed within the larger urban metabolism to predict the material, energy, food and waste regime of the altered system. Lastly, an LCA could estimate the environmental consequences of the new metabolism.

4.2 Shortcomings of the proposed taxonomy

One major disadvantage of the UA taxonomy is the small number of studies on which it relies and its anecdotal nature. Modern statistical methods that use significant sample populations to 'bin' like-types are the norm for developing typologies. This was the method promoted by Kostrowicki (1977) in his foundational paper, and has been employed by others in demarcating the different social and economic aims of UA (Dossa et al. 2011). Because of this shortcoming, the developed UA taxonomy is propositional in nature; able to evolve dynamically as new findings arise, or be cast aside if its utility is ultimately low.

Another major caveat is the proposed taxonomy is singular in focus, ignoring the equally important socio-economic characteristics of UA. There are many reasons to practice UA besides environmental food production; leisure, community building, education and food equity to name but a few (Sanyé-Mengual et al. 2015a; Thomaier et al. 2015). Decisions surrounding the implementation should equally weigh economic, social and environmental outcomes where relevant though this might not always be the case. For instance, social and environmental performance might be secondary to economic returns in private business scenarios, or environmental and economic performance might be secondary to community building for a more socially oriented project.

It is also by viewing UA projects with competing motives (primarily economic vs. primarily social) through the lens of the proposed UA taxonomy that interesting observations might emerge. For instance, what are the environmental tradeoffs between high-efficiency, high-input economically driven building-integrated, conditioned projects, and low-efficiency, low-input socially focused ground-based, non-conditioned operations? How do the auspices of an UA project affect environmental performance? Can we generalize their material and energy throughputs of these operations? These questions remain largely unanswered to date and warrant exploration if UA is going to scaled-up in cities, usually in concert with a larger environmental sustainability agenda.

5. Conclusion

The environmental impacts from food consumed in cities are large, but cities have design tools to address them. However, urban design interventions should be adopted after due consideration of whether they actually achieve the expressed goal of increasing the sustainability of urban systems. This review shows that if UA is to promoted on environmental grounds, then there remain a number of unanswered questions about the environmental performance of individual systems and less certainty regarding how an 'edible city' would perform. Where solid evidence does exist it has normally been performed on only one type of UA out of the panoply that exist, leaving a bric-a-brac picture of the larger environmental impacts of food production in cities. Significant differences in environmental performance of similar systems illustrate this well (Sanyé-mengual et al. 2015), and in at least one study UA was not preferable to conventional supply chains for specific products and methods (Kulak et al. 2013). Though environmental benefits may very well be conferred to UA adopting cities, it would be wise to test these assertions deeper before committing to scaling-up.

This paper compliments earlier work to develop a structured understanding of UA's environmental integrity. We have developed a taxonomy of four general UA types based on their operating characteristics, capital inputs and how they interact with urban systems. The types have significantly different behavior across these echoing the need for an organizing typology for and further assessment of UA. The proposed taxonomy is illustrative in its focus on important drivers behind the overall environmental performance of the UA systems, and covers the majority of UA operating styles. The typologies differentiate between material and energy loading, but not how these are provisioned, and therefore, sub-types exist within the derived framework for ‘organic’, ‘conventional’, ’integrated’ or other cultivation techniques. Nonetheless, keeping product and location the same, combining the framework with environmental assessment methods would allow comparisons of the relative environmental performance of UA systems or conventional urban food supply-chains. We have also outlined a path forward to apply the typology to a larger urban system to assess the environmental consequences of an altered urban metabolism through coupling with LCA, and better understand whether UA is in fact a good environmental initiative. Such an appraisal is essential at this critical juncture where a fecundity of UA cases exist for analysis but expensive and potentially deleterious experiments at the urban scale have not yet come to fruition.

Society should not solely seek technological fixes to the environmental challenges feeding an increasingly urban planet will entail. Simple actions such as reducing animal product consumption, increasing seasonal and local consumption and stymieing edible food waste will also having significant positive environmental benefits (Saxe 2014; Tilman and Clark 2014; Heller and Keoleian 2015). However, if cities can evolve to shoulder some of the burdens of their food provision, while concomitantly providing ancillary environmental, social and economic benefits to the city with UA, then this strategy is worth pursuing.

Table 1 - Summary of sustainability claims and quantitative support surrounding urban agriculture. Asterisks (*) indicate a field trial.

Sustainability Claim	Support
Building Energy	
Heating load reduction (Smit et al. 2001)	-Green roofs on Chinese buildings appreciably reduced heating loads, benefits diminished with building insulation (Kokogiannakis et al. 2011)* -Green roofs found to be beneficial in cold European climates (Jaffal et al. 2012) -41% heating energy reduction modeled with rooftop urban agriculture in northern climate (Delor 2011) -5% reduction through insulation, 79% reduction through air Exchange from rooftop greenhouse in Mediterranean environment (Ceron 2012)
Cooling load reduction (IBID; Ackerman, 2012; RUAF, 2010)	-Modeled 23% cooling reduction with rooftop greenhouse in Toronto, CA (Bass and Baskaran 2003)* -Indoor temperature and annual building energy reduced by 2°C and 6%, respectively (Jaffal et al. 2012) -Life-cycle building energy for diminished from 16% in extensive to 4% in intensive green roofs (Wong et al. 2003)
Urban Symbiosis	
Nutrient capture and recycling (RUAF 2006; Mougeot 1994; Specht et al. 2013)	-Wastewater recycling performed in African (Ruma and Sheikh 2010)**,** Asian urban agriculture (Khai et al. 2007)* and 1800s Paris (Barles 2007) -Compost application to urban agriculture in Cuba (Hernandez et al. 2014)* and UK (Edmondson et al. 2014)* -Rooftop farm in Paris utilized local food waste to generate a compost substrate (Grard et al., 2015)*
Rainwater capture and use (Havaligi 2011; Despommier 2010)	-Osmosis filtration and rainwater capture satisfied water needs of greenhouse in Manhattan, US (Nelkin and Caplow 2008)* -Hypothetical storm-water farm outside of Melbourne, AU had numerous benefits to local water management (Liebman et al. 2011) -Fertilecity rooftop greenhouse in Barcelona, ES collects rainwater for irrigation reducing water impacts by 98% compared to conventional tomato production (Sanyé-Mengual et al. 2015)*
Excess building heat utilization (Ackerman 2012)	-Modeled urban greenhouse showed potential benefits of using air from host building for heat in Barcelona, ES (Sanyé-Mengual et al., 2015) -Lufa Greenhouses in Montreal, CA utilizes energy of site building for heating*
Supply-Chain Efficiencies	
Reduced food-miles (Knowd and Mason 2006; Ackerman 2012; Specht et al. 2013)	-Local production around Osaka, JP reduced embodied energy in vegetables by 25% (Hara et al. 2013)
Improved yields (Smit et al. 2001; Despommier 2013a; Besthorn 2013)	-Urban greenhouse in NL provided improved yields above traditional agricultural for numerous products (Besthorn 2013)*
Reduced food waste (Sanyé-Mengual et al. 2012, 2015)	-Assumed 17% reduction in food losses over distribution (Sanyé-Mengual et al. 2012)

Reduced packaging (IBID)	-Packaging savings potentially reduce carbon footprint with urban agriculture in Barcelona, ES (Sanyé-Mengual et al. 2012, 2015)
In-situ Ecosystem Improvement	
Improved biodiversity (Knowd and Mason 2006; Havaligi 2011)	-Green roofs shown to increase local biodiversity (Hoffman 2007; Oberndorfer et al. 2007; Forman 2014)
Urban heat island attenuation (Pearson et al. 2010; Wong et al. 2003)	-Satellite models showed appreciable UHI reduction in New York City, US with hypothetical urban agriculture scenario (Ackerman 2012) -50% green roof cover could reduce ambient temperatures by 2°C in Toronto (Bass and Baskaran 2003)
Storm-water attenuation (Ackerman 2012; Sida 2003)	-Significantly slower runoff rate and runoff retention observed at green roofs around North America (Oberndorfer et al. 2007)* -Green roof significantly mitigated runoff in Mediterranean (Fioretti et al. 2010)*
Soil quality (Smit et al. 2001; Jansson 2013)	-Compost on UK urban agriculture improved soil structure and nutrients (Edmondson et al. 2014)*
Air quality Improvement (Hampwaye 2013)	-Models linked urban forest cover in China (Jim and Chen 2009)and Green roofs in Chicago (Yang et al. 2008) with reduced local NOx, SOx, O_3 and particulates
Ex-situ Ecosystem Improvement	
Carbon sequestration (Sida 2003; Despommier 2013b)	-Urban green infrastructure in Toronto, CA and Salt Lake City, US sequestered <1% of urban CF (Kennedy 2012; Pataki et al. 2009)
Carbon Footprint Reduction (IBID)	-Significant greenhouse gas reduction for urban agriculture except for polytunnel strawberries (Kulak et al., 2013) -Rooftop greenhouse tomatoes in Barcelona showed lower embodied carbon than conventional supply chain from 33% (Sanyé-Mangual et al., 2015) to 63% (Sanyé-Mengual et al., 2012)
Lower ecological footprint (RUAF 2006)	-None encountered
Improved biodiversity (same as above)	-None encountered
Soil quality (same as above)	-None encountered

	GB-NC	GB-C	BI-NC	BI-C
Example	Edgemere Farm: http://www.edgemerefarm.org	Bright Farms: http://www.brightfarms.com	Brooklyn Grange: http://brooklyngrangefarm.com	Gotham Greens: http://gothamgreens.com
Operating Characteristics				
Water Use Efficiency	Low: runoff and evaporation	High: contained environment with recycling	Moderate: runoff control possible	Same as GB-C
Nutrient Use Efficiency	Low: lost in runoff or bacterial digestion	High: contained environment with recycling	Moderate: lost in runoff or bacterial digestion	Same as GB-C
Potential Soil Erosion	High	Low	High	Low
Light Inputs	None	*Passive:* None *Active:* At night or overcast	None	Same as GB-C
Heat Input	None	*Passive:* None *Active:* At night or winter	None	Same as GB-C
Other Energy Inputs	Low	High: pumps, computers, louvers	Low to High: potentially pumps, computers	Same as GB-C
Substrate	Soil	Soil or soil-less	Soil or soil-less	Soil-less
Pest Risks	High	Low	High	Low
Ambient Pollution Risk	High	Low	High	Low
Growing Season	Seasonal: extended with hoop houses	Year-round	Same as GB-NC	Year-round
Yields	Low	High	Low	High
Capital Inputs				
Supporting Structure	None	None	Low to High: reinforcing building or adding extra capacity to new building	Same as BI-NC
Conditioned Space	Low: potentially small hoop houses	High: greenhouse frame and cladding	Same as GB-NC	Same as GB-C
Roof Protection	None	None	High: Root barrier, waterproof membrane and drainage layer	Moderate: Potentially waterproof membrane
Substra	None to Low:	Low to High:	Same as GB-NC	High:

te Containers	potential raised beds	raised beds or hydroponic tables/towers		hydroponic tables/towers
Irrigation Equipment	Low to medium: potentially distribution system and pumps	High: pumps and distribution system	Same as GB-NC	Same as GB-C
Mechanical Components	Low	High: motors, fans, heater/air-conditioning	Low	Same as GB-C
Computers and Sensors	Low	Low to High	Low to High	High
Urban Symbiosis Potentials				
Solid Waste Assimilation	High: compost	Low to High: compost derived nutrient solution or compost	Moderate: compost (within roof's capacity)	Low: compost derived nutrient solution
Liquid Waste Assimilation	Low	Low	High: direct access to building gray/black water	Same as BI-NC
Building Energy Coupling	Low: indirect UHI mitigation	Same as GB-NC	Moderate: increase roof albedo, insulation	High: same as BI-NC, evaporative cooling, heat capture in winter
Runoff Mitigation	High: over entire site	Low: limited rainwater capture	High: over entire site	Low to Medium: potential recycling from building gutter
Other General Traits				
Economic Motivation	Likely non-profit or supplemental income	Likely profit driven	Very likely profit driven	Very likely profit driven
Community Engagement	High	Low to High	Likely low	Likely low

Table 2 - Properties of the developed urban agriculture typologies: ground-based-non-conditioned (GB-NC), ground-based-conditioned (GB-C), building-integrated-non-conditioned (BI-NC) and building-integrated-conditioned (BI-C). Note that the GB-C and BI-C have passive and active sub-types for light and heat inputs.

Figure 1. (A) ground-based-non-conditioned, (B) ground-based-conditioned, (C) building-integrated-non-conditioned and (D) building-integrated-conditioned systems in the Northeastern United States. First author's own photographs.

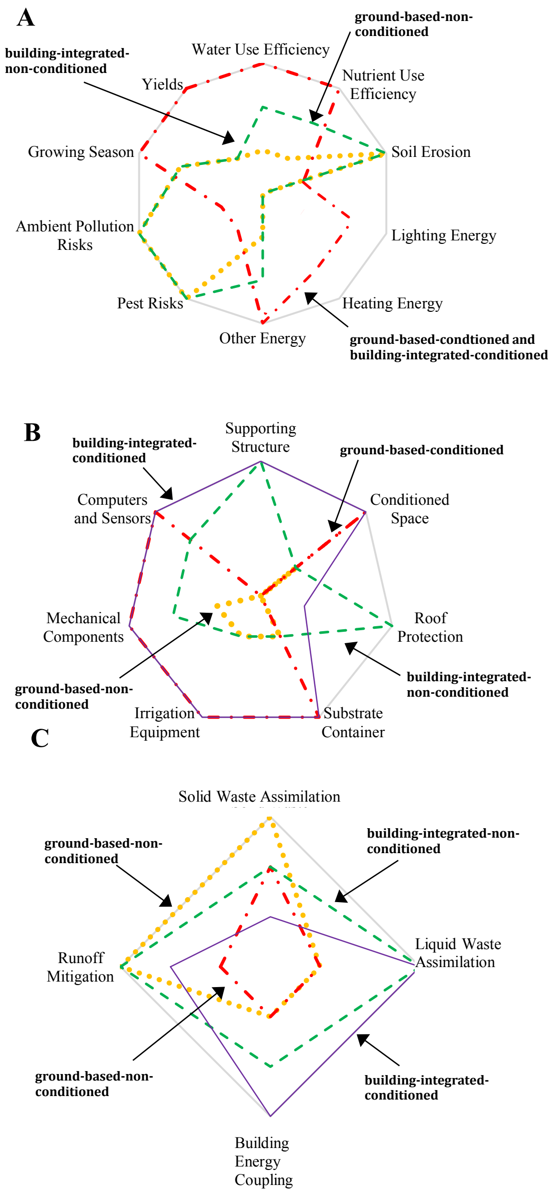

Figure 2 - A) Operational characteristics B) Capital Inputs and C) Urban symbiosis potential. Note that in (A) conditioned types *can* have a large operational energy inputs, though this might be contingent on the climate and lattitude, which would affect the amount of external energy to supply lighting and heating. Comparisons based on the site-visits to urban farms in 2015, the results of Sanye-Mengual et al. (2015a, 2015b), and the findings of Kulak et al. (2013).

Table 3: Comparison of a building-integrated-non-conditioned operation (system 1) with a building-integrated-conditioned (passive) operation (system 2). CO2 eq. represents the equivalent amount of CO_2 to have the same radiative forcing effect on the atmosphere as the greenhouse gases released during the production of the tomato. Cumulative energy demand is the total amount of energy embodied within the production of materials and energy directly consumed by the UA system. As can be seen, the capital inputs and energy demands of the conditioned system are higher. Water use is lower for the conditioned system. The energy and capital inputs for the conditioned system make it less favorable in terms of carbon footprint, despite the higher yields.

	Unit	**System 1:** building-Integrated-non-conditioned (Sanyé-mengual et al., 2015)	**System 2:** building-integrated-conditioned (Sanyé-Mengual, Oliver-sola, Montero, & Rieradevall, 2015)
Capital Inputs			
Metals	kg/kg tomato	0.004	0.037
Aluminum	kg/kg tomato	-	0.003
Steel	kg/kg tomato	0.004	0.034
Biomaterials	kg/kg tomato	0.26	-
Wood	kg/kg tomato	0.25	-
Bamboo	kg/kg tomato	0.01	-
Plastics	kg/kg tomato	0.0002	0.017
LDPE	kg/kg tomato	0.0001	0.004
HDPE	kg/kg tomato	-	0.004
Polycarbonate	kg/kg tomato	-	0.006
Polyester	kg/kg tomato	-	0.0003
Polystyrene	kg/kg tomato	-	0.001
Polypropylene	kg/kg tomato	0.001	-
PVC	kg/kg tomato	0.0003	0.002
Aggregates	kg/kg tomato	-	0.02
Perlite	kg/kg tomato	-	0.02
Operational Inputs			
Water	m^3/kg tomato	0.5	0.03
Electricity	kWh/kg tomato	0.0002	0.04
Fertilizer (N)	g/kg tomato	0.33	0.39
Fertilizer (P_2O_5)	g/kg tomato	0.25	2.47
Fertilizer (K_2O)	g/kg tomato	0.53	0.76
Compost	g/kg tomato	16	-
Soil	g/kg tomato	155	-
Outcomes			
Yields	kg tomato/m^2	13-14	25
Carbon Footprint	kg CO_2 eq./kg tomato	0.068-0.075	0.22
Cumulative Energy Demand	MJ/kg tomato	1.14-1.26	3.25

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Appendix C

Article 3: Testing the environmental performance of urban agriculture as a food supply in northern climates

Title: Testing the environmental performance of urban agriculture as a food supply in northern climates

Authors: Benjamin Goldstein[a*], Michael Hauschild[a], John Fernandez[b], Morten Birkved[a]

[a]Technical University of Denmark, Quantitative Sustainability Assessment Division Producktionstorvet, Building 424, Kongens Lyngby, Denmark, 2800

[b]Massachusetts Institute of Technology, Department of Architecture
77 Massachusetts Avenue, 5-419, Cambridge, United States, 02139

*Corresponding Author: Technical University of Denmark, Quantitative Sustainability Assessment Division, Producktionstorvet, Building 424, Room 225, Kongens Lyngby, Denmark, 2800 T: +45 45 25 45 61 E-mail: bgol@dtu.dk

Abbreviations[1]

1. Introduction

Food consumption is a major driver of a city's total environmental burdens; often on par with mobility, building energy and construction activities (Goldstein et al., 2016a). By virtue of their majority shares of both population and wealth, cities consume the bulk of global food, the production of which is a leading cause of greenhouse gas (GHG) emissions, natural habitat appropriation, chemical pollution (nutrients and pesticides) and water consumption (Foley et al., 2011; Gliessman, 2015). Agriculture is also resource intensive; dependent on non-renewable fossil fuels and minerals for agrichemicals to meet growing food demands on approximately 36% of the globally available ice-free land, with scarce room for sustainable expansion (Foley et al., 2011; Steffen et al., 2015). For cities to become sustainable, environmental impacts from their food demands must be reduced, especially considering predicted urbanization, economic development and increasing consumption of environmentally-burdensome animal-proteins for an increasing share of humanity (Goldstein et al., 2016a; Tilman and Clark, 2014).

Urban agriculture (UA) , *the production of food in and adjacent to cities, leveraging pre-existing urban material energy flows as production factors* (Koc et al., 1999), is commonly touted as an urban design solution to the environmental impacts of urban food needs (IPCC, 2014; Pearson et al., 2010). A recent review by Goldstein et al. (2016c) found that UA is posited to have numerous advantages over conventional agriculture that will supposedly result in UA's superior environmental performance, grouped here into three categories:

1. Supply-chain efficiency; reduced distance from farm to consumer ('food miles'), attenuating overall environmental burdens from production and distribution;
2. Urban symbiosis potential; interacting with a city's material and energy fluxes, reducing a farm's operational inputs, absorbing urban waste flows (e.g. food waste), lowering building energy demand (i.e. through insulation or reducing the urban heat

[1] UA – urban agriculture, BI-C – building-integrated-conditioned, BI-NC – building-integrated-non-conditioned, GB-C – ground-based-conditioned, GB-NC – ground-based-non-conditioned, LCA – life cycle assessment, IP – impact potential , CC – climate change, FE – freshwater ecotoxicity, ME – marine eutrophication, WRD – water resource depletion, LU – land use, RD – minerial and fossil resource depletion

island effect) and other local environmental benefits (e.g. tempering stormwater runoff);

3. Ex-situ environmental benefits; supposed reductions in agricultural land occupation, carbon sequestration and other benefits to ecosystems beyond the city boundary.

Despite the fanfare, a paucity in evidence exists where literature reviews of UA been performed (Born and Purcell, 2006; Goldstein et al., 2016c; Specht et al., 2013).

Sanye-Mengual and colleagues' recent work on UA in Barcelona, ES has started addressing this, comparing the environmental performance of rooftop greenhouse tomatoes against conventional supply chains, finding that the former can have lower life-cycle GHG emissions and toxicity impacts (2015b, 2012). Rothwell and colleagues also found that lettuce from local farms could reduce GHG impacts compared to conventional produce in Sydney, AU (2015). Though promising, both studies considered UA in warm climes, more amenable to food production than many of the wealthy, northern cities where food related environmental impacts are typically highest and UA is often promoted to reduce these burdens.

Year-round UA in colder cities will likely rely heavily on controlled agriculture (i.e. greenhouses) to produce food to potentially negative environmental results. Leafy greens from a Japanese automated, conditioned, indoor farm ('plant factory') produced lettuce at 6.4 kg CO_2 equivalents per kg fresh lettuce, well above conventional production due to the system's energy demands (Shiina et al., 2011), due to the high energy requirements for 100% artificial lighting and temperature control. Kulak and colleagues found that low-tech greenhouse UA strawberries in London, UK , had a higher carbon footprint than conventional counterparts (2013) showing that embodied impacts in capital and equipment can also drive burdens.

Our contention here is that UA might not always have the intended positive effect on a city's environmental performance, particularly in a northern context. The generality of UA as an environmentally preferable urban food supply chain is questionable, since its three general environmental benefits may be largely contextual. UA's environmental efficacy is a pressing question in northern cities considering its renaissance at a grassroots level (Mok et al., 2014), active promotion by many northern cities through the Milan Urban Food Policy pact (City of Milan, 2015) and codification in land use policies (City of Boston, 2014). Further consideration of UA's environmental performance could help balance what has hitherto been a pro-UA narrative and assuage data gaps in an evolving dialogue. In this study we test the performance of six UA systems, covering four distinct UA types, in Boston, US using environmental life cycle assessment (LCA), to see whether UA is a true environmental benefit to Boston, and by proxy, similar cities.

2. Methods

LCA is applied here to compare the potential environmental benefits of lettuce and tomato production with UA and conventional farming. Here we focus on the aspects most relevant to the study at hand. For a richer treatment of LCA methodology, see existing standards (ISO, 2006a, 2006b) and the European Commission's LCA handbook (European Union Joint Research Council, 2011).

2.1. Modelling Framework

Process-based LCA is applied here using detailed data for the processes throughout the life cycle (e.g. fertilizer application, freight transport, etc.), maximizing geographic, temporal and technological representativeness. A consequential LCA (CLCA) approach is used here as opposed to attributional-LCA (ALCA). CLCA models consumption as a mix of ‘unconstrained suppliers’ that respond to the next unit demand (‘marginal producers’), not the average mix of historic suppliers as in ALCA (Weidema et al., 2013). In keeping with CLCA practice, we model multi-functional processes using system-expansion, and not the allocation approach of ALCA. For example, rooftop farms produce food and also minimize the energy consumption of the building on which they are situated; CLCA credits the farm for the avoided energy consumption, while ALCA uses economic value to allocate the environmental burdens between the energy savings (taken as the price of the energy that would have been consumed otherwise) and the value of the food. CLCA aligns best with ISO recommendations (ISO, 2006b) and is implemented here using the ecoinvent 3.1 database embedded within SimaPro 8 product system modelling software.

2.2. System boundaries and functional unit

Our scope is cradle-to-shelf; cultivation, harvesting and distribution of food to market are modeled. This is justified given that these aspects of the product system are controlled by producers and the study aim of comparing relative environmental performance (post purchase transport and preparation are identical and can be excluded).

Tomato and lettuce production using UA and conventional methods are modeled here. The functional unit *is 1 kg of fresh food item delivered to the point of purchase in Boston*, for each item (tomato or lettuce). Tomatoes and lettuce were chosen as subjects of study due to their prominence in the North American farming system and diet. According to the FAO (2016), in the US tomatoes and lettuce account for 14% (2nd) and 10% (3rd) harvested vegetable area, respectively, and 37% (1st) and 10% (3rd) harvested vegetable mass, respectively. In the US diet they both the most consumed fresh vegetables by mass, with 21% a piece (Heller and Keoleian, 2015).

2.3. UA cases and life cycle inventory

Goldstein et al. (2016c) identified four overarching UA types based on material and energy regimes and expected disparate environmental performance: building-integrated-conditioned (BI-C), building-integrated-non-conditioned (BI-NC), ground-based-conditioned (GB-C) and ground-based-non-conditioned (GB-NC). Building-integrated identifies whether the farm is standalone or physically attached to a building, while conditioning refers to control of growing space variables (light, temperature, CO_2 levels, etc.) System particularities are outlined in Table 1, while detailed descriptions can be found in the aforementioned review.

Six urban farms were assessed; four in Boston, one in New York City (assumed to operate in Boston), and a single hypothetical rooftop greenhouse in Boston. Six examples were chosen to capture the potentially disparate environmental performances between architectures as hinted by Sanye-Mengual et al. (2015) results, allowing a richer discussion of UA’s environmental performance, departing from previous work on single UA types. Table 2 outlines the studied farms’ characteristics. Two of the UA cases (GB-NC1 and GB-C1) produce leafy greens (arugula) instead of lettuce. We assume here that leafy greens are directly substitutable for lettuce, fulfilling the same function (salad or sandwich topping), despite potential nutritional mismatches in terms of calories, micronutrients, macronutrients, etc. This is acceptable since the assessment is performed on a mass basis, but future studies could look into the environmental impacts per unit of nutritional value generated, as has been recommended by Heller et al. (2013).

The LCIs for the studied UA operations were based on primary data collected through site visits, interviews, financial records and, where necessary, estimation. For the BI-C, a rooftop greenhouse designer was consulted to develop a reasonable facsimile of the operation, complimenting this with publically available data from an operating rooftop greenhouse in

Montreal, Canada. For detailed case descriptions and LCIs see appendices A-F. There were some important data gaps that could not be filled. Information for structural buttressing of the supporting buildings for both BI systems was not available. A licensed structural engineer was employed to make reasonable estimates of these inputs based on site photos (see appendix A), but the findings should be viewed with this caveat in mind. The lack of a participating BI-C farm in the project also meant that the LCI for this system was built using expert input from a greenhouse designer and publically available information. Finally, data was not available for irrigation in many instances, which meant this information was estimated.

The UA produce are compared to conventional tomato and iceberg lettuce production. Tomatoes and lettuce were modeled from the 'Tomato {GLO}' and 'Lettuce {GLO}' unit processes in the ecoinvent 3.1 database, respectively. These processes represent LCIs for current conventional farming technologies in Europe which will result in an overestimation of heating inputs, it is assumed that they are technologically representative of North American production (Stoessel et al., 2012). Of note is that the conventional tomato used here was modeled on a heated greenhouse. An unheated greenhouse would improve the relative performance for energy related impacts, though the countering effect of the lower yields remains unknown. In the same vain, field tomatoes have significantly lower yields than their greenhouse counterparts, which would have an impact on land use related indicators. Nonetheless, keeping these study limitations in mind, the conventional cases should provide robust enough yardstick of comparison to test UA's environmental performance.

Distances from conventional farms are taken as weighted average source distances for US tomatoes (2550 km) and lettuce (2962 km) to Iowa (Pirog and Benjamin, 2003), since food miles for the Northeast US are unknown. These values represent intermediate estimates considering the coast to coast distance of United States (~5000 km). Post-harvest, pre-consumer losses of 11% are assumed for both products (USDA, 2014). For more information on the conventional produce see appendix G.

2.4. Impact categories included

Six metrics that are broadly representative of agriculture's environmental impacts were included: climate change (CC) from agricultural land expansion, energy inputs, enteric fermentation; freshwater ecotoxicity (FE) from fertilizer and biocide application; marine eutrophication (ME) from fertilizer application; water resource depletion (WRD) from irrigation; land use (LU) from agricultural land expansion and degradation; and mineral, fossil and renewable resource depletion (RD) from agrochemical consumption. There exists numerous methodologies to convert LCIs to impact categories of potential impacts (herein 'impact potentials' or 'IPs'), opting here for the ILCD method.

It should be noted that the study boundary of the point of purchase ignores potential contamination in the UA from local pollution and its adverse effects on human health. Though human health IPs were not assessed here, precluding contamination impacts from showing up in the results, it is important to note that UA is susceptible to local contamination, either in soil or from aerial deposition (Säumel et al., 2012; Wortman and Lovell, 2013), hinting that this may be an important factor in future LCAs of UA.

3. Results

Table 3 shows the results for the UA and conventional product systems for tomatoes and lettuce across all impact categories, while appendix H situates these relative to previous studies. No UA system is superior to conventional production across all impact categories, although select UA systems may appear preferable (based on equal weighting of considered impact categories) in that they have lower IPs for a majority of impact categories. The reasons for the disparate performance vary by system, but trends exist. For conditioned UA systems, energy consumption for space conditioning drives most IPs; CC (> 90%), FE (> 70%), ME (> 80%) and RD (> 70%) for both tomato and lettuce UA systems. Capital inputs seldom mattered with the exception of the BI-NC system's structural steel which was more than half of the IPs for CC, FE, ME, LU and RD impact categories. GB-NC systems are inefficient in land and water use, but low intensity for other aspects. Between tomatoes and lettuce, UA generally performs better when producing the former, as the greater yield of tomatoes per unit area ensures that capital and energy inputs, which are applied evenly across the case farms, are best utilized, reducing IPs. The following sections detail the findings for the food products.

3.1. Tomatoes

Figure 1 outlines findings for tomatoes. UA IPs are classified as related to capital inputs (equipment and structures), operational inputs (supplies and distribution) and urban symbiosis (interaction between farm and built environment's material and energy fluxes). IPs for conventional tomatoes are classified as related to cultivation or distribution.

UA was found to be ubiquitously superior for freshwater ecotoxicity, FE and marine eutrophication, ME (Figures 1b and c), a consequence of the use of inorganic fertilizers for nutrients and pesticides during conventional tomato production. All the UA cases generally avoid pesticides using beneficial insects for pest control, though GB-C1 was applying small doses of natural pesticides to combat aphids (this was not modeled, decreasing this IP). Fertilization levels were generally low for the UA cases (typically with fish emulsion), recycled in closed hydroponic systems (BI-C) or even zero (GB-NC1), though natural gas for operational energy elevates the conditioned urban farms above their counterparts for ME due to NOx emissions. Notably, BI-NC's FE and ME IPs deviate from most agricultural LCAs, driven not by operations, but capital; structural steel affects FE while natural gas to produce expanded shale/clay growth medium drives ME.

Conversely, for both water resource depletion, WRD and mineral and fossil resource depletion, RD (Figures 1d and f) the UA systems generally perform equally or worse than the conventional tomato cultivation. WRD, driven by irrigation, puts inefficient soil based systems at a handicap compared to the BI-C and market hydroponic systems. Of note is that rainwater irrigation, reducing municipal water demands, nonetheless deprives the surrounding catchment of water; depleting local water resources. This captured rainwater is called 'green water' in the water footprint method of embodied water impacts (Mekonnen and Hoekstra, 2011), its presence here highlighting that harvesting flows from the built-environment can come at a price, potentially reducing ecosystem quality or redirecting potentially potable water. For RD, the main driver of the UA systems' poorer performances varies. For the BI-C it is the natural gas for operational energy. For the BI-NC system it is the capital inputs that drive RD IPs, particularly additional steel for structural buttressing. The GB-C1 is influenced strongly by produce distribution, which despite the short distance, is done in small batches by pickup truck resulting in high capital inputs for the vehicle, contradicting claims that reducing 'food miles' is a universal environmental good (Born and Purcell, 2006; Sanyé-Mengual et al., 2012). Contributions from the greenhouse structure (steel) and irrigation (piping in municipal water system) also hamper the GB-C1 system. Similar to the GB-C1 the GB-NC1 RD IP is driven by distribution, and less so, the plastics for the irrigation system. The GB-

NC2 performs on par with the conventional tomato, having minor impacts for nutrient demands.

For climate change (CC) IPs (Figure 1a), no general pattern between UA and conventional production is seen. Energy space heating leads to discouraging CC results for the conditioned UA types. Clearly heating greenhouses to grow tomatoes over winter in a northern clime has a high energy cost, with commensurate CC IPs if fossil fuels are relied on, as in Boston (natural gas as marginal fuel for heat and electricity). Comparing these greenhouse tomatoes to those from earlier LCAs in appendix H shows that our results align well with prēvious findings (1.27-1.97 kg CO_2 eq/kg tomato in heated greenhouses). Moreover, the BI-C is modeled as procuring half of its heat through symbiosis with its host building (in the same manner as Montreal farm on which selected system aspects were taken), hinting at the true unsuitability of this UA practice in Boston where free, dissipative energy is lacking. Alternatively, both GB-NC farms are markedly superior to conventional practices for CC IPs, a consequence of their minimal capital and energy inputs. Once again the BI-NC stands out: embodied carbon in the structural steel has significant CC IPs, but these are offset by energy savings at the host building (assumed 3% of heating and 5% of cooling), elucidating UA's potentially meaningful mutualisms with the urban system. The land use (LU) IPs in Figure 1e unexpectedly showed that BI UA, though ostensibly devoid of direct land use, nonetheless has substantial indirect land use. For the BI-C, land occupation for natural gas extraction is substantial, while the BI-NC is affected by structural steel (mine infrastructure, energy inputs) and natural gas used in producing expanded clay media. The conventional tomato, also reliant on natural gas as heating source, has similar LU IPs for the production stage as the BI-C, while also having burdens for road area and diesel during distribution. The GB-C1 LU is a mix of natural gas demands, direct urban LU and greenhouse capital (wood and steel). Lastly, for the GB-NC farms, the combination of direct land occupation and low yields lead to elevated LU IPs above conventional tomatoes.

3.2. Lettuce

Figure 2 shows the IPs in the six impact categories for the production systems broken down in the same manner as in Figure 1. The lettuce results diverge markedly from the tomato results in that across the different sites UA does not perform consistently better than conventional agriculture for any single indicator. For CC IPs the conditioned systems are again hampered by heating demands and accompanying natural gas inputs (GB-C2 incurs additional penalties for 100% artificial light demands), which when combined with the low yields of lettuce, result in CC IP levels similar to red meat (Nijdam et al., 2012). Non-conditioned ground-based farms perform slightly better than the conventionally cultivated market lettuce for the same reasons as the tomatoes. Lettuce results in FE and ME mirror each other. Energy inputs to the conditioned systems results in high IPs, while the capital inputs for the BI-NC farm mar its performance. The GB-NC farms perform better than conventional lettuce cultivation due to the latter's fertilizer and biocide use. For WRD only the GB-C2's closed-loop hydroponics performs comparably to market lettuce, with all of the others consuming significant irrigation volumes, as municipal water or 'green water'. All of the ground-based UA farms' LU IPs are elevated above conventional lettuce; with GB-C1, GB-NC1 and GB-NC2 all driven by the same inputs as the tomato production. GB-C2 occupies very little land directly, but its high electricity consumption makes it a prohibitively costly for LU. Lastly, none of the UA systems provide an attractive alternative to conventional lettuce for RD where the same pattern as for the tomatoes continues for the BI-NC, GB-C1, GB-NC1 and GB-NC2 systems, while the GB-C2 performs poorly due to space conditioning and equipment.

4. Discussion

The results confirm the intimations of a number of earlier reviews of UA (Goldstein et al., 2016c; Mok et al., 2014; Specht et al., 2013) that UA has the potential, in certain contexts, to be a far more environmentally damaging food source than conventional agriculture. Furthermore, the performance of UA is as varied as the types that exist, hinting at the need for a more nuanced discussion of UA, one that departs from the pro UA bias that has heretofore dominated the discourse. In the following sections we re-visit the three themes raised in the introduction (supply-chain efficiency; urban symbiosis potential; ex-situ environmental benefits) to see how our results align with these claims. To balance the discussion we also touch on some of the non-environmental considerations of UA.

4.1. UA and supply-chain efficiency

A primary argument for UA is reduced 'food miles' and the belief that this makes food more environmentally sustainable (Born and Purcell, 2006; Weber and Matthews, 2008). Here we have shown that in Boston's case, this is not a defensible claim. Firstly, the food miles argument overestimates the importance of transport from a life-cycle perspective, which was never a dominant driver for either conventional case here, as supported by other LCAs and reviews of the topic (Born and Purcell, 2006; Garnett, 2011), although expanding the scope to a full life-cycle, including consumer transport, might affect this. Bulk-freight by ground over long distances, though certainly imparting environmental burdens, is relatively efficient on a per mass basis. The RD IP for the BI-C1 displays this through the significant contribution from distribution using a pickup truck, despite the short intra-city distances.

By overemphasizing the importance of transport, UA advocates underestimate the environmental impacts of producing food (Garnett, 2011; Weber and Matthews, 2008), which is where the majority of burdens lie. Efficient use of agricultural inputs is intricately related to system performance, and in this regard the conventional cases appear to make for leaner supply chains than many of the UA alternatives in Boston. Table 4 explores this, showing per square-meter performance for the different production systems for CC and WRD IPs, as well as cumulative energy demand (sum of all direct and indirect energy during production).

From this perspective one can see that the conventional systems, despite their considerable cumulative energy (heating, agrochemicals and capital) and irrigation demands relative to the UA options produce in substantial volumes to make for environmentally superior systems across a number of IPs. Where UA does provide a more sustainable substitute for the conventional cases, the UA system falls within one of two scenarios; very low inputs or high inputs matched by high yields. The GB-C2 system producing lettuce provides an example of the high input-high yields nexus for WRD IPs. Looking at the system it is clear that per unit growing space, the GB-C2 operation has the highest WRD IPs relative to other UA systems. Countering the substantial irrigation demand is a high yield with the end result that it is the only UA system out of the five that competes with conventional lettuce by this metric as visible in Table 3. Conversely, the cumulative energy demand, driven by heating and lighting, is too great to be offset by the system's efficiency, resulting in the system's elevated CC IPs.

The GB-NC systems both display the benefits of having low inputs. For instance, the GB-C1 and GB-NC1 have the same yield of lettuce, but the high energy inputs of the former result in a CC IP that is orders of magnitude larger than the latter, since GB-NC1 has almost minimal direct energy requirements over its lifetime. Therefore, it might appear that UA in northern climates is best suited for summer production, where heating needs are negated, in line with previous studies in mild climates (Rothwell et al., 2015; Sanyé-Mengual et al., 2015b). Despite their positive performance in a number of areas low-input UA systems have very high LU IPs, exacerbated by the lengthy cold periods of the year where the land is unused. Boston land is amongst the priciest in the United States (Davis and Palumbo, 2008), while the percentage of income spent on food by Americans is amongst the lowest globally (FAO, 2016), hampering the economic tractability of these systems in Boston, or similar markets. The low efficiency of GB-NC UA hints at why it is primarily applied on patches of underutilized municipal land, as a usufruct exercise between private owner and community-members prior to development or as a food source in shrinking cities with abundant space (e.g. Detroit) (Smit et al., 2001).

4.1.1. **Different Energy Scenarios**

Given that UA will likely have to produce in large quantities in order to compete with other uses of space in the city, it is worth investigating whether high yield UA systems, despite their relative inefficiency, could provide a sustainable alternative to conventional systems in Boston or other northern cities using alternative energy sources. We re-performed our analysis on the high-yield cases (BI-C and GB-C2), comparing the conventional lettuce and tomato to these systems with the existing marginal energy source (natural gas) replaced by photovoltaic, on-shore wind and hydroelectric power. The conventional production system was not altered since electricity is not a major input. We only show results for the CC, LU and RD IPs since these are most tied to the energy consumption in the previous assessment. Results are shown in table 5.

Grid changes profoundly affect the results. For the BI-C tomato the wind and hydro power options clear preferences over the conventional tomato. The photovoltaic powered BI-C, though an improvement in terms of CC IPs, is a step backwards for RD (metals in PV-panels), and unsurprisingly considering the energy demands, LU; once again calling into question the claim of UA’s ability to lower the land usage of agriculture. Despite the hydro tomatoes’ strong performance, hydro power in the Northeast Power Coordinating Council (NPCC) region is not expected to grow within the operational lifetime of the BI-C project (U.S. Energy Information Administration, 2015), with natural gas remaining the marginal fuel for the foreseeable future. Electricity generation from wind and solar power in the Northeast US are likely to grow in coming decades, plausibly supporting UA production of tomatoes that is preferable to the status quo.

The analysis for the lettuce deviates from the tomato. Although hydro is clearly the best choice amongst the energy sources, as mentioned above it is not a realistic marginal power source for Boston. Photovoltaic, though a marked improvement from the original analysis is still worse than conventional lettuce, while a farm utilizing wind power appears to match conventional lettuce for CC and RD, and performing slightly better for LU, offering the potential for this design to reduce the burdens of urban food demand within this constrained context. Since only energy intensive forms are really competitive in terms of yields (and hence LU), the marginal electricity source is crucial. If we have a future with unrestricted access to renewable energy, intensive UA may compete with conventional, though this is currently hypothetical.

4.2. Urban symbiosis of UA and scaling up

Another frequent claim in UA literature is the practice’s ability to be weaved within the urban fabric and affect pre-existing material and energy flows to reduce inputs to the farm and positively affect the urban environment (Goldstein et al., 2016c). Examples include solid (food) and liquid (toilet and kitchen water) waste assimilation, energy exchanges between farm and host building, runoff attenuation and mitigating the urban heat island effect (IBID). Where they were present in this study, these mutualisms seldom resulted in large benefits for the UA systems. Energy exchanges with the host building was one exception; BI-NC energy savings to the host building, meager as they are, counteracted the embodied burdens of the structural steel. For instance, the CC IP for tomato production was reduced by 77% from a no-energy-savings scenario, while ME and LU were both reduced by about 20%. Bootstrapping on the dissipative energy of the host building, the BI-C reduces heating needs by 50%, essentially halving the CC IP for this system (2.14 instead of 4.11 kg CO_2 eq./kg tomato), while also showing that such synergies cannot overcome the system's low energy efficiency, thought shifts towards renewable energy systems in the future would reduce these advantages. Rainwater capture and subsequent runoff avoidance, though beneficial to the systems employing it, does not translate into significant environmental savings in this study, evidenced by the small, negative, black bars in Figures 1 and 2 (less than 1% across all IPs for BI-C and GB-C variants). The rainwater capture did have significant affect some IPs for the GB-NC farms (CC: 32-72%, FE: 18-43%, ME: 27-60%, RD: 13-32% reductions), but in an absolute sense the actual reductions of the IPs are small since the environmental burdens of these systems are very low for these impact categories. Moreover, this rainwater capture did nothing to ameliorate the water stress caused by irrigation since UA diverts water from other anthropogenic or ecosystems uses.

The use of compost by the farms, though a potentially meaningful synergy between the city and farm, does not affect the results. This is because composting benefits (avoided landfilling, avoided fertilizer production) are given to the original waste generator who decided to forego landfilling. In other words, the mere presence of UA in Boston was not a driver of compost production, and hence, from consequential LCA thinking is not credited with related benefits. However, if UA were scale up within cities, inducing a market, allocating the benefits of composting to an UA operation would be justified. An existing example is the use of sewage sludge in UA in developing countries, where the production systems are prevalent enough to act as continual repositories for the waste, providing a disposal route where other options are absent (Qadir et al., 2010), though challenges of pathogenic contamination posed by low-tech nutrient capture techniques cannot be downplayed (Srikanth and Naik, 2004).

Looking at single UA sites it is difficult to gauge the UA's latent ability to affect large scale change. If densely applied throughout a city, scaling effects could manifest, whereby some of the espoused UA's benefits, such as urban heat island mitigation (Pearson et al., 2010), attenuation of runoff from rain events (Ackerman, 2012) and local biodiversity increases (Havaligi, 2011) result in substantial changes to city's environmental performance. Most salient is whether a city-wide food production system would produce in appreciable volumes to satisfy a substantial proportion of urban food demands. Studies have been varied in their findings in this arena. An assessment of Oakland, US found that intensive utilization of suitable open space in the city by UA would only supply ~1% of the city's fruit and vegetable needs (McClintock et al., 2013) – in consonance with the nominal yields found in this study for the GB-NC systems. Conversely, Orsini et al. (2014) estimated that rooftop UA could supply 77% of the fruit and vegetable demands in Bologna, IT. Future work could involve modelling the changes in a city's total environmental burdens through the application of the UA systems that are preferential to conventional supply chains, answering the question about UA's true ability to contribute to sustainable urban consumption regimes.

The antagonisms of scaled up UA also require further exploration. If GB-NC UA is to be employed as an efficient food production means and an economically competitive land use, the application of both fertilizer and pesticides will likely increase, with a potential for adverse local environmental changes (eutrophication and contamination) and human health impacts (ambient pesticide exposure in densely populated urban settings). Moreover, the ability for plants to release toxic chemicals when stressed is also a consideration when scaling up to city-wide UA (Pataki et al., 2011). Ambient pollution uptake during cultivation is both a real risk and of primary concern to the purchasing public, requiring further exploration to help UA gain traction and support responsible application (Wortman and Lovell, 2013).

4.3. Ex-situ environmental benefits and urban land use

It is often espoused that UA could have a number of environmental benefits beyond the city boundary (Goldstein et al., 2016c), two of which, reduction of agricultural land occupation and sequestration of carbon we explore here. Results show that UA in Boston can actually have larger LU IPs than conventional agriculture for the GB-NC tomatoes where low yields mixed with direct land occupation. UA lettuce also had greater LU IPs than conventional lettuce since GB-NCs suffered from low yields, heating fuels exacerbated the conditioned farms, while capital inputs (mainly steel) elevated the BI-NC results. This finding casts doubt on the generality of the claim that UA could reduce net land occupation. The direct carbon uptake potential of the case farms is likely limited, since the bulk of the atmospheric CO_2 converted to biomass is harvested for human consumption, digestion and subsequent release to the atmosphere, as opposed to long term storage in biomass or soil (Gliessman, 2015). Low-tillage UA would increase the amount of GHG sequestration in soil, but given that globally the application of such agricultural systems would only sequester 3-6% of anthropogenic GHGs (Hutchinson et al., 2007), the contribution from UA appears meager. Moreover, the fact that conventional farming occurs directly on land, it is hypothetically better able to accumulate biomass in the soil than systems limited by growing medium mass (rooftop farms) or hydroponic systems.

Nonetheless, it is worthwhile considering if Boston UA, under the best scenarios, could lead to significant reductions in farmland and carbon sequestration. Here we assume that BI-C is operating with solar power sources and that their production allows farmland in Massachusetts to return to forest, sequestering carbon. Yields for field tomatoes in Massachusetts are 1.4 kg/m^2(USDA, 2013). Subtracting the space occupied by solar panels, every square meter of BI-C frees 48.5 m^2 of farmland, with each square meter free farmland sequestering 0.95 kg CO_2 annually (30 year timeframe) (Schmidinger and Stehfest, 2012). The BI-C is 3492.8 m^2, producing 244.5 tons of tomatoes annually, resulting in 187 tonnes of CO_2 eq avoided annually; 27.8 by replacing conventional tomatoes, the remainder through off-site carbon sequestration. A similar assessment with the GB-C2 shows a net GHG reduction of 11.4 kg CO_2 eq/year through UA substitution of conventional produce and carbon sequestration. Appendix I outlines the underlying calculations.

These hypothetical outcomes must be viewed skeptically, since US agricultural land (cropland and pasture) with little room for expansion (USDA, 2011; World Bank, 2015) will probably continue operating at full capacity to accommodate a growing US population (FAO, 2016). Thus it is unlikely to see prime agricultural land in Massachusetts (or any other part of the US) returning to forest on account of UA production. A conservative appraisal, until contrary evidence can be found, is that it is improbably that UA result in these types of land use changes outside the city.

4.3.1. **Urban land use**

If uncertainties preclude making solid assessments of the effects of UA on land use beyond the city boundary, LCA does allow us to evaluate the efficacy of UA compared to other uses of the space within the city. Figure 3 compares the amount of GHGs reduced annually for different application of a square meter in Boston. Here land either generates solar power which replaces electricity from the NPCC grid or substitutes conventional produce with UA production (high yield UA forms are assessed using NPCC grid and 'clean' electricity). Generation of solar power turns out to be far superior in terms of GHG reductions compared to both lettuce and tomato production. Thus if Boston is looking to enact land use policies that result in optimal GHG reductions per unit area, then promotion of solar generating capacity appears to be the superior choice over UA. This finding can likely be extrapolated to other northern cities with fossil fuel dominated energy grids. Appendix I outlines the calculations supporting Figure 3. However, façade integrated solar power (Quesada et al., 2012) combined with BI UA, could provide double dividends of conventional produce and electricity substitutions making for a more efficient use of urban space than either technology on its own.

4.4. Profit vs. non-profit UA

One dichotomy that emerged in this LCA was the divergence between for- and non-profit UA. Non-profit UA is typically part of a larger exercise, be it community building, nutritional literacy, food-desert amelioration, parks and recreation, after-school programs, or any other number of intangibles (Sanyé-Mengual et al., 2015a). These benefits should be considered when one compares the tradeoffs of 26.5 kg CO_2 eq/kg lettuce from a GB-C operation running natural gas heaters in winter to a 0.92 kg CO_2 eq/kg lettuce from the prevailing supply-chain. Providing urbanites the experience to produce food (potentially stymieing food waste) or fostering alternative spaces in low-income neighborhoods for inner-city youth are vital activities. Moreover, though less common in Boston than cities in the emerging economies, the ability to provide income and nutrition to locals cannot be discounted.

Transparency in motives is essential. If a farmer partakes in UA for reasons aside from environmental sustainability, as all of the non-profit systems in this study do, then there is no contradiction between their motives and the environmental performance of their systems. However, if UA is done under the auspices of providing an environmentally preferable alternative to the status quo food system, as is often the case with for-profit UA, then it should operate in a way that aligns with this goal, such as tomato production for the BI-NC case. Contrarily, when these environmental goals are not met and no ancillary services are provided (e.g. lettuce from GB-C2), different objectives should be evoked; quality-control, freshness, etc.

5. Conclusions

This study has tested the urban environmental legend that UA provides environmentally superior food to conventional agriculture. We have used LCA to show that three of the common claims of UA advocates appear to be largely questionable: reducing food miles does not lead to more efficient supply-chains and reduced environmental impacts; the potential for symbiosis between farm and urban environment seem overstated at the farm scale; UA does not necessarily lead to reductions in land use and carbon sequestration. Though some of the UA systems do perform well against their conventional counterparts for certain IPs, a general recommendation for UA over conventional would be premature at this point. The UA systems that performed best are generally low-input systems with low yields, which cannot realistically compete with other land uses in the competitive markets of Boston and of most other large cities in northern climates. The conditioned UA forms with high yields, though ostensibly more financially tractable, are hampered by energy demands of year-round production in a northern climate and are environmentally deleterious given the underlying energy grid. Moreover, even when UA performs better than conventional agriculture, it turns out that an equivalent amount of space producing solar power would better combat GHGs, though tradeoffs in other IPs should then also be considered. Though this study was performed on UA in Boston, the conclusions likely apply to UA in other northern cities with cold winters and fossil fuel energy sources.

Ultimately, shifting towards a well-fed world that respects the finite carrying capacity of the planet will require a manifold agenda, including a reduction in the consumption of high-intensity foods (meat and dairy) (Goldstein et al., 2016b; Tilman and Clark, 2014) and a reduction of the 1/3 of global edibles discarded annually (FAO, 2013). Perhaps the agency of cities would be better applied towards these ends than towards UA in combating their food-borne environmental impacts. Even if UA's gross environmental benefits may be limited, there is still a place for it in cities as a constructive, social enterprise, as the non-profit cases in this study have shown.

One unanswered question is whether a city that fully utilizes UA will actually make a dent in their food related environmental impacts. An assessment at the urban level which looks at available space and production capacity given different UA forms, and then assesses the amount of conventional food displaced by the urban production system would serve as a good starting point to understand if UA is a meaningful design intervention to combat the environmental challenges of prevailing urban food supply chains, whilst also testing scaling effects on some of the purported environmental benefits to the city at large. However, in light of the reality that the meat and dairy are the dominant drivers of food related environmental impacts (Foley et al., 2011; Tilman and Clark, 2014), the ability of UA to significantly alter a city's environmental burdens, even given potential urban symbiosis, appears limited.

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Attribute Example	**BI-C** Gotham Greens: www.gothamgreens.com	**BI-NC** Brooklyn Grange: http://brooklyngrangefarm.com	**GB-C** Bright Farms; http://www.brightfarms.com	**GB-NC** Edgemere farm: http://www.edgemerefarm.org
Capital inputs	High: greenhouse, supporting building buttressing, irrigation, HVAC	Medium: supporting building buttressing, irrigation, green roof membranes and media	High: same as BI-C minus structural buttressing	Low: irrigation, growing beds
Operational inputs	Low for water. Potentially high for nutrients and space conditioning (if heated)	Medium: can capture nutrients at parapet and rainwater	Same as BI-C	High: loss of nutrients and runoff
Urban symbiosis potential	Medium: interacts with host building energy system and can capture rainwater	High: same as BI-C, but can better utilize organic waste (compost)	Low: no building interaction, less likely to accept organic waste or harvest rainwater	Medium: accepts rainwater and compost, but no links to buildings
Urban environmental benefits	Medium: reduced urban heat island (UHI) and potentially runoff	High: reduced UHI and runoff. Potential biodiversity hotspot.	Same as BI-C	Same as BI-NC
Production efficiency	High	Medium	High	Low

Table 1 – Four predominant UA types based on predicted material and energy regimes as identified by Goldstein et al. (2016c) Examples of each are included in the current study (see Table 2).

UA Case	Farm Size (m^2)	Technology	Location	Growing Season	For Profit	Crop(s) assessed
GB-NC1	560	field	Boston	April to October	no	tomato, arugula
GB-NC2	1269	field	NYC	April to October	no	tomato, lettuce
GB-C1	558	soil media in greenhouse (heated)	Boston	All year	no	tomato, salad greens
GB-C2	30	modular hydroponic unit	-	All year	yes	lettuce
BI-NC	1469	soil media on green roof	Boston	April to October	yes	tomato, lettuce
BI-C	3493	hydroponic greenhouse (heated)	Boston	All year	yes	tomato

Table 2 – General attributes of the UA cases assessed in this study. The GB-C2 is a portable unit and therefore has no fixed location, though the LCI is for east coast US operation. All farms are operational with the exception of the hypothetical BI-C. See appendices A-F for system descriptions.

	BI-C	BI-NC	GB-C1	GB-NC1	GB-NC2	Conventional	BI-NC	GB-C1	GB-C2	GB-NC1	GB-NC2	Conventional
	Tomato						**Lettuce**					
CC (kg CO_2 eq)	2.15	0.26	1.58	0.08	0.07	0.59	0.40	26.51	8.65	0.23	0.08	0.92
FE (CTU eq)	1.30	1.35	0.86	0.15	0.13	9.09	4.55	8.83	6.08	0.38	0.27	1.97
ME (kg N eq)	$1.0*10^{-3}$	$4.3*10^{-4}$	$6.3*10^{-4}$	$8.9*10^{-5}$	$8.1*10^{-5}$	$1.4*10^{-3}$	$1.4*10^{-3}$	$7.9*10^{-3}$	$3.8*10^{-3}$	$3.9*10^{-4}$	$1.4*10^{-4}$	$2.9*10^{-3}$
WRD (m^3 H_2O eq)	$9.0*10^{-3}$	$6.8*10^{-2}$	$6.6*10^{-2}$	$5.1*10^{-2}$	$7.4*10^{-2}$	$3.0*10^{-3}$	0.20	0.18	0.02	0.17	0.22	0.02
LU (kg C deficit)	2.03	1.25	2.43	5.10	3.63	3.35	4.23	30.23	8.78	30.48	18.73	6.55
RD (kg Sb eq)	$1.4*10^{-5}$	$1.1*10^{-4}$	$3.1*10^{-5}$	$1.3*10^{-5}$	$1.0*10^{-5}$	$1.3*10^{-5}$	$3.6*10^{-4}$	$2.3*10^{-4}$	$4.3*10^{-4}$	$3.6*10^{-4}$	$2.4*10^{-5}$	$2.3*10^{-5}$

Table 3 – Results per functional unit of tomato and lettuce for CC in kilogram CO_2 equivalents (kg CO_2 eq), FE in comparative toxicity units for ecotoxicity (CTU_e), ME in kilogram nitrogen equivalents (kg N eq), WRD in m^3 H_2O equivalents (m^3 H_2O eq), LU in kilogram carbon deficit (kg C deficit) and RD in kilogram antimony equivalent (kg Sb eq). Color spectrum traverses white (lowest IP) to dark grey (highest IP).

Indicator	BI-C	BI-NC	GB-C1	GB-C2	GB-NC1	GB-NC2	Conventional
Tomato							
CC (kg CO_2/m^2)	150	2.1	15.5	-	0.3	0.5	14.7
Cumulative Energy Demand (kWh/m^2)	1090	15	90	-	8	5	114
WRD (m^3/m^2)	0.63	1.11	0.64	-	0.23	0.51	0.099
Yields (kg/m^2)	70.0	16.3	9.8	-	4.4	6.9	40.6
Lettuce							
CC (kg CO_2/m^2)	-	1.9	19.2	250.6	0.2	0.11	11.0
Cumulative Energy Demand (kWh/m^2)	-	14	104	1830	7	3	94.2
WRD (m^3/m^2)	-	0.94	0.13	0.52	0.12	0.29	0.3
Yields (kg/m^2)	-	4.8	0.7	53.6	0.7	1.3	15.8

Table 4 – CC, cumulative energy demand, WRD and yields per unit growing area for the different UA systems and crops.

	NPCC grid	Photovoltaic	On-shore Wind	Hydro	Conventional	NPCC grid	Photovoltaic	On-shore Wind	Hydro	Conventional
	Tomato (BI-C)					Lettuce (GB-C2)				
CC (kg CO_2 eq.)	2.15	0.478	0.323	0.241	0.591	8.65	1.65	0.903	0.515	0.925
LU (kg C deficit)	2.03	28.9	0.735	0.379	3.35	8.78	136	3.63	1.95	6.55
RD (kg Sb eq.)	$1.4*10^{-4}$	$1.9*10^{-4}$	$4.0*10^{-6}$	$1.9*10^{-6}$	$1.3*10^{-5}$	$4.3*10^{-5}$	$8.8*10^{-4}$	$2.4*10^{-5}$	$1.4*10^{-5}$	$2.3*10^{-5}$

Table 5 – LCA results for BI-C producing 1 kg tomatoes and GB-C2 producing 1 kg lettuce using different marginal electricity sources. Northeast Power Coordinating Council (NPCC) grid is the default scenario from the earlier results.

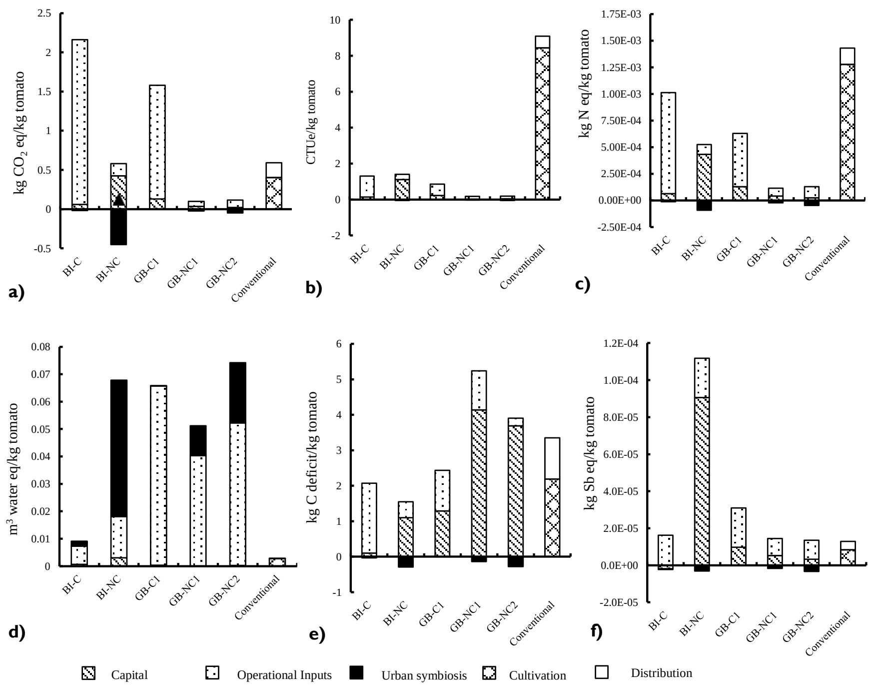


Figure 1) IPs for production of 1 kg fresh tomatoes with the studied UA systems and a conventional system for CC (a), FE (b), ME (c), WRD (d), LU (e) and RD (f). Black triangle is the aggregate BI-NC CC IP.

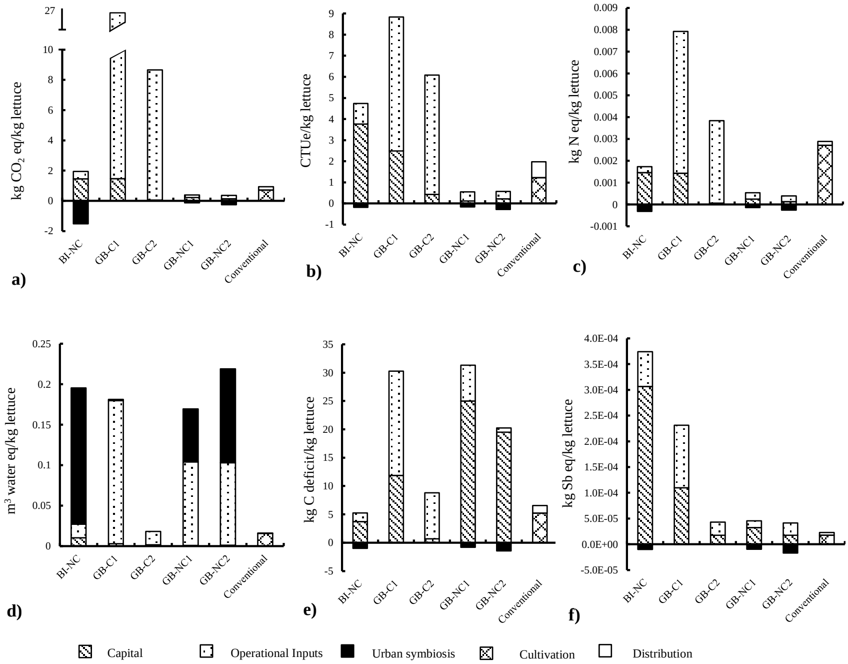


Figure 2) IPs for production of 1 kg fresh tomatoes with the studied UA systems and a conventional system for CC (a), FE (b), ME (c), WRD (d), LU (e) and RD (f).

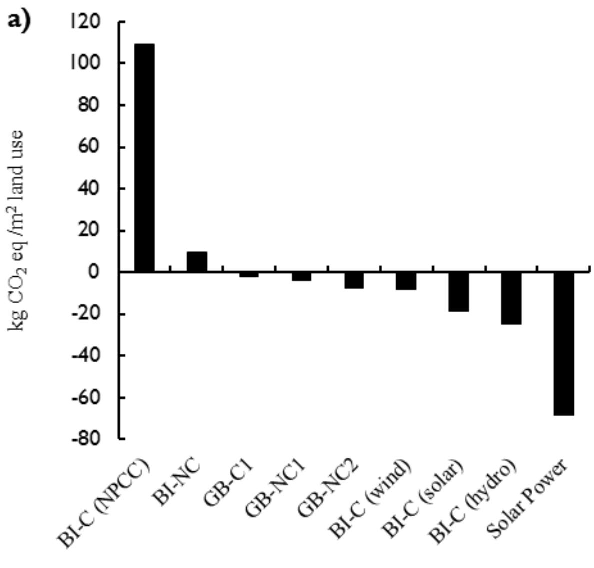
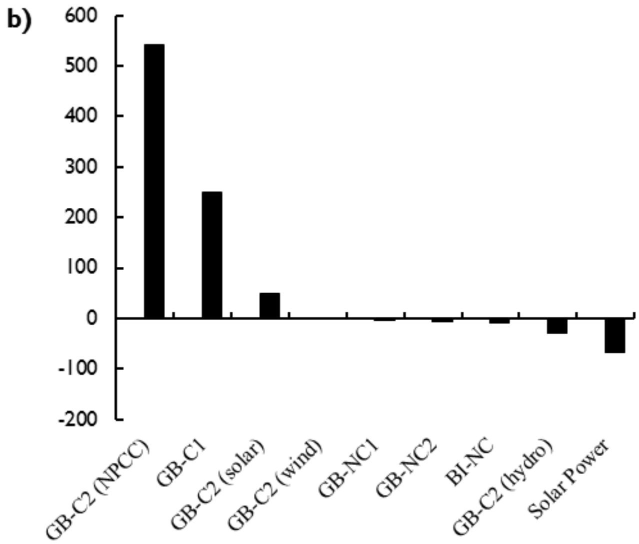


Figure 3 – CC impacts per m^2 of different farms assuming a substitution of conventional tomato (a) and lettuce (b) for UA produce. Net impact of 1 m^2 solar panel installation is also shown.



Appendix A: Life cycle inventory for building integrated conditioned farming system (BI-C)

The supplementary information is arranged as follow: description of estimation of capital inputs where primary data was lacking; description of operational inputs where primary data was lacking; component lifetimes and recycling rates; life cycle inventory for functional unit.

BI-C: Capital Inputs

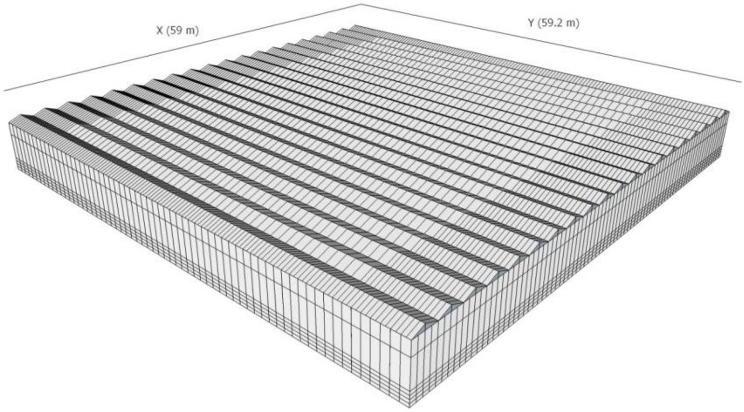


Figure 1 - BI-C system in Google SketchUp

The building integrated conditioned UA (referred to as BI-C in the article text) site is intended to approximate the operation of a BI-C farm in Boston, US with a growing space of 3493 m^2. The life cycle inventory is based off of discussions with a rooftop greenhouse designer who has experience designing BI-C farms in this climate. Through discussions with the designer a base design for the BI-C was developed including the structural, heating-ventilation-air-conditioning (HVAC) system, irrigation system, operational energy, yields and irrigation needs. Grey literature and information publically available about a BI-C farm operating were used to better refine the model. Figure 1 shows the envelope and exterior structural elements of the BI-C system. Transport of capital is assumed to be included within the inventories of the materials taken from the ecoinvent 3.1 database, which are global averages for transport from place of production to the global market.

Steel Structural Components

The system consists of a system of 11 columns along the x-axis and 17 columns along the y-axis. Along the exterior rows of columns along the y-axis, there are extra columns between the 17 columns, bringing the total along these two rows up to 32 columns. All columns are 5.8 meters high. Beams are connected to the columns running along the x-axis, making for a total of 17 girders including the exterior walls. Interior x-axes have truss systems running the length of the farm. Each y-axis row of columns also has two braces for additional structural buttressing. At the bottom of the greenhouse are 4 radiator tubes, each running the entire perimeter of the operation (these are not structural in nature, but have been included here since they are in the figure).

Figure 2 outlines the steel structural components of the BI-C system. Figure 3 shows a section of the BI-C system along with all of the different constituent components. Table 1 outlines the amounts of steel required for the BI-C's structural support.

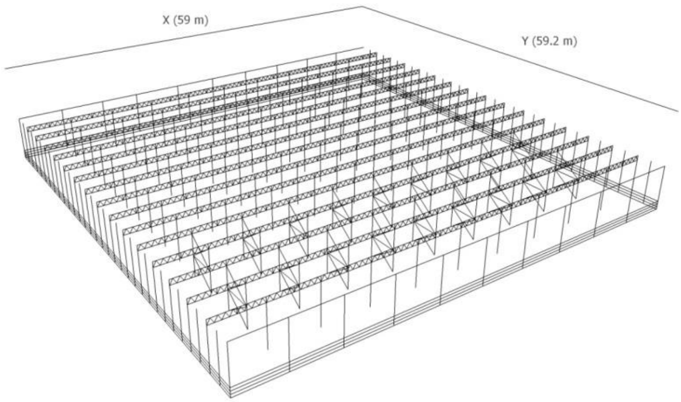


Figure 2 - Steel structural components of the BI-C system

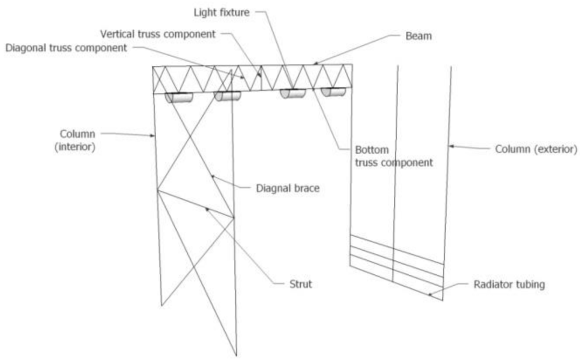


Figure 3 - Different components of the BI-C system's structural components. The perimeter radiator tubing and light fixtures are also shown.

Table 1 - Steel components

Component	**Number**	**Length(m)**	**Profile**	**Volume(m^3)**	**Mass (kg)**	**Notes**
Column	217	5.8	0.18 m x 0.18 m thickness(t) = 0.0055 m	2.45	19729	Profile provided by consultant
Beam	170	5.9	0.1 m x 0.05 m t = 0.004 m	0.59	4715	Profile provided by consultant
Bottom truss component	150	5.9	0.1 m x 0.05 m t = 0.004 m	0.52	4161	Profile provided by consultant
Vertical truss component	150	0.63	0.05 m x 0.05 m t = 0.0025 m	0.023	185	Profile provided by consultant

Diagonal truss component	2700	0.725	0.05 m x 0.05 m t = 0.0025 m	0.42	3338	Profile provided by consultant
Strut	16	3.72	0.05 m x 0.05 m t = 0.0025 m	0.015	117	Profile provided by consultant
Diagonal brace	64	4.72	0.05 m x 0.05 m t = 0.0025 m	0.074	593	Profile provided by consultant
				Total	32838	

Aluminum Structural Components

Rows of aluminum girders run along the y-axis, 2 rows per column, making for 21 rows of beams in total including the edges. 20 gables run the length of the greenhouse along the y-axis. Each gable has an aluminum ridge running the length of the building. Around the greenhouse perimeter, on the x-dimension, there four aluminum columns running the height of the greenhouse per gable, along with a post that meets the ridge peak in the center. Along the y-dimension there are four exterior columns per set of steel columns running in that direction around the perimeter of the building. Along each ridge are aluminum rafters, for every set of steel columns running in the y direction. The density of the rafters are doubled along both sides of the exterior ridges running the length of the y-axis, and between the first two sets of columns running the x-axis on both sides of the greenhouse. Lastly, there are two periphery braces wrapped around the building. Figure 4 shows the entire aluminum structure, figure 5 highlights the individual components, while table 2 shows the amounts of aluminum used in the BI-C farm.

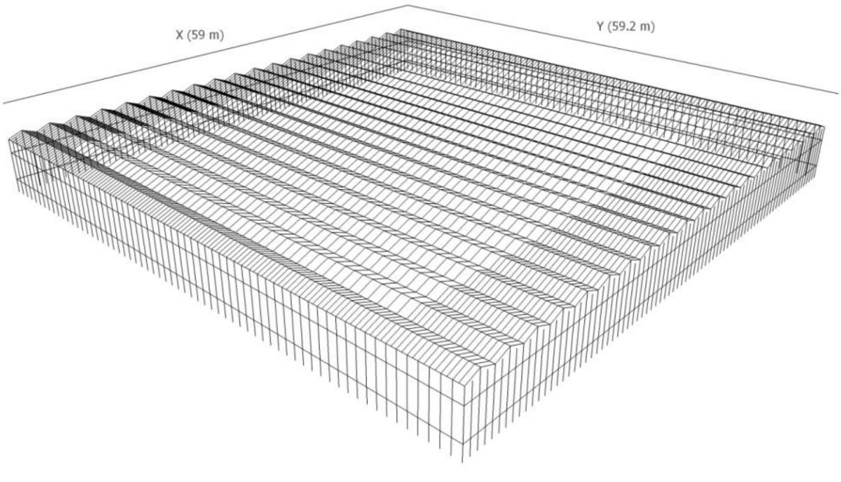


Figure 4 - Aluminum components of BI-C operation

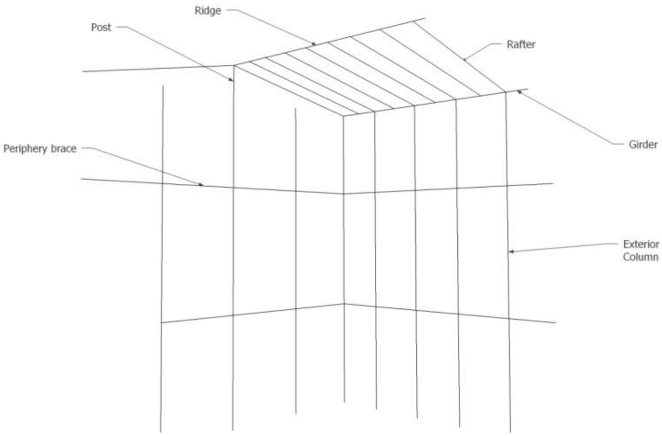


Figure 5 - Detailed shot of aluminum components of BI-C operation

Table 2 - Aluminum components

Component	Number	Length (m)	Linear Density (kg/m)	Mass (kg)	Notes
Girder	21	59.2	4	4973	Linear density estimated by consultant
Ridge	20	59.2	2	2368	Linear density estimated by consultant
Post	40	0.57	3	68	Linear density estimated by consultant
Rafter	2992	1.57	3	14092	Linear density estimated by consultant
Exterior Column	296	5.8	3	5144	Linear density estimated by consultant
Periphery Brace	2	236.4	3	1418	Linear density estimated by consultant
			Total	28064	

Glass Skin

The BI-C operation is assumed to be clad with a 4 mm thick glass membrane. The area was calculated from the Google SketchUp model. Figure 6 shows the glass skin and table 3 shows the inventory for the glass skin.

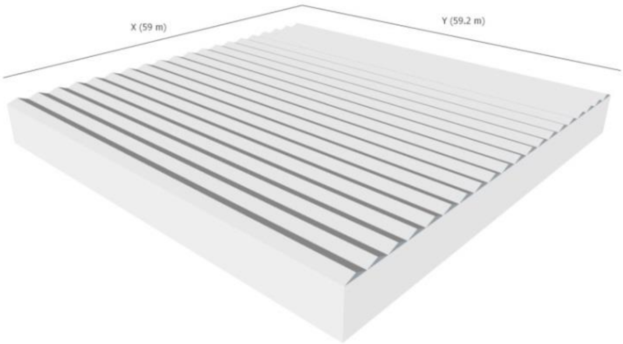


Figure 6 - Glass cladding

Table 3 - Glass components

Component	**Area (m^2)**	**Thickness (m)**	**Density (kg/m^3)**	**Mass (kg)**
Glass	4788	0.004	2600	49795

Floor

According to discussions with the consultant, roof surface is not different than a traditional flat roof with the exception of the placement of a polypropylene protective layer. Table 4 shows the amount of propylene estimated for the greenhouse.

Table 4 - Floor components

Component	**Area (m^2)**	**Density (kg/m^2)**	**Mass (kg)**	**Notes**
Polypropylene floor cover	3493	0.1	349	Density taken from a standard ground cover used in industry: http://bit.ly/1JGI6qE

Mechanical Components

The mechanical system is assumed to consist of a louver system to provide ventilation along the ridges and a system of screens that provide insulation and provide shading when necessary. Each ridge has three louvers, making 20 louvers each with a rack box. There are three motors controlling the louvers along the entire greenhouse. The shading system consists of shading curtains above the growing area, while the walls have blackout curtains. All of the curtains are retracted by means of retracting a write using a motor – both the motor and the wire have been ignored. Table 5 outlines the different components of the system.

Table 5 - Components of the mechanical system

Component	**Amount**	**Units**	**Unit Mass**	**Units**	**Mass (kg)**	**Notes**
Metal Components						
Rack box	60	units	10	kg/unit	600	Assumed to be steel. Model THG25R, 1100 mm stroke: http://bit.ly/1R5tqaM
Motor	3	units	33.4	kg/unit	100	Assumed to be Wadsworth 100A: http://bit.ly/1kpIWS3 with a mass of 75% steel, 10%

						copper, 10% aluminum, 5% PET
				Total	700	
Polyester Components						
Roof	3493	m^2	0.0435	kg/m^2	304	Taken as industry standard shading material: http://bit.ly/1MEJcDh
Wall	1369	m^2	0.06732	kg/m^2	184	Taken as 66/34 polyolefin/polyester blend: http://bit.ly/1SkxCUX
				Total	488	
Polyolefin Components						
Wall	1369	m^2	0.13068	kg/m^2	358	Taken as 66/34 polyolefin/polyester blend: http://bit.ly/1SkxCUX
				Total	358	

Heating-Ventilation-Air-Conditioning (HVAC)

The HVAC system consists of a ventilation system (discussed above), a perimeter radiator system, a radiator system along the floor, air handling units connected to convection tubing and a system of evaporative cooling pads. The perimeter radiator system can be seen in figures 2 and 3 running along the bottom of the structure. The radiator tubing along the floor runs in the y-direction in the same direction of the ridges. For every row of steel columns running along the y-direction there are three sets of radiator tubes and 4 grow gutters. Each grow gutter has a convection tube underneath it running the length of the greenhouse. The evaporative cooling pads are along the length of one of the walls running in the y-direction. It is assumed that a negative air pressure is maintained in the greenhouse obviating the need for a fan.

There is one air handling unit (AHU) per convection tube. According the consultant, at typical AHU consists of a fan driving air passing over a copper coil. The AHU was sized to maintain a temperature of 20° C when the outside temperature is -25° C, using a simple heat balance equation (heat conducted through glass is equal to the amount to be supplied by the AHU), in order to provide an order of magnitude estimate of the material equipments for this piece of equipment.

Figure 7 shows the setup of HVAC and growing equipment (to be explained below) within each set of steel columns running along the y direction. Figure 8 shows the detailed components of the HVAC and growing system. Table 6 shows the material inputs to the HVAC system.

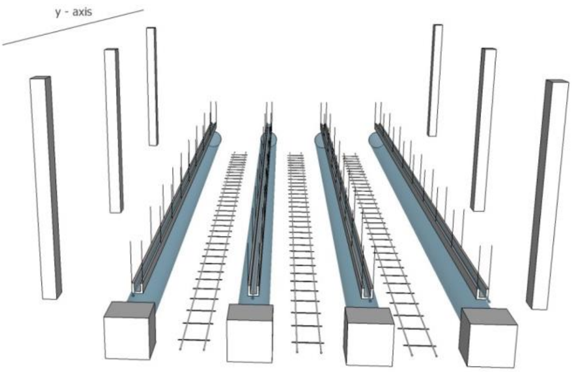


Figure 7 - HVAC and grow system looking along the y-direction of the BI-C operation. 4 grow gutters, convection tubes and AHU's between each row of columns in the y-direction. 3 sets of radiator tubes in the same direction.

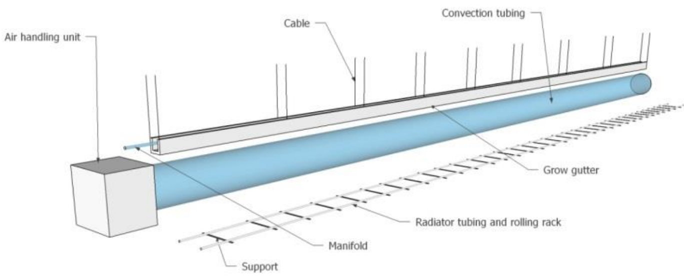


Figure 8 - Detailed diagram of the HVAC and grow system.

Table 6 - Components of the HVAC system

Component	Amount	Length (m)	Dimensions	Density (kg/m^3)	Mass (kg)	Notes
Steel Components						
Perimeter radiative tubing	4	236.4	d = 0.025 m t = 0.002 m	8050	2295	Dimensions estimated by consultant.
Floor radiator tubing	60	55	d = 0.025 m t = 0.002 m	8050	8008	Dimensions estimated by consultant. Tubes are shorter than building length to allow for mobility within the farm
Support	1800	-	-	-	900	Assumed to weigh 0.5 kg per unit
AHU	40	-	-	-	351	AHU sized to the heating demand of the BI-C weighs 10.45 kg/

						http://bit.ly/1OWWM6A. Material breakdown given by Nyman and Simonson (2004).
				Total	11555	
Copper Components						
AHU	40	-	-	-	8	See above
				Total	8	
Polyurethane Components						
AHU	40	-	-	-	0.3	See above
				Total	0.3	
Wool Components						
AHU	40	-	-	-	28	See above
				Total	28	
Fiberglass Components						
AHU	40	-	-	-	3	See above
				Total	3	
Aluminum Components						
AHU	40	-	-	-	34	See above
				Total	34	
Rubber Components						
AHU	40	-	-	-	0.4	See above
				Total	0.4	
Low Density Polyethylene Components						
Convection tubing	40	55	-	0.2 kg/m	440	Density taken from: http://bit.ly/1NOcnYG
				Total	440	
Paper Components						
Evaporative pad	1	59.2	-	18 kg/m	1048	Density taken from: http://bit.ly/1R5OwWs
				Total	1048	

Electrical and Lighting

The BI-C is light by high power sodium lamps. There are assumed to be four lamps hanging from each truss section (see figure 3), making for a total of 600 lamps in the BI-C farm. The material composition of the lamps was taken from the work of Rosado et al. (2014). It is assumed that these lamps are all connected by medium voltage cables to a central position along one of the walls running in the y-direction. Tables 7 and 8 outline the rough material inputs to electrical and lighting system. Additionally, it has been assumed that at least one computer it used as hub for controlling the greenhouse operations, but this is not included in tables 7 or 8, but it is in the life cycle inventory for the system (see table 23).

Table 7 - Components of the lighting and electrical system

Component	Amount	Unit	Total	Units	Notes
Lamp	600	lamps	2820	kg	Assumed to be 1000W model of brand: http://bit.ly/1QXUYjN
Cable	15	rows	885	m	59 m per row

Table 8 - Material inputs to lighting and electrical system

Component	Amount	Unit	Total (kg)	Notes
Copper Components				
Copper conductor (cable)	6.8	kg/m	6034	Cable components taken as http://bit.ly/1YPVbF9
Copper shield (cable)	2.1	kg/m	1833	Same source as conductor

Copper element (lamp)	0.17	kg/kg lamp	484	(Rosado et al. 2014)
		Total	8351	
Steel Components				
Steel (lamp)	0.02	kg/kg lamp	60	(Rosado et al. 2014)
		Total	60	
Aluminum Components				
Aluminum (lamp)	0.15	kg/kg lamp	420	(Rosado et al. 2014)
		Total	420	
High Density Polyethylene				
HDPE (lamp)	0.04	kg/kg lamp	106	(Rosado et al. 2014)
		Total	106	
Glass Components				
Glass (lamp)	0.6	kg/kg lamp	1692	(Rosado et al. 2014)
		Total	1692	
Expanded Polypropylene Rubber Components				
Inner screen (cable)	0.09	kg/m	79	Same source as conductor
Insulator (cable)	2.7	kg/m	2377	Same source as conductor
Outer screen (table)	0.2	kg/m	175	Same source as conductor
		Total	2631	
Polyvinylchloride Components				
Jacket (cable)	0.53	kg/m	468	Same source as conductor
		Total	468	

Irrigation System

The grow system consists of 4 grow gutters running between each set of steel columns for the length of the y-axis of the greenhouse (see figure 7), making for a total of 40 gutters. The gutters serve as a base for rockwool grow cubes, of which there are 112 per grow gutter. Along the length of each grow gutter there is assumed to be a PVC manifold which allows for drip irrigation (see figure 8). The grow gutter hang from the truss system by means of steel cables – the cables have been ignored in the model. The sizing of the pump is based on a 2.4 L/d/plant at maturity, and a planting density of approximately 2.5 plants/m^2 (Selina and Bledsoe 2002). It is also assumed that two tanks are located in the supporting building in order to store rainwater for irrigation in line with a similar farm in Montreal, CA (Lufa Farms 2014a). Piping to deliver the water from the storage tanks to the irrigation manifold is ignored. There is also a UV filter which is used to clean the captured irrigation water, but this has been ignored due to its negligible size and lack of information. Each grow gutter is fed by a soft polyvinylchloride dripline running its length, which itself is fed by a manifold running along the length of one of the x-direction walls. Table 9 outlines the material inputs to the irrigation system.

Table 9 - Material components of BI-C irrigation system

Component	Number	Amount	Units	Dimensions	Mass (kg)	Notes
Steel Components						
Grow gutter	40	54.2	m	0.3 m x 0.09 m t = 0.0008 m	4607	Taken as an average model from this range of products http://bit.ly/1RR8ciy
Pump	2	27	kg/pump-	-	54	One pump operating and one for backup: http://bit.ly/1mQMONE
				Total	4661	
Polyurethane Components						
Grow gutter	40	54.2	m	0.3 m x 0.09	34	Taken as an average model

coating				m t = $4*10^{-5}$ m		from this range of products http://bit.ly/1RR8ciy
				Total	34	
Copper Components						
Pump	2	3.6	kg/pump-	-	7.2	
				Total	7.2	
Aluminum Components						
Pump	2	3.6	kg/pump-	-	7.2	
				Total	7.2	
Polyvinylchloride Components						
Dripline	40	59.2	m	d = 0.03175 m Schedule 40	1549	-
Manifold	1	59	m	d = 0.03175 m Schedule 40	38	-
				Total	1587	
Rubber Components						
Pump	2	1.8	kg/pump-	-	3.6	
				Total	3.6	
High Density Polyethylene Components						
Tank	2	2506	kg/tank	20 000 gal	5012	Tank taken from http://bit.ly/1QZ7QX0
				Total	5012	
Rock Wool Components						
Grow Cubes	4480	0.3	kg/cube	-	1425	-
				Total	1425	

Waste Management System

The BI-C farm is equipped with a composter to deal with a portion of the organic waste generated at the site and produce compost. The composter type and size were based off of an existing BI-C operation in Montreal, QC with a similar growing area and yield (Church 2013). The composter is of the rotary drum variety and the mass is assumed to consist entirely of steel.

Because the compost is sold on the market, implying that some of the burdens for the production of the compost should be the onus of the purchaser. The burden sharing between the agent generating the refuse and the agent purchasing the daughter products of its treatment has been a matter of debate in LCA since the methods inception (Allacker et al. 2014). Here we opt for a 50:50 split between the BI-C operation producing the waste and the end user of the compost.

Table 10 - BI-C waste management system material inputs

Component	Number	Unit Mass (kg)	Mass (kg)	Notes
Composter	1	2000	2000	Taken as 16' model: http://bit.ly/1VpueYC

Structural Support

See S2.1 of the BI-NC farm to see the assumptions made in calculating the extra steel demands for the roof with the greenhouse on top. It was estimated that no extra concrete would be required but 7.6 tonnes of steel would be needed in the farm scenario.

Table 11 – Steel and concrete needs for the BI-C in the absence of a farm

Component	Load	Unit	Mass	Unit
corrugated sheet t=0.7	556.866325	kN	56.8	t
secondary beams IPN160	144.89692	kN	14.8	t
primary beams(truss) C200	457.69724	kN	46.7	t
total steel weight	1159.460485	kN	118.2	t

total concrete weight	3.7311246	kN	0.4	t

Table 12 - Steel and concrete needs for the BI-C with a farm incorporated

Component	Load	Unit	Mass	Unit
corrugated sheet t=1	556.866325	kN	56.8	t
secondary beams IPN180	144.89692	kN	14.8	t
primary beams(truss) C220	531.86952	kN	54.2	t
total steel weight	1233.632765	kN	125.8	t
total concrete weight	3.7311246	kN	0.380339	t

BI-C: Operating Characteristics and Inputs

The operating inputs for the BI-C operation are based off of secondary literature sources and discussions with a rooftop greenhouse designer, and should thus approximate the characteristics of a BI-C operation in the northeast of North America. It is assumed that the BI-C only produces tomatoes, and the operating inputs are in accordance with this.

Yields

A literature search was performed to understand the range of possible yields for tomatoes grown in advanced greenhouse operations. Table 13 outlines these findings. Based on these findings and discussions with the greenhouse designer a base production rate of *70 kg of tomatoes per square meter greenhouse space per year (70 kg/m^2/year)* is adopted for this study. With a total greenhouse space of 3492.8 m^2, a gross production volume of 244 496 kg of fresh tomatoes per year is expected for this operation.

Table 13 - Greenhouse tomato yields in literature

Yield (kg/m^2/year)	Study
56.2	(De Gelder et al. 2005)
60	(De Gelder et al. 2012)
65	(Asci et al. 2013)
94.5	(Unknown 2015)
80-100	(Ho 2004)
100	(Kubota 2009)

Irrigation

The BI-C has irrigation water supplied from both the municipal water system and rainwater that is stored in tanks in the supporting building's basement. The first step was the calculation of gross irrigation demands for the tomato crops. An initial guess can be gleaned from data on the irrigation inputs per unit output of tomato in greenhouse production schemes, shown in table 14.

Table 14 - Greenhouse tomato irrigation demands in literature

Irrigation Input (kg water/kg tomato)	Study	Notes
4, 15 or 16	(Ruijs 2011)	High efficiency with water capture and recycling
26	(Selina and Bledsoe 2002)	High efficiency with water capture and recycling
136	(Stefanelli et al. 2013)	Standard hydroponic method without water recycling

In discussion with the aforementioned greenhouse designer, the Selina and Bledsoe number of 26 kg water/kg tomato was taken, as this is within the operating range of a high-tech greenhouse, but does not assume an absolute best-case scenario. Taking this number and multiplying it by the estimated yield, the total irrigation demands are 6347 m^3 per annum, supplied from both the municipal water system and rain fed irrigation.

To determine the split between rain fed and municipal irrigation, the amount of rain available and tank capacity were considered. The amount of rain was based on that of rainfall in the Allston neighborhood of Boston.

Daily rainfall for Boston in the year 2014 (March to December) and 2015 (January and February) were taken from the Boston Water and Sewer Commission (Boston Sewer and Water Commission 2015). The amount of water that could be used for irrigation was limited by the storage capacity of the tanks (151.6 m^3). The volume of water (V_t) at any given time in the tank was calculated as,

$$V_t = V_{t-1} + V_{in,t} - V_{out,t}$$

Where, V_{t-1} is the amount of water left in the tank from the previous day, $V_{in,t}$ is the amount of water entering the tank from precipitation (taken as the greenhouse area times the precipitation amount), and $V_{out,t}$ is the amount of water leaving the tank to irrigate the plants on the same day. When daily deluges exceeded the capacity of the tank, $V_{(t)}$ was set to the tank volume, and $V_{in,t}$ subsequently modeled as zero until space was generated. If there was no water in the tank, then it was assumed that $V_{out,\,t}$ was also zero.

To determine irrigation demands the field measurements of Selina and Bledsoe (2002) were taken, whereby plants seedlings were assumed to have an irrigation demand of 0.05 L/plant/day, which rises linearly over a 42 day period to 2.4 L/plant/day, with this level sustained for a 126 day period. After the 168 day period it was assumed that a week-long break occurred, which was then followed by another round of growing. This meant that assuming a first planting on March 1st a second planting would occur on August 23rd. The planting on March first makes sense as it would allow the greenhouse to be in full bloom during the coldest months of the year, thereby maximizing the energy dissipated from plant respiration. The number of plants in the greenhouse was also calculated from Selina and Bledsoe (2002), whose 78078 m^2 greenhouse contained an estimated 200 000 plants, or roughly 2.56 plants/m^2, which works out to *8947 plants per growing period*. Checking this assumption the annual yield can be divided by the total number of plants to give approximately 14 kg, well aligned with general hothouse tomato production characteristics (Ford 2013).

Using the above methods it was found that the annual irrigation demand of the greenhouse would be 6353 m^3, *supplied as 3190 m^3 from rainwater and 3164 m^3 from municipal sources*. Discrepancies between the above noted total irrigation demand based off of the 26 kg water/kg tomato (6357 m^3 total water) are assumed to be due to rounding. It should also be noted that in the LCA, the rainwater used as irrigation was also modeled as avoided runoff to the wastewater treatment plant as the water would have went to the plant in the absence of the BI-C operation.

Rockwool cubes are assumed to be used as a growing medium. Each cube is used once and then disposed. There is one cube per plant, which amounts to 17894 cubes per year. Each cube has a mass of 0.3 kg, amounting to 5368 kg of rock wool per year.

Heating Energy

Heating energy was taken from a review of greenhouse performance which found that best-practice greenhouses in the Netherlands used 520 kWh/m^2/year for an intensive edible crop (e.g. tomato) (Carbon Trust 2011). This was scaled to the Boston, US location using climate data, assuming that the amount of heating input would be directly related to the number of heating degree days (HDD) in the city. Furthermore, it was assumed that 50% of the heating demands were met through symbiosis with the building below, in line with the claims of an existing BI-C operation in Montreal, Canada (Hage 2012). Ten year average HDD in Boston from 2005-2015 were taken from the US National Oceanic and Atmospheric Administration centers database (National Oceanic and Atmospheric Administration 2016), while the HDD for the Netherlands were averaged from 2000-2009 (Eurostat 2013). The details of the calculation are outlined in table 15. Electricity mix from 2012-2040 was used for the Northeast (see BI-NC SI2.2 for details on grid composition).

Table 15 - Details for heating of the BI-C operation

Heating Demand in High Efficiency Dutch Greenhouse ($kWh/m^2/year$)	HDD (Netherlands)	HDD (Boston, US)	Ratio (HDD_{MTL}/HDD_{NL})	Estimated heating demand for BI-C ($kWh/m^2/year$)	Estimated heating demand - 50% supplied from building ($kWh/m^2/year$)
520	2644 ± 116	3032 ± 128	1.15	596	298

It should be noted that this back-of-the envelope calculation may be an overestimate since we have not accounted for the increased the higher solar irradiance in Boston compared to the Netherlands which would be captured as heat in the greenhouse. *Taking the total area of the BI-C farm, the heating demands are calculated as 1 041 393 kWh/year.*

Lighting Energy

Lighting energy was taken in the same manner as the heating energy; an initial estimate was taken from high-tech Dutch greenhouses (Carbon Trust 2011), and then scaled to the Boston site, but this time using solar irradiance. Table 16 outlines the calculation. Solar irradiance for The Netherlands was taken as the national average over a 22 year period (Solar Electricity Handbook 2015). Solar irradiance for Boston was taken as the 1999-2008 average from the National Renewable Energy Laboratory's solar resource map (2015).

Table 16 - Lighting energy demands at BI-C

Lighting Demand in High Efficiency Dutch Greenhouse ($kWh/m^2/year$)	Solar Irradiance (Irr), Netherlands ($kWh/m^2/d$)	Solar Irradiance (Irr), Boston, US ($kWh/m^2/year$)	Ratio (Irr_{MTL}/Irr_{NL})	Estimated lighting demand for BI-C ($kWh/m^2/year$)	Total lighting energy (kWh/year)
10	2.85	4	1.40	7.3	25 547

Nutrient Inputs

The follow macro nutrients were included in the study: calcium, potassium, magnesium, nitrogen and phosphorous. There exist a number of recipes for supplying these to hydroponic tomatoes in masses of the individual elements (see table 17). In order to convert the elemental masses to masses of commercial applied fertilizers, the following algorithm was applied:

1. Calculate the amount of calcium nitrate required to supply elemental calcium;
2. Calculate the elemental nitrogen in the calcium nitrate in step 1, and augment any deficiencies with potassium nitrate to get the needed amount of elemental nitrogen;
3. Calculate the potassium phosphate that would assuage elemental phosphorous demands;
4. Calculate the elemental potassium in the potassium phosphate and potassium nitrate, and top-up with potassium sulfate as needed;
5. Calculate magnesium sulfate to support magnesium requirements.

Using this formula, the amounts of the various salts needed over a year of operation were calculated from the total irrigation rate, since the various fertilization recipes were given in concentrations for the irrigation water (g/L). Table 17 outlines various nutrient solutions found in literature and the total amount of salts required to support them given our estimated yield. Not all nutrient solutions included all of the salts.

Moreover, the solutions two and three provided the nutrient demands per kg of fresh yield, so the above noted algorithm was not used.

Table 17 - Recipes for various nutrient solutions. A dash indicates that the salt was either not prescribed for that recipe. A zero indicates that the nutrient demands were already met according to the above calculation method.

Solution	**$MgSO_4*7H_2O$ (g/L) [kg/a]**	**KH_2PO_4 (g/L) [kg/a]**	**KNO_3 (g/L) [kg/a]**	**$Ca(NO_3)_2$ (g/L) [kg/a]**	**K_2SO_4 (g/L) [kg/a]**	**$KMgSO_4$ (g/L) [kg/a]**
1 (Zekki et al. 1996)	0.287 [2099]	0.16 [1155]	0.53 [3845]	0.492 [3599]	0	-
2 (Sanyé-mengual et al. 2015)	-	[1080]	[9535]	-	-	-
3 (Hatirli et al. 2006)	-	[2247]	[3115]	-	[1092]	-
4 (Arizona State University - Center for controlled agriculture 2013)	0.46 [2390]	0.22 [1394]	0.58 [3679]	0.76 [4819]	0.22 [1427]	-
5 (IBID)	0.49 [3125]	0.17 [1087]	0.58 [3653]	0.70 [4428]	0.22 [1373]	-
6 (Mattson and Peters 2014)	0.59 [3756]	0.19 [1235]	0.55 [3507]	0.78 [4967]	0.18 [1153]	-
7 (Hochmuth 2012)	0.5 [3176]	0.27 [1715]	0.2 [1462]	0.62 [3939]	0.1 [635]	-
8 (Vegetable Research and Information Center 2011)	0.25 [1597]	0.14 [907]	0.25 [1597]	0.46 [2908]	0	-
9 (IBID)	0.50 [3143]	0.27 [1731]	0.20 [1294]	0.5 [3177]	0	-
10 (IBID)	-	-	0.18 [1126]	0.95 [6051]	0.35 [2185]	0.44 [2807]
11 (IBID)	0.51 [3261]	0.26 [1664]	0.58 [3715]	1.0 [6387]	0	-

For this assessment, solution 4 was employed, since it included all five salts and its nutrient loading tended to lie within the middle of the range of potential values according to all of the recipes.

Waste Management

The first aspect of the waste management tackled is the organic solid waste on the site. Solid organic waste is assumed to be composted. We base our estimates off of a BI-C operation in Montreal, CA which also grows tomatoes and is of a similar technological level. The Montreal, CA greenhouse reports a 900 kg per week of solid waste from a site as; 73% organic; 22% recyclables; and 5% inorganic (Church 2013). This farm is reported by the owners to produce 70 metric tons of produce annually (Lufa Farms 2014b). *Taking the ratio of our estimated yield and the yield from the operating greenhouse and multiplying by the waste generation rate we get an estimated waste generation rate of 163 463 kg/year in the same ratios as before.*

The 119 328 kg/year of organic waste is assumed to be composted on site. During the composting it is assumed that a mass reduction of 32% occurs in line with the composting of garden waste; table 9.3.5 (Christensen 2011), which when combined with the 44% water content of the compost (table 9.3.6 – IBID) results in *45 440 kg of compost (dry weight) produced annually*. The compost composition was estimated based on experiments of composting tomato plant waste as 0.7% nitrogen, 0.15% phosphorous and1.52% potassium (table 9.3.6 – IBID). From here, there equivalent amounts of avoided fertilizer production were estimated using molecular masses. In the consequential LCA model, this was modeled as avoided virgin fertilizer production, since the compost is sold to the market (Church 2013). Table 18 outlines the details of this.

Table 18 - Compost properties and avoided fertilizer from organic solid waste at BI-C site

Component	Mass ratio in finished compost (dry mass)	Elemental mass produced (kg/a)	Avoided fertilizer production (kg/a)
Nitrogen	0.007	318	318 as N
Phosphorous	0.0015	68	167 as P_2O_5
Potassium	0.0152	691	821 as K_2O

For the recycled materials, we assume that these are covered in the inventories of the other annual operations and the waste treatment of the retired capital. The amount of landfilling was taken as the balance of the 5% of total waste mentioned above (8173 kg/a) that was not already accounted in the waste from all of the other processes (6592 kg/a), giving a final total of 1581 kg/a.

Distribution

According to the operators of the Montreal farm, they use $15/day in fuel to deliver their food (Hage 2012). Assuming a gas price of $1/liter at the time of reference, we can assume 15 L fuel/day, every day of the year, equally 5460 L fuel per annum. The fuel is assumed to be diesel with a specific mass of 0.83, giving an annual diesel demand for distribution of 4532 kg. We use the Montreal numbers because both Boston and Montreal have similar population densities (5100 and 4500 people/km^2 for Boston and Montreal, respectively), so we assume that delivery distances and distribution of supermarkets are on a similar scale.

Food is assumed to be delivered in re-usable HDPE totes, much like the operating rooftop greenhouse in Montreal (Lufa Farms 2014c), and that the *volume* of delivered food is identical. According to the Montreal greenhouse, they use 2000 bins to deliver their produce, which are then returned the next week. In order to maintain the operation of the system it is assumed that as a bin is delivered, an empty one is received, which would mean that they have 4000 bins on hand, each one being used once per week, amounting to 208 000 trips per year for all of the bins. According to the farm, each bin lasts for approximately 1000 trips (Lufa Farms 2014c), which means that 208 new bins are purchased per year. The manufacturing lists the bin's mass as 2.85 kg (Monoflo 2016), resulting in an HDPE demand of 596 kg/year.

Building Symbiosis

Due to its siting on a building, the hypothetical BI-C operation is assumed to confer energy savings to its host building both in summer by increased albedo and evapotranspiration, and in the winter from insulation and evapotranspiration. Assuming that the base building is already well insulated to code, as is typically the case according to the rooftop greenhouse designer, these savings can expected to be modest. For our model a cooling load reduction of 5% and heating load reduction of 3% were assumed, based off of field studies of the energy saving benefits of green roofs in Chicago, US (La Roche and Berardi 2014). Base heating and cooling loads were taken from statistics on energy consumption in warehouse buildings in the United States in New England (U.S. Energy Information Administration 2008). We assume that the farm is situated above an industrial building due to the greenhouse size. Heat was assumed to be supplied through natural gas. Cooling was assumed to rely on electricity. Table 19 outlines the calculations and the total energy savings assuming the same footprint for the BI-C operation and supporting building.

Table 19 - Energy savings for the supporting building due to the BI-C operation

	Base Load (MJ/m^2/year)	Percentage Saved	Amount Saved (MJ/m^2/year)	Total Saved (MJ/year)
Heating	219	3%	6.6	22954
Cooling	50	5%	2.5	8793

BI-C: Component lifetimes and recycling rates

BI-C operations are comprised of durable goods. This poses two challenges in an LCA. First, it is difficult to know when system components are retired and enter the end-of-life stage. Second, because some of these components enter the end of life phase decades in the future, it is impossible to know what recycling technologies and legislation will be in place, and consequently, what amounts of the various components will be recycled, landfilled or incinerated.

The lifetimes of different components, were informed by Sanye-Mengual et al.'s earlier LCA of a BI-C system (Sanyé-mengual et al. 2015), LCAs of regular buildings (Scheuer et al. 2003), LCAs of green roofs (Peri et al. 2012), manufacturer warranties for specific products within a category, and where no other options were available, the opinion of the greenhouse designer.

For the recycling rates, the general heuristic employed was that as components become smaller and more intermingled in terms of different materials contained, the recycling costs begin to outweigh the value of the recoverable materials, and the recycling rate drops, as has been found in practice (Gutowski and Dahmus 2005). For instance, the recycling rate of an electronic screwdriver in the US in the 00's was almost zero, while that of a automobile was high (Gutowski and Dahmus 2005). Both automobile and electric screwdriver are complicated assemblages of different materials of low value (plastics) and high value (metals), but due to the car's girth, easily accessible high value components, and potentially legislation, the car sees a much higher recycling rate. Applying the same thinking here, smaller intermingled components (pumps, motors) see lower recycling rates, while the larger less co-mingled components where recycling has an economic incentive (steel beams, plastic bins) see higher reuse rates.

Current recycling rates were taken from existing literature on construction and demolition (C&D) recycling rates in Northeastern North America (Nisbet et al. 2004) as 95% for steel structural components and 15% for aluminum and copper components. The overall recycling rate in the State of Massachusetts was 42% in 2009 (Massachusetts Department of Environmental Protection 2013). For the smaller, co-mingled goods (pumps), recycling rate of 50% after 30 years was assumed with linear interpolation between current recycling rate of all C&D waste in Massachusetts (33%) (Government of Massachusetts 2010). Recycling rates for large, metallic structural components were assumed to increase to 97.5% at the demolition time. Because of Massachusetts C&D waste landfilling ban (Government of Massachusetts 2015), it is assumed that large plastic components will also see a high recycling rate in 30 years (95%) since these components are easily source-segregated and we assume that there will exist incentives to promote their collection at the time of demolition (fines or fees for inclusion with landfilled waste), but we linearly interpolate between current recycling rate of all C&D waste in Massachusetts (33%). All materials not recycled are assumed to be landfilledFor plastics it was assumed that popular thermoplasts (high- and low-density polyethylene, polypropylene, polyethylene terephthalate) are mechanically recycled to virgin plastics, others (rubber, polystyrene, polyvinylchloride, nylon and Acrylonitrile-butadiene-styrene) are mechanically downcycled to filler (assumed to be gravel) (Hopewell et al. 2009; Rebeiz and Craft 1995).

Table 20 outlines the lifetimes and recycling rates for the various components of the hypothetical BI-C operation.

Table 20 - Recycling rates and lifetimes of various components of the BI-C farm.

Component	**Lifetime (years)**	**Current recycling rate (%)**	**Future rate (%)**						**Applied Rate (%)**	**Replaces**
			5	10	15	20	25	30		
Structural Components										
Steel	30	95	-	-	-	-	-	-	97.5	virgin steel
Glass	30	95	-	-	-	-	-	-	97.5	gravel
Aluminum	30	95	-	-	-	-	-	-	97.5	virgin aluminum
Floor	30	0	-	-	-	-	-	-	0	single use assumed
Mechanical Components										
Gearbox	20	95	-	-	-	95	-	95	95	virgin steel
Motor	20	43	-	-	-	50	-	50	50	virgin metals
Curtains[1]	5	43	100	100	100	100	100	100	100	fill
HVAC Components										
Convection	5	43	80	83	85	88	90	95	87	virgin

[1] Assumed recycling rate of 100% since they are very easily source separated

tubing (LDPE)										LDPE
Steel radiator tubing	30	95	-	-	-	-	-	-	97.5	virgin steel
Cooling pads	5	53	80	83	85	88	90	95	87	pulp
AHU[2]	20	43	-	-	-	47	-	50	48	virgin steel
Electrical and Lighting										
Lamps	10	43	43	44	45	47	48	50	48	virgin metals and plastics
Cables	30	43	-	-	-	-	-	-	50	virgin metals and plastics
Computer	10	-	-	100	-	100	-	100	100	Goes to recycling center
Irrigation										
Tubing (PVC)	30	43	80	-	-	-	-	95	95	fill
Grow gutters	25	43	80	-	-	-	90	95	92.5	virgin steel
Tanks (HDPE)	30	43	80	-	-	-	-	95	97.5	virgin HDPE
Pump[2]	15	43	-	-	45			50	47.5	
Waste Management System										
Composter	30	95	-	-	-	-	-	95	95	virgin steel
Structural Buttressing										
Steel	30	95	-	-	-	-	-	-	97.5	virgin steel
Supplies										
Rockwool[3]	0.5	0	0	0	0	0	0	0	0	single use assumed
Plastic totes[4]	-	95	95	95	95	95	95	95	95	virgin HDPE

BI-C: Life Cycle Inventory (LCI)

Capital inputs (tables 1 to 13) were divided by their lifetimes (table 18) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 20). Annual capital inputs, outputs (recycled and landfilled), and annual operating inputs were then normalized per kilogram fresh tomato produced by dividing by the annual production volume calculated above. Table 21-33 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1 processes used to model them. Produce is not refrigerated at any point in the supply chain. For more details on the custom processes for the US please see SI2.2 for the BI-NC system.

[2] Only steel, copper and aluminum components are recycled, the rest is landfilled.
[3] Assumed recycling rate of zero since rockwool is not a traditionally considered a recyclable material and it is not likely to be high on the City of Montreal's recycling agenda. Therefore, we assume that it is not likely to be recycled. Moreover, it will be entrained with biomass by the end of the growing period which would complicate the recycling process.
[4] Accounts for the 208 plastic totes that are discarded each year (see 'Distribution' section above)

Structural Components

Table 21 - LCI for BI-C structural components

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Steel	0.004477	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Aluminum	0.0038261	kg/FU	Aluminium, primary, ingot {US-NPCC}| production | Conseq, U	
Polypropylene	4.762E-05	kg/FU	Flat glass, coated {US-NPCC}| market for | Conseq, U	
Glass	0.0067888	kg/FU	Flat glass, coated {CA-QC}| market for | Conseq, U	
Material Processing				
Plastic floor shaping	4.762E-05	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
Steel recycling	0.004365	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Aluminum recycling	0.0037304	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Landfilling	0.0004249	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Avoided Products				
Gravel	0.0066191	kg/FU	Gravel, crushed {US-NE}| production | Conseq, U	

Mechanical System

Table 22 - LCI for BI-C mechanical system

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Steel	0.0001381	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	2.049E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
Aluminum	2.049E-06	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	Includes manufacturing energy for generic aluminum part
Polyethylene terephthalate	1.025E-06	kg/FU	Polyethylene terephthalate, granulate, amorphous {GLO}| market for | Conseq, U	
Polyester	0.0003993	kg/FU	Polystyrene, general purpose {GLO}| market for | Conseq, U	
Polyolefin	0.0002926	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.000693	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Copper forming	2.049E-06	kg/FU	Wire drawing, copper {US-NPCC}| processing |	

			Conseq, U	
Waste Treatment				
Steel recycling	0.0001243	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	1.025E-06	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Aluminum recycling	1.025E-06	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
PET recycling	1.025E-06	kg/FU	PET (waste treatment) {US-NPCC}| recycling of PET | Conseq, U	
Curtain recycling	0.0006585	kg/FU	PS (waste treatment) {US-NPCC}| recycling of PS | Conseq, U	
Landfilling	5.11256E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

HVAC System

Table 23 - LCI for BI-C HVAC system

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Steel (pipes)	0.0015256	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Steel (AHU)	7.158E-05	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	1.706E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
LDPE	0.0004295	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Linerboard	0.0042945	kg/FU	Linerboard {CA-QC}| linerboard production, kraftliner | Conseq, U	
Polyurethane	6.258E-08	kg/FU	Polyurethane, flexible foam {GLO}| market for | Conseq, U	
Glass wool	5.685E-06	kg/FU	Glass wool mat {GLO}| market for | Conseq, U	
Glass fiber	6.83E-07	kg/FU	Glass fibre {GLO}| market for | Conseq, U	
Aluminum	6.953E-06	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Rubber	9.08E-08	kg/FU	Seal, natural rubber based {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0004295	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Copper forming	1.706E-06	kg/FU	Sheet rolling, copper {GLO}| market for | Conseq, U	
Steel Pipe Forming	0.0015256	kg/FU	Drawing of pipe, steel {US-NPCC}| processing | Conseq, U	
Waste Treatment				
Steel recycling	0.0015234	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	

Copper recycling	8.221E-07	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Aluminum recycling	3.346E-06	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
LDPE recycling	0.0003722	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U, U	
Paper recycling	0.0044582	kg/FU	Paper (waste treatment) {GLO}| recycling of paper | Conseq, U	
PET recycling	3.018E-08	kg/FU	PET (waste treatment) {US-NPCC}| recycling of PET | Conseq, U	
Landfilling	0.0008788	kg/FU	Inert waste, for final disposal {GLO}| market for | Conseq, U	

Irrigation System

Table 24 - LCI for BI-C irrigation system

Input/Process	**Amount**	**Unit**	**ecoinvent 3 process**	**Notes**
Material Inputs				
Steel	0.0007685	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	1.958E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
Aluminum	1.958E-06	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Rubber	9.791E-07	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
HDPE	0.0006834	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
LDPE	0.0116604	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
PVC	0.0002164	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Polyurethane	5.618E-06	kg/FU	Polyurethane, rigid foam {GLO}| market for | Conseq, U	
Material Processing				
Copper Wiring	1.958E-06	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	
Plastic tube forming	0.0002164	kg/FU	Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	
Plastic film forming	0.0123494	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
Steel recycling	0.0007043	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	9.334E-07	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Aluminum recycling	9.334E-07	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
PVC Recycling	0.0001876	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
LDPE and HDPE recycling	0.0107549	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0016849	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Electrical System

Table 25 - LCI of BI-C electrical system

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Copper	0.0012707	kg/FU	Copper {GLO}| market for | Conseq, U	
Aluminum	0.0001719	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Steel	2.468E-05	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Polystyrene	0.0003587	kg/FU	Polystyrene, expandable {GLO}| market for | Conseq, U	
HDPE	4.337E-05	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
PVC	6.386E-05	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Glass	0.0006922	kg/FU	Flat glass, coated {US-NPCC}| market for | Conseq, U	
Computer	0.1	p/FU	Computer, laptop {GLO}| market for | Conseq, U	
Material Processing				
Copper Wiring	0.0012707	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	
Plastic tube forming	6.386E-05	kg/FU	Extrusion, plastic pipes {US-NPCC}| production | Conseq, U	
Plastic film forming	4.337E-05	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
Steel recycling	1.177E-05	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	0.0006307	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Aluminum recycling	8.192E-05	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
PVC Recycling	3.193E-05	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
HDPE recycling	2.067E-05	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0015184	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Computer disposal	0.315	kg/FU	Used laptop computer {GLO}| market for | Conseq, U	
Avoided Products				
Gravel	0.0003299	kg/FU	Gravel, crushed {US-NE}| production | Conseq, U	Glass recycled to gravel

Composter

Table 26 - LCI of BI-C on-site composter

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Steel	0.0006354	kg/FU	Steel, low-alloyed, hot rolled {CA-QC}| market for | Conseq, U	
Waste Treatment				

Steel recycling	0.0001554	kg/FU	Steel and iron (waste treatment) {CA-QC}| recycling of steel and iron | Conseq, U	
Landfilling	8.18E-06	kg/FU	Inert waste, for final disposal {GLO}| market for | Conseq, U	

Structural Buttressing

Table 27 - LCI of BI-C structural buttressing

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Steel	0.0017874	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Waste Treatment				
Steel recycling	0.001742667	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	4.46838E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation

Table 28 - LCI for BI-C annual irrigation

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
Irrigation water	0.012941	m^3/FU	Tap water {US-Boston}| market for | Conseq, U	
Rock wool	0.0219554	kg/FU	Rock wool {GLO}| market for | Conseq, U	
Waste Treatment				
Landfilling	4.46838E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Avoided Products				
Wastewater treatment	0.01305	m^3/FU	Wastewater, unpolluted, from residence {GLO}| market for | Conseq, U	Assumed to have same pollutant loading as household wastewater.

Space Conditioning

Table 29 - LCI for BI-C space conditioning

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Energy and Fuels				
Lighting	0.104407	kWh/FU	Electricity, medium voltage, 2012-2032 average {NPCC, US only}| market for | Conseq, U	
Heating	4.25	kWh/FU	Electricity, medium voltage, 2012-2032 average {NPCC, US only}| market for | Conseq, U	

Nutrient Demands

Table 30 - LCI for BI-C nutrient demands

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Material Inputs				
$MgSO_4*7H_2O$	0.005847	kg/FU	Magnesium sulfate {GLO}| market for | Conseq, U	Adjusted for water content to convert to anhydrous mass
KH_2PO_4	0.0029759	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	Converted to P_2O_5 mass

KNO_3	0.0150491	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
$Ca(NO_3)_2$	0.0197109	kg/FU	Calcium nitrate {GLO}| market for | Conseq, U	
K_2SO_4	0.0031538	kg/FU	Potassium sulfate, as K2O {GLO}| market for | Conseq, U	Converted to K_2O mass

Distribution

Table 31 - LCI for BI-C distribution

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Energy and Fuels				
Fuel	0.0185353	kg/FU	Diesel, low-sulfur {Europe without Switzerland}| market for | Conseq, U	
Material Inputs				
HDPE	0.0024362	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0024362	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
HDPE recycling	0.0023144	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0001218	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Building Symbiosis

Table 32 - LCI for BI-C building symbiosis

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Avoided Products				
Cooling	0.0359638	MJ/FU	Electricity, medium voltage, 2012-2032 average {NPCC, US only}| market for | Conseq, U	
Heating	0.0938816	MJ/FU	Heat, district or industrial, natural gas {CA-QC}| market for | Conseq, U	

Waste Treatment

Table 33 - LCI for BI-C waste treatment

Input/Process	Amount	Unit	ecoinvent 3 process	Notes
Waste Treatment				
Landfilling	0.0064671	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Avoided Products				
Nitrogen fertilizer	0.0013	kg/FU	Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	
Phosphorous fertilizer	0.00068	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium fertilizer	0.00336	kg/FU	Potassium fertiliser, as K2O {GLO}| market for | Conseq, U	
Landfilling	0.48672	kg/FU	Inert waste, for final disposal {US}| market for | Conseq,	Organic waste that would have gone to landfill in

			U	absence of composting system

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Appendix B: Life cycle inventory for building integrated non-conditioned farming system (BI-NC)

The building integrated non-conditioned UA site (referred to as 'BI-NC' in the article text) is an operating rooftop urban farm in Metropolitan Boston, US. The farm has 423 m^2 of growing area, 1469 m^2 of total green roof space and grows approximately 80 crop varieties. The supplementary information is arranged as follow: description of estimation of capital inputs where primary data was lacking; description of operational inputs where primary data was lacking; component lifetimes and recycling rates; life cycle inventory for functional unit. Capital equipment is spread evenly across the site, and according to interviews with the site operator supplies and operating inputs are also spread evenly across the site. Therefore, inputs and outputs are allocated to different crops based on their growing space with the exception of irrigation which was crop specific.

BI-NC: Capital Inputs

Information on the capital inputs for the farm are based off of detailed construction plans from the farm operator and procurement lists from the contracting company that built the farm. The only aspect of the capital equipment that is estimated is the structural buttressing to support the farm weight. Inputs for structural buttressing are outlined below, otherwise, the details for the capital equipment can be found in the life cycle inventory. For all components manufactured from recycled materials, no burdens were assumed for the raw material manufacturing, since these are allocated to the previous life-cycle.

Structural Support

The extra steel necessary to carry the farm was calculated using the following assumptions:

1) The roof area supporting the green roof is considered as separated from the rest of the roof in the sense that it is physically disconnected with a moment-not-transferring connection to the rest of the roof. This is a realistic assumption since pictures of the roof show that the portion of the rooftop that does not hold the farm has different construction.
2) Since it is not clear from the pictures neither from the drawings which cross section is used, we assume that the roof consists in a grid of primary and secondary beams where the primary beam is a truss with 2L section and the secondary beams used to connect the steel corrugated sheets to the primary system have C section.
3) Some machinery (for the heat and air condition system) is present on the roof. We do not have information on how much they weigh. In standard design this is a very important issue because they are usually very heavy and represent a concentrated permanent-load. However, with respect to the aim of evaluating the difference in the amount of steel between a roof with and without garden this is not important because you would have this machinery in both cases.
4) We assume that the primary beams (truss) are continuous and supported on concrete beams.
5) Size of the area of the roof x=129'≈39.4 m and y=116'≈35.4 m. The span is equally divided in 4 spans in x-direction and 3 spans in y-direction (see video), i.e. l1=13.13m and l2=11.80m.
6) In the drawings the weight of the green roof as average value is listed as: 71.83LBS/PSF for the saturated cultivated area that it is equal to 3.44kN/m^2 to which corresponds a characteristic value (10% coefficient of variation) of 4kN/m^2. The weight of ballast and not cultivated area is lower than the cultivated one, so we consider a uniform weight on the roof equal to the weight of cultivated area since this stays on the safe side resulting in a slight overestimation.
7) The snow load for flat roofs in Boston area is 1.5kN/m^2.
8) The load combination of dead and permanent and the capacity check at Ultimate Limit State and Service Limit State are done according to Eurocode 3 and 4.
9) For the flat roof load, a combination of the different spans are necessary, however since the difference with and without the snow load and the live load (due to people walking) is very small, we consider just the roof as fully loaded in all spans.
10) We do not take into account the steel from the bolts, since we do not compute the joints

Figure 1 and 2 show the profiles of the roof without and with the farm on top of it, respectively. Tables 1 and 2 show the amount of steel and concrete required for the two roof scenarios. The estimated extra steel and concrete due to the presence of the BI-NC farm are 36.9 tonnes and 0.1 tonnes, respectively.

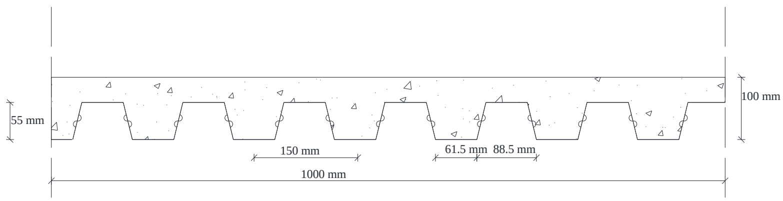

Figure 9 - Profile of building roof without farm

Table 34 - Steel and concrete for the roof without the farm

Component	Load	Unit	Mass	Unit
corrugated sheet t=0.7	72.96582054	kN	7.4	t
secondary beams IPN160	96.761672	kN	9.9	t
primary beams(truss) C160	209.66904	kN	21.4	t
total steel weight	379.3965325	kN	38.2	t
total concrete weight	1.335125736	kN	0.1	t

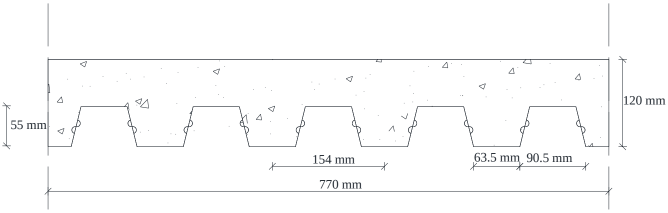

Figure 10 - Profile of roof with farm on top

Table 35 - Steel and concrete for the roof without the farm

Component	Load	Unit	Mass	Unit
corrugated sheet t=1	249.9751357	kN	25.5	t
secondary beams IPN180	118.384392	kN	12.1	t
primary beams(truss) C240	368.30752	kN	37.5	t
total steel weight	736.6670477	kN	75.1	t
total concrete weight	2.15585623	kN	0.2	t

BI-NC: Operating Characteristics and Inputs

The operating inputs for the BI-NC farm were taken from primary data provided by the farm operator, though some aspects had to be estimated, namely, space conditioning to grow seedlings, irrigation demands and runoff retention, electricity consumption, nutrient loading in runoff, waste management, imported compost production and energy savings to the building.

Seedling Production

Production of seedlings (small plants grown in greenhouses that are transferred to outdoor soil) is based off of the methods of Stoessel at al. (2012) for lettuce and tomatoes. The basis is the inputs to raise the amount of seedlings required to cultivate 1 kg fresh produce. Inputs are shown below in table 3.

Table 36 – Inputs for seedling production

Input	Tomato Seedling	Lettuce seedling	Units
Peat	$2.1*10^{-3}$	$2.5*10^{-2}$	kg
Transport	$1.5*10^{-2}$	$1.8*10^{-1}$	tkm
Heating	$6.0*10^{-3}$	$7.1*10^{-2}$	MJ

Irrigation and Runoff Retention

Lacking primary data on evapotranspiration rates from various plants at the site, general irrigation rates were used for tomatoes and lettuce were taken from FAO guidelines for estimating crop water needs (Brouwer and Heibloem 1986). Irrigation needs were calculated as the gross irrigation demands minus retained rainwater over the growing period. A retention rate of 50% was assumed based off a review of rainwater retention of intensive green roofs (Czemiel Berndtsson 2010) though this may be on the high side for an expanded clay substrate (Stovin et al. 2015). Rainfall for the Boston, US area for the 2014 growing season (assumed March through September) was taken from the Boston Sewer and Water Commission (2015). Details outlined below in table 4. The municipality is not operating combined sewers in the vicinity of the site, so the avoided runoff is not accounted as avoided wastewater treatment.

Table 37 - Irrigation demands and avoided runoff for BI-NC operation

Crop	Total Irrigation Requirements (m)	Rainfall (m)	Planted Area[5] (m^2)	Green Roof Area (m^2)	Rainfall Irrigation (m^3)	Runoff avoided (m^3)	Irrigation supplied (m^3)
Tomato	0.9	0.53	29	100	7.6	57.8	21.1
Lettuce	0.5	0.53	17	57	4.4	33.3	3.9

Electricity

The only electrical input to the BI-NC's operation is for the small pump and associated digital irrigation controller. According to the manufacturer's specifications (withheld for confidentiality reasons) the pump runs at around 7W when pumping at 37.9 L/min. Pumping energy for the entire site was taken as the estimated total irrigation needs of the site and dividing by the aforementioned flowrate and then multiply this pumping time by the wattage. The digital controller has 0.06 amps at 120 volts (AC) for a 37 week growing period, which is then converted to electricity requirements with the assumption that it runs continuously over this period. Details are in table 5 below.

Table 38 - Energy demands for BI-NC operation

User	Amount	Unit	Time (hours)	Energy (MJ)
Pump	6.72	W	16.8	0.4
Controller	0.06	A @ 120 V(AC)	6216	161.1
			Total	161.5

Because of the long-lifetime of the BI-NC operation it can be expected that the electricity mix will evolve over the duration of the project. To account for this, all electricity inputs have been modeled based off the average mix for the period 2012-2040, based off of the projections of electricity mix for the Northeast Power Coordinating Council (NPCC) during that period (U.S. Energy Information Administration 2015). For recycling, the 2040 grid was used since most components will retire at this time. Table 6 shows the average grid mix over the operating period for the NPCC region and the 2040 grid, which are essentially identical due to the unwavering reliance on natural gas as a fuel throughout the 2012-2040 period. This mix is also applied in the assessments of the other farms with the exception of the BI-C farm.

Table 39 – Estimated average grid mix over 2012-2040. Values do not add to 100% due to rounding.

Fuel	% of Grid Mix (2012-2040)	% of Grid Mix (2040)
Coal	2	2
Petroleum	< 1	< 1
Natural Gas	53	52
Nuclear – Boiling Water	10	10
Nuclear – Pressure Water	16	16
Pumped Storage	1	1
Hydro	7	7
Geothermal	1	1
MSW Incineration	1	1

[5] This is the amount of planted area which is less than the actual amount of green roof space (planted space plus soil covered areas between planted spaces) with the latter being allocated to the various crops based on the formers share occupied by a given crop. Runoff mitigated is calculated based on the green roof area and accounts for all rain that fell throughout 2014 (1.17 m). Rainwater irrigation is calculated based on planted area.

Biomass Incineration	1	1
Biomass Co-firing	< 1	< 1
Solar Thermal	1	1
Photovoltaic	< 1	< 1
Wind	1	1
Total	~97	~97

Nutrients in Runoff

Nutrients released to the ambient environment were calculated based on the amount of rainwater not captured during the growing seasons times the estimated nutrient loading in the runoff. Nutrient loading in runoff was taken from Emilsson et al.'s (2007) study of green roofs which related 'low, medium and high' levels of fertilization (N, P and K, respectively) with nutrients in green roof runoff. To determine the level of fertilization, the total nutrient input from fertilization was provided by the farm operator was divided by the total green roof area (not the planted area, since the runoff containing the nutrients comes from the entire site), giving a low level of fertilization according to the study. Nutrients are also brought on to the site through the imported fertilizer, but it is assumed that these are stably entrained within the soil matrix and do not contribute significantly to runoff. Details of the nutrient loading are in table 7.

Table 40 - Nutrient loading in runoff from BI-NC operation

Nutrient	Fertilization (g/m^2)	Runoff (mg/L)	Nutrient Loading (kg/a)
Total-N	1.72	0.5	0.73
Total-P	0.94	0.009	0.013
K	0.99	0.09	0.13

Waste Management

According to the operator around 682 kg of organic waste are generated at the site on a per capita basis for composting. During the composting it is assumed that a mass reduction of 32% occurs, and that 44% of the final is water assuming that it has the same composting properties as garden waste (see table 9.3.5 in Christensen 2011), resulting in 260 *kg of compost, dry mass, produced annually*. The compost composition was estimated based on experiments of composting tomato plant waste as 0.7 % nitrogen , 0.15% phosphorous and 1.52% potassium (table 9.3.6 IBID). From here, there equivalent amounts of avoided fertilizer production were estimated using molecular masses. In the consequential LCA model, this was modeled as avoided virgin fertilizer production. Table 8 outlines the details of this. Table 9 outlines the energy and material inputs to treat a single kilogram of organic waste and convert it to compost according to table 9.3.5 in Christensen (2011).

Table 41 - Compost properties and avoided fertilizer from organic solid waste at BI-NC site

Component	Mass ratio in finished compost	Elemental mass produced (kg/a)	Avoided fertilizer production (kg/a)
Nitrogen	0.007	1.8	1.8 as N
Phosphorous	0.0015	0.4	0.9 as P_2O_5
Potassium	0.0152	3.9	4.8 as K_2O

Table 42 – LCI for the treatment of one kilogram of garden waste

	Amount	Unit
Inputs		
Diesel	0.00113	kg
Composting facility	$4*10^{-9}$	units
Electricity	0.149	kWh
Outputs		
Carbon dioxide, fossil, to air	0.0038	kg

Landfill	0.141	kg
Avoided Products		
Avoided landfill through diversion to composting	1	kg
Nitrogen fertilizer, as N	0.00256	kg
Phosphate fertilizer, as P_2O_5	0.00127	kg
Potassium sulfate, as K_2O	0.00764	kg

Imported Compost Production

It is assumed that no burdens for the compost production are allocated to the BI-NC operation, though the freight to transport the compost from the waste management center to the farm are allocated to the BI-NC system.

Building Symbiosis

Due to its siting on a building, the hypothetical BI-NC operation confers energy savings to its host building, which is an operating supermarket. According to contractor that built the farm, the roof is well insulated below the green roof components, meaning that the energy savings for the host building from the farm will not be significant. To calculate the energy savings from the BI-NC's presence the energy consumption for an operating supermarket from the same chain was taken and the energy savings estimated as a percentage of this using the same savings assumed for the hypothetical BI-C farm (3% for cooling, 5% for heating). Because the BI-NC farm only covers a portion of the host buildings (1392 m^2), energy savings are only applied to the area of the host building covered by the farm. Table 10 outlines the predicted energy savings.

Table 43 - Energy savings for the supporting building due to the BI-NC operation

	Base Load ($MJ/m^2/year$)	**Percentage Saved**	**Amount Saved** ($MJ/m^2/year$)	**Total Saved** (MJ/year)
Heating	556	3%	2.0	62034
Cooling	68	5%	16.7	4748

BI-NC: Component lifetimes and recycling rates

BI-NC operations are comprised of durable goods. This poses two challenges in an LCA. First, it is difficult to know when system components are retired and enter the end-of-life stage. Second, because some of these components enter the end of life phase decades in the future, it is impossible to know what recycling technologies and legislation will be in place, and consequently, what amounts of the various components will be recycled, landfilled or incinerated. The contractor estimates that such a farm will operate for at least 20 years, but maybe as long as 50 years. A 30 year system lifetime has been adopted in this study.

The lifetimes of different components, were informed by Sanye-Mengual et al.'s earlier LCA of a BI-C system (Sanyé-mengual et al. 2015), LCAs of regular buildings (Scheuer et al. 2003), LCAs of green roofs (Peri et al. 2012), manufacturer warranties for specific products within a category, and where no other options were available, the opinion of the greenhouse designer.

For the recycling rates, the general heuristic employed was that as components become smaller and more intermingled in terms of different materials contained, the recycling costs begin to outweigh the value of the recoverable materials, and the recycling rate drops, as has been found in practice (Gutowski and Dahmus 2005). For instance, the recycling rate of an electronic screwdriver in the US in the 00's was almost zero, while that of a automobile was high (Gutowski and Dahmus 2005). Both automobile and electric screwdriver are complicated assemblages of different materials of low value (plastics) and high value (metals), but due to the car's girth, easily accessible high value components, and potentially legislation, the car sees a much higher recycling rate. Applying the same thinking here, smaller intermingled components (pumps, motors) see lower recycling rates, while the larger less co-mingled components where recycling has an economic incentive (steel beams, plastic bins) see higher reuse rates.

Current recycling rates were taken from existing literature on construction and demolition (C&D) recycling rates in Northeastern North America (Nisbet et al. 2004) as 95% for steel structural components and 15% for aluminum and copper components. The overall recycling rate in the State of Massachusetts was 42% in 2009 (Massachusetts Department of Environmental Protection 2013). For the smaller, co-mingled goods (pumps), recycling rate of 50% after 30 years was assumed with linear interpolation between current recycling rate of all C&D waste in Massachusetts (33%) (Government of Massachusetts 2010). Recycling rates for large, metallic structural components were assumed to increase to 97.5% at the demolition time. Because of Massachusetts C&D waste landfilling ban (Government of Massachusetts 2015), it is assumed that large plastic components will also see a high recycling rate in 30 years (95%) since these components are easily source-segregated and we assume that there will exist incentives to promote their collection at the time of demolition (fines or fees for inclusion with landfilled waste), but we linearly interpolate between current recycling rate of all C&D waste in Massachusetts (33%). All materials not recycled are assumed to be landfilled. Table 11 outlines the lifetimes and recycling rates for the various components of the BI-NC operation.

Table 44 - Recycling rates and lifetimes of various components of the BI-NC farm.

Component	**Lifetime (years)**	**Current recycling rate (%)**	**Future recycling rate (%)**						**Applied**	**Substitutes at market**
			5	10	15	20	25	30		
Green roof components										
Root barrier (HDPE)	30	95	-	-	-	-	-	95	95	virgin steel
Moisture mat (PP)	30	33	-	-	-	-	-	95	95	virgin plastic
Drainage mat (HDPE)	30	33	-	-	-	-	-	95	95	virgin plastic
Aggregate (expanded shale)	30	0	-	-	-	-	-	0	0	landfill assumed
Gravel	30	0	-	-	-	-	-	0	0	-
Filter fabric (LDPE)	30	95	-	-	-	-	-	95	95	virgin plastic
Medium (expanded clay)	30	0	-	-	-	-	-	0	0	-
Drain cover (steel)	30	95	-	-	-	-	-	95	95	virgin steel
Shore edge (steel)	30	95	-	-	-	-	-	95	95	virgin steel
Irrigation Components										
Irrigation box (HDPE)	30	95	-	-	-	-	-	95	95	virgin plastics
Irrigation control computer[6]	10	33	-	39	-	45	-	50	45	virgin metals
Pressure regulator[2]	15	33	-	-	42	-	-	50	46	virgin metals
Pump[2]	15	33	-	-	42	-	-	50	46	virgin metals
Tubing (plastics)	30	95	-	-	-	-	-	95	95	virgin plastics
Fencing system										
Steel frame	30	95	-	-	-	-	-	95	95	virgin metals
Weights (HDPE)	30	95	-	-	-	-	-	97.5	97.5	virgin plastics

[6] Only metals recovered; plastic and glass components assumed to go to landfill

Structural components										
Steel	30	95	-	-	-	-	-	97.5	97.5	virgin metals
Operations										
Distribution totes	10	95	-	95	-	95	-	95	95	virgin metals and plastics

BI-NC: Life Cycle Inventory (LCI) - Tomato

Capital inputs, as provided by the contractor, were divided by their lifetimes (table 11) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 9). As the capital inputs apply to the entire farm area, a percentage was allocated to the tomatoes based on the percentage of total growing area occupied by tomatoes (6.8%). For operating inputs the same method was applied with the exception of irrigation demands which were crop specific. Annual capital inputs, outputs (recycled and landfilled), and annual operating inputs were then normalized per kilogram fresh tomato produced by dividing by the annual production of 469 kg/annum. Table 12-22 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1 processes used to model them. Produce is not refrigerated at any point in the supply chain.

Green Roof Components

Table 45 - LCI for BI-NC green roof components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.01170539	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Polypropylene	0.0041951	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Shale	0.07103755	kg/FU	Expanded shale {US-Boston} | Market for | Conseq, U	Transport adjusted to assume distance of 50 miles between extraction site and farm as determined through conversation with supplier.
Gravel	0.12400727	kg/FU	Crushed gravel {US-Boston} | market for | conseq, U	Same as with shale.
Expanded Clay	0.77029327	kg/FU	Expanded clay {US-Boston} | Market for | Conseq, U	Same as with shale.
Steel	0.00064681	kg/FU	Steel, low-alloyed {US}| market for | Conseq, U	Steel assumed to be produced in U.S., since 75% of steel consumption is from domestic production (World Steel Association 2013). Hot rolling assumed to occur using Uses Midwest Reliability Organization (MRO) grid according to manufacturer's location
Material Processing and Freight				
Plastic forming	0.0159005	kg/FU	Extrusion, plastic film {US-MRO}| production | Conseq, U	MRO electrical grid according to manufacturer's location
Component freight	0.0264627	tkm/FU	Transport, freight, loryy >32 metric ton,	Components come from Chicago, U.S. area according to

			EURO6 {RoW}| transport, freight, lorry >32 metric ton, EURO6 | Conseq, U	manufacturer (~1600 km)
Waste Treatment				
Steel recycling	0.00062998	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	Same as BI-C process but with energy grid changed to NPCC region
HDPE recycling	0.00398265	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	Same as above
Polypropylene recycling	0.01112247	kg/FU	PP (waste treatment) {US-NPCC}| recycling of PP | Conseq, U	Same as BI-C process but with energy grid changed to NPCC region
Landfilling	0.96597474	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	Trains using U.S. data

Irrigation System

Table 46 - LCI for BI-NC irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	2.41373E-05	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Polypropylene	4.74176E-05	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Nylon	4.10738E-06	kg/FU	Nylon 6 {GLO}| market for | Conseq, U	
LDPE	0.000407312	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Aluminum	7.7777E-07	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	Aluminum assumed to be produced in U.S., since 75% of steel consumption is from domestic production (U.S. Geological Survey 2013) . Includes energy for shaping.
Steel	3.81113E-05	kg/FU	Steel, low-alloyed, hot rolled {US-WECC}| market for | Conseq, U	Western Electricity Coordinating Council (WECC) grid used for forming, based off of manufacturer's location
Glass	5.59546E-08	kg/FU	Glass, for liquid crystal display {GLO}| production | Conseq, U	
Copper	2.51796E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing and Freight				

Plastic film forming	0.00040695	kg/FU	Extrusion, plastic film {US-WECC}| production | Conseq, U	WECC grid used based on location of manufacture
Plastic pipe forming	7.5598E-05		Extrusion, plastic pipes {US-NPCC}| production | Conseq, U	Assumes manufacturer occurs locally
Copper forming	2.5199E-07	kg/FU	Wire drawing, copper {US-WECC}| processing | Conseq, U	WECC grid used based on location of manufacture
Component freight	0.000267924	tkm/FU	Transport, freight, lorry >32 metric ton, EURO4 {RER}| transport, freight, lorry >32 metric ton, EURO4 | Conseq, U	Transport from West Coast, U.S. to Boston, U.S. (~4880 km)
Waste Treatment				
Steel recycling	1.9117E-05	kg/FU	Steel and iron (waste treatment) {CA-QC}| recycling of steel and iron | Conseq, U	
Copper recycling	3.8958E-07	kg/FU	Copper (waste treatment) {GLO}| recycling of copper | Conseq, U	
Aluminum recycling	3.8958E-07	kg/FU	Aluminium (waste treatment) {CA-QC}| recycling of aluminium | Conseq, U	
LDPE and HDPE recycling	0.00040956	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	Uses local electrical grid
Landfilling	9.3846E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Fence System

Table 47 - LCI for BI-NC fence system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Steel	0.01245883	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0081628	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	For fencing counter-weights which are made of recycled HDPE
Freight	0.01535132	kg/FU	Transport, freight, lorry >32 metric ton, EURO5 {RoW}| transport, freight, lorry >32 metric ton, EURO5 | Conseq, U	Freight from upstate New York
Waste Treatment				
Steel recycling	0.01215072	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
HDPE recycling	0.00774807	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	

Landfilling	0.00071977	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Structural Buttressing

Table 48 - LCI of BI-NC structural buttressing

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Steel	0.17620229	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Waste Treatment				
Steel recycling	0.17179723	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.00440506	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation

Table 49 - LCI for BI-NC annual irrigation and nutrient loading in runoff

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.045002	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Outputs to Nature				
Nitrogen	0.00010592	kg/FU	-	To river
Phosphorous	1.8862E-06	kg/FU	-	To river
Potassium	1.8862E-05	kg/FU	-	To river

Nutrient Demands

Table 50 - LCI for BI-NC nutrient demands

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Nitrogen fertilizer, as N	0.00036749	kg/FU	Ammonium nitrate, as N {GLO}| market for | Conseq, U	
Phosphate fertilizer, as P_2O_5	0.00047121	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium sulfate, as K_2O	0.00025493	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
Rock dust	0.01975976	kg/FU	Basalt {GLO}| market for | Conseq, U	
Transport				
Compost hauling	0.00373987	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	22.5 km between farm and composting center

Distribution

Table 51 - LCI for BI-NC distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.001086178	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Totes used to transport produce to supermarket floor
Steel	0.000217236	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	Totes used to transport produce to supermarket floor

Material Processing				
Plastic forming	0.001086178	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
HDPE recycling	0.001042731	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Steel recycling	0.000208546	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	5.2137E-5	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Building Symbiosis

Table 52 - LCI for BI-NC building symbiosis

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Avoided Products				
Cooling	0.41125325	MJ/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	
Heating	5.60433347	MJ/FU	Heat, central or small-scale, natural gas {CH}| heat production, natural gas, at boiler condensing modulating <100kW | Conseq, U	

Waste Treatment

Table 53 - LCI for BI-NC waste treatment

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Waste Treatment				
Organic waste treatment	0.0987453	kg/FU	garden waste treatment {US-NPCC} | at farm | conseq, U	
Transport				
Organic waste hauling	0.00222173	tkm/FU	Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	

Energy and Fuels

Table 54 - LCI for BI-NC energy and fuels

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Energy and Fuels				
Electricity	0.023392808	kWh/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	
Site visits	0.4729366	km/FU	Transport, passenger car, small size, petrol, EURO 5 {GLO}| market for | Conseq, U	25.5 km one way, twice per week for 37 weeks

Other

Table 55 - LCI for BI-NC other inputs

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Energy and Fuels				
Seedling	1	kg/FU	tomato seedling {at farm}	Inputs scaled to number of seedlings required to grow 1 kg of tomatoes

BI-NC: Life Cycle Inventory (LCI) - Lettuce

LCIs calculated in the same manner as the tomatoes, allocating 3.9% of the growing area to the lettuce and having a harvest of 80 kg/annum. Table 23-33 outline the LCI for a single kilogram of lettuce (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1 processes used to model them. Produce is not refrigerated at any point in the supply chain.

Green Roof Components

Table 56 - LCI for BI-NC green roof components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.039605844	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Polypropylene	0.014194365	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Shale	0.240359476	kg/FU	Expanded shale {US-Boston} | Market for | Conseq, U	Transport adjusted to assume distance of 50 miles between extraction site and farm as determined through conversation with supplier.
Gravel	0.41958546	kg/FU	Crushed gravel {US-Boston} | market for | conseq, U	Same as with shale.
Expanded Clay	2.606329966	kg/FU	Expanded clay {US-Boston} | Market for | Conseq, U	Same as with shale.
Steel	0.002188534	kg/FU	Steel, low-alloyed {US}| market for | Conseq, U	Steel assumed to be produced in U.S., since 75% of steel consumption is from domestic production (World Steel Association 2013). Hot rolling assumed to occur using Uses Midwest Reliability Organization (MRO) grid according to manufacturer's location
Material Processing and Freight				
Plastic forming	0.053800209	kg/FU	Extrusion, plastic film {US-MRO}| production | Conseq, U	MRO electrical grid according to manufacturer's location
Component freight	0.08967009	tkm/FU	Transport, freight, loryy >32 metric ton, EURO6 {RoW}| transport, freight, lorry >32 metric ton, EURO6 | Conseq, U	Components come from Chicago, U.S. area according to manufacturer (~1600 km)
Waste Treatment				
Steel recycling	0.0021315	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	Same as BI-C process but with energy grid changed to NPCC region
HDPE recycling	0.01347501	kg/FU	PE (waste treatment) {US-NPCC}| recycling	Same as above

			of PE | Conseq, U	
Polypropylene recycling	0.03763204	kg/FU	PP (waste treatment) {US-NPCC}| recycling of PP | Conseq, U	Same as BI-C process but with energy grid changed to NPCC region
Landfilling	3.26830327	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	Trains using U.S. data

Irrigation System

Table 57 - LCI for BI-NC irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	8.16667E-05	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Polypropylene	0.000160434	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Nylon	1.3897E-05	kg/FU	Nylon 6 {GLO}| market for | Conseq, U	
LDPE	0.001378111	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Aluminum	2.63153E-06	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	Aluminum assumed to be produced in U.S., since 75% of steel consumption is from domestic production (U.S. Geological Survey 2013) . Includes energy for shaping.
Steel	0.000128947	kg/FU	Steel, low-alloyed, hot rolled {US-WECC}| market for | Conseq, U	Western Electricity Coordinating Council (WECC) grid used for forming, based off of manufacturer's location
Glass	1.89318E-07	kg/FU	Glass, for liquid crystal display {GLO}| production | Conseq, U	
Copper	8.51933E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing and Freight				
Plastic film forming	0.0013769	kg/FU	Extrusion, plastic film {US-WECC}| production | Conseq, U	WECC grid used based on location of manufacture
Plastic pipe forming	0.00025578		Extrusion, plastic pipes {US-NPCC}| production | Conseq, U	Assumes manufacturer occurs locally
Copper forming	8.526E-07	kg/FU	Wire drawing, copper {US-WECC}| processing | Conseq, U	WECC grid used based on location of manufacture
Component freight	0.000906501	tkm/FU	Transport, freight, lorry >32 metric ton, EURO4 {RER}| transport,	Transport from West Coast, U.S. to Boston, U.S. (~4880 km)

			freight, lorry >32 metric ton, EURO4 | Conseq, U	
Waste Treatment				
Steel recycling	6.468E-05	kg/FU	Steel and iron (waste treatment) {CA-QC}| recycling of steel and iron | Conseq, U	
Copper recycling	1.3181E-06	kg/FU	Copper (waste treatment) {GLO}| recycling of copper | Conseq, U	
Aluminum recycling	1.3181E-06	kg/FU	Aluminium (waste treatment) {CA-QC}| recycling of aluminium | Conseq, U	
LDPE and HDPE recycling	0.00138572	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	Uses local electrical grid
Landfilling	0.00031752	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Fence System

Table 58 - LCI for BI-NC fence system

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
Steel	0.04215353	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
HDPE	0.02761821	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.02761821	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Freight	0.05194005	kg/FU	Transport, freight, lorry >32 metric ton, EURO5 {RoW}| transport, freight, lorry >32 metric ton, EURO5 | Conseq, U	Freight from upstate New York
Waste Treatment				
Steel recycling	0.04111104	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
HDPE recycling	0.02621503	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0024353	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Structural Buttressing

Table 59 - LCI of BI-NC structural buttressing

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
Steel	0.59616726	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Waste Treatment				
Steel recycling	0.58126308	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.01490418	kg/FU	Inert waste, for final disposal {US}| market for |	

			Conseq, U	

Irrigation

Table 60 - LCI for BI-NC annual irrigation and nutrient loading in runoff

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.048888	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Outputs to Nature				
Nitrogen	0.0003577	kg/FU	-	To river
Phosphorous	6.37E-06	kg/FU	-	To river
Potassium	6.37E-05	kg/FU	-	To river

Nutrient Demands

Table 61 - LCI for BI-NC nutrient demands

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Nitrogen fertilizer, as N	0.00124338	kg/FU	Ammonium nitrate, as N {GLO}| market for | Conseq, U	
Phosphate fertilizer, as P_2O_5	0.0015943	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium sulfate, as K_2O	0.00086253	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
Rock dust	0.06685567	kg/FU	Basalt {GLO}| market for | Conseq, U	
Transport				
Compost hauling	0.01265355	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	22.5 km between farm and composting center

Distribution

Table 62 - LCI for BI-NC distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.003675	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Totes used to transport produce to supermarket floor
Steel	0.000735	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	Totes used to transport produce to supermarket floor
Material Processing				
Plastic forming	0.01086178	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
HDPE recycling	0.003528	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Steel recycling	0.0007056	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.0001764	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Building Symbiosis

Table 63 - LCI for BI-C building symbiosis

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Avoided Products				
Cooling	1.39144459	MJ/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	
Heating	18.961843	MJ/FU	Heat, central or small-scale, natural gas {CH}| heat production, natural gas, at boiler condensing modulating <100kW | Conseq, U	

Waste Treatment

Table 64 - LCI for BI-NC waste treatment

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Waste Treatment				
Organic waste treatment	0.3340912	kg/FU	garden waste treatment {US-NPCC} | at farm | conseq, U	
Transport				
Organic waste hauling	0.00751705	tkm/FU	Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	

Energy and Fuels

Table 65 - LCI for BI-NC energy and fuels

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Energy and Fuels				
Electricity	0.07914782	MJ/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	
Site visits	1.6001456	km/FU	Transport, passenger car, small size, petrol, EURO 5 {GLO}| market for | Conseq, U	25.5 km one way, twice per week for 37 weeks

Other

Table 66 - LCI for BI-NC other inputs

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Energy and Fuels				
Seedling	1	kg/FU	lettuce seedling {at farm}	Inputs scaled to number of seedlings required to grow 1 kg of lettuce

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Appendix C: Life cycle inventory for ground-based conditioned farming system 1(GB-C1)

The ground-based conditioned UA site (referred to as 'GB-C1' in the article text) is an operating urban greenhouse in Metropolitan Boston, US. The greenhouse is in a traditional gable style and consists of 1169 m^2 of growing area; 557.75 m^2 allocated to commercial production and 488.75 m^2 for communal growing space. There is additional building space for offices and storage (~122.5 m^2), and outdoor space for vehicle parking and waste management. *The LCA only concerns itself with the commercial growing space which is operated by the farm itself*. The space is used to grow three crops; tomatoes form March through September, salad greens (lettuce) from October through February and a small amount of bean shoots throughout the year. The operation is soil-based, with the plants grown in raised beds that receive annual top-ups with fertilizer, with water supplied through drip irrigation.

Primary data was available for all capital and operating inputs. Capital was accounted through site visits, engineering drawings and manufacturer literature. Operating inputs were taken from interviews with the farm operator and utilities invoices. Because multiple crops are grown throughout the year the various inputs had to be allocated to the tomatoes and lettuce. Table 1 outlines the allocation methods.

Table 67 - Allocation key for GB-C1

Aspect	**Allocation Method**	**Tomato**	**Lettuce**	**Pea Shoots**
Equipment, site visits, compost inputs, land occupation	% growing period length as a fraction of the year times the fraction of total mass of food produced in that period	0.49	0.41	0.1
Nutrient inputs	Mass of produced crop	0.9	0.07	0.03
Utilities (water, heat, electricity), distribution	Consumption during growing period times the fraction of total mass of food produced in that period	0.98 (Mar - Sept)	0.69 (Oct - Feb)	0.02 (Mar - Sept), 0.31 (Oct - Sept)

The supplementary information is arranged as follow: description of estimation of capital inputs where primary data was lacking; description of operational inputs where primary data was lacking; component lifetimes and recycling rates; life cycle inventory for functional unit.

GB-C1: Capital Inputs

Primary data was available for all equipment. In some instances, the mass of a component was known (e.g. fans, light fixtures, etc.), but not the material composition (steel, aluminum, etc.). In these cases the breakdown of materials was estimated using the same method employed by Rosado et al. (2014). Table 2 outlines the material composition of various categories of equipment used in this study.

Table 68 - The material composition of various pieces of capital equipment. The category of material according to Rosado et al. (2014) is shown, with the author's assumption of the specific material in brackets. The totals do not add up to 100% as some categories were ignored if they were too vague to make a reasonable assumption about (e.g. non-specified biomass, precious metals, etc.), though the ignored categories tended to be minor fractions of the masses of the components.

Material	Multilayer circuit	Standard Air Conditioner (also used for furnace)	Fluorescent light fixture
Plastics (HDPE)	0.2	0.25	0.04
Ferrous metals (Steel)	0.32	0.66	0.02
Light metals	0.2	0.03	0.15
Nonferrous heavy metals (copper)	0.28	0.02	0.17
Glass	0	<0.01	0.6

GB-C1: Operating Characteristics and Inputs

The operating inputs for the GB-C1 farm were taken from primary data provided by the farm operator, though some aspects had to be estimated, namely, space conditioning to grow seedlings, runoff retention and imported compost production. In general, the same electricity mixes were used as employed for the BI-NC operation (see S1.2 for details). No nutrient runoff is expected since all of the fertilization occurs within the greenhouse.

Seedling Production

Production of seedlings (small plants grown in greenhouses that are transferred to outdoor soil) is based off of the methods of Stoessel at al. (2012) for lettuce and tomatoes. For details see S1.2 of the BI-NC system.

Runoff Retention

A rainwater collection and storage system is present at the site, but according to interviews with the farm operator it is seldom used for irrigation and they had no estimate for how much it was used. We have taken a conservative estimate that the tank only captures its own volume (11.37 m^3) of rainwater throughout the year. This is much less than the volume that fell on the site in 2014 (1.17 m * 557.75 m^2 = 652 m^3) (Boston Sewer and Water Commission 2015), but trusting the head farmer, who is on site for almost every operating day of the year, it is a safe assumption that rainwater capture plays a small role in the irrigation of the site. All captured rainwater is assumed to avoid runoff to combined sewers, and is modeled as avoided wastewater treatment demand.

Imported Compost Production

Method applied to BI-NC system was also used (see S1.2 for further details).

Waste Management

The GB-C1 operation produces compost on site, but this is not included in the inventory since the method is very low input (manual turning of open piles). The resulting compost is given away to low-income residents of Boston, meaning that it is not replacing fertilizer purchases on the market, and, thus, is not included in a consequential LCA, although the system is credited for the amount of organic waste that would have been sent to landfill in lieu of the on-site composting. The farm also sends a small amount of inorganic waste to landfill, but this has not been included in the inventory, since the farm manager did not have an estimate for this.

GB-C1: Component lifetimes and recycling rates

Same heuristics for the lifetimes and recycling rates as applied to the BI-NC were applied here, since they both operate within the Metropolitan Boston, U.S. region, and rely on the same waste management system. The assumed lifetime of the overall GB-C1 operation is 30 years. Table 3 outlines the different component lifetimes and disposal rates in the GB-C1 system.

Table 69 - Lifetime and recycling rate of different components in the GB-C1 system

Component	Lifetime (years)	Current recycling rate (%)	Future recycling rate (%)						Applied rate (%)	Substitutes at market
			5	10	15	20	25	30		
Structural components										
Steel columns	30	95	-	-	-	-	-	97.5	97.5	virgin metal
Steel beams	30	95	-	-	-	-	-	97.5	97.5	virgin metal
Skin (polycarbonate)	30	33	-	-	-	-	-	95	95	virgin plastic
Aluminum girders	30	15	-	-	-	-	-	97.5	97.5	virgin metal
Interior dividing wall	30	33	-	-	-	-	-	95	95	virgin plastic
Concrete foundation[7]	30	61	-	-	-	-	-	92	92[8]	gravel
Mechanical components										
Gearbox	30	95	-	-	-	-	-	-	95	virgin metal
Motor	20	33	-	-	-	50	-	-	50	virgin metals
Shaft	30	95	-	-	-	-	-	-	95	virgin metal
HVAC components										
Inlet fans	20	33	-	-	-	50	-	50	50	virgin metals
Overheard fans	20	33	-	50	-	50	-	50	50	virgin metals
Furnace	30	33	-	-	-	-	-	50	50	virgin metals
Control unit	10	33	-	50	-	50	-	50	50	virgin metals
Tubing	30	95	-	-	-	-	-	50	50	virgin metals
Electrical components										
Lamp	10	33	-	50	-	50	-	50	50	virgin metals
Light fixture	30	95	-	-	-	-	-	95	95	virgin metals
Irrigation components										
Tubing	10	33	-	95	-	95	-	95	95	virgin plastic
Tanks	30	33	-	-	-	-	-	95	95	virgin plastic
Manifold	30	33	-	-	-	-	-	95	95	virgin plastic
Pump	15	33	-	-	50	-	-	50	50	virgin metals
Hose	30	33	-	-	-	-	-	-	95	virgin plastic
Other										
Plastic drums	20	33	-	-	-	95	-	-	95	virgin plastic
Distribution totes	10	33	-	95	-	95	-	95	95	virgin metal and plastic
Wood[9]	30	33	-	-	-	-	-	0	0	-

[7] Assumed to be left in the ground upon demolition of structure.

[8] Massachusetts aims for 80% decrease in landfilling by from 2010 rate by 2050 (Government of Massachusetts 2015). The concrete recycling rate has been estimated assuming that this goal is achieved by the demolition date.

[9] Part of the raised beds. Assumed to be incinerated.

GB-C1: Life Cycle Inventory (LCI) - Tomato

Capital inputs, as provided by the contractor, were multiplied by the time allocated to tomato growing (see table 1) by their lifetimes (table 3) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 3). For operating inputs the allocation key in table 1 was applied. The allocated capital inputs, outputs (recycled and landfilled), and operating inputs were then normalized per kilogram fresh tomato produced by dividing by the annual production of 5455 kg/annum. Table 4-13 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1.1 processes used to model them. Transport was generally ignored unless the distances were large (e.g. greenhouse components coming from California) or the freight's mass significant (e.g. concrete).

Greenhouse components

Table 70 - LCI for BI-C1 greenhouse components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				
Land Occupation	0.0502745	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Steel	0.0190884	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Aluminum	0.0004673	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	
Polycarbonate	0.0030936	kg/FU	Polycarbonate {GLO}| market for | Conseq, U	
PVC	0.0011117	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Concrete	0.0001717	m^3/FU	Concrete, normal {US-NPCC}| production | Conseq, U	
Wood	0.00820899	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Material Processing and Freight				
Plastic forming	0.0042052	kg/FU	Extrusion, plastic film {US-WSCC}| production | Conseq, U	WSCC electrical grid according to manufacturer's location
Component freight	0.0038612	tkm/FU	Transport, freight, lorry >32 metric ton, EURO5 {RER}| transport, freight, lorry >32 metric ton, EURO5 | Conseq, U	Greenhouse structural components (sans concrete) come from California manufacturer (~4875 km)
Concrete freight	0.0167657	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {RoW}| transport, freight, lorry 16-32 metric ton, EURO4 | Conseq, U	Assumed to be transported 40 km to site
Waste Treatment				
Steel recycling	0.018134	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Aluminum recycling	0.0004556	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	

Polycarbonate recycling	0.0029389	kg/FU	Polycarbonate (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.0010561	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Concrete recycling	0.0001583	kg/FU	Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	
Landfilling	0.0011898	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.00820899	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

HVAC System

Table 71 - LCI for GB-C1 HVAC system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Steel	0.0017208	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	0.0002652	kg/FU	Copper {GLO}| market for | Conseq, U	
Aluminum	0.000224	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	
HDPE	0.0004568	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Glass	9.656E-06	kg/FU	Flat glass, coated {GLO}| market for | Conseq, U	
Material Processing and Freight				
Plastic film forming	0.0004568	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	2.5199E-07	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Pipe forming	6.916E-05	kg/FU	Drawing of pipe, steel {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0008604	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	4.275E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Aluminum recycling	0.000112	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
HDPE recycling	0.0002284	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0014329	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Mechanical System

Table 72 - LCI for GB-C1 mechanical system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				

Steel	0.01245883	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	7.466E-05	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Copping forming	7.466E-05	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0015693	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	3.733E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Landfilling	0.0007093	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation System

Table 73 - LCI of GB-C1 irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.0006958	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
HDPE	0.000519	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
PVC	0.0002829	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Polypropylene	1.475E-05	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Rubber	3.278E-05	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
Copper	1.639E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0015452	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	1.639E-06	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
HDPE recycling	0.00066104	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
LDPE recycling	0.00049303	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.00026874	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	Inventory the same as PE recycling but with PVC as input and avoided product
Copper recycling	8.1944E-07	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of	

			aluminium | Conseq, U	
Rubber recycling	3.1139E-05	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Landfilling	9.2093E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Lighting System

Table 74 - LCI of GB-C1 lighting system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	1.2941E-06	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Aluminum	5.128E-06	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Steel	0.000201	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Glass	2.0653E-05	kg/FU	Flat glass, coated {GLO}| market for | Conseq, U	
Copper	5.9127E-06	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	1.2941E-06	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	5.9127E-06	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.00019064	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Aluminum recycling	2.564E-06	kg/FU	Aluminum (waste treatment) {US-NPCC}| recycling of Aluminum | Conseq, U	
Copper recycling	2.9564E-06	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Landfilling	3.7203E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Space Conditioning

Table 75 - LCI for GB-C1 space conditioning during tomato cultivation

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Heating	2.8069362	kWh/FU	Heat, central or small-scale, natural gas {CH}| heat production, natural gas, at boiler condensing modulating <100kW | Conseq, U	
Electricity	1.2735134	kWh/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	

Irrigation

Table 76 - LCI for GB-C1 irrigation and avoided runoff during tomato growing season

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				

Irrigation water	0.1945531	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.0010248	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands

Table 77 - LCI for GB-C1 nutrient consumption during tomato cultivation period

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
Nitrogen fertilizer, as N	0.00102137	kg/FU	Ammonium nitrate, as N {GLO}| market for | Conseq, U	
Phosphate fertilizer, as P_2O_5	0.00221773	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium sulfate, as K_2O	0.0006698	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
Transport				
Compost hauling	0.0253244	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	14.1 km between farm and composting center

Waste Management

Table 78 - LCI for GB-C1 waste management during tomato cultivation period

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Avoided Production				
Waste treatment	0.8767327	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	14.1 km between farm and composting center

Distribution

Table 79 - LCI for GB-C1 distribution

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
HDPE	0.0064016	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Totes
Paper	0.0080927	kg/FU	Kraft paper, unbleached {GLO}| market for | Conseq, U	Bags
Material Processing and Transport				
Plastic forming	0.0064016	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Distribution to markets	0.1135645	km	Transport, passenger car, large size, petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	Driven in van to various markets around the Boston area
Waste Treatment				
HDPE recycling	0.006081481	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Paper recycling	0.0040464	kg/FU	Paper (waste treatment) {GLO}| recycling of paper | Conseq, U	
Landfilling	0.0101278	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Other

Table 80 - LCI for GB-C1 other inputs

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	3.0229E-05	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Waste storage drums (55 gal)
Material Processing				
Plastic forming	3.0229E-05	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Energy and Fuels				
Seedling	1	kg/FU	tomato seedling {at farm}	Inputs scaled to number of seedlings required to grow 1 kg of tomatoes
Waste Treatment				
LDPE recycling	2.87178E-05	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	1.51146E-06	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

GB-C1: Life Cycle Inventory (LCI) - Lettuce

Same method that was used for the tomatoes was applied to the lettuce. The allocated capital inputs, outputs (recycled and landfilled), and operating inputs were then normalized per kilogram fresh tomato produced by dividing by the annual production of 403 kg/annum. Table 14-23 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1.1 processes used to model them. Produce is not refrigerated at any point in the supply chain.

Greenhouse components

Table 81 - LCI for GB-C1 greenhouse components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				
Land Occupation	0.5626089	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Steel	0.2136138	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Aluminum	0.0052294	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	
Polycarbonate	0.0346195	kg/FU	Polycarbonate {GLO}| market for | Conseq, U	
PVC	0.0124402	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Concrete	0.0019215	m^3/FU	Concrete, normal {US-NPCC}| production | Conseq, U	
Wood	0.0918646	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Material Processing and Freight				
Plastic forming	0.0470598	kg/FU	Extrusion, plastic film {US-	WSCC electrical grid

			WSCC}| production | Conseq, U	according to manufacturer's location
Component freight	0.0432092	tkm/FU	Transport, freight, lorry >32 metric ton, EURO5 {RER}| transport, freight, lorry >32 metric ton, EURO5 | Conseq, U	Greenhouse structural components (sans concrete) come from California manufacturer (~4875 km)
Concrete freight	0.1876204	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {RoW}| transport, freight, lorry 16-32 metric ton, EURO4 | Conseq, U	Assumed to be transported 40 km to site
Waste Treatment				
Steel recycling	0.2029331	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Aluminum recycling	0.0050987	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Polycarbonate recycling	0.0328886	kg/FU	Polycarbonate (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.0118182	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Concrete recycling	0.0017717	kg/FU	Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	
Landfilling	0.0133142	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.2029331	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

HVAC System

Table 82 - LCI for GB-C1 HVAC system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Steel	0.0192573	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	0.0029675	kg/FU	Copper {GLO}| market for | Conseq, U	
Aluminum	0.0025063	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	
HDPE	0.0051124	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Glass	0.0001081	kg/FU	Flat glass, coated {GLO}| market for | Conseq, U	
Material Processing and Freight				
Plastic film forming	0.0051124	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	0.0029675	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Pipe forming	0.0007739	kg/FU	Drawing of pipe, steel {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				

Steel recycling	0.0096286	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	0.0004784	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Aluminum recycling	0.0012531	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
HDPE recycling	0.0025562	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0160352	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Mechanical System

Table 83 - LCI for GB-C1 mechanical system

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
Steel	0.0250814	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	0.0008355	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Copping forming	0.0008355	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0015693	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	3.733E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Landfilling	0.0007093	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation System

Table 84 - LCI of GB-C1 irrigation system

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
LDPE	0.0077869	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
HDPE	0.0058077	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
PVC	0.0031657	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Polypropylene	0.0001651	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Rubber	0.0003668	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
Copper	1.834E-05	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0172922	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq,	Local production assumed

			U	
Copper forming	1.834E-05	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
HDPE recycling	0.0073975	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
LDPE recycling	0.0055173	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.0030074	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	Inventory the same as PE recycling but with PVC as input and avoided product
Copper recycling	9.17E-06	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Rubber recycling	0.0003485	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Landfilling	0.0010306	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Lighting System

Table 85 - LCI of GB-C1 lighting system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	1.448E-05	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Aluminum	5.739E-05	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Steel	0.0022496	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Glass	0.0002311	kg/FU	Flat glass, coated {GLO}| market for | Conseq, U	
Copper	6.617E-05	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	1.448E-05	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	6.617E-05	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0021334	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Aluminum recycling	2.869E-05	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Copper recycling	3.308E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Landfilling	0.0004163	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Space Conditioning

Table 86 - LCI for GB-C1 space conditioning during tomato cultivation

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Heating	82.293991	kWh/FU	Heat, central or small-scale, natural gas {CH}| heat production, natural gas, at boiler condensing modulating <100kW | Conseq, U	
Electricity	18.225504	kWh/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	

Irrigation

Table 87 - LCI for GB-C1 irrigation and avoided runoff during tomato growing season

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.6033146	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.0140969	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands

Table 88 - LCI for GB-C1 nutrient consumption during tomato cultivation period

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Nitrogen fertilizer, as N	0.00102137	kg/FU	Ammonium nitrate, as N {GLO}| market for | Conseq, U	
Phosphate fertilizer, as P_2O_5	0.00221773	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium sulfate, as K_2O	0.0006698	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
Transport				
Compost hauling	0.0253244	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	14.1 km between farm and composting center

Waste Management

Table 89 - LCI for GB-C1 waste management during lettuce cultivation period

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Avoided Production				
Waste treatment	0.8767327	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	14.1 km between farm and composting center

Distribution

Table 90 - LCI for GB-C1 distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.0064016	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Totes
Paper	0.0080927	kg/FU	Kraft paper, unbleached {GLO}| market for | Conseq, U	Bags
Material Processing and Transport				
Plastic forming	0.0064016	kg/FU	Extrusion, plastic film {US-NPCC}|	

			production | Conseq, U	
Distribution to markets	0.8194745	km	Transport, passenger car, large size, petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	Driven in van to various markets around the Boston area
Waste Treatment				
HDPE recycling	0.006081481	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Paper recycling	0.0040464	kg/FU	Paper (waste treatment) {GLO}| recycling of paper | Conseq, U	
Landfilling	0.0101278	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Other

Table 91 - LCI for GB-C1 other inputs

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.0003383	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Waste storage drums (55 gal)
Material Processing				
Plastic forming	0.0003383	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Energy and Fuels				
Seedling	1	kg/FU	lettuce seedling {at farm}	Inputs scaled to number of seedlings required to grow 1 kg of lettuce
Waste Treatment				
LDPE recycling	0.000321373	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	1.69144E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

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Appendix D: Life cycle inventory for ground-based conditioned farming system 2 (GB-C2)

The ground-based conditioned UA site (referred to as 'GB-C2' in the article text) is a modular farming system that is designed to be completed isolated from the elements; requiring artificial lighting throughout the year, heat in the winter and cooling in the summer. Because of the proprietary nature of the technology, numerous details regarding the calculation of the LCI are, unfortunately, omitted from the following sections.

Primary data was available for all capital and operating inputs. Capital was accounted using parts lists from the farm designer. Operating inputs were taken from interviews with the farm operator and utilities invoices, for the production of lettuce.

The supplementary information is arranged as follow: capital inputs; operating inputs; life cycle inventory for functional unit.

GB-C2: Capital Inputs
Primary data was available for all equipment. In some instances, the mass of a component was known (e.g. fans, light fixtures, etc.), but not the material composition (steel, aluminum, etc.). In these cases the breakdown of materials was estimated using the same method employed by Rosado et al. (2014). Table 2 outlines the material composition of various categories of equipment used in this study. Components made of recycled materials have no burdens attributed to the BI-C2 system (unless the material comes from a constrained market, such as steel, see: http://consequential-lca.org/) since the waste treatment is allocated to the previous life-cycle of the material, though any further manipulation or freight of the material is allocated to the BI-C2.

GB-C2: Operating Characteristics and Inputs
The operating inputs for the GB-C1 farm were taken from primary data provided by the farm operator. No estimations were necessary.

GB-C2: Component lifetimes and recycling rates
Same heuristics for the lifetimes and recycling rates as applied to the BI-NC were applied here, since they both operate on the eastern seaboard of the U.S., and rely on similar waste management systems. The assumed lifetime of the overall GB-C2 operation is 30 years. Table 2 outlines the different component lifetimes and disposal rates in the GB-C2 system.

Table 92 - Lifetime and recycling rate of different components in the GB-C1 system

Component	Lifetime (years)	Current recycling rate (%)	Future recycling rate (%)						Applied rate (%)	Substitutes at market
			5	10	15	20	25	30		
Structural components										
Steel components	30	95	-	-	-	-	-	95	95	virgin metal
Aluminum components	30	15	-	-	-	-	-	95	95	virgin metal
Rubber	10	33	-	95	-	95	-	95	95	virgin plastic
Insulation	30	33	-	-	-	-	-	95	95	virgin plastic
Wood	30	61	-	-	-	-	-	95	95	incinerated
HVAC components										
Fans	10	33	-	50	-	50	-	50	50	virgin metals
AC/Heater	20	33	-	-	-	50	-	50	50	virgin metals
Tubing	30	33	-	-	-	-	-	50	50	virgin plastics
Electrical components										
Lights	15	33	-	-	50	-	-	50	50	virgin metals
Sensors	15	33	-	-	50	-	-	50	50	virgin metals
Computer control system	15	33	-	-	50	-	-	50	50	virgin metals and plastics
Cables	30	33	-	-	-	-	-	50	50	virgin metals
Irrigation components										
Tubing	30	33	-	95	-	95	-	95	95	virgin plastic
Tanks	30	33	-	-	-	-	-	95	95	virgin plastic
Pump	15	33	-	-	50	-	-	50	50	virgin metals

GB-C1: Life Cycle Inventory (LCI) - Lettuce

Table 4-13 outline the LCI for a single kilogram of lettuce (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1.1 processes used to model them.

Structural components

Table 93 - LCI for GB-C2 greenhouse components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				
Land Occupation	0.034375	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Steel	0.1248958	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Aluminum	0.0068406	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	
Fibreboard	2.739E-05	m^3/FU	Medium density fibreboard {GLO}| market for | Conseq, U	
Material Processing and Freight				
Component freight	0.0083188	tkm/FU	Transport, freight, lorry >32 metric ton, EURO5 {RER}| transport, freight, lorry >32 metric ton, EURO5 | Conseq, U	
Waste Treatment				
Steel recycling	0.0899479	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	

Aluminum recycling	0.0057979	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
LDPE recycling	0.0097052	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	original material was recycled –no raw material production burdens
Rubber recycling	1.501E-06	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	same as above
Landfilling	0.0457188	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.0152396	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

Electrical Wiring

Table 94 - LCI for GB-C2 wiring system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Steel	8.261E-05	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	0.0004136	kg/FU	Copper {GLO}| market for | Conseq, U	
PVC	8.846E-05	kg/FU	Aluminium, primary, ingot {US}| market for | Conseq, U	
LDPE	9.396E-05	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Material Processing and Freight				
Plastic film forming	0.0001822	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	0.0004136	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	2.063E-05	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Copper recycling	0.0002051	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
PVC recycling	3.781E-05	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
LDPE recycling	4.698E-05	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0003678	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Seedling System

Table 95 - LCI for GB-C2 seedling system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Steel	0.0008204	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	0.0002739	kg/FU	Copper {GLO}| market for | Conseq, U	
ABS	0.0002739	kg/FU	Acrylonitrile-butadiene-styrene copolymer {GLO}| market for | Conseq, U	

Rubber	1.742E-05	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
PVC	4.858E-05	kg/FU	Polyvinylidenchloride, granulate {GLO}| market for | Conseq, U	
Material Processing				
Copping forming	0.0002739	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Plastic forming	0.0003392	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
Steel recycling	0.0004102	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
PVC recycling	4.618E-05	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
ABS recycling	0.0001364	kg/FU	ABS (waste treatment) {US-NPCC}| recycling of ABS | Conseq, U	
Rubber recycling	0.0000165	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Copper recycling	0.0001364	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Landfilling	0.0006875	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Electrical plates and covers

Table 96 - LCI of GB-C2 for electrical plates and covers

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Polycarbonate	0.0006958	kg/FU	Polycarbonate {GLO}| market for | Conseq, U	
Nylon	0.000519	kg/FU	Nylon 6 {GLO}| market for | Conseq, U	
PVC	0.0002829	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0015452	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Nylon recycling	8.25E-07	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.0000033	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Polycarbonate recycling	4.159E-05	kg/FU	Polycarbonate (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	2.406E-06	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Electrical Panels and Service Equipment

Table 97 - LCI of GB-C2 lighting system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Nylon	8.468E-05	kg/FU	Nylon 6 {GLO}| market for | Conseq, U	
Steel	2.12E-05	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}|	

			market for | Conseq, U	
Copper	0.0001753	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0001753	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	8.468E-05	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	3.369E-06	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Nylon recycling	4.24E-05	kg/FU	Nylon (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Copper recycling	8.766E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Landfilling	0.0001409	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Electronic Monitoring System

Table 98 - LCI of GB-C2 electronic monitoring system (sensors, etc.)

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.0004411	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Steel	0.0006841	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Glass	7.643E-08	kg/FU	Flat glass, coated {US-NPCC}| market for | Conseq, U	
Aluminum	0.0001765	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Copper	0.0003518	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0004411	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Copper forming	0.0003518	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0003426	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Aluminum recycling	8.353E-05	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
HDPE recycling	0.00022	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Glass recycling	3.816E-08	kg/FU	Glass (waste treatment) {US-NPCC}| recycling of glass | Conseq, U	
Copper recycling	0.0001765	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Landfilling	0.0008273	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Lighting System

Table 99 - LCI of GB-C2 lighting system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
PVC	0.0017073	kg/FU	Polyvinylidenchloride, granulate {GLO}| market for | Conseq, U	
Steel	0.0022917	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Glass	5.729E-05	kg/FU	Flat glass, coated {US-NPCC}| market for | Conseq, U	
Aluminum	0.0002292	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Material Processing				
Plastic forming	0.0017073	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0011458	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Aluminum recycling	0.0001146	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
PVC recycling	0.0008514	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Landfilling	0.0021656	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation System

Table 100 - LCI of GB-C2 irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.0002418	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
PVC	0.001375	kg/FU	Polyvinylidenchloride, granulate {GLO}| market for | Conseq, U	
PTFE	3.816E-06	kg/FU	Tetrafluoroethylene {GLO}| market for | Conseq, U	Modeled as base monomer
Nylon	7.986E-06	kg/FU	Nylon 6 {GLO}| market for | Conseq, U	
Brass	0.0001094	kg/FU	Brass {GLO}| market for | Conseq, U	
Wood	2.601E-05	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Steel	0.0003552	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Glass	7.643E-08	kg/FU	Flat glass, coated {US-NPCC}| market for | Conseq, U	
Aluminum	0.0001765	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Copper	0.0001157	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic sheet forming	0.0001157	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production

				assumed
Plastic tube forming	0.001501	kg/FU	Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	
Steel pipe forming	0.0001169	kg/FU	Drawing of pipe, steel {US-NPCC}| processing | Conseq, U	
Copper forming	0.0001157	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.000181	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Brass recycling	0.0001039	kg/FU	Brass (waste treatment) {US-NPCC}| recycling of brass | Conseq, U	
HDPE recycling	0.0001776	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.0012948	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Nylon recycling	7.585E-06	kg/FU	Nylon (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PTFE recycling	3.632E-06	kg/FU	PTFE (waste treatment) {US-NPCC}| recycling of PTFE | Conseq, U	
Copper recycling	5.809E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
Landfilling	0.0004022	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Grow Media

Table 101 - LCI of GB-C2 grow media

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
PVC	0.0366667	kg/FU	Polyvinylidenchloride, granulate {GLO}| market for | Conseq, U	
Steel	7.815E-05	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Aluminum	0.0002956	kg/FU	Aluminium removed by milling, small parts {US-NPCC}| aluminium milling, small parts | Conseq, U	
Material Processing				
Plastic forming	0.1970833	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	0.0348333	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Aluminum recycling	7.425E-05	kg/FU	Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	
PETE recycling	0.0002807	kg/FU	PET (waste treatment) {US-NPCC}| recycling of PET | Conseq, U	original material was recycled –no raw material production burdens
PVC recycling	0.1523958	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Landfilling	0.0098542	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Electronic Devices

Table 102 - LCI of GB-C2 electronic devices

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Nylon	2.051E-05	kg/FU	Nylon 6 {GLO}| market for | Conseq, U	
Steel	5.122E-06	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Copper	2.555E-05	kg/FU	Copper {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	2.555E-05	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Wire forming	2.051E-05	kg/FU	Wire drawing, copper {US-NPCC}| processing | Conseq, U	Local production assumed
Waste Treatment				
Steel recycling	2.555E-06	kg/FU	Steel (waste treatment) {US-NPCC}| recycling of Steel | Conseq, U	
Nylon recycling	1.024E-05	kg/FU	Nylon (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Copper recycling	1.283E-05	kg/FU	Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	
Landfilling	2.555E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Boxes and Cans

Table 103 - LCI of GB-C2 boxes and cans

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
PVC	0.0008147	kg/FU	Polyvinylidenchloride, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0008147	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
PVC recycling	0.0007746	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Landfilling	4.079E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Electrical Conduits

Table 104 - LCI of GB-C2 electrical conduits

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
PVC	0.0013865	kg/FU	Polyvinylidenchloride, granulate {GLO}|	

			market for | Conseq, U	
Material Processing				
Plastic forming	0.0013865	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
PVC recycling	0.0013177	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Landfilling	6.955E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Space Conditioning

Table 105 - LCI for GB-C2 space conditioning during tomato cultivation

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Electricity	33.229167	kWh/FU	Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	

Irrigation

Table 106 - LCI for GB-C2 irrigation and avoided runoff during tomato growing season

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.0180657	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.0010248	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands

Table 107 - LCI for GB-C2 nutrient consumption during

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Nitrogen fertilizer, as N	0.0028848	kg/FU	Ammonium nitrate, as N {GLO}| market for | Conseq, U	
Phosphate fertilizer, as P_2O_5	0.0012348	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium sulfate, as K_2O	0.005852	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
Potassium carbonate	0.0037448	kg/FU	Potassium carbonate {GLO}| market for | Conseq, U	

Distribution

Table 108 - LCI for GB-C2 distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.022	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	Bags
Material Processing and Transport				
Plastic forming	0.022	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Distribution to markets	0.5958333	km	Transport, passenger car, large size, petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	Driven in van to various markets around the Boston area

Waste Treatment				
LDPE recycling	0.0073	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.0147	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Waste Management

Table 109 - LCI for GB-C2 waste management

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Avoided Production				
Nitrogen fertilizer, as N	0.0001568	kg/FU	Ammonium nitrate, as N {GLO}| market for | Conseq, U	
Phosphate fertilizer, as P_2O_5	1.114E-05	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium sulfate, as K_2O	0.0002386	kg/FU	Potassium nitrate {GLO}| market for | Conseq, U	
Waste Treatment				
Paper recycling	0.01375	kg/FU	Paper (waste treatment) {GLO}| recycling of paper | Conseq, U	
Landfilling	0.075	kg/FU	Inert waste, for final disposal {GLO}| market for | Conseq, U	

Miscellaneous Operational Inputs

Table 110 - LCI for GB-C2 miscellaneous operational inputs

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.001	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Waste covered in waste management
Cotton	0.0041905	kg/FU	Textile, woven cotton {GLO}| market for | Conseq, U	Waste covered in waste management
Tissue Paper	0.0275	kg/FU	Tissue paper {GLO}| market for | Conseq, U	Waste covered in waste management

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Appendix E: Life cycle inventory for ground-based non-conditioned farming system 1(GB-NC1)

The ground-based non-conditioned UA site (referred to as 'GB-NC1' in the article text) is an operating urban farm in Metropolitan Boston, US. The farm consists of a 560 m^2 lot surrounded primarily be residential land uses, located about 10 km from the center of Boston. The space is used to grow ten crops from May through October directly in the sites overburden. The cultivated area covers 83% of the total lot. Water is supplied primarily through rainfall and is supplemented by municipal water during dry periods using a single hose. No fertilizer is currently employed at the site since the farm received a fresh dressing of fertilizer over the entire site a year earlier (14 cm in depth). The plan is to continue to topping up the site with fresh compost every 5 years or thereabouts in lieu of fertilizer application.

Primary data was available for all capital and operating inputs. Capital was accounted through site visits. Operating inputs were taken from interviews with the farm operator and utilities invoices. Because multiple crops are grown throughout the year the various inputs had to be allocated to the tomatoes and greens (arugula). Table 1 outlines the allocation methods.

Table 111 - Allocation key for GB-NC1

Aspect	Allocation Method	Tomato	Greens
Equipment, site visits, compost inputs, land occupation, runoff mitigation	% of growing area occupied	0.2	0.02
Irrigation	Crop specific water needs	0.26	0.01
Distribution	% mass produced	0.43	0.01

The supplementary information is arranged as follow: description of estimation of capital inputs where primary data was lacking; description of operational inputs where primary data was lacking; component lifetimes and recycling rates; life cycle inventory for functional unit.

GB-C1: Capital Inputs

Primary data was available for all equipment.

GB-C1: Operating Characteristics and Inputs

The operating inputs for the GB-C1 farm were taken from primary data provided by the farm operator, though some aspects had to be estimated, namely, space conditioning to grow seedlings, runoff retention and imported compost production. In general, the same electricity mixes were used as employed for the BI-NC operation (see S1.2 for details). No nutrient runoff is expected since all of the fertilization occurs within the greenhouse.

Seedling Production

Production of seedlings (small plants grown in greenhouses that are transferred to outdoor soil) is based off of the methods of Stoessel at al. (2012) for lettuce and tomatoes. For details see S1.2 of the BI-NC system.

Runoff Retention

The volume of rain to fall on the site was taken as the amount of precipitation over a calendar year (1.14 m) in the Dorchester neighborhood (Boston Sewer and Water Commission 2015) times the site area (560 m^2), amounting to 638.4 m^3. It was assumed that 50% of the rain was lost as runoff and the 50% captured on site and transferred to the groundwater system (Forman 2014), amounting to 319.2 m^3.

To determine the increase in the amount of avoided runoff as a result of the farm's presence, we looked at the amount of impermeable coverage in the vicinity of the farm. Impermeable coverage on lots on the same block is 48% as calculated using Geographic Information System software (see figure 1). It was assumed that the presence of the farm was thus only increasing the amount of permeable area on the lot by the same amount, compared to a scenario where the lot would be built up in accordance with the historic precedent of the area. Thus 48% of the avoided runoff was attributed to the farm's presence, equaling 154.2 m^3. Because the farm is located in an area with combined sewers, the increased rainwater capture was modeled as avoided wastewater treatment in accordance with the consequential LCA modelling employed here.

Figure 11 - GB-NC1 site with surroundings shown. Green blocks represent buildings, while yellow forms are other impermeable surfaces. On average a built up plot in the area would be covered with 48% impermeable surfaces.

Irrigation

Though data on the amount of municipal rainwater used for irrigation was available from the farmer, the distribution amongst the different crops was unknown. To estimate the irrigation demands of the different crops we used the general guidelines for crop irrigation set out by the United Nations Food and Agricultural Organization (Brouwer and Heibloem 1986) to determine the amount of water needed for a given crop based on its area at the site. The difference between the crop irrigation needs and the captured rainfall (see above section) over the growing period were taken as the estimated supplemental irrigation that the farmer would need to supply. Dividing the estimated supplemental irrigation for a single crop by the supplemental irrigation for all crops provided a means to allocate the recorded irrigation at the site to an individual crop. Table 2 outlines this method.

Table 112 - Crop specific irrigation demands for GB-NC1 operation

Crop	**% Growing Area**	**Site Growing Area (m^2)**	**Captured rainfall (m/m^2)**	**Water Demand (m/m^2)**	**Irrigation Demand (m/m^2)**	**Hypothetical Irrigation to Crop (m^3)**	**% Total Irrigation** [10]	**Allocated Irrigation to Crop (m^3)**
Tomato	20	89.6	0.28	0.8	0.52	47	26	31.6
Greens	2	9	0.28	0.5	0.22	2	1	1.3

Imported Compost Production

Method applied to BI-NC system was also used (see S1.2 for further details). Interviews with the farmer revealed that they intend to maintain nutrient levels purely through continued application of compost. They believed that they would be re-applying compost every fifth year. We have assumed that this application volume will be equal to 25% of the initial compost applied at the

[10] Total estimated irrigation demand of 178 m^3

Waste Management

According to the site operator, organic waste is handled on site and no inputs or outputs are attributed to this.

Nutrients in Runoff

Fertilization levels are so low at the site that their concentrations in runoff are assumed to be negligible. Moreover, they are captured by the combined sewer and collected downstream at the wastewater treatment plant.

GB-NC1: Component lifetimes and recycling rates

Same heuristics for the lifetimes and recycling rates as applied to the BI-NC were applied here, since they both operate within the Metropolitan Boston, U.S. region, and rely on the same waste management system. The assumed lifetime of the overall GB-NC1 operation is 30 years. Table 3 outlines the different component lifetimes and disposal rates in the GB-NC1 system.

Table 113 - Lifetime and recycling rate of different components in the GB-NC1 system

Component	Lifetime (years)	Current recycling rate (%)	Future recycling rate (%)						Applied rate (%)	Substitutes at market
			5	10	15	20	25	30		
Structural components										
Wood[11]	30	33	-	-	-	-	-	95	95	virgin metal
Concrete foundation	30	61	-	-	-	-	-	92	92[12]	gravel
Irrigation components										
Hose	10	33	-	95	-	95	-	95	95	virgin plastic
Tanks	30	33	-	-	-	-	-	95	95	virgin plastic
Other										
Wheelbarrow	10	33	-	50	-	50	-	50	50	virgin plastic
Distribution totes	10	33	-	95	-	95	-	95	95	virgin metal and plastic
Row cover	4	0	0	0	0	0	0	0	0	assumed to go to landfill

GB-NC1: Life Cycle Inventory (LCI) - Tomato

Capital inputs, were multiplied by the tomato allocation key (see table 1) and divided by their lifetimes (table 3) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 3). For operating inputs the allocation key in table 1 was applied. The allocated capital inputs, outputs (recycled and landfilled), and operating inputs were then normalized per kilogram fresh tomato produced by dividing by the annual production of 411 kg/annum. Table 4-10 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1.1 processes used to model them. Transport was generally ignored unless the distances were large (e.g. greenhouse components coming from California) or the freight's mass significant (e.g. concrete).

Structural components

Table 114 - LCI for BI-NC1 structural components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				

[11] Assumed to be incinerated.

[12] Massachusetts aims for 80% decrease in landfilling by from 2010 rate by 2050 (Government of Massachusetts 2015). The concrete recycling rate has been estimated assuming that this goal is achieved by the demolition date.

Land Occupation	0.272358487	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Concrete	4.07565E-05	m^3/FU	Concrete, normal {US-NPCC}| production | Conseq, U	
Wood	2.06701E-05	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Waste Treatment				
Concrete recycling	0.045668682	kg/FU	Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	
Landfilling	0.005301263	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.012985664	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

Irrigation System

Table 115 - LCI of GB-NC1 irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.000797621	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Rubber	0.001741149	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.002533907	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
LDPE recycling	0.001653605	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Rubber recycling	0.000753849	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Landfilling	0.000126939	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Ground Cover

Table 116 - LCI of GB-NC1 ground cover

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Polypropylene	0.006954869	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.006954869	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Landfilling	0.006954869	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Miscellaneous Capital

Table 117 - LCI of GB-NC1 miscellaneous capital

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.001167251	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Rubber	0.000375952	kg/FU	Synthetic rubber {GLO}| market for |	

			Conseq, U	
Steel	0.000676033	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Wood	5.69035E-08	kg/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Material Processing				
Plastic forming	0.001206159	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
HDPE recycling	0.001108888	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Rubber recycling	1.87733E-05	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Steel recycling	0.000338016	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.000452796	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation

Table 118 - LCI for GB-NC1 irrigation and avoided runoff for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.077744959	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.074986705	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands and Growing Medium

Table 119 - LCI for GB-NC1 nutrient consumption for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Wood Chips	0.184814688	kg/FU	Wood chips, wet, measured as dry mass {RoW}| market for | Conseq, U	
Transport				
Compost hauling	0.170224055	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	58 km between farm and composting center and 40 km between farm and wood chips

Distribution

Table 120 - LCI for GB-C1 distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.003792322	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	Plastic bags
Material Processing and Transport				
Plastic forming	0.003792322	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Distribution to	0.106395714	km	Transport, passenger car, large size,	Driven in van to

markets			petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	various markets around the Boston area
Waste Treatment				
LDPE recycling	0.001896161	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.001896161	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

GB-NC1: Life Cycle Inventory (LCI) - Arugula

Capital inputs, were multiplied by the tomato allocation key (see table 1) and divided by their lifetimes (table 3) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 3). For operating inputs the allocation key in table 1 was applied. The allocated capital inputs, outputs (recycled and landfilled), and operating inputs were then normalized per kilogram fresh arugula produced by dividing by the annual production of 7 kg/annum. Table 11-17 outline the LCI for a single kilogram of lettuce (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1.1 processes used to model them. Transport was generally ignored unless the distances were large (e.g. greenhouse components coming from California) or the freight's mass significant (e.g. concrete).

Structural components

Table 121 - LCI for BI-NC1 structural components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				
Land Occupation	1.6426667	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Concrete	0.0002458	m^3/FU	Concrete, normal {US-NPCC}| production | Conseq, U	
Wood	0.0001247	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Waste Treatment				
Concrete recycling	0.27544	kg/FU	Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	
Landfilling	0.0319733	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.07832	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

Irrigation System

Table 122 - LCI of GB-NC1 irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.0048107	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Rubber	0.0105013	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0152827	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
LDPE recycling	0.0099733	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Rubber recycling	0.0045467	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	

Landfilling	0.0007656	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Ground Cover

Table 123 - LCI of GB-NC1 ground cover

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Polypropylene	0.0419467	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0419467	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Landfilling	0.0419467	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Miscellaneous Capital

Table 124 - LCI of GB-NC1 miscellaneous capital

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.00704	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	
Rubber	0.0022675	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
Steel	0.0040773	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Wood	3.432E-07	kg/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Material Processing				
Plastic forming	0.0072747	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
HDPE recycling	0.006688	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Rubber recycling	0.0001132	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Steel recycling	0.0020387	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.0027309	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation

Table 125 - LCI for GB-NC1 irrigation and avoided runoff for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.1998088	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.452265	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands and Growing Medium

Table 126 - LCI for GB-NC1 nutrient consumption for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Wood Chips	1.1146667	kg/FU	Wood chips, wet, measured as dry mass {RoW}| market for | Conseq, U	
Transport				
Compost hauling	1.0266667	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	58 km between farm and composting center and 40 km between farm and wood chips

Distribution

Table 127 - LCI for GB-C1 distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.003792322	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	Plastic bags
Material Processing and Transport				
Plastic forming	0.003792322	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Distribution to markets	0.106395714	km	Transport, passenger car, large size, petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	Driven in van to various markets around the Boston area
Waste Treatment				
LDPE recycling	0.001896161	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	0.001896161	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

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Appendix F: Life cycle inventory for ground-based non-conditioned farming system 1(GB-NC2)

The ground-based non-conditioned UA site (referred to as 'GB-NC2' in the article text) is an operating urban farm in Brooklyn, New York City, US. The farm consists of a 2081 m^2 lot surrounded by residential and commercial uses. The total growing space on the site is 1285 m^2; 693 m^2 is operated by the farm itself with the remaining 592 m^2 used as community growing space. This assessment is concerned with the growing space operated by the farm organization itself. The farm grows around 60 varieties of fruits, vegetables and herbs during any given growing season. Water is supplied primarily through rainfall and is supplemented by municipal water during dry periods using a single hose. Artificial fertilizer and compost provide nutrients at the farm.

Primary data was available for all capital and operating inputs. Capital was accounted through site visits. Operating inputs were taken from interviews with the farm operator and utilities invoices. Because multiple crops are grown throughout the year the various inputs had to be allocated to the tomatoes and lettuce. Table 1 outlines the allocation methods.

Table 128 - Allocation key for GB-NC2

Aspect	Allocation Method	Tomato	Lettuce
Equipment, site visits, compost inputs, land occupation, runoff mitigation	% of growing area occupied	0.04	0.02
Irrigation	Crop specific water needs	0.06	0.01
Distribution, waste management	% mass produced	0.04	0.004

The supplementary information is arranged as follow: description of estimation of capital inputs where primary data was lacking; description of operational inputs where primary data was lacking; component lifetimes and recycling rates; life cycle inventory for functional unit.

GB-C1: Capital Inputs

Primary data was available for all equipment.

GB-C1: Operating Characteristics and Inputs

The operating inputs for the GB-C1 farm were taken from primary data provided by the farm operator, though some aspects had to be estimated, namely, space conditioning to grow seedlings, runoff retention and imported compost production. In general, the same electricity mixes were used as employed for the BI-NC operation (see S1.2 for details). No nutrient runoff is expected since all of the fertilization occurs within the greenhouse.

Seedling Production

Production of seedlings (small plants grown in greenhouses that are transferred to outdoor soil) is based off of the methods of Stoessel at al. (2012) for lettuce and tomatoes. For details see S1.2 of the BI-NC system.

Runoff Retention

Runoff retention was calculated in the same manner as GB-NC1 (see S1.5 for details). Total precipitation over the site was 1.29 m during the 2014 growing season (U.S. Climate Data 2015). From this the amount of runoff estimated to be subsumed by the farm was 1349 m^3. Considering that similar built up lots in the neighborhood are on average covered by 89% impermeable area (see figure 1), and that 61% of the farms area is run by the farm organization, the total amount of avoided runoff attributed to the farm operation is 739 m^3 per annum. Because the farm is located in an area with combined sewers (NYC Environmental Protection 2015), the increased rainwater capture was modeled as avoided wastewater treatment in accordance with the consequential LCA modelling employed here.

Figure 12 - GB-NC2 site with surroundings shown. Brown blocks represent buildings, while green forms are other impermeable surfaces. On average a built up plot in the area would be covered with 89% impermeable surfaces.

Irrigation

Primary data on irrigation was not available at the site. Irrigation from municipal water were estimated using the same method for the BI-NC operation (see S1.2 for further information); total irrigation demands for a crop were estimated using the crops growing space and rules of thumb for water demands, and then the captured rainwater was subtracted from this to estimate the amount of municipal water that would be required to meet this. Total municipal water demand was taken as the sum of this for all crops, with the allocation key in table 1 used to re-allocate to the individual crops. Table 2 outlines this method for the tomatoes and lettuce.

Table 129 - Crop specific irrigation demands for GB-NC1 operation

Crop	% Growing Area	Site Growing Area (m^2)	Captured rainfall (m/m^2)	Water Demand (m/m^2)	Irrigation Demand (m/m^2)	Hypothetical Irrigation to Crop (m^3)	% Total Irrigation[13]
Tomato	4	68.3	0.40	0.8	0.4	27.2	6
Lettuce	2	34.2	0.40	0.5	0.1	3.4	1

Imported Compost Production

Method applied to BI-NC system was also used (see S1.2 for further details).

Waste Management

The GB-NC1 operation produces compost on site, but this is not included in the inventory since the method is very low input (passively aerated compost piles gravity fed through plastic bins). The resulting compost does not substitute for fertilizer purchases, so the system is not credited for such, though the avoided waste sent to landfills is accounted.

[13] Total estimated irrigation demand of 472 m^3

Nutrients in Runoff

Though some of the nutrients from the applied fertilizer are lost in runoff from the site, they are collected by the combined sewer system and captured at the wastewater treatment plant. The concentration of nutrients in the runoff is very low ($8*10^{-4}$ g N/L, $2*10^{-4}$ g P/L, $4*10^{-3}$ g/L according to Emilsson et al.'s method (2007) – see S1.2 for more details) that is not assumed to not affect the normal operation of the receiving wastewater treatment plant.

Refrigeration

Crops are refrigerated overnight prior to delivery to the market. Energy inputs for refrigeration were taken from the Stoessel et al. (2012).

Table 130 - Energy inputs for refrigeration during distribution

Product	Amount	Unit
Tomato	$7.53*10^{-3}$	kWh/day
Lettuce	10^{-3}	kWh/day

GB-NC1: Component lifetimes and recycling rates

Same heuristics for the lifetimes and recycling rates as applied to the GB-NC2 were applied here, since they operate on the Eastern Seaboard of the U.S. The assumed lifetime of the overall GB-NC2 operation is 30 years. Table 3 outlines the different component lifetimes and disposal rates in the GB-NC2 system.

Table 131 - Lifetime and recycling rate of different components in the GB-NC2 system

Component	Lifetime (years)	Current recycling rate (%)	Future recycling rate (%)						Applied rate (%)	Substitutes at market
			5	10	15	20	25	30		
Structural components										
Wood[14] - Hothouse	30	33	-	-	-	-	-	95	95	incinerated
Steel – Hothouse	30	33	-	-	-	-	-	95	95	virgin metal
Skin - Hothouse	15	33	-	-	64	-	-	95	80	virgin plastic
Concrete foundation – Hothouse	30	61	-	-	-	-	-	92	92[15]	gravel
Concrete foundation - Fence	30	61	-	-	-	-	-	92	92	gravel
Steel - Fence	30	33	-	-	-	-	-	95	95	virgin steel
Raised beds	30	33	-	-	-	-	-	95	95	incinerated
Irrigation components										
Tubing	30	33	-	-	-	-	-	95	95	virgin plastic
Wood	30	33	-	-	-	-	-	95	95	incinerated
Tanks	30	33	-	-	-	-	-	95	95	virgin plastic
Other										
Wheelbarrow	10	33	-	50	-	50	-	50	50	virgin plastic
Distribution totes	10	33	-	95	-	95	-	95	95	virgin metal and plastic
Row cover	4	0	0	0	0	0	0	0	0	assumed to be landfilled

[14] Assumed to be incinerated.

[15] Assumed that same as concrete in Massachusetts

GB-NC1: Life Cycle Inventory (LCI) - Tomato

Capital inputs, were multiplied by the tomato allocation key (see table 1) and divided by their lifetimes (table 3) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 3). For operating inputs the allocation key in table 1 was applied. The allocated capital inputs, outputs (recycled and landfilled), and operating inputs were then normalized per kilogram fresh tomato produced by dividing by the annual production of 184 kg/annum. Table 5-13 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1 processes used to model them. Transport was generally ignored unless the distances were large (e.g. greenhouse components coming from California) or the freight's mass significant (e.g. concrete).

Structural components

Table 132 - LCI for BI-NC2 structural components

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				
Land Occupation	0.2353414	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Concrete	2.07774E-06	m^3/FU	Concrete, normal {US-NPCC}| production | Conseq, U	
Wood	1.37046E-05	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Steel	0.004165614	kg/FU	Steel, low-alloyed {GLO}| market for | Conseq, U	
LDPE	0.000131688	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.000131688	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
Concrete recycling	0.004650721	kg/FU	Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	
LDPE recycling	0.000105575	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Steel recycling	0.00395739	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.001022418	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.008600407	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

Irrigation System

Table 133 - LCI of GB-NC2 irrigation system

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.00273133	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Wood	1.34273E-06	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
PVC	0.002168384	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Material Processing				

Tube forming	0.004607817	kg/FU	Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	Local production assumed
Tank forming	0.000287728	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
LDPE recycling	0.002606231	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.00205788	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Wood waste treatment	0.000842334	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	
Landfilling	0.000289813	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Row Cover

Table 134 - LCI of GB-NC2 row cover

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Polypropylene	0.003502775	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.003502775	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Landfilling	0.003502775	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Miscellaneous Capital

Table 135 - LCI of GB-NC2 miscellaneous capital

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Rubber	9.67433E-05	kg/FU	Synthetic rubber {GLO}| market for | Conseq, U	
Steel	0.001740962	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Wood	2.00159E-07	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Material Processing				
Plastic forming	9.67433E-05	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Rubber recycling	4.83716E-05	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Steel recycling	0.000869439	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.001042492	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation

Table 136 - LCI for GB-NC2 irrigation and avoided runoff for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes

Material Inputs				
Irrigation water	0.153269665	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.151683389	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands and Growing Medium

Table 137 - LCI for GB-NC2 nutrient consumption for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Wood Chips	0.552638883	kg/FU	Wood chips, wet, measured as dry mass {RoW}| market for | Conseq, U	
Nitrogen fertilizer	0.000763897	kg/FU	Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	
Phosphate fertilizer	0.000306028	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium fertilizer	0.000936978	kg/FU	Potassium sulfate, as K2O {GLO}| market for | Conseq, U	
Transport				
Compost and wood chips hauling	0.002105881	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	Wood chips and compost assumed to be delivered 40 km

Waste Management

Table 138 - LCI for GB-NC2 waste management for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Avoided Production				
Landfilling	0.018772263	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	Some solid waste is produced but this is offset by composting

Energy and Fuels

Table 139 - LCI for GB-NC2 waste management for tomatoes

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Transport and Energy				
Gasoline	0.000328466	kg/FU	Petrol, unleaded {RoW}| market for | Conseq, U	For on farm equipment
Transport	0.131687881	km/FU	Transport, passenger car, large size, petrol, EURO 4 {GLO}| market for transport, passenger car, large size, petol, EURO 4 | Conseq, U	Weekly trips to garden center on Staten Island, NY in van
Outputs to Environment				
Carbon dioxide (fossil)	0.00111173	kg/FU	Carbon dioxide (fossil, to air)	Combustion of fuel in farm equipment

Distribution

Table 140 - LCI for GB-C1 distribution

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
HDPE	0.000700103	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Distribution totes
Material Processing, Transport and Energy				
Plastic forming	0.000700103	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Distribution to markets	0.02468161	km	Transport, passenger car, large size, petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	Driven in van to market approximately 1 km from site
Cooling	1	kg/FU	ENYF, tomato cooling	
Waste Treatment				
HDPE recycling	0.000665098	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	3.50052E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

GB-NC1: Life Cycle Inventory (LCI) - Lettuce

Capital inputs, were multiplied by the tomato allocation key (see table 1) and divided by their lifetimes (table 3) to determine the annual inputs of materials to the system. Annual outputs were also estimated, with the fractions going to recycling and landfill calculated using the recycling rates (table 3). For operating inputs the allocation key in table 1 was applied. The allocated capital inputs, outputs (recycled and landfilled), and operating inputs were then normalized per kilogram fresh lettuce produced by dividing by the annual production of 22 kg/annum. Table 14-22 outline the LCI for a single kilogram of tomatoes (abbreviated as 'FU' for functional unit) delivered to the point of pickup by the customer, including the ecoinvent 3.1 processes used to model them. Transport was generally ignored unless the distances were large (e.g. greenhouse components coming from California) or the freight's mass significant (e.g. concrete).

Structural components

Table 141 - LCI for BI-NC2 structural components for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Inputs From Nature				
Land Occupation	1.243395324	m^2*a/FU	Occupation, urban, continuously built	
Material Inputs				
Concrete	1.09775E-05	m^3/FU	Concrete, normal {US-NPCC}| production | Conseq, U	
Wood	7.24067E-05	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Steel	0.022008478	kg/FU	Steel, low-alloyed {GLO}| market for | Conseq, U	
LDPE	0.000695756	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.000695756	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
Concrete recycling	0.024571475	kg/FU	Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	
LDPE recycling	0.000557794	kg/FU	PE (waste treatment) {US-NPCC}| recycling of	

			PE | Conseq, U	
Steel recycling	0.020908351	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.005401812	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	
Wood disposal	0.045439124	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	

Irrigation System

Table 142 - LCI of GB-NC2 irrigation system for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
LDPE	0.014430626	kg/FU	Polyethylene, low density, granulate {GLO}| market for | Conseq, U	
Wood	7.09414E-06	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
PVC	0.011456375	kg/FU	Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	
Material Processing				
Tube forming	0.024344797	kg/FU	Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	Local production assumed
Tank forming	0.001520173	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Waste Treatment				
LDPE recycling	0.013769682	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
PVC recycling	0.010872541	kg/FU	PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	
Wood waste treatment	0.004450361	kg/FU	Waste wood, untreated {US-NPCC}| market for | Conseq, U	
Landfilling	0.001531189	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Row Cover

Table 143 - LCI of GB-NC2 row cover for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Polypropylene	0.018506452	kg/FU	Polypropylene, granulate {GLO}| market for | Conseq, U	
Material Processing				
Plastic forming	0.018506452	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Landfilling	0.018506452	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Miscellaneous Capital

Table 144 - LCI of GB-NC2 miscellaneous capital for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Rubber	0.000511131	kg/FU	Synthetic rubber {GLO}| market for |	

			Conseq, U	
Steel	0.009198147	kg/FU	Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	
Wood	1.05751E-06	m^3/FU	Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	
Material Processing				
Plastic forming	0.000511131	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	Local production assumed
Waste Treatment				
Rubber recycling	0.000255565	kg/FU	Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	
Steel recycling	0.004593566	kg/FU	Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	
Landfilling	0.005507873	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

Irrigation

Table 145 - LCI for GB-NC2 irrigation and avoided runoff for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Irrigation water	0.29977701	m^3/FU	Tap water {US}| market for | Conseq, U	Ground water changed to U.S. location
Avoided Production				
Wastewater treatment	0.801399371	m^3/FU	Wastewater, unpolluted {GLO}| market for | Conseq, U	

Nutrient Demands and Growing Medium

Table 146 - LCI for GB-NC2 nutrient consumption for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Material Inputs				
Wood Chips	2.919795343	kg/FU	Wood chips, wet, measured as dry mass {RoW}| market for | Conseq, U	
Nitrogen fertilizer	0.000262045	kg/FU	Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	
Phosphate fertilizer	0.000306028	kg/FU	Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	
Potassium fertilizer	0.000936978	kg/FU	Potassium sulfate, as K2O {GLO}| market for | Conseq, U	
Transport				
Compost and wood chips hauling	0.011126145	tkm/FU	Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	Wood chips and compost assumed to be delivered 40 km

Waste Management

Table 147 - LCI for GB-NC2 waste management for lettuce

Input/Process	Amount	Unit	ecoinvent 3.1 process	Notes
Avoided Production				

Landfilling	0.016074258	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	Some solid waste is produced but this is offset by composting

Energy and Fuels

Table 148 - LCI for GB-NC2 waste management for lettuce

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Transport and Energy				
Gasoline	0.001735405	kg/FU	Petrol, unleaded {RoW}| market for | Conseq, U	For on farm equipment
Transport	0.695755713	km/FU	Transport, passenger car, large size, petrol, EURO 4 {GLO}| market for transport, passenger car, large size, petol, EURO 4 | Conseq, U	Weekly trips to garden center on Staten Island, NY in van
Outputs to Environment				
Carbon dioxide (fossil)	0.005873679	kg/FU	Carbon dioxide (fossil, to air)	Combustion of fuel in farm equipment

Distribution

Table 149 - LCI for GB-C1 distribution for lettuce

Input/Process	**Amount**	**Unit**	**ecoinvent 3.1 process**	**Notes**
Material Inputs				
HDPE	0.000599482	kg/FU	Polyethylene, high density, granulate {GLO}| market for | Conseq, U	Distribution totes
Material Processing, Transport and Energy				
Plastic forming	0.021134297	kg/FU	Extrusion, plastic film {US-NPCC}| production | Conseq, U	
Distribution to markets	0.02468161	km	Transport, passenger car, large size, petrol, EURO 5 {RER}| transport, passenger car, large size, petrol, EURO 5 | Conseq, U	Driven in van to market approximately 1 km from site
Cooling	1	kg/FU	ENYF, lettuce cooling	
Waste Treatment				
HDPE recycling	0.000569508	kg/FU	PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	
Landfilling	2.99741E-05	kg/FU	Inert waste, for final disposal {US}| market for | Conseq, U	

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Appendix G: Carbon sequestration and solar energy calculations

Carbon sequestration

First need to determine the amount of land made available if UA can substitute for farmland. This means determining how much land would need to be occupied for solar energy and how much farmland would be freed.

(1) Net land change = farmland freed – land occupied for solar panels to supply UA energy

The yield for both field tomatoes and lettuce in Massachusetts are 1.5 kg/m^2/annum (USDA 2013). Using this we can see how much farmland would be freed by the BI-C and GB-C2 producing tomatoes and lettuce, respectively.

Farm	UA yield (kg/a)	Conventional Yield (kg/m^2)	Conventional land with equivalent capacity (m^2)
BI-C	244496	1.404305038	174104.6
GB-C2	53.6	1.49979778	1400.2

The amount of land necessary for the solar array to power the farm can be made using the following assumptions: Massachusetts receives on average 4 kWh/m^2/day solar radiation (National Renewable Energy Laboratory 2015), which if converted to electricity at 15% efficiency (reasonable for monocrystalline silicon panels) supplies 219 kWh/m^2/year. Assuming 25% extra space is required for access to the panels, then we find that each square meter of solar array produces 175.2 kWh/year. Using per unit area energy demands of the farms and the total areas of the farms the size of the required solar array can be estimated.

Farm	Unit energy requirements (kWh/m^2/year)	Farm Area (m^2)	Supporting solar array area (m^2)
BI-C	305	3492.8	6086.8
GB-C2	967	30	165.5

We are now in a position to calculate the carbon sequestrated if we allow the farmland to return to forest. We assume a sequestration rate of 0.95 kg CO_2 eq./year/m^2 (Schmidinger and Stehfest 2012) and combine with the net land use change to estimate the carbon capture.

Farm	Farmland to forest (m^2)	Land occupation for solar (m^2)	Net land change (m^2)	Annual carbon sequestered (kg CO_2 eq./m^2)
BI-C	174104.6	6086.8	168017.9	159617.0
GB-C2	1607.273	165.5	1267.8	1172.9

To get the final total the savings from the substituted produce are added to the carbon sequestration totals. Each kg BI-C tomato from solar power results in a 0.11 kg CO_2 eq reduction. Each kg GB-C2 from solar power results in a -0.72 kg CO_2 eq reduction (an increase).

Farm	kg CO_2 reduced per kg conventional replaced	Yield (kg/annum)	Total CO_2 savings (kg CO_2 eq.)
BI-C	0.11	244496	27778.54507
GB-C2	-0.72	1607.273	-1161.524326

Total savings are 187 tonnes CO_2 eq. for the BI-C and 11.4 kg CO_2 eq. for the GB-C2.

GHG savings for various land uses

Savings for tomatoes (negative indicates net increase in GHGs)

Farm	kg CO_2 eq/kg tomato	Yields (kg/m^2)	kg CO_2 avoided/kg tomato replaced	kg avoided CO^2/m^2 UA	kg avoided CO^2/m^2 solar
BI-C (NPCC)	2.1472301	70	-1.55604409	-108.9230863	68.328
BI-NC	0.12811897	16.3	0.46306704	7.547992752	68.328
GB-C1	1.5804366	9.8	-0.98925059	-9.694655782	68.328
GB-NC1	0.075838218	4.4	0.515347792	2.267530285	68.328
GB-NC2	0.13540287	6.9	0.45578314	3.144903666	68.328
BI-C (wind)	0.32284504	70	0.26834097	18.7838679	68.328
BI-C (solar)	0.47757047	70	0.11361554	7.9530878	68.328
BI-C (hydro)	0.24055064	70	0.35063537	24.5444759	68.328
Conventional	0.591	-	-	-	-

Savings for lettuce (negative indicates net increase in GHGs)

Farm	kg CO_2 eq/kg lettuce	Yields (kg/m^2)	kg CO_2 avoided/kg lettuce replaced	kg avoided CO^2/m^2 UA	kg avoided CO^2/m^2 solar
GB-C2 (NPCC)	8.6549462	70	-7.73003145	-541.1022015	68.328
BI-NC	0.400418808	16.3	0.524495942	8.549283851	68.328
GB-C1	26.50507659	9.8	-25.58016184	-250.685586	68.328
GB-NC1	0.232652097	4.4	0.692262653	3.045955672	68.328
GB-NC2	0.440830366	6.9	0.484084384	3.340182249	68.328
GB-C2 (wind)	0.90314289	70	0.02177186	1.5240302	68.328
GB-C2 (solar)	1.6475826	70	-0.72266785	-50.5867495	68.328
GB-C2 (hydro)	0.51505105	70	0.4098637	28.690459	68.328
Conventional	0.925	-	-	-	-

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1. Raw Results
    1.1. Tomato

Indicator	**BI-C**	**BI-NC**	**GB-C1**	**GB-NC1**	**GB-NC2**	**ecoinvent**
Climate Change (CO_2 eq/kg tomato)	0.34	0.13	1.58	0.08	0.14	0.47
Freshwater Ecotoxicity (CTU/kg tomato)	0.7	1.4	0.9	0.1	0.2	7.8
Marine Eutrophication (kg N eq./kg tomato)	0.00036	0.00043	0.00063	$8.9*10^{-5}$	0.00015	0.00141

Land Use (kg C deficit/kg tomato)	2.1	1.2	2.4	5.1	6.8	2.4
Mineral, fossil and ren resource depletion (kg S eq./kg tomato)	$7.2*10^{-6}$	$1.1*10^{-4}$	$3.1*10^{-5}$	$1.3*10^{-5}$	$1.9*10^{-5}$	$9.8*10^{-6}$
Water resource depletion (m^3 water eq./kg tomato)	0.0028	0.068	0.066	0.05	0.074	0.0025
Water scarcity index (m^3 water eq./kg tomato)[16]	0.011	0.089	0.087	0.128	0.15	0.0022

1.2. Lettuce

Indicator	**BI-NC**	**GB-C1**	**GB-C2**	**GB-NC1**	**GB-NC2**	**ecoinvent**
Climate Change (CO_2 eq/kg tomato)	0.40	26.5	8.7	0.23	0.44	0.23
Freshwater Ecotoxicity (CTU/kg tomato)	4.6	8.9	6.1	0.4	0.9	0.4
Marine Eutrophication (kg N eq./kg tomato)	0.0014	0.0079	0.0038	$3.9*10^{-4}$	$5.2*10^{-4}$	0.0016
Land Use (kg C deficit/kg tomato)	4.2	30.3	8.8	30.4	35.5	1.53
Mineral, fossil and ren resource depletion (kg S eq./kg tomato)	$3.7*10^{-4}$	$2.3*10^{-4}$	$4.3*10^{-5}$	$3.6*10^{-5}$	$6.8*10^{-5}$	$1.39*10^{-5}$
Water resource depletion (m^3 water eq./kg tomato)	0.195	0.18	0.018	0.17	0.22	0.010
Water scarcity index (m^3 water eq./kg tomato)[1]	0.26	0.26	0.029	0.47	0.57	0.010

2. Tomato Results

Results from previous LCAs of tomatoes as point of comparison.

Study	**Climate Change (CO_2 eq/kg tomato)**	**Water Scarcity (m^3 water eq./kg tomato)**	**Notes**
Page et al. (2011)	-	0.002-0.024	Greenhouse and field tomatoes in the Sydney, AU region
Cellura et al. (2012)	0.74	-	Italian greenhouse production
Torrellas et al. (2012)	0.25	-	Mediterranean greenhouse tomatoes
Jones at al. (2012)	0.06-0.75	-	Open field tomato production in Florida, US
Page et al. (2012)	0.39-1.97	0.005-0.053	Greenhouse and field tomatoes in the Sydney, AU region
Sanye-Mengual et al. (2012)	0.26	-	UA rooftop greenhouse in Barcelona, ES*
Sanye-Mengual et al. (2012)	0.7	-	Greenhouse tomato in Spain*
Bojaca et al. (2014)	0.074	-	Columbian greenhouse tomatoes
Manfredi and Vignali (2014)	0.181	0.104	Italian field tomatoes
Sanye-Mengual et al. (2014)	0.178-0.297	-	Rooftop greenhouse in Barcelona, ES
Theurl et al. (2014)	0.109	-	Organic tunnel in Austria*
Theurl et al.	0.609	-	Conventional multi-tunnel in

[16] Pfister et al. (2009) method also checked here to compare with other studies

(2014)			Spain*
Theurl et al. (2014)	0.281	-	Conventional open field in Italy*
Theurl et al. (2014)	1.296	-	Greenhouse in Austria*
Sanye-Mengual et al. (2015a)	0.708	-	Rooftop greenhouse in Barcelona, ES*
Sanye-Mengual et al. (2015a)	1.54	-	Conventional multi-tunnel in Spain*
Sanye-Mengual et al. (2015b)	0.0679-0.0753	0.0881-0.0980	Rooftop hydroponic (BI-NC) in Bologna, IT
He at al. (2016)	0.207	0.060	Chinese organic production
He at al. (2016)	0.261	0.059	Chinese conventional production

* includes distribution

3. Lettuce Results

Results from previous LCAs of lettuce as point of comparison.

Study	Climate Change (CO_2 eq/kg lettuce)	Water Scarcity (m^3 water eq./kg lettuce)	Notes
Hospido et al. (2009)	0.33	-	British field production*
Hospido et al. (2009)	0.45	-	Spanish field production*
Hospido et al. (2009)	0.24	-	British heated greenhouse*
Shiina et al. (2011)	6.4	-	Japanese plant factory
Gunady et al. (2012)	3.75	-	Field lettuce in western Australia
Maraseni et al. (2012)	0.17-0.22	-	Field lettuce in eastern Australia
Plawecki et al. (2014)	0.198	-	Unheated Michigan, US greenhouse*
Plawecki et al. (2014)	0.857	-	California, UA field lettuce*
Romero-Gamez et al. (2014)	0.025-0.214	-	Open field and unheated greenhouse in Spain
Hall et al. (2014)	0.08-0.32	-	Field production in Sydney, AU region
Bartzas et al. (2015)	0.205-0.243	-	Open field and unheated greenhouses in Spain and Italy
Foteinis and Chatzisfymeon (2016)	1.282	-	Organic field production in Greece
Foteinis and Chatzisfymeon (2016)	0.631	-	Conventional field production in Greece
Rothwell et al. (2015)	0.2-0.9	0.02-0.1	Field production in Sydney, AU region*
Sanye-Mengual (2015b)	0.567-1.08	0.0395-0.0904	Floating hydroponic BI-NC in Bologna, IT
Sanye-Mengual (2015b)	2.51-4.88	0.0855-0.196	Nutrient film hydroponic BI-NC in Bologna, IT
Sanye-Mengual (2015b)	0.323	0.389	Soil BI-NC in Bologna, IT

* includes distribution

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Appendix D

Article 4: Contributions of local agriculture to urban sustainability in the Northeast United States

Contributions of local farming to urban sustainability in the Northeast United States

Benjamin P. Goldstein†, Michael Z. Hauschild†, John E. Fernández‡, Morten Birkved†*
†Technical University of Denmark, Quantitative Sustainability Assessment Division, Produktionstorvet, Building 424, Kongens Lyngby, 2800, Denmark
‡Massachusetts Institute of Technology, Department of Architecture, 77 Massachusetts Avenue, 5-419, Cambridge, 02139, United States
*Corresponding author: Phone: +45 45254561; fax: +45 45933435; e-mail: bgol@dtu.dk

Abstract

Food consumption is an important contributor to a city's environmental impacts (carbon emissions, land occupation, water use, etc.) Urban farming (UF) has been advocated as a means to increase urban sustainability by reducing food-related transport and tapping into local resources. Taking Boston as an illustrative Northeast US city, we developed a novel method to estimate sub-urban, food-borne carbon and land footprints using multi-region-input-output modeling and nutritional surveys. Computer simulations utilizing primary data explored UF's ability to reduce these footprints using select farming technologies, building on previous city-scale UF assessments which have hitherto been dependent on proxy data for UF. We found that UF generated meagre food-related carbon footprint reductions (1.1-2.9% of baseline 2211 kg CO_2 equivalents/capita/annum) and land occupation increases (<1% of baseline 9000 m^2 land occupation/capita/annum) under optimal production scenarios, informing future evidence-based urban design in the region. Notwithstanding UF's marginal environmental gains, UF could help Boston meet national nutritional guidelines for vegetable intake, generate an estimated $160 million US in revenue to growers and act as a pedagogical and community building tool, though these benefits would hinge on large-scale UF proliferation, likely undergirded by environmental remediation of marginal lands in the city.

Introduction

Food consumption is a key driver of a city's environmental burdens[1,2], and in the United States (US) per capita impacts are amongst the highest globally[3,4]. Many cities in the Northeast US are promoting urban farming (UF) – food production within the city, allowing for material and energy exchange between city and farm[5] – as a joint environmental and social sustainability exercise[6,7]. Up to 20% of global food supply already comes from within cities, primarily in the Global South[8,9], but the potential in the Global North's cities to produce their own food on the ground and buildings is believed to be substantial[10–12]. Hypothetical assessments of UF at the city-scale have demonstrated reduced food related GHG emissions[13,14] and land occupation[14], giving the impression that pro-UF policies can contribute to more sustainable urban food supply networks. Despite UF's perceived environmental benefits, the recent spurt of pro-UF actions by the cities of the Northeast US that include codification in land use laws[7,15] and multi-city commitments to increased local food production[6] require deeper reflection about their systemic environmental implications.

UF advocates tend to focus on the distance from farm to fork, equating local food with environmentally sustainable food[16,17], oversimplifying the complexity of food sustainability to a single aspect. Reducing distribution burdens and wastage by co-locating food production and consumption can lead to environmentally leaner production networks[18,19], but contrasting results have been found when large energy inputs are needed for space heating and lighting[20,21]. UF in the Northeast US has demonstrated lower embodied greenhouse gas (GHG) emissions compared to conventional supply networks in some instances, but with tradeoffs in other indicators (land occupation, water scarcity) and potentially significant burdens from farm capital[21]. UF studies at neighborhood and city scales have estimated reductions in food-borne GHG emissions[13,14] and land occupation[14], although these findings are not transferable to the US Northeast due to climatic differences. The use of data for conventional agricultural production (minus transport and wastage) as a proxy for UF production due to data gaps[13,14] biased the assessments in UF's favor.

This article provides a level of analysis that has been absent in previous UF sustainability work. We used primary data from multiple urban farms in the US Northeast to evaluate the environmental tradeoffs of substituting UF for conventional produce at the city-scale in this region (assessing strictly horticultural products), including interactions with the host city's material and energy systems. Multi-region input-output based environmental life cycle assessment (MRIO-LCA) was combined with nutritional surveys to model baseline food-borne environmental burdens at sub-urban granularity, in contrast to earlier work that has assumed equivalency between per capita city and national food intakes. Potential nutritional and economic benefits of UF were also considered.

Boston, US was used as a representative case city for the Northeast US. Though denser than many cities in the region, Boston's monocentric layout typifies most Northeast US cities, particularly in the densely populated Northeast Megalopolis. Importantly, Boston's climate mirrors that of the Northeast US, with an outdoor growing season roughly from April through October, and cold winters necessitating indoor growing reliant on external heating from the region's predominantly fossil-fuel driven grid.

Methods

Two overarching tasks were performed here: estimating baseline environmental impacts from Boston's food demands and modeling UF in Boston.

Baseline environmental performance

EXIOBASE v2.3 MRIO model was applied to estimate Boston's food related environmental burdens in 2010. EXIOBASE is a trade-linked model accounting for global economic activity in 2007, detailing domestic production, bilateral trade and final consumption of 43 regions accounting for ~90% of global GDP[22]. MRIO-LCA has been described in detail elsewhere[23,24], but the method's core are environmental extensions coupling production activities to resource and pollution intensities per unit economic output, facilitating the allocation of environmental impacts and resource draws to end consumers. Such top-down analysis is suitable for consumption based environmental accounting of large systems, having been applied at the national[25–27] and urban scales[28,29]. We chose EXIOBASE due to the high level of disaggregation (200 products), including pertinent food items.

The assessed indicators were land use and global warming potential (GWP). Land use accounts for crop, pasture and forest land occupation in m^2. The GWP extension includes CO_2, CH_4, N_2O and SF_6 emissions, employing IPCC 2013 methodology to convert emissions to the radiative forcing in equivalent mass CO_2 over a 100 year time horizon (kg CO_2e).

Input-output models take the product of national footprint multipliers (e.g. kg CO_2e/$ final demand product) and final consumption ($ final demand for product) to estimate demand-side footprints, insinuating that doubling food expenditures doubles food consumption and environmental stress. Whilst there is a correlation between income and food related environmental burdens at the national scale, it appears to follow a logarithmic trend, hinting at an income level beyond which food intake and environmental impacts plateau[1,3]. For a wealthy nation such as the US with a low Engel's ratio[30] (food expenditures as a percentage of total income), assuming a linear relationship between food expenditures and consumption is erroneous. US nutritional surveys show slight differences between the food consumption of high and low income residents, most notably for environmentally intensive foods (less than 10% difference between the groups for per capita meat and dairy intake by mass)[31], despite markedly elevated food expenditures by wealthy Americans[32]. Lastly, the higher prevalence of obesity in poorer Americans highlights the incongruences between food expenditures and intake[33].

We circumnavigated this challenge using a top down approach, ascribing total GWP and land use from US final food consumption to total available calories in the US, akin to Jones and colleagues[28,29]. Using a concordance matrix linking calorific availability for over 200 foods with products in the EXIOBASE model (e.g. calories of grains with the EXIOBASE product 'Cereals'), embodied environmental intensity per calorie was estimated. Total calories available for different foods were taken as the product of the 2007 US population and average US calorific intake for the years 2007-2010 from the Center for Disease Control's National Health and Nutrition Examination Survey (NHANES)[34] corrected for supply chain losses using US Department of Agriculture (USDA) loss adjusted availability data[35]. Tables S1 to S24 in the supplementary information document this process.

Intakes of foods for US demographics based on sex and age group were taken from the NHANES data for 2007-2010. Sex and age were chosen to develop population sub-groups as these are both strong determinants of food volume consumed (males tend to eat more than females at most ages) and dietary habits (e.g. dark green vegetable intake is nearly zero before age 14 and then proceeds parabolically with age)[34]. This sub-grouping also allowed for more nuanced modelling than the low/high income binary afforded by the publically available NHANES data relating income to food intake[31]. Demographics data for Boston at the block-group level (geographies of population 600-3000) were taken from the 2010 US Census. Combining census data, calorific intakes for different demographics and embodied GWP and land use per calorie delivered, food-borne environmental burdens for 560 block-groups in Boston were calculated. Figure 1 outlines this workflow while the supporting information details the data manipulation and calculations.

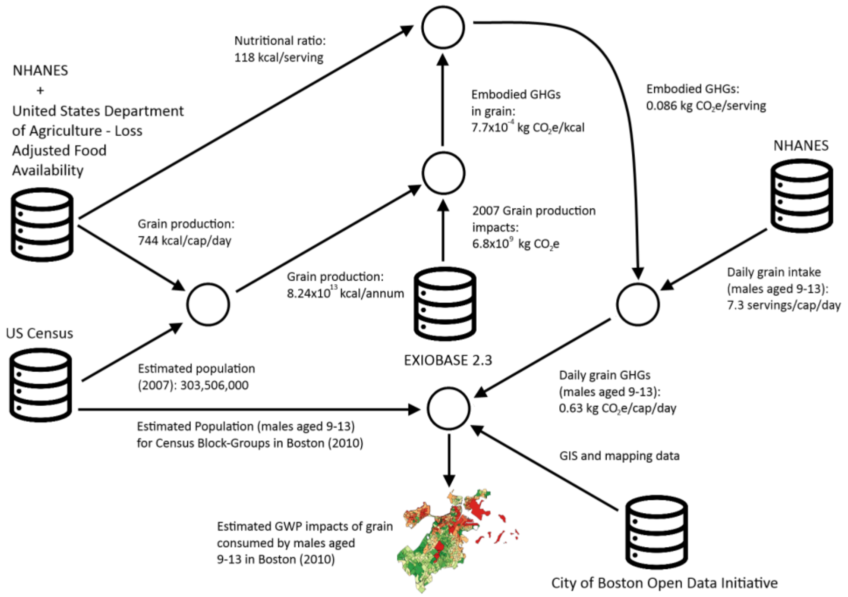


Figure 1. Workflow in generating baseline environmental performance, using grain consumption by adolescent males as an example.

Modelling UF in Boston

We assessed two UF forms: empty-lot and rooftop farms, both open to the ambient environment. Data were also collected on additional UF forms (ground/rooftop greenhouses and automated precision agriculture), but were not included in the model since they displayed worse environmental performance compared to conventional produce[21], and therefore, were poor candidates when assessing UF's substitution benefits. Resource use and yield data for two empty-lot farms and one rooftop farm were collected over the 2015 growing season for 14 vegetables covering 32% by mass (44% excluding potatoes) and 24% by calories (50% excluding potatoes) of total average US vegetable consumption[35]. Although only some of the 14 vegetables were environmentally preferable to conventional when produced with UF (see supporting information), including all vegetables was representative of actual production scenarios where farmers are free to choose their crops.

GWP and land use impacts from UF production were modeled with process-LCA from cradle to point of purchase, aligning with the EXIOBASE scope (transport impacts were added manually to EXIOBASE results) and capturing the majority of food related impacts[36–38]. Conventional crop GHG intensities were taken as mean values from Keolian and Heller's review of food LCAs[39], corrected for distribution losses and average transport distances. Land use was taken as direct agricultural land occupation from USDA production data[40], corrected for distribution losses[35], and excluded final distribution burdens. LCA modelling details are in the supporting information.

Ground space potentially available for UF was determined using *additive* and *subtractive* approaches. The additive approach assessed over 160,000 individual properties in Boston, calculating total UF space as the sum of properties with land uses amenable to UF (vacant lots, pasture, open land, cropland, transitional, etc.) The subtractive approach started with the city's entire land area and subtracted land uses unsuitable for UF (steep slopes, impermeable surfaces, protected parkland, etc.) to arrive at an upper estimate from the opposite direction. Rooftop UF space was estimated using a dataset of 80,000 buildings in Boston. Lacking structural data, the year of construction was used as a proxy for load bearing ability. We tested cutoff years from 1900 to 2000 to quantify the effect of this choice on model results since this range covered ~80% of the city's buildings. Buildings over 30 meters high, having sloped roofs or historically protected were assumed unusable for UF. As some buildings lacked data on roof-type, 100-run Monte-Carlo simulations were performed for each cutoff year, examining the impact of probabilistic roof-type assignment. The supporting information details the UF area estimates.

In considering UF interactions with the city we included avoided runoff, municipal organic solid waste assimilation and building energy impacts. High and low estimates of runoff reduction were taken as the average rainfall in Boston times the formally impermeable UF area, using upper and lower retention values from previous studies[41,42]. Solid waste assimilative capacity was taken from primary data on urban farm compost consumption converted to mass of original organic waste. The same dataset used in calculating roof space includes heating and air conditioning data which were combined with commercial[43] and residential[44] energy surveys to estimate building energy loads. Previous studies of heating and cooling savings from vegetated roofs were used to estimate energy savings from building situated UF[45]. UF interactions with Boston's hydrological, waste and energy systems are detailed in the supporting information.

We modeled the most efficient application of Boston's UF space towards both land use and GWP reduction. An algorithm was run whereby each block-group produced vegetables that resulted in the largest reduction in GWP or land use depending on optimization goal, while respecting local demands for each crop as a constraint. Space was allocated to a vegetable until the block-group was satiated (at which point the next best vegetable for the optimization goal was produced), UF space was exhausted or all vegetable needs were met. After all blocks-groups had the chance to produce for themselves, those with extra capacity produced for those lacking space until total vegetable needs for the city were met or Boston's UF space was exhausted. See supplementary information for detailed explanation of optimization algorithm.

Given the different UF space estimation methods and optimization goals, four scenarios were run. Within each scenario 10 different building age cutoffs were considered using 100-run Monte-Carlo simulations. Table 1 outlines these scenarios.

Scenario	Description
GWP(+)	Optimization for GWP reduction using additive method to estimate UF space
GWP(-)	Optimization for GWP reduction using subtractive method to estimate UF space
Land(+)	Optimization for land use reduction using additive method to estimate UF space
Land(-)	Optimization for land use reduction using subtractive method to estimate UF space

Results and Discussion

Figure 2a shows the average, baseline GWP for Boston's food demands according to the NHANES usual daily intakes for different demographics. Calculated GWP was 2211±55 kg CO_2e/cap/a aligning with national assessments using EXIOBASE[27], with the main impacts emanating from the meat and dairy products (54%). Figure 2c focuses on GWP impacts for the individual block-groups which varied between 2078-2211 kg CO_2e/cap, where those with greater proportions of adults and males sat at the upper end. The influence of meat and dairy agrees with other assessments of the US diet[29,38,39]. GWP estimates are larger than process-LCA accounts of the US diet[39], but well aligned with other input-output analyses of US food consumption[28,38], a result of the latter method's enhanced value-chain coverage when building inventories. The tight spread around Boston's mean and proximity of city and national averages support the use of the latter as a proxy for urban level impacts, though caution is warranted when using this simplification in settings with substantial income inequality.

Figure 2b presents land related impacts which averaged 9077±198 m^2/cap/a (8578 to 10554 m^2/cap/a), agreeing with the earlier national EXIOBASE work[27] and studies that have pegged average US food-related land occupation between 0.86-1 ha/a[46,47]. Meat and dairy were again dominant (~50%), while fruits and vegetables were also key (20%). The focus on animal based products agrees with previous work, but the percentage of total land burdens is reduced. Peters et al.'s assessments of US diets have found that animal products accounted for approximately 75% of the ~1 ha/a land use burdens[46,47], and Eshel and colleague's calculate over 1 ha/cap/a for animal-sourced foods alone[48]. Misalignment with these other studies might stem from the calorie allocation method employed here, since EXIOBASE products divided between animal- and vegetal-sourced foods (e.g. 'Food products nec' a catch-all EXIOBASE product for various processed foods, accounting for 39% of total land use) are disproportionately allotted to the latter, due to the poorer energetic returns per unit area when moving up trophic levels[47,48].

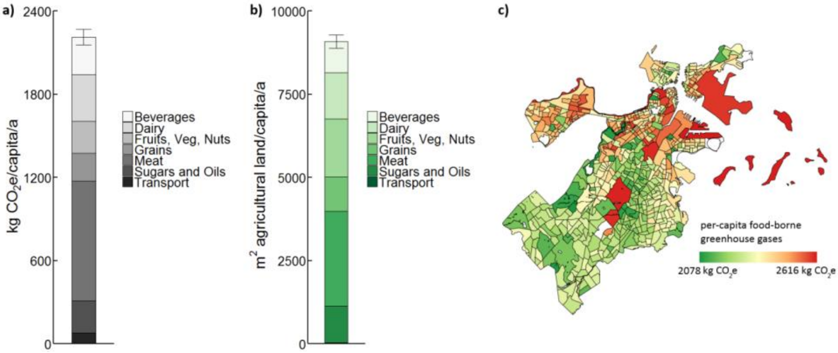


Figure 2a-c. Average baseline food related (A) GWP and (B) land use impacts for Boston in 2010 based on NHANES demographic usual daily intakes. Error bars represent standard error amongst city population. (C) Average GWP at the block-group level, with uninhabited blocks shown as white.

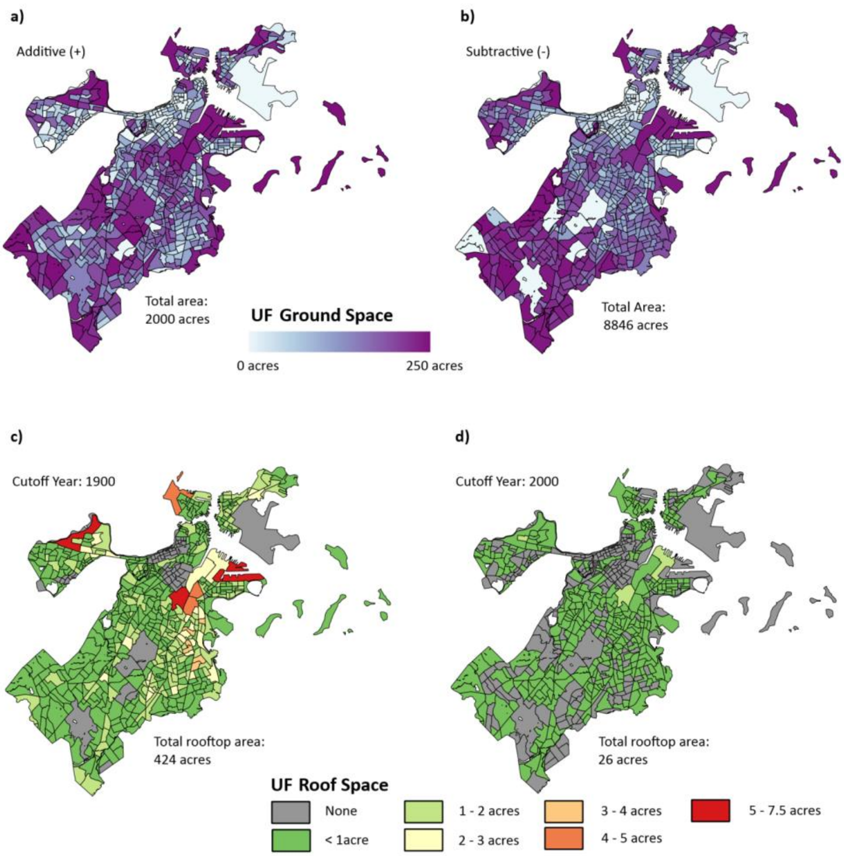
Available UF space

Figures 3a-b show Boston's available ground UF space calculated with subtractive and additive methods estimated at 8846 and 2000 acres (28.8% and 6.7% Boston's area), respectively. Naturally, the lower density block-groups with dispersed built forms tended to have more UF potential, but appreciable area was also found in the former industrial areas and port lands. These estimates ignored contaminated land that would likely be precluded from UF without remediation, but are suitable approximations of where UF could be placed without disturbing Boston's built form. Figure 3c-d presents 100-run Monte Carlo average UF available roof area in each block-group for the lowest (1900) and highest (2000) construction cutoff years, respectively. A 1900 cutoff resulted in 8828 available UF buildings with average area 195 m^2 netting 424±8 total acres. Using 2000 as a cutoff year left only 700 buildings with a mean area of 379 m^2, covering a mere 26±3 acres. The majority of Boston's buildings were built prior to 1920, and accordingly, estimated rooftop UF space remained below 200 acres at cutoffs above this year (see supplementary information figures for further details).

Figure 3a-d. Ground UF space in individual block-groups in Boston using (a) additive, (b) subtractive and rooftop space using construction year cutoffs of (c) 1900 and (d) 2000.

Environmental performance of UF

Figure 4a exhibits the changes in GWP potential through the introduction of UF into Boston for the four scenarios. Results average all Monte-Carlo runs and all years for each scenario. The GWP(+) scenario provided 20% greater GHG reductions compared to the Land(+) (18066±432 vs. 15045±523 tons CO_2e/a). With the subtractive method both GWP and land optimizations approached each other (~24000 tons CO_2e/a) since they both produced until Boston's demands for the UF crops are met, with slight differences due to allocation choices (ground vs. roof) for select vegetables. In the best cases, UF reduced Boston's total food-bo



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GWP burdens by approximately 1.1% (12% of fruit and vegetable burdens) when limited by space, and by 1.3% (15% of fruit and vegetable burdens) when producing until vegetable demands were met.

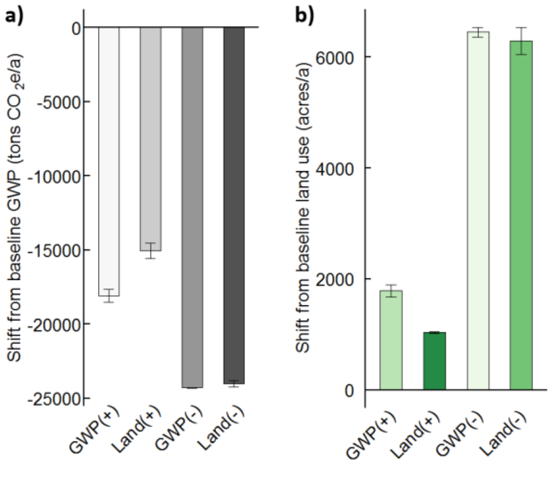

Figure 4. Impacts of UF on (a) GWP and (b) land use for all model scenarios. Error bars show variance over all building construction cutoff years.

Figure 4b shows the change in land use for the four scenarios. In all cases UF led to net increases in land occupation. The Land(+) optimization minimized these to 57% of those from the GWP(+) scenario (1033 vs. 1786 acres/a increase). Akin to the GWP results, land use for both optimization scenarios approached each other using the upper bound of UF space (~6400 acre/a increase). In the context of Boston's total food-related land occupation, these increases were a mere 0.07-0.5%, and although hinting at UF's potential to worsen a city's environmental performance, are not reason to outright discount UF as a food source for Boston. Land use increases stemmed from the low-yield, ground-based UF which is the dominant farm type in the model. Whilst UF frees some land beyond city boundaries, the practice requires more land within city borders to produce an equivalent volume, highlighting the comparative advantage of conventional production. Although already appropriated from the wild and hence imparting low ecological 'costs' in converting to UF, it is worth considering whether vacant urban land is best utilized for UF when solar farms net significantly greater environmental benefits per unit area, particularly for GHG reductions[21], but this could change in regions with lower GHG grid intensities (the Northeast US is primarily fossil fuel supplied). Rooftop UF performed quite well compared to conventional agriculture (exceptions being low yield vegetables where embodied land use in capital is large), but the relatively small rooftop area cultivated was not enough to counteract increases from ground UF. Although UF increased food related land use, the conversion of urban space to farms could increase urban biodiversity[49,50] and contribute to green corridors through the city, potentially justifying the practice.

UF impacts on Boston's energy and material metabolism

Naturally, the more buildings employed for UF, the larger the building energy related GWP reductions in Boston. In the Land(+) scenario, building energy savings accounted for 19% and 1% of total GWP reductions to the city using 1900 and 2000 as construction cutoffs, respectively, compared to 17% and 1% for the GWP(+) simulations. Both optimizations resulted in approximately the same building energy GWP reductions (3.2×10^6 kg CO_2e), but they took on increased importance for the land optimization due to its poorer GWP performance. When the simulations ran until Boston vegetable demands were met, building energy reductions contributed a maximum of 5% to total GWP reductions as building UF took on a diminished share of total production. In terms of contributions to total building energy demand, reduction from UF's was in the single digits. UF's potential urban heat island mitigation was excluded here, which could reduce ambient temperatures by 1-2° C[51], affecting cooling energy loading. However, cooling energy pales in comparison to heating demands in New England (1% and 59% of total residential end use, respectively in Massachusetts)[43,44], hinting at the limited ability of UF to affect baseline urban energy metabolism, although more detailed modeling is required.

Figures 5a-b outline UF's impacts on surface runoff and solid waste flows in Boston. Building space was highly influential on these interactions since it is the majority of UF area that shifts from non- to permeable and its significant compost needs due to soil losses and expanded shale grow-media devoid of nutrients[21]. Here we focus on building cutoff years of 1900 and 2000 (other years shown in the supplementary information graphics). Figure 4a shows that the GWP(+) and Land(+) simulations (averaged due to similarity) provided significantly greater runoff retention, since they were forced to use all available building area. The subtractive scenarios provided less runoff reduction as they favored ground UF when optimizing (particularly the GWP(+) scenario) and did not convert any impermeable area on the ground to UF. The maximum estimated runoff retention was 2.0 Mm^3/a, or 2.0% of annual stormwater (1.11 m annual precipitation falling on 67.8 km^2 impermeable area and 57.4 km^2 permeable with 50% assumed retention[49]).

Yard and kitchen solid waste assimilation as compost (also averaged for land and GWP optimizations) was highest for the subtractive scenarios (~12000 tons/a), decreasing as building space was removed to approximately 8200 tons/a by cutoff year 2000. Additive scenarios provided lower waste assimilation capacity (10648-4026 tons/a) and were more sensitive to building space removal as this constituted a larger proportion of UF area. By our estimates UF could absorb at most 9% of Boston's municipal organic solid waste fraction at 2009 generation rates[52].

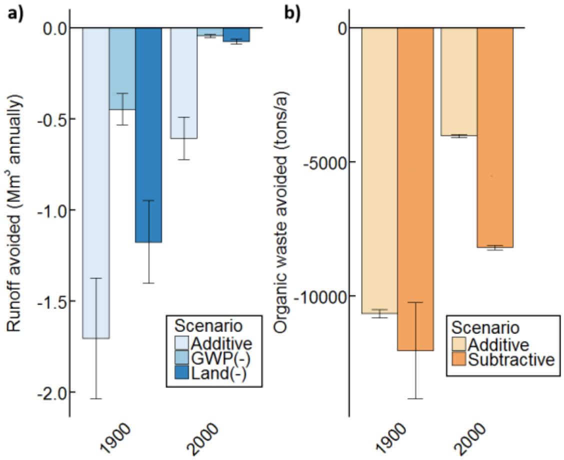

Figure 5a-b. Effects of UF on Boston's material metabolism for (a) runoff and (b) organic solid waste uptake for cutoff years 1900 and 2000. Error bars display range for high and low retention values for the runoff and variance over 100 Monte Carlo simulations for waste uptake.

Alternative motives for UF

Given UF's meager improvements in food related GWP and potential exacerbation of land use impacts, urban designers in the Northeast US should reconsider their enthusiasm for UF as a component of an environmentally sustainable urban food system, especially compared to higher environmental gains from other land applications[21]. Urban farms in the region do not tackle the animal-sourced foods that drive dietary environmental burdens. Other cities in the Global North are actively promoting reduced meat intake as explicit environmental initiatives, recognizing the importance of diet, and not technology, as means to more sustainable cities[53,54]. In a US context, shifting from average to vegetarian and vegan diets would reduce GWP by 30% and 50%, respectively[39] and reduce land use by a factor two or greater[47].

Effects of UF on Boston's building energy demands and surface runoff were both meagre, though the latter's ability to stymie sewage overflow events during heavy rains is notable[55]. UF incorporated a sizeable amount of organic solid waste, although meaningful shifts towards a circular metabolism should tackle wastewater management systems, where most imported nutrients end up[56,57].

Notwithstanding, UF is also often promoted as a social enterprise in the US Northeast[7]. The slight environmental gains should be compared against performance in other domains to see if current policies are justified given alternative motivations. In Boston, a significant percentage of residents do not meet recommended fruit and vegetable guidelines, and some of the city's neighborhoods have elevated poverty rates[58]. Here we test UF's potential contributions towards alleviating these challenges.

Nutritional Improvements

UF's nutritional contributions were assessed as the percentage of USDA recommended annual vegetable intake met for the three vegetable types grown by our case farms: 'dark green' (e.g. spinach, kale, broccoli), 'red and orange' (tomatoes, carrots, squash) and 'other' (lettuce, onions, cucumbers)[59]. USDA guidelines for these vegetable types at different ages and sexes were combined with census data to calculate Boston's total vegetable needs. We estimated that Boston currently consumes 64%, 64% and 85% of its dark green, red/orange and other vegetable needs, respectively (see Table S47 in the supplementary information). Nutrition optimization algorithms were run for both additive (+) and subtractive (-) grow area estimates, where the farms supplied equal nutritional requirements for each vegetable type.

Figure 6 shows average nutritional output for the previous scenarios and nutrition optimizations. Both GWP and land optimizations provided appreciable percentages of red/orange and other vegetable needs, but dark greens were not produced in volumes greater than 11% of recommended intakes. The Nutrition(+) optimization reduced red/orange and other vegetable production, but provided a fourfold jump in dark green cultivation, while the Nutrition(-) scenario supplied all of Boston's needs in the three categories.

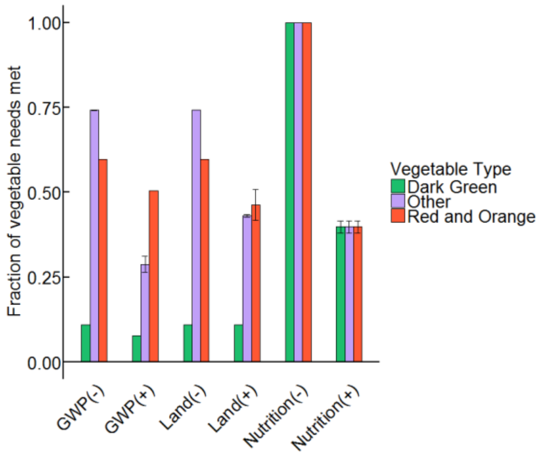

Figure 6. Fraction of vegetable needs met by UF in Boston for the different scenarios according to USDA dietary guidelines.

In terms of GWP reduction the nutritional scenario was similar to the Land(+) simulation (15726±733 tons CO_2e/a) and provided the largest reductions of all scenarios when producing until all nutritional needs were assuaged (39312±25 tons CO_2e/a or 2.9% of average dietary GWP), since it substituted the greatest volume of conventional produce. When producing to meet all nutritional demands, land use impacts were reduced relative to the other optimizations (3746±77 acres/a), since the Nutrition(-) scenario grew significantly more dark green vegetables, which provide high marginal land use reductions. The scale of interactions with the city remained largely unchanged, but the Nutrition(-) scenario had 25% greater solid waste assimilation capacity since cultivated area was the largest of all scenarios.

We also tested when UF acts as an *ancillary food supply* that can be used to alleviate the aforementioned gaps between USDA guidelines and current consumption. As UF would not substitute conventional produce here, no crediting was provided to the city and full burdens of UF production were ascribed to Boston. By our estimates Boston could actually close its nutritional gap for these food groups within the UF space estimated by the additive method, with the downside of increasing land use by 2608±89 acres/a (0.2%) and GWP by 2950±138 tons CO_2e/a (0.2%). However, the ecological costs should be weighed against the benefits of closing nutritional gaps, particularly in inner city neighborhoods bereft of fresh vegetable choices where lifestyle related diseases are more prominent[60,61]. Nutritional gaps would remain for other vegetable types ('starchy' and 'legumes') and fruits, but promoting UF as a public health measure appears justified.

Economic benefits

Lastly we looked at the ability for UF to provide economic returns to the block-groups for all of the GWP, land and nutritional optimization scenarios. Because supplying Boston's vegetable demands or nutritional needs required ~50% and 64% of total UF area, respectively, we also explored Boston's potential to export beyond its borders to the larger metropolitan area. Vegetable prices were taken US Bureau of Labor Statistics and USDA data[62,63]. The 191 acres of UF applied to surface parking in the additive scenarios were removed here, since this area already generates revenue.

Figure 7a shows that when restricted to intra-block-group trading, estimated UF market value was lowest (~1.5×10^7 USD) for the GWP(-) and Land(-) trials, as more block-groups were self-sufficient. Market value for internal trading is maximized (~4.9×10^7 USD) when the model aimed to meet its nutritional needs, as this left the most block-groups in production deficits, necessitating purchases from block-groups with surplus production capacity. Figure 7b shows an estimated market value of ~1.6×10^8 USD when the city used all UF space, with exports to the metropolitan region accounting for ~90% of that when producing to meet current vegetable *demands* and dropping to 67% when satisfying nutritional *needs*. Situating this within the Boston-Cambridge-Newton metropolitan area, estimated UF market value amounted to less than 0.5% of regional GDP[64]. Notwithstanding, Figure 7c maps potential UF revenue in the Nutrition(-) scenario along with household poverty rates in Boston, demonstrating UF as a latent revenue stream to some of Boston's impoverished neighborhoods. ~$2.5*10^7$ USD could be generated in low income block-groups housing ~81 thousand residents (1/3 of Boston's residents in low income blocks). However, most of the market value (~$1.0*10^8$ USD) would benefit blocks with poverty rates below 25%.

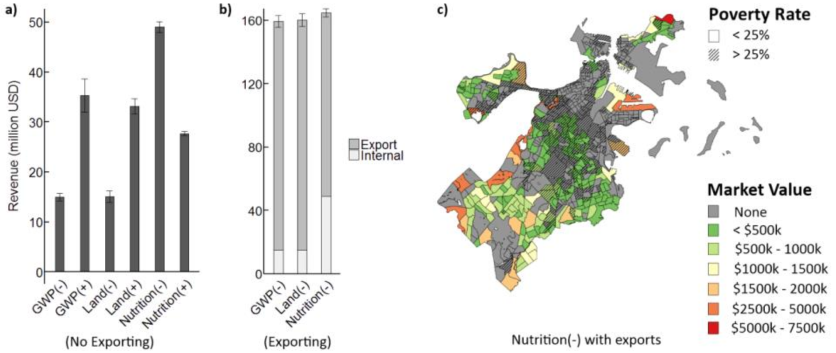

Figure 7a-c. Potential UF revenue in Boston when (a) limited to intra-block-group trading and (b) exports outside of city allowed. (c) Revenue production (with exporting) in block-groups after supplying cities nutritional needs with household poverty rates overlaid.

UF and study challenges

One challenge to the diffusion of UF into the city is pollution in soil and groundwater, as well as aerial deposition of contaminants from the concentration of industry and traffic in urban areas[65,66]. Of particular concern is the legacy of lead in soil from lead-containing fossil fuel combustion, although minimal uptake outside of the root zone occurs, and oral intake can be obviated through discarding of root portions and proper rinsing of edible portions[67]. Polycyclic aromatic hydrocarbons pose a similar issue, more so from aerial deposition than plant uptake, and can usually be made safe for consumption by rinsing edible portions[67]. Actual ingestion of toxic substances through UF remains understudied, and is a serious concern despite these positive signs. The presence of contamination is site specific, but it is correlated to age and density of the city[66], and in Northeast US cities the amount of current UF suitable area is certainly lower than our estimates.

UF is also at odds with other more economically competitive land uses that are usually preferred by municipal governments, further reducing long term production capacity[68]. Securing UF's role as a nutrition source in the Northeast US will likely require more than making the practice legal, but active protection of UF suitable space to avoid transitory UF application. This could easily be done for city-owned vacant properties as a start.

By including potentially contaminated land in our models this study represents an optimistic take on the potential for UF to affect a city's environmental performance. At the same time, using process-based LCA for crop production may have underestimated the burdens of both UF and substituted vegetables due to inventory gaps, depressing or inflating UF substitution effects. Furthermore, UF practice is constantly evolving, with improvements to current systems and new systems entering the market[17]. Although previous work has demonstrated that technologically advanced urban farms in the study region are the most burdensome due to energy impacts[21], future developments might shift the balance in the opposite direction. Our findings are only a snapshot of the current best-practices in the study region, which should be reevaluated as UF technology and the region's electrical grid mix evolve. However, given the marginal impacts of UF in this study, such shifts need to be seismic in order for UF contribution meaningfully to making Northeast US urban food consumption more environmentally sustainable.

Results should also be viewed in light of Boston's relatively dense built form, which produces high competition for the scarce open space remaining, reducing UF's tenability in the city and its environmental and nutritional impacts. Less-dense or warmer Northeast US cities may have greater production capacities per capita and resultant UF benefits, requiring care in directly applying our results directly to other Northeast US cities. A more complete assessment of local farming would look beyond political boundaries, including low-density suburbs and peri-urban regions where higher production volumes are possible[14,19]. Regional food system strategies, such as Vancouver, Canada's[69], could help distinct political entities coordinate their disparate land use regimes to maximize production and more effectively harness residual resources, increasing local farming's benefits. Although focusing on Boston's geopolitical boundaries precluded such a regional perspective, this study reveals the current limits of a lone, urban municipality to reduce the environmental burdens of its food demands through technology.

Despite these methodological challenges, we have shown that when embedded within a complex city system, UF's environmental performance is more nuanced than the previous studies at the farm scale or using hypothetical UF data at the city scale would suggest. We have demonstrated that it cannot be assumed that UF by default results in leaner supply chains. Policy makers and other urban designers in the Northeast US will hopefully benefit from this and future work when considering UF as a sustainable design intervention in the region.

ABBREVIATIONS

CO_2e, carbon dioxide equivalents; UF, urban farming; USDA, United States Department of Agriculture; NHANES, National Health and Nutrition Examination Survey; MRIO, multi-region-input-output; LCA, life cycle assessment; GWP, global warming potential.

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Supporting Information Available

Background data for the development of the food borne environmental footprints. Description of methods for urban agriculture area estimates, urban agriculture life cycle assessment and algorithms for urban farming proliferation in the city. Raw results from the models of impacts of urban farming in the city.

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Supplementary Information

Contributions of local farming to urban sustainability in the Northeastern United States

Benjamin P. Goldstein†, Michael Z. Hauschild†, John E. Fernández‡, Morten Birkved†*

†Technical University of Denmark, Quantitative Sustainability Assessment Division, Produktionstorvet, Building 424, Kongens Lyngby, 2800, Denmark

‡Massachusetts Institute of Technology, Department of Architecture, 77 Massachusetts Avenue, 5-419, Cambridge, 02139, United States

*Corresponding author: Phone: +45 45254561; fax: +45 45933435; e-mail: bgol@dtu.dk

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Linking MRIO food impacts to different nutritional categories

As outlined in the article, ascribing the embodied impacts of from food consumption to different food products is done using kilo calories. The starting point of the assessment are the individual categories of nutrition as outlined by the United States Department of Agriculture's (USDA) 2015-2020 dietary guidelines[1].

USDA loss-adjusted food availability (LAFA) data[2] provide kilo calories per nutritional equivalent for individual foods within the broader nutritional categories (e.g. kcal of broccoli per cup equivalent of 'dark green vegetables'), which are then used to develop availability weighted averages of kilo calories per nutritional equivalent. Food losses are also included in the weighted average, so that the kilo calories per nutritional unit approximate the amount of kilo calories provided by the economy for consumption and not just those actually consumed. Tables S1-19 outline the calculations of embodied kilo calories in the nutritional group.

Table S1 - Vegetables: Dark Green

	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily	Food pattern equivalents available daily	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number -*	*-- Cups --*	*kcal*	*kcal produced/cup consumed*
Fresh broccoli	12	39	12	0.947	0.031	2.195	71.855
Fresh collard greens	37	43	38	0.029	0.002	0.248	100.999
Fresh escarole	48	14	24	0.030	0.004	0.092	24.664
Fresh kale	39	39	38	0.022	0.001	0.154	200.145
Fresh leaf lettuce	14	21	24	0.995	0.077	2.089	27.290
Fresh mustard greens	64	7	38	0.023	0.002	0.116	75.014
Fresh spinach	14	28	9	0.222	0.016	0.412	25.959
Fresh turnip greens	41	30	38	0.031	0.002	0.164	95.294
Frozen broccoli	6	-	12	0.585	0.011	0.707	62.863

Frozen spinach	6	-	34	0.131	0.002	0.212	104.771
						Weighted Average:	43.617

Table S2 – Vegetables: Other

	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily	Food pattern equivalents available daily	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number -*	*-- Cups --*	*kcal*	*kcal produced/cup consumed*
Fresh artichokes	19	60	18	0.149	0.002	0.840	428.102
Fresh asparagus	9	47	18	0.084	0.003	0.264	85.157
Fresh green bell pepper	8	18	39	0.427	0.014	1.078	75.645
Fresh Brussels sprouts	19	10	12	0.094	0.002	0.149	59.965
Fresh cabbage	14	20	24	1.093	0.050	2.271	45.721
Fresh cauliflower	14	61	9	0.125	0.005	0.485	104.660
Fresh celery	5	11	39	0.546	0.034	1.151	33.721
Fresh cucumbers	6	27	32	0.332	0.024	0.864	36.376
Fresh eggplant	21	19	26	0.101	0.005	0.233	46.222
Fresh garlic	7	13	43	1.665	0.008	4.087	498.354
Fresh head lettuce	9	16	24	1.557	0.097	2.843	29.216
Fresh mushrooms	13	3	21	0.421	0.020	0.634	31.651
Fresh okra	24	14	20	0.081	0.002	0.163	66.153
Canned olives	6	0	25	2.126	0.014	3.016	219.858
Fresh onions	10	10	43	4.283	0.067	10.099	150.892
Fresh radishes	21	10	47	0.035	0.002	0.102	55.947
Fresh snap beans	19	12	24	0.415	0.013	0.796	59.478
Fresh squash	12	17	25	0.373	0.021	0.735	35.456
Canned asparagus	6	0	2	0.026	0.001	0.028	49.935
Canned snap beans	6	0	24	0.416	0.014	0.582	41.993
Canned cabbage	6	0	16	0.083	0.003	0.105	34.195
Canned cucumbers	6	0	3	0.876	0.011	0.960	89.384
Canned mushrooms	6	0	9	0.250	0.006	0.292	45.593
Frozen asparagus	6	0	26	0.007	0.000	0.011	46.003
Frozen snap beans	6	0	24	0.444	0.012	0.622	53.191
Frozen cauliflower	6	0	27	0.041	0.001	0.060	49.548
Dehydrated onions	6	0	4	0.442	0.005	0.490	108.599
Fresh avocados	9	26	32	2.516	0.011	6.607	622.305
						Weighted Average	88.607

Table S3 – Vegetables: Red and Orange

	Loss from retail/ institutional to consum	Loss at consumer level		Kilo calories available daily	Food pattern equivalents available daily	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and				

	er level		uneaten food)				
Component	-- *Percent* --	-- *Percent* --	-- *Percent* --	-- *Number* -	-- *Cups* --	*kcal*	*kcal produced/cup consumed*
Fresh red bell pepper	8	18	39	0.427	0.014	1.077	75.644
Fresh carrots	5	11	34	2.056	0.039	3.942	99.668
Fresh pumpkin	11	30	69	0.012	0.001	1.436	3379.731
Fresh tomatoes	13	9	7	3.370	0.082	4.621	56.220
Canned carrots	6	0	31	0.136	0.003	0.210	55.504
Canned chili peppers	6	0	4	1.022	0.035	1.132	32.136
Canned tomatoes	6	0	28	3.750	0.098	5.541	56.146
Other canned vegetables	6	0	16	0.503	0.010	0.637	62.056
Frozen carrots	6	0	34	0.236	0.004	0.381	87.040
						Weighted Average	66.720

Table S4 – Vegetables: Starchy							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily	Food pattern equivalents available daily	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	-- *Percent* --	-- *Percent* --	-- *Percent* --	-- *Number* --	-- *Cups* --	*kcal*	*kcal produced/cup consumed*
Fresh sweet corn	1	64	32	0.361	0.003	9.091	3144.228
Fresh potatoes	6	10	16	28.391	0.229	41.031	178.492
Fresh sweet potatoes	14	28	44	1.172	0.010	4.879	474.417
Canned sweet corn	6	0	7	3.642	0.033	4.166	125.829
Canned green peas	6	0	24	0.476	0.004	0.666	166.573
Canned potatoes	6	0	28	0.314	0.003	0.465	159.574
Frozen sweet corn	6	0	36	1.638	0.012	2.722	222.739
Frozen green peas	6	0	24	1.167	0.009	1.634	174.972
Frozen lima beans	6	0	27	0.293	0.001	0.427	275.429
Frozen potatoes	6	0	16	20.345	0.143	25.761	179.837
Misc. frozen vegetables	6	0	26	0.787	0.009	1.132	113.858
Dehydrated potatoes	6	0	4	7.227	0.068	8.008	117.464
Potato chips	6	0	4	28.465	0.186	31.544	169.547
Dry lima beans	6	0	10	0.118	0.002	0.139	63.829
						Weighted Average	183.947

Table S5. Fruits – Citrus							
	Loss from retail/	Loss at consumer level		Kilo calories	Food pattern	Produced kilo	Calorific density

	institutional to consumer level	Nonedible share	Other (cooking loss and uneaten food)	available daily	equivalents available daily	calories	
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*-- Cups --*	*kcal*	*kcal produced/cup consumed*
Fresh oranges	12	27	36	1.382	0.016	4.225	261.314
Fresh tangerines	20	26	52	0.279	0.003	1.595	588.447
Fresh grapefruit	13	50	20	0.286	0.003	1.095	313.588
Fresh lemons	7	47	44	0.081	0.001	0.964	728.504
Fresh limes	8	16	44	0.293	0.015	0.800	54.536
Fresh blueberries	5	5	8	0.314	0.004	0.380	101.898
Fresh cranberries	6	2	26	0.038	0.001	0.055	75.332
Fresh honeydew	23	54	43	0.017	0.000	0.754	2697.253
Fresh kiwi	13	14	45	0.108	0.001	0.302	212.224
Fresh raspberries	10	4	20	0.074	0.001	0.108	93.312
Fresh strawberries	10	6	35	1.220	0.025	2.291	92.591
Fresh watermelon	17	48	13	1.573	0.034	4.845	141.727
Frozen blackberries	6	0	40	0.042	0.000	0.074	171.986
Frozen raspberries	6	0	24	0.191	0.003	0.268	102.184
Frozen strawberries	6	0	24	0.151	0.003	0.211	72.788
Frozen other berries	6	0	30	0.023	0.000	0.035	147.416
						Weighted Average	162.457

Table S6. Fruits - Juice							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*-- Cups --*	*kcal*	*kcal produced/cup consumed*
Orange juice	6	0	10	18.071	0.161	21.361	132.388
Grapefruit juice	6	0	10	1.252	0.013	1.480	113.475
Lemon juice	6	0	10	0.283	0.005	0.335	63.830
Lime juice	6	0	10	0.050	0.001	0.059	70.922
Apple juice	6	0	10	9.712	0.085	11.480	134.752
Cranberry juice	6	0	10	0.990	0.009	1.170	137.116
Grape juice	6	0	10	3.163	0.021	3.739	179.669
Pineapple juice	6	0	10	1.055	0.008	1.247	156.028

Prune juice	6	0	32	0.223	0.001	0.349	284.731
						Weighted Average	135.490

Table S7. Fruits: Other							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*-- Cups --*	*kcal*	*kcal produced/cup consumed*
Fresh apples	9	10	20	6.597	0.118	10.314	87.546
Fresh apricots	35	7	10	0.045	0.001	0.084	141.924
Fresh bananas	8	36	20	11.644	0.091	28.754	314.855
Fresh cantaloupe	12	49	43	0.261	0.005	3.716	804.376
Fresh cherries	4	9	51	0.294	0.003	0.766	221.060
Fresh grapes	8	4	33	3.670	0.035	6.303	178.627
Fresh mangoes	14	31	13	0.714	0.007	1.490	206.683
Fresh papaya	55	33	20	0.116	0.002	0.547	292.434
Fresh peaches	12	7	42	1.045	0.016	2.326	146.909
Fresh pears	18	10	20	1.209	0.013	2.095	156.860
Fresh pineapple	15	49	37	0.352	0.004	2.945	686.082
Fresh plums	17	6	27	0.304	0.004	0.548	137.189
Canned apples	6	0	8	2.210	0.022	2.555	117.946
Canned apricots	6	0	27	0.032	0.001	0.046	72.865
Canned sweet cherries	6	0	32	0.006	0.000	0.010	178.348
Canned tart cherries	6	0	32	0.051	0.001	0.079	137.672
Canned peaches	6	0	9	1.176	0.020	1.374	68.974
Canned pears	6	0	9	1.099	0.015	1.285	83.002
Canned pineapple	6	0	9	1.335	0.017	1.561	92.354
Canned plums	6	0	26	0.007	0.000	0.011	146.636
Frozen blueberries	6	0	29	0.158	0.002	0.237	118.370
Frozen sweet cherries	6	0	29	0.149	0.001	0.223	346.119
Frozen tart cherries	6	0	29	0.187	0.003	0.280	106.383
Frozen apples	6	0	35	0.195	0.002	0.320	135.843
Frozen apricots	6	0	35	0.015	0.000	0.025	135.843
Frozen peaches	6	0	35	0.258	0.003	0.422	135.843
Frozen plums	6	0	10	0.002	0.000	0.002	98.109
Dried apples	6	0	11	0.287	0.003	0.343	124.313
Dried apricots	6	0	11	0.312	0.002	0.373	187.664
Dried dates	6	10	25	0.265	0.001	0.434	338.789
Dried figs	6	0	25	0.203	0.001	0.288	263.830
Dried peaches	6	0	11	0.118	0.001	0.141	228.305
Dried plums	6	0	11	0.732	0.004	0.875	249.821
Raisins	6	0	26	3.653	0.017	5.252	311.961
						Weighted Average	183.640

Table S8. Grains							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/ Oz consumed*
White and whole wheat flour	12	0	20	376.435	4.591	534.709	116.477
Rye flour	12	0	20	2.152	0.027	3.056	113.636
Rice	12	0	33	54.012	0.524	91.607	174.695
Corn flour and meal	12	0	20	60.937	0.952	86.558	90.909
Corn hominy and grits	12	0	20	28.810	0.389	40.923	105.114
Corn starch	12	0	20	11.548	0.241	16.404	68.182
Barley products	12	20	14	1.716	0.017	2.955	172.176
Oat products	12	20	14	12.920	0.121	22.246	184.229
Durum flour	12	0	20	40.140	0.379	57.018	150.568
						Weighted Average	118.150

Table S9. Dairy - Cheese							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Cup*	*kcal*	*kcal produced/ cup consumed*
Cheddar cheese	6	0	11	42.890	0.248	51.267	206.789
Other American cheese	6	0	28	8.707	0.053	12.866	241.578
Provolone cheese	6	0	14	3.751	0.025	4.640	184.315
Romano cheese	6	0	8	1.154	0.007	1.335	190.796
Parmesan cheese	6	0	8	2.818	0.016	3.258	206.984
Mozzarella cheese	6	0	31	25.454	0.204	39.245	192.414
Ricotta cheese	6	0	12	1.384	0.007	1.673	233.922
Other Italian cheese	6	0	16	0.671	0.004	0.850	205.547
Swiss cheese	6	0	50	2.803	0.017	5.965	344.681
Brick cheese	6	0	40	0.063	0.000	0.111	280.142
Muenster cheese	6	0	35	0.933	0.006	1.526	255.319
Blue cheese	6	0	43	0.678	0.005	1.266	279.955
Other miscellaneous cheese	6	0	42	3.554	0.023	6.519	287.417
Regular cottage cheese	12	0	31	0.705	0.002	1.162	602.767
Low-fat cottage cheese	12	0	4	1.063	0.003	1.259	384.706

Weighted Average	214.276

Table S10. Dairy - Milk

	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Cup*	*kcal*	*kcal produced/ cup consumed*
Plain whole milk	12	0	20	28.079	0.188	39.885	211.648
2 percent milk	12	0	20	26.152	0.214	37.148	173.295
1 percent milk	12	0	20	8.044	0.079	11.426	144.886
Skim milk	12	0	20	8.050	0.097	11.435	117.898
Whole flavored milk	12	0	45	1.096	0.005	2.264	429.752
Low-fat flavored milk	12	0	45	5.433	0.030	11.226	380.165
Buttermilk	12	0	18	0.603	0.006	0.836	135.809
Ice cream	12	0	24	24.519	0.060	36.661	613.038
Ice milk	12	0	24	8.826	0.022	13.197	613.038
Other frozen	12	0	33	4.895	0.021	8.303	388.399
Evap condensed canned whole milk	12	0	15	15.950	0.094	21.323	225.936
Evap condensed bulk whole milk	12	0	15	40.737	0.241	54.462	225.936
Evap condensed skim milk	12	0	15	4.289	0.043	5.733	133.690
Dry whole milk	1	0	41	0.562	0.004	0.962	272.214
Nonfat dry milk	1	0	41	7.366	0.068	12.610	186.612
Dry buttermilk	1	0	41	0.649	0.006	1.111	198.596
Eggnog	12	0	51	0.044	0.000	0.101	347.866
						Weighted Average	228.156

Table S11. Dairy - Yoghurt

	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Cup*	*kcal*	*kcal produced/cup consumed*
Yoghurt	12	0	21	5.804	0.041	8.349	205.696

Table S12. Protein – Eggs							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/Oz consumed*
Eggs	9	12	13	37.843	0.485	63.978	131.868

Table S13. Protein - Legumes							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/ Oz consumed*
Fresh lima beans	12	56	27	0.005	0.000	0.036	310.524
Dry Peas and lentils	6	0	10	0.317	0.007	0.375	56.865
Dry black beans	6	0	10	0.769	0.014	0.909	64.291
Dry great northern beans	6	0	10	0.334	0.007	0.395	57.460
Dry navy beans	6	0	10	1.450	0.025	1.714	68.246
Dry pinto beans	6	0	10	4.078	0.069	4.821	69.708
Dry red kidney beans	6	0	10	0.649	0.012	0.767	61.855
Other dry beans	6	0	10	1.806	0.033	2.134	65.581
						Weighted Average	66.797

Table S14. Protein - Meat							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/Oz consumed*
Beef	4	0	20	170.871	2.084	223.290	107.155
Veal	25	0	20	0.557	0.009	0.934	108.934
Pork	4	0	29	94.164	1.405	138.666	98.665
Lamb	12	0	20	2.140	0.026	3.050	118.300
						Weighted Average	103.855

Table S15. Protein - Poultry							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/Oz consumed*
Chicken	4	0	15	141.470	2.143	173.318	80.858
Turkey	3	0	35	20.498	0.380	32.666	86.055
						Weighted Average	81.640

Table S16. Protein - Nuts							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/Oz consumed*
Peanuts	6	0	4	40.179	0.502	44.524	88.652
Almonds	6	0	21	6.518	0.079	8.777	110.423
Hazelnuts	6	0	20	0.296	0.003	0.394	118.351
Pecans	6	0	14	3.095	0.032	3.828	121.227
Walnuts	6	0	18	2.999	0.032	3.890	120.654
Macadamia	6	0	8	0.835	0.008	0.965	117.946
Pistachios	6	0	16	1.267	0.016	1.605	101.317
Other tree nuts	6	0	18	6.525	0.073	8.466	116.373
						Weighted Average	97.164

Table S17. Protein - Fish and Seafood							
	Loss from retail/ institutional to consumer level	Loss at consumer level		Kilo calories available daily[4]	Food pattern equivalents available daily[5]	Produced kilo calories	Calorific density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	*-- Percent --*	*-- Percent --*	*-- Percent --*	*-- Number --*	*Oz*	*kcal*	*kcal produced/ Oz consumed*
Fresh and frozen fish	9	0	40	5.598	0.158	10.218	64.802
Fresh and frozen shellfish	9	0	40	3.265	0.131	5.998	45.925
Canned Salmon	6	0	17	0.406	0.010	0.521	49.987
Canned Sardines	6	0	36	0.304	0.005	0.505	98.072

Canned Tuna	6	0	17	3.035	0.092	3.889	42.297
Canned shellfish	6	0	17	0.419	0.015	0.536	35.888
Other canned fish	6	0	17	0.402	0.010	0.515	49.987
Cured fish	6	0	17	0.336	0.010	0.431	42.297
						Weighted Average	52.438

Table S18. Fats and Oils

	Loss from retail/ institutional to consumer level	Loss at consumer level		Energy content	Per capita availability adjusted for loss	Per capita availability adjusted for loss	Calorific Density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	-- *Percent* --	-- *Percent* --	-- *Percent* --	kcal/g	g/d	kcal/d	kcal produced/g consumed
Butter	7	0	35	9	2.815	25.333	14.888
Margarine	7	0	35	9	2.037	18.335	14.888
Lard	50	0	35	9	0.651	5.858	27.692
Edible beef tallow	50	0	35	9	1.189	10.705	27.692
Shortening	21	0	35	9	13.368	120.316	17.527
Salad and cooking oils	21	0	15	9	41.885	376.969	13.403
Other edible fats and oils	5	0	25	9	1.488	13.388	12.632
Light cream	12	0	12	9	1.556	14.000	11.622
Sour cream	12	0	8	9	0.819	7.371	11.117
Cream cheese	12	0	13	9	0.744	6.695	11.755
Eggnog	12	0	51	9	0.011	0.098	20.872
						Weighted Average	14.630

Table S19. Sugars

	Loss from retail/ institutional to consumer level	Loss at consumer level		Energy content	Per capita availability adjusted for loss	Per capita availability adjusted for loss	Calorific Density
		Nonedible share	Other (cooking loss and uneaten food)				
Component	-- *Percent* --	-- *Percent* --	-- *Percent* --	kcal/g	kcal/d	kcal/d	kcal produced/kcal consumed
Cane and beet sugar	11	0	34	-	169.976	289.370	1.702
Edible syrups	11	0	15	-	2.221	2.936	1.322
Honey	11	0	15	-	3.340	4.415	1.322
High fructose corn sweetener	11	0	34	-	155.278	264.349	1.702
Glucose	11	0	34	-	38.070	64.811	1.702
Dextrose	11	0	34	-	8.256	14.056	1.702

Weighted Average	1.697

We take the average US usual daily food intake for each nutritional group over the years 2007-2010 (in cup equivalents, ounce equivalents, etc) from the National Nutritional Health and Nutrition Examination Survey (NHANES)[3] and multiply this by the population (301,231,207) and kilo calories per serving nutritional component to arrive at the total calorific demand of the US population. Table S20 outlines these findings.

Table S20. US average per capita food equivalent, daily calorific and annual calorific intake (2007-2010)

Dietary Component	**NHANES Usual Daily Intake**	**Unit**	**kcal produced per nutritional equivalent**	**Daily per capita kcal**	**Annual per capita kcal**
Vegetables					
Dark Green	0.1	cup eq.	43.61664	4.36166418	1592.007427
Red and Orange	0.4	cup eq.	65.72036	26.2881426	9595.172043
Other	0.5	cup eq.	88.60711	44.3035567	16170.79821
Starchy	0.4	cup eq.	183.9475	73.5789929	26856.33242
Fruits					
Citrus	0.2	cup eq.	162.4571	32.4914191	11859.36796
Juice	0.4	cup eq.	135.4902	54.1960631	19781.56303
Other	0.5	cup eq.	183.6407	91.8203589	33514.431
Grains					
Total	6.3	Oz eq.	118.1499	744.344173	271685.6231
Dairy					
Milk	1	cup eq.	228.1564	228.156392	83277.083
Cheese	0.7	cup eq.	214.2764	149.99351	54747.6311
Yoghurt	0.1	cup eq.	205.6962	20.5696203	7507.911392
Protein					
Meat	2.5	Oz eq.	103.8546	259.636553	94767.34176
Poultry	1.5	Oz eq.	81.64011	122.46017	44697.96204
Eggs	0.5	Oz eq.	131.8681	65.9340659	24065.93407
Legumes	0.5	Oz eq.	66.79709	33.3985472	12190.46974
Nuts	0.6	Oz eq.	97.16419	58.2985134	21278.9574
Seafood	0.5	Oz eq.	52.43769	26.2188457	9569.878694
Fats and Oils					
Total	56.8	g	14.63028	830.99974	303314.905
Sugars					
Total	268	kcal	1.696806	454.744058	165981.5812
Beverages					
Total*	-	-	-	447	163230

* NHANES does not overtly track the kilo calories consumed through beverages. Estimated here as the difference between the US average total available kilo calories daily according to LAFA data for 2010[2] (3769 kcal) and the sum of the food/juice intake estimated here.

Total GWP and land use impacts for US final demands were taken from the EXIOBASE v2.3 default final demand vector which represents consumption for the year 2007 (www.exiobase.eu). This only accounts for impacts for production, excluding final transport to the consumer. To account for transport impacts, the transport margins are taken from the EXIOBASE data for each product and multiplied by the final demands vector to generate the resulting final transport needs for each good in 2007 USD. The modal share is then taken from the United States Commodity Flow Survey for the year 2007[4] using best judgement to link EXIOBASE products to the commodity groups covered in the survey. Table S21 displays the transport margins and modal share for each EXIOBASE product we include.

Table S21. Transport margins and modal shares for EXIOBASE products

EXIOBASE Code	**Transpor**	**Modal**	**Modal**	**Modal**	**Modal**

	t Margin	Share – Road	Share - Rail	Share - Water	Share – Air
Paddy rice	0.115998	0.488242	0.363942	0.147816	0
Wheat	0.09643	0.488242	0.363942	0.147816	0
Cereal grains nec	0.10418	0.488242	0.363942	0.147816	0
Vegetables, fruit, nuts	0.131523	0.913812	0.043094	0.043094	0
Oil seeds	0	0.913812	0.043094	0.043094	0
Sugar cane, sugar beet	0	0.913812	0.043094	0.043094	0
Crops nec	0	0.913812	0.043094	0.043094	0
Cattle	0.006992	1	0	0	0
Pigs	0	1	0	0	0
Poultry	0.034421	1	0	0	0
Meat animals nec	0	1	0	0	0
Animal products nec	0.026805	0.991251	0.005661	0.003088	0.004117
Raw milk	0	0.936752	0.063248	0	0
Fish and other fishing products; services incidental of fishing (05)	0.061087	1	0	0	0
Products of meat cattle	0.053237	1	0	0	0
Products of meat pigs	0.063048	1	0	0	0
Products of meat poultry	0.054744	1	0	0	0
Meat products nec	0.077171	1	0	0	0
products of Vegetable oils and fats	0.040329	0.972208	0.025695	0.002098	0.001049
Dairy products	0.07116	0.936752	0.063248	0	0
Processed rice	0.063115	0.969017	0.029915	0.001068	0
Sugar	0.065962	0.972208	0.025695	0.002098	0.001049
Food products nec	0.077061	0.972208	0.025695	0.002098	0.001049
Beverages	0.101754	0.966173	0.033827	0	0
Fish products	0.08422	1	0	0	0

All transport is modeled using GWP and land use intensities for the US economy. The transport processes used here and their environmental intensities as taken from EXIOBASE are shown in Table S22.

Table S22. GWP and land use intensities for different transport modes				
EXIOBASE Code	**Country**	**Mode**	**GWP Intensity (kg $CO_2e/10^6$ EUR)**	**Land Use Intensity ($km^2/10^6$ EUR)**
Railway transportation services	US	Rail	1412976.522	1.256751515
Other land transportation services	US	Truck	778740.6385	0.26280198
Sea and coastal water transportation services	US	Water	3476526.152	0.536899867
Air transport services	US	Air	2832246.05	0.309460771

Finally the MRIO calculations are performed; yielding the total production and transport related impacts related to US consumption for the year 2007. Table S23 outlines the results of the MRIO manipulations.

Table S23. EXIOBASE results for 2007 US final consumption				
	Production		**Transport**	
EXIOBASE Code	**GWP (kg CO_2e)**	**Land Use (km^2)**	**GWP (kg CO_2e)**	**Land Use (km^2)**
Poultry	5822816965	22376.74	80842374	27.28191
Products of meat poultry	32860973488	89477.22	9.56E+08	322.5918
Cattle	588291879	1731.563	402182.3	0.135725
Products of meat cattle	149156495799	461445	8.89E+08	299.8428

Products of meat pigs	17356597166	71345.54	6.9E+08	232.879
Pigs	0	0	0	0
Fish and other fishing products; services incidental of fishing	1267172044	1974.987	54883840	18.52168
Fish products	3860016147	8001.427	2.19E+08	73.96871
Meat products nec	7465819719	15791	3.94E+08	132.8264
Animal products nec	5768762921	32870.85	1.03E+08	34.80589
Dairy products	50604671333	225524.6	1.73E+09	686.0962
Processed rice	1615329595	3673.16	32217730	11.78406
Paddy rice	188415433	874.5804	12077810	5.703473
Cereal grains nec	3150569995	14125.08	77873089	36.77381
Wheat	1849616004	15205.19	50254915	23.73175
Products of Vegetable oils and fats	8879027984	63894.03	1.13E+08	40.58251
Vegetables, fruit, nuts	32863666424	355058.8	4.14E+09	1424.003
Sugar	3131171454	10810.13	99482266	35.80383
Beverages	80757542526	231330.2	5.48E+09	2031.049
Crops nec	955989660	58753.4	0	0
Food products nec	270641831417	1194618	9.84E+09	3541.019

Allocating global warming potential (GWP) and land use impacts from the EXIOBASE is done through a concordance matrix matching nutritional groups to relevant product groups. Concordances are made based on the descriptions provided in the United Nations International Standard Industrial Classification of All Economic Activities classification codes[5]. The total impact from US final demand in 2007 in each relevant EXIOBASE product is divided amongst the kilo calories for all nutritional components ascribed to that product. Letting I_j represent the total impacts (production and transport) from final demand for EXIOBASE food product j, and C_i the total kilo calories produced of nutritional category *i*, then the impacts of the EXIOBASE product *j* attributed to supplying a single kilo calorie of nutritional category *x*, $i_{x,j}$, is given by equation (1), where the denominator is the sum of kilo calories from all nutritional categories linked to that EXIOBASE product.

(1) $i_{x,j} = \frac{I_j}{\sum_{i=1}^{n} C_i}$

A single nutritional category could be matched to multiple EXIOBASE products, and hence, embodied impact per kilo calorie delivered in a nutritional category, i_x, is the sum of the components from each EXIOBASE product assigned to it, according to (2).

(2) $i_x = \sum_{j=1}^{n} i_{x,j}$

Table S24 shows the concordance between different EXIOBASE products and the nutritional categories. Table S25 shows the embodied GWP and land use impacts per kilo calorie nutritional category produced.

Table S24. Concordance matrix between EXIOBASE products and nutritional categories

EXIOBASE Code	**USDA Nutritional Category**
Poultry	Poultry, Eggs
Products of meat poultry	Poultry
Cattle	Meat
Products of meat cattle	Meat
Products of meat pigs	Meat
Pigs	Meat

Fish and other fishing products; services incidental of fishing	Fish and Seafood
Fish products	Fish and Seafood
Meat products nec	Meat
Animal products nec	Poultry, Meat, Fish and Seafood, Milk, Cheese, Yoghurt, Eggs
Dairy products	Milk, Cheese, Yoghurt
Processed rice	Grains
Paddy rice	Grains
Cereal grains nec	Grains
Wheat	Grains
Products of Vegetable oils and fats	Fars and Oils
Vegetables, fruit, nuts	Dark Green Vegetables, Red and Orange Vegetables, Starchy Vegetables, Citrus Fruits, Juice, Other Fruits, Nuts
Sugar	Sugars
Beverages	Beverages, Milk, Juice
Crops nec	Other Vegetables, Starchy Vegetables
Food products nec	Dark Green Vegetables, Red and Orange Vegetables, Starchy Vegetables, Citrus Fruits, Juice, Other Fruits, Nuts, Poultry, Meat, Fish and Seafood, Milk, Cheese, Yoghurt, Eggs, Grains, Legumes, Sugars, Beverages

Table S25. GWP and land use impacts per kilo calorie produced

USDA Nutritional Category	**GWP (kg CO_2e/kcal produced) - production**	**GWP (kg CO_2e/ kcal produced) - transport**	**Land Use (km^2/ kcal produced) - production**	**Land Use (km^2/ kcal produced) - transport**
Poultry	0.003409	9.90E-05	1.09E-08	3.39E-11
Citrus, melons, berries	0.001418	0.00012	1.12E-08	4.18E-11
Other Fruits	0.001418	0.00012	1.12E-08	4.18E-11
Meat	0.006777	9.32E-05	2.23E-08	3.20E-11
Grains	0.000731	2.57E-05	3.27E-09	9.43E-12
Dark Greens	0.001418	0.00012	1.12E-08	4.18E-11
Yoghurt	0.001854	6.37E-05	8.31E-09	2.44E-11
Red and Orange	0.001418	0.00012	1.12E-08	4.18E-11
Sugars	0.00071	2.55E-05	3.08E-09	9.19E-12
Nuts	0.001418	0.00012	1.12E-08	4.18E-11
Cheese	0.001854	6.37E-05	8.31E-09	2.44E-11
Fish and Seafood	0.002473	0.000119	6.64E-09	4.07E-11
Juice	0.002417	0.000188	1.40E-08	6.70E-11
Beverages	0.001647	9.14E-05	5.72E-09	3.36E-11
Starchy	0.001491	0.00012	1.57E-08	4.18E-11
Legumes and Soy	0.000648	2.36E-05	2.86E-09	8.48E-12
Other Vegetables	0.001491	0.00012	1.57E-08	4.18E-11
Fats and Oils	0.000745	2.48E-05	3.56E-09	8.92E-12
Milk	0.002853	0.000132	1.12E-08	4.95E-11
Eggs	0.000987	2.85E-05	4.27E-09	1.01E-11

Determining food related GWP and land use impacts for Boston final consumption

Embodied kilo calories per nutritional serving (Tables S1-19) can be connected with the GWP and land use impacts per kilo calorie nutritional category delivered to market (Table S25) to estimate environmental pressure exerted by different levels of food consumption. We use the NHANES 2007-2010 usual daily intake data for different demographics and US census data to estimate Boston's food related environmental burdens for the year 2010.

Table S26 shows the usual daily intake for different population segments based on sex and age, which when multiplied by 365 provide estimates of annual food demands for US citizens. It should be noted that NHANES, being self-reported, is plagued by underreporting by participants, particularly in foods that have negative health stigmas attached to them (red meat, sugar, highly processed foods, etc.) and is considered at the lower end of food consumption estimates[6]. Notwithstanding the above shortcoming, NHANES provides the most comprehensive and consistent data for US food consumption, with the added benefit of recording including important demographics data, and is therefore chosen here to model Boston's consumption.

Table S26. Usual daily intake for different demographics from NHANES 2007-2010

		Males								**Females**							
		Age								**Age**							
Nutritional Category	**Unit**	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+
Citrus	cup eq.	0.2	0.2	0.2	0.2	0.1	0.2	0.3	0.3	0.2	0.2	0.2	0.1	0.2	0.2	0.3	0.3
Other Fruits	cup eq.	0.6	0.6	0.5	0.4	0.4	0.5	0.6	0.7	0.6	0.5	0.5	0.4	0.4	0.5	0.7	0.7
Juice	cup eq.	0.7	0.5	0.4	0.4	0.4	0.3	0.3	0.4	0.7	0.4	0.4	0.3	0.4	0.2	0.3	0.4
Dark Greens	cup eq.	0	0	0	0.1	0.1	0.1	0.2	0.1	0	0	0	0.1	0.1	0.1	0.2	0.1
Red and Orange	cup eq.	0.2	0.3	0.3	0.4	0.5	0.5	0.4	0.4	0.2	0.2	0.3	0.3	0.4	0.4	0.4	0.4
Other	cup eq.	0.1	0.2	0.2	0.3	0.6	0.6	0.7	0.5	0.2	0.2	0.3	0.3	0.5	0.6	0.7	0.5
Starchy	cup eq.	0.2	0.3	0.4	0.4	0.5	0.5	0.6	0.5	0.2	0.3	0.4	0.4	0.4	0.4	0.4	0.4
Grains	oz eq.	4.1	6.1	7.3	8.2	8.1	7.8	6.9	6	3.7	5.6	6.5	6.1	5.9	5.5	5.1	4.9
Milk	cup eq.	1.9	1.5	1.6	1.5	0.8	0.9	0.9	1.1	1.9	1.5	1.3	0.9	0.7	0.8	0.8	0.9
Cheese	cup eq.	0.4	0.6	0.8	1	1	0.9	0.7	0.4	0.4	0.6	0.6	0.7	0.7	0.6	0.5	0.3
Yoghurt	cup eq.	0.1	0.1	0	0	0.1	0	0.1	0	0.1	0.1	0	0	0.1	0.1	0.1	0.1
Meat	oz eq.	1.2	1.8	2.3	3.2	3.4	3.8	3.3	2.7	1.1	1.7	2	1.6	2.1	2.1	1.9	1.8
Poultry	oz eq.	0.9	1.1	1.5	1.8	2.2	1.9	1.8	1.1	0.8	1	1.2	1.6	1.5	1.4	1.3	1
Eggs	oz eq.	0.4	0.4	0.4	0.4	0.6	0.7	0.7	0.7	0.3	0.3	0.4	0.4	0.4	0.5	0.5	0.5
Legumes	oz eq.	0.2	0.2	0.3	0.4	0.6	0.7	0.6	0.4	0.2	0.2	0.3	0.3	0.4	0.5	0.5	0.3
Nuts	oz eq.	0.3	0.4	0.5	0.4	0.5	0.8	0.9	0.7	0.2	0.3	0.4	0.3	0.4	0.6	0.7	0.5
Fish and Seafood	oz eq.	0.1	0.1	0.2	0.3	0.6	0.7	0.8	0.6	0.1	0.2	0.2	0.2	0.4	0.5	0.6	0.5
Fats and Oils	g	39.1	50.1	59.2	68	67.2	72.1	66.4	55.5	36.6	47	53.4	51.1	50.8	50.3	50.3	44.1
Added Sugars	kcal	9.4	15.7	21.5	24.6	23.5	20.5	16.5	14	8.4	14.3	17.8	17.5	16.7	15.1	12.5	10.9
Beverages*	-	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

* Lacking demographic data for beverage intake, all respondents are assumed to have the same daily calorific intake from beverages

To move from NHANES usual daily intake to annual environmental impacts for a nutritional category, Y_x, the usual daily intake for nutritional component x, UDI_x, is combined with the produced kilo calories per nutritional unit, $kcal_x$, and the impacts per kilo calorie supplied to the market, i_x, and corrected for the number of days in a year:

(3) $$Y_x = (UDI_x \times kcal_x \times i_x) \times 365$$

Tables S27 and S28 show food related GWP and land use impacts for different demographics, respectively.

Table S27. Food related GWP impacts for different demographics in CO_2e/a/cap

	Males								Females							
	Age								Age							
Nutritional Category	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+
Poultry	91.4	111.7	152.4	182.9	223.5	193.0	182.9	111.7	81.3	101.6	121.9	162.5	152.4	142.2	132.1	101.6
Citrus	16.8	16.8	16.8	16.8	8.4	16.8	25.2	25.2	16.8	16.8	16.8	8.4	16.8	16.8	25.2	25.2
Fish and Seafood	4.7	4.7	9.5	14.2	28.4	33.1	37.9	28.4	4.7	9.5	9.5	9.5	18.9	23.7	28.4	23.7
Other Fruits	57.0	57.0	47.5	38.0	38.0	47.5	57.0	66.5	57.0	47.5	47.5	38.0	38.0	47.5	66.5	66.5
Meat	308.3	462.4	590.9	822.1	873.5	976.2	847.8	693.6	282.6	436.7	513.8	411.0	539.5	539.5	488.1	462.4
Grains	129.2	192.2	230.0	258.4	255.2	245.8	217.4	189.1	116.6	176.5	204.8	192.2	185.9	173.3	160.7	154.4
Dark Greens	0.0	0.0	0.0	2.3	2.3	2.3	4.5	2.3	0.0	0.0	0.0	2.3	2.3	2.3	4.5	2.3
Red and Orange	6.8	10.2	10.2	13.6	17.0	17.0	13.6	13.6	6.8	6.8	10.2	10.2	13.6	13.6	13.6	13.6
Sugars	4.1	6.9	9.5	10.8	10.3	9.0	7.3	6.2	3.7	6.3	7.8	7.7	7.3	6.6	5.5	4.8
Nuts	15.1	20.1	25.1	20.1	25.1	40.2	45.3	35.2	10.1	15.1	20.1	15.1	20.1	30.2	35.2	25.1
Milk	451.4	356.4	380.1	356.4	190.1	213.8	213.8	261.3	451.4	356.4	308.8	213.8	166.3	190.1	190.1	213.8
Cheese	58.0	87.0	116.0	145.0	145.0	130.5	101.5	58.0	58.0	87.0	87.0	101.5	101.5	87.0	72.5	43.5
Juice	83.7	59.8	47.8	47.8	47.8	35.9	35.9	47.8	83.7	47.8	47.8	35.9	47.8	23.9	35.9	47.8
Beverages	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9	268.9
Fats and Oils	155.5	199.2	235.4	270.4	267.2	286.7	264.0	220.7	145.5	186.9	212.3	203.2	202.0	200.0	200.0	175.4
Legumes and Soy	3.2	3.2	4.7	6.3	9.5	11.1	9.5	6.3	3.2	3.2	4.7	4.7	6.3	7.9	7.9	4.7
Other Vegetables	4.8	9.6	9.6	14.5	28.9	28.9	33.8	24.1	9.6	9.6	14.5	14.5	24.1	28.9	33.8	24.1
Yoghurt	13.9	13.9	0.0	0.0	13.9	0.0	13.9	0.0	13.9	13.9	0.0	0.0	13.9	13.9	13.9	13.9
Starchy	20.0	30.0	40.0	40.0	50.1	50.1	60.1	50.1	20.0	30.0	40.0	40.0	40.0	40.0	40.0	40.0
Eggs	19.0	19.0	19.0	19.0	28.5	33.2	33.2	33.2	14.2	14.2	19.0	19.0	19.0	23.7	23.7	23.7
Transport	72.5	75.7	82.6	88.7	86.2	89.4	88.3	80.4	70.9	71.7	74.6	67.1	70.2	70.6	73.1	69.7
Total	1784	2004	2296	2636	2617	2729	2561	2222	1719	1906	2030	1825	1955	1950	1919	1805

	.4	.8	.2	.2	.9	.5	.7	.7	.0	.4	.3	.6	.0	.8	.7	.3

Table S28. Food related land use for different demographics in km^2/a/capita

Nutritional Category	Males								Females							
	Age								Age							
	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+
Poultry	0.000301	0.000368	0.000501	0.000602	0.000735	0.000635	0.000602	0.000368	0.000267	0.000334	0.000401	0.000535	0.000501	0.000468	0.000435	0.000334
Citrus	0.000156	0.000156	0.000156	0.000156	7.80E-05	0.000156	0.000234	0.000234	0.000156	0.000156	0.000156	7.80E-05	0.000156	0.000156	0.000234	0.000234
Fish and Seafood	1.34E-05	1.34E-05	2.67E-05	4.01E-05	8.02E-05	9.36E-05	0.000107	8.02E-05	1.34E-05	2.67E-05	2.67E-05	2.67E-05	5.35E-05	6.68E-05	8.02E-05	6.68E-05
Other Fruits	0.000529	0.000529	0.000441	0.000353	0.000353	0.000441	0.000529	0.000617	0.000529	0.000441	0.000441	0.000353	0.000353	0.000441	0.000617	0.000617
Meat	0.001007	0.00151	0.00193	0.002685	0.002853	0.003188	0.002769	0.002265	0.000923	0.001426	0.001678	0.001342	0.001762	0.001762	0.001594	0.00151
Grains	0.000666	0.000991	0.001186	0.001332	0.001316	0.001267	0.001121	0.000975	0.000601	0.00091	0.001056	0.000991	0.000958	0.000893	0.000829	0.000796
Dark Greens	0	0	0	2.09E-05	2.09E-05	2.09E-05	4.19E-05	2.09E-05	0	0	0	2.09E-05	2.09E-05	2.09E-05	4.19E-05	2.09E-05
Red and Orange	6.31E-05	9.47E-05	9.47E-05	0.000126	0.000158	0.000158	0.000126	0.000126	6.31E-05	6.31E-05	9.47E-05	9.47E-05	0.000126	0.000126	0.000126	0.000126
Sugars	1.79E-05	2.99E-05	4.10E-05	4.69E-05	4.48E-05	3.91E-05	3.14E-05	2.67E-05	1.60E-05	2.72E-05	3.39E-05	3.33E-05	3.18E-05	2.88E-05	2.38E-05	2.08E-05
Nuts	3.04E-05	4.06E-05	5.07E-05	4.06E-05	5.07E-05	8.12E-05	9.13E-05	7.10E-05	2.03E-05	3.04E-05	4.06E-05	3.04E-05	4.06E-05	6.09E-05	7.10E-05	5.07E-05
Milk	0.001767	0.001395	0.001488	0.001395	0.000744	0.000837	0.000837	0.001023	0.001767	0.001395	0.001209	0.000837	0.000651	0.000744	0.000744	0.000837
Cheese	0.00026	0.00039	0.00052	0.00065	0.00065	0.000585	0.000455	0.00026	0.00026	0.00039	0.00039	0.000455	0.000455	0.00039	0.000325	0.000195
Juice	0.000555	0.000396	0.000317	0.000317	0.000317	0.000238	0.000238	0.000317	0.000555	0.000317	0.000317	0.000238	0.000317	0.000158	0.000238	0.000317
Beverages	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934	0.000934
Fats and	0.000	0.000	0.001	0.001	0.001	0.001	0.001	0.001	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000

Oils	742	951	124	291	276	369	261	054	695	892	014	97	964	955	955	837
Legumes and Soy	1.40E-05	1.40E-05	2.09E-05	2.79E-05	4.19E-05	4.88E-05	4.19E-05	2.79E-05	1.40E-05	1.40E-05	2.09E-05	2.09E-05	2.79E-05	3.49E-05	3.49E-05	2.09E-05
Other Vegetables	4.26E-05	8.51E-05	8.51E-05	0.0000128	0.0000255	0.0000255	0.0000298	0.0000213	8.51E-05	8.51E-05	0.0000128	0.0000128	0.0000213	0.0000255	0.0000298	0.0000213
Yoghurt	6.24E-05	6.24E-05	0	0	6.24E-05	0	6.24E-05	0	6.24E-05	6.24E-05	0	0	6.24E-05	6.24E-05	6.24E-05	6.24E-05
Starchy	0.0000177	0.0000265	0.0000353	0.0000353	0.0000442	0.0000442	0.000053	0.0000442	0.0000177	0.0000265	0.0000353	0.0000353	0.0000353	0.0000353	0.0000353	0.0000353
Eggs	8.23E-05	8.23E-05	8.23E-05	8.23E-05	0.0000123	0.0000144	0.0000144	0.0000144	6.17E-05	6.17E-05	8.23E-05	8.23E-05	8.23E-05	0.0000103	0.0000103	0.0000103
Transport	2.67E-05	2.76E-05	2.99E-05	3.22E-05	3.11E-05	3.18E-05	3.14E-05	2.88E-05	2.62E-05	2.62E-05	2.71E-05	2.45E-05	2.55E-05	2.54E-05	2.63E-05	2.53E-05
Total	0.007446	0.008335	0.009382	0.010613	0.010566	0.010964	0.010484	0.009227	0.007226	0.007858	0.008403	0.007548	0.008089	0.00804	0.008125	0.007675

Census data are taken from American Fact Finder at the block-group level[7]. These data provide population based on sex and age group. The age groups in the census data do not precisely align with those in NHANES, so concordance was made based on best judgement, as shown in Table S29. Census data is also adjusted for incarcerated population since their usual daily intakes are likely not well represented by NHANES. This means subtracting 1418 adults (taken from age groups based on proportion of unaltered population) from block group ‘250250801001’ as it contains the Suffolk County Correctional Facility[8].

Table S29. Concordance between NHANES and US Census age groups

NHANES age group	Census age groups
1-3	‘under 5 years’
4-8	‘5 to 9 years’
9-13	’10 to 14 years’
14-18	’15 to 17 years’, ’18 and 19 years’
19-30	’20 years’, ’21 years’, ’22-24 years’, ’25-29 years’
31-50	’30-34 years’, ’35-39 years’, ’40-44 years’, ’45-49 years’
51-70	’50-54 years’, ’55-59 years’, ’60 and 61 years’, ’62 to 64 years’, ’65 and 66 years’, ‘67 to 69 years’
71+	’70 to 74 years’, ’75 to 79 years’, ’80 to 84 years’, ‘85+ years’

With the block-group demographics data in hand and estimated environmental burdens for the different age groups and sexes, Boston’s food related environmental impacts are calculated for the 560 block-groups that comprise the city. Figures 1-2 show the estimated GWP impacts and land use for Boston’s food consumption for the year 2010.

Figure S1. 2010 food-borne GHG emissions for Boston block-groups (white indicates uninhabited block groups devoid of impacts)

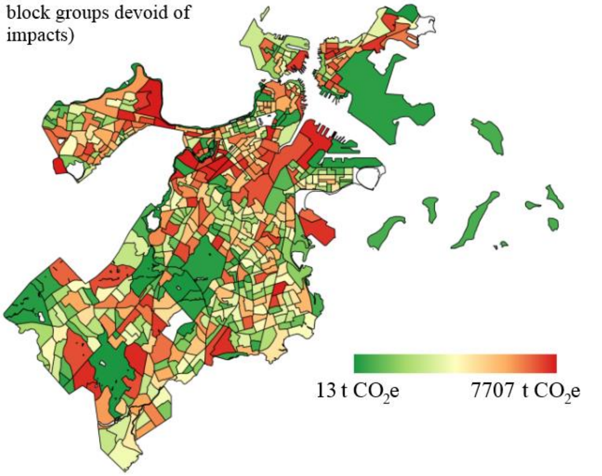
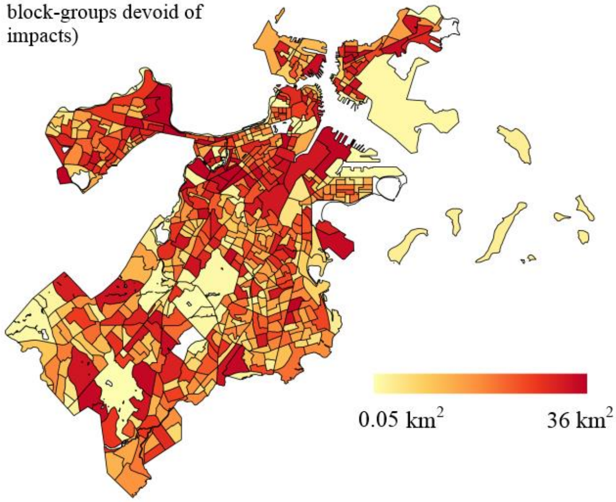


Figure S2. 2010 food-borne land use for Boston block-groups (white indicates uninhabited block-groups devoid of impacts)



Life Cycle Inventories and LCA for Urban Agriculture

Life cycle inventories (LCI) for the urban farms build upon those from an earlier study of farms growing tomato and lettuce in Boston in New York City[9]. Of the six farms covering five UF forms from the earlier study, only three of the farms and two farm-types are used in this study, for a number of reasons:

- They produced the widest variety of crops, useful when modelling city-wide impacts of UF (difficult to model a city only consuming tomatoes)
- They represent the predominant UF forms in the study region at the time of publishing: open plots and rooftop farms (see Figure 3 for examples of each). See Goldstein et al. (2016)[10] for more information about the nuances between UF types and their divergent environmental performance.
    - Open plots typically low-tech operations, growing crops directly in local overburden or raised beds
    - Rooftop farms are identical in most respects to green roofs with the exception that they grow edible crops. Soil depth is typically equal to greater than 12", and hence, rooftop farms qualify as intensive green roofs.
- Have superior environmental performance than conventional agriculture for some foods and by some metrics, as opposed to the other forms which had higher environmental intensities compared to conventional UF[9]. Although this skews the results in UF's favor, it is useful in a hypothetical study of large scale urban design to quantify the potential best-case, hypothetical benefits of UF. Additionally, since UF is not universally preferable to conventional produce, this will still provide opportunities to discuss trade-offs when adapting UF.

Figure 3 – Open rooftop farm (left) and open lot farm (right). Authors own photographs.

The attributes of the utilized UF systems are outline in Table S30.

Table S30. Urban farm characteristics and crops

Location	Farm	Farm Type	Area (m^2)	For profit?	Crops
Boston, MA	1	open plot	560	No	tomato, bell pepper, eggplant, lettuce*
Boston, MA	2	open rooftop	1469	Yes	turnip, tomato, scallion, radish, bell pepper, lettuce, kale, cucumber, carrot, green bean
New York City	3	open plot	1269	No	turnip, tomato, squash, scallions, bell pepper, lettuce, kale, cucumber, collard greens, carrot, cabbage, beet, green bean

* Technically 'arugula' but assumed lettuce here since it performs the same function as lettuce (salad greens, sandwich topping, etc.)

Process-based LCA methodology is applied here. The LCA scope is production of crops and distribution to final consumers – in line with the MRIO model used to assess city-wide impacts. Where by-products occur, system expansion is applied to credit the urban farm in accordance with the ISO 14040 family[11]. The ecoinvent database version 3.2 was used to provide data on background processes and to perform the life cycle impact assessment for the different foods. Primary data was collected over the 2015 growing season. Tables S31-33 outline the Life Cycle Inventories to produce 1 kilogram of different crops from the modeled farms.

Table S31. Life Cycle Inventories per kilogram crop from farm 1

	Unit	Tomato	Bell Pepper	Eggplant	Arugula
Materials and Energy Inputs					
Capital					
Concrete, normal {US-NPCC}| production | Conseq, U	m^3	2.07E-05	3.64E-05	3.35E-05	1.25E-04
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	1.24E-02	2.19E-02	2.02E-02	7.50E-02
Occupation, urban, continuously built	m^2	2.53E-01	4.46E-01	4.10E-01	1.52E+00
Polyethylene, high density, granulate {GLO}| market for | Conseq, U	kg	1.17E-03	2.06E-03	1.89E-03	7.04E-03
Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	m^3	4.08E-05	7.19E-05	6.61E-05	2.46E-04
Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	kg	6.76E-04	1.19E-03	1.09E-03	4.08E-03
Synthetic rubber {GLO}| market for | Conseq, U	kg	8.35E-04	1.47E-03	1.35E-03	5.04E-03
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.70E-01	3.00E-01	2.76E-01	1.03E+00
Wood chips, wet, measured as dry mass {RoW}| market for | Conseq, U	m^3	1.85E-01	3.26E-01	2.99E-01	1.11E+00
Operations					
Polyethylene, low density, granulate {GLO}| market for | Conseq, U	kg	3.49E-03	6.15E-03	5.66E-03	2.11E-02
Polypropylene, granulate {GLO}| market for | Conseq, U	kg	6.95E-03	1.23E-02	1.13E-02	4.19E-02
Tap water {US-Boston}| market for | Conseq, U	m^3	7.77E-02	1.31E-01	1.02E-01	1.28E-01
Transport, passenger car, large size, petrol, EURO 4 {RER}| transport, passenger car, large size, petrol, EURO 4 | Conseq, U	km	4.91E-02	8.66E-02	7.96E-02	2.96E-01
Waste					
Inert waste, for final disposal {GLO}| market for | Conseq, U	kg	1.27E-04	2.24E-04	2.06E-04	7.65E-04
Inert waste, for final disposal {US}| market for | Conseq, U	kg	1.36E-02	2.39E-02	2.20E-02	8.19E-02
PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	kg	3.64E-03	6.41E-03	5.89E-03	2.19E-02
Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	kg	7.73E-04	1.36E-03	1.25E-03	4.66E-03
Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	kg	3.38E-04	5.96E-04	5.47E-04	2.04E-03
Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	kg	4.57E-02	8.05E-02	7.40E-02	2.75E-01
Waste wood, post-consumer {GLO}| market for | Conseq, U	kg	1.30E-02	2.29E-02	2.10E-02	7.83E-02

Table S32. Life Cycle Inventories per kg crop for farm 2

	Unit	Turnip	Tomato	Scallion	Radish	Bell Pepper
Materials and Energy Inputs						
Capital						
Aluminium, primary, ingot {US}| market for | Conseq, U	kg	2.0E-06	7.8E-07	3.9E-06	1.2E-06	1.5E-06
Copper {GLO}| market for | Conseq, U	kg	6.4E-06	2.5E-06	1.3E-05	3.8E-06	4.9E-06
Crushed gravel {US-Boston} | market for | conseq, U	kg	3.2E-01	1.2E-01	6.3E-01	1.9E-01	2.4E-01
Expanded clay {US-Boston} | Market for | Conseq, U	kg	2.0E+00	7.7E-01	3.9E+00	1.2E+00	1.5E+00

Expanded shale {US-Boston} | Market for | Conseq, U	kg	1.8E-01	7.1E-02	3.6E-01	1.1E-01	1.4E-01
Extrusion, plastic film {US-MRO}| production | Conseq, U	kg	4.1E-02	1.6E-02	8.1E-02	2.4E-02	3.1E-02
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	2.4E-02	9.3E-03	4.7E-02	1.4E-02	1.8E-02
Extrusion, plastic pipes {US-NPCC}| production | Conseq, U	kg	1.0E-03	4.1E-04	2.1E-03	6.1E-04	7.9E-04
Glass, for liquid crystal display {GLO}| production | Conseq, U	kg	1.4E-07	5.6E-08	2.8E-07	8.4E-08	1.1E-07
Nylon 6 {GLO}| market for | Conseq, U	kg	1.1E-05	4.1E-06	2.1E-05	6.2E-06	7.9E-06
Polyethylene, high density, granulate {GLO}| market for | Conseq, U	kg	3.3E-02	1.3E-02	6.5E-02	1.9E-02	2.5E-02
Polypropylene, granulate {GLO}| market for | Conseq, U	kg	1.1E-02	4.2E-03	2.1E-02	6.4E-03	8.2E-03
Steel, low-alloyed, hot rolled {US-MRO}| market for | Conseq, U	kg	1.7E-03	6.5E-04	3.3E-03	9.8E-04	1.2E-03
Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	kg	4.8E-01	1.9E-01	9.6E-01	2.9E-01	3.7E-01
Steel, low-alloyed, hot rolled {US-WECC}| market for | Conseq, U	kg	9.7E-05	3.8E-05	1.9E-04	5.7E-05	7.3E-05
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.1E-01	4.3E-02	2.2E-01	6.5E-02	8.3E-02
Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.9E-02	7.5E-03	3.8E-02	1.1E-02	1.4E-02
Wire drawing, copper {US-WECC}| processing | Conseq, U	kg	6.4E-06	2.5E-06	1.3E-05	3.8E-06	4.9E-06
Operations						
Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	kg	1.2E-03	4.7E-04	2.4E-03	7.1E-04	9.1E-04
Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	kg	9.4E-04	3.7E-04	1.9E-03	5.5E-04	7.1E-04
Potassium nitrate {GLO}| market for | Conseq, U	kg	6.5E-04	2.5E-04	1.3E-03	3.8E-04	4.9E-04
Transport, passenger car, small size, petrol, EURO 5 {GLO}| market for | Conseq, U	km	1.2E+00	4.7E-01	2.4E+00	7.1E-01	9.1E-01
Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	MJ	6.0E-02	2.3E-02	1.2E-01	3.5E-02	4.5E-02
Basalt {GLO}| market for | Conseq, U	kg	5.0E-02	2.0E-02	1.0E-01	3.0E-02	3.8E-02
compost {US-NPCC} | at farm | conseq, U	kg	4.3E-01	1.7E-01	8.4E-01	2.5E-01	3.2E-01
garden waste treatment {US-NPCC} | at farm | conseq, U	kg	1.3E-01	4.9E-02	2.5E-01	7.5E-02	9.5E-02
Tap water {US-Boston}| market for | Conseq, U	m^3	1.4E-02	5.6E-03	2.9E-02	8.5E-03	1.1E-02
Waste						
Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	kg	8.9E-07	3.5E-07	1.8E-06	5.3E-07	6.8E-07
Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	kg	8.9E-07	3.5E-07	1.8E-06	5.3E-07	6.8E-07
Inert waste, for final disposal {US}| market for | Conseq, U	kg	2.5E+00	9.7E-01	4.9E+00	1.5E+00	1.9E+00
PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	kg	5.2E-02	2.0E-02	1.0E-01	3.1E-02	3.9E-02
PP (waste treatment) {US-NPCC}| recycling of PP | Conseq, U	kg	1.0E-02	4.0E-03	2.0E-02	6.0E-03	7.7E-03
Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	kg	4.7E-01	1.9E-01	9.4E-01	2.8E-01	3.6E-01

Table S32 contd. Life Cycle Inventories per kg crop from farm 2

	Unit	**Lettuce**	**Kale**	**Cucum bers**	**Carrots**	**Green Bean**
Materials and Energy Inputs						
Capital						
Aluminium, primary, ingot {US}| market for | Conseq, U	kg	2.6E-06	2.9E-06	9.1E-07	2.3E-06	3.3E-06
Copper {GLO}| market for | Conseq, U	kg	8.5E-06	9.4E-06	2.9E-06	7.5E-06	1.1E-05
Crushed gravel {US-Boston} | market for |	kg	4.2E-01	4.6E-01	1.4E-01	3.7E-01	5.2E-01

conseq, U						
Expanded clay {US-Boston} | Market for | Conseq, U	kg	2.6E+00	2.9E+00	9.0E-01	2.3E+00	3.2E+00
Expanded shale {US-Boston} | Market for | Conseq, U	kg	2.4E-01	2.6E-01	8.3E-02	2.1E-01	3.0E-01
Extrusion, plastic film {US-MRO}| production | Conseq, U	kg	5.4E-02	5.9E-02	1.9E-02	4.7E-02	6.7E-02
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	3.1E-02	3.4E-02	1.1E-02	2.7E-02	3.9E-02
Extrusion, plastic pipes {US-NPCC}| production | Conseq, U	kg	1.4E-03	1.5E-03	4.7E-04	1.2E-03	1.7E-03
Glass, for liquid crystal display {GLO}| production | Conseq, U	kg	1.9E-07	2.1E-07	6.5E-08	1.7E-07	2.4E-07
Nylon 6 {GLO}| market for | Conseq, U	kg	1.4E-05	1.5E-05	4.8E-06	1.2E-05	1.7E-05
Polyethylene, high density, granulate {GLO}| market for | Conseq, U	kg	4.3E-02	4.8E-02	1.5E-02	3.8E-02	5.4E-02
Polypropylene, granulate {GLO}| market for | Conseq, U	kg	1.4E-02	1.6E-02	5.0E-03	1.3E-02	1.8E-02
Steel, low-alloyed, hot rolled {US-MRO}| market for | Conseq, U	kg	2.2E-03	2.4E-03	7.5E-04	1.9E-03	2.7E-03
Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	kg	6.4E-01	7.0E-01	2.2E-01	5.6E-01	8.0E-01
Steel, low-alloyed, hot rolled {US-WECC}| market for | Conseq, U	kg	1.3E-04	1.4E-04	4.4E-05	1.1E-04	1.6E-04
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.5E-01	1.6E-01	5.0E-02	1.3E-01	1.8E-01
Transport, freight, lorry 16-32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	2.5E-02	2.8E-02	8.7E-03	2.2E-02	3.1E-02
Wire drawing, copper {US-WECC}| processing | Conseq, U	kg	8.5E-06	9.4E-06	2.9E-06	7.5E-06	1.1E-05
Operations						
Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	kg	1.2E-03	1.4E-03	4.3E-04	1.1E-03	1.5E-03
Basalt {GLO}| market for | Conseq, U	kg	6.7E-02	7.3E-02	2.3E-02	5.9E-02	8.3E-02
compost {US-NPCC} | at farm | conseq, U	kg	5.6E-01	6.2E-01	1.9E-01	4.9E-01	7.0E-01
Electricity, low voltage, 2012-2040 average {NPCC, US only}| market for | Conseq, U	MJ	7.9E-02	8.7E-02	2.7E-02	6.9E-02	9.8E-02
garden waste treatment {US-NPCC} | at farm | conseq, U	kg	1.7E-01	1.8E-01	5.8E-02	1.5E-01	2.1E-01
Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	kg	1.6E-03	1.8E-03	5.5E-04	1.4E-03	2.0E-03
Potassium nitrate {GLO}| market for | Conseq, U	kg	8.6E-04	9.5E-04	3.0E-04	7.6E-04	1.1E-03
Tap water {US-Boston}| market for | Conseq, U	m^3	1.9E-02	2.1E-02	6.6E-03	1.7E-02	2.4E-02
Transport, passenger car, small size, petrol, EURO 5 {GLO}| market for | Conseq, U	km	1.6E+00	1.8E+00	5.5E-01	1.4E+00	2.0E+00
Waste						
Aluminium (waste treatment) {US-NPCC}| recycling of aluminium | Conseq, U	kg	1.2E-06	1.3E-06	4.1E-07	1.0E-06	1.5E-06
Copper (waste treatment) {US-NPCC}| recycling of copper | Conseq, U	kg	1.2E-06	1.3E-06	4.1E-07	1.0E-06	1.5E-06
Inert waste, for final disposal {US}| market for | Conseq, U	kg	3.3E+00	3.6E+00	1.1E+00	2.9E+00	4.1E+00
PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	kg	6.9E-02	7.6E-02	2.4E-02	6.0E-02	8.5E-02
PP (waste treatment) {US-NPCC}| recycling of PP | Conseq, U	kg	1.4E-02	1.5E-02	4.7E-03	1.2E-02	1.7E-02
Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	kg	6.3E-01	6.9E-01	2.2E-01	5.5E-01	7.8E-01

Table S33. Life Cycle Inventories per kg crop from farm 3

	Unit	Turnip	Tomato	Squash	Bell Pepper	Lettuce
Material and Energy Inputs						
Capital						
Concrete, normal {US-NPCC}| production |	m^3	1.2E-05	9.5E-06	1.6E-05	1.6E-05	5.0E-05

Conseq, U						
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	5.1E-03	4.1E-03	7.0E-03	7.0E-03	2.2E-02
Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	kg	5.6E-03	4.6E-03	7.8E-03	7.8E-03	2.4E-02
Polyethylene, low density, granulate {GLO}| market for | Conseq, U	kg	3.6E-03	3.0E-03	5.0E-03	5.1E-03	1.6E-02
Polypropylene, granulate {GLO}| market for | Conseq, U	kg	4.3E-03	3.5E-03	5.9E-03	6.0E-03	1.9E-02
Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	kg	2.7E-03	2.2E-03	3.7E-03	3.7E-03	1.1E-02
Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	m^3	4.1E-05	3.4E-05	5.7E-05	5.7E-05	1.8E-04
Steel, low-alloyed {GLO}| market for | Conseq, U	kg	7.4E-04	6.1E-04	1.0E-03	1.0E-03	3.2E-03
Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	kg	2.1E-03	1.7E-03	2.9E-03	3.0E-03	9.2E-03
Straw {GLO}| market for | Conseq, U	kg	4.1E-02	3.4E-02	5.7E-02	5.7E-02	1.8E-01
Synthetic rubber {GLO}| market for | Conseq, U	kg	1.2E-04	9.7E-05	1.6E-04	1.6E-04	5.1E-04
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	6.5E-02	5.3E-02	8.9E-02	9.0E-02	2.8E-01
Operations						
Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	kg	8.2E-04	6.7E-04	1.1E-03	1.1E-03	3.5E-03
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	7.0E-04	7.0E-04	7.0E-04	7.1E-04	6.0E-04
Occupation, urban, continuously built	m^2a	2.9E-01	2.4E-01	4.0E-01	4.0E-01	1.2E+00
Petrol, unleaded {RoW}| market for | Conseq, U	kg	7.4E-04	6.1E-04	1.0E-03	1.0E-03	3.2E-03
Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	kg	3.3E-04	2.7E-04	4.5E-04	4.6E-04	1.4E-03
Polyethylene, high density, granulate {GLO}| market for | Conseq, U	kg	7.0E-04	7.0E-04	7.0E-04	7.1E-04	6.0E-04
Potassium sulfate, as K2O {GLO}| market for | Conseq, U	kg	1.0E-03	8.2E-04	1.4E-03	1.4E-03	4.4E-03
Tap water {US-Boston}| market for | Conseq, U	ton	3.4E-01	1.6E-01	3.3E-01	3.3E-01	3.1E-01
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.2E-02	9.6E-03	1.6E-02	1.6E-02	5.1E-02
Transport, passenger car, large size, petrol, EURO 4 {GLO}| market for transport, passenger car, large size, petol, EURO 4 | Conseq, U	km	3.7E-02	3.1E-02	5.1E-02	5.2E-02	1.6E-01
Transport, passenger car, large size, petrol, EURO 4 {RER}| transport, passenger car, large size, petrol, EURO 4 | Conseq, U	km	2.5E-02	2.5E-02	2.5E-02	2.5E-02	2.1E-02
Direct Emissions						
Carbon dioxide, fossil	kg	2.5E-03	2.1E-03	3.5E-03	3.5E-03	1.1E-02
Waste						
Inert waste, for final disposal {US}| market for | Conseq, U*	kg	-1.8E-01	-1.5E-01	-2.5E-01	-2.5E-01	-7.7E-01
PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	kg	4.1E-03	3.5E-03	5.4E-03	5.4E-03	1.5E-02
PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	kg	2.5E-03	2.1E-03	3.5E-03	3.5E-03	1.1E-02
Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	kg	5.9E-05	4.8E-05	8.2E-05	8.2E-05	2.6E-04
Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	kg	1.8E-03	1.4E-03	2.4E-03	2.5E-03	7.6E-03
Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	kg	2.6E-02	2.1E-02	3.5E-02	3.6E-02	1.1E-01
Waste wood, post-consumer {GLO}| market for | Conseq, U	kg	1.9E-02	1.6E-02	2.6E-02	2.7E-02	8.3E-02

* Negative number due to avoided waste from the use of used jute bags for ground cover

Table S33 contd. Life Cycle Inventories per kg crop from farm 3						
	Unit	**Kale**	**Cucumber**	**Collard Greens**	**Carrot**	**Cabbage**
Material and Energy Inputs						

Capital						
Concrete, normal {US-NPCC}| production | Conseq, U	m^3	8.6E-06	1.2E-05	9.8E-05	2.5E-05	8.6E-06
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	3.7E-03	5.3E-03	4.3E-02	1.1E-02	3.8E-03
Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	kg	4.2E-03	5.9E-03	4.8E-02	1.2E-02	4.2E-03
Polyethylene, low density, granulate {GLO}| market for | Conseq, U	kg	2.7E-03	3.8E-03	3.1E-02	7.8E-03	2.7E-03
Polypropylene, granulate {GLO}| market for | Conseq, U	kg	3.2E-03	4.5E-03	3.6E-02	9.2E-03	3.2E-03
Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	kg	2.0E-03	2.8E-03	2.2E-02	5.7E-03	2.0E-03
Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	m^3	3.1E-05	4.3E-05	3.5E-04	8.9E-05	3.1E-05
Steel, low-alloyed {GLO}| market for | Conseq, U	kg	5.5E-04	7.8E-04	6.3E-03	1.6E-03	5.5E-04
Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	kg	1.6E-03	2.2E-03	1.8E-02	4.6E-03	1.6E-03
Straw {GLO}| market for | Conseq, U	kg	3.0E-02	4.3E-02	3.5E-01	8.8E-02	3.1E-02
Synthetic rubber {GLO}| market for | Conseq, U	kg	8.8E-05	1.2E-04	1.0E-03	2.5E-04	8.8E-05
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	4.8E-02	6.8E-02	5.5E-01	1.4E-01	4.8E-02
Operations						
Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	kg	6.1E-04	8.6E-04	6.9E-03	1.8E-03	6.1E-04
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	7.0E-04	7.2E-04	7.0E-04	7.0E-04	7.0E-04
Occupation, urban, continuously built	m^2a	2.1E-01	3.0E-01	2.4E+00	6.2E-01	2.1E-01
Petrol, unleaded {RoW}| market for | Conseq, U	kg	5.5E-04	7.8E-04	6.3E-03	1.6E-03	5.5E-04
Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	kg	2.4E-04	3.4E-04	2.8E-03	7.1E-04	2.4E-04
Polyethylene, high density, granulate {GLO}| market for | Conseq, U	kg	7.0E-04	7.2E-04	7.0E-04	7.0E-04	7.0E-04
Potassium sulfate, as K2O {GLO}| market for | Conseq, U	kg	7.4E-04	1.1E-03	8.5E-03	2.2E-03	7.5E-04
Tap water {US-Boston}| market for | Conseq, U	ton	5.2E-02	1.3E-01	6.0E-01	7.2E-01	5.3E-02
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	8.7E-03	1.2E-02	9.9E-02	2.5E-02	8.7E-03
Transport, passenger car, large size, petrol, EURO 4 {GLO}| market for transport, passenger car, large size, petol, EURO 4 | Conseq, U	km	2.8E-02	3.9E-02	3.2E-01	8.0E-02	2.8E-02
Transport, passenger car, large size, petrol, EURO 4 {RER}| transport, passenger car, large size, petrol, EURO 4 | Conseq, U	km	2.5E-02	2.6E-02	2.5E-02	2.5E-02	2.5E-02
Direct Emissions						
Carbon dioxide, fossil	kg	1.9E-03	2.6E-03	2.1E-02	5.4E-03	1.9E-03
Waste						
Inert waste, for final disposal {US}| market for | Conseq, U*	kg	-1.3E-01	-1.9E-01	-1.5E+00	-3.8E-01	-1.3E-01
PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	kg	3.2E-03	4.3E-03	3.0E-02	8.0E-03	3.2E-03
PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	kg	1.9E-03	2.6E-03	2.1E-02	5.4E-03	1.9E-03
Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	kg	4.4E-05	6.2E-05	5.0E-04	1.3E-04	4.4E-05
Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	kg	1.3E-03	1.8E-03	1.5E-02	3.8E-03	1.3E-03
Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	kg	1.9E-02	2.7E-02	2.2E-01	5.5E-02	1.9E-02
Waste wood, post-consumer {GLO}| market for | Conseq, U	kg	1.4E-02	2.0E-02	1.6E-01	4.1E-02	1.4E-02

* Negative number due to avoided waste from the use of used jute bags for ground cover

Table S33 contd. Life cycle inventories per kg crop from farm 3				
	Unit	**Beet**	**Green**	**Scallion**

			Bean	
Capital				
Concrete, normal {US-NPCC}| production | Conseq, U	m^3	1.8E-05	9.5E-06	5.3E-05
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	7.8E-03	4.1E-03	2.3E-02
Extrusion, plastic pipes {US-NPCC}| market for | Conseq, U	kg	8.7E-03	4.6E-03	2.6E-02
Polyethylene, low density, granulate {GLO}| market for | Conseq, U	kg	5.6E-03	3.0E-03	1.7E-02
Polypropylene, granulate {GLO}| market for | Conseq, U	kg	6.6E-03	3.5E-03	2.0E-02
Polyvinylchloride, bulk polymerised {GLO}| market for | Conseq, U	kg	4.1E-03	2.2E-03	1.2E-02
Sawnwood, hardwood, air dried, planed {RoW}| market for | Conseq, U	m^3	6.4E-05	3.4E-05	1.9E-04
Steel, low-alloyed {GLO}| market for | Conseq, U	kg	1.1E-03	6.1E-04	3.4E-03
Steel, low-alloyed, hot rolled {US-NPCC}| market for | Conseq, U	kg	3.3E-03	1.7E-03	9.8E-03
Straw {GLO}| market for | Conseq, U	kg	6.4E-02	3.4E-02	1.9E-01
Synthetic rubber {GLO}| market for | Conseq, U	kg	1.8E-04	9.7E-05	5.4E-04
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.0E-01	5.3E-02	3.0E-01
Operations				
Ammonium nitrate, as N {RER}| ammonium nitrate production | Conseq, U	kg	1.3E-03	6.7E-04	3.8E-03
Extrusion, plastic film {US-NPCC}| production | Conseq, U	kg	7.0E-04	7.0E-04	6.8E-04
Occupation, urban, continuously built	m^2a	4.5E-01	2.4E-01	1.3E+00
Petrol, unleaded {RoW}| market for | Conseq, U	kg	1.2E-03	6.1E-04	3.4E-03
Phosphate fertiliser, as P2O5 {GLO}| market for | Conseq, U	kg	5.1E-04	2.7E-04	1.5E-03
Polyethylene, high density, granulate {GLO}| market for | Conseq, U	kg	7.0E-04	7.0E-04	6.8E-04
Potassium sulfate, as K2O {GLO}| market for | Conseq, U	kg	1.6E-03	8.2E-04	4.6E-03
Tap water {US-Boston}| market for | Conseq, U	ton	5.2E-01	2.7E-02	4.5E-02
Transport, freight, lorry >32 metric ton, EURO4 {GLO}| market for | Conseq, U	tkm	1.8E-02	9.6E-03	5.4E-02
Transport, passenger car, large size, petrol, EURO 4 {GLO}| market for transport, passenger car, large size, petol, EURO 4 | Conseq, U	km	5.8E-02	3.1E-02	1.7E-01
Transport, passenger car, large size, petrol, EURO 4 {RER}| transport, passenger car, large size, petrol, EURO 4 | Conseq, U	km	2.5E-02	2.5E-02	2.4E-02
Direct Emissions				
Carbon dioxide, fossil	kg	3.9E-03	2.1E-03	1.2E-02
Waste				
Inert waste, for final disposal {US}| market for | Conseq, U*	kg	-2.8E-01	-1.5E-01	-8.2E-01
PE (waste treatment) {US-NPCC}| recycling of PE | Conseq, U	kg	6.0E-03	3.5E-03	1.6E-02
PVC (waste treatment) {US-NPCC}| recycling of PVC | Conseq, U	kg	3.9E-03	2.1E-03	1.2E-02
Rubber (waste treatment) {US-NPCC}| recycling of rubber | Conseq, U	kg	9.2E-05	4.8E-05	2.7E-04
Steel and iron (waste treatment) {US-NPCC}| recycling of steel and iron | Conseq, U	kg	2.7E-03	1.4E-03	8.1E-03
Waste concrete gravel {US-NPCC}| treatment of, recycling | Conseq, U	kg	4.0E-02	2.1E-02	1.2E-01
Waste wood, post-consumer {GLO}| market for | Conseq, U	kg	3.0E-02	1.6E-02	8.8E-02

* Negative number due to avoided waste from the use of used jute bags for ground cover

Two metrics are assessed in this LCA: GWP and land use. GWP is assessed using the IPCC 2013 methodology over a 100 year time horizon[12]. Land use is assessed using the ReCiPe LCIA methodology[13], which is an un-weighted method for accounting land use (it is time weighted in that it measure area × time, but since the time component is equal to a single year for all UF operations here and the MRIO model, the time weighting is inconsequential here). ReCiPe does differentiate between urban and agricultural land occupation. Here we sum both land uses to account for total land use by UF, both indirect and direct. Table S34 outlines the impacts for each product from the UF operations for both GWP and land use.

Table S34. GWP and land use for different UF crops

Crop	Farm	GWP (kg CO^2e/kg crop)	Land use (m^2/kg)
Beet	3	0.399	0.713
Bell Pepper	1	0.156	1.542
Bell Pepper	2	1.165	0.245
Bell Pepper	3	0.304	0.638
Cabbage	3	0.116	0.342
Carrot	2	1.793	0.377
Carrot	3	0.549	0.989
Collard Greens	3	1.218	3.851
Cucumber	2	0.706	0.149
Cucumber	3	0.181	0.479
Eggplant	1	0.127	1.417
Green Beans	2	2.547	0.536
Green Beans	3	0.114	0.374
Kale	2	2.256	0.475
Kale	3	0.153	0.518
Lettuce	1	0.263	5.244
Lettuce	2	2.088	0.437
Lettuce	3	0.448	1.798
Radish	2	0.915	0.193
Scallion	2	3.062	0.644
Scallion	3	0.551	2.092
Squash	3	0.302	0.633
Tomato	1	0.104	0.880
Tomato	2	0.625	0.129
Tomato	3	0.134	0.344
Turnip	2	1.547	0.326
Turnip	3	0.261	0.462

Comparative performance of UF and conventional agriculture

GWP for the conventional food are taken from Heller and Keoleian's work on the GWP impacts of the US diet[14]. Their work includes a review of LCAs of different food products, including the range of reported findings and average across studies. Here we use their reported averages as a proxy for conventional agriculture. As their numbers are only for production, we add on transport impacts in accordance with Pirog and Benjamin's work on 'food miles' for conventional food products heading to Iowa (data for the US northeast remain in absentia)[15]. Transport impacts are taken as $9.7*10^{-5}$ kg CO_2e/kgkm (ecoinvent 3.2 process 'Transport, freight, lorry >32 metric ton, EURO5 {RER}| transport, freight, lorry >32 metric ton, EURO5 | Conseq, U'). Land use is taken as direct land occupation: calculated as the total 3 year average (2012-2014) annual US production divided by the total US cultivated area from the USDA annual vegetable statistics (beet, eggplant, kale, collards, turnip, scallion taken from 2002 survey)[16,17]. Direct land use is taken here as this is far and away the largest driver of this indicator for vegetal foods and should cover nearly 100% of land use. Values are corrected for food losses from the USDA LAFA statistics[2]. Table S35 outlines the impacts of the conventional goods for both GWP and land use.

Table S35. GWP and land use for conventional produce

Product	Transport (km)	Losses (%)	GWP – production (kg CO_2e/kg)	GWP – transport (kg CO_2e/kg)	GWP – total (kg CO_2e/kg)	Land Use (m^2/kg)
Beet	1759	6.5	0.33	0.28	0.65	0.40
Bell Pepper	1589	7.8	0.88	0.25	1.23	0.29
Cabbage	719	6.5	0.12	0.11	0.25	0.27
Carrot	1838	5.1	0.53	0.29	0.86	0.28
Collard Greens	1815	37.5	0.33	0.29	0.99	1.10
Cucumber	1277	6.1	0.66	0.20	0.92	0.48
Eggplant	1277	21.3	1.30	0.20	1.91	0.43
Green Beans	1313	18.4	0.73	0.21	1.15	1.93
Kale	1815	39.2	0.33	0.29	1.01	0.75
Lettuce	1823	7.7	1.08	0.29	1.48	0.27
Radish	1759	21	0.33	0.28	0.77	1.25
Scallion	1759	9.8	0.33	0.28	0.67	0.18
Squash	1277	12.5	0.09	0.20	0.33	0.64
Tomato	1569	5.2	0.67	0.25	0.97	0.34
Turnip	1815	6.5	0.33	0.29	0.66	0.80

Combining primary data on yields from the UF operations, we calculate the marginal change in environmental performance of Boston per meter UF cultivating vegetable *x*, $\frac{dI_x}{dA}$, as:

(4) $$\frac{dI_x}{dA} = \frac{dm_i}{dA}\left(i_{x,UA} - i_{x,conv}\right)$$

Where $\frac{dm_i}{dA}$ is the change in mass of vegetable per unit area in kilograms (annual yield), $i_{x,UA}$, the environmental impact from producing one kilogram of vegetable *x* with UF, and $i_{x,conv}$ the environmental impact of producing one kilogram of vegetable *x* with conventional agriculture (crediting for the substituted conventional crop). Table S36 outlines the predicted change in Boston's food-borne environmental impacts by implementing UF. It should be noted that the yield for UF includes 'dead space' on the farm where cultivation is not occurring (e.g. sheds, footpaths, etc.) and not just productive area. Where farms 1 and 3 produce the same product, the average yield and environmental burdens have been used.

Table S36. UF yields and marginal shifts in GWP and land use per m2 UF implemented in Boston

Crop	Farm(s)	Yield (kg/m^2)	Marginal GWP Shift (kg CO_2e/m^2 UF)	Marginal Land Use Shift (m^2/m^2 UF)
Beet	3	2.26	-0.57	0.70
Bell Pepper	1 and 3	2.30	-2.29	1.84
Bell Pepper	2	2.44	-0.15	-0.65
Cabbage	3	4.70	0.43	0.36
Carrot	2	1.59	1.47	0.16
Carrot	3	1.63	-0.51	1.16
Collard Greens	3	0.41	0.10	1.14
Cucumber	2	5.28	-1.11	-1.73
Cucumber	3	3.34	-2.46	-0.01
Eggplant	1	2.27	-4.04	2.24
Green Beans	3	4.27	-3.06	-4.36
Green Beans	2	1.12	1.57	-1.56
Kale	2	1.26	1.57	-0.35
Kale	3	4.72	-4.24	-1.96
Lettuce	1 and 3	0.80	-0.90	2.61
Lettuce	2	0.80	0.49	0.14
Radish	2	3.11	0.45	-3.29

Scallion	2	0.93	2.22	0.43
Scallion	3	0.76	-0.25	1.42
Squash	3	2.54	-0.08	-0.01
Tomato	1 and 3	2.94	-2.50	0.80
Tomato	2	4.70	-1.61	-0.99
Turnip	2	1.84	1.63	-0.86
Turnip	3	3.50	-1.39	-1.17

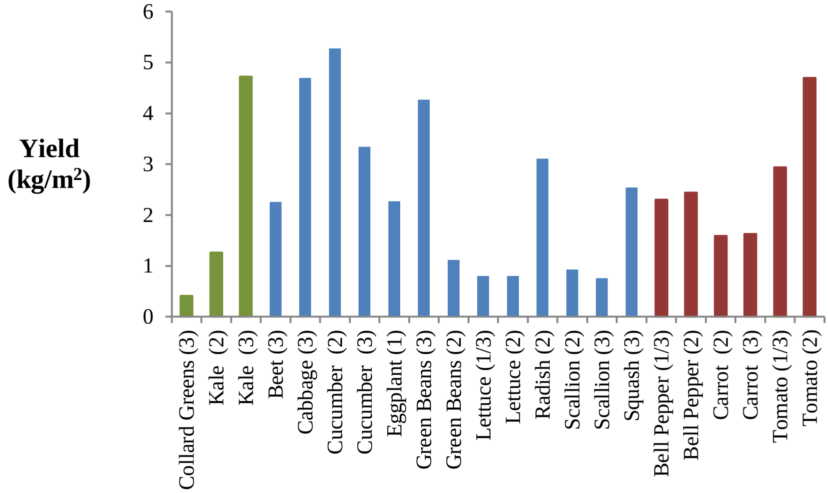


Figure 4. Yield for dark green (green), other (blue) and red and orange (red) vegetables. Farm(s) listed in brackets.

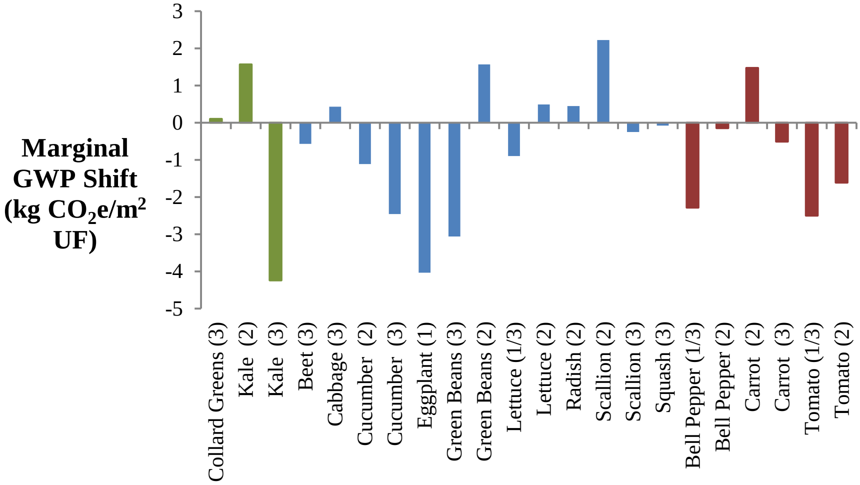


Figure 5. Marginal GWP shift per square meter UF grown for dark green (green), other (blue) and red and orange (red) vegetables. Farm(s) listed in bracets.

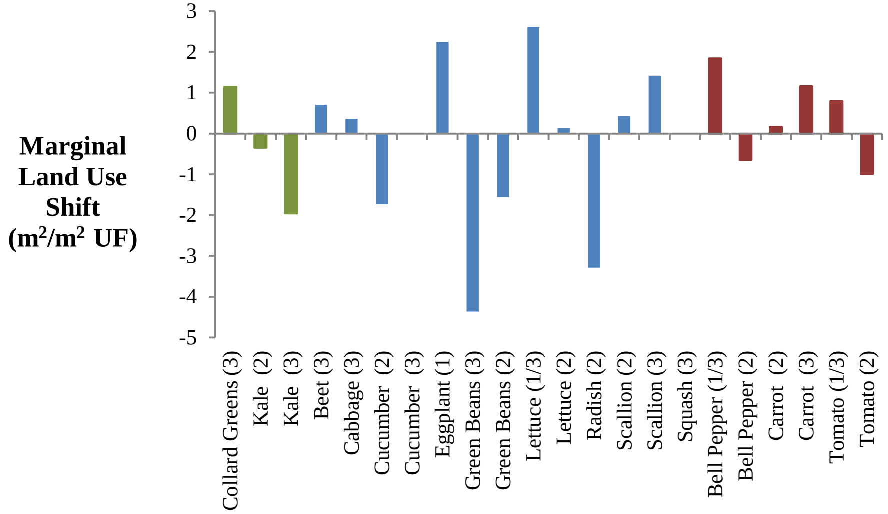


Figure 6. Marginal land use shift per square meter UF grown for dark green (green), other (blue) and red and orange (red) vegetables. Farm(s) listed in bracets.

UF Space Availability

Space for UF in Boston was estimated for ground and roof. Potential ground space was estimated by two methods: subtractive and additive. Roof space is performed in an additive manner. In this assessment, soil contamination is not considered when assessing the suitability of a piece of land for UF. Soil contamination is a major issue in US cities, particularly older cities with industrial heritage[18]. Moreover, shading effects from buildings are also ignored. As such, these estimates should be viewed as upper bounds for UF available space in Boston for both ground methods.

Ground Space – Additive

The additive approach for UF space starts with the assumption that the area of UF space in Boston is 0 m^2. Then utilizing a variety of data sources, we look at individual pieces of land, assesse their suitability for UF and add them to amount of space suitable for UF. The data sources are the 2016 Tax Assessment Parcel and open space maps, sourced from the City of Boston's Open Data Initiative[19] and the Massachusetts land use map from their geographic information system (GIS) data repository[20].

Tax assessment parcels data for the year 2016 includes all tax assessment parcels in Boston (166,248) including their land use according to the Massachusetts property classification system under the 'PTYPE' field in the raw data. Table S37 outlines the land uses we consider suitable for UF as they are not currently occupied by buildings or other productive land uses.

Table S37. UF suitable property types from tax assessment data

Land Use Code ('PTYPE')	Description
130	Residential land
131	Residential land (secondary)
132	Residential land (unusable)
390	Commercial land
391	Commercial land (secondary)
392	Commercial land (unusable)
440	Industrial land
441	Industrial land (secondary)
442	Industrial land (unusable)

337	Parking lot
359	Condo parking (commercial)
387	Pay parking lot
108	Condo parking (residential)
119	Residential parking lot

Parking lots have been included here to test the impact of their inclusion on the results, since they could be considered transitional land uses. Moreover, some of the parking lots are subterranean, making them unsuitable for the UF forms considered here, though this is not indicated by the parcel assessment data. Results include assessments with and without parking included to gauge the sensitivity of the results to their inclusion.

Community garden data includes the locations of existing UF in the city as designated by the Open Space map in the city's data repository. We assume that all operating community gardens are valid for this assessment.

Lastly, the state land use map from 2005 is used to include the land uses outlined in table S38 as described by the field 'LUCODE' in the data.

Table S38. UF suitable land uses from state land use map

Land Use Code ('LUCODE')	**Description**
1	Cropland
2	Pasture
6	Open Land
17	Transitional
36	Nursery
40	Brushland/Successional

Data are imported into the GIS software QGIS 2.4.0 and corrected for two issues:

- Residential and condo parking lots are checked for double counting, since the same assessment parcel are listed multiple times if the different parking spots on the same piece of area are owned by different individuals
- Where UF suitable plots intersected, the overlapping portion is subtracted from one of the layers. See figure 7.
- Plots with average slopes greater than 10°, as determined from digital elevation models provided by the National Oceanic and Atmospheric Administration (NOAA)[21], were deemed too steep for agriculture and excluded.

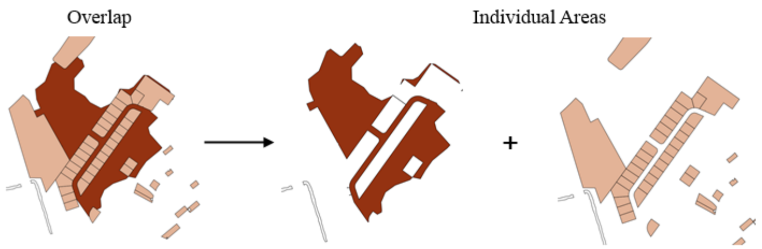


Figure 7. The disaggregation of overlapping areas in QGIS.

With a unique set of non-intersecting UF suitable plots, the area of each plot is calculated in QGIS and added to estimate total UF available space for Boston. Results are listed in table S39. QGIS also allows one to determine if an UF suitable plot lies within a block-group, providing area totals for each block-group (not shown here, but shown in Figures 2 a-b of the article).

Table S39. UF space using additive method				
Land Use	**Number of UF Suitable Sites**	**Average Site Size (standard deviation) m^2**	**Total UF Space (m^2)**	**Total UF Space (acres)**
Vacant residential*	5865	412 (1637)	2421080	599
Vacant commercial*	1267	1026 (3826)	1300847	322
Vacant industrial*	162	1375 (3668)	222860	55
Existing community gardens	123	1122 (2235)	138063	34
Pasture	1	13184 (0)	13184	3
Transitional	34	10271 (13569)	349228	86
Nursery	15	10127 (11699)	151906	38
Cropland	14	6395 (7739)	89525	22
Brushland/Successional	17	24979 (53946)	421542	104
Open Land	159	13825 (29049)	2198200	544
Residential Parking*	262	805 (4080)	210850	52
Commercial Parking*	630	892 (2698)	562124	139
Boston Total	8549	944 (5740)	8079409	2000

* Sum of their respective sub-uses

Ground Space – Subtractive

Contrasting with the additive method, here we start with the assumption that 100% of Boston is suitable for UF and then subtract those areas deemed unsuitable for farming. As with the additive approach, overlapping areas are removed to avoid double counting. Table S40 lists the land types considered unsuitable for ground-based US, their areas and the total UF available land in Boston using the subtractive estimation method.

Table S40. Boston UF ground space using subtractive method			
Land Type	**Data Source**	**Total Area (m2)**	**Total Area (acres)**
Steep areas	NOAA[21]	480475	119
Parks and sports fields	MassGIS (‘OpenSpace’ dataset)[22]	2818969	698
Protected open space	MassGIS (‘OpenSpace’ dataset)[22]	19461607	4817
Temporarily protected open space	MassGIS (‘OpenSpace’ dataset)[22]	11443	3
Cemeteries	MassGIS (‘OpenSpace’ dataset)[22]	3186303	789
Buildings	Boston Open Data(‘Buildings’ dataset)[19]	21946457	5433
Impervious surfaces (roads, sidewalks, etc.) – buildings removed	Boston Open Data(‘Impervious Surfaces’ dataset)[19]	35926938	8893
Airport	Boston Open Data[19]	6172008	1528
Total	-	90004200	22278
Total Boston Area			
Boston Total Area	Boston Open Data (‘Boundary’ dataset)[19]	125095606	30964
Total UF Area			
Total UF Area	-	35736010	8846

As with the additive scenario, QGIS is used to allocate available space to the block-groups in Boston.

Rooftop Space

The first step in estimating the amount of rooftop area available for UF in Boston is to get a clean data set of pertinent information of the Boston building stock. Davila and colleagues already outlined the process in detail[23], but in a nutshell it involves combining three datasets: the Boston property tax assessment for the year 2014[24], the 2016 tax parcel assessment data[19] and the geospatial building data for Boston[19].

The property tax assessment is required as it is the most up to date and detailed assessment of building attributes for the city and contains all buildings and sub-units within buildings. Because of the latter point, it contains double counting of buildings that contain multiple apartment units. Double counted units were removed using a Python 2.7 script which identifies buildings with multiple units based on the 'CM_ID' field. While consolidating multiple units to a single entry, we also assign the heating and cooling type of the building based on the majority heat and cooling types for the units within the building. This initial data parsing reduces the tax records from 164,092 entries to 100,858 entries.

Although the tax records data contains the most detailed information, they contain no spatial data and cannot be mapped nor attributed to block-groups. To overcome this we link the 10 digit property ID key 'Parcel_ID' in the tax record with the synonymous 'PID_LONG' key in the spatially explicit tax assessment data. Minor mismatches between the datasets shave the number of entries down to 98,865. Table S41 outlines the various fields utilized in this process and their purposes.

Table S41. Fields used in joining tax data sets

Field	**Dataset**	**Purpose**
CM_ID	2014 Tax records	Identify duplicate building entries
U_Heat	2014 Tax records	Identify the predominant heating type in multi-unit dwellings
U_AC	2014 Tax records	Identify if a majority of units have air conditioning in multi-unit dwellings
Parcel_ID	2014 Tax records	Join data tax record data with tax parcel assessment polygons
PID_Long	Tax parcel assessment data	Join tax parcel assessment polygons with 2014 tax record data

Although this dataset is spatially explicit, it still contains numerous errors in terms of non-existent buildings, improper building footprints and building types that are not suitable for UF applications. Moreover, the tax data does not contain reliable estimates of building heights. To overcome this we join the generated parcel data set with the building data set, since the latter contains information on building types and can be used to more accurately calculate building footprints. The spatially explicit data is then mapped as polygons in QGIS, converted to centroids and spatially joined to building polygon data. Figure 5 illustrates this process in QGIS.

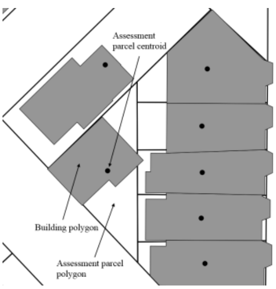


Figure 8. Joining the tax data with the building polygons

Building data is also spatially joined with data on historic preservation districts from the Boston Open Data Initiative[19], as buildings in these districts are not permitted to make changes to their exterior appearance, and should be excluded from UF use. The four joined datasets contain the necessary information for estimating UF building space in Boston, outlined in table S42.

Table S42. Properties used for calculating UF suitability		
Building Property	**Key**	**Dataset**
Year of construction	'YR_BUILT'	2014 Tax assessment data
Number of floors	'NUM_FLOORS'	2014 Tax assessment data
Roof type	'R_ROOF_TYP'	2014 Tax assessment data
Heating type	'R_HEAT_TYP'	2014 Tax assessment data
Presence of air conditioning	'R_AC'	2014 Tax assessment data
Ground elevation	'GROUND_ELE'	Building data
Roof elevation	'ROOF_ELE'	Building data
Building Type	'IEL_TYPE'	Building data
Building Area	Calculated in QGIS	Building data
Presence in historic preservation district	Generated in QGIS with spatial join	Historic Districts

The final step in cleaning the building data is to remove buildings lacking information on year built, height (either no data on number of floors or incomplete elevation data) and unsuitable for UF. The latter is done using the 'IEL_TYPE' key of the building data by excluding ruins, foundations, etc. After all of the manipulations, the cleaned dataset of collated building and tax data contains 76,170 buildings (69,857 when historical buildings are excluded).

Actually determining the area of Boston's buildings available for UF is impossible since we lack structural analyses of the buildings that would allow us to determining their individual capacities for supporting the load of a rooftop farm. However, we can use three indicators to estimate UF rooftop space: building age, building height and roof type.

Building age is justified in the sense that the introduction of building codes and standards has led to the gradual infiltration of more structurally sound buildings through mandated snow loading capacity, etc. Thus, here we assume that older buildings are less likely to be suited for UF than new ones. This is a gross simplification, since old buildings, particularly older factories and cast iron buildings are designed to support significantly heavier loads than they are burdened with not in their post-industrial uses. As such, we run multiple scenarios building age is used as a cutoff for UF consideration. The cutoff construction years range from 1900 to 2000 in ten year intervals. This covers around ¾ of Boston's building stock by both number of buildings and area. Figure 6 is a histogram showing the effects of different construction year cutoffs on the number and area of buildings considered.

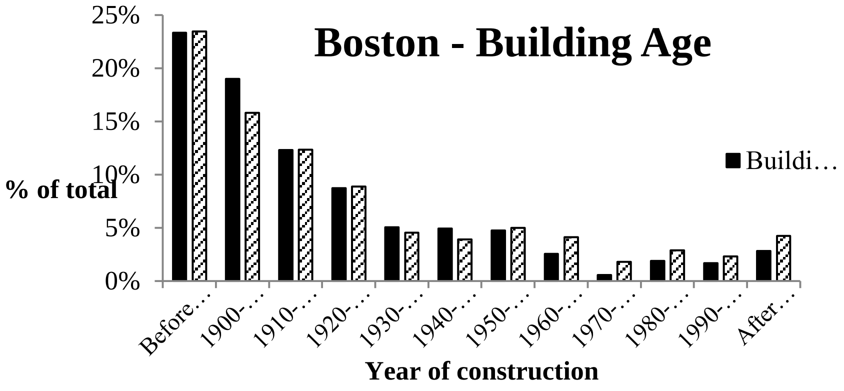


Figure 9. Histogram of building numbers and area within different intervals of construction years.

Height is also a natural limiting factor on UF suitability, since stronger winds above certain heights not only pose a challenge to the stability of the growing medium, but also a safety issue to farm workers. Heights are taken as the difference of ground and roof elevations in the building data, and where these are lacking, the number of floors times the average floor height of 3.42 m as determined from those buildings within the building data set that contain both number of floors and elevation data. Looking at the histogram of number of buildings and building area with within different height ranges in Figure 7, it can be seen that only small fraction of Boston's building stock is over 30 m tall, and hence this is taken as the maximum allowable height for a building to be considered UF appropriate.

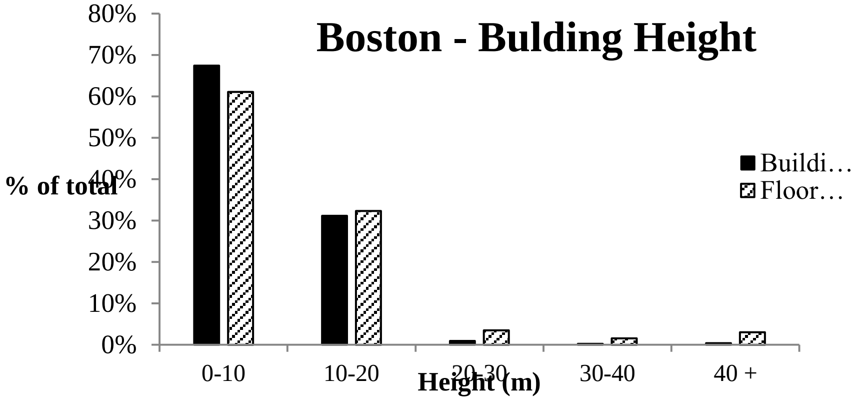


Figure 10. Number of buildings and building area in Boston for different building height ranges

Lastly, roof type is a natural indicator of UF suitability, since rooftop farms necessitate flat roofs. Only some of the tax data entries specify the roof type. Where no data on roof type was given, the roof type was assigned to a building probabilistically based on the representation of flat roofs in the general New England building stock. For commercial buildings this is a 25.2% probability according to the commercial building energy consumption survey[25]. As this data is lacking in the analogous residential survey, it is estimated as the percentage of buildings in our generated building data set with flat roofs, 21.3% (11735 of 55026 entries with roof data).

Table S43 shows the results of the different cutoff years, the height limit and the probabilistic assignment of roof top averaged over 100 runs.

Table S43. Estimates of rooftop suitable UF based on different cutoff years, a height limit of 30 m and the presence of a flat roof dictated by probabilistically over 100 Monte Carlo simulations

Cutoff Year	Building Space (m^2)	SD (m^2)	Building Space (acres)	SD (acres)
1900	1714149	34113	424	8
1910	942317	35025	233	9
1920	672546	32088	166	8
1930	512520	31076	127	8
1940	440620	29107	109	7
1950	406841	30041	101	7
1960	334557	24713	83	6
1970	253619	19861	63	5
1980	215278	19309	53	5
1990	156922	15967	39	4
2000	104950	12094	26	3

Urban metabolism interactions

This study accounts for three direct material/energy exchanges between farm and city: runoff retention, solid waste assimilation and building energy reductions.

Runoff retention

The engineering of the modern city has seen the channelizing, rerouting and burying of urban streams. The proliferation of impermeable surfaces throughout cities that prevent the penetration of the rainwater into the soil mean that much of this water is directed towards stormwater sewers, eventually bound for wastewater treatment instead of recharging groundwater aquifers or surface waters. The net effect is that when it rains, large volumes of water are unnecessarily sent for treatment or during intermittent heavy rain events, sewer capacity is exceeded and water from sewage pipes is vented to local surface waters[26]. If the stormwater is combined with sanitary water in a combined sewer, then heavy rain events can lead to the release of raw sewage when the sewers overflow in combined-sewer-overflow events (CSO)[27]. Boston has over 235 miles of combined-sewers and 37 CSO outfalls and is negatively impacted by CSO events during particularly intense or long rainfalls[27]. Since UF occasionally replaces impermeable area with soil that can either retain water for crop uptake or provide a hydraulic conduit between surface and groundwater it is important to model how the potential runoff mitigation provided by UF in Boston.

Here we consider to situations where UF implementation in Boston obviates runoff to the sewers: where UF replaces ground parking and where it is placed on buildings. We provide upper and lower bounds of runoff retention based on field studies of extensive green roofs. Lower and upper retention rates are taken as 50%[28] and 74% [29], respectively. The same rates are applied to ground UF since they are also representative of runoff retention on permeable land[26]. This method ignores the heterogeneity of soil characteristics and resultant runoff retention, but as a basic estimate to gauge the impact of UF on Boston's hydrology it should suffice to identify whether the scale of these impacts are significant or miniscule. Moreover, this method ignores the ability for UF to reduce the prevalence of CSOs and toxic fallout from sewage releases. However, quantifying such impacts would require detailed information on CSO outfall locations and local pollution assimilation capacity that is beyond the scope of this exercise.

In assessing the GWP and land use impacts from avoided stormwater treatment, we use the ecoinvent 3.2 process 'Wastewater, unpolluted {RoW}| treatment of, capacity 5E9l/year | Conseq, U' to model wastewater treatment in Boston. Using the aforementioned GWP and land use methods we calculate 0.293 kg CO_2e and 0.0260 m^2 in avoided impacts per m^3 avoided wastewater treatment.

Precipitation is taken as the 2000-2015 annual Boston average of 1.11 m[30].

Solid waste assimilation

We use primary data collected from the urban farms we have the following compost application rates:

- Roof based UF: 2.8 kg compost/m^2
- Ground based UF: 0.3 kg compost/m^2

Though the lower compost usage for roof based UF seems counterintuitive, it is a result of wind-related soil losses from green roofs and the need to supplement the expanded shale/concrete grow media with medium rich in nutrients and organic carbon. Ground-based UF is less affected by soil loss and tends to occur in a top-soil matrix rich in organic carbon and with greater nutrient sorption capacity, and hence, does not demand the same volume of nutrient/organic additions as the rooftop farms.

To convert from deposited compost to mass of avoided waste, we assume a mass loss of 32% from waste compost. This is a conservative estimate based on the open windrow composting of garden waste in the US[31]. Applying this factor we find that rooftop farms and ground-based UF can assimilate 4.1 and 0.4 kg organic waste/m^2, respectively. In modelling the environmental impacts of waste assimilation, we allocate the waste treatment and related avoided fertilizer production to the previous life-cycle of the waste, and the delivery of the waste to the UF site to the farms.

Building Energy

In modelling the potential interactions between a building's energy system and farm the following assumptions are made:

- No direct coupling of the building energy system and urban farm are made (e.g. no heat ventilation into the growing media to extend growing periods, etc.)
- Energy savings only apply to the floor directly below the roof. This will underestimate the energy savings to the entire building, since the attenuation of temperature shifts on the top floor will have a spillover effect on energy use on subsequent floors that diminishes with distance from the roof.
- We assume that the energy impacts of rooftop urban farming are similar to those from extensive green roofs.
- Effects at the city level are modeled in an additive manner, ignoring the multiplicative effect of large numbers of farms in proximity. This will underestimate total energy savings as reduced air conditioning use from an attenuated urban heat island effect are not counted here.
- Insulation values and heating fuels are assumed to be independent of other building characteristics (e.g. age, height, size, etc.) during the Monte Carlo simulations, as the building energy surveys lack data on relating these characteristics for the New England region.

Modeling building energy savings start first by characterizing the level of insulation on the building and the energy consumption per unit area for heating and cooling. Both of these parameters are taken from the residential and commercial building energy consumption surveys[25,32]. Heating and cooling energy intensities are assumed to be constant for all commercial and residential buildings in the city, while insulation levels are assigned probabilistically to each building at the start of each simulation. Likewise, the heating fuel and presence of air conditioning are assigned in the same manner to buildings that are lacking these data in the tax assessment survey. The prevalence of different heating types and air conditioning presence are also taken from the building energy surveys. Table S44 outlines these parameter values and their likelihood in the New England building stock.

Table S44. Building parameters

	Residential		**Commercial**	
Parameter	**Value**	**Probability**	**Value**	**Probability**
Energy Intensity				
Heating Intensity	352 $MJ/m^2/a$*	-	465 $MJ/m^2/a$	-
Cooling Intensity	6 $MJ/m^2/a$*	-	51 $MJ/m^2/a$	-
Insulation Levels				
Well	-	0.36	-	0.36**
Adequate	-	0.44	-	0.44**
Poor	-	0.2	-	0.2**
None	-	0	-	0**
Air Conditioning Present				
Yes	-	0.76	-	0.62
No	-	0.24	-	0.38
Heating Present				
Yes	-	1	-	0.87
No	-	0	-	0.13
Heating Fuel				
Electricity	-	0.12	-	0.18
Natural Gas	-	0.52	-	0.36
Fuel Oil	-	0.32	-	0.46
Propane	-	0.04	-	0

* Taken as the total energy intensity for residential buildings (Table CE1.2-RECS2009)[32] times the percentage going to different end uses[33]
** Not available in the commercial energy consumption survey. Assumed that same as residential values

To link UF with energy savings, a relation between insulation level and amount of cooling and heating attenuation is needed. Results from La Roche and Berardi's field work measuring energy savings of green roofs at different insulation thicknesses was useful in building this concordance[34]. In using their numbers we assume equivalent percentage savings for buildings in Chicago, US and Boston. Although Chicago has a continental climate with slightly warmer summers and cooler winters, the data adequate for the cursory analysis performed here. Table S45 outlines the concordance between the insulation levels here and the predicted energy savings from building-integrated UF.

Table S45. Predicted energy savings at different insulation levels

Insulation level from energy consumption survey	Insulation thickness from La Roche and Berardi (m)[34]	Heating attenuation (%)	Cooling Attenuation (%)
Well	0.20	0	7.5
Adequate	0.10	0	7.5
Poor	0.05	2.5	8
None	0	7.5	15

With these parameters in hand for each building, the UF related energy savings are estimated as the product of energy intensity, area and percentage attenuation. Embodied greenhouse gas impacts are taken from the Boston's own carbon footprint accounting since these represent the intensities for the local grid and fuel delivery systems[35]. Table S46 outlines emissions intensities for the different fuels used in Boston buildings for space conditioning.

Table S46. Carbon intensities for different fuels in Boston

Energy source	kg CO2e/MJ supplied
Electricity	0.102
Natural Gas	0.050
Fuel Oil	0.070
Propane*	0.050

* Assumed equivalent to natural gas here. Minor role in energy system should not influence general results.

City-wide optimization simulations

In assessing the impacts of UF at the city level, all of the disparate pieces described in the preceding sections were tied together. A Python 2.7 script acts as a scaffolding with which to model the impacts of UF on Boston's food-borne GWP impacts and land use, and to model interactions between the urban farms and the city's energy and material metabolism. The script can optimize UF in Boston to maximize any one of three indicators at a time: GWP savings, land use savings and nutritional content. As many of the building parameters were assigned probabilistically, we run each optimization scenario 100 times in a Monte Carlo manner, randomly assigning UF suitability and building energy use characteristics. Despite the low number of runs, little variation is seen around the mean for the results, hinting at the suitability of our choice of simulation length. Requests for the script can be made through the corresponding author.

Figure 11 outlines the algorithm.

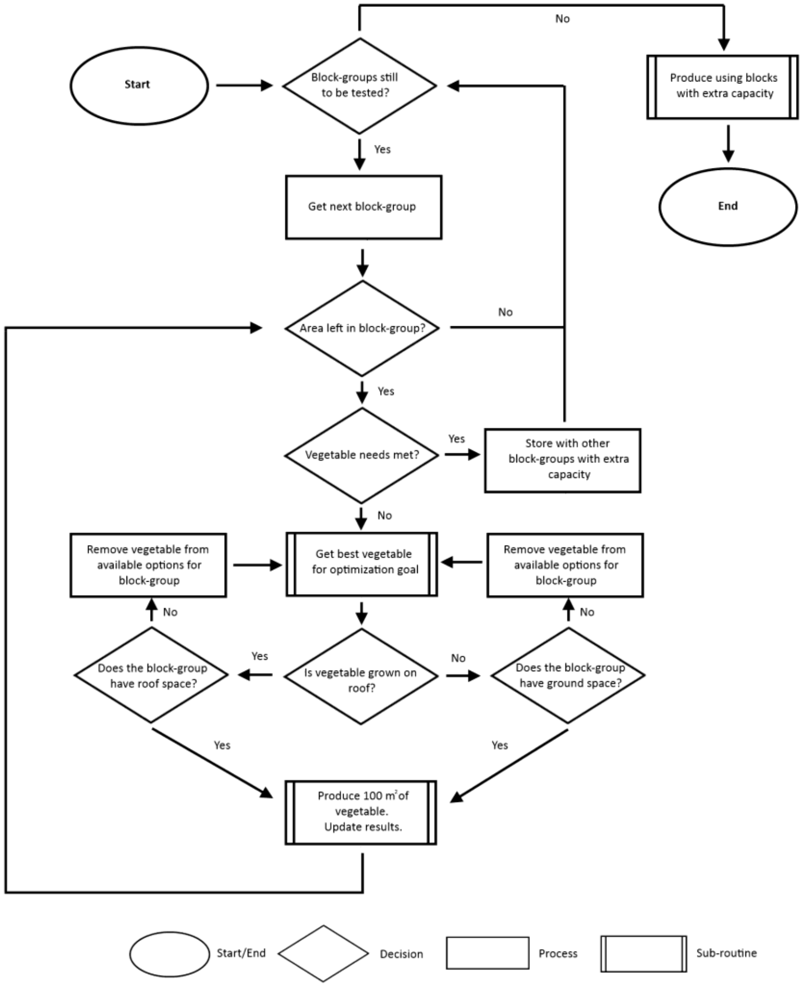


Figure 11. Optimization algorithm outline.

GWP Optimization

In optimizing the GWP impacts of the city's UF system, a 'greedy' algorithm is used. 'Greedy' algorithms work by picking items with the largest marginal benefit in terms of the parameter being optimized. In this case that means growing vegetables with the largest GWP impact reduction per unit area grown. Table S47 lists the UF produce with the largest reductions in GWP per area cultivated.

Table S47. List of UF vegetables in order of decreasing reductions in GWP impacts per m^2 planted

Vegetable	**Siting**
Kale	Ground
Eggplant	Ground
Green Beans	Ground
Tomato	Ground
Cucumber	Ground

Bell Pepper	Ground
Tomato	Roof
Turnip	Ground
Cucumber	Roof
Lettuce	Ground
Beet	Ground
Carrot	Ground
Scallion	Ground
Bell Pepper	Roof
Squash	Ground
Collard Greens	Ground
Cabbage	Ground
Radish	Roof
Lettuce	Roof
Carrot	Roof
Green Beans	Roof
Kale	Roof
Turnip	Roof
Scallion	Roof

Each run of the algorithm cycles through all of Boston's block groups and performs the following sub-routine for each individual block group:

Block-group GWP optimizing sub-routine

Is there area left in the block-group?

- **Yes:** *Are the block-group's needs met for all vegetables?*
  - **Yes:** Store the block-group with others with extra capacity. End sub-routine.
  - **No:** Get the UF vegetable with the largest marginal GWP impact reduction.
    - *Are all of the blocks needs met for this vegetable?*
      - **Yes:** Remove vegetable from list of potential vegetables and get the next vegetable in the list.
      - **No:** *Where is the vegetable grown?*
        - **Roof:** *Is there building space?*
          - **Yes:** Produce 100 m^2 of the vegetable (or remainder of roof space if less than 100 m^2 left). Update results. Rerun sub-routine.
          - **No:** Remove vegetable from list of potential vegetables and attempt with next vegetable.
        - **Ground:** *Is there ground space?*
          - **Yes:** Produce 100 m^2 of the vegetable (or remainder of ground space if less than 100 m^2 left). Update results. Rerun sub-routine.
          - **No:** Remove vegetable from list of potential vegetables and attempt with next vegetable.
- **No:** End sub-routine.

In this way each block-group will attempt to satisfy as much of its vegetable demands using those UF crops that minimize the GWP impacts of the block-group's residents. At the completion of a single cycle through all of Boston's block-groups, if there are block-groups that are able to satiate there vegetable demands while having surplus space, a separate sub-routine is run on those blocks:

City GWP optimizing sub-routine

Is there area left in the block?

- **Yes:** *Are all of the city's needs met for all vegetables?*
  - **Yes:** End sub-routine.
  - **No:** Get the UF vegetable with the largest marginal GWP impact reduction.
    Are all of the city's needs met for this vegetable?
    - **Yes:** Remove vegetable from list of potential vegetables and get the next vegetable in the list.
    - **No:** *Where is the vegetable grown?*
      - **Roof:** *Is there building space?*
        - **Yes:** Produce 100 m^2 of the vegetable (or remainder of roof space if less than 100 m^2 left). Update results. Rerun sub-routine.
        - **No:** Remove vegetable from list of potential vegetables and attempt with next vegetable.
      - **Ground:** *Is there ground space?*
        - **Yes:** Produce 100 m^2 of the vegetable (or remainder of ground space if less than 100 m^2 left). Update results. Rerun sub-routine.
        - **No:** Remove vegetable from list of potential vegetables and attempt with next vegetable.
- **No:** End sub-routine.

This sub-routine is run on all of the block-groups with auxiliary space until all are exhausted or the city's vegetable needs are met. This algorithm can be run with additive or subtractive UF space estimates, including or excluding parking.

In determining the block-group and city-wide vegetable demands we use the 2010 LAFA data for average demands at the household prior to household wastage and multiply by the population for each block-group. This assumes that wastage from the urban farms is negligible, which was observed during in the field while working with the case farms. We do not attempt to satiate the needs for all vegetables listed in the USDA LAFA data[2], but only those that UF produces or where UF crops act as reasonable substitutes. Table S48 shows the average intake of relevant vegetables from the LAFA data.

Table S48. LAFA data and per capita demand of UF producible vegetables

Vegetable	Raw LAFA (lb/a)	Per capita demand of UF crop (kg/a)	Total Boston Demand (kg/a)	Fraction of total vegetables
Beans		2.31	1424548.816	0.045155
Fresh	1.44			
Canned	2.07			
Frozen	1.58			
Beet*	0.24	0.11	68551.87793	0.002173
Bell Pepper	8.77	3.98	2456900.259	0.077879
Cabbage		2.90	1786523.718	0.056629
Fresh	5.96			
Canned	0.41			
Carrots		3.83	2361388.616	0.074851
Fresh	7.14			

Canned	0.53			
Frozen	0.76			
Collard Greens	0.51	0.23	143684.578 8	0.004555
Cucumbers		3.28	2023733.81	0.064148
Fresh	5.81			
Canned	1.41			
Eggplant	0.53	0.24	147267.998 2	0.004668
Kale		0.89	549605.579 5	0.017421
Kale	0.24			
Spinach	1.27			
Frozen Spinach	0.45			
Lettuce		10.52	6484207.70 4	0.205537
Leaf	13.52			
Romaine	9.62			
Radish	0.38	0.17	106540.984 4	0.003377
Scallion	0.24	0.11	68551.8779 3	0.002173
Squash		3.13	1931088.17 8	0.061212
Squash	3.40			
Pumpkin	3.49			
Tomato		19.34	11926557.8 7	0.378048
Fresh	15.2			
Canned	27.4			
Turnip*	0.24	0.11	68551.8779 3	0.002173

* No LAFA data on beets and turnips. Assumed to be the same as the lowest consumed food for which LAFA data exists, Kale.

Land use optimization

This method is identical to the GWP impact algorithm except that UF crops are now listed in order of their ability to reduce land use. Table S49 shows the list of vegetables when ordered in this manner.

Table S49. List of UF vegetables in order of decreasing reductions in land use per m^2 planted	
Vegetable	**Siting**
Green Beans	Ground
Radish	Roof
Kale	Ground
Cucumber	Roof
Green Beans	Roof
Turnip	Ground
Tomato	Roof
Turnip	Roof
Bell Pepper	Roof
Kale	Roof
Squash	Ground
Cucumber	Ground
Lettuce	Roof
Carrot	Roof

Cabbage	Ground
Scallion	Roof
Beet	Ground
Tomato	Ground
Collard Greens	Ground
Carrot	Ground
Scallion	Ground
Bell Pepper	Ground
Eggplant	Ground
Lettuce	Ground

Nutritional Optimization

In optimizing for nutrition, the algorithm is moved away from a greedy mode. This is because the boundary for knowing when to stop producing the best-option vegetable in the greedy mode is the demand for that crop at the block-group or city level. In the nutritional optimizing algorithm we are in fact attempting to change the boundary condition, that is, the amount of certain foods consumed, eliminating this indicators candidacy for this role. Instead we randomly pick vegetables in a manner that reflect the consumption patterns of the average consumer, whilst trying provide as much nutrition as possible, beyond their typical demands, aiming to satisfy the nutritional needs as outlined by the USDA guidelines[1].

USDA guidelines provide recommended intakes for four vegetable types: starchy, dark green, red and orange and other. Here we focus on the last three since none of the vegetables in the first group are produced by any of the case farms. The nutritional algorithm has two variants. The first attempts to produce as much vegetables as possible and satiate the entire vegetable demands of the block-group (and city using surplus land). The second version attempts to close the gap between current consumption and USDA guidelines. In both versions the vegetable group with the largest distance to target is always prioritized, so that in the first variant it will end up producing nearly the same fraction of USDA guidelines, while the second may or may not end up satisfying all groups to the same extent. Importantly, the second variant models a situation where UF is not substituting conventional supply chains, but supplementing, and hence, no crediting for avoided conventional production is accounted for.

Vegetable demands are taken from the USDA guidelines for different sexes and age groups. The deficiency is taken as the difference between the USDA guidelines and usual daily intake from the NHANES data[36]. Individual nutritional demands and deficiencies are then combined with census demographics data for each block-group to get the total demands at block-group and city level. Table S50 shows the nutritional demands and deficits for different demographics.

Table S50. Nutritional demands and deficits (in brackets) for different demographics

		Males								**Females**							
		Age								**Age**							
Vegetable Category	**Unit**	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+	1-3	4-8	9-13	14-18	19-30	31-50	51-70	71+
Dark Greens	cup eq.	0.5 (0.5)	1 (1)	1.5 (1.5)	2 (1.3)	2.5 (1.8)	2.5 (1.8)	2 (0.6)	2 (1.3)	1 (1)	1.5 (1.5)	1.5 (1.5)	1.5 (0.8)	1.5 (0.8)	1.5 (0.8)	1.5 (0.1)	1.5 (0.8)
Red and Orange	cup eq.	2.5 (1.1)	3 (0.9)	5.5 (3.4)	6 (3.6)	7 (3.5)	7 (3.5)	6 (3.2)	6 (3.2)	2.5 (1.1)	3 (1.6)	5.5 (3.4)	5.5 (3.4)	5.5 (2.7)	5.5 (2.7)	5.5 (2.7)	5.5 (2.7)
Other	c	1	3.	5	6	7	7	6	6	2	3	5	5	5	5	5	5

	up eq.	(1.3)	5 (1.4)	(3.6)	(3.9)	(3.8)	(3.8)	(0.6)	(2.5)	(0.6)	(1.6)	(2.9)	(2.9)	(1.5)	(0.8)	(0.1)	(1.5)

Taking the institution adjusted population of Boston of 616,602 in 2010 and the demographic spread, we estimate the city-wide nutritional demands as 5.68×10^{7}, 1.83×10^{8} and 1.75×10^{8} cup eq. of dark green, red and orange, and other vegetables, respectively. City-wide nutritional deficits are estimated as 1.96×10^{7}, 6.64×10^{8} and 2.62×10^{7}cup eq. for dark green, red and orange, and other vegetables, respectively.

Table S51 summarizes the UF crops in terms of their vegetable type and the amount of nutritional units supplied per area planted.

Table S51. UF Crops and their nutritional properties

Vegetable	USDA Category	Siting	Cup eq./m^2
Beans	Other	Ground	28
Beet	Other	Ground	17
Bell Pepper	Red and Orange	Ground	19
Cabbage	Other	Ground	52
Carrots	Red and Orange	Ground	13
Collard Greens	Dark Green	Ground	6
Cucumbers	Other	Ground	28
Eggplant	Other	Ground	28
Kale	Dark Green	Ground	36
Lettuce	Other	Ground	7
Squash	Other	Ground	22
Tomato	Red and Orange	Ground	17
Turnip	Other	Ground	27
Scallion	Other	Ground	8
Beans	Other	Roof	10
Bell Pepper	Red and Orange	Roof	20
Carrots	Red and Orange	Roof	13
Cucumbers	Other	Roof	44
Kale	Dark Green	Roof	10
Lettuce	Other	Roof	7
Radish	Other	Roof	25
Scallion	Other	Roof	9
Tomato	Red and Orange	Roof	28
Turnip	Other	Roof	14

Block-group nutritional optimization sub-routine

Is there area left in the block-group?

- **Yes:** *Are the block-group's nutritional demands (or deficit) met?*
  - **Yes:** Store the block-group with others with extra capacity. End sub-routine.
  - **No:** Determine the vegetable type with largest distance to gap.
    - *Is there building space?*
      - **Yes:** Randomly choose vegetable from amongst those within the vegetable category that are grown on buildings, with probability based on usual intake rates. Produce 100 m^2 of the vegetable (or remainder of roof space if less than 100 m^2 left). Update results. Rerun sub-routine.
      - **No:** Randomly choose vegetable from amongst those within the vegetable category that are grown on the ground, with probability based on usual intake rates. Produce 100 m^2 of

the vegetable (or remainder of roof space if less than 100 m^2 left). Update results. Rerun sub-routine.

No: End sub-routine.

After all block-groups are given the chance to produce for themselves, those with surplus growing area attempt to produce to satisfy Boston's nutritional needs.

<u>City nutritional optimization sub-routine</u>

Is there area left in the block-group?

Yes: *Are the city's nutritional demands (or deficit) met?*

Yes: End sub-routine.

No: Determine the vegetable type with largest distance to gap at the city level.

Is there building space?

Yes: Randomly choose vegetable from amongst those within the vegetable category that are grown on buildings, with probability based on usual intake rates. Produce 100 m^2 of the vegetable (or remainder of roof space if less than 100 m^2 left). Update results. Rerun sub-routine.

No: Randomly choose vegetable from amongst those within the vegetable category that are grown on the ground, with probability based on usual intake rates. Produce 100 m^2 of the vegetable (or remainder of roof space if less than 100 m^2 left). Update results. Rerun sub-routine.

No: End sub-routine.

In the same manner is the GWP and land use sub-routines, this algorithm can be used with additive and subtractive UF land use estimates.

UF Revenue

Crop prices are taken from consumer expenditure data (averaged over the available years)[37] or from USDA retail reports on specialty crops[38]. Table 49 outlines the crop prices used here in current US dollars.

Table S52. Crop prices

Vegetable	**USD/kg**	**Source**
Beans	3.20	Consumer Expenditure
Beet	2.19	USDA
Cabbage	1.42	Consumer Expenditure
Carrots	1.72	Consumer Expenditure
Collard Greens	2.13	USDA
Cucumbers	2.85	USDA
Eggplant	3.01	USDA
Iceberg	2.28	Consumer Expenditure
Kale	2.28	USDA
Peppers	5.38	Consumer Expenditure
Radish	3.51	USDA
Scallion	1.22	USDA
Squash	1.86	USDA
Tomato	3.71	Consumer Expenditure
Turnip	2.19	USDA

When the city only produces for its residents, the above algorithms are unaltered, and the revenue from block-group trade is calculated and recorded along with all of the environmental and nutritional results.

The only shift to the algorithm is when the block group starts exporting the conurbation. A crude method would simply produce the most profitable crop, but this would actually lead to a glut of one or two crops on the market, leading to a crash in prices. To avoid this, the city's extra space is allocated to crops based on the usual demand for the crop according to the USDA LAFA data (see rightmost column of Table 45). The above algorithms remain unaltered from the above cases with the exception of a sub-routine that is run at the end on all block-groups with surplus land:

Is there area left in the block-group?

Yes: Randomly select vegetable based on usual intake probability.

Is there suitable UF space to grow the crop (either roof or ground)?

Yes: Grow 100 m^2 (or available area) of that vegetable and update results. Rerun sub-routine.

No: End sub-routine.

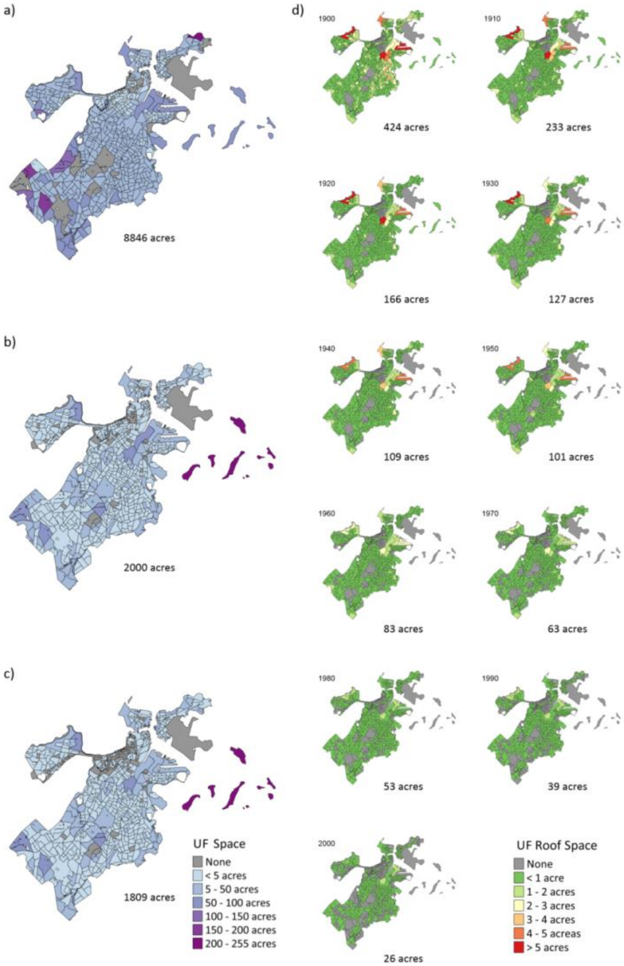


Figure 12. UF space results for (a) subtractive, (b) additive, (c) additive minus parking and (d) rooftop

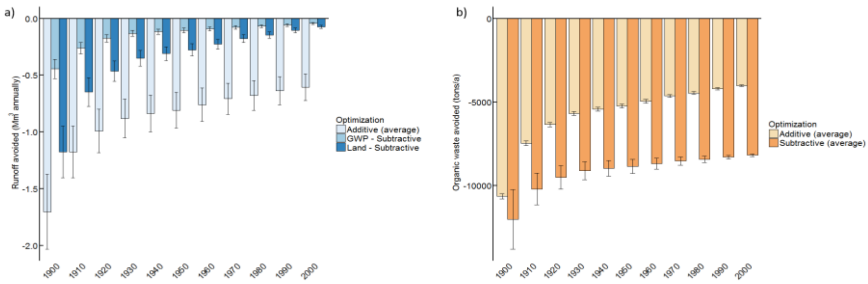


Figure 13. (a) avoided runoff results for the different scenarios and years . Additive results for both land and GWP are averaged due to similarity. (b) Organic waste uptake from UF averaged for both optimizations due to similarities

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Appendix E

Article 5: Ethical aspects of life cycle assessments of diets

Ethical aspects of life cycle assessments of diets

Benjamin Goldstein (bgol@dtu.dk), Steffen Foss Hansen, Mickey Gjerris, Alexis Laurent and Morten Birkved

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Abstract: Since the turn of the century a growing chorus of researchers has been espousing reduced meat and dairy intake as a partial strategy to transition towards a sustainable food system. Many of these studies have been predicated on a life-cycle assessment (LCA) methodology and though transparent in communicating their work within that framework, it has largely gone unmentioned that LCA involves a number of choices by the assessor and LCA methodology developers that are ultimately subjective. This study uses a consequential LCA of the average Danish diet in comparison to model vegetarian and vegan diets, leveraging the cultural perspectives afforded by the ReCiPe methodology, as starting point to explore the ways that subjectivity influences the LCA process and to test the robustness of the results against these different viewpoints. Mirroring earlier studies, we find vegetarian and vegan diets generally perform better environmentally compared to a standard Danish diet, but that there was minimal difference between the two no-meat options. Results were resilient to varying cultural perspectives applied in the model. LCA methodology, though loaded with value judgments, remains a dependable tool for assessing environmental dietary performance, but is less suited for estimating environmental pressures that are highly dependent on local conditions (e.g. chemical toxicity).

- **Keywords:** Life-cycle-assessment; Ethics; Diets; Vegetarian; Vegan; Sustainable consumption

Introduction

At the global level, food production is estimated to be responsible for between 20% and 50% of anthropogenic environmental impacts (McLaren, 2010, Notarnicola et al., 2012 and Roy et al., 2012). Irrespective of this pressure's true value, it is clear that global food consumption affects the performance of ecosystems negatively (locally and globally) through contributions to a variety of environmental issues including: climate change, water stress, toxic chemical release, air quality degradation, eutrophication of water bodies, soil erosion, and biodiversity losses (Cribb, 2010 and Foley et al., 2011). Ecosystem damages aside, current intensive agricultural systems rely on non-renewable resources (fossil fuels, land, and minerals) that are being exhausted and inefficiently employed (IBID). A projected 33% population growth – from 7 billion today to 12 billion by 2100 (Gerland et al., 2014) – with concurrently increased global economic activity (Price Waterhouse Cooper, 2010) will challenge the global agriculture system to produce more food with less resources while minimizing environmental impacts synchronously. Recent trends have been discordant with these ambitions, showing reduced growth in yields per unit production factor (land, fertilizer, etc.) in a number of countries as well as increased gross, non-renewable resource consumption from 1985–2005 (Foley et al., 2011 and Tilman and Clark, 2014).

Meat and dairy products are central to food-related impacts, having large environmental burdens including agricultural land degradation due to overgrazing, surface and groundwater contamination from uncontrolled waste management, biodiversity loss through the proliferation of grazing land (and land for feed production), and greenhouse gas (GHG) emissions related to livestock digestion (particularly ruminants) (Asner et al., 2004, Cribb, 2010, FAO, 2006, Modernel et al., 2013 and Nijdam et al., 2012). Due to the inherent inefficiencies of producing biomass at higher trophic levels (McMichael et al., 2007 and Pimentel and Pimentel, 2003), livestock production also requires calorific inputs amounting to 40% of global grain production (IBID; Foley et al., 2011). These feed requirements have environmental impacts embodied within their production, exacerbating the direct environmental disturbances of animal husbandry. Accounting for pastures and animal feed, livestock production is estimated to commandeer nearly one third of global, ice-free surface area (McMichael et al., 2007). These environmental pressures and land constraints are key issues if the predicted global animal product demand doubles from year 2000 levels by 2050 in response to population and economic drivers (FAO, 2006, Feeley and Machovina, 2014, McMichael et al., 2007 and Tilman and Clark, 2014).

Technological improvements to livestock production can mitigate some environmental harm, but eco-efficiency gains have failed thus far to mitigate net environmental impacts. Conversely, tackling this challenge on the demand side by reducing meat and dairy consumption has been championed as a way to improve the environmental integrity of nourishing humanity (FAO, 2006, Foley et al., 2011 and Tilman and Clark, 2014). This approach has been most salient in the United States Department of Agriculture's (USDA) 2015 dietary guidelines (2015). Indeed, environmental audits using life-cycle assessment (LCA) have shown that, low meat, vegetarian (no meat), and vegan (no meat or dairy) diets can have significant environmental benefits in comparison to prevailing dietary trends in wealthy countries (see Table 1). LCA estimates the resultant environmental impacts in a number of pertinent indicators from the supply chain (raw material extraction, processing, use, disposal, and related transport) required to deliver a product or service. These studies have shown univocally that vegetarian and vegan diets have reduced GHG emissions over standard omnivorous diets in a wealthy context. For other environmental impacts, LCA conclusions vary, showing that reduced animal product consumption reduces all accounted environmental impacts (Baroni et al., 2007), reduces particulate matter formation and land occupation (Saxe, 2014) or, conversely, exacerbates water consumption (Meier and Christen, 2013).

Table 1 - Previous environmental life cycle assessments of dietary habits

Reference	Country	Impacts Included				GHG Reduction (% change relative to omnivorous diet)	Other Comments
		Non-toxic	Toxic	H_2O Use	Land Use		
Heller and Kaoleian	United States	X				Vegetarian: 33%	

(2014)						Vegan: 53%	
Saxe (2014)	Denmark	X		X	X	New Nordic Diet: 30% w/ reduced transport: 35% w/ organics: 32%	- land occupation reduction with reduced meat diet. - organic content of diet raised particulate matter and land occupation impacts.
Scarborough et al. (2014)	United Kingdom	X				Medium Meat: 21% Low Meat: 35% Pescatarian: 46% Vegetarian: 47% Vegan: 60%	- comprehensive diet survey used
van Dooren et al. (2014)	Netherlands	X			X	Vegetarian: 21% Vegan: 37%	
Meier et al. (2013)	Germany	X		X	X	Vegetarian: 25% Vegan: 50%	- water use inversely proportional to meat intake
Berners-Lee et al. (2012)	United Kingdom	X				Vegetarian: 22% Vegan: 36%	
Roy et al. (2012)	Japan	X				Not Applicable	
Saxe (2012)	Denmark	X				New Nordic Diet: 6% w/ optimization: 27% Vegetarian: 27%	- select local, organic and meat consumption performed equal to vegetarian
Macdiarmid et al. (2012)	United Kingdom	X				Reduced meat: 36%	- unrealistic sustainable diet achieved 90% reduction in GHGs
Tukker et al. (2011)	Europe	X	X			Reduced red meat: 8% Mediterranean: 5%	
Baroni et al. (2007)	Italy	X	X	X	X	Vegetarian: 74% w/ organic: 87% Vegan: 90% w/ organic: 97%	- ubiquitous superior performance across all impact categories with reduced meat
Wallén et al. (2004)	Sweden	X				Reduced meat: 5%	

Though compelling, the veracity of environmental benefits from reducing meat consumption has shortcomings. The common application of single issue indicators, chiefly the GHG burdens, dominates relevant literature (Berners-Lee et al., 2012, Heller et al., 2013, Roy et al., 2012, Saxe et al., 2012 and Wallén et al., 2004), running the risk that reduced meat diets may increase other environmental impacts (i.e. environmental burden shifting). Moreover, where expanded indicator sets covering more types of environmental pressures have been applied, paucities exist in illuminating the latent assumptions within the LCA framework and their potential consequences. Baroni et al. (2007) explored this theme with their analysis of the robustness of LCA results of dietary shifts to changes in assessor concern for different environmental impacts, both in terms of impact type and time-frame, finding that in general little change was seen with shifting assessor perspective. Aside from nascent investigation, there has been sparse discussion surrounding how the choice of indicators included in and LCA or the way that chemicals are modeled in the environment might affect dietary study results. Moreover, environmental efficacy has been ascribed to dietary choices even when the compared diets perform within the margins of error typically applied to LCA assessments.Herrmann et al. (2014) note that the margin of error can be significantly larger than the 10% uncertainty used in some of the reviewed studies. Lastly, with the exception of Saxe's work, studies have utilized attributional LCA models which are not representative of production systems at play with market forces (Plevin et al., 2014). Clearly, even within the LCA framework which strives for scientific objectivity, subjective values influence assessments, although this is only one aspect of the power of personal preferences in the discussion of the sustainability of diets.

A number of food related ethical discussions have gained momentum the past 20 years (Mepham, 1996) such as livestock welfare, food waste, food safety, food security, rural development, agricultural practices related to conventional, organic, and transpersonal agroecology, crops as biofeuls and the use of biotechnology as breeding tools on both animals and plants. "Sustainability" can mean many things in regard to food (Gamborg and Sandsøe, 2005) and various aspects of sustainability can easily conflict leaving one to choose between different values (e.g. land use efficiency and animal welfare) (Gjerris et al., 2011). Consumer-driven sustainability on food thus faces serious challenges, since it can be confusing as a consumer to determine the more sustainable choice. This is both because of scientific uncertainty, but also because of different and value-driven definitions of what "sustainability" actually is (Gjerris et al., 2016). From an environmental perspective, sustainability is roughly equated with humanity's stewardship of the environment in a way to not undermine its long- and short-term ability to provide natural resources, pollution assimilation and other ecosystem services to mankind, whilst concurrently supporting a meaningful proportion of the planet's wildlife and biodiversity. However, since sustainability is multi-faceted in nature (encompassing economic, environmental, social and institutional traits), the environmental performance concerning different diets is interconnected to discussions of food, culture, animals, humans relationship to nature, economics and values. Therefore, even though environment assessments (LCA, ecological footprint, emergy, etc.) are important to understanding and communicating environmental impacts related to diets, no assessment strategy completely covers all quantifiable (e.g. environmental and economic impacts) nor less-quantifiable (e.g. social issues) aspects of sustainability. In relation to this article the task therefore becomes to show what values drive different LCA methodologies to clarify the extent to which they affect the conclusion. Considering LCAs as value-neutral decision tools is precarious, as the values informing the political process used to develop LCA methodologies thus become hidden. Leaving decisions about sustainability to LCA experts does not make the decisions value-free, but merely ensures that it is the values of the experts that inform the decisions.

This paper presents an LCA comparing predicted environmental performance of average omnivorous Danish and conceptual vegetarian and vegan diets. Denmark provides an interesting case, because it has high per capita meat consumption (97 kg/a, 11th globally) (The Economist, 2012), produces a significant portion of its consumed meat and dairy (FAO, 2014), and enforces stringent energy and environmental controls on agricultural production. Moreover, this paper utilizes the full suite of LCA indicators, consequential LCA modelling methodology, and supporting databases not yet used in literature for dietary assessments at the time of writing. This study also explores the normative nature of environmental assessments and deduces the tractability of LCA as tool for comparing diets, with a discussion of the axiological ethical positions implicit in modeling environmental impacts using LCA.

Materials and Methods

LCA framework

LCA attempts to quantify the materials and energy consumed, and chemicals emitted to the environment during resource extraction, manufacturing, distribution, use, and end-of-life stages of a product/service (Guinée et al., 2002). LCA utilizes the *functional unit*concept in comparing different food products. In essence the functional unit strives to provide a common basis of comparison between different means of achieving the same end, strictly defined as a service that the assessed system(s) must fulfill (e.g. provide containment for a certain volume of liquid). The amounts of mass and/or energy required to meet that functional unit (e.g. the amount of ceramic or polystyrene needed to hold the amount of fluid defined by the functional unit) are called the reference flows.

Through the entire system life cycle, the LCA accounts exchanges (resource consumptions, energy, pollutant emissions) between different, well-defined environmental compartments (water, land and air in their different permutations) and the system (herein the 'product-system') providing the functional unit. Summing like flows of these resource inputs or pollutant outputs along the entire supply chain, a system inventory is generated for the total resource needs and pollution loading related to the functional unit. Lastly, the chemical and energy exchanges between society and environment are converted to environmental pressure potentials pertaining to the environment and human health. Resources used by the system and pollutants leaving the system are assessed for contributions to specific environmental problems (e.g. climate change, freshwater ecotoxicity, etc.) or resource scarcity issues (e.g. metal depletion). These scores represent *estimated* contributions to environmental and resource challenges imparted by the product system to fulfill the functional unit, called impact potentials (IPs).

IPs can be characterized at midpoint or at endpoint. Endpoint indicators model the entire impact pathway up to damages to 3 areas of protection (ecosystem quality, human health, and natural resources). Midpoint indicators stop earlier than endpoints along the cause-effect chain. For example, climate change impacts at the midpoint level are measured as the equivalent amount of carbon dioxide emitted to the atmosphere by the product system, while the product systems contribution to the damage to human health endpoint accounts for the estimated rise in temperatures and resultant loss of healthy years of living from disease, sea level rise and other factors. An assessor starts with the midpoint IPs and uses conversion factors which weight the contributions of that midpoint to a given endpoint category in order to create common unit that can be summed.

Though endpoint indicators are more meaningful from a decision-making perspective, they are less certain than midpoint (Hauschild, 2005). Lastly, the endpoint IPs themselves can be further weighted and summed to generate a single score, though this is hindered by subjectivity regarding the how the weighting should be done. LCA has seen increased harmonization in recent years, with the basic requirements outlined by the ISO 14040 series of standards, and detailed best practices guidance in the International Reference Life Cycle Data System (ILCD) handbook (EC, 2010, Finkbeiner et al., 2006 and Owsianiak et al., 2014). Lastly, LCA has seen increased application to food in recent years, viewed as an effective assessment method for environmental impacts food products.

Functional unit and scoping of the assessed diets

Different functional units for food LCAs have been proposed in the past: they can relate to agricultural areas, entire farms, a single livestock unit, quantities of food produced or consumed, nutritional values of meals (Haas et al., 2000). In this study, the primary function is considered to be the supply of adequate energy and nutrient levels to an adult person. The functional unit in this study will be taken as the provision of 2000 kcal per day of food excluding beverages aside from dairy. The United States Department of Agriculture recommends a daily calorific intake of 2000 kilo calories (kcal) per average adult (weighted for gender and age) (Venti and Johnston, 2002), with this standard adopted throughout Europe (Meier and Christen, 2013 and Van Dooren et al., 2014). It should be stressed that consuming 2000 kcal per day does not automatically equate to a nutritionally adequate diet. The inclusion of other nutritional metrics to ensure compatibility of the compared systems would improve the study (Heller et al., 2013), but as a rough guide for nutritional equivalency, calories suffices for the study at hand.

Three dietary patterns are assessed: the average Danish diet, and two recommended diets – an ovo-lacto vegetarian diet (no meat consumed, herein 'vegetarian') and a vegan diet (no meat or dairy products consumed). The scope of the assessment will stretch from the extraction of the raw materials necessary for the system up to the manufacture and production of the food products, with all processes beyond agricultural production excluded. Though this will underestimate total environmental impact by excluding processes downstream from the farm, it has been shown that food production is the dominant contributor to food-related environmental burdens (Davis et al., 2010, Meier and Christen, 2013 and Roy et al., 2012). For the use stage it has been shown that the processes of refrigeration and transport are typically the most important activities. Food miles tend to contribute marginally towards final environmental burdens, excepting cases involving air transport or long-distance refrigeration (Born and Purcell, 2006). Refrigeration itself, both in-store and at-home, can also be important contributors to life-cycle energy consumption and environmental impacts, though this is uncommon and not a priority in affecting food system sustainability. Furthermore, the impacts from cold storage speak more about the supporting energy system than the dietary choices themselves (Garnett, 2011). With regard to the disposal stage, impacts related to the incineration or composting of organic waste, both representative for Denmark, are not deemed to vary significantly between the three diet systems analyzed in this study. Thus, truncating their life cycles should not impact their comparative environmental performance. Finally, packaging is excluded from the assessments. The variety of possible packaging and cooking methods precludes sensible modelling, their inclusion adding marginal completeness in terms of impacts at the price of model robustness (Muñoz et al., 2010).

Data sources and inventory settings

The assessed diets were constructed from two sources. The standard Danish omnivore diet was taken from Danish consumption surveys for 2003 to 2008 and scaled from 10 MJ supplied energy to the functional unit (DTU Fødevareinstituttet, 2010). For the vegetarian and vegan diets, where actual consumption data was lacking, the recommended vegetarian and vegan diets were based on the 'Vegetarian food guide pyramid' (Loma Linda University – School of Public Health, 2008), which in turn relied on the US Department of Agriculture's nutritional guidelines (Haddad et al., 1999 and Venti and Johnston, 2002). The recommended diets list the required servings of broad food groups (e.g. whole grains, legumes, and soy, etc.) to meet the nutritional requirements of a balanced 2000 kcal/day diet. The broad food groups were disaggregated into the individual food components found in the Danish diet (e.g. the food group 'fruits and vegetables' is broken down into the food items like: 'tomatoes, cucumbers, and peppers'). The ratios of different food products available to the average Danish consumer according to Statistics Denmark food balance sheets (2014) were maintained for the vegetarian and vegan diets, but scaled to the amount required to meet recommendations for 'vegetables' in the food pyramid (this was done for all food groups). As such, the conceptual vegan and vegetarian diets reflect Danish consumer habits assuming that the food balance sheet expresses consumer demand. Moreover, in keeping with the system boundary of the farm, certain foods had to be dissected to their base agricultural constituents (e.g. bread was converted to grains), with the exception of vegetable oils from complex bio-refineries.

Food losses occur due to pests, damage during harvesting, processing losses from aesthetic or functional quality control, rough handling and spoilage during distribution, and at the retail and consumer due to further spoilage (FAO, 2011). Farm losses were internalized within individual modelling processes, since these scale total inputs and outputs at the farm to mass of product delivered at farm gate. Post farm-gate loss factors (in an OECD context) of 8%, 19%, 31%, 26%, and 32% for meat, dairy products, cereals, fruits and vegetables, and roots and tubers, respectively were taken from FAO (IBID) and applied to the diets. As such, the reference flows in Table 2 are inflated above actual consumption, representing demand at farm-gate necessary to supply the 2000 kcal/day for the given diets. Calculations are outlined in Supplementary Material S1 and S2.

Table 2 - Food demands at farm gate to meet a functional unit of 2000 kcal/day for the three considered diets and associated processes used in modelling. Diets do not include drinks (barring dairy) and vegan and vegetarian have high water content in foods consumed.

Food Item	Omnivorous (g/day)	Vegetarian (g/day)	Vegan (g/day)	Process (All ecoinvent 3 unless other sources are listed)

Dairy and Eggs				
Milk	278.4	449.8	-	(LCA Food, 2007)
Cream	31.3	-	-	(Weidema and Schmidt, 2014)
Creme fraiche	7.8	-	-	IBID
Butter	6.0	-	-	IBID
Cheese	28.2	19.1	-	IBID
Eggs	14.5	31.0	-	(Nielsen et al., 2013)
Meat				
Beef and veal	47.7	-	-	(Nguyen et al., 2010)
Edible offals of cattle	1.4	-	-	IBID
Pig meat	54.2	-	-	(Reckmann et al., 2013)
Edible offals of pigs	1.9	-	-	IBID
Poultry meat	22.7	-	-	Chicken for slaughtering
Mutton and lamb	2.0	-	-	Goat for slaughtering
Grains				
Wheat flour	141.6	271.0	316.2	Wheat
Durum wheat e.g.	15.1	0.0	0.0	Wheat
Rye flour	33.8	21.7	25.4	Rye grain, rye production
Oat-meal	24.7	0.0	0.0	Wheat
Rice and rice flour	15.8	144.9	169.1	Rice, production
Potato flour etc.	2.0	0.0	0.0	Potato
Other flour and groats, etc.	24.0	0.0	0.0	Wheat
Fruits and Vegetables				
Potatoes	115.6	238.4	238.4	Potato
Cucumbers	25.3	47.1	47.1	Cucumber
Spring-white cabbage	9.0	16.7	16.7	Cabbage white
Spring-red cabbage	9.0	16.7	16.7	Cabbage red
Brussels sprouts	0.6	1.1	1.1	Radish
Broccoli	11.8	11.0	11.0	Broccoli
Cauliflower	11.8	11.0	11.0	Cauliflower
Chinese cabbage	6.8	11.0	11.0	Cabbage red
Leeks	6.8	12.6	12.6	Celery
Beetroots	5.4	10.1	10.1	Radish
Celery	2.6	4.8	4.8	Celery
Carrots	44.5	82.8	82.8	Carrot
Onions	31.2	58.1	58.1	Onion
Lettuce	25.0	46.6	46.6	Lettuce
Tomatoes	106.2	272.3	272.3	Tomato
Cherries sour and sweet	6.6	16.9	16.9	(Carlsson-Kanyama and Emmenegger, 2000)
Strawberries	9.7	24.8	24.8	(Gunady et al., 2012)
Apples	169.9	435.6	435.6	Apple

Pears	23.9	61.2	61.2	Pear
Protein Substitutes				
Beans	0	135.1	135.1	Fava bean
Tofu	0	94.6	94.6	(Ercin et al., 2012)
Soy Beverage	0	32.4	32.4	(Ercin et al., 2012)
Peanuts	0	20.3	40.5	(University of Arkansas, 2012)
Cashews	0	20.3	40.5	(Figueiredo et al., 2014)
Oils and Sugar				
Vegetable Oil	0.0	7.6	7.6	(Stevenson, 2014)
Margarine	30.9	7.6	7.6	(Nilsson et al., 2010)
Sugar	30.1	0.0	0.0	Sugar from beet

System modelling

Two types of LCA modelling frameworks exist, namely the attributional and the consequential modelling, the choice of which has been a continuous source of polemic in the LCA community (e.g. Ekvall and Weidema, 2004 and Weidema, 2003). Consequential LCA differs from attributional LCA in two main ways: (1) the processes encompassed in the study are those which are most likely to respond to a change in demand, and (2) the co-product allocation is avoided by system expansion (Schmidt and Weidema, 2008). In this study, we have opted for a consequential modelling to reflect the environmental consequences that the change in diets may imply on the systems within and outside the primary agricultural processes, e.g. market reactions to proposed future consumption (ex-ante modelling). This approach is also in compliance with the ISO14044 requirements (ISO, 2006). For instance, in Denmark, butter (a by-product of milk) requires milk fat, which is re-allocated from high fat cheese and powdered milk production. Thus, when butter is demanded, powdered milk manufacturers substitute palm oil for milk fat, while high fat cheese production decreases forcing consumers to purchase other comparable fats (low fat cheese). Thus, in a consequential model butter is modeled as the amount of palm oil and low fat cheese produced in response to market demand for butter which are then translated into estimated environmental impacts (Weidema and Schmidt, 2014).

The implementation of the consequential modelling was facilitated by the use of the ecoinvent 3.1 database, which exist in 2 versions dedicated to attributional and consequential modelling, respectively (Weidema et al., 2013). The consequential database, containing inventories of resource consumption and pollutant releases for the different foodstuffs, was therefore utilized in the study. In conjection with LCA software, the database can model interactions with other systems by use of marginal data, which model supplies of products by taking a mix of all unconstrained suppliers in the market, i.e. those suppliers who can respond to the next unit of demand for that good in the market (Weidema et al., 2013). This database is deemed to be a marked improvement over those utilized in earlier dietary comparison studies, since it includes an expanded set of food production processes and utilizes a full-fledged consequential LCI modelling framework. Where appropriate processes were lacking in ecoinvent 3, custom processes were built using inventories from reliable sources such as peer review LCAs or the Danish LCA Food database (Bengoa, 2005, Cederberg et al., 2009, Meier and Christen, 2013 and Nilsson et al., 2010). These were kept consistent with the consequential modelling by using system-expansion with marginal data, where necessary. The ecoinvent processes and data sources utilized for custom processes are outlined inTable 2. Breakdowns of custom processes are in Supplementary Material S3.

Typically relevant in the modelling of agri-food systems, indirect land use change is defined as the life cycle consequences of the land use in the analyzed system, e.g. deforestation or cropland intensification taking place as a result of the change in demand from the system (Schmidt et al., 2015). The inclusion of indirect land use change (iLUC) effects may alter the IPs of an LCA through increased GHG emissions and biodiversity loss from deforestation (e.g. Dalgaard et al. (2014)) potentially changing the best performing product-system, and it is widely accepted that the problems related to iLUC should be integrated into decision-making (Schmidt et al., 2015). However, despite the recent release of frameworks for performing iLUC, e.g. Schmidt et al. (2015), there is yet no consensus on the approaches to integrate iLUC into

LCA modelling, which is still the source of debate, particularly in the assessments of biofuels (Finkbeiner, 2014,Finkbeiner, 2013, Munoz et al., 2014 and Schmidt et al., 2015). For this reason and due to the lack of insights into indirect land use change mechanisms triggered by the dietary changes, as reflected in the review by (Hallström et al., 2015), iLUC effects were not considered in the present assessment. As also recommended by Hallström et al. (2015), this important source of uncertainties, of which it is difficult to predict the influence on the results of the study, should however be addressed in future studies.

Impact assessment methods

There exist a number of competing life-cycle-impact assessment (LCIA) methodologies for modelling midpoint and endpoint IPs in LCA. The dissimilarities come from the varying choices used for modelling how chemicals disperse through the environment and to what extent they affect encountered organisms (Hauschild et al., 2012). The ReCiPe 2008 methodology was selected as it covers the whole spectrum of relevant environmental indicators at both midpoint and endpoint levels, and includes the possibility for differentiating across three cultural perspectives, namely the egalitarian, individualist, and hierarchist perspectives. The egalitarian perspective is sensitive to all environmental impacts (long and short term), uses preventive thinking in assessing pollutants, and aims for minimizing society's impacts on the ecosphere. Opposing this is the *individualist*, which is concerned with current environmental impacts within their lifetime. This assumes that technological progress can solve eventual environmental woes and that ecosystems are resilient against human intervention. The hierarchist lies between these two representing a intermediary (Goedkoop et al., 2009). Table 3 outlines the indicators used in ReCiPe and how the different cultural perspectives view them. It should be noted that ReCiPe's water scarcity and land use indicators were not used here as more nuanced methods were deemed necessary for the assessment.

Table 3 - Assumptions behind cultural perspectives in ReCiPe 2008 (Goedkoop et al., 2009)

Midpoint Indicator	**Assumptions at midpoint level**			**Assumptions moving from midpoint to endpoint**		
	Egalitarian	**Hierarchist**	**Individualist**	**Egalitarian**	**Hierarchist**	**Individualist**
Climate Change	500 year time horizon	100 year	20 year	no societal adaptation, high human health impacts and biodiversity loss	medium societal adaptation, mean human health impacts and biodiversity loss	full societal adaptation, low human health impacts and biodiversity loss
Ozone Depletion	Identical			Identical		
Terrestrial Acidification	500 year time horizon	100 year	20 year	500 year time horizon	100 year	20 year
Freshwater Eutrophication	Identical			Identical		

	Marine Eutrophication	Identical			Identical
	Human Toxicity	Infinite time horizon, all exposure routes for all chemicals, chemical toxicity considered	Same as egalitarian, except 100 year time horizon	100 year time horizon, limited exposure pathways for metals, selected chemical toxicity considered	Identical
	Photochemical oxidant formation	Identical			Identical
	Particulate matter Formation	Identical			Identical
	Terrestrial Ecotoxicity	Infinite time horizon, all exposure routes for all chemicals, chemical toxicity considered	Same as egalitarian, except 100 year time	100 year time horizon, limited exposure pathways for metals, selected chemical toxicity considered	Identical
	Freshwater Ecotoxicity	Identical to Terrestrial Ecotoxicity			Identical
	Marine Ecotoxity	Infinite time horizon, all exposure pathways possible	Same as egalitarian	100 years, limited exposure pathways for some chemicals	Identical
	Ionising	100 000	Same as	100 year time	Identical

Radiation	year time horizon	egalitarian	horizon			
Mineral Resource Depletion	Identical			Identical		
Fossil Fuel Depletion	Identical			Technology will slowly substitute fossil fuels	Same as egalitarian	Technology will quickly substitute fossil fuels

Results

Table 4 provides the impact indicator results for the three diets in terms of percentage difference from the omnivorous diet. Dark grey indicates the worst performing diet for that indicator, black the medium performing diet (where applicable), light grey the best performing, and white a tie across all diets. In our assessment results of the diet with a 25% standard deviation assumed, whereby IPs with overlapping confidence intervals were assumed to have no appreciable difference. Minute dissimilarities were thus ignored and claims about superior diet performance could not be made based on these.

	Omnivorous			**Vegetarian**			**Vegan**		
Midpoint Impact Category	**I**	**H**	**E**	**I**	**H**	**E**	**I**	**H**	**E**
Climate Change	-	-	-	-56%	-46%	-38%	-70%	-60%	-52%
Ozone Depletion	-	-	-	-3%	-3%	-3%	0%	-1%	0%
Terrestrial Acidification	-	-	-	-64%	-65%	-66%	-79%	-81%	-81%
Freshwater Eutrophication	-	-	-	-6%	-7%	-6%	-24%	-24%	-24%
Marine Eutrophication	-	-	-	-33%	-33%	-33%	-72%	-72%	-72%
Human Toxicity	-	-	-	-33%	14%	30%	-53%	5%	25%
Photochemical Oxidant Formation	-	-	-	6%	6%	6%	0%	-1%	0%
Particulate Matter Formation	-	-	-	-47%	-47%	-47%	-60%	-60%	-60%
Terrestrial Ecotoxicity	-	-	-	-6%	-8%	0%	45%	41%	43%
Freshwater Ecotoxicity	-	-	-	2%	2%	2%	-1%	-2%	-1%
Marine Ecotoxicity	-	-	-	3%	1%	24%	-5%	-7%	20%
Ionization Radiation (human health)	-	-	-	66%	15%	14%	67%	15%	14%
Metals Depletion	-	-	-	11%	11%	11%	8%	7%	8%
Fossil Depletion	-	-	-	-18%	-18%	-18%	-22%	-22%	-22%
Water Scarcity Index		-			26%			31%	
Land Use		-			-67%			-78%	
Endpoint Impact Category	I	H	E	I	H	E	I	H	E

Human Health	-	-	-	-54%	-44%	-10%	-68%	-57%	-19%
Ecosystems Damage	-	-	-	-56%	-46%	-38%	-70%	-60%	-52%
Resource Depletion	-	-	-	-9%	-14%	-14%	-13%	-19%	-19%

Table 4 - Relative environmental performance of the different diets shown as percentage deviation from the omnivorous diet. Light grey indicates best, black medium, dark grey worst performing. Where two diets had the same performance in an indicator, they will share the relevant color. White indicates a tie across all diets. Note that a 25% divergence from the omnivorous diet does not guarantee superior performance in a category, since possible values may overlap.

The results mirror those of previous diet comparison studies, since they show a clear difference between the omnivorous and non-meat diets, with the latter showing superior performance in a number of categories (see light grey cells in Table 4). The source of the poor performance of the omnivorous diet is the reliance on animal based products, as outlined in the climate change impacts and freshwater eutrophication impacts in Table 4. Beef is particularly pernicious in that it requires large quantities of inputs (feed, water and land) and results in large amounts of digestive waste (affecting eutrophication), and greenhouse gases (Nijdam et al., 2012). In terms of compatibility with similar studies, climate change provides the best comparative indicator due to its ubiquity. Relative dietary climate change performance was within the ranges found previously (see Table 1). Climate change IPs also agreed with earlier studies for the omnivore; 4.27 kg CO_2eq/day compared to 4.1, 3.02 and 4.09 kg CO_2eq/day for the average Dutch, US MyPate and average French diets respectively (van Dooren et al., 2014) and 5.6 CO_2eq/day for the average Dane (Saxe, 2014), though low compared to other studies that included transport and processing impacts (Berners-Lee et al., 2012 and Saxe et al., 2012).included transport and processing impacts (Berners-Lee et al., 2012; Saxe et al., 2012).

Impacts from pollution at farm

For climate change, methane emissions from cattle increase the IPs of the omnivorous diet well beyond the error threshold; this is reasonable considering that bovine enteric fermentation accounts for 18% of global methane emissions (McMichael et al., 2007) and significant N_2O release (Nguyen et al., 2010). Livestock production also perturbs the environment through feces and urine, which contain ammonia and nitrates. This also contributes to acidification or particulate matter formation if evaporated, or marine eutrophication (Gliessman, 2015). Plant production contributes to these IPs through over fertilization, which can result in nutrients runoff into receiving waters, or tilling, which activates the production of gaseous NOx by soil bacteria (IBID). In the assessment, excrement from livestock was the dominant factor resulting in the declined performance of the diets with increased animal product intake. Freshwater eutrophication is a consequence of phosphorous release to freshwater bodies from both animal excrement and fertilizer runoff, with all of the diets having similar performance in this regard as shown in Table 4. Lastly, though animal waste strongly influences photochemical oxidant formation (smog), the vegan's higher consumption of greenhouse-produced cucumbers and tomatoes led to similar scale IPs due to external heating needs.

Impacts from agricultural production inputs

A number of IPs can be traced to the chemicals and energy consumed in food production. Ozone depletion IPs are linked to diesel used on farms, but also pesticide production, with no differentiation in diet performance. Fossil fuel based fertilizer impacts are the same for all systems, though the non-meat diets are borderline superior, which is logical due to the exorbitant feed requirements for animal production (FAO, 2006). For fossil depletion the 25% standard deviation may be too liberal considering the reduced uncertainty surrounding the modelling of fossil fuel consumption; allowing for defensible prima facie conclusions here. Land occupation is adversely impacted by the imported livestock feed requirements and grazing

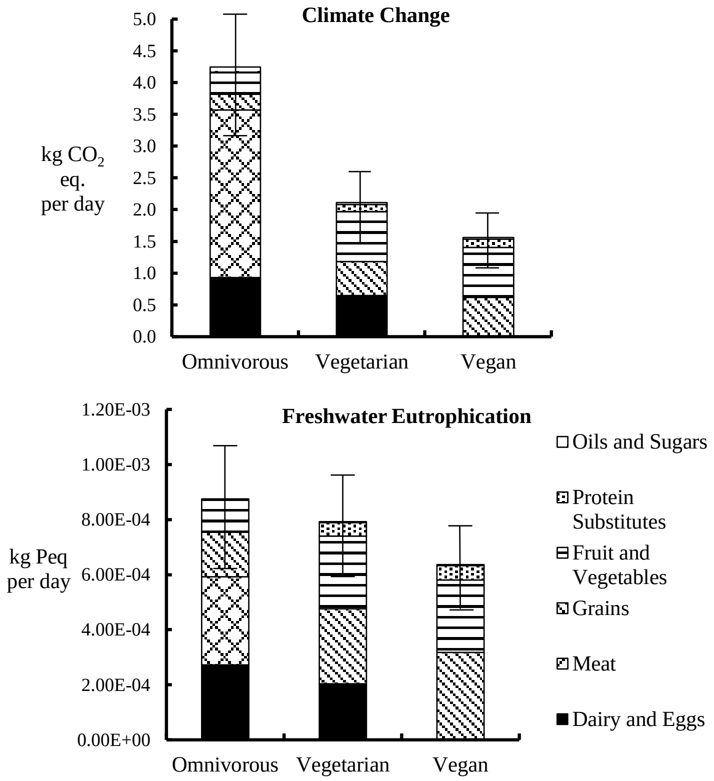

Figure 1 - Climate change and freshwater eutrophication midpoint IPs using hierarchist perspective in kg CO_2eq/day and kg Peq/day (Peq – phosphorous equivalents), respectively

territory (Foley et al., 2011), creating a large gap between omnivorous and non-meat diets. Of interest is that the meat-protein substitute, fava beans, contributed significantly to land occupation (~12%), which is of note since this is a proxy for all types of beans consumed by the meat-free diets. Moreover, the ecoinvent 3.1 inventory shows that land occupation is low compared to other potential LCIs (Abeliotis et al., 2013) by up to a factor of three, however overall results remained robust to this uncertainty when land occupation for the fava beans was increased by this factor (vegetarian and vegan land occupation IPs relative to omnivorous diet changed to −62% and −73%, respectively). No differences were noted for metal depletion across the different diets or ionizing radiation (primarily related to pesticide production) since they all are heavily reliant on these inputs. However, when ionization radiation is compared at the individualist level the meat-free diets have worse performance due to significant pesticide inputs to fruit and vegetable production (particularly apples), but these differences dissolve at the egalitarian and hierarchist level, as a longer timeframe for potential impacts are taken (see Section 'Difficulties with toxicity impacts'). Water impacts are also worse for the two non-meat diets, as a result of the large irrigation inputs (exacerbated by large losses from runoff), echoing the findings of Meier and Christen (2013) and hinting a need for future research in more accurately quantifying impacts in this category. Lastly, animal feed requirements compound climate change IPs through fossil fuel based fertilizer needs and deforestation associated with soy protein feeds (Flysjö et al., 2012).

Difficulties with toxicity impacts

Ambiguous results were also illustrated in midpoint categories dominated by inputs to agricultural production. Pesticides dominated all of the toxicity IPs; a natural consequence of the toxic properties for which they are produced. Non-meat diets had lower human toxicity IPs, though only from an *individualist* perspective, since this viewpoint focuses on malicious substances with short term impacts (i.e. some chemicals are bad), while the other perspectives account for more chemicals (i.e. many chemicals are bad) (Goedkoop et al., 2009), thus blurring the performance of the diets as more pesticides are considered, with human toxicity IPs articulating this clearly. This trend can also be seen for marine ecotoxicity IPs where the egalitarian results for the non-meat diets are highest relative to the other methods, though no discernible difference was seen between diet choices. For terrestrial ecotoxicity, the vegan diet performed worst, though this was a shortcoming of the LCIA method. Soybean feed coproduces soybean oil resulting in avoided palm oil production on the market, with the avoided IPs credited to the feed. Pesticides used in palm production are included in ReCiPe LCIA methodology, while some of those for soy are not. Thus, the avoided palm impacts outweigh the soy impacts, producing a net negative IP. Thus, the animal product diets (which include feed) appear to perform better. However, with LCIA methods – using more complete chemical inventories (e.g. UseTox) (Owsianiak et al., 2014) – contradicting results are seen. Such deficiencies accentuate the difficulties of chemical toxicity modelling within LCA, forcing the question of whether a 25% uncertainty level is valid for toxicity impacts. Moreover, this modelling artefact explains why the vegan diet was borderline worse for fresh- and marine water eco-toxicity.

Endpoint impacts

Table 4 displays the ReCiPe IPs aggregated and assessed at the endpoint level, providing a comparative overview of the diets' IPs. The endpoint results succinctly showed what the midpoints communicated that the average Danish diet has larger IPs for ecosystems damage and human health than the vegetarian or vegan diets. However, the latter has ambiguous results when seen from the egalitarian perspective, due to the uncertainties in the toxicity modelling (see Section 'Difficulties with toxicity impacts'). Lastly, no discernible difference was observed for resource depletion, although – as discussed above – fossil fuel use appears higher in the livestock dependent diets. Notwithstanding differences at midpoint level, both non-meat diets performed equally at the endpoint.

Discussion

Results supported those of other studies in that the standard omnivorous diet performed poorly compared to model vegetarian and vegan diets. This was evident by the comparable performance of the meat-free vegetarian diet to the meat- and dairy-free vegan diet. Only within the realms of toxicity and eutrophication can we see appreciable divergence between these two choices, due to pesticide regimes and animal waste, respectively. Though conclusions could be drawn about the comparative performance for some specific IPs, ambiguity is present in others. Fig. 1 exemplifies this with the clear distinction between the omnivorous and non-meat diets for climate change, but inconclusive results for freshwater eutrophication. Table 4 takes this further by displaying both the various trade-offs between the dietary choices and the dependence of the results on the perspective of the assessor, thus challenging the objectivity of the LCA process, and necessitating a re-inspection of the cardinal ethical precepts embedded within the methodology. These ethical issues are related not only to the cultural perspectives applied in this specific study, but also to questions about the use of the LCA as a decision-making tool. Also, the ethical values that this decision-making tool entails, opens a wider debate, which must be taken up. This re-evaluation may not eradicate the elucidated uncertainties, but will at least support the validity of using LCA as decision support tool.

Sensitivity of results to ReCiPe cultural perspectives

We have taken some ethical aspects of food production and its ability to utilize differing cultural and ethical perspectives (individualist, hierarchist, and egalitarian) in characterizing environmental impacts into consideration with ReCiPe 2008. One of the purposes of the cultural perspectives is to allow for results interpretation in the face of uncertainty. For instance, with toxicity IPs where there is higher uncertainty due to challenges of adequately including the toxilogical properties in LCA models of all of the chemicals in commercial use, the user can adopt an ethical perspective that deals with this data gap in a way that aligns with their thinking of nature how nature works and the potential risks of underestimating IPs.

In ReCiPe the *egalitarian* is most worried by environmental impacts (long and short term), the individualist the least, while the hierarchist represent a middle view, although with a valence towards the egalitarian's stance (see Table 4) (Goedkoop et al., 2009). According to the cultural theories as presented here, egalitarians tend to perceive nature as an ephemeral entity, highly sensitive to perturbations, whereas hierarchists view nature as surprising in the sense that it "may hide the response when exposed to stress and at some time flip to another state in a more or less irreversible manner." (Finnveden, 1997). Obviously, the moral theory most closely related to egalitarians is egalitarianism (equal treatment for all agents affected by a situation) as it is reflected in environmental ethics such as Deep Ecology (Naess, 1973). The moral theory that might be related to the hierarchist point of view is utilitarianism, i.e. this posits maximizing happiness and minimizing pain. Finally, individualists tend to perceive nature as resilient in the sense that it will vacillate from its baseline state when exposed to stress, but can return to the baseline state if the stress is lessened or removed (Finnveden, 1997 and Shwarz and Thompson, 1990). The moral position of libertarianism and especially "Green libertarianism", which opposes regulation and advocates the maximum freedom of individual action compatible with equal freedom for all (Davidson, 2009), invoking IsaiahBerlin's notion of negative liberty (1964), is strongly related to the individualist viewpoint is.

As shown above, the LCA results were robust against the application of these attitudes, with the exception of conclusions about impacts to human health through toxicity and ionizing radiation (midpoint level) and damages to human health (endpoint level). Moving from the individualist to the hierarchist and egalitarian standpoints resulted in the accounting of more and more uncertain or long-term environmental impacts. This was illustrated by the human toxicity midpoint IPs where the individualist perspective narrowly focused on chemicals with well documented and/or acute toxicities, while the other outlooks included less immediately harmful chemicals and took a precautionary stance towards those *suspected* of being toxic (Goedkoop et al., 2009), with the general effect of blurring the comparative performance of the diets with this indicator and the endpoint human health IPs. Toxicity IPs are some of the most difficult impacts to predict with LCA, since actual toxicological impacts are extremely dependent on the unique assimilative capacity of the receiving body (ecosystem or animal). This is a major methodological challenge for an assessment tool such as LCA that does not yet spatially disaggregate chemical releases (IBID). Thus, the individualist's skepticism about the toxicity impacts may be warranted, especially when LCA is applied to numerous foods from a global market composited into a single diet.

In general all of the midpoint categories were robust against the cultural perspectives aside from the two exceptions noted above, with the further consequence of these exceptions promulgating through the LCIA calculations to affect the endpoint IP damage to human health. Table 5 outlines how the cultural perspectives affected those IPs that were sensitive to them.

Midpoint Impact Category	**Effect of cultural perspective**
Human Toxicity	Inclusion of increasing number of chemicals included in the LCIA when moving from I to E perspective made the model more sensitive to the herbicides and pesticides in vegetarian and vegan diets, erasing differences in environmental performance of the diets
Ionization Radiation (human health)	Longer timeframe of impacts considered when moving towards H and E perspectives meant that the impacts from more of the pesticides and herbicides used attenuated differences between the omnivorous and non-meat diets
Endpoint Impact Category	
Human Health	The impacts of the increased sensitivity of the human toxicity and ionization radiation (human health) affected the conversion from midpoint to endpoint IPs introducing ambiguity between dietary performance when the E perspective was adopted

Table 5 - Midpoint and endpoint IPs sensitive to cultural perspective chosen in the comparison of diets

The implicit values embedded within LCAs

The framework of LCA is built upon a number of implicit values. This section attempts to lift the veil of the many philosophical and ethical principles that an LCA assessor or LCA study commissioner accepts in choosing LCA as method to assess the environmental performance of product systems.

LCA methodology is predicated on the belief that we are able to predict environmental impacts. This belief is in itself based on our views of nature (Finnveden, 1997), which may differ considerably amongst both decision-makers and stakeholders. Our faith in LCA's ability to inform decisions hinges on the belief that nature is complex yet predictable rather than inherently random, influencing not only how we model IPs, but also whether LCA is even capable of providing answers to the fundamental questions being asked. If the general belief is that we are not able to predict environmental impacts, LCAs are nonsensical and one should seek alternative valuation methods that circumvent evaluating environmental damages (Finnveden, 1997).

If we accept that LCAs provides valuable insights for decision-makers, it is important to realize that LCA studies are grounded in different theoretical constructs and that the choice of assessment approaches entails ethical implications as well (Ekvall et al., 2005 and Finnveden, 1997). In our study, we used a consequential modelling approach, which falls into what Ekvall et al. (2005) term a "prospective life cycle assessment (LCA)", which provides information on the environmental consequences of individual actions in a dynamic system. This is contrasted with "retrospective" LCA's assessment of the environmental impacts in a static system without constrained suppliers. Ekvall et al. (2005) analyzed different LCA methodologies against different normative moral philosophy theories and found that each LCA type, as well as each of the moral theories, can be criticized from the alternative point of departure and that both prospective and retrospective LCAs had pros and cons. According to Ekvall et al. (2005) the use of prospective LCA is valid *if* the audience equates positive environmental outcomes with 'good' changes to a production system. It follows that decision-makers and people in general have differing opinions on what constitutes a good environmental action depending on their ethical values, since they may actually be indifferent to the state of the environment. For instance prospective LCA methodology is valid from a teleological framework, whereby consequences of an action are the criterion for success or failure (utilitarianism with its maximization of universal pleasure employs this thinking) (Finnveden, 1997), which is in opposition to deontological ethics which evaluates good or bad according to the principles underpinning actions and not necessarily outcomes (Ekvall et al., 2005).

Ekvall et al. (2005) found that "the sheer diversity of ethical theories makes it impossible (sic) to decide whether an environmentally good action is an action that reduces the environmental burdens of the total life cycle or an action with good consequences for the total environment". In this study, the former ethical foundation was implicit within the "prospective" LCA methodology that we used; whereby the proposed changes to vegetarian or vegan diets were assessed in terms of the environmental consequences of these actions relative to the status-quo, omnivorous Danish diet.

The fact that it is impossible to model the full consequences of an action in an LCA (or any model for that matter) has been noted as the most important limitation of prospective LCI methodology. Prospective LCA accounts only for simple causal relationships, whereas full outcomes depend on a variety of causal loops and delays. Often we do not know the significance of these excluded causal relationships or how well modeled outcomes accurately reflect reality (Ekvall et al., 2005). The issue at hand is that LCA practitioners endeavor to provide robust decision support, whilst being realistic about data and model weaknesses. How do we know if the results of an LCA are defensible and can agents make ethical decisions based on these results? As a rule, an LCA is meritorious if through judicious design, working within the limits of those aspects of external reality that can be known and modeled, and accounting for those aspects that are most salient to the model outcomes, it directs decision makers towards a reasonable facsimile of the outcomes of the modeled scenarios. The LCA in this paper was successful towards these ends, insofar that is transparent about model shortcomings while robustly identifying meat-free diets as viable alternative with superior environmental performance over the Danish status quo. Thus, people who equate actions that reduce impacts on the environment with 'good' actions would be justified in moving from omnivorous to meat-free diets in a Danish context.

Subjectivity in LCA modelling

This section departs from the previous in that these decisions do not relate to the choice of using LCA to assess environmental performance of diets, but those choices made by the LCA practitioner in developing their LCA that are based on the assessor's or study comissioner's values.

To start the user of the LCA must decide on the scope of the LCA, clarifying what aspects of the many facets of sustainability they will try to quantify. In the current study there was a pure focus on the environmental aspects, explicitly avoiding social and economic aspects of sustainability that could be addressed through the nascent social-LCA tool and the life-cycle costing methodologies, respectively. Making this choice implies either a low valuation of these sustainability aspects on the part of the LCA assessor or the belief that these issues are better handled within other assessment frameworks. Even if these sustainability aspects were assessed within their respective frameworks, there would still be an anthropocentric lean to the results due to the omission of issues related to animal welfare.

Although many strive to make LCA as objective, detailed, and scientifically robust as possible, it is well known that the use of LCA as a decision-making tool is not value free (Hellweg and Frischknecht, 2004 and Hertwich et al., 2000). For instance, the 25% error threshold employed here was based on the authors' professional judgment and experience that this is reasonable for the assessment, though a different threshold could have been used with nontrivial implications. The exclusion of post farm-gate impacts from the model, though grounded in previous findings, is a value-laden decision, whereby our focus on comparative performance implicitly eschews quantifying the complete environmental footprints of the diets (e.g. cradle to grave). The notion of absolute sustainability is likewise ignored, since the results cannot relate the food consumption of a typical Dane to the planet's seemingly limited ability to absorb impacts and continue operating in a manner amenable to human life. More crudely: we cannot determine whether the 'footprint' fits the 'shoe'. As such, this study adopts a weaker sustainability stance: We assume that acting to minimize current environmental harm is 'good', even in the absence of knowing whether this action is enough in an absolute sense. Ironically, one can end up employing ostensibly deep ecological principles through the egalitarian perspective to support opposing weak sustainability actions.

Moving from midpoint to endpoint in this study involved the acceptance of all of the weighting factors to aggregate to the three endpoint categories and their implicit assumptions, whilst the potential to move to a weighted single score, if taken, would have been imbued with values choices of the weighting factors and the belief that this is good scientific practice; decisions that are all loaded with implicit fundamental ethical and ideological judgments (Goedkoop et al., 2009). Hauschild (2005) notes that ethical values do not only come into play in the valuation step of LCA, but already in the definition of impact categories and how emissions are classified and characterized (e.g. toxicity in the individualist, hierarchist, and egalitrian perspectives in this study). This is most pronounced through the near ubiquity of carbon footprint as the preferred assessment in previous diet comparisons (see Section 'Introduction') focusing on climate change over other environmental challenges. One of the most important aspects of LCA, where societal and ethical values come into play, is the weighting of environmental impact potentials, as the weighting factor for an environmental impact reflects the importance of the impact category relative to the other environmental impact categories considered in the LCA (Hauschild and Barlaz, 2011). Accordingly, the determination of the weighting factors should therefore involve both an analysis of the causal relationships subject to the LCA as well as an analysis of the ethical values of the major stakeholders of the study who the LCA practitioner wishes to accept the result of the LCA. If the major stakeholders do not share the ethical values inherent in the weighting this can change the outcome of the LCA (Hauschild and Barlaz, 2011). This is especially important when one considers that LCAs can be funded by companies and industry groups (e.g. an association of a particular type of farmer) that might have a stake in presenting a certain outcome to the public, potentially leading to the weighting of selected midpoint IPs or the exclusion of others to achieve results that align with the aims of the funding entity.

LCA Aspect	**Note**
Implicit within LCA	
Use of LCA	Focus on environmental aspects of sustainability. Inclusion of social-LCA and life-cycle costing expand the scope of sustainability assessment, but still eschews discussions on animal welfare. Also implicitly believes that the behavior of nature is in many ways predictable and equates an environmentally preferable choice with the adoption of a

	set of technologies that provides a function with potentially lower burdens than other comparable sets of technologies.
Choices in LCA process	
Selection of IPs	Involves the valuation of available indicators and the prioritization of those included. Can be used to obfuscate poor performance of a product-system through the purposeful exclusion of those IPs where the system has negative performance. Funders of a study may influence this. Various levels of certainty and consensus in modelling methodology exist for different LCIA indicators. At the midpoint level only climate change, ozone depletion and particulate matter formation indicators are widely considered to be the most mature (Hauschild et al., 2013). Results in all other categories should be viewed with a higher level of skepticism and require significant divergence between assessed systems in those categories before conclusions regarding comparative performance should be drawn.
Use of cultural perspectives	Egalitarian, hierarchist and individualist perspectives prescribe to egalitarianism, utilitarianism and libertarianism, respectively. Can be used as a lens to deal with uncertainty in modelling by adopting precautionary principle or as a way to focus on short- to mid-term impacts.
Weighting of IPs	Moving to endpoint IPs or generating a single score (after normalization) both involve weighting which involves a subjective valuation of the importance of various IPs. Can be used to minimize the impacts of IPs and obfuscate poor LCA results for a system. Funders of a study may influence this.
Life cycle stages included	Choice of excluding life cycle stages (e.g. assessing from cradle to farm-gate) ignores full impacts and precludes any assessment of absolute sustainability

Table 6 - Overview of the ethical perspectives built within LCA and the subjective choices made while performing LCAs

The current case displays this clearly, as the toxicity and water scarcity index results remain ambiguous or even antagonistic to the general trend. One could easily reverse the conclusions of this study by employing single indicator methodologies focused on these IPs (à la carbon footprint) or through hefty weighting factors when moving towards a single indicator score. Relating this to the discussion of toxicity impacts in Section 'Sensitivity of results to ReCiPe cultural perspectives', the favored adoption of a global environmental indicator, such as carbon footprint, in comparing diets may be more appropriate considering the lack of spatial differentiation in LCA IPs. An adoption ofHeller et al.'s (2003) spatially disaggregated food product environmental assessment method may actually be better equipped to deal with other agricultural related IPs (erosion, eutrophication, etc.) than the traditional LCA tool. Moreover, the carbon footprint's cynosure is also product of a larger environmental community's valuation of climate change as the defining environmental issue of our epoch, requiring amelioration on ethical grounds.

Robustness of LCA on diets

On the whole it would seem that using LCA as a method to environmentally assess diets (or anything) is fraught with uncertainty, value judgments, and even value judgments about uncertainty, begging the question: does LCA show that switching to lower animal product diets reduces environmental burdens? The fact that this assessment, along with earlier diet LCAs, all point in the same direction hints either that these models are all similarly flawed, or their conclusions are substantiated. The former is unlikely considering the methodological variability employed across the studies (system boundaries, LCIA models, consequential vs. attributional methods, databases utilized, etc.), which would have identified large flaws in competing methods through contradictory results. We thus accept that movements towards vegetarian or vegan diets generally constitute environmental 'goods', but only if one is disposed to value pristine environmental state.

Though this study has shown that LCA does have a role to play in assessing select aspects of the sustainability of diets, the discussion has shown that there remain a number of challenges in the application of LCA to this domain. Table 6 sums up the value choices implicit within the LCA methodology and the subjective choices made by an assessor while performing an LCA in hopes of providing the reader with the tools to critically interpret LCAs of diets.

Conclusions

Assessing diets from an environmental perspective is a complex task. Technical difficulties aside, the value systems embedded within assessment methods question the objectivity of such an endeavor, as evidenced by the normative values embedded within LCA, and the various ways these judgments influence model outcomes. Accepting that LCA can be used to predict environmental impacts, the assessment found that the results were robust against changing the 'cultural perspectives' allowed within the ReCiPe 2008 LCIA methodology, adding credence – along with earlier studies – to the idea that shifts from diets with high meat intake towards vegetarian or vegan diets generally predicts positive environmental outcomes, with the exception of water scarcity, which was influenced by the higher grain, fruit, and vegetable intake of these diets.

While our results support the general argument for reducing food-related environmental impacts through behavioral changes, difficulties in assessing toxicity impacts with LCA were noted. These require further methodological development or different assessment tools for those impacts – preferably at the local level – to account for the idiosyncrasies of receiving ecosystems (e.g. environmental risk assessment) or containing larger inventories of agrochemicals (e.g. USETox). Moreover, following vegetarian or vegan diets should not be conflated with sustainable lifestyles, since one can adhere to a low meat diet while causing negative environmental impacts in other aspects of life (e.g. commuting long distances by private vehicle, frequent air travel, large dwelling, etc.) that more than negate the positive environmental impacts of food choices. Dietary habits are only one of many areas where individuals can actively reduce their ecological burdens (Gjerris and Gaiani, 2014).

It should be elucidated that polemical dietary shifts that completely eliminate meat or dairy products are not necessary to induce positive environmental change. Animal husbandry methods that are well situated within ecological cycles can be positive for the environment. However, these remain the exception, since ecologically destructive factory farming is still the conventional approach (Cribb, 2010). Saxe et al.'s (2012) work showed that a more environmentally focused omnivorous diet in a Nordic context (reduced food miles, strategic organic content, reduced ruminant consumption) could potentially have similar environmental performance to a fully vegetarian diet. However, given organic agriculture's typically lower yields, a societal scale change to consuming primarily organic agriculture though positive in terms of fossil fuel reductions and toxicity attenuation, would consume more of the quintessential, constrained agricultural resource: land (Seufert et al., 2012). Notwithstanding, even shifting diets away from beef consumption would provide considerable environmental benefits (Nijdam et al., 2012).

LCA is limited insofar as it is an environmental assessment tool that ignores numerous other issues surrounding food consumption. The positive health impacts of vegetarian and vegan diets (Singh et al., 2010) have been neglected here for instance, though these effects may also result from generally healthier lifestyle choices amongst their proponents (more active, lower rates of smoking, etc.) and not solely the diets (Chang-Claude and Frentzel-Beyme, 1993). Furthermore, active lifestyles also have their own related environmental impacts (sports facilities, physiotherapy centers, etc.) that warrant consideration if a complete assessment of lifestyles is performed. Issues of animal welfare have also been ignored here, even though these could have significantly changed our comparison of the vegetarian and vegan diets, likely supporting a switch to a vegan diet despite their generally similar environmental performances. Despite these exclusions, the evidence of the environmental benefits of lower meat and dairy consumption continues to mount, not only in Denmark, but also in countries with similar food cultures.

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S.1 – Omnivorous Danish Diet

Food Product	Consumption at plate according to Danish Household Survey[i] (g/day)[a]	Apparent consumption according to Danish Statistics[ii] (g/day) [year][b]	Breakdown of consumption within food group[c]	Edible Losses[iii] (post farm)[d]	Demand at farm for 2000 kcal (g/day)[e]
Milk Products	356	-	-	8%	-
Milk	-	253.4 [2011]	86.0%	8%	278.4
Cream	-	28.5 [2011]	9.7%	8%	31.3
Crème fraiche	-	7.1 [2011]	2.4%	8%	7.8
Butter	-	5.5 [2011]	1.9%	8%	6.0
Cheese	31	-	-	8%	28.2
Eggs	16	-	-	8%	14.5
Meat	105	-	-	19%	-
Beef and Veal	-	77.0 [2011]	44.0%	19%	47.7
Offals of cattle	-	2.2 [2011]	1.3%	19%	1.4
Pork	-	87.4 [2011]	50.0%	19%	54.2
Offals of pigs	-	3.0 [2011]	1.7%	19%	1.9
Mutton	-	3.3 [2011]	1.9%	19%	2.0
Game	-	1.9 [2011]	1.1%	19%	1.2
Poultry	22	-	-	19%	22.7
Cereals	212	-	-	31%	-
Wheat flour	-	157.3 [2010]	55.1%	31%	141.6
Durum wheat	-	16.7 [2009]	5.9%	31%	15.1
Rye flour	-	37.5 [2010]	13.2%	31%	33.8
Oats	-	27.4 [2010]	9.6%	31%	24.7
Rice and rice flour	-	17.5 [2009]	6.1%	31%	15.8
Other flour and groats	-	26.6 [2010]	9.3%	31%	24.0
Potato flour	-	2.2 [2010]	0.8%	31%	2.0
Vegetables	153	-	-	26%	-
Potatoes	94	-	-	32%[f]	115.6
Cucumbers	-	23.6 [2006]	14.7%	26%	25.3
Pepper	-	0 [2011]	0.0%	26%	9.0
White cabbage	-	8.4 [2006]	5.2%	26%	9.0
Red cabbage	-	8.4 [2006]	5.2%	26%	0.6
Brussels sprouts	-	0.5 [2006]	0.3%	26%	11.8
Cauliflower and Broccoli	-	11.0 [2006]	6.8%	26%	11.8
Chinese cabbage	-	5.5 [2006]	3.4%	26%	6.8
Leeks	-	6.3 [2006]	3.9%	26%	6.8
Beetroots	-	4.7 [2006]	2.9%	32%[f]	5.4
Celeriac	-	2.2 [2006]	1.4%	32%[f]	2.6
Carrots	-	38.1 [2006]	23.7%	32%[f]	44.5
Onions	-	29.0 [2006]	18.1%	26%	31.2
Lettuce	-	23.3 [2006]	14.5%	26%	25.0
Fruits	280	-	-	26%	-
Tomatoes	-	84.1 [2006]	33.6%	26%	106.2
Cherries (sweet and sour)	-	5.2 [2006]	2.1%	26%	6.6

Strawberries	-	7.7 [2006]	3.1%	26%	9.7
Apples	-	134.5 [2006]	53.7%	26%	169.9
Pears	-	18.9 [2006]	7.5%	26%	23.9
Sugar	36	-	-	0%	30.1
Oils	34	-	-	8%	-
Margarine	-	-	-	8%	27.8
Margarine: Rapeseed Oil[g]	-	-	-	8%	16.1
Margarine: Sunflower Oil[g]	-	-	-	8%	1.5
Margarine: Maize Oil[g]	-	-	-	8%	1.5
Margarine: Palm Oil[g]	-	-	-	8%	5.9
Margarine: Palm Kernel Oil[g]	-	-	-	8%	5.9

Grey rows indicate that the food item was disaggregated into its constituent items which were then considered in the final consumption.
[a] Average Danish consumption to provide 10MJ energy per day, in the broad food groups defined and selected staples (e.g. potatoes)
[b] Taken from Danish Statistics for the most recent year available for every food item. Provided by source in kg consumed per capita per annum, and adjusted to grams/day by multiply by a factor of (1000/365).
[c] Taken is the mass of that food item divided by the sum of masses of all other food items within that food group. For example, 'milk' was taken as 253.4/(253.4+28.5+7.1+5.5)=86%.
[d] Taken as losses for the 'Processing', 'Distribution' and 'Consumption' for the food groups.
[e] Calculated as percentage of that food item in its' food group times amount consumed of that food group in first column. Adjusted for food losses with a factor of 1/(1-food losses). Adjusted to 2000 kcal/day with a factor of 2/(10/4.18).
[f] Taken as losses for 'Roots and Tubors'
[g] Breakdown of constituent oils taken from Nilsson et al. (2010)[iv]

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[i] DTU Fødevareinstituttet. (2010). *Danskernes kostvaner*. Søborg [In Danish]

[ii] Statistics Denmark, 'Food Consumption', http://www.dst.dk/en/Statistik/emner/forbrug/foedevareforbrug.aspx, last accessed: November 19, 2014

[iii] FAO. (2011). *Global Food Losses and Food Waste - Extent, Causes and Prevention*. Rome, IT. Retrieved from http://www.fao.org/docrep/014/mb060e/mb060e00.pdf

[iv] Nilsson, K., Flysjö, A., Davis, J., Sim, S., Unger, N., & Bell, S. (2010). Comparative life cycle assessment of margarine and butter consumed in the UK, Germany and France. *The International Journal of Life Cycle Assessment*, *15*(9), 916–926. doi:10.1007/s11367-010-0220-3

S.2 – Vegetarian and Vegan Diets

Food Product	Vegetarian recommended daily servings for 2000 kcal[iv]	Vegan recommended daily servings for 2000 kcal[ii]	Mass per serving (g)[i]	Breakdown of food items within food group[a]	Edible losses (post farm)[iv]	Vegetariandemand at farm for 2000 kcal (g/day)[e]	Vegan demand at farm for 2000 kcal (g/day)
Dairy	2	-	-	-	8%	-	-
Milk	-	-	250	79.1/0%	8%	449.8	-
Cheese	-	-	42	20.9/0%	8%	19.1	-
Eggs	0.5	-	57	100/0%	19%	31.0	-

Whole Grains	6	7	-	-	33%	-	-
Bread[b]	-	-	30	16.6%	33%	44.8	52.2
Pasta[b]	-	-	100	16.6%	33%	149.3	174.1
Bun[b]	-	-	30	16.6%	33%	44.8	52.2
Breakfast Cereal[b]	-	-	30	16.6%	33%	44.8	52.2
Cracker[b]	-	-	30	16.6%	33%	44.8	52.2
Wheat: bread	-	-	-	-	33%	9.0	10.4
Wheat: pasta	-	-	-	-	33%	149.3	174.1
Wheat: bun	-	-	-	-	33%	31.3	36.6
Wheat: Breakfast cereal	-	-	-	-	33%	44.8	52.2
Wheat: cracker	-	-	-	-	33%	44.8	52.2
Rye: bread	-	-	-	-	33%	21.8	25.3
Rice	-	-	100	16.6%	33%	145.0	316.2
Vegetables	8	8	50	-	26%	-	-
Potatoes	-	-	-	40.5%	26%	238.4	238.4
Cucumbers	-	-	-	8.7%	26%	47.1	47.1
Pepper	-	-	-	0.0%	26%	0	0
Spring-white cabbage	-	-	-	3.1%	26%	16.7	16.7
Spring-red cabbage	-	-	-	3.1%	26%	16.7	16.7
Brussels sprouts	-	-	-	0.2%	26%	1.1	1.1
Cauliflower and broccoli	-	-	-	4.1%	26%	21.9	22.0
Chinese cabbage	-	-	-	2.0%	26%	11.0	11.0
Leeks	-	-	-	2.3%	26%	12.6	12.6
Beetroots	-	-	-	1.7%	32%	10.1	10.1
Celeriac	-	-	-	0.8%	32%	4.8	4.8
Carrots	-	-	-	14.1%	32%	82.8	82.8
Onions	-	-	-	10.7%	26%	58.1	58.1
Lettuce	-	-	-	8.6%	26%	46.6	46.6
Fruits	4	4	150	-	26%	-	-
Tomatoes	-	-	-	33.6%	26%	272.3	272.3
Cherries (sweet and sour)	-	-	-	2.1%	26%	16.9	16.9
Strawberries	-	-	-	3.1%	26%	24.8	24.8
Apples	-	-	-	53.7%	26%	435.5	435.5
Pears	-	-	-	7.5%	26%	61.2	61.2
Legumes and Soy	3	3	-		26%	-	-
Beans	-	-	100	33.3%	26%	135.1	135.1

Tofu	-	-	125	33.3%	26%	94.6	94.6
Soy beverage[c]	-	-	250	33.3%	26%	337.8	337.8
Soy beverage: soy beans	-	-	-	-	26%	23.6	23.6
Soy beverage: sugar cane	-	-	-	-	26%	8.5	8.5
Soy beverage: maize starch	-	-	-	-	26%	0.1	0.1
Nuts	1	2	30	-	26%	-	-
Peanuts	-	-	-	50%	26%	20.3	40.5
Cashews	-	-	-	50%	26%	20.3	40.5
Vegetable Oils	2	2	14	-	8%	-	-
Vegetable Oil	-	-	-	50%	8%	7.6	7.6
Palm Oil	-	-	-	-	8%	3.5	3.5
Soybean Oil	-	-	-	-	8%	2.4	2.4
Rapeseed Oil	-	-	-	-	8%	1.7	1.7
Margarine[d]	-	-	-	50%	8%	7.6	7.6
Margarine: rapeseed oil	-	-	-	-	8%	2.5	2.5
Margarine: sunflower oil	-	-	-	-	8%	0.2	0.2
Margarine: maize oil	-	-	-	-	8%	0.2	0.2
Margarine: palm oil	-	-	-	-	8%	0.9	0.9
Margarine: palm kernel oil	-	-	-	-	8%	0.9	0.9

Grey rows indicate that the food item was disaggregated into its' constituent items which were then considered in the final consumption.
[a] Splitting of the food groups was done using the same breakdown of foods consumed according to the Danish Statistics or evenly between foods within that food group where these statistics were lacking.
[b] Food items broken into constituent items using the LCA Food[iv] or assumed to be comprised only of wheat where a breakdown was lacking.
[c] Soy beverage disaggregated using Ercin et al. (2012)[iv]
[d] Breakdown of constituent oils for margarine taken from Nilsson et al. (2010)[iv]
[e] Calculated as the total number of servings for that food group multiplied by that food items share of consumption in that food group and then multiplied by the factor accounting for food losses. For example, since 'Milk' accounts for 79.1% of the total dairy needs, it is calculated as 2 servings * 0.791 * 250 g/serving * 1/(1-0.08).

S.3 Custom Process Inventories

Inputs	**Amount**	**Units**	**Amount**	**Units**
Spring Barley	91.9	t/a	0.18	kg/kg milk
Soy meal	77.2	t/a	0.15	kg/kg milk

Lubricant	1068	t/a	0.002	kg/kg milk
Fertilizer, calcium ammonium nitrate	6602	t N/a	0.01	kg/kg milk
Fertilizer P	909	t P/a	0.002	kg P/kg milk
Fertilizer K	2549	t K/a	0.005	kg K/kg milk
P, mineral feed[a]	137	t N/a	0.0003	kg N/kg milk
Electricity	42162	kWh/a	0.08	kWh/kg milk
Heating	690	MJ/a	0.001	MJ/kg milk
Traction	376043	MJ/a	0.75	MJ/kg milk
Land	65	ha	0.0001	ha/kg milk
Outputs				
Bread wheat	12.1	t/a	0.02	kg/kg milk
Rapeseed	1.1	t/a	0.002	kg/kg milk
Milk	499.3	t/a	1	kg
Beef meat[b]	20.6	t/a	0.01	kg/kg milk
Air Emissions				
Methane	12316	t/a	0.02	kg/kg milk
Ammonia	3426	t/a	0.007	kg/kg milk
N_2O	920	t/a	0.002	kg/kg milk
Emissions to Water				
Nitrate	31112	t/a	0.06	kg/kg milk
Phosphate	113	t/a	0.0007	kg/kg milk

Milk Production – Process inventory for milk production on the marginal Danish farm according to Food LCA[iv]
Numbers have been rounded for legibility. The model also assumes that 1.12 kg of milk are required to produce 1 kg of milk at market according to LCA Food[i].
[a] Mineral feed assumed to consist of 40% dolomite and 60% zeolite by volume[iv].
[b] Slaughter weight of cows taken as the weighted average of sucklers reaching market from Nguyen et al. (2010)[iv]. Amount of avoided beef production avoided at market taken from Cederberg et al. (2003)[iv].

Cream Production – Dynamic market reactions to the production of cream[iv].

Cream is a constrained by-product of milk.
The utility of cream is its fat content.
Consuming butter will means fat content in cream typically used for other products must be procured from elsewhere
25% of cream fat would have been used as fat content in powdered milk - this is substituted with marginal vegetable oil (palm oil)
75% of cream fat actually sees consumers switch from high fat to low fat cheese

Palm Oil substitution for fat content allocated to butter production

Cream Lost	0.25	kg
Product	**Fat Content**	**Notes**
Cream	25%	Estimated
Palm Oil	100%	Estimated
	0.0625	kg

Low fat cheese produced as substitution for high fat cheese

Cream Lost	0.75	kg
Product	**Fat Content**	**Notes**
Cream	25%	Estimated
High Fat Cheese	35%	Estimated

Low Fat Cheese	11%	Estimated
High Fat Cheese Lost	0.54	kg
Low Fat Cheese Produced	1.70	kg

Crème Fraiche Production – Dynamic market reactions to the production of crème fraiche.

Same market reactions as outlined for cream demand market.

Palm Oil substitution for fat content allocated to crème fraiche production

Cream Lost	0.25	kg
Product	**Fat Content**	**Notes**
Crème Fraiche	40%	Estimated
Palm Oil	100%	Estimated
Palm Required	0.1	kg

Low fat cheese produced as substitution for high fat cheese

Cream Lost	0.75	kg
Product	**Fat Content**	**Notes**
Crème Fraiche	40%	Estimated
High Fat Cheese	35%	Estimated
Low Fat Cheese	11%	Estimated
High Fat Cheese Lost	0.88	kg
Low Fat Cheese Produced	2.73	kg

Butter Production – Dynamic market reactions to the production of butter.

Same market reactions as outlined for cream demand market.

Palm Oil substitution for fat content allocated to butter production

Cream Lost	0.25	kg
Product	**Fat Content**	**Notes**
Butter	81%	Estimated
Palm Oil	100%	Estimated
Palm Required	0.2	kg

Low fat cheese produced as substitution for high fat cheese

Cream Lost	0.75	kg
Product	**Fat Content**	**Notes**
Butter	81%	Estimated
High Fat Cheese	35%	Estimated
Low Fat Cheese	11%	Estimated
High Fat Cheese Lost	1.74	kg
Low Fat Cheese Produced	5.52	kg

Cheese Production

Assumes only whey produced as a single byproduct. Values taken for soft cheese in a US context[iv].

Input	Amount	Unit
Milk	8.4	kg/kg dry. wt. cheese
Milk[a]	13.8	kg/kg cheese
Output		
Cheese	1	kg
Whey	12.8	kg

[a] Water content taken as 39%[iv]. Total milk taken as 1/(1-% wet wt.) Adjusted for 12% loss of milk at dairy.

Beef Production

Beef production taken from Nugyen et al. (2010)[iii].

Inputs	**Unit**	**per 1000 kg slaughter weight**	**per 1000 kg meat at market[a]**	**per kg meat at market**
Farm Supplied Feed				
Outdoor Grazing				
Grazed Grass	kg	9021	16174	16.17
Indoor Grazing				
Grass silage	kg	5446	9764	9.76
Maize silage	kg	2404	4310	4.31
Spring Barley	kg	2254	4041	4.04
Straw	kg	1726	3095	3.09
Imported Feed				
Soy meal	kg	12	22	0.02
Mineral Feed[b]	kg	131	235	0.24
Land Use				
Grass grazed (low)	ha a	3.01	5	0.005
Grass siliage (high)	ha a	0.68	1	0.001
Cereals	ha a	0.6	1	0.001
Fertilizer Import				
Nitrogen	kg	478	857	0.86
Phosphorous	kg	21.5	39	0.04
Direct on-farm energy use				
Electricity used in stables	MWh	1.07	2	0.0021
Electricity used in crop processing	MWh	0.64	1	0.001
Diesel	GJ	14	25	0.03
Transport				
Feed				
By ship	tkm	162	290	0.29
By Truck	tkm	12	22	0.02
Ouputs				
Gaseous Emissions				
N_2O	kg	26.2	47	0.06
CH_4				
Enteric fermentation	kg	417.6	749	0.75
Manure management	kg	58.5	105	0.10
NH_3	kg	95.6	171	0.17

Liquid Emissions				
NO_3	kg	123.1	221	0.22
PO_4	kg	2.7	5	0.005
Soil carbon loss	kg	145	260	0.26

Numbers have been rounded for ease of reading.
[a]Meat produced per slaughtered cow taken as weighted average of cattle reaching market from the article.
[b]Same assumptions as for mineral feed in the milk system.

Pork Production[iv]

Inputs	**Amount**	**Units**	**Amount**	**Units**
Feed				
Wheat	1090	kg	1.09	kg/kg pork
Barley	440	kg	0.44	kg/kg pork
Rye	161	kg	0.16	kg/kg pork
Soybean Meal	188	kg	0.19	kg/kg pork
Others	648	kg	0.65	kg/kg pork
Energy/Transport				
Heat (oil)	130.2	kWh	0.13	kWh/kg pork
Electricity	117.6	kWh	0.12	kWh/kg pork
Transport				
Ship	3375	tkm	3.38	tkm/kg pork
Truck 28t	868	tkm	0.87	tkm/kg pork
Traction	206	MJ	0.21	MJ/kg pork
Water	353	m^3	0.35	m^3/kg pork
Land	71	ha	0.0004	ha/kg pork
Outputs				
Air Emissions				
Methane	26.7	kg	0.03	kg/kg pork
N_2O	1	kg	0.001	kg/kg pork
NO_2	-2.4	kg	-0.002	kg/kg pork
Ammonia	20.7	kg	0.02	kg/kg pork
Water Emission				
NO_3	12	kg	0.01	kg/kg pork
PO_4	0.5	kg	0.0005	kg/kg pork
Avoided Fertilizer				
N	49	kg	0.05	kg/kg pork
P	13	kg	0.01	kg/kg pork
K	12	kg	0.01	kg/kg pork

Assumed that for every 120 kg of biomass produced, 94.7 kg of meat enters the market[viii]. Numbers have been rounded for ease of reading.

Cherry Production

Assumed cherry farmers supplying Denmark have similar technological level of development as Californian system. Values taken from Carlsson-Kanyama et al. (2000)[iv].

Inputs	**Amount**	**Units**	**Amount**	**Units**

Diesel	288	L	0.02	L/kg cherries
Gasoline	96	L	0.008	L/kg cherries
N-fertilizer	112	kg	0.009	kg/kg cherries
P-fertilizer	34	kg	0.003	kg/kg cherries
K-fertilizer	152	kg	0.01	kg/kg cherries
Land	1	ha	0.00008	ha/kg cherries
Outputs				
Cherries	12125	kg	1	kg

Numbers have been rounded for ease of reading.

Strawberry Production

Assumed strawberry farmers supplying Denmark have similar technological level of development as Californian system. Values taken from Carlsson-Kanyama et al. (2000)[iv].

Inputs	**Amount**	**Units**
$CaNO_3$	0.02	kg/kg strawberries
KNO_3	0.03	kg/kg strawberries
NH_3PO_4	0.005	kg/kg strawberries
$MgSO_4$	0.003	kg/kg strawberries
Traction	102	MJ/kg strawberries
Irrigation	1	L/kg strawberries
Ouputs		
Strawberries	1	kg
N_2O (to air)	0.009	kg/kg strawberries

Numbers have been rounded for ease of reading.

Tofu

Assumes that 0.56 kg of soybeans are required for 1 kg of produced tofu (the balance being water and coagulant)[iv].

Soy Beverage[xi] – agricultural inputs that go into soy beverage manufacturing.

Inputs	**Amount**	**Units**
Soybean	0.07	kg/kg soy beverage
Sugar cane	0.03	kg/kg soy beverage
Maize starch	0.00003	kg/kg soy beverage
Outputs		
Soy Beverage	1	kg

Peanuts[iv]

Inputs	**Amount**	**Units**
Rye Seed	0.73	g/kg peanuts
Boron	0.17	g/kg peanuts
Lime/Gypsum	317	g/kg peanuts
Energy		
Pre-harvest fuel[a]	18	g/kg peanuts
Harvesting fuel[a]	26	g/kg peanuts
Electricity	0.06	kWh/kg peanuts
Outputs		
Peanuts	1	kg

[a] Assumed to be diesel.

Margarine[iv] - agricultural inputs that go into margarine manufacturing.

Inputs	Amount	Units
Rapeseed oil	0.36	kg/kg margarine
Sunflower oil	0.03	kg/kg margarine
Maize oil	0.03	kg/kg margarine
Palm oil	0.13	kg/kg margarine
Palm kernel oil	0.13	kg/kg margarine
Outputs		
Margarine	1	kg

Eggs[iv] **-** Based on LCA of organic eggs which likely have lower production efficiency per unit input, which may elevate the results, but only marginally when taken in the context of the diets. Waste by-products should be interpreted as the goods at the market that processed chicken waste would substitute, not products directly resulting from egg production.

Inputs	Amount	Units
Transport	0.2	kg/kg eggs
Wheat	1.37	kg/kg eggs
Rapeseed	0.20	kg/kg eggs
Soybean meal	0.29	kg/kg eggs
Barley	0.10	kg/kg eggs
Maize	0.31	kg/kg eggs
Soybeans	0.18	kg/kg eggs
Oats	0.11	kg/kg eggs
Protein Pea	0.07	kg/kg eggs
Limestone	0.20	kg/kg eggs
Water	5.79	L/kg eggs
Silage	0.09	kg/kg eggs
Straw	0.09	kg/kg eggs
Sand	0.04	kg/kg eggs
Electricity	0.32	kWh/kg eggs
Diesel	0.0054	kg/kg eggs
Gas	0.05	MJ/kg eggs
Outputs		
Eggs	1	kg/kg eggs
Fertilizer as N (manure by-product)	0.0082	kg/kg eggs
Barley (waste treatment by-product)	0.00084	kg/kg eggs
District Heat (waste treatment by-product)	0.0005	kg/kg eggs
Maize (waste treatment by-product)	0.041	kg/kg eggs
N_2O (to air)	0.00082	kg/kg eggs
Methane (to air)	0.0016	kg/kg eggs

Vegetable Oil Mix[iv] **-** Blend of the top 3 vegetable oils by production volume in 2014, accounting for over 2/3 of global production

Inputs	Amount	Units
Palm oil	0.47	kg/kg oil mix
Soybean oil	0.33	kg/kg oil mix
Rapeseed oil	0.2	kg/kg oil mix
Output		
Vegetable oil mix	1	kg

Cashews[iv]

Inputs	Amount	Units
Land	$5.8*10^{-5}$	ha/kg cashews
Limestone	0.63	kg/kg cashews

Gypsum	0.029	kg/kg cashews
Copper	$9.6*10^{-6}$	kg/kg cashews
Manganese	$2.4*10^{-5}$	kg/kg cashews
Molybdenum	$1.2*10^{-6}$	kg/kg cashews
Zinc	$1.1*10^{-4}$	kg/kg cashews
Iron	$3.6*10^{-5}$	kg/kg cashews
Urea	0.20	kg/kg cashews
Phosphate	0.47	kg/kg cashews
KCl	0.05	kg/kg cashews
Glyphosate	0.0014	kg/kg cashews
Diesel	0.089	kg/kg cashews
Water	5.48	L/kg cashews
Outputs		
Cashews	1	kg/kg cashews
Wood	3.89	kg/kg cashews

Appendix F

Article 6: Potential to curb the environmental burdens of American beef consumption using a novel plant based beef substitute

Potential to Curb the Environmental Burdens of American Beef Consumption Using a Novel Plant Based Beef Substitute

Benjamin Goldstein[1*], Rebekah Moses[2], Norman Sammons[2] and Morten Birkved[1]

[1]Department of Management Engineering, Quantitative Sustainability Assessment Division, Technical University of Denmark, Kongens Lyngby, Denmark

[2]Impossible Foods, Redwood City, CA USA

* Corresponding author

E-mail: bgol@dtu.dk

Abstract

The food demands of the United States (US) impart significant environmental pressures. The high rate of consumption of beef has been shown to be the largest driver of food-borne greenhouse gas emissions, water use and land occupation in the US diet. The environmental benefits of substituting animal products with vegetal foods are well documented, but significant psychological barriers persist in enacting dietary transitions. Here we use life cycle assessment to appraise the environmental performance of a novel vegetal protein source in the mean US diet where it replaces ground beef, and in vegetarian and vegan diets where it substitutes for legumes, tofu and other protein sources. We find that relative to the mean US diet, vegetarian and vegan diets significantly reduce per-capita food-borne greenhouse gas emission (32% and 67%, respectively), water use (70% and 75%, respectively) and land occupation (70% and 79%, respectively). The substitution of 10%, 25% and 50% of ground beef with plant-based burger (PBB) at the national scale results in substantial reductions in annual US dietary greenhouse gas emissions (4.55-45.42 Mt CO_2 equivalents), water consumption (1.30-12.00 km^3) and land occupation (22300-190100 km^2). Despite PBB's elevated environmental pressures compared to other vegetal protein sources, our analysis shows that minimal risk is present for the disservices of PBB substitution in non-meat diets to outweigh the benefits of ground-beef substitution in the omnivorous American diet. Demand for plant-based oils in PBB production has the potential to increase land use pressures in biodiversity hotspots, though these could be obviated through responsible land stewardship. Although the apparent environmental benefits of the PBB are contingent on actual uptake of the product, this study demonstrates the potential for non-traditional protein substitutes to play a role in a transition towards more sustainable consumption regimes in the US and abroad.

1. Introduction

The food-related environmental footprint of the United States (US) is among the highest in the world per capita [1,2], driven largely by animal-sourced products [3–5]. Of all livestock products, beef is the most environmentally taxing, both in terms of total global impacts from the sector and normalized per unit mass [1,6–9]. Studies of the US diet have pinpointed beef as a main driver of greenhouse gas (GHG) emissions (enteric fermentation, deforestation) [3,10], water use (hydration and feed irrigation) [11] and land occupation (pasture and cropland for feed) [11,12]. Although domestic consumption has waned in recent years, beef remains a staple of the American diet [13–15], representing a key opportunity to attenuate the environmental impacts of US food demands both through supply- and demand-side initiatives.

Beef production is one of the least efficient animal agriculture systems at any scale, due to the metabolic requirements associated with using cattle to convert plant materials into human-available protein and energy [6,11,16]. Streamlining existing cattle systems through improved feed quality and consolidation in feedlots offer limited opportunities for improving resource efficiencies and feed conversion ratios [17,18]. Further, the US already maintains an industrialized beef production landscape with some 97% of beef finished on feedlots [19]. Since the late 1970s, the land, feed and water resources required to produce a given volume of beef have dropped significantly [20], yet production of beef still occupies 88% of the US land footprint allocated to animal agriculture (or around 41% of the contiguous US) [11]. Given production-side constraints, an alternative is to substitute beef with plant-based protein sources (legumes, beans, tofu, seitan, etc.) providing identical nutritional functionality with lower embodied environmental burdens [21,22]. The nutritional role of beef in the US diet could be performed by plant-sourced foods using 10% of the land while producing 4% of the GHGs [12], and shifts from standard US to vegetarian and vegan diets could reduce dietary GHGs by 30% and 50%, respectively [3].

Although plant-based diets could reduce US diet environmental pressures, and demand for beef in the US is elastic [12,13], behavioral hurdles exist in getting Americans to trade beef for beans. Eating beef (and meat in general) is tied to a host of social, psychological and hedonic factors: taste, the perception that a meal requires meat, communal eating practices, dietary guidelines and advertising espousing meat as an essential part of a healthy diet, etc. [23–26]. Given the challenge of shifting consumer practices surrounding meat, lower-intensity beef alternatives that obviate these psychological impasses provide a mechanism to reduce US beef intake, promote reduced-meat diets and lower dietary environmental impacts (barring currently unrealistic bans or taxes on beef).

A plethora of technologies to produce novel protein substitutes that more authentically mimic meat than existing vegetal foods (e.g. soy-based, mycroprotein or gluten products) have emerged in recent years, including 'cellular agriculture,' yeast culture, bioprinting, scalable arthropod farming, and plant-based functional equivalents [27]. Though most of these technologies were initially developed in university labs, such as the '$300,000 test-tube hamburger' produced by Post et al. [27,28], the most market-ready technologies are those that have been adopted and refined in industry. Hence, focusing on the commercially available products coming out of private labs provides the best opportunity to gauge the potential short- to mid-term environmental benefits of the emerging generation of animal-protein alternatives [27].

One such technology is the plant-based burger (PBB) by Impossible Foods, which is a substitute designed to match the experience of cooking and consuming ground beef [27]. By fulfilling the same gustatory, culinary and nutritional functions as traditional beef, the adoption barrier associated with consumption of vegetal in lieu of animal proteins is reduced. The primary ingredients of the PBB include texturized wheat protein (wheat TVP), coconut oil, and potato protein. In order to deliver the same sensory characteristics of animal-sourced beef, the company developed a modified yeast culture to produce "heme" (leghemoglobin), a protein which occurs naturally in the root nodules of leguminous plants and functions as an analog for the myoglobin that gives beef its distinct flavor and cooking characteristics.

Resource use in producing plant-based beef is much lower than traditional production via cattle [29,30]. Compared to a typical US beef production system, PBB requires less than one quarter of the resources as modeled according to pilot scale production data collected in 2015 and refined in 2016 to account for minor supply chain changes [31]. As is the case with beef, land and water use associated with production of PBB is dominated by raw ingredients (agricultural products, maintenance of the yeast culture) rather than by production or formulation. While the majority of emissions within the beef supply chain are derived from cattle (raw materials), PBB emissions impacts are split between raw materials and production stages.

Aside two studies of in-vitro cultured meat production relying on estimates for production inputs [30,32], there exists no published environmental assessments using primary data from operations above bench-scale of novel protein sources that truly mimic the essential sensory aspects of ground beef from cattle. Here we look at the potential environmental and resource implications of substituting ground beef in the 2010 mean US diet (MUD), and plant-based proteins in the hypothetical vegetarian (VEG) and vegan (VGN) diets with PBB, at individual and national scales. Lastly, the potential for negative environmental trade-offs due to PBB adoption by VEG and VGN Americans are examined.

2. Materials and Methods

Life cycle assessment (LCA) is a widely used method to quantify the environmental impacts of food production systems [1,3]. LCA focuses at the processes along a supply chain that deliver a service, accounting for material and energy inputs, and chemical emissions to the environment (herein, life cycle inventory or LCI), thus providing an appraisal of system-wide environmental impacts and resource draws [33,34]. We apply LCA to the US food supply chain, setting the system boundaries as the agricultural and processing stages, excluding the distribution (transport and packaging), preparation and disposal phases of the life cycle. These omissions underestimate environmental impacts and resource use [35,36], but given uncertainties surrounding relevant data for these stages and their typically marginal contribution to the outcomes of other food LCAs [10,37,38], the majority of life-cycle impacts should be captured here.

Three different archetypical American dietary patterns are modeled: the mean-US (MUD), vegetarian (VEG) and vegan (VGN) diets. The MUD is constructed from the 2010 USDA's loss-adjusted-food-availability estimates of per capita consumption of ~250 food items in the US [39]. The VEG and VGN are built from the USDA's 2010 dietary guidelines for vegetarian and vegan diets consuming 2000 kcal per day [40] (in line with measured adult vegetarian energy intake [41]), adapted to actual US consumption regimes using the 2010 loss-adjusted data. For instance, USDA guidelines suggest 1.5 cups of dark green vegetables per week for the VEG. Here the constituent dark green vegetables (e.g. broccoli, kale, spinach, etc.) were provided in the same ratios as found in the 2010 MUD. USDA data on food waste at the consumer and retail levels are also included so the diets represent the production volumes drawn by each diet to meet final ingestion. See S1 Spreadsheet for a full breakdown of the components of the modeled diets.

The effects of substituting 10%, 25% and 50% of total protein in American diets are examined using MUD, VEG and VGN as baseline diets, with PBB as replacement protein. PBB is nutritionally similar to ground beef in most respects, besides lacking cholesterol and containing carbohydrates (see S2 Table for laboratory analytics) and is substituted on a 1:1 mass basis in the MUD. A nutritionally equivalent mass of PBB replaces the blend of protein foods in the VEG and VGN diets (see S1 Spreadsheet for further information). Given uncertainties in the amount of total beef as ground-beef ingested by Americans, values of 30% (see S3 Text for estimation method) and 50% [42] were used to assess the upper and lower PBB market penetration.

GHG emissions, water use and land occupation are evaluated: all metrics to which LCA is widely applied and accepted, and relevant to the impacts of beef production in the context of net environmental burdens from the US diet. GHG emissions are assessed using the IPCC 2013 methodology to convert from atmospheric chemical emissions to the equivalent mass of carbon dioxide to affect the same degree of radiative forcing over a 100 year period (kg CO_2e). Water use is calculated as 'blue water', the volume of surface or groundwater used and evaporated or incorporated into a product [43]. Lastly, agricultural land occupation is assessed as the physical area occupied in m^2 arable land according to the Impact 2002+ method [44].

Hybrid-LCA methodology is employed here, whereby LCIs of on-farm resource use and chemical emissions are derived from studies of individual agricultural operations, while those for food processing (slaughterhouse operation, vegetable and fruit canning, etc.) are taken using a top down methods, based on national economic input-output accounts. Previous LCAs of on-farm operations are used to gather the production inputs and emissions data for foods, which were combined with inventories of individual inputs (fertilizers, fuels, irrigation, etc.) from the ecoinvent 3.2 database (www.ecoinvent.org) to build a complete LCI for that food. Ecoinvent 3.2 also contains complete LCIs for several relevant foods, which are adapted to US production conditions (e.g. US electricity and irrigation). The Carnegie Mellon 2002 US input-output database (www.eiolca.net), providing LCIs per dollar economic output for 428 economic sectors, is linked with US food production volumes to estimate average resource and emissions inventories per mass food produced in (e.g. per kilogram canned vegetables or fruit). Combining these two data streams provides a complete LCI for the agricultural and processing stages. S3 Text further details the LCA method employed here and the construction of the LCIs from the supporting literature.

LCI data for PBB production are from early-stage, low-volume (hundreds of kilos/day capacity) production scale of both heme and burger manufacturing for current bill of material. The PBB model relies on certain literature-derived assumptions to estimate commercial scale production (mainly associated with fermentation substrate and energy use) and the results of the PBB LCA reflect both known and projected bill of material and production processes specific to 2015-2016 LCA development period. Because the PBB product continues to evolve, these impacts are likely to change as formulations and processes continue to improve, and should be viewed as a snapshot of current production technology. To ensure validity the PBB life cycle inventory and subsequent analysis presented in this paper, the inventory and assessment were independently vetted by an external independent LCA consultant and again by Quantis US following inventory updates. Data management is done in the LCA software SimaPro 8.2.0.0.

3. Results

Fig 1 outlines the baseline results for the MUD, VEG and VGN for annual per capita GHG emissions (Fig 1A), water use (Fig 1B) and land occupation (Fig 1C). Error bars around the MUD represent different conversion rates from cattle live weight to beef (see supplementary info S4 for detailed results).

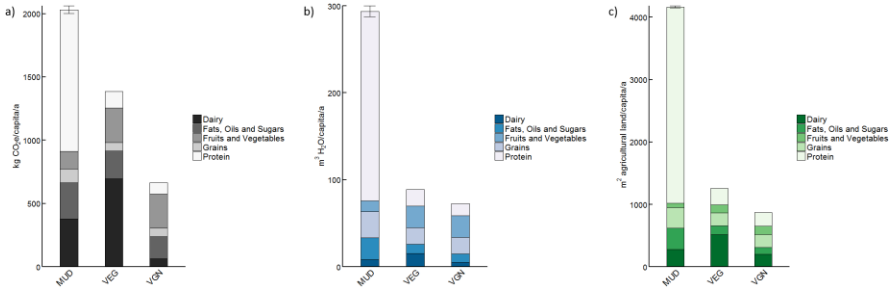

Fig 1. Results for the mean US Diet (MUD), vegetarian (VEG) and vegan (VGN).
(A) GHG in kg CO_2e. (B) Water use in m^3 blue water consumption. (C) Land occupation in m^2 organic arable land. Error bars indicate range of results for different proportions of ground beef in baseline MUD and varying carcass yields.

GHG emission results align with other US diet assessments, with shifts away from the MUD resulting in reduced GHG emissions for the VEG (-32%) and VGN (-67%). Of note is that if isocaloric diets were compared (total intake for MUD, VEG and VGN of 2481 kcal/day in line with 2010 USDA loss-adjusted numbers for the MUD), the reductions would have been -15% and -40% for the VEG and VGN, respectively. Protein dominates MUD impacts, with meat as the primary driver (50% total GHG emissions), itself impacted by beef (40-42% of total GHG emissions). The VEG is burdened by higher reliance on dairy as a protein and fat source, which elevate this dietary component's impacts well above the MUD. Fruits and vegetables are the area of largest potential improvement for the VGN. Our findings are comparable to Heller et al.'s estimate of 5 kg CO_2e/cap/d for the average American and reductions of 33% and 53% for vegetarian and vegan diets [3] and are in agreement with the scale of GHG emissions and reduction potentials through dietary shifts in nations with similar diets [35,45–47].

Water use follows the same pattern as GHGs: relative to the MUD, approximately 200 fewer cubic meters of water per annum are required to support the VEG (-70%) and VGN (-75%) though reductions shrunk when isocaloric diets were compared (-62% and -70% for VEG and VGN, respectively) . The majority of the MUD's impacts here stem from meat intake (74%), especially beef (56-58%), which requires sufficient animal hydration and significant embodied water inputs in feed via pasture, roughage, and concentrates. The VEG differs slightly from the VGN due to the former's dairy and egg intake, but these differences are marginal when compared to the MUD. Eshel et al. [11] found that 150 m^3/cap/a are needed for feed production for the US diet, in close alignment with our estimate of 140 m^3/cap/a. Jalava et al. [48] also estimated significant reductions when moving from MUD to VGN, though their alternative method estimated larger savings of 438-657 m^3/cap/a.

Significant decreases in land occupation also follow from a shift away from animal-based foodstuffs. The VEG and VGN occupy 70% and 79% less land than the MUD, respectively (VEG = -63% and VGN = -74% for isocaloric diet comparison). Of the MUD's ~4100 m^2 annual occupation, 75% is from meat, 67% from beef alone, where grazing land and feed production predominate. Similar to GHG emissions, the VEG is greater than the VGN, exerted by dairy consumption and related agricultural space for feed crops. Our results match other LCAs of similar diets, where vegetarian and vegan diets effect 50-90% reductions in agricultural land occupation from omnivorous alternatives [35,45,47,49]. Of note is that inedible portions of plants can feed livestock to produce nutritionally dense animal products with limited environmental cost, and hence a diet with limited animal products could potentially have similar or lower land use to VEG and VGN diets contingent on the optimal balancing of residual resource and livestock production [50].

3.1 Beef substitution with PBB

Fig 2 displays the potential impacts of PBB diffusion into the modeled diets at rates of 10%, 25% and 50%, where PBB substitutes for ground beef in the MUD, and a mix of vegetal proteins for the VEG and VGN. Upper and lower bounds of the MUD results represent high ground beef share of total beef intake (50%) combined with low conversion from live weight to beef (39%) and lower ground beef share of total beef (30%) combined with higher carcass yield (43%), respectively (see S3 Text for estimation methods).

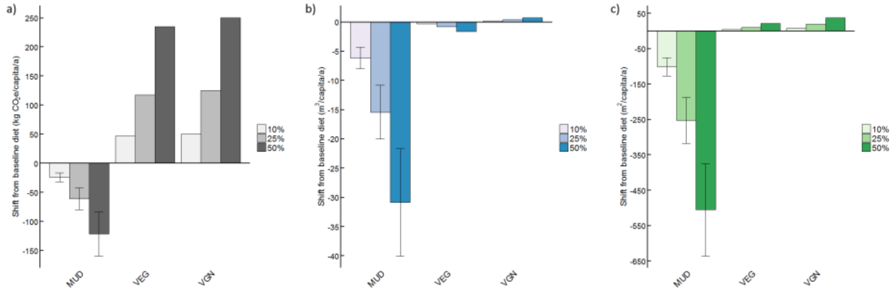

Fig 2. Per capita shifts in environmental burdens give PBB substitution in the mean US diet(MUD), vegetarian (VEG) and vegan (VGN).
Substitution rates of 10%, 25% and 50% ground beef (MUD) and total protein foods (VEG and VGN). (A) GHG in kg CO_2e. (B) Water use in m^3 blue water consumption. (C) Land occupation in m^2 organic arable land. Error bars indicate range of results for different proportions of ground beef in baseline MUD and varying carcass yields.

By all three metrics the introduction of PBB improves the MUD's environmental performance. Total dietary GHG emissions are reduced by 24 (1.2%), 61 (3.0%) and 122 (6.0%) kg CO_2e/cap/a at increasing levels of diffusion. Of note is that a 50% PBB diffusion generates nearly half the GHG savings as an isocaloric switch to a vegetarian diet. Similarly, water use is reduced by 6 (2.1%), 15 (5.2%) and 31 (10.4%) m^3/cap/a, while agricultural land occupation shrinks by 101 (2.4%), 252 (6.1%) and 505 (12.1%) m^2/cap/a. For the MUD, PBB provides an ecologically leaner protein option for GHGs (6.9 kg CO_2e/kg PBB vs 30.1 kg CO_2e/kg ground beef), water consumption (0.18 m^3/kg PBB vs. 6.07 m^3/kg ground beef) and land use (3.5 m^2/kg PBB vs. 101.1 m^2/kg ground beef). These reductions for the PBB are similar to those estimated for in-vitro meat production in GHGs (-75%) and land use (-94%) based on extrapolations from bench-scale data [32]. The results are more complex for the VEG and VGN diets. Notably increases are seen for GHG emissions (VEG: 3-17% and VGN: 8-38%), water impacts drop slightly for the VEG (0.4-1.8%) and rise for the VGN (0.2-1%), while land occupation increases marginally for both the VEG (0.3-1.7%) and VGN (0.9-4.4%).GHG emission increases stem largely from the energy inputs for the PBB, which are higher than soy and nut-based protein sources due to production processes and inclusion of leghemoglobin. Water and land remain essentially unaltered when moving from pulses, nuts and eggs to alternative plants sources, although the tendency for higher land occupation aligns with the lower protein content of wheat used in PBB compared to the fava beans used to model the legumes in the VEG and VGN.

The marginal shifts in dietary performance of the MUD at the individual level mask the true scope of reducing dietary environmental burdens from potential diffusion of such novel protein substitutes at the national scale. Considering the 299.40 million omnivores, 8.35 million vegetarians and 1.55 million vegans in the US [51], a hypothetical 10% introduction of PBB into all three diets would net annual reductions of 4.6-9.1 Mt CO_2e GHG emissions, 1.3-2.4 Gm^3 water use and 22300-38000 km^2 agricultural land occupation. To bring these numbers into context, this is the equivalent of removing 1.1-2.2 million cars from American roads annually (4400 kg CO_2e/car/a [52]), eliminating the direct water consumption of 10.5-19.3 million Americans (124 m^3/cap/a [53]) and freeing up an area equal to 1-1.6 times that of the state of New Hampshire. Table 1 highlights the potential impacts from PBB diffusions at higher levels.

Table 1. Net impacts of PBB at different substitution rates for protein in the MUD, VEG and VGN at US scale

Indicator	% shift to PBB	Net Change	Analogue	Unit
GHGs	10%	4.55 – 9.08 Mt CO_2e	1.13 – 2.25	million average US drivers removed
	25%	11.39 - 22.71 Mt CO_2e	2.82 – 5.62	
	50%	22.78 - 45.42 Mt CO_2e	5.64 – 11.24	

Water consumption	10%	1.30 - 2.40 km^3	10.48 – 19.34	million fewer US water consumers
	25%	3.25 - 6.00 km^3	26.20 – 48.37	
	50%	6.50 - 12.00 km^3	52.41 – 96.74	
Land occupation	10%	22300 - 38000 km^2	1 – 1.6	area of New Hampshire
	25%	55900 - 95100 km^2	1 – 1.7	area of Illinois
	50%	111800 - 190100 km^2	1 – 1.7	area of Nevada

4. Discussion

Dietary shifts from the MUD to the VEG or VGN, and substitution of PBB for ground beef, reduce food related pressures exerted from typical US residents. Actually achieving net gains is contingent on the adoption of PBB by a proportion of the 97% of US residents that are omnivores, since PBB uptake risks increasing some environmental pressures of the non-meat diets. Fig 3 compares GHG emissions for different common protein sources per kilogram protein delivered to the consumer’s plate. The PBB, though significantly lower in burdens than beef, is similar to other animal-sourced proteins and elevated above the other plant-based choices. It should be kept in mind that our GHG estimates for animal-sourced proteins could be considered conservative [54]. PBB appears more burdensome than protein from mealworms, though the numbers for the mealworm LCA are for a live product [55], excluding the processing inputs to convert live insects to more palatable end products.

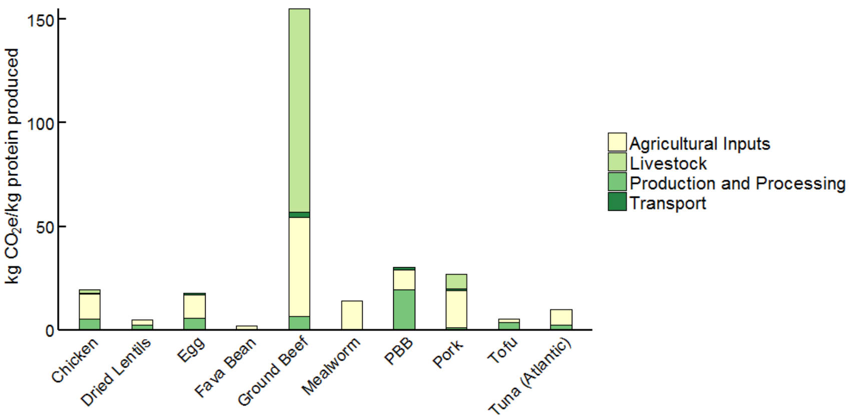

Fig 3. Embodied GHG for different foods.
GHG emissions in kg CO2e/kg protein produced.

Notwithstanding, a small risk exists that increases in VEG and VGN environmental burdens for GHGs and land (and water for the VGN) from PBB adoption, might not be counteracted through uptake by the MUD. Fig 4 explores the required substitution of ground-beef with PBB in the MUD to balance 0-100% substitution of total protein with PBB in the VGN and VEG diets. In the extreme case that all vegetarians and vegans in the US source 100% of their protein from PBB, a replacement rate of around 6% ground beef (averaged ground beef as percentage of total beef and slaughtering efficiency) by PBB in the MUD would avoid a net increase, hinting that the potential risk for unintended increases of GHG emissions at the US aggregate is marginal. The same is true for land use, where a MUD penetration rate below 1% would suffice to counterbalance a net increase. For water consumption, the negative slopes indicate that the MUD would have to increase beef consumption to counteract net reductions of PBB uptake by the VEG and VGN; unlikely given falling US beef demand in recent decades [13].

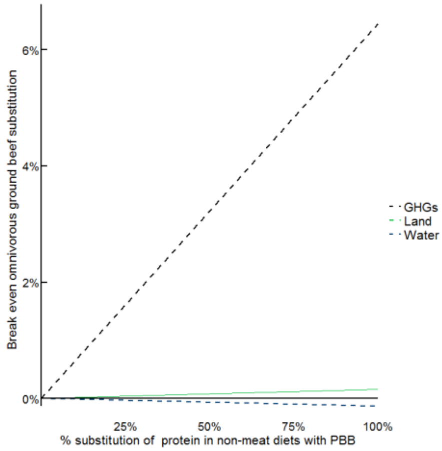

Fig 4. Required substitution rates of Plant-Based Beef (PBB) in mean US diet (MUD) to counteract impacts from uptake by vegetarian and vegan Americans.

Much of the beef reduction in the US diet has been matched by increases in poultry intake. Such a trend would be preferable from a GHG reduction perspective, even over PBB substitution or other novel meat substitutes [30,32]. However, the large scale industrialization of poultry since the 1970s has been undergirded by higher animal stocking densities and an undercutting of genetic diversity and resilience through a producer preference for fast-growing breeds [56], practices connected to the transmission of communicable diseases in the avian livestock population and high rates of antibiotic administration to industrial broilers to combat disease and hasten growth [56,57]. Similar practices have also been noted in US pork production [57,58]. Applications of antibiotics are linked to the increase of multi-drug resistant disease strains, diminishing the effectiveness of medicines in the human population [58–60]. Likewise, only considering GHG related impacts for tuna obscures the fact that shifting towards pescatarian habits would further stress marine ecosystems that have seen precipitous declines in population size, species richness and functional diversity at current fishing levels [61,62]. Such costs in other sustainability domains should be counted when comparing PBB to livestock products with potentially lower GHG emission intensities.

Large-scale sourcing for plant-based lipids could eventually present a land use change (LUC) risk, though that risk is dwarfed by the deforestation and LUC driven by beef production. Pasture and feedcrop expansion is the leading driver of forest loss and landscape intensification in Central and South America, [63–65] and one of several leading contributors to global net carbon losses (~880 MT $C*yr^{-1}$) from wooded area conversion [66]. Of concern is the sourcing of coconut oil for the PBB, as coconuts are grown in plantations in the humid tropics, regions that are rich in biodiversity and thus at elevated risk of habitat and species loss. While coconut palm systems are of lower biodiversity value than intact forests [67], thus far, there has been limited recent detectable demand-driven extensification of coconut plantations in the source region, based on FAOSTAT land use and production volume data. Further, yield gaps in copra production indicate that production could be theoretically doubled without acreage expansion via cultivar selection and use of best management practices in coconut production (though such intensification is not without off-site environmental impacts) [68]. So while a net reduction in human appropriation of land in biodiversity hotspots seems probable when moving to PBB, oil sourcing will remain a potential sustainability challenge in the novel protein economy.

4. 1 Scaling up and future production efficiencies of PBB

Improvement potential exists for PBB, since significant impacts are borne in the processing of raw inputs to PBB, in addition to acquisition of raw materials. Future shifts from fossil fuel based electricity sources could net improvements in PBB GHG emission performance, since electricity accounts for 80% of emissions during the processing stage. Potential reductions in impacts from heme production appear likely as the technology matures and improved conversion efficiencies of raw inputs to heme are attained.

Taking the development of biofuels in recent decades as a barometer, considerable performance improvements are to be expected once PBB production reaches industrial scale using mature technologies. Precisely estimating the upscaling and technology maturation benefits and the resultant impacts on GHG performance of PBB production are difficult due to the novel nature of PBB and the scarcity of data on upscaling and maturation effects on environmental performance. Barlow et al. [69] showed that the net GHG emissions of algal derived biofuel improved from 80 g C02-eq/MJ to -44 g C02-eq/MJ as a result of scaling efficiencies of energy use (stirring, heating, etc.), elucidating the potential for improvement as biotechnologies move beyond pilot phase. Previous work on in-vitro meat production also assumed significant efficiencies with scaling and maturation [30,32], supporting the supposition that the environmental burdens of novel meat analogues such as a the PBB will likely decrease in the future.

Greening of the power needed by Impossible Foods may occur due a multitude of causes including changing the location of production to countries with more desirable grid mixes, construction of own (low GHG intensity) power supply and/or shifts of the regional US grid mix away from carbon intense fuels. The fossil fuel based energy mix used in this assessment accounts for a significant part of the climate change impacts produced by the PBB (see Fig 4). The global variation in the climate change intensity of one kWh ranges more than 2 orders of magnitude [70] meaning that choice of grid is paramount. For instance, the PBB's electricity related impacts could be reduced by a factor 7-8 by producing on a grid with similar GHG intensity to France. LCAs of cultured meat production linked water use and GHG performance to production location, underscoring the importance of geographic specificity [30].

In contrast, significant improvements in the US livestock supply chain do not appear immediately forthcoming in most regards. The majority of beef GHG impacts stem from enteric fermentation, which is physiologically constrained, and though higher quality feeds do have the potential to mitigate a portion of these, North American beef production systems are already amongst the world's leanest in this regard, limiting improvement potential through this route [9]. Long-term analysis of the US livestock production shows that feed to final product ratios have remained stable for all the staple livestock proteins throughout the 20^{th} century, with the exception of broilers which have seen substantial improvements [71]. The same can be said for current manure management practices [IBID]. Animal feed is also a major GHG source. Reduced tillage practices and improved yields would mitigate these, but given the advanced state of the majority of suppliers US production systems, such improvements are more salient to the low-tech livestock producers in the emerging economies [9,72]. Similarly, water use is physiologically constrained and strongly related to feed production. Exceptionally, land occupation could be significantly improved by switching from pasture to feedlot methods, though this potentially expands demand for arable cropland, and reduces demand for marginal rangeland.

4. 2 Additional aspects of US adoption of PBB

PBB adoption potentially reaps additional benefits not directly addressed through this assessment, including the reduction of reactive nitrogen runoff, a precursor to marine hypoxia and eutrophication. Livestock production is an important driver of these impacts at the regional and global from lax manure and urine management and runoff from fertilized feed crops [11,12,73]. PBB obviates both excrement production and the inefficiencies of converting feed to animal protein, ostensibly ameliorating eutrophication impacts in US beef supply regions, though more in depth analysis should buttress this claim.

Predicted increases in meat consumption at the global aggregate, particularly beef, will likely exacerbate stress on the planet's bio-geochemical cycles. Production improvements [9] and proactive land stewardship [21] can mitigate these to an extent, but stymying beef demand with PBB appears to be an alternative means towards more sustainable diets in the US and beyond. In a globalized and interconnected world, the ability for US dietary trends to diffuse into other cultures is more pronounced than ever, including cultures 'locked-in' to similar consumption levels of beef (Europe, Australia) and those only now ramping up their beef demands (China, India, Africa, etc.) Capturing the latter countries is particularly important before high beef consumption becomes the norm, since they have the capacity to significantly affect future global beef demands based on economic and population projections [1,71]. Moreover, as the US is currently a net exporter of beef [74], it is possible that US beef producers might simply export surplus production, hinting at the importance of dietary shifts beyond US borders.

It should be noted that contrary to the US and Latin America where ground beef is predominantly produced from dedicated beef herds, much of the ground beef in many European countries is a byproduct of spent dairy cattle and breeding overhead [75,76]. From an LCA perspective the GHG impacts of this type of beef are generally two thirds lower than those of a segregated beef herd due to co-product allocation [17,77], but still higher than those of PBB [76]. This relates again to the role of the Americas as a beef export region, since importing countries could be sourcing ground beef from more impacting locales, emphasizing the importance of consumption dynamics beyond the US border.

Essential to any discussion of the adoption of the PBB is the human factor. Changing diets is difficult and eating meat normalized in the United States [23–26]. Despite the PBB's superior performance at the product level when compared to beef, estimates of aggregate changes from large-scale adoption are speculative. These benefits hinge on the uptake of the PBB, and the results at the country level only express the potential of novel protein sources to reduce environmental impacts at their current production efficiencies. Lastly, the PBB is one of numerous novel protein sources [27], each having a signature resource profile, meaning that the environmental outcome of their uptake at the national level is conjectural.

5. Conclusions

It has been long known that reducing meat intake can play an important role in reducing the environmental impacts of the US diet and similar meat oriented diets. The challenge now is less about identifying the problem, but rather getting people to make a switch. This is a difficult proposition in the US where meat heavy diets are deeply enmeshed within its eating culture. Novel protein sources that substitute for environmentally deleterious livestock products while circumnavigating tough psychological hurdles offer a means to improve the environmental integrity of the MUD.

Novel protein substitutes, such as the PBB, could make important inroads to reducing the impacts of the MUD. When projected to the national level, the introduction of the PBB (and potentially other novel ground-beef replacements [30,32]) could generate substantial savings in GHG emissions, water consumption and agricultural land occupation. PBB has elevated net GHG emissions compared to other animal protein sources, but considering the age of the technology, it has substantial potential for improvement over animal-sourced foods, while providing benefits in additional realms of sustainability. PBB adoption can have slightly negative impacts on the VEG and VGN by some metrics, but a marginal uptake rate by the average American could counterbalance these.

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Supporting Information

S1 Spreadsheet. S1_File. Spreadsheet outlining the development of the baseline MUD, VEG and VGN diets, as well as their PBB substituted counterparts. (ONLINE ONLY)

S2 Table. S2_File. Table outlining the nutritional properties of PBB and ground beef. Analytic methods for PBB are also noted. (ONLINE ONLY)

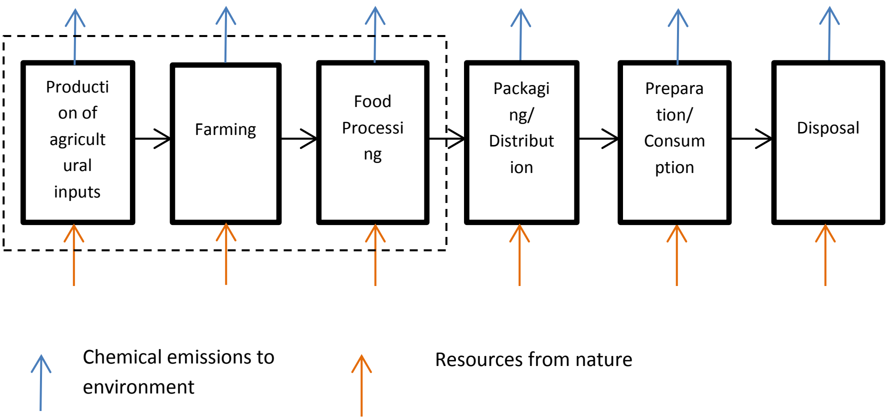
S3 Text. S3_File. PDF document detailing the life cycle assessment methodology employed here and the development of the life cycle inventories for the food products.

S4 Spreadsheet. S4_File. Spreadsheet showing the life cycle environmental impacts for individual food items, diets on the per capita level and national level consumption for GHG emissions, water consumption and land occupation. Also includes changes on impacts from PBB diffusion at different rates. (ONLINE ONLY)

S3 File. Life assessment methodology and life cycle inventories

Supporting information for the article 'Potential to curb the environmental burdens of American beef consumption using a novel plant based beef substitute'

1. Overview:



This study uses a hybrid life cycle assessment (LCA) approach that combines two different LCA methodologies: process-based LCA (P-LCA) and environmentally-extended-input-output LCA (EEIO-LCA). Figure 1 outlines the P-LCA methodology, which breaks down the life cycle of a product into different stages (material extraction, fabrication, distribution, etc.), accounts for the resource use and chemical emissions across all stages (aka. 'life cycle inventory'), converts different emissions into a common unit for each impact category (e.g. CO_2 equivalents for global warming) and then sums across the life cycle to estimate total environmental impacts. Here the scope of the assessment is on the agricultural production and processing stages as represented by the dashed line.

Here, all of the life cycle inventories (LCI) for agricultural production were built using P-LCA thinking, either with the ecoinvent 3.2 database or published results from previous LCAs. Data for food processing are not as readily available. EEIO-LCA was used to overcome this data gap.

EEIO-LCA works by augmenting standard economic input-output tables with environmental extensions outside of the economy. The foundation is **A,** the direct requirements matrix, representing interindustry (or sectoral) monetary exchanges. Each entry in the **A**, $\mathbf{a_{i,j}}$, represents the dollars demanded from industry **i** required by industry **j** to produce one dollar output from industry **j**. Vector **Y** represents demands from final consumers from each sector in dollars (i.e. excludes purchases from other industries that are used to produce goods for final consumption), where each element $\mathbf{y_i}$ represents the total final demand from industry **i**. Vector **X** represents the total economic activity of both the final consumption and the interindustry exchanges. Thus the total demand, **X**, can be represented in matrix notation as [1]:

[1] $$X = X \cdot A + Y$$

Solving for total demand yields [2]:

[2] $$X = (I - A)^{-1} Y$$

Where **I** is the identity matrix, and $\mathbf{(I\text{-}A)^{-1}}$ is known as the Leontief inverse.

The equation is linear and scalable in that a change in final consumption $\Delta\mathbf{Y}$ can be related to a change in total production $\Delta\mathbf{X}$ in the form [3]:

[3] $$\Delta X = (I - A)^{-1} \Delta Y$$

Environmental impacts can be included if emissions inventories from each industry (or sector) are known. For instance if total CO_2 emissions are known for sector **i**, then the CO_2 emissions per unit output, $\mathbf{R_i}$, can be determined as [4]:

[4] $R_i = total\ emissions_i / X_i$

From [4] the emissions, $\Delta\mathbf{b}$, due to an incremental change in final demand for sector can be taken as [5], where **R** is a vector of emissions factors per monetary unit output from each sector:

[5] $\Delta b = R(I - A)^{-1}\Delta Y$

This type of environmental extension can be applied wherever inventories exist (land and water use, acidification, etc.) A wealth of resources exist for those further interested in EEIO-LCA [1,2].
Here EEIO data from the Carnegie Mellon IO database for the US economy were used to account for environmental impacts from food processing for US final consumption (www.eiolca.net). The Carnegie Mellon model accounts for 428 economic sectors in the US economy in the year 2002. Although the database is dated for some industries, seismic technological transitions have not occurred in US agricultural production (nor background energy systems) in the meantime, and so impacts should be generally representative of 2010 production. Moreover, using the database assumes that all foods imported from foreign economies are technologically equivalent to the US, an assumption that should hold given the US's food security and low percentage of imports in the final consumption mix.
We only included the impacts from energy and chemical usage in food processing, since capital is typically of marginal contribution to total impacts. This is done by placing 0's in the direct requirements matrix, A, for those supplying industries that are not related to energy supply or chemical production. Thus, for each food processing process, zeros are placed in all rows with the exception of the following industries:

- Oil and gas extraction
- Coal mining
- Other nonmetallic mineral mining and manufacturing
- Electric power generation, transmission and distribution
- Natural gas distribution
- Petroleum refineries
- Petroleum lubricating oil and grease manufacturing
- Industrial gas manufacturing
- Synthetic dye and pigment manufacturing
- Alkalies and chlorine manufacturing
- Carbon black manufacturing
- All other basic inorganic chemical manufacturing
- Other basic organic chemical manufacturing
- Biological product (except diagnostic) manufacturing
- Soap and cleaning compound manufacturing
- All other chemical product and preparation manufacturing

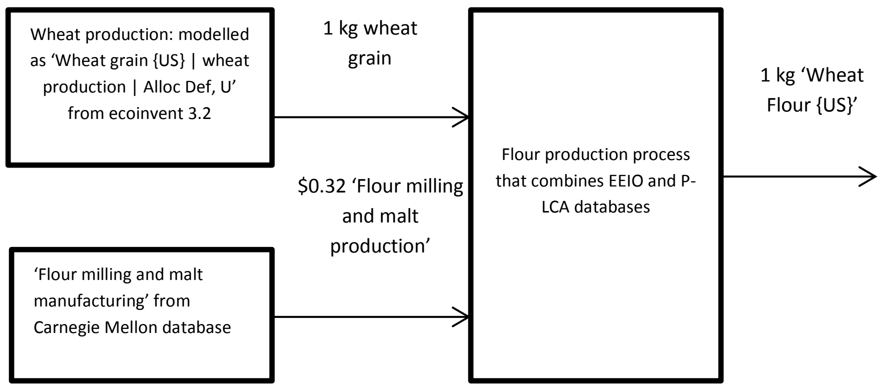
Because EEIO databases provide results in impacts per dollar final demand, it is necessary to convert this to impacts per unit mass so that they align with the mass-based P-LCA framework employed here. This was done using the USDA's Loss Adjusted Food Availability (LAFA)[3] data which allowed us to match the outputs from US sectors to the masses of food items produced in the baseline year. Table 1 shows the sectors used from the EEIO database, the dollars output from the sector[4], the corresponding foods in each sector, the output of those foods for the baseline year from the LAFA data, and the dollars output per kilogram produced. As not all food items are available in the LAFA (e.g. fresh and frozen fish are lumped into a single value), the total mass output of each sector will be underestimated in some instances, inflating the conversion factor. However, the overestimations should be slight since the LAFA numbers include those foods consumed in the highest volumes by Americans. LAFA numbers also represent food produced for US consumption, excluding exports, which should also inflate some of the numbers. Thus, where possible USDA production statistics were used[5], since these account for total US production of goods, including exports. This method generates the dollar demands for the average product produced by a sector, underestimating for some products and overestimating for others. Nonetheless it will provide a reasonable estimate of food processing related impacts.

EEIO Sector	**Output (10^6 2002 USD)**	**Food Items**	**2002 Production (t)**	**Conversion (USD/kg)**
Frozen Food Manufacturing	33177	All frozen food*	21498825	1.54
Fruit and vegetable canning, pickling, and drying	39283	Canned Fruit Dried Fruit Canned Vegetables Dried Vegetables Dried Beans	2282605 1439323 13763065 4398280 945994 22829274	1.72
Seafood product preparation and packaging	4869	Total Fish	2140374	2.27
Poultry processing	45242	Total Poultry**	23554466	1.92
Flour milling and malt manufacturing	6030	Total wheat flour	18762937	0.32
Wet corn milling	1724	Total corn products	4071886	0.42
Animal (except poultry) slaughtering, rendering, and processing	89239	Total red meat**	22563360	4.15
Total frozen dairy products	4420	Frozen Dairy	3498392	1.26
Dry, condensed, and evaporated dairy product manufacturing	12948	Evaporated/Condensed Milk	6674302 508517 7182189	1.80
Seasoning and dressing manufacturing	16303	Salads and cooking oils	16303110	2.14

* LAFA data do not include data on highly-processed, ready-made meals, so total production volume taken as 2002 US total frozen food consumption [6]

** Taken from USDA production reports[5]

With these conversion factors the P-LCA data can then be combined with EEIO-LCA data to generate LCIs encompassing both on-farm activities and upgrading from food processing. Figure 2 illustrates how the ecoinvent 3.2 P-LCA and Carnegie Mellon EEIO-LCA databases are combined to produce flour for the US market.



2. Life cycle inventories:

2.1. Ecoinvent based

Ecoinvent 3.2 provides full LCIs for some food products, including processing. These were used where possible and adjusted for US production conditions when appropriate (e.g. changing the supplying electrical grid to US). Table 2 notes these processes, as they are named in S3, the ecoinvent 3.2 process that each one is based upon and any alterations to the original ecoinvent process (foods sourced primarily from the global market were not altered). Proceeding sections outline custom LCIs for food products for which no ecoinvent 3.2 surrogates could be found. Where multi-functional processes occurred, system expansion was attempted (e.g. producing system was credited for avoided production of by-products), but mass-based allocation was a performed for beef from culled dairy cattle and cashew nuts, since the former is unfairly biased by system expansion [7] and the latter had a plurality of by-products that are difficult to model with system expansion.

Process used here	ecoinvent 3.2 basis	Adaptions for study
Broccoli {US}	Broccoli {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Spinach {US}	Spinach {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Iceberg Lettuce {US}	Iceberg Lettuce {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Green Bell Pepper	Green Bell Pepper {GLO}| production | Alloc Def, U	None
Cabbage red {US}	Cabbage red {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Celery {US}	Celery {GLO}| 675 production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Cucumbers {US}	Cucumber {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Aubergine {US}	Aubergine {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'

Onion {US}	Onion {GLO}| 855 production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Fava bean, organic {GLO}	Fava bean, organic {GLO}| Market for | Alloc Def, U	Transport to market removed
Avocado {GLO}	Avocado {GLO}| production | Alloc Def, U	None
Carrot {US}	Carrot {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Tomato {US}	Tomato {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Potato {US}	Potato {US}| production | Alloc Def, U	None
Citrus {US}	Citrus {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Strawberry {US}	Strawberry {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Melon {US}	Melon {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Apple {US}	Apple {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Banana {GLO}	Banana {GLO}| production | Alloc Def, U	None
Grape {US}	Grape {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U' Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Pear {US}	Pear {GLO}| production | Alloc Def, U	Electricity changed to 'Electricity, low voltage {US}| market group for | Alloc Def, U'

		Irrigation changed to 'Irrigation {US}| market for | Alloc Def, U'
Cheese {US}	Cheese, from cow milk, fresh, unripened {GLO}| cheese production, soft, from cow milk | Alloc Def, U	Electricity changed to 'Electricity, medium voltage {US}| market group for | Alloc Def, U' Cream and milk changed to US production scenarios (see below)
Milk {US}	Cow milk {CA-QC}| milk production, from cow | Alloc Def, U	Soybean feed changed to 'Soybean, feed {GLO}| production | Alloc Def, U' Maize changed to 'Maize grain, feed {US}| production | Alloc Def, U' 1% of impacts allocated to by-product beef production (see below)
Yoghurt {US}	Yogurt, from cow milk {CA-QC}| production | Alloc Def, U	Milk changed to 'Milk {US}' Electricity changed to 'Electricity, medium voltage {US}| market group for | Alloc Def, U'
Soybean Beverage {US}	Soybean beverage {CA-QC}| production | Alloc Def, U	Electricity changed to 'Electricity, medium voltage {US}| market group for | Alloc Def, U'
Tofu {US}	Tofu {CA-QC}| production | Alloc Def, U	Electricity changed to 'Electricity, medium voltage {US}| market group for | Alloc Def, U'
Palm oil, refined {GLO}	Palm oil, refined {GLO}| market for | Alloc Def, S	None
Cream {US}	Cream, from cow milk {CA-QC}| yogurt production, from cow milk | Alloc Def, U	Electricity changed to 'Electricity, medium voltage {US}| market group for | Alloc Def, U'
Sugar, from sugarcane {GLO}	Sugar, from sugarcane {GLO}| production | Alloc Def, U	None

2.2. Literature based

The following sections detail the life cycle inventories built from previous LCAs.

2.2.1. Frozen Broccoli

Output to Market		
Item	Amount	Unit
Frozen Broccoli {US}	1	kg
Production Factors		
Item	Amount	Unit
Broccoli {US}	1	kg
Frozen Food Manufacturing	1.543200617	2002 USD

2.2.2. Frozen Spinach

Output to Market		
Item	Amount	Unit
Frozen Spinach {US}	1	kg
Production Factors		
Item	Amount	Unit
Spinach {US}	1	kg
Frozen Food Manufacturing	1.543200617	2002 USD

2.2.3. Canned Olives [8]

Output to Market		
Item	Amount	Unit
Canned Olives {US}	1	kg
Production Factors		
Item	Amount	Unit
Pesticide, unspecified {GLO}| market for | Alloc Def, U	0.003333	kg
Irrigation {ES}| market for | Alloc Def, U	1	L
Nitrogen fertiliser, as N {GLO}| market for | Alloc Def, U	0.000145	kg
Diesel {GLO}| market group for | Alloc Def, U	0.008	kg
Lubricating oil {GLO}| market for | Alloc Def, U	0.000475	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD
Emissions to Air		
Substance	Amount	Unit
CO_2 (fossil)	0.283789	kg

2.2.4. Canned Mushrooms

Reliable LCI data on mushrooms are lacking. We estimate mushroom impacts as the average of all fresh vegetables, akin to Heller and Keoleian [9]. We add to the average the impacts from canning, taken as $1.80 2002 USD demand from the 'Fruit and vegetable canning, pickling, and drying' sector per kilogram canned mushrooms on the market.

2.2.5. Canned Beans

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Canned Beans {US}	1	kg
Production Factors		
Item	Amount	Unit
Fava bean, organic {GLO}	1.14	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.6. Canned Cucumbers

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Canned Cucumbers {US}	1	kg

Production Factors		
Item	Amount	Unit
Cucumber {US}	0.916	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.7. Frozen Beans

Output to Market		
Item	Amount	Unit
Frozen Beans {US}	1	kg
Production Factors		
Item	Amount	Unit
Fava bean, organic {GLO}	1	kg
Frozen Food Manufacturing	1.543200617	2002 USD

2.2.8. Dried Onions

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Dried Onion {US}	1	kg
Production Factors		
Item	Amount	Unit
Onion {US}	9.1	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.9. Canned Carrots

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Canned Carrots {US}	1	kg
Production Factors		
Item	Amount	Unit
Carrots {US}	1.12	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.10. Canned Peppers

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Canned Peppers {US}	1	kg
Production Factors		
Item	Amount	Unit
Pepper {US}	1.04	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.11. Canned Tomatoes

Output to Market		
Item	Amount	Unit
Canned Tomatoes {US}	1	kg
Production Factors		
Item	Amount	Unit
Tomato {US}	1	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.12. Frozen Carrots

Output to Market		
Item	Amount	Unit
Frozen Carrots {US}	1	kg
Production Factors		
Item	Amount	Unit
Carrot {US}	1	kg
Frozen Food Manufacturing	1.543200617	2002 USD

2.2.13. Canned Corn

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Canned Corn {US}	1	kg
Production Factors		
Item	Amount	Unit
Maize grain {US}| production | Alloc Def, U	0.79	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.14. Frozen Corn

Output to Market		
Item	Amount	Unit
Frozen Corn {US}	1	kg
Production Factors		
Item	Amount	Unit
Maize grain {US}| production | Alloc Def, U	1	kg
Frozen Food Manufacturing	1.543200617	2002 USD

2.2.15. Frozen Potatoes

Output to Market		
Item	Amount	Unit
Frozen Potatoes {US}	1	kg
Production Factors		
Item	Amount	Unit
Potato {US}| production | Alloc Def, U	1	kg
Frozen Food Manufacturing	1.543200617	2002 USD

2.2.16. Dried Potatoes

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Dried Potatoes {US}	1	kg
Production Factors		
Item	Amount	Unit
Potato {US}| production | Alloc Def, U	4.819277108	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.17. Orange Juice

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Orange Juice {US}	1	kg
Production Factors		
Item	Amount	Unit
Citrus {US}	2	kg

2.2.18. Grapefruit Juice

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Grapefruit Juice {US}	1	kg
Production Factors		
Item	Amount	Unit
Citrus {US}	2.25	kg

2.2.19. Apple Juice

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Apple Juice {US}	1	kg
Production Factors		
Item	Amount	Unit
Apple {US}	1.5	kg

2.2.20. Cranberry Juice

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market

Item	Amount	Unit
Cranberry Juice {US}	1	kg
Production Factors		
Item	Amount	Unit
Strawberry {US}*	1.3	kg

* Strawberry taken as surrogate for cranberry. Strawberry {US} process adapted for energy and irrigation in same manner as shown in table 2

2.2.21. Grape Juice

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Grape Juice {US}	1	kg
Production Factors		
Item	Amount	Unit
Grape {US}*	1.3	kg

* Grape {US} process adapted for energy and irrigation in same manner as shown in table 2

2.2.22. Pineapple Juice

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Pineapple Juice {US}	1	kg
Production Factors		
Item	Amount	Unit
Pineapple {GLO}| production | Alloc Def, U	1.88	kg

2.2.23. Canned Apples

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Canned Apples {US}	1	kg
Production Factors		
Item	Amount	Unit
Apple {US}	1.018	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2

2.2.24. Raisins

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Raisins {US}	1	kg
Production Factors		
Item	Amount	Unit
Grape {US}	4.52	kg

Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

2.2.25. Wheat Flour

Output to Market		
Item	Amount	Unit
Wheat Flour {US}	1	kg
Production Factors		
Item	Amount	Unit
Wheat grain {US}| wheat production | Alloc Def, U	1	kg
Flour milling and malt manufacturing	0.32137825	2002 USD

2.2.26. Processed Rice

Output to Market		
Item	Amount	Unit
Processed Rice {US}	1	kg
Production Factors		
Item	Amount	Unit
Maize grain {US}| production | Alloc Def, U	1	kg
Wet corn milling	0.423391068	2002 USD

2.2.27. Oat Products

Output to Market		
Item	Amount	Unit
Oat Products {US}	1	kg
Production Factors		
Item	Amount	Unit
Oat grain {CA-QC}| oat production | Alloc Def, U	1	kg
Flour milling and malt manufacturing	0.32137825	2002 USD

2.2.28. Ice Cream

Output to Market		
Item	Amount	Unit
Ice Cream {US}	1	kg
Production Factors		
Item	Amount	Unit
Cream {US}	1	kg
Ice cream and frozen dessert manufacturing	1.263437634	2002 USD

2.2.29. Condensed Milk

Mass change during production taken from the USDA Food Intakes Converted to Retail Commodities Database (FICRCD) [10]

Output to Market		
Item	Amount	Unit
Condensed Milk {US}	1	kg
Production Factors		
Item	Amount	Unit

Milk {US}	1.23	kg
Evaporation of milk {US}| milk evaporation | Alloc Def, U*	1.23	kg

* Energy and heating sources changed to US

2.2.30. Eggs [11]

Output to Market		
Item	Amount	Unit
Layer Feed	1	kg
Production Factors		
Item	Amount	Unit
Maize grain {GLO}| market for | Alloc Def, U	0.712555066	kg
Soybean meal {GLO}| market for | Alloc Def, U	0.212555066	kg
Lime {GLO}| market for | Alloc Def, U	0.074889868	kg

Output to Market		
Item	Amount	Unit
Pullet	1	pullet
Production Factors		
Item	Amount	Unit
Layer Feed	5.27	kg
Irrigation {US}| market for | Alloc Def, U	9.22	kg
Electricity, low voltage {US}| market group for | Alloc Def, U	3.015	MJ
Diesel, low-sulfur {GLO}| market group for | Alloc Def, U	0.00246	kg
Petrol, unleaded {RoW}| market for | Alloc Def, U	0.0022	kg
Propane {GLO}| market for | Alloc Def, U	0.0357	kg
Natural gas, high pressure {US}| market for | Alloc Def, U	0.180	ft^3
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	0.61734	kg

Output to Market		
Item	Amount	Unit
Eggs {US}	1	kg
By-products		
Item	Amount	Unit
Chicken for slaughtering, live weight {GLO}| chicken production | Alloc Def, U	0.02	kg
Production Factors		
Item	Amount	Unit
Pullet	0.036	pullets
Layer Feed	1.980	kg
Irrigation {US}| market for | Alloc Def, U	4.26	kg
Electricity, low voltage {US}| market group for | Alloc Def, U	0.557	MJ
Diesel, low-sulfur {GLO}| market group for | Alloc Def, U	0.00162	kg
Petrol, unleaded {RoW}| market for | Alloc Def, U	0.000207	kg
Propane {GLO}| market for | Alloc Def, U	0.00175	kg
Natural gas, high pressure {US}| market for | Alloc Def, U	0.00386	ft^3
Poultry processing	1.920739745	2002 USD
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	0.02084	kg

2.2.31. Dried Beans

Output to Market		
Item	Amount	Unit
Dried Beans {US}	1	kg
Production Factors		
Item	Amount	Unit
Fava bean, organic {GLO}	1.13*	kg
Fruit and vegetable canning, pickling, and drying	1.720729275	2002 USD

* Adjusted for 11.33% water content from USDA for raw pinto beans (item 16042): https://ndb.nal.usda.gov/ndb/search/list

2.2.32. Beef, feedlot finished [12]

The LCI is provided for one kilogram of live weight beef for slaughter, which is then adjusted for the carcass yield (the ratio of meat to live weight). An upper carcass yield of 0.428 was taken from the Cornell Small Farms Program [13]. A lower carcass yield of 0.390 was taken as the product of the dressing percentage of 0.59 (11902714932 kg carcass weight produced/34407300 head slaughtered/580 kg live weight per head in 2011) [14] and carcass cutting yield of 0.65 (24.6 kg beef on market/person * 314100000 people/11902714932 kg carcass weight produced in 2011) [3].

Output to Market		
Item	Amount	Unit
Beef, feedlot finished {US}	yield	kg
By-products		
Item	Amount	Unit
Liquid manure spreading, by vacuum tanker {GLO}| market for | Alloc Def, U	0.0041	m^3
Production Factors		
Item	Amount	Unit
Natural gas, high pressure {CA-AB}| natural gas production | Alloc Def, U	0.798	m^3
Diesel {GLO}| market group for | Alloc Def, U	0.00303	kg
Electricity, low voltage {MRO, US only}| market for | Alloc Def, U	0.08067	kWh
Alfalfa-grass silage {GLO}| market for | Alloc Def, U	0.431	kg
Maize silage {GLO}| market for | Alloc Def, U	1.04	kg
Maize grain, feed {GLO}| market for | Alloc Def, U	3.49	kg
Energy feed, gross {GLO}| soybean meal to generic market for protein feed | Alloc Def, U	0.00676	MJ
Irrigation {US}| market for | Alloc Def, U*	0.152	m^3
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	1.56957	kg
Methane	0.238468168	kg
Ammonia	0.0193	kg
Nitrate	0.248	kg

* Blue water taken from Mekonnen and Hoekstra, 2012 [15]

2.2.33. Beef, backgrounding-feedlot finished [12]

Yield calculated in the same manner as above.

Output to Market		
Item	Amount	Unit
Beef, backgrounding-feedlot finished {US}	yield	kg
By-products		

Item	Amount	Unit
Liquid manure spreading, by vacuum tanker {GLO}| market for | Alloc Def, U	0.0041	m^3
Production Factors		
Item	Amount	Unit
Occupation, pasture and meadow*	85.7	m^2a
Natural gas, high pressure {CA-AB}| natural gas production | Alloc Def, U	0.798	m^3
Diesel {GLO}| market group for | Alloc Def, U	0.00303	kg
Electricity, low voltage {MRO, US only}| market for | Alloc Def, U	0.08067	kWh
Alfalfa-grass silage {GLO}| market for | Alloc Def, U	0.364	kg
Maize silage {GLO}| market for | Alloc Def, U	3.43	kg
Wheat Pasture {US-midwest}	12.5	kg
Irrigation {US}| market for | Alloc Def, U**	0.234	m^3
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	1.56957	kg
Methane	0.238468168	kg
Ammonia	0.0193	kg
Nitrate	0.248	kg

* Land occupation taken as the ratio of total land for beef/veal/lamb pasture from Table 8 of the USDA 2012 Census of Agriculture ($1.68 * 10^{12}$ m^2) [16] divided by total market availability of red meat in live weight for that year ($1.9*10^{10}$ kg)[14]

** Blue water for taken from Mekonnen and Hoekstra, 2012 [15]

2.2.34. Beef, culled dairy cattle

Beef and milk are co-products in this system. Mass based allocation is used to apportion the impacts to each product. Average US dairy cow has 2.63 years of milk production [17], producing on average 9566 kg milk/annum in the U.S. [18] and a slaughter weight of 566 kg for the top three dairy breeds [19], then given the two carcass yields of 0.390 and 0.428 there is either 0.0087 kg beef at market/kg milk or 0.0095 kg beef at market/kg milk produced, respectively. Impacts from the milk production system are thus allocated accordingly. Alternatively an economic allocation of $600 [17] for the slaughtered dairy cow and 0.531 USD/kg milk [20] would yield an allocation of 4.3% of milk impacts to the beef by-product, which is relatively significant, but not critical to the results here (~1% decrease in results for MUD).

2.2.35. Ground beef

Beef for ground beef comes from two streams, dedicated beef herds (split 50/50 between feedlot and backgrounding-feedlot finished [12]) and culled dairy cattle. We assume that all culled dairy cattle go to ground beef due to the low quality of the meat. Assuming a steady state cow population of 9252272 dairy cows in 2012 [16] and an average life of 4.8 years [17], implies 1927557 culled dairy cattle per year. At 566 kg/head and the aforementioned carcass yields, between 425488932 and 466946828 kg of ground beef from dairy cattle are hitting the market annually.

Of the 7781113636 kg of total beef on the U.S market in 2012, it is estimated by industry that 50% of this was ground beef [21]. We also used the Center for Disease Control and Preventions National Nutrition and Health Examination Survey (NHANES) from 2011/2012 to estimate the amount of ground beef for all beef consumers [22]. 8-digit USDA food codes were used to disaggregate total mass of food consumed into constituent masses using the FICRCD [10]. Those food codes that corresponded to ground beef as the main ingredient were added to the ground beef total. A text search was also performed of the meal descriptions to capture any meals that contain ground-beef as a secondary ingredient, with the amount of beef in meals meeting the search criteria added to the ground beef total. The total amount of ground beef was then divided by the total beef amount to come to an estimate of 30% ground beef as percentage of total beef consumption. The difference between our estimate and the industry estimate could come from the underreporting of unhealthy foods in self-reporting dietary surveys or due to the difficulty of capturing all meals containing ground beef as second ingredients in the USDA food descriptions.

Upper and lower bounds for total ground beef on the market are thus 3890556818 and 2334334091 kg/annum, respectively. At a carcass yield of 0.390, between 10.9% and 18.2% of ground beef in the US originates from the dairy herd. At a carcass yield of 0.428, between 12.0% and 20.0% of US ground beef originates from the dairy herd.

Output to Market		
Item	Amount	Unit
Ground beef {US}	1	kg
Production Factors		
Item	Amount	Unit
Beef, feedlot finished {US}	(1-% dairy)/2*	kg
Beef, backgrounding-feedlot finished {US}	(1-% dairy)/2*	kg
Beef, culled dairy cattle {US}	% dairy	kg
Animal (except poultry) slaughtering, rendering, and processing	4.150212451	2002 USD

* Divide by two to account for the 50/50 split between supplying streams

2.2.36. Beef {US}

Taken as 50/50 split between feedlot and backgrounding-feedlot finished, as these are the two dominant production systems in the US [12].

Output to Market		
Item	Amount	Unit
Beef {US}	1	kg
Production Factors		
Item	Amount	Unit
Beef, backgrounding-feedlot finished {US}	1	kg
Beef, culled dairy cattle {US}	1	kg
Animal (except poultry) slaughtering, rendering, and processing	4.150212451	2002 USD

2.2.37. Pork [23]

Austrian pork production used as surrogate for US production.

Output to Market		
Item	Amount	Unit
Pork {US}	1	kg
By-products		
Item	Amount	Unit
Nitrogen fertiliser, as N {GLO}| field application of ammonium chloride | Alloc Def, U	0.030176	kg
Phosphate fertiliser, as P2O5 {GLO}| market for | Alloc Def, U	0.010938	kg
Potassium chloride, as K2O {GLO}| market for | Alloc Def, U	0.027637	kg
Production Factors		
Item	Amount	Unit
Maize grain {GLO}| market for | Alloc Def, U	1.484375	kg
Wheat grain, feed {GLO}| market for | Alloc Def, U	0.742188	kg
Barley grain, feed {GLO}| market for | Alloc Def, U	0.742188	kg
Protein feed, 100% crude {GLO}| soybean meal to generic market for protein feed | Alloc Def, U	0.283203	kg
Protein feed, 100% crude {GLO}| rape meal to generic market for protein feed | Alloc Def, U	0.146484	kg
Sunflower silage {RoW}| catch crop growing, sunflower, August-October, organic fertiliser 20 kg N | Alloc Def, U	0.146484	kg
Irrigation {US}| market for | Alloc Def, U	0.011738	m^3
Electricity, low voltage {US}| market group for | Alloc Def, U	0.341797	kWh
Heat, central or small-scale, other than natural gas {GLO}| market group for | Alloc Def, U	0.185547	kWh
Animal (except poultry) slaughtering, rendering, and processing	4.150212451	2002

		USD
Emissions to Air		
Substance	Amount	Unit
Methane	0.032353516	kg
Ammonia	6.84E-05	kg
Nitrous Oxide	0.015615	kg

2.2.38. Chicken

Output to Market		
Item	Amount	Unit
Chicken {US}	1	kg
Production Factors		
Item	Amount	Unit
Chicken for slaughtering, live weight {GLO}| chicken production | Alloc Def, U	1.33*	kg
Poultry processing	1.920739745	2002 USD

* Carcass yield taken from [13]

2.2.39. Peanuts [24]

Output to Market		
Item	Amount	Unit
Peanuts {US}	1	kg
Production Factors		
Item	Amount	Unit
Occupation, agriculture	5.88	m^2a
Rye seed, for sowing {GLO}| market for | Alloc Def, U	0.005317073	kg
Lime {GLO}| market for | Alloc Def, U	0.317195122	kg
Diesel, low-sulfur {GLO}| market group for | Alloc Def, U	0.044927	kg
Borax, anhydrous, powder {GLO}| market for | Alloc Def, U	0.000170732	kg
Pesticide, unspecified {GLO}| market for | Alloc Def, U	0.007634146	kg
Electricity, low voltage {US}| market group for | Alloc Def, U	0.062317	kWh
Irrigation {US}| market for | Alloc Def, U	0.246341	m^3
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	0.141951	kg

2.2.40. Almonds [25]

Output to Market		
Item	Amount	Unit
Almonds {US}	1	kg
Production Factors		
Item	Amount	Unit
Occupation, agriculture	0.445	m^2a
Petrol, unleaded {RoW}| market for | Alloc Def, U	0.0297	kg
Diesel, low-sulfur {GLO}| market group for | Alloc Def, U	0.0398	kg
2,4-dichlorophenol {GLO}| market for | Alloc Def, U	0.000145	kg
Sulfur {GLO}| market for | Alloc Def, U	0.034	kg
Nitrogen fertiliser, as N {GLO}| market for | Alloc Def, U	0.0609	kg
Zinc {GLO}| market for | Alloc Def, U	0.00125	kg
Irrigation {US}| market for | Alloc Def, U	4.98	m^3
Electricity, low voltage {WECC, US only}| market for | Alloc Def, U	1.5	kWh
Emissions to Air		

Substance	Amount	Unit
Carbon dioxide (fossil)	0.1557	kg

2.2.41. Walnuts [25]

Output to Market		
Item	Amount	Unit
Walnuts {US}	1	kg
Production Factors		
Item	Amount	Unit
Occupation, agriculture	0.178	m^2a
Petrol, unleaded {RoW}| market for | Alloc Def, U	0.0162	kg
Diesel, low-sulfur {GLO}| market group for | Alloc Def, U	0.0078	kg
2,4-dichlorophenol {GLO}| market for | Alloc Def, U	0.000284	kg
Nitrogen fertiliser, as N {GLO}| market for | Alloc Def, U	0.0183	kg
Irrigation {US}| market for | Alloc Def, U	4.98	m^3
Electricity, low voltage {WECC, US only}| market for | Alloc Def, U	0.164	kWh
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	0.0409	kg

2.2.42. Cashew [26]

Austrian pork production used as surrogate for US production.

Output to Market		
Item	Amount	Unit
Cashew Nuts {BR}	1	kg
By-products		
Item	Amount	Unit
See source*	1.27	kg
Production Factors		
Item	Amount	Unit
Arable land use, soy bean, Brazil	0.0577	m^2a
Limestone, crushed, washed {GLO}| market for | Alloc Def, U	0.63473745	kg
Gypsum, mineral {GLO}| market for | Alloc Def, U	0.028851702	kg
Copper {GLO}| market for | Alloc Def, U	9.60185E-6	kg
Manganese {GLO}| market for | Alloc Def, U	2.40046E-5	kg
Molybdenum {GLO}| market for | Alloc Def, U	1.20023E-6	kg
Zinc {GLO}| market for | Alloc Def, U	0.000108021	kg
Iron pellet {GLO}| market for | Alloc Def, U	3.60069E-5	kg
Urea, as N {GLO}| market for | Alloc Def, U	0.203084247	kg
Phosphate fertiliser, as P2O5 {RoW}| single superphosphate production | Alloc Def, U	0.465089152	kg
Potassium chloride, as K2O {GLO}| market for | Alloc Def, U	0.054010387	kg
Glyphosate {GLO}| market for | Alloc Def, U	0.001436815	kg
Diesel {GLO}| market group for | Alloc Def, U	0.089118725	kg
Irrigation {BR}| market for | Alloc Def, U	5.476053087	m^3

* Mass allocation performed

2.2.43. Fresh and frozen fish/Canned Fish/Canned Sardines

Output to Market		
Item	Amount	Unit
Salmon, Atlantic {ES}	1	kg
Production Factors		

Item	Amount	Unit
Freight ship, transoceanic {GLO}| market for | Alloc Def, U	7.10997E-09	ship
Diesel, low-sulfur {GLO}| market group for | Alloc Def, U	0.359335038	kg
Alkyd paint, white, without water, in 60% solution state {GLO}| market for | Alloc Def, U	8.95141E-05	kg
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	1.135549872	kg

Taken from Hospido et al., 2005 [27]

Output to Market		
Item	Amount	Unit
Fish feed	1	kg
Production Factors		
Item	Amount	Unit
Rape seed, organic {CH}| production | Alloc Def, U	0.061	m^2a
Rape oil, crude {CH}| market for | Alloc Def, U	0.041	kg
Rape meal {GLO}| to generic market for protein feed | Conseq, U	0.051	kg
Protein pea {GLO}| market for | Alloc Def, U, U	0.051	kg
Maize grain, feed {GLO}| market for | Alloc Def, U	0.092	kg
Energy feed, gross {GLO}| skimmed milk, from cow milk to generic market for protein feed | Alloc Rec, U	0.056	MJ
Wheat grain, feed {GLO}| market for | Alloc Def, U	0.133	kg
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	1.2848	kg
Sulfur dioxide	0.0076	kg

Taken from Pelletier et al., 2009 [28]

Output to Market		
Item	Amount	Unit
Salmon, farmed {US}	1	kg
Production Factors		
Item	Amount	Unit
Fish feed	1.313	m^2a
Transport, freight, lorry >32 metric ton, EURO6 {GLO}| market for | Alloc Def, U	0.3192	tkm
Electricity, high voltage {MRO, US only}| market for | Alloc Def, U	2326.9	MJ

Taken from Pelletier et al., 2009 [28]

Output to Market		
Item	Amount	Unit
Fresh and frozen fish {US}	1	kg
Production Factors		
Item	Amount	Unit
Tuna, Atlantic {ES}	0.875	kg
Salmon, farmed {US}	0.125	kg
Seafood product preparation and packaging	2.274835635	2002 USD

87.5% of fish consumed in US is wild (http://www.seafoodhealthfacts.org/seafood-choices/overview-us-seafood-supply), Atlantic Tuna and farmed Salmon used as proxies for all wild and farmed fish consumed in US respectively

Output to Market		
Item	Amount	Unit
Canned fish {US}	1	kg
Production Factors		
Item	Amount	Unit
Tuna, Atlantic {ES}	0.865	kg
Salmon, farmed {US}	0.124	kg
Seafood product preparation and packaging	2.274835635	2002 USD

87.5% of fish consumed in US is wild (http://www.seafoodhealthfacts.org/seafood-choices/overview-us-seafood-supply), Atlantic Tuna and farmed Salmon used as proxies for all wild and farmed fish consumed in US respectively. Mass change from fresh to canned taken from FICRCD [10], the difference being water.

Output to Market		
Item	Amount	Unit
Canned sardines {US}	1	kg
Production Factors		
Item	Amount	Unit
Tuna, Atlantic {ES}	0.868	kg
Salmon, farmed {US}	0.124	kg
Seafood product preparation and packaging	2.274835635	2002 USD

87.5% of fish consumed in US is wild (http://www.seafoodhealthfacts.org/seafood-choices/overview-us-seafood-supply), Atlantic Tuna and farmed Salmon used as proxies for all wild and farmed fish consumed in US respectively. Mass change from fresh to canned taken from FICRCD [10], the difference being water.

2.2.44. Fresh and frozen shellfish [29]

Output to Market		
Item	Amount	Unit
Fresh and frozen shellfish {TH}	1	kg
Production Factors		
Item	Amount	Unit
Occupation, water bodies, artificial	0.411	m^2a
Fish feed	1.9	kg
Diesel {GLO}| market group for | Alloc Def, U	0.0397	kg
Liquefied petroleum gas {RoW}| market for | Alloc Def, U	0.08	kg
Seafood product preparation and packaging	2.274835635	2002 USD
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	0.366	kg

* Taken from Pelletier et al., 2009 [28]

2.2.45. Cured fish

Output to Market		
Item	Amount	Unit
Cured fish {US}	1	kg
Production Factors		
Item	Amount	Unit
Tuna, Atlantic {ES}	2.36	kg
Salmon, farmed {US}	0.338	kg
Seafood product preparation and packaging	2.274835635	2002 USD

87.5% of fish consumed in US is wild (http://www.seafoodhealthfacts.org/seafood-choices/overview-us-seafood-supply), Atlantic Tuna and farmed Salmon used as proxies for all wild and farmed fish consumed in US respectively. Mass change from fresh to canned taken from FICRCD [10], losses due to dehydration.

2.2.46. Plant based burger (PBB)

Output to Market		
Item	Amount	Unit
PBB Ingredients	1	kg
Production Factors		
Item	Amount	Unit
Ammonia, liquid {RER}| market for | Alloc Rec, U	0.0156	kg

Ammonium sulfate, as N {GLO}| market for | Alloc Rec, U	0.00547	kg
Chemical, organic {GLO}| market for | Alloc Rec, U	0.00472096	kg
Boric acid, anhydrous, powder {GLO}| market for | Alloc Rec, U	0.00000005	kg
Calcium Sulfate Dihydrate*	0.00015	kg
Acetic acid, without water, in 98% solution state {GLO}| market for | Alloc Rec, U	0.13833825	kg
Cobalt {GLO}| market for | Alloc Rec, U	0.00000108	kg
Copper sulfate {GLO}| market for | Alloc Rec, U	0.000013	kg
Iron sulfate {GLO}| market for | Alloc Rec, U	0.00014	kg
Magnesium sulfate {GLO}| market for | Alloc Rec, U	0.00408	kg
Manganese sulfate {GLO}| market for | Alloc Rec, U	0.00000648	kg
Potassium carbonate {GLO}| market for | Alloc Rec, U	0.00328	kg
Sodium hydroxide, without water, in 50% solution state {GLO}| market for | Alloc Rec, U	0.14826	kg
Sodium {GLO}| market for | Alloc Rec, U	0.0000006	kg
Sulfuric acid {GLO}| market for | Alloc Rec, U	0.0000108	kg
Water, completely softened, from decarbonised water, at user {GLO}| market for | Alloc Rec, U	13.991	kg
Zinc {GLO}| market for | Alloc Rec, U	0.0000257	kg
Maize grain {GLO}| market for | Alloc Rec, U	0.4181749	kg
Acetic acid, without water, in 98% solution state {GLO}| market for | Alloc Rec, U	0.13833825	kg
Glycine {GLO}| market for | Alloc Rec, U	0.0169156	kg
Lactic acid {GLO}| market for | Alloc Rec, U	0.00088	kg
Chemical, organic {GLO}| market for | Alloc Rec, U	0.00472096	kg
Sodium hydroxide, without water, in 50% solution state {GLO}| market for | Alloc Rec, U	0.14826	kg
Coconut oil, crude {PH}| production | Alloc Rec, U	0.166	kg
Acetic acid, without water, in 98% solution state {GLO}| market for | Alloc Rec, U	0.13833825	kg
Potato protein*	0.066	kg
Soybean {GLO}| market for | Alloc Rec, U	0.00209	kg
Wheat gluten meal, consumption mix, at feed compound plant/NL Economic	0.25	kg
Water, completely softened, from decarbonised water, at user {GLO}| market for | Alloc Rec, U	13.991	kg

* Taken from Agri-footprint database

Output to Market		
Item	Amount	Unit
Electricity, low voltage, US average	1	kWh
Production Factors		
Item	Amount	Unit
Electricity, low voltage {ASCC}| market for | Alloc Rec, U	0.002	kWh
Electricity, low voltage {FRCC}| market for | Alloc Rec, U	0.052	kWh
Electricity, low voltage {HICC}| market for | Alloc Rec, U	0.003	kWh
Electricity, low voltage {MRO, US only}| market for | Alloc Rec, U	0.057	kWh
Electricity, low voltage {NPCC, US only}| market for | Alloc Rec, U	0.064	kWh
Electricity, low voltage {RFC}| market for | Alloc Rec, U	0.227	kWh
Electricity, low voltage {SERC}| market for | Alloc Rec, U	0.269	kWh
Electricity, low voltage {SPP}| market for | Alloc Rec, U	0.055	kWh
Electricity, low voltage {TRE}| market for | Alloc Rec, U	0.089	kWh
Electricity, low voltage {WECC, US only}| market for | Alloc Rec, U	0.182	kWh

Average US grid mix used since final production location is unknown

Output to Market		
Item	Amount	Unit
PBB	1	kg
Production Factors		
Item	Amount	Unit
PBB Ingredients	1.1*	kg

Electricity, low voltage, US average	5.165	kWh
Transport, freight, lorry 16-32 metric ton, EURO6 {GLO}| market for | Alloc Rec, U	1.35	tkm
Heat, central or small-scale, natural gas {RoW}| market for heat, central or small-scale, natural gas | Alloc Rec, U	0.161	kWh
Compressed air, 800 kPa gauge {GLO}| market for | Alloc Rec, U	3.22	m^3
Carbon dioxide, liquid {RER}| market for | Alloc Rec, U	0.297	kg

* 10% loss assumed

2.2.47. Vegetable Oil Mix

Taken as margarine from Nilsson et al., 2010 [30]

Output to Market		
Item	Amount	Unit
Vegetable Oil Mix {GLO}	1	kg
Production Factors		
Item	Amount	Unit
Vegetable oil, refined {GLO}| palm oil, refined, to generic market for vegetable oil | Conseq, S	0.474074074	kg
Soybean oil, refined {GLO}| to generic market for vegetable oil, refined | Conseq, U	0.331851852	kg
Rape oil, crude {Europe without Switzerland}| rape oil mill operation | Conseq, U	0.194074074	kg

2.2.48. High Fructose Corn Syrup (HFCS)[31]

Output to Market		
Item	Amount	Unit
HFCS {US}	1	kg
By-products		
Item	Amount	Unit
Maize grain, feed {GLO}| market for | Alloc Def, U	0.348	kg
Palm kernel oil, crude {GLO}| market for | Alloc Def, U	0.027	kg
Production Factors		
Item	Amount	Unit
Maize grain {US}| production | Alloc Def, U	1.5	kg
Lime {GLO}| market for | Alloc Def, U	0.0003	kg
Sulfuric acid {GLO}| market for | Alloc Def, U	0.00045	kg
Sulfur dioxide, liquid {GLO}| market for | Conseq, S	0.00306	kg
Urea, as N {GLO}| market for | Alloc Def, U	0.000208	kg
Sodium chloride, powder {GLO}| market for | Alloc Def, U	0.000065	kg
Sodium hydroxide, without water, in 50% solution state {GLO}| market for | Alloc Def, U	0.000282	kg
Cyclohexane {GLO}| market for | Alloc Def, U	0.000055	kg
Chlorine, liquid {GLO}| market for | Alloc Def, U	0.000012	kg
Water, decarbonised, at user {GLO}| market for | Alloc Def, U	4.9	kg
Electricity, low voltage {US}| market group for | Alloc Def, U	0.934	MJ
Natural gas, high pressure {US}| market for | Alloc Def, U	0.0596	m^3
Emissions to Air		
Substance	Amount	Unit
Carbon dioxide (fossil)	0.17	kg

2.2.49. Glucose from corn

Output to Market		
Item	Amount	Unit

Glucose from corn {US}	1	kg
Production Factors		
Item	Amount	Unit
Maize grain {US}| production | Alloc Def, U	1.6	kg

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The past decade has seen a renaissance of urban farming in the Northeast US. One of the key motivations is the production of more environmentally sustainable food for cities. However, key questions remain unanswered: Is urban farming more resource efficient than conventional forms of food procurement in the Northeast US? Could a city such as Boston or New York City make considerable reductions in its food-borne greenhouse gas emissions through local farming? How much does food contribute to a city's overall environmental footprint? This Ph.D. responds to these practical problems, while bringing theoretical and methodological advances in the way that industrial ecologists conceptualize and assess urban sustainability.

DTU Management Engineering
Department of Management Engineering
Technical University of Denmark

Produktionstorvet
Building 424
DK-2800 Kongens Lyngby
Denmark
Tel. +45 45 25 48 00
Fax +45 45 93 34 35

www.man.dtu.dk