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A Global Geospatial Ecosystem Services Estimate of Urban Agriculture

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Key Points:
- We present an original, data-driven approach to quantify country-scale multifunctionality of urban agriculture (UA) adoption
- We estimate that UA can provide a range of ecosystem services, which highlights the role of built environments for increasing adaptive capacity to climate change
- We estimate that China, Japan, Germany, and the United States have a suitable combination of factors to be consistently among the top beneficiaries of UA in terms of estimated dollar potential and associated ecosystem services

Supporting Information:
- Table S1.

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Abstract Though urban agriculture (UA), defined here as growing of crops in cities, is increasing in popularity and importance globally, little is known about the aggregate benefits of such natural capital in built-up areas. Here, we introduce a quantitative framework to assess global aggregate ecosystem services from existing vegetation in cities and an intensive UA adoption scenario based on data-driven estimates of urban morphology and vacant land. We analyzed global population, urban, meteorological, terrain, and Food and Agriculture Organization (FAO) datasets in Google Earth Engine to derive global scale estimates, aggregated by country, of services provided by UA. We estimate the value of four ecosystem services provided by existing vegetation in urban areas to be on the order of $33 billion annually. We project potential annual food production of 100–180 million tonnes, energy savings ranging from 14 to 15 billion kilowatt hours, nitrogen sequestration between 100,000 and 170,000 tonnes, and avoided storm water runoff between 45 and 57 billion cubic meters annually. In addition, we estimate that food production, nitrogen fixation, energy savings, pollination, climate regulation, soil formation and biological control of pests could be worth as much as $80–160 billion annually in a scenario of intense UA implementation. Our results demonstrate significant country-to-country variability in UA-derived ecosystem services and reduction of food insecurity. These estimates represent the first effort to consistently quantify these incentives globally, and highlight the relative spatial importance of built environments to act as change agents that alleviate mounting concerns associated with global environmental change and unsustainable development.

1. Introduction

More than half of the world's population lives in cities, a proportion expected to increase to 67% by 2050 (United Nations, 2012). Since urbanized (i.e., built-up area) land represents less than 1% of the Earth's land surface (Liu et al., 2014; Zhou et al., 2015), efforts to increase resiliency, adaptive capacity, and habitability of cities will require substantial investments over small areas that will impact a disproportionately large fraction of society. This issue is particularly prominent for regions whose local relative share of global urban population continues to rise. For example, while Asia's share of global urban population is projected to peak around 2030, urban populations in Africa are expected to grow at increasingly rapid rates for many decades to come (Georgescu et al., 2015). Consequently, Africa will make up an increasingly larger fraction of total urban land (Güneralp & Seto, 2013). Such recognition of anticipated urban expansion has implications for prioritization of strategies guiding sustainable urban development (i.e., retrofitting relative to planning of future cities), which have largely focused on reduction of greenhouse gas emissions and on land-based solutions such as green and cool roofs (Georgescu et al., 2014; Sailor, 2008), increased vegetation fraction (Krayenhoff et al., 2014; Middel et al., 2015), and local engineering-based solutions (Meggers et al., 2016). In addition to potentially beneficial aspects of landscape configuration (Connors et al., 2013), deployment of natural capital within built environments may provide considerable additional human and environmental cobenefits.

Urban agriculture (UA) is a form of natural capital for growing food and other crops within cities (van Veenhuizen & Danso, 2007). Here we define natural capital to mean the biotic and abiotic components of a plant growth system, maintained in an area otherwise classified as urban (we exclude livestock, aquaculture and other secondary productivity). UA offers potential to ameliorate a host of urban environmental problems by increasing vegetation cover and therefore contributing to a decrease in the urban heat island (UHI) intensity (Susca et al., 2011), improving the livability of cities (Frumkin, 2003; Turner et al., 2004) and providing enhanced food security to over half of Earth's population (de Bon et al., 2009; Pearson et al., 2010). UA is connected to multiple metabolic pathways in the urban ecosystem including food provisioning (Zezza & Tasciotti, 2010), regulation of local microclimate and hydrology (Oberndorfer et al., 2007), consumption of nutrient rich "waste" water and biosolids/organic matter (Armstrong, 2009; de Zeeuw et al., 2011; Smit & Nasr, 1992), and fixation of atmospheric nitrogen (Herridge et al., 2008) and carbon (Beniston & Lal, 2012). For pollinators and other wildlife, habitat is created in the city (Goddard et al., 2010). It has been suggested that UA can also alleviate poverty (van Veenhuizen & Danso, 2007; Zezza & Tasciotti, 2010), increase resiliency (to market fluctuations and climate change) (de Zeeuw et al., 2011), serve as a repository of agricultural knowledge (Koohafkan & Altieri, 2010) and an incubator of new technologies (Despommier, 2010), provide measurable improvements to human health and wellbeing (Joye, 2007; Ulrich, 2006), and reunite urbanites with natural systems from which they have been separated (McClintock, 2010; Turner, 2011). There is evidence that UA shifts dietary intake toward more fresh fruits and vegetables (McCormack et al., 2010), which may reduce emissions from fossil fuels (Weber & Matthews, 2008) and nitrogenous waste (Sutton et al., 2011), while contributing to human nutrition and reducing the risk of multiple chronic diseases (Boeing et al., 2012). Conversely, UA may introduce disease and agricultural pollutants to the urban ecosystem (Smit et al., 2001), create conflicts over land use (Schmelzkopf, 1995), and add complicated, maintenance intensive systems to the urban infrastructure. Based on the economies of scale enjoyed by industrial agriculture, some have argued vigorously against the potential of UA to provide economic and environmental benefits (Desrochers & Shimizu, 2012). Meharg (2016) noted that UA will require monitoring to avoid consumption of food contaminated by urban pollutants.

The incorporation of agroecosystems into cities represents a significant paradigm shift in urban planning and design (Doron, 2005), and its success will require a sufficiently broad inquiry from the scientific community that addresses existing (e.g., environmental) concerns. Given the many and varied potential benefits of UA, it is largely absent from discussions of global food security (Foley et al., 2011; Godfray et al., 2010; Nature Editors, 2010; Tilman et al., 2002) and urban ecology (Grimm et al., 2008). Despite "a groundswell of interest" (Whitfield, 2009), little is known about the cumulative impact of broad UA adoption. In the United States and elsewhere, UA implementation is outstripping policy (Viljoen & Bohn, 2012). Recent reviews of UA in developing (Hamilton et al., 2013) and developed countries (Mok et al., 2013) still have not established a consistent, quantitative framework for analyzing the costs and benefits of UA at global scale, despite acknowledging that "the time is ripe for urban agriculture to be taken seriously and its potential contribution assessed rigorously" (Hamilton et al., 2013). Recent efforts (Badami & Ramankutty, 2015; Martellozzo et al., 2014) estimated the extent to which UA production of vegetables for urban dwellers is constrained by urban area, concluding that UA has minimal potential to alleviate food insecurity in developing countries. However, because these "nutritional footprint" approaches focus on constraints of urban area to meet dietary needs (100% devotion of urban area to agriculture is infeasible in any reasonable scenario), they may not appropriately assess UA potential multifunctionality (i.e., food production, ecosystem services, or economic returns) from realistic scenarios of UA adoption. A recent attempt has been made to estimate the amount of urban area under active cultivation (Thebo et al., 2014). Prior work therefore highlights the need to carry out a comprehensive examination that simultaneously incorporates multiple environmental and societal aspects, setting the basis for the data-driven quantitative approach used here.

It is important to emphasize that urban ecosystem services in general are not well quantified even at regional scales, let alone global. To our knowledge there exists only one global estimate for the value of ecosystem services (Costanza et al., 2014), and our paper adds to this burgeoning body of literature by extending this dialogue to the contribution made by UA. Here we endeavor to answer the following question: To what extent is UA a biological option for enhancing the urban infrastructure? Specifically, the purpose of this study is to quantify, at global scale:

1. Ecosystem services provided by existing urban vegetation.
2. Spatial availability for UA (and natural capital in general) in urban areas.
3. Ecosystem services provided when spaces identified in (1) are put to agricultural use.
4. Economic benefits derived from such services.

We estimate (1) directly from remotely sensed data. We used a data-driven approach for (2) in order to obtain a globally consistent estimate of building surface area and vacant lands from remotely sensed data and published studies. For (3) and (4), we used national scale datasets of agricultural productivity and prices combined with global-scale geographic datasets of 2010 urban area, population, and meteorological information. We aggregated and summarized the data in Google Earth Engine (hereafter Earth Engine), a cloud-based geospatial processing platform for analysis of planetary scale geospatial data (Gorelick et al., 2017).

This paper is not an estimate of actual UA production, though we do estimate services from existing vegetation, which may or may not be agriculture. Rather, our objective is to improve our understanding associated with the extent to which agroecological systems could enhance urban areas and address issues such as food insecurity, nitrogen pollution, flashy runoff, energy consumption and habitat (or lack thereof) for important species such as pollinators and predators of insect pests. We present a framework for global scale analysis of UA in cities at 500 or 1000-m resolution in Earth Engine and apply it to a scenario of development in which agroecosystems are fully incorporated to the urban infrastructure. Our intent is that these results inform the ongoing conversation regarding the role biological systems play in adapting urban areas to an uncertain future.

2. Materials and Methods

2.1. Estimation of Vegetation and Available Space from UA

We used a threshold on the Normalized Difference Vegetation Index (NDVI) computed from a cloud-free median composite of 3 years (2009–2011) of the Landsat 5 Collection 1 archive (30 m resolution). Specifically, we used a threshold of 0.3 to determine vegetation, a relatively conservative estimate based on previous research (Huang et al., 2016). We designate urban vegetation as any 30 m pixels with NDVI >0.3 that are within 500 m pixels classified as urban in the 2010 Moderate Resolution Imaging SpectroRadiometer (MODIS) land cover (MCD12Q1) (Friedl et al., 2002, 2010) dataset.

We estimated the space for natural capital in three mutually exclusive urban niches: vertical, rooftop, and vacant land. We specified these areas as follows.

2.1.1. Vertical and Facades

Semi-portable cultivation containers enable the installation of natural capital into vertical directions and growth environments unavailable otherwise (such as inside spaces in colder climates) and into the technological fabric of the city itself (e.g., through wastewater systems and indoor lighting). A simple apparatus consists of a 1-m^2 cultivation bin or bag filled with soil; a more complex apparatus is a hydroponic system with increased technological intensity and potential yield. We assume that UA in the vertical dimension is either appended to the outside of buildings or inside the building.

2.1.2. Rooftops

Although green roofs are typically composed of low maintenance species of mosses or succulents (e.g., *Sedum* spp.), growth of agricultural crops is constrained, but not precluded, by the rooftop environment. As stated in Oberndorfer et al. (2007), "In theory, almost any plant taxon could be used for green-roof applications, assuming it is suited to the climatic region, grown in appropriate substrate at an adequate depth, and given adequate irrigation … many possibilities have yet to be realized."

2.1.3. Vacant Land

We considered agriculture as an opportunistic use of vacant land, as defined by Pagano and Bowman (2000), in the matrix of residential, commercial, or other development.

We only estimate these spaces in areas classified as urban. Areas that are classified as agriculture for the purpose of land cover mapping at 500–1000 m scale are not included in this analysis, though may be defined as

UA for other purposes. Here we compared surface area within urban environments from two urban datasets at 500-m resolution. We used the MCD12Q1 dataset described previously. We also used the Landscan 2010 population dataset (http://www.ornl.gov/sci/landscan/) on 24-h ambient population per 30-arc sec pixel. The Landscan dataset contains structural typologies at 30-arc sec resolution (roughly 1 km). We used the "multistory structure urban" (MSSU) type, to ensure a vertical component to every urban pixel, as an alternative to MCD12Q1 urban area.

To estimate building area, we used Google's proprietary buildings database to establish an empirical relationship between building area (here we treat building roofs as the vertical projection of the building footprint) and the square root of Landscan population. We used areas of complete coverage of the building database in San Francisco, Rio de Janeiro, Hong Kong, Prague, London, Atlanta, Cairo, Lillehammer, and Nagano to compute building area per Landscan pixel over a diverse set of development typologies. From these data we computed the following linear relationship between building area per square meter and population per square meter ($n = 2856$ pixels, $R^2 = 0.678$):

$$B_{p(1)} = 2.170 \times population_{p(1)}^{0.5} + 0.015 \tag{1}$$

where $p(s)$ denotes a pixel at scale s, $B_{p(1)}$ is total building area per square meter pixel and $population_{p(1)}$ is population per square meter pixel.

UA is not feasible on all roof area. To estimate the fraction of rooftop area suitable for UA, we used data for green roofs and photovoltaics, which share similar requirements. Specifically, the roof must be accessible, well illuminated, relatively flat, capable of load bearing as necessary and may need access to water resources depending on climate. Wiginton et al. (2010) estimates 19% of rooftop area in Ontario to be suitable for photovoltaics in terms of slope, illumination and infrastructure. Izquierdo et al. (2008) estimated that 19.5% of total rooftop area is suitable for photovoltaic installation in Spain. Green roof suitability was estimated at 25% rooftop area for Athens, Georgia, United States (Carter & Keeler, 2008). MacRae et al. (2010) report that 37% of roofs in Toronto, Canada are "suitable for greening of some form." Here we adopt a conservative 19% suitability of estimated rooftop area for UA (i.e., the lowest value retrieved from the peer-reviewed literature). This results in an estimate roof area per pixel at scale s as.

$$B_{p(s)} = 0.19 B_{p(1)} A_{p(s)} \tag{2}$$

where $A_{p(s)}$ denotes pixel area at scale s.

To estimate building façade area, we required a globally consistent estimation of building heights (which did not previously exist). We used morphological operations on the AW3D30 30-m resolution digital surface model (DSM) provided by Japan Aerospace Exploration Agency (JAXA) (n.d.) to obtain height estimates (Marconcini et al., 2014). We compared estimated heights to areas of complete building height coverage in the Google database for San Francisco and London, and to the Philadelphia buildings database (OpenDataPhilly, n.d.). Estimated heights at 30 m resolution were averaged to obtain mean height in each 1-km Landscan pixel. We compared the aggregated estimates to the mean height of buildings in each 1-km Landscan pixel. The accuracy assessment indicated an acceptable linear relationship between our estimate ($\hat{h}$) and reported heights ($h = 1.30\hat{h} * +5.83$, $n = 470$ pixels, $R^2 = 0.416$). Note that this is a conservative estimate of actual building height. We used our estimated heights at 30-m resolution, aggregated to the mean in each 1-km Landscan pixel as our estimate of building height ($\hat{h}$).

Mean height of buildings and total building area per 1-km pixel are insufficient to determine façade area without an estimate of the distribution of building sizes. Observing the size distribution of buildings in the set of cities used for area estimation, we noted a power law distribution of building area, a scaling relationship reported previously (Batty et al., 2008). Therefore, we modeled building area according to the following distribution:

$$P(B) = CB^{-\alpha}, \tag{3}$$

where C is a normalizing constant and B is area of an individual building. We fit the parameters of the distribution according to the method in Newman (2004), and noted small variation in α across geographic regions ($1.18 < \alpha < 1.23$). Based on these fits, we set $\alpha = 1.22$. The distribution is bounded above by total building area in a pixel (i.e., individual building size cannot exceed total building area), enabling computation of expected individual building area in a pixel as:

$$\mathrm{E}[B] = C\int_{1}^{B_{p(1000)}} BB^{-\alpha}dB \tag{4}$$

where $B_{p(1000)} = B_{p(1)}A_{p(1000)}$ is the total building area in a 1-km pixel (with area $A_{p(1000)}$) and E is the expectation operator. We assume that a 1-km pixel is populated by square, mean sized buildings as computed in equation 4, with height $\hat{h}$ estimated as described previously. A square building has four sides of surface area $\hat{h} * \mathrm{E}\left[B^{0.5}\right]$, the expectation of the square root building area computed as:

$$\mathrm{E}\left[B^{0.5}\right] = C\int_{1}^{B_{p(1000)}} B^{0.5}B^{-\alpha}dB \tag{5}$$

We modeled $n_{p(1000)} = A_{b(1000)}/\mathrm{E}[B]$ to obtain mean-sized buildings in a 1-km pixel, for a total building façade area of $n_{p(1000)}4\hat{h}\mathrm{E}\left[B^{0.5}\right]$ per 1-km pixel. At maximum, two sides of a building can receive direct illumination at a given time. We assume one side has sufficient illumination (i.e., one out of four sides, or 25% illumination, is used as a conservative estimate) and that only 10% of buildings are suitable for development in this manner, yielding $0.1n_{p(1000)}\hat{h}\mathrm{E}\left[B^{0.5}\right]$ of vertical area per Landscan pixel.

Vacant land is not possible to estimate from remotely sensed data alone and required an extensive review of the literature to obtain internationally representative values. A survey of 70 US cities found an average of 15.4% vacant land in urban areas, with responses ranging from 0.6% to 45% (Pagano & Bowman, 2000). Specifically for UA, 6.5% of Cleveland, OH, United States (Grewal & Grewal, 2012), 5.3% of Detroit, MI, United States (Colasanti, 2010), 2.5% of New York City, United States (Ackerman, 2012), 2.4% of Oakland, CA, United States (McClintock et al., 2013), and 42.6% of Phoenix, AZ, United States (Kittrell, 2012) were estimated to be suitable vacant land. More recently, Smith et al. (2017) developed a systematic approach applied to the Phoenix metropolitan area that combines high-resolution remote sensing and cadastral data to identify vacant parcels of land. They conclude that 6.8% of available urban land is suitable for greening through deployment of UA for the Phoenix metropolitan area. International vacant land availability has been reported as highly variable, ranging from 2% in Santiago, Chile (Morandé et al., 2010), to 21% for Atibaia, Brazil (Sperandelli et al., 2013), to as much as 35% for Herat, Afghanistan (French et al., 2016). Based on the aforementioned literature, we constrain vacant land by bracketing our calculations using a minimum constraining value and a maximum value.

We used a linear function (based on a least median squares regression computed from the data in Pagano and Bowman (2000)), evaluated for each pixel, which predicts the US average rate of vacant land, reduced by population density, and we utilize this as the maximum vacant land availability:

$$V_{p(s)} = 0.154A_{p(s)} - 0.0007^{*}\ population_{p(s)} \tag{6}$$

where $V_{p(s)}$ is vacant area per pixel at scale s.

Following the above, we examine the sensitivity to this specification and repeat this calculation using vacant land availability of 1%, as a minimum constraining value:

$$V_{p(s)} = 0.01A_{p(s)} \tag{7}$$

2.2. Quantification of Agricultural Production

In small-scale UA, crop choice can and should respond to local climatic conditions, price of inputs, market fluctuations, and consumer preferences or needs. These choices will be based in part on culture, availability, familiarity, and local knowledge. Given this variability, we assume that the mix of urban crops is drawn from the list of crops produced nationally (as reported by the FAO), with important differences in crop proportion due to constraints on growth media, spatial and temporal scale of operations. Of all the crops in production globally, as reported by FAO for 2010 (http://faostat.fao.org), we ranked crop suitability for UA on a 0–3 scale: 0 = unsuitable, 3 = most suitable. We considered horticultural crops to be most suitable, with greatest emphasis placed on high-yielding, easily grown, nutrient dense (kcal, protein, vitamins) annual fruits and vegetables (de Bon et al., 2009). Specifically, we used the following general ranking:

10.1002/2017EF000536

- Rank 3 (most suitable). Pulses (*Fabaceae*), root crops (e.g., carrots, beets, turnips, etc.) and tubers (e.g., yams, sweet potatoes, cassava and *Solanum* potatoes, etc.), and vegetables such as okra, brassicas (*Brassicaceae*), spinach and squash.
- Rank 2 (medium suitability). More difficult to grow or process grains (*Poaceae*), small-stature perennial fruiting plants, vines and shrubs, and less nutrient dense vegetables such as lettuce, eggplant, tomatoes alliums and peppers.
- Rank 1 (least suitable). Large-stature perennial vegetables (e.g., artichokes), small stature (i.e., "dwarf") perennial fruiting trees (e.g., apples, pears, citrus, banana, etc.), process-intensive seed crops (e.g., sunflower, sesame and safflower), and seasoning and nonfood consumables such as tea, spices, ginger and sugar crops/sweeteners.

Due to constraints on available area, growth media, rooting depth, processing requirements, and land tenure, we consider the following crops to be unsuitable for UA and unlikely to be cultivated (ranked zero): large-stature and long-lived perennial tree crops (e.g., almonds, pistachio, walnuts, carob, dates, avocado, mango, etc.), oil (e.g., olive, oil palm) and fiber crops (e.g., jute, cotton), animal fodder, and biofuels (i.e., energy crops such as *Miscanthus* and switchgrass). We assigned growing space proportionally to rank, with unsuitable crops not being allocated any growing space.

Yield (hectograms per hectare) and annual producer price (2010 USD) are also reported by FAO for most countries and most crops. We estimated aggregate UA crop production at country level as:

$$\textit{Annual production} = \sum_{crops} \sum_{i} A_i^* \, weight_{crop}^* \, yield_{crop} \tag{8}$$

where A_i is the area per pixel of subspace i (e.g., rooftop) as described previously, *weight* is derived from the rank of all crops reported for a country:

$$weight_{crop} = rank_{crop} \Big/ \sum_{crops} rank \tag{9}$$

and $yield_{crop}$ is a country-crop specific value from the 2010 FAO data.

Within each country, the set of crops with reported yield may not equal the set with reported producer price. We used the average of all producer price data for 2009–2011 to increase the number of crops with reported price in each country. For country-crop combinations with no reported producer prices in 2009–2011, we used the 2009–2011 global mean. Approximately 25% of the total economic value of UA crops was estimated in this manner. The economic return is simply annual producer price multiplied by annual production.

We estimated partial input costs according to country level use (kg/ha) of agricultural fertilizers as reported by World Bank (2014a, 2014b). We assumed these inputs are applied to all crops, uniformly to all areas estimated as above, as a mix of nitrogen, phosphorus and potassium fertilizers in the form of urea, diammonium phosphate and muriate of potash (potassium chloride), respectively. We estimated the cost of urea from the 2010 global average commodity price and the latter two inputs obtained from FAO (2012). It is important to acknowledge that fertilizer application may be higher, and more variable, in cities in part because dwellers within urban environments have a reduced understanding of food system processes, including appropriate application of fertilization (Metson & Bennett, 2015b).

2.3. Other Ecosystem Services

We quantified ecosystem services other than food provisioning in the following categories: nitrogen fixation (of the legumes), pollination, biological control (of pests), avoided storm water runoff, and energy conservation (resulting from the insulation properties of rooftop UA substrate).

We estimated nitrogen fixation from global average fixation rates (kilograms/hectare/year) (Herridge et al., 2008) in seven categories of legume linked to the FAO crop tables (Table S1). Here we value nitrogen as a component of the agroecosystem, when it is fixed by *Rhizobium* spp. Nitrogen in the soil or root is an avoided fertilization cost, therefore we used 2010 average global commodity price of urea (288.59 USD/tonne) as the value of atmospheric nitrogen fixed during agricultural production.

Because UA may use nutrient inputs (and in cases of nonrain fed situations, water input aimed to increase yield) there is the potential to actually augment, rather than lower, runoff. Nevertheless, previous studies have demonstrated rainfall retention characteristics of green roofs (Mentens et al., 2006; Simmons et al., 2008; VanWoert et al., 2005). For example, VanWoert et al. (2005) showed that for a 10 cm thick substrate and a heavy rainfall event, defined as exceeding 6 mm rainfall per day, 54% of precipitation is retained by the growth medium. To compute the change in runoff, we used the Natural Resource Conservation Service (NRCS) method for urban hydrology (NRCS, 1986). Specifically, we estimated runoff from an "Open space, Good condition, grass cover > 75%" in the least pervious soil hydrologic group according to (NRCS, 1986):

$$Runoff = A_i S \frac{\left[R - 0.2\left(\frac{1000}{c} - 10\right)\right]^2}{R + 0.8\left(\frac{1000}{c} - 10\right)} = A_i S \frac{(R-0.5)^2}{R+2}, \text{for } c = 80 \tag{10}$$

where A_i is area in subspace i, S is a conversion factor between inches and meters, R is rainfall in inches and $c = 80$ is an index corresponding to the open space surface described previously (minimum c is 39 for urban surfaces). For R, we used 2010 PERSIANN-CDR (Ashouri et al., 2015) daily rainfall data, at 0.25° spatial resolution, available between −60° and 60° latitude. We estimated avoided storm water runoff in urban areas as the difference between PERSIANN rainfall on an impervious surface (i.e., $c = 100$) and runoff estimated for open space as described previously. We estimated total annual avoided runoff as the sum of daily avoided runoff in 2010 for roof and vacant areas with PERSIANN coverage.

We use estimated values for thermal resistance (R in m^2 K/W) to compute potential energy savings resulting from the insulative characteristics of rooftop UA substrate. For UA, we assume a 10 cm rooftop substrate with relatively high moisture content and thermal resistance (R_0) of $R_0 = 3.66$, a representative value for well-insulated conventional roofs (Clark et al., 2008; Niachou et al., 2001; Vacek et al., 2017; Van Hooff et al., 2014). We also performed a sensitivity analysis, assuming $R_0 = 0.25$ for poorly insulated roofs (Castleton et al., 2010). Assuming indoor temperatures are maintained between 18°C to 20°C, we computed avoided heating or cooling loads from the temperature gradient (in units of Kelvins) between indoor and outdoor air temperature (ΔT) and the added thermal resistance from R_a. We computed heating or cooling loads (energy in Watts) as:

$$Flux = B_{p(s)} \Delta T / R \tag{11}$$

where $B_{p(s)}$ is computed according to equation 2 for each urban pixel. ΔT was computed from 2010 Global Land Data Assimilation System 2.0 (GLDAS-2) climatologically consistent surface temperatures (Rodell et al., 2004), available at 3-hourly temporal frequency and 0.25° spatial resolution. Since thermal resistance is additive, we computed avoided heating or cooling energy consumption resulting from adding an R_a layer to an existing R_0 roof as:

$$Difference\ in\ energy\ flux = B_{p(s)} \Delta T / R_0 - B_{p(s)} \Delta T / \left(R_0 + R_a\right) = B_{p(s)} \Delta T^{*}\ R_a / \left(R_0\left(R_0 + R_a\right)\right) \tag{12}$$

To place an opportunity cost on the avoided energy consumption, we assume the marginal change in flux is compensated with electrical heating or cooling. Electricity prices vary widely at global scale (OVO Energy, n.d.). Here we use the 2010 USA price of 0.10 USD/kWh (US Energy Information Administration, n.d.), among the lowest energy prices in the world.

Costanza et al. (2014) reported per hectare values of various ecosystem services from agriculture. We used these values, adjusted to 2010 USD, to calculate additional ecosystem services. Specifically, we use \$23.87 USD/ha for pollination services, \$35.80 USD/ha for biocontrol services, \$445.88 USD/ha for climate regulation services and \$577.15 USD/ha for soil formation. In the urban context, we consider climate regulation important in terms of UHI abatement and soil formation important to creating a growth medium for supporting any urban vegetation, including ornamentals, lawns or street trees.

The analysis was implemented in Google Earth Engine (Gorelick et al., 2017), a cloud based geospatial processing platform. The code and results that correspond to the analysis are available to registered Earth Engine users at https://goo.gl/YyfsWB.

Table 1.
Estimated Area (ha) Available for Natural Capital, Comparing Urban Masks from Landscan 2010 "Multi-Story Structure Urban" Type and MODIS MCD12Q1 Urban Type

	Landscan	MCD12Q1
Vertical	30,000	70,000
Rooftops	1,370,000	1,340,000
Vacant land	5,660,000	9,860,000
Total	7,060,000	11,280,000

Note. Rounded by significant figures. MODIS MCD12Q1 = 2010 Moderate Resolution Imaging SpectroRadiometer land cover.

3. Results

Global urban area availability, according to MCD12Q1, is approximately 641,000 km^2, and is regarded as an intermediate estimate of urban "extent" (Potere & Schneider, 2009). Landscan MSSU area is approximately 367,000 square kilometers, at the low end of global estimates of urban extent.

Table 1 shows global aggregate space for natural capital estimated in each category, for both urban area datasets. The vertical category is the smallest space, with 30–70 thousand hectares available globally. Rooftop area is estimated to be larger, with approximately 1.3 million hectares. Vacant land is by far the largest category, and out of approximately 7–11 million hectares of total available land for natural capital, represents between 80% and 87% of the total area available for UA. In aggregate, we estimate the available space for UA as approximately 1–7 million hectares or 1.4%–11% of the urban area we considered in this study. We estimate 31 million hectares of urban vegetation in 2010 or approximately 48% of the MODIS urban area, roughly consistent with (though slightly higher than) previously published estimates (Dobbs et al., 2014).

Figure 1 shows available space by country using the MCD12Q1 urban mask. With over 2 million hectares, the United States is estimated to have more urban area available for natural capital than any other country. China is second with just over 1.4 million hectares. Other relatively urbanized countries with extensive areas for UA include Brazil, India, Russia, Germany and Japan.

Table 2 shows ranges of potential food production, energy savings from reduced heating and cooling costs, nitrogen sequestration from leguminous plants, and avoided storm water reduction from increased rainfall interception. On an aggregate level, natural capital provides a spectrum of benefits.

Millions of tonnes of food, modest sequestrations of nitrogen, billions of kilowatt hours of energy savings and billions of cubic meters of storm-water avoidance are estimated. For comparison, our results indicate approximately 164 billion cubic meters of stormwater avoidance is provided by existing vegetation in urban areas (i.e., nearly three times as much as the UA amount estimated using MODIS (5.7141×10^{10} m^3; Table 2), which is equivalent to 34% of the volume of Lake Erie (United States Environmental Protection Agency, 2012). In addition, assuming that existing urban vegetation is fully converted to UA, we estimate roughly 500 million tonnes of food could be produced on an annual basis (i.e., nearly three times as much as the UA amount estimated using MODIS (1.7831×10^8 tonnes; Table 2). Finally, to put the energy savings into perspective, the estimates represent 10% of the 2014 electricity consumption of Argentina (CIA, 2014).

The distribution of benefits varies geographically. For example, Figure 2 shows the high and low estimate of UA yield by country. We estimate that in the United States, 40 million tonnes of annual production are possible. China is second with 20 million tonnes, followed by Germany and Brazil. As shown in Figure 2, countries with large urban areas and crop mixtures suitable for both the climate and urban cultivation could produce millions of tonnes from UA.

The United States and China could both experience about 2.4 billion kWh of energy savings by installing natural capital on roofs. Figure 3 shows that Russia and India also stand to experience over 1 billion kWh in savings resulting from enhanced rooftop insulation by growth substrate. We characterize this as a conservative

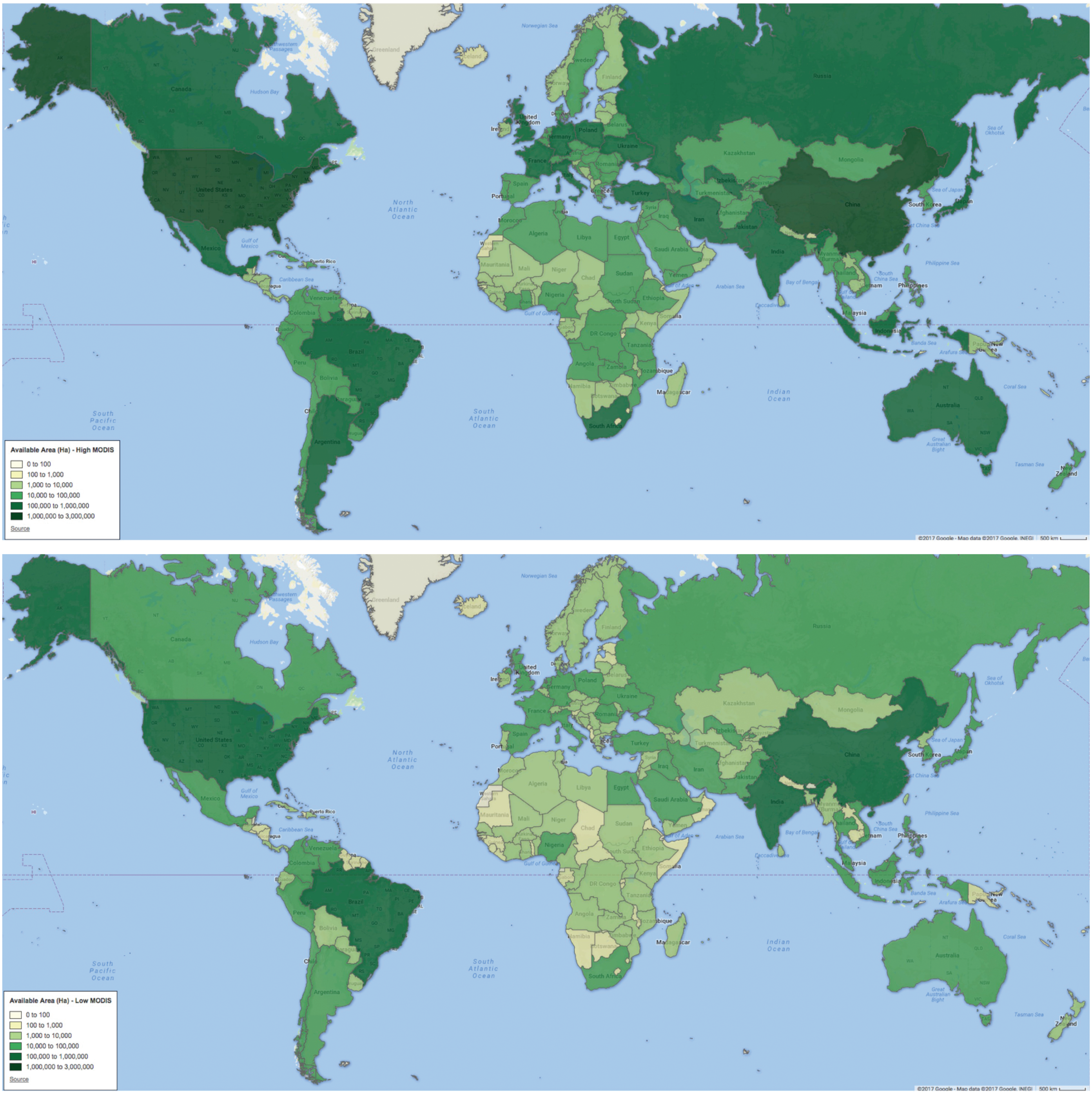

Figure 1. Available area (ha), by country, using the MCD12Q1 urban mask. The top panel, designated as "High MODIS", is based on a maximum vacant land availability as specified by Equation 6. The bottom panel, designated as "Low MODIS" is based on a minimum constraining value of vacant land availability as specified by Equation 7. See Section 2.1 for details. Figure 1 online.

estimate since it does not include additional likely savings associated with biophysical effects from added vegetation (Georgescu et al., 2015; Li et al., 2014).

Stormwater avoidance (Figure 4) shows a different pattern, with countries that experience consistent or high rainfall estimated to benefit more. Though we estimate the United States has the highest storm water remediation potential, at 3.7 billion cubic meters annually, monsoon affected countries such as China,

Table 2.
Estimates of Services Provided by Urban Agriculture (UA) in Physical Units

	Landscan	MCD12Q1
Food production (tonnes)	98,000,000	178,310,000
Energy savings (kWh)	13,977,000,000	15,141,000,000
Nitrogen sequestration (tonnes)	110,000	170,000
Stormwater reduction (cubic meter)	45,469,000,000	57,141,000,000

Note. Rounded by significant figures.

Japan, India, Brazil, and Indonesia all are estimated to benefit significantly, with over 1 billion cubic meters of avoidance annually.

Aggregate ecosystem service values are shown in Table 3. These estimates include economic benefits from agricultural production, energy savings, pollination, biocontrol, climate regulation, soil formation and nitrogen fixation. Of the ecosystem services evaluated, food production provides the largest fraction of returns, though the results also indicate substantial benefits from climate regulation, soil formation and energy savings. The results indicate that existing vegetation provides roughly $33 billion of services from biocontrol, pollination, climate regulation and soil formation.

The mixture of benefits by country is a function of urban area, space within urban area (vertical, rooftop or vacant), national agricultural output (productivity, price and combination of crops suitable for UA) and climate (temperature and precipitation patterns). Figure 5 shows the distribution of ecosystem service value, by country, corresponding to the MCD12Q1 urban mask. We estimate that China, Japan, Germany, and the United States have a suitable combination of factors to be consistently among the top beneficiaries of UA in terms of estimated dollar potential and associated ecosystem services. Relatively urbanized countries such as the UK, Brazil and Netherlands (though Netherlands is in this category, we suspect anomalously high reported prices for berries and spices in the FAO data) also indicate potentially high value of ecosystem services from UA.

It is essential to place the suggested benefits of UA in broader context. For example, as mentioned previously, we note that the estimated storm water avoidance (i.e., reduction in urban stormwater runoff; see Table 2) is nearly a third of the volume of Lake Erie, the 11th largest freshwater body on Earth (by surface area). There are no global (or regional) measurements of *actual* urban stormwater runoff (volumes) to compare our estimates to, aside from notional estimates based on varying levels (peak, for instance) of surface imperviousness. In urban catchments, runoff volumes are usually estimated via hydrological (or statistical) models that simulate (or estimate) relationships between rainfall and coefficients for streamflow and runoff for different surfaces. Modeled estimates suggest urban stormwater runoff can be substantial. For example, (Walsh et al., 2012) estimate that the urban stormwater runoff in Melbourne, Australia ranges from 57% to over 140% of the city's total water demand (variation is due to performance of different catchments in the city). It has been estimated that urban stormwater runoff could be as much in volume as the total incoming water supply of an urban water system (Mitchell et al., 2003). Based on estimates from Aquastat (FAO 1, 2016), worldwide municipal water withdrawals exceed 466 billion cubic meters annually. This means our avoided runoff estimates are substantial, at approximately 10%–12% of this municipal (i.e., urban) water use.

To put the energy savings into perspective, we may consider energy use by commercial and residential sectors mainly for the lighting, heating and cooling of buildings. This is the largest electricity end-use category, and a growing one, particularly in the developing world (with rising urbanization). This is also the end-use sector where the energy saving benefits of UA would be most directly observed. While total energy savings from UA would be less than 1% of worldwide electricity use in the residential and commercial sectors (based on figures for 2010) (EIA, 2017), it represents about 1.2% of electricity use in commercial buildings in the U.S. (based on figures for 2012) (EIA, 2016). The energy savings we estimate from UA is also equivalent to displacing the annual electricity use by air conditioners in nearly 9 million U.S. households (EIA, 2009).

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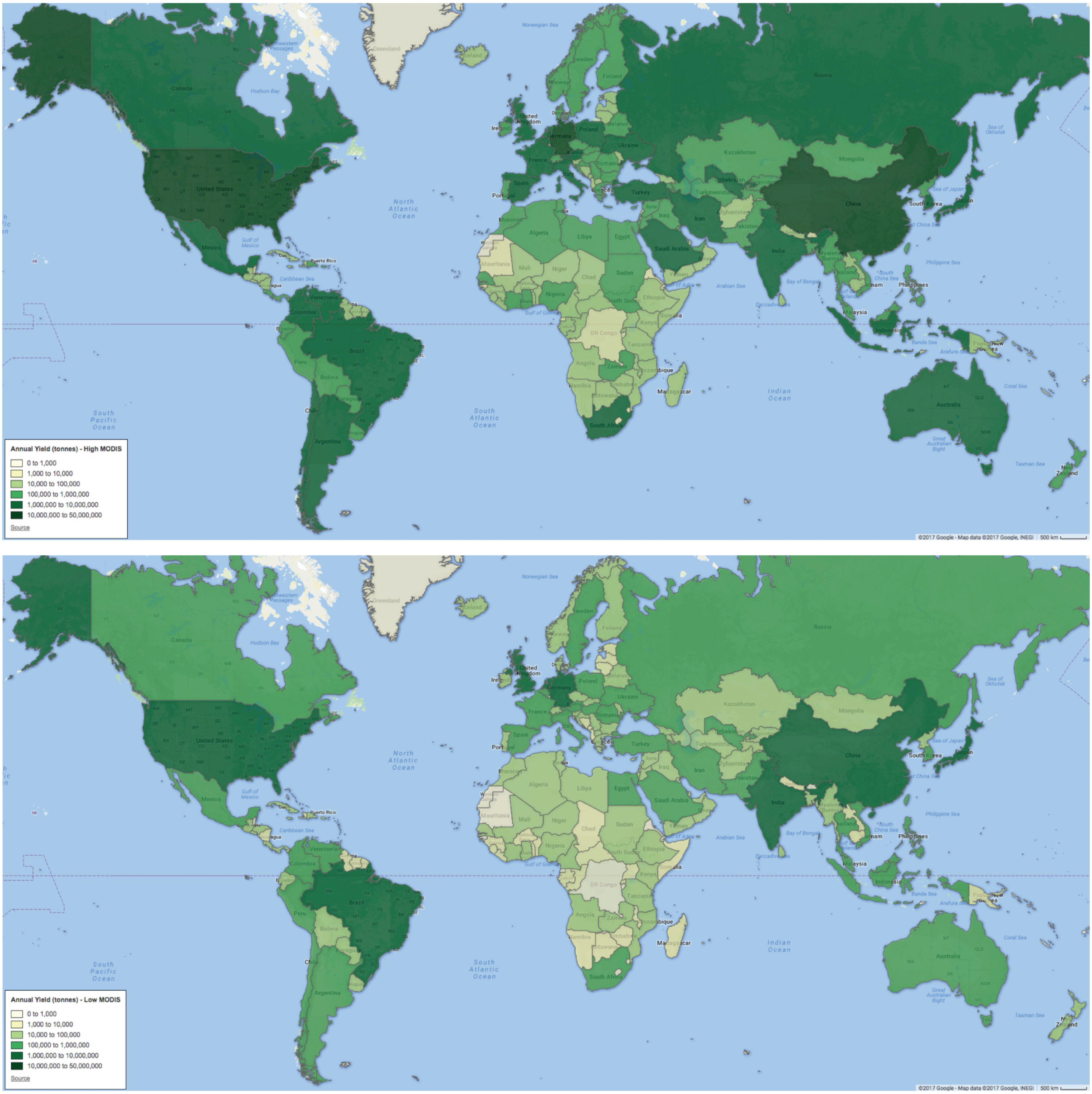

Figure 2. Potential yield (tonnes) from UA, by country, based on the MCD12Q1 urban mask. The top panel, designated as "High MODIS", is based on a maximum vacant land availability as specified by Equation 6. The bottom panel, designated as "Low MODIS" is based on a minimum constraining value of vacant land availability as specified by Equation 7. See Section 2.1 for details. Figure 2 online.

We also quantify the amount of nitrogen sequestration from UA to range from 110 to 170 thousand tonnes. According to figures from the FAOSTAT (FAO 2, 2016) database, the consumption of total nitrogen fertilizers worldwide has averaged about 105,488 thousand tonnes between 2010 and 2014. The sequestration benefits we estimate from UA correspond to less than 0.2% of world nitrogen fertilizer use for agriculture.

Our estimates of a 98–178.3 million tonnes per year annual food production from UA can be placed into context by referring to the FAOSTAT (FAO 2) database for a comparison with the global production of crops

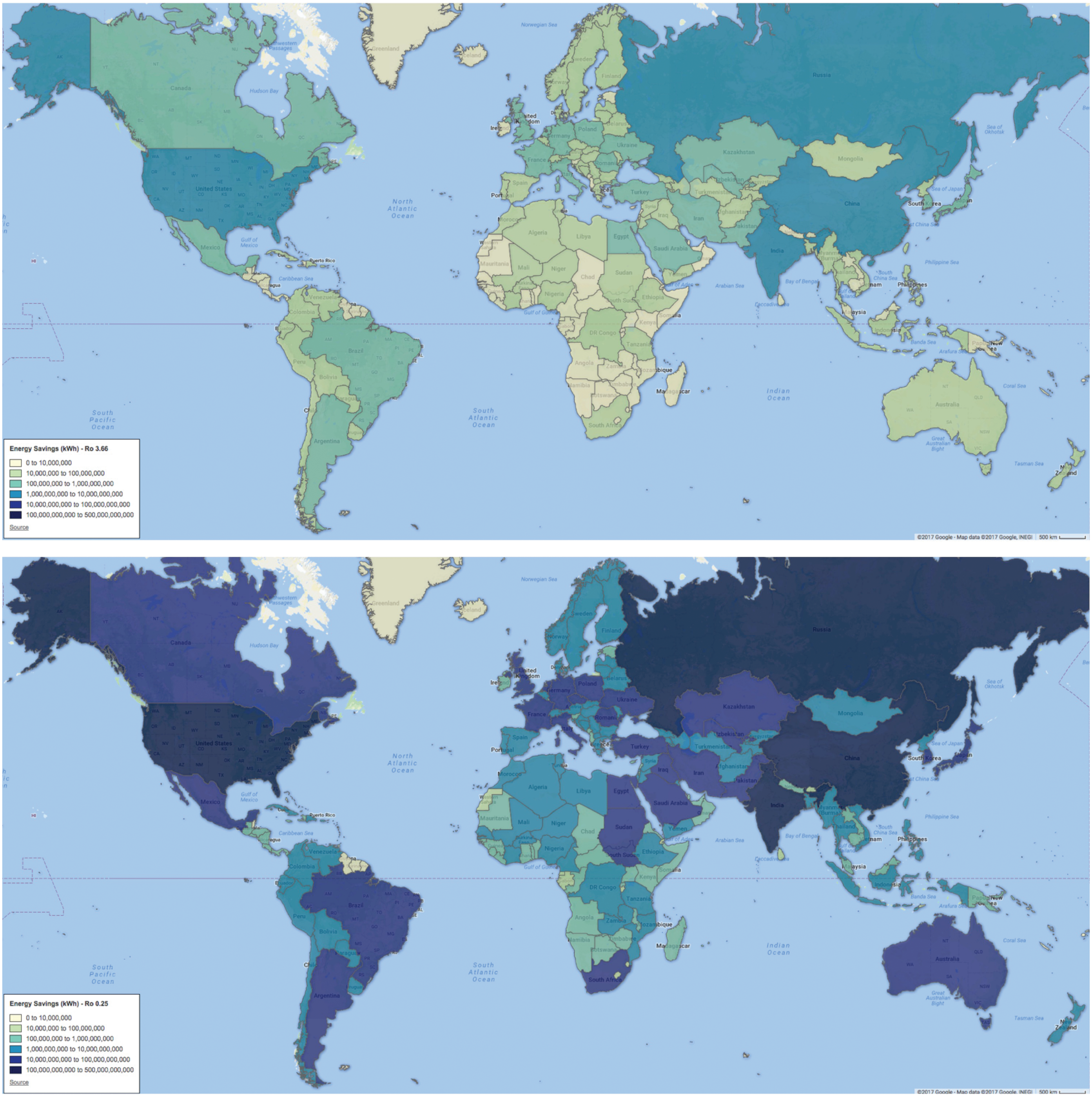

Figure 3. Energy savings (kilowatt hours) from rooftop substrate, by country, based on the MCD12Q1 urban mask. The top panel, designated as "Ro 3.66", is based on a Ro value of 3.66 (m^2 K/W), representative of well-insulated buildings. The bottom panel, designated as "Ro 0.25", is based on a Ro value of 0.25 (m^2 K/W), representative of poorly insulated buildings. See Section 2.3 for details. Figure 3 online.

(recall that we exclude livestock). According to the FAO, in 2010, worldwide production of all crops (including fiber crops) was about 6433 million tonnes. Nearly 30% of that amount, or 1875 million tonnes, involved the production of pulses, roots and tubers, and vegetables that are suitable for UA. The estimated output from UA is about 5%–10% of this global production of pulses, roots and tubers, and vegetables.

Our estimated benefits (value) of food production from UA ranges from \$78.6 to \$150.1 million (in 2010 U.S. dollars) (Table 3). According to FAO 2, the value of total agricultural production (which includes livestock and

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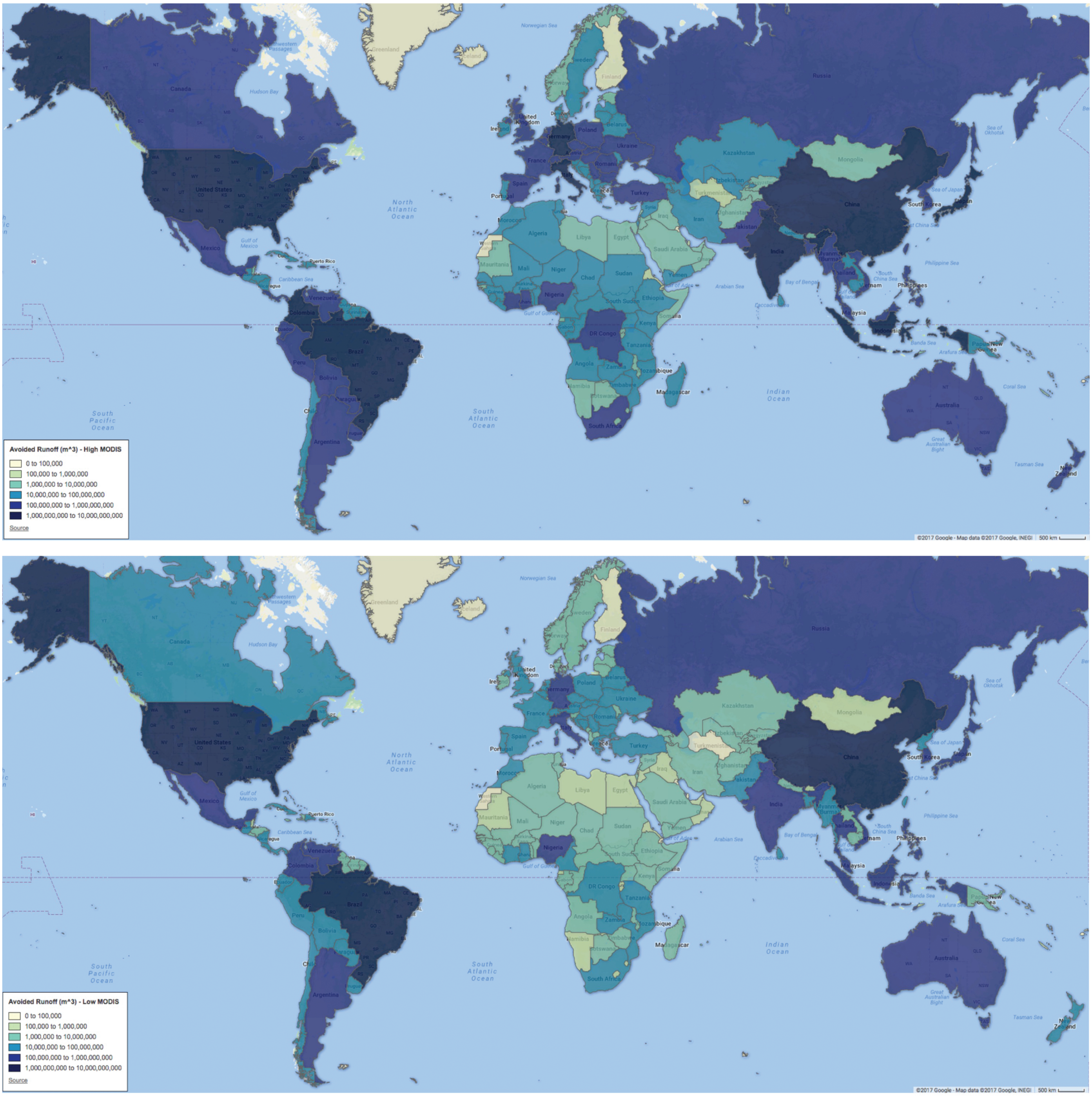

Figure 4. Avoided runoff (cubic meters) from UA, by country, based on the MCD12Q1 urban mask. The top panel, designated as "High MODIS", is based on a maximum vacant land availability as specified by Equation 6. The bottom panel, designated as "Low MODIS" is based on a minimum constraining value of vacant land availability as specified by Equation 7. See Section 2.3 for details. Figure 4 online.

nonfood crops) in 2010 was $2578 billion. The value of crop production, including cereals, was $2157 billion; and excluding cereals, it was $1509 billion. This last figure is the most relevant benchmark to compare our estimates with: therefore, the benefits we estimate from UA food production represent 5%–10% of world noncereal crop production (in 2010 U.S. dollars).

Finally, let us place the entire set of ecosystem benefits we estimate from UA into a broader context. We estimate these benefits to range from nearly $88 billion to $164 billion (in 2010 U.S. dollars; Table 3). The

Table 3.
Value of Ecosystem Services Provided by Urban Agriculture (UA) Assuming Full Implementation Compared to Existing Vegetation, in 2010 USD

	Landscan	MCD12Q1	Existing vegetation
Food production	$78,600,000,000	$150,100,000,000	$440,000,000,000
Nitrogen sequestration	$31,500,000	$49,700,000	
Energy savings	$1,400,000,000	$1,510,000,000	
Biocontrol	$253,000,000	$404,000,000	$1,120,000,000
Pollination	$169,000,000	$269,000,000	$750,000,000
Climate regulation	$3,150,000,000	$5,030,000,000	$13,930,000,000
Soil formation	$4,080,000,000	$6,510,000,000	$18,030,000,000
Total	$87,700,000,000	$163,900,000,000	$33,830,000,000[a]

Note. Rounded by significant figures.
[a]The total ecosystem services provided by existing vegetation does not include hypothetical food production.

world GDP for 2010 was $65,906 billion (The World Bank, 2017). The total ecosystem benefits we estimate from UA represent 0.1%–0.2% of this measure of global economic output.

4. Discussion

This paper serves as an initial attempt to examine potential gains associated with conversion of existing urban spaces to multifunctional green spaces. We do not perform an actual accounting of UA, which is notoriously difficult due to the complexity of distinguishing agriculture from ornamental plants, street trees, or other urban vegetation with a phenology that differs from the natural environment. Nevertheless, recent results (d'Amour et al., 2017) report that 36% of urban areas were used for crop production in 2000. While this value is a likely overestimate given the probable inclusion of peri-urban areas, these results highlight the already critical importance of crop growth within a reasonable distance of existing built environments. To better inform dialogue on this topic and to promote rapid advances that can utilize the framework developed here, the code we provide as part of our data-driven approach more easily enables such investigations when input data are available, allowing for inventory or simulation-based planning.

Our results suggest that $80–160 billion (2010 USD) of ecosystem services could be realized annually, depending primarily on the extent to which local governments interpret lands to be vacant and/or available, to a lesser extent on the amount of suitable rooftop space. Table 3 shows that this is a relatively small number compared to what is attainable from a hypothetical conversion of existing vegetation to agriculture. However, existing vegetation consists of parks, street trees, lawns and other private spaces that are not likely to be suitable for UA, so the actual amount of vegetation available for conversion (and corresponding UA returns) is likely to be substantially less.

These results place often-repeated anecdotal estimates of UA contributions into perspective. For example, Zezza and Tasciotti (2010) trace the entrenchment of the suppositions that 800 million urban residents are "actively engaged urban farmers," or that 15–20% of the world's food supply comes from UA (van Veenhuizen & Danso, 2007). Even our most optimistic estimates of area and yield cast doubt on the claimed 15% of global production, though it is possible the "urban" definition that led to the 15%–20% estimate is substantially more broad than ours and includes periurban areas and livestock operations as well. With maximal utilization of space and intensive production practices, we estimate that UA could produce ~5% of FAO reported production of the same crops globally. With more conservative estimates of space and productivity, we estimate that UA results in 1% or less of global production of the same crops.

In some climates, irrigation infrastructure will be needed to support UA. However, water inputs to UA are fundamentally different from those in large-scale agriculture that has displaced natural ecosystems. Existing urban water metabolism has pathways that can be exploited for use as UA inputs (Rojas-Valencia et al., 2011; Zhang et al., 2010). Perversely, the "waste" water resource, containing nutrients essential for agriculture (Armstrong, 2009; de Zeeuw et al., 2011; Smit & Nasr, 1992), is treated as garbage, the disposal of which

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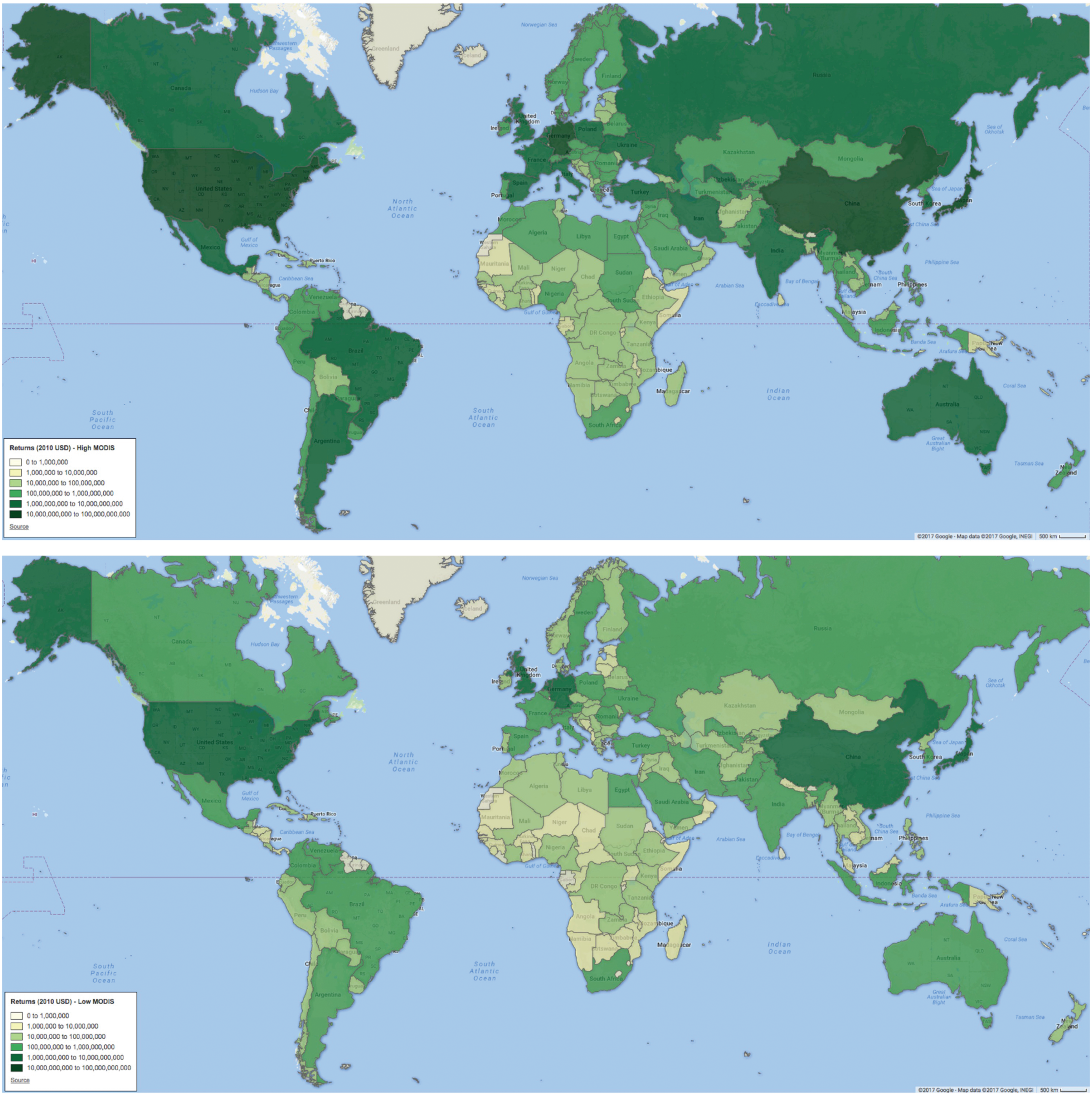

Figure 5. Value (2010 USD) of ecosystem services from UA, by country, based on the MCD12Q1 urban mask. The top panel, designated as "High MODIS", is based on a maximum vacant land availability as specified by Equation 6. The bottom panel, designated as "Low MODIS" is based on a minimum constraining value of vacant land availability as specified by Equation 7. See Section 2.3 for details. Figure 5 online.

is often subsidized (Rogers et al., 2002). Properly treated, nonindustrial wastewater is safe and effective for both irrigation and fertilization of UA (Rojas-Valencia et al., 2011). Therefore, we view the cost of inputs to UA as a question of necessary technology and infrastructure for recycling rather than demand for scarce resources (irrigation water). Other infrastructural costs could be minimized, for example with readily available, lightweight, portable planting bags which are possible to install almost anywhere without special equipment. Costs to modify existing infrastructure as necessary are unknown and their estimation beyond the scope of this effort, but an excellent topic for future research.

Taking costs associated with rooftop development of UA as an example, it is possible to identify break-even points for a better understanding of incentives. For example, we estimated possible returns from rooftop UA of around $20 billion annually. If infrastructural changes of $10 per square meter are necessary, this represents an approximately $134 billion investment. Assuming a guiding rate of 8%, and a $20 billion annual return on investment, the break-even point occurs in 10 years. While this is an interesting scenario, it is not sufficient to characterize private sector decision boundaries. That is because many ecosystem service revenues are public benefit returns that are realized on an aggregate societal level and many private sector returns (e.g., real estate values, roof longevity, rental income, etc.) are unknown and not accounted for here. A comprehensive present net value analysis is therefore beyond our scope.

In addition to input costs and fixed costs associated with infrastructural development, there are risks associated with UA that could impose additional economic constraints or limit the food provisioning service (Smit et al., 2001). The urban environment may contain air pollutants that adversely affects or limits growth, quality and pest resistance of UA (Bell et al., 2011). Soil pollutants pose a potentially more serious problem since they not only affect plant growth, but can also be absorbed by plants and lead to health hazards in the ultimate consumers (Beniston & Lal, 2012; Meharg, 2016). Other health risks may arise from the exposure to agrichemicals, use of untreated animal fertilizers or wastewater, or the transmission of disease from crops to other urban inhabitants not associated with agriculture (Smit et al., 2001). Given that many of these risks are manageable through appropriate testing, treatment, and appropriate management practices, the balance between potential risks and rewards is dependent on the establishment of monitored environments that recognize UA as a primary or ancillary land use with suitable remediation where needed.

High accuracy, high spatial resolution data about urban infrastructure and land use (e.g., cadastral data) are needed to reduce uncertainty in potential services from natural capital. Though some of these data exist, they are highly inconsistent (Open Street Map), proprietary (Google Maps data) or difficult to generate (i.e., global scale processing of high resolution imagery). Other critical variables are much more difficult to obtain from remotely sensed data. For example, FAO reported crop productivities are likely lower than biointensive yields on intensively managed small plots (Jeavons, 2002). Increased yields, commensurate with increased intensity, are possible in small-scale cultivation (Altieri & Nicholls, 2008) or technology-based growth environments (Despommier, 2009). (See Table S1 for crop yields in the FAO data and under biointensive cultivation.) In addition to mapping and/or inventory of UA, reporting of actual yields is necessary to reduce uncertainty in yield. Socioeconomic data such as producer prices for urban produce, costs of energy, water or critical inputs are by necessity collected locally.

We only explicitly account for the cost of fertilizer applications as a private cost. While it is necessary to account for a set of representative costs to be able to compare net benefits from UA, the global scope of our study, and total lack of representative cost data for UA for (nearly) all countries prohibits us from initiating any instructive calculations. It is important to highlight that this dearth of empirical data has been noted previously (e.g., FAO, 2007, chap. 3) and provides a critical opportunity for additional future research (e.g., targeted survey-based data collections). Systematic cataloguing of key cost items that would influence the private net benefits (returns) from UA, for a representative set of countries for each region, is necessary for similar research endeavors in the future. These items include: urban land rents, effort (labor) used in UA, capital intensiveness of UA (use of structures, machinery and equipment, automation), rate of other resource use (water and electricity), and policies that confer economic advantages to urban farmers (preferential tax treatments, subsidies). Augmenting this data with socioeconomic characteristics of urban farmers—demographics, outside employment, and experience with farming, would yield a rich and profoundly valuable dataset. Finally, a more comprehensive list of policies that support UA is also needed, since such policies can either reduce costs for producers directly, or do so indirectly by improving their access to land, financing, and markets.

Frameworks for collecting and aggregating such data, archiving them, and cohosting them with environmental data would greatly enhance accounting of services from natural capital in cities. Such accounting is desperately needed to understand cumulative impacts of local land use policies and hyperlocal design choices such as between green roofs (for agriculture or not) or photovoltaic panels.

Despite the aforementioned caveats, our results demonstrate the potential for a significant range of benefits across the food, water, and energy nexus in managed urban ecosystems. In some ways, such ecosystems are an ideal place to test and experiment with much needed adaptive capacity in areas vulnerable to climate change (Heffernan, 2012). The services of UA we estimated simultaneously address needs to mitigate temperature, flood, energy and food security vulnerabilities (de Zeeuw et al., 2011). Although potential agricultural returns from UA are modest, the food and revenues generated from UA could mean the difference between survival and famine (Zezza & Tasciotti, 2010). Reduced storm runoff and energy savings are less on an aggregate level than agricultural returns, but Figures 2–4 illustrate how such benefits vary in importance geographically based on climate. However, our results do highlight that natural capital can take a varying role based on context and geography, in developed countries providing locally produced, lucrative agricultural outputs, in temperate areas contributing to a reduction in heating or cooling, and in tropical locations mitigating storm runoff.

In developing regions, UA may be critical to survival or a necessary adaptation to changing climate. In temperate, developed countries, UA increases access to recommended daily consumption of fresh fruits and vegetables (McCormack et al., 2010). Recommended consumption of vegetables for the urban population may be met almost entirely through UA (Martellozzo et al., 2014), leading to reduced emissions from transportation of agricultural products (Weber & Matthews, 2008), and decreased food waste (Kulak et al., 2013). Additional positive externalities may arise from waste cycling (Metson & Bennett, 2015a, 2015b; Smit & Nasr, 1992), UHI abatement (Clinton & Gong, 2013), and improvements to human health and well-being (Tzoulas et al., 2007). To quantify such effects will require substantial efforts in both monitoring and modeling. Specifically, sensor networks that monitor energy and water flows within buildings, high-resolution thermal data, energy balance models and even spatiotemporally resolved biometric data (for example from wearable sensors) may be needed for a complete understanding of the trade-offs between natural capital and more sterile building practices.

5. Conclusions

The results documented here illustrate potentially large incentives to incorporate agricultural ecosystems within existing and planned urban areas. We show potential benefits with unique features spanning the food-energy-water nexus, arising from the natural capital of agroecosystems installed in cities. To our knowledge, this study represents the most comprehensive effort to consistently estimate urban ecosystem services at global scale. We view the research presented here as a concrete step toward a more informed discussion of aggregate level ecosystem services from natural capital in urban areas. For consistency, the data sources we use are global scale and the models are simple, by necessity. Access to high spatiotemporal resolution data would enable more complex models and decreased uncertainty. Methods for collection and aggregation of socioeconomic data and synthesis with other environmental datasets would also enable much-needed synthesis of interdisciplinary spatial information. However, with existing data synthesized in Google's Earth Engine cloud-computing platform, our estimates of ecosystem services show potential for millions of tonnes of food production, thousands of tonnes of nitrogen sequestration, billions of kilowatt hours of energy savings, and billions of cubic meters of avoided storm runoff from natural capital in urban areas. Our findings justify implementation of systems for reporting, monitoring, aggregation and synthesis of data about productive ecosystems in the urban infrastructure, if not direct implementation of UA, as a useful practice in urban areas.

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