Environmental Research Communications

LETTER • OPEN ACCESS

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To cite this article: Gaston Small *et al* 2019 *Environ. Res. Commun.* **1** 091007

Environmental Research Communications

OPEN ACCESS

RECEIVED
2 May 2019

REVISED
1 August 2019

ACCEPTED FOR PUBLICATION
15 August 2019

PUBLISHED
13 September 2019

LETTER

Excess phosphorus from compost applications in urban gardens creates potential pollution hotspots

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Keywords: nutrient use efficiency, compost, phosphorus, urban agriculture

Supplementary material for this article is available online

Abstract

Urban sustainability initiatives often encompass such goals as increasing local food production, closing nutrient loops through recycling organic waste, and reducing water pollution. However, there are potential tradeoffs among these desired outcomes that may constrain progress. For example, expansion of urban agriculture for food production may create hotspots of nutrient pollution if nutrient recycling is inefficient. We used gardener and urban farmer survey data from the Twin Cities Metropolitan Area (Minnesota, USA) to characterize phosphorus (P) and nitrogen (N) inputs and harvest in order to determine nutrient use efficiencies, and measured soil P concentrations at a subset of these sites to test whether excess soil P was common. All survey respondents ($n = 142$) reported using some form of soil amendment, with plant-based composts being the most common. Median application rates were 300 kg P/ha and 1400 kg N/ha. Median nutrient use efficiencies were low (2.5% for P, 5.0% for N) and there was only a weak positive relationship between P and N inputs and P and N harvested in crop biomass. Garden soils had a median Bray P value of 80 ppm, showing a buildup of plant-available P far exceeding recommended levels. Our results show that urban gardens are characterized by high nutrient inputs and inefficient conversion of these nutrients into crops, leading to buildup and potential loss of P and N from garden soils. Although urban gardens make up only 0.1% of land area in the Twin Cities, compost application to these urban gardens still constitutes one of the largest inputs of P to the watershed. In order to maximize desired outcomes from the expansion of urban agriculture (UA), it will be necessary to target soil amendments based on soil nutrient levels and crop nutrient demand.

Introduction

Rapid and extensive urbanization over the past century has resulted in urban ecosystems driving biogeochemical fluxes on local, regional, and global scales (Grimm *et al* 2008). Cities are hotspots of resource consumption and waste production, relying on areas of ecologically productive land 100–300 times larger than the geographic area of the city for the provisioning of food and disposal of waste (Rees and Wackernagel 1994, Folke *et al* 1996, Alberti *et al* 2003). The expected growth of urban populations in the coming decades (McDonnell and MacGregor-Fors 2016), combined with increased consumption of animal protein and postconsumer food waste among urban residents (Seto and Ramankutty 2016), will continue to increase the magnitude of these fluxes. Whereas urban sustainability initiatives may include such goals as closing nutrient loops through the recycling of organic waste, increasing local food production, and reducing water pollution, potential tradeoffs among these desired outcomes may arise, necessitating careful management in order to optimize sustainability goals. Here, we examine the potential for urban agriculture to create hotspots of nutrient pollution due to inefficient uptake, and how this can undermine the goals of urban sustainability.

Closing nutrient cycles in urban ecosystems has been identified as essential to food system and aquatic ecosystem sustainability (Childers *et al* 2011, Burger *et al* 2012). Increasing recycling of urban organic waste, notably through composting food and green waste, is necessary to move from linear to more circular nutrient flows (Baker 2011), and many municipalities have ambitious goals to keep organic waste out of landfills through increased composting (e.g. Metson and Bennett 2015a). Recycled urban nutrients have the potential to reduce dependence on mined phosphorus (P) and synthetic nitrogen (N) fertilizers for food production. Price fluctuations related to the geopolitical availability of mineral phosphate may constrain agricultural production in the next century, and as such is of particular concern (Cordell and White 2011, Childers *et al* 2011). Although much of the P and N in urban organic waste originates in food produced in rural areas, returning these nutrients back to these rural agricultural systems poses economical challenges, as the cost of transportation can often exceed the value of the resource. The rapid expansion of urban agriculture (UA) in the United States (Fox 2011) could act as one productive use for recycled urban nutrients (Nogeire-McRae *et al* 2018). Urban agriculture can take a variety of different forms, but most outdoor urban agriculture (including backyard and community gardens, as well as commercial urban farms) typically relies heavily on compost-derived nutrients (Metson and Bennett 2015a). If UA can increase urban nutrient recycling, that would add to a number of other environmental, social, and cultural benefits that have been described (Recknagel *et al* 2016, Artmann and Sartison 2018, Clinton *et al* 2018).

There may be a downside of nutrient recycling through UA, however. There may be an implicit assumption among UA practitioners and policy makers that all compost applied to UA is recycled, without further consideration of the fate of those nutrients. However, if crops only recover a fraction of nutrient inputs, excess nutrients could be lost to the environment, and the expansion of UA could result in hotspots of nutrient pollution. The local loss of nutrients is in large part a function of the area of land under cultivation, the rate of nutrient inputs per unit area, and the nutrient use efficiency (defined as mass of nutrients removed in crop biomass relative to mass of nutrient inputs) of the crops under cultivation (Schröder *et al* 2011). Crop production from rural agriculture has P use efficiencies (PUE) between 60%–100% (Suh and Yee 2011) and N use efficiencies (NUE) typically ranging from 20%–70% (Swaney *et al* 2018). Losses of P and N from agricultural lands are significant at regional and global scales because of the vast area under conventional agricultural production (Vitousek *et al* 1997, Carpenter *et al* 1998, Rockström *et al* 2009).

Compared to rural agriculture, UA may have nutrient application rates that are much higher per unit area, and low nutrient use efficiency, potentially resulting in nutrient buildup and loss. Furthermore, compost and other organic soil amendments often have low N:P ratios relative to requirements for crop production, so that if compost is applied as the sole source of nutrients (i.e., based on crop N-demand), soils would receive an excess of P (Kleinman *et al* 2007, 2011). If nutrient efficiency is low, then the potential for nutrient losses could be significant locally or regionally, especially under scenarios of UA intensification (Grewal and Grewal 2012, Clinton *et al* 2018). The few available published UA nutrient budgets indicate N and (especially) P application rates exceed crop demand (e.g., Smith 2001, Cofie *et al* 2003, Metson and Bennett 2015a, Wielemaker *et al* 2019) which has been linked to nutrient leaching to ground water (Abdulkadir *et al* 2015). Compared to conventional rural agriculture, urban farmers or gardeners may be more likely to apply compost or other soil amendments far in excess of crops' nutrient demand (Taylor and Taylor Lovell 2014), as the cost of nutrient inputs at small scales is minimal.

Because only a handful of quantitative nutrient studies exist on UA (each with their own focus areas and system boundaries), we still know little about the factors affecting UA nutrient input rates and nutrient use efficiency, and how nutrient fluxes from UA compare with other nutrient fluxes within urban ecosystems. Considering the growing importance of UA in many urban areas, it is important to understand the potential for nutrient recycling and loss from UA, and to examine what steps are needed to maximize desired outcomes while minimizing unintended consequences. Such data are needed by municipal governments to inform regulations and develop educational outreach efforts guiding UA.

In order to quantify the role of UA in nutrient cycling in one well-studied urban ecosystem, we used data collected from a survey of gardeners and urban farmers in the Twin Cities Metropolitan Area (Minnesota, USA) to characterize P and N inputs, surpluses, and nutrient use efficiencies, and also measured garden soil P concentrations at a subset of these sites. We combine these data with analysis of areal images to estimate cumulative nutrient fluxes associated with UA, relative to other N and P fluxes previously documented in this watershed (Hobbie *et al* 2017).

Methods

Survey data collection

An electronic survey was developed to collect data on the various soil amendment practices and garden yield across gardens/farms throughout the Twin Cities, based on survey questions used by Metson and Bennett (2015a). The survey was approved by the University of St. Thomas Institutional Review Board (1007423-1). The survey was distributed through the Minneapolis Food Council monthly newsletter, newsletters from Saint Paul district councils, and on social media.

Participants were asked to identify the source of fertilizers or soil amendments they used (selecting from 28 choices) and the amount use of each type (table S1 is available online at stacks.iop.org/ERC/1/091007/mmedia). Participants were also asked to identify the size of their garden/farm, length of operation, land history, physical address, the list of individual crops grown and their respective yields in the survey (appendix).

Total nutrient input, crop nutrient use efficiency and surplus nutrient estimations

The total mass of different soil amendments applied to a farm was either directly reported by survey respondents, or was estimated by multiplying the reported volume of soil amendments added by their bulk density (table S2). The amount of soil amendments reported as used was multiplied by soil amendment total N and P content (values of total N and P content for different soil amendments are detailed in table S3), and summed across all the soil amendment types applied at a garden/farm. Total P and N in the harvested crops were calculated by multiplying crop tissue P and N content (table S4) by respective crop yields and summed across all crop types at a garden/farm. When nutrient content could not be obtained for certain crops, the average nutrient content of its closest relatives was used as proxy. Total crop nutrient use efficiency (N use efficiency; NUE, or P use efficiency; PUE) for each farm was calculated as a ratio of nutrients removed in plant biomass relative to the total amount of nutrient inputs to soil. Surplus nutrients are defined as the amount of input nutrients above nutrients removed through crop harvest.

Garden soil P measurements

In order to test whether excess P was common in garden soils, we collected soil samples from the gardens of 35 survey respondents. We collected an aggregated soil sample from within the garden (approximately six 10-cm deep cores at each site), and another aggregated soil sample from outside of the garden (typically adjacent lawn) for comparison. Soil samples were analyzed for Bray-1 P (a measure of plant-available P) at the University of Minnesota Research Analytical Laboratory (Frank *et al* 1998).

Statistical analysis

Because of skewed distributions, data are presented as medians and quartiles. Relationships between yield (in terms of P) and P-inputs, yield (in terms of N) and N-inputs, surplus P and P-inputs, surplus N and N-inputs, PUE and P-inputs, and NUE and N-inputs were analyzed using linear regression with log-transformed data. We note that PUE and NUE are not independent of P and N inputs, respectively, but we are including these relationships to facilitate comparisons with other published studies (e.g. Anglade *et al* 2018). Multiple linear regression was used to test for the effects of surplus P and garden age on garden soil Bray P-1. All analyses were conducted using JMP Pro 13.2.

Estimating total N and P flux from UA

We followed Taylor and Taylor Lovell's (2012) approach and used Google Earth to map and enumerate urban gardens in the Capitol Region Watershed District (CRWD), encompassing 10,600 ha in Ramsey County, Minnesota, including Saint Paul and parts of neighboring suburbs (figure 1). Survey response gardens located outside of CRWD were used to develop a set of reference images for small household gardens, community gardens, and urban farms. The imagery was viewed at maximum resolution and methodically searched in an east-west direction, between the watershed boundaries. Identified gardens were marked and categorized as either a household garden or an urban farm/community garden. The tree canopy in the Twin Cities obscures many smaller gardens, so we used known locations of known survey gardens within CRWD to estimate our detection efficiency. We estimated the total number of gardens in CRWD as:

$$\text{Total CRWD gardens located using Google Earth} \times \frac{\text{Total survey gardens in CRWD}}{\text{Survey gardens in CRWD located using Google Earth}} \tag{1}$$

We used this approach to separately estimate the number of community gardens/urban farms and small household gardens, and used the median size for each category (from survey data) to estimate the total area of

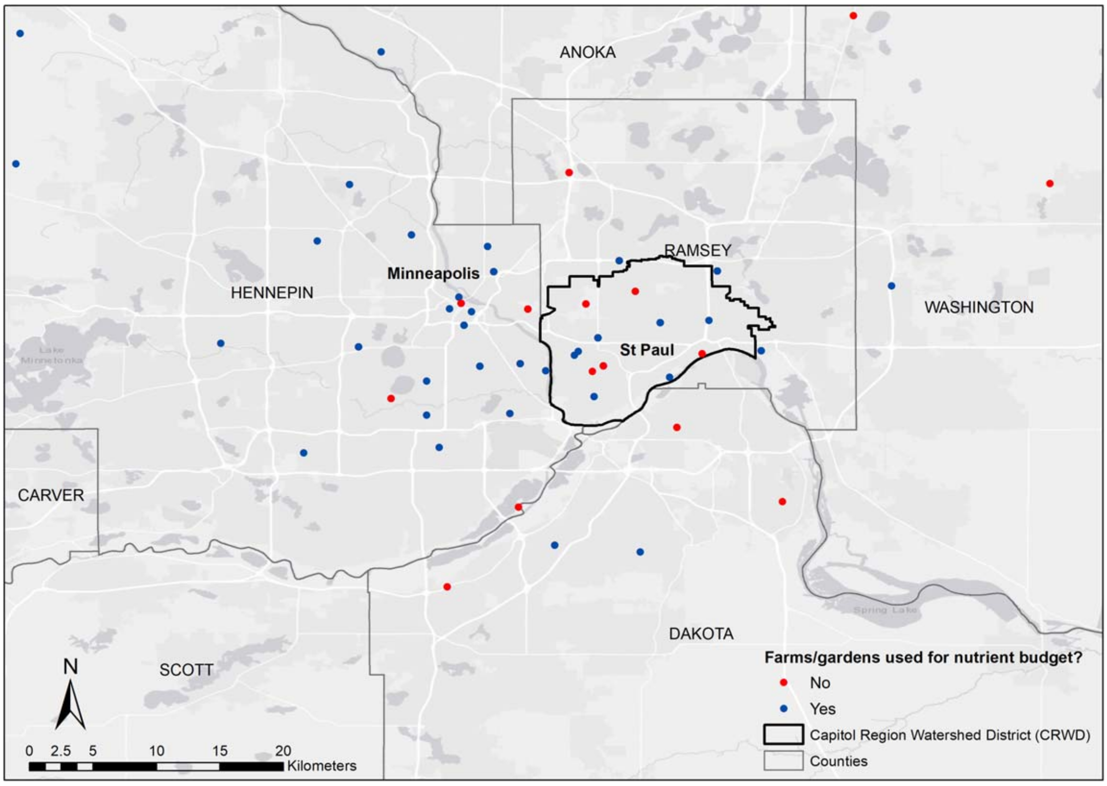

Figure 1. Location of survey gardens and location of Capitol Region Watershed District (CRWD). Two of the 142 survey gardens are outside of this map, and some points represent multiple gardens that are in close proximity. Blue dots depict gardens for which sufficient data was available to create nutrient budgets; red dots depict gardens where partial data sets were available.

gardens in the watershed. We multiplied this area by the median P and N application rates based on survey data to estimate the total N and P inputs to the watershed from UA activities. We note that, because median values for soil amendment application rate are considerably lower than mean values, our approach is conservative in estimating watershed-scale nutrient inputs from urban gardens.

Results

Total nutrient input, crop nutrient use efficiency and surplus nutrient

We obtained a total of 142 survey responses including: 109 private gardens, 20 community gardens, 2 commercial urban farms, and 11 unspecified entities (figure 1). Median garden size was 0.0014 ha, and median length of operation was 5 years (table 1). All 142 respondents reported using some form of soil amendment. Plant-based inputs were most common: 86 respondents (60.6%) reported using plant-based compost, 35 (24.6%) reported applying leaves, 28 (19.7%) used woodchips, and 26 (18.3%) used hay. Manure composts were also commonly applied: 16 respondents (11.3%) reported applying cow manure, 15 (10.6%) reported using chicken manure, and another 26 respondents reported using other manures (e.g, horse, sheep, goat, rabbit). Inorganic soil amendments were less common. Eleven respondents (7.7%) reported using fertilized potting mixes, 8 respondents (5.6%) reported using liquid fertilizers, and 4 respondents (2.8%) reported using synthetic fertilizers (table S1).

The median P application rate from soil amendments was 300 kg P/ha, and median N application rate was 1400 kg N/ha (table 1). 40 out of 142 respondents (28%) have had soil tested, but we found no differences in P or N application rates between gardeners who have had soil tested and gardeners who have not. There was no relationship between garden area and area-specific N- or P-application rates, although larger gardens (>100 m^2) all had N application rates <1200 kg N/ha, whereas several smaller gardens had N application rates as high as 100,000 kg N/ha. We note that omitting these gardens with extremely high soil amendment application rates does not change any of the results reported here.

The median reported crop yield (wet mass) was 20,900 kg/ha. In terms of P, median crop yield was 9 kg P/ha, and, in terms of N, median crop yield was 53 kg P m^{-2} (table 1). Yield-P increased significantly with P inputs ($P = 0.0247$), and yield-N increased with N inputs ($P = 0.0339$), but these relationships were weak (R^2 values of 0.065 and 0.055, respectively) (figures 2(A) and (B)).

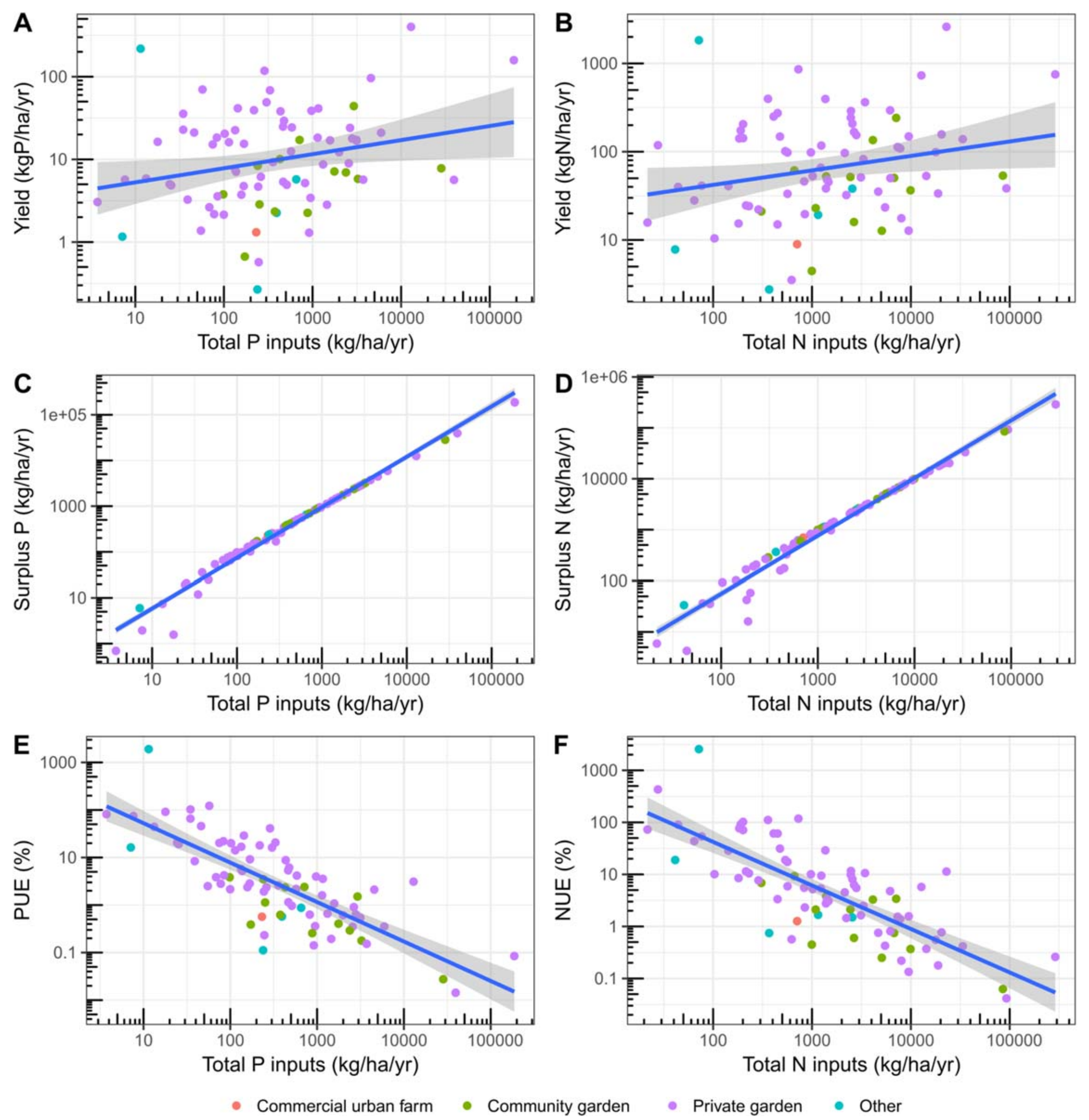

Figure 2. Relationship between total inputs of P and N, and crop yield (A) and (B), surplus nutrients (C) and (D), and nutrient use efficiency (E) and (F). The shaded gray areas represent the 95% confidence interval for the regression lines.

Table 1. Characteristics of gardens/farms from the survey (n = 142).

	Q1	Median	Q3
Garden size (ha)	0.00059	0.0014	0.0037
Years in operation	3	5	9
P application rate (kg P/ha)	100	300	1200
N application rate (kg N/ha)	400	1400	5500
Crop yield (kg wet mass/ha)	2100	20,900	62,400
Crop yield (kg P/ha)	4	9	230
Crop yield (kg N/ha)	24	53	150
P surplus (kg P/ha)	91	259	1170
N surplus (kg N/ha)	251	125	5570
PUE (%)	0.6	2.5	16.2
NUE (%)	1.2	5	13.1
Garden soil Bray-1 P (ppm)	58.3	80	131.5
Lawn soil Bray-1P (ppm)	38.8	60	126.5

Table 2. Estimated number of gardens and cumulative garden area in Capitol Region Watershed District.

	Community gardens/urban farms	Backyard gardens
Number found in Google Earth analysis	21	36
Total Survey Gardens in CRWD	7	82
Survey Gardens found during Google Earth Analysis	3	2
Estimated detection efficiency	43%	2%
Estimated total number of gardens in CRWD	49	1476
Median garden area (ha)	0.0721	0.0056
Estimated cumulative garden area in CRWD (ha)	3.5329	8.2656

Median P surplus was 259 kg P/ha, and median N surplus was 125 kg N m^{-2} (table 1). Surplus P increased with P inputs ($P < 0.0001$; $R^2 = 0.977$), and Surplus N increased with N inputs ($P < 0.0001$; $R^2 = 0.965$) (figures 2(C) and (D)). The median PUE was 2.5%, half of the median NUE value (5.0%) (table 1). PUE and NUE decreased with P inputs ($P < 0.0001$, $R^2 = 0.608$) and N inputs ($P < 0.0001$, $R^2 = 0.603$) (figures 2(E) and (F)).

Garden soil P measurements

Bray- P for garden soils had a median value of 80 ppm; 4 of 26 samples (15%) were below 50 ppm. Bray-1 P for lawn soils had a median value of 60 ppm, with 11 of 26 samples (42%) below 50 ppm (table 1). Garden Bray-1 P was positively related to the number of years the garden had been in operation ($P = 0.0237$) and was marginally significantly related to the reported P surplus ($P = 0.0619$). The interaction term between these factors was not significant ($P = 0.1537$).

Estimating total N and P flux from UA

Using areal imagery, we identified 21 community gardens/urban farms in the CRWD watershed, including three out of seven that had responded to the survey (estimated detection rate of 43%), resulting in an estimate of 49 urban farms/community gardens. We identified 36 backyard gardens within the CRWD boundary from areal imagery, including 2 of 82 survey gardens (estimated detection rate of 2%), resulting in an estimate of 1476 small household gardens in the watershed (table 2). Based on these calculations, we estimate a survey response rate of 14% (7 out of 49 within CRWD) for community gardens and urban farms, and 5.5% (82 out of 1476 within CRWD) for small household gardens. The median area of urban farms/community gardens was 0.0721 ha, so we estimate a cumulative area of 3.5 ha for these large gardens. The median area of smaller backyard gardens was 0.0056 ha, so we estimated a total area of 8.2 ha for these gardens. The combined estimated total garden area in the watershed was 11.8 ha, approximately 0.1% of the total watershed area (10,600 ha). Using the median reported garden nutrient application rates of 318 kg P/ha/yr and 1381 kg N/ha/yr, we estimate that 3,752 kg P and 16,295 kg N are applied to gardens annually. Averaged across the 10,600 ha watershed, these input rates correspond to 0.354 kg P/ha/yr and 1.537 kg N/ha/yr. This P input rate would constitute approximately 19% of the P budget that Hobbie *et al* (2017) calculated for this same watershed; total P input as UA soil amendments is larger than P inputs from weathering (0.14 kg P/ha/yr), similar in magnitude to atmospheric deposition (0.49 kg P/ha/yr), and 43% of the magnitude of P inputs from pet waste (the largest P input of 0.82 kg P/ha/yr). It is important to note that, due to Minnesota's ban on phosphorus lawn fertilizers, Hobbie *et al* assumed no lawn P inputs, thereby increasing the relative importance of other inputs (including compost). Estimated N inputs from UA amendments, by contrast, are a minor component of the watershed N budget, making up approximately 3% of total N inputs (Hobbie *et al* 2017).

Discussion

Desired outcomes for UA, in the context of sustainability, include increasing local food production while recycling organic waste through composting (Metson and Bennett 2015b). UA has previously been characterized as resource-efficient based on its use of vacant spaces, the potential use of stormwater for irrigation, and the reduction of food miles (Artmann and Sartison 2018), although UA is inefficient in terms of human labor requirements (McDougall *et al* 2019). Our results show that urban gardens in one major US metropolitan area are characterized by high nutrient inputs and inefficient conversion of these nutrients into crops. Median garden P inputs (300 kg P/ha/yr) were approximately 30-fold greater than mean P application rates in US conventional agriculture (IPNI (2010)), and median garden N inputs (1400 kg N/ha/yr) were approximately ten-fold greater compared to N inputs in conventional agriculture in the United States (Swaney *et al* 2018). Because UA yields were similar to yields from conventional agriculture (Swaney *et al* 2018), nutrient use efficiencies from UA (2.5% for P, 5.0% for N) were much lower than corresponding efficiencies from conventional agriculture (89% PUE (IPNI (2010)); 31% NUE (Swaney *et al* 2018)). It is important to note that the majority of UA practitioners (i.e., backyard gardeners) are not trained farmers, and, unlike large-scale farmers, have little economic or regulatory incentive to carefully manage soil nutrients. The ability of urban farms and gardens to assimilate organic waste may be a desired outcome of UA (Metson and Bennett 2015b), but optimizing this goal necessarily leads to low nutrient recycling efficiencies and the potential for nutrient loss through leachate (Small *et al* 2017, Small *et al* 2018).

Although UA occupies only 0.1% of the CRWD watershed, the associated inputs of soil amendments still constitute one of the largest P fluxes in the watershed. If UA is scaled up, the impact of P over-application would be amplified accordingly. For example, if UA expanded to cover 3% of the land area of CRWD (less than the current extent of UA in Montreal (Metson and Bennett 2015a) and less than Scenario 1 in Grewal and Grewal (2012), in which 80% of vacant lots in Cleveland are used for UA), then the resulting P inputs would be more than 5 $\times$ the current cumulative P inputs to the watershed. Clinton *et al* (2018) estimated that UA could produce 100–180 million tonnes of food per year globally, providing ecosystem services worth \$80-160 billion. However, achieving these yields with a 2.5% PUE would result in the buildup of P in urban soils of approximately 1.7 Tg P/yr, a significant component of the global P budget (Bennett *et al* 2001).

Large mass transfers of P into a relatively small area of agricultural production can contribute to the buildup and subsequent export of soil nutrients, as has been documented in agricultural areas with intensive livestock production, where manure is applied for fertilization (Sharpley *et al* 2014, Jia *et al* 2015). Our results indicate that UA represents an even greater concentration of P compared to livestock manure application; for example, the median P input rate to urban gardens in our study was six-fold higher than the manure P loading rate to peri-urban regions around Beijing, and average soil P in urban gardens was twice as high as manure-fertilized soils under vegetable production around Beijing (Jia *et al* 2015). The surplus P builds up in garden soil, as illustrated by the positive relationship we observed between garden age and garden soil Bray P. The lack of a strong relationship between reported P surplus and measured Bray-1 P may be due to reporting errors or inter-annual variability in P-inputs, but P loss in the form of leachate or runoff would also contribute to deviation from this relationship. Long-term over-application of P may contribute to water pollution for decades or longer (Sharpley *et al* 2014). Legacy P transmission from soils can occur not only from surface runoff (which may be high in cities due to the high density of impervious surface; Janke *et al* 2014), but also from infiltrated leachate, especially where soil P levels are high (Vadas *et al* 2007, Holman *et al* 2008, Domagalski and Johnson 2011). Data from one long-term agricultural P addition experiment show that soils with less than 60 ppm Olsen-P strongly retained P, but P losses to drainage water increased with increasing soil P above this level (Heckrath *et al* 1995). We found garden soils to consistently have higher concentrations of plant-available P, sometimes by large margins (e.g., soil Bray P concentrations reached up to 259 ppm).

A limitation of our survey data is that values for inputs and yield are self-reported and typically are based on estimates from respondents. Survey responses are also a one-year snapshot, so reported soil amendment application rates may not necessarily reflect long-term average annual inputs. However, these values are generally in agreement with published values. The reported crop yield values from our survey are similar to values reported for gardens from Montreal (Metson and Bennett 2015a), New York (Ackerman 2011), Oakland (McClintock *et al* 2013), and the Twin Cities (Small *et al* 2017, 2018). Nutrient inputs from soil amendments are also consistent with values reported by (Metson and Bennett 2015a) and Small *et al* (2017, 2018) for small urban gardens, and, for larger gardens in our dataset, N application rates were similar to those reported by Anglade *et al* (2016) for organic market gardens. The wide range of reported inputs, yields, and management strategies may be due in part to the range of motivating factors and management strategies among urban gardeners (Metson and Bennett 2015b, Lewis *et al* 2018). There are few estimates of UA land area, and they diverge widely, likely due to methodological differences (e.g., definition of urban boundary) as well as actual differences in land usage among cities. Our estimated garden area of 0.1% in CRWD falls between reported values of 0.04% for Chicago (Taylor and Taylor Lovell 2012) and 3.6% for Montreal (Metson and Bennett 2015a).

We note that our analysis focuses on soil-based outdoor growing, and that other forms of UA have different implications for nutrient recycling. For example, intensive indoor growing can achieve high crop yields year-round without nutrient losses to the environment (Kozai *et al* 2016), but this approach typically relies on hydroponic nutrient delivery and presents limited opportunities for recycling urban nutrients. Although our study focused on a limited array of UA activities in one US city, we suggest that the general findings regarding the widespread over-application of compost-derived nutrients hold true for most soil-based outdoor UA, including rooftop farms (e.g. Whittinghill *et al* 2015, 2016).

We emphasize that the low nutrient use efficiencies resulting from over-application of compost are not a consequence of recycling urban organic waste *per se*; indeed, over-application of synthetic fertilizer is also common in urban gardens (e.g. Taylor and Taylor Lovell 2014). For small-scale urban vegetable gardens, there is little incentive for gardeners to have their soil tested and to carefully target soil amendment applications to soil conditions and crop demands, and there is minimal financial or regulatory disincentive against over-applying nutrients. Furthermore, many home gardeners have a 'more is better' mentality regarding nutrient application (Taylor and Taylor Lovell 2014), and compost may also be added to gardens for reasons other than nutrient inputs (e.g., to improve soil structure or buffer pH). Targeted educational campaigns, combined with free garden soil testing, are needed to raise awareness among UA practitioners of the importance of avoiding excessive nutrient inputs. Similar campaigns have been effective in reducing excessive fertilizer inputs to urban lawns, especially due to 'spillover effects' where citizens share information they have learned with their neighbors (Martini *et al* 2014, Martini and Nelson 2015). As many UA practitioners may already be interested in sustainability (McDougall *et al* 2019), this population may be amenable to making behavioral changes that decrease negative environmental impacts from their actions.

Our findings do underscore the mass balance challenges inherent in using UA to recycle nutrients from urban organic waste back into the human food system, regarding the land area of UA required to assimilate urban organic waste (Metson and Bennett 2015a, Wielemaker *et al* 2018) and the low N:P ratio of compost relative to crop demand. As cities (including Minneapolis and Saint Paul) engage in increasing organics recycling, the production of urban compost will continue to increase, and as UA continues to grow, so will demand for that compost. Our results underscore the importance of municipalities considering nutrient management implications as they develop strategies to manage organics recycling and urban agriculture.

Conclusions

Our results highlight the challenge of recycling P and N in urban ecosystems through composting coupled with UA. As the potential ecosystem services from UA are beginning to be quantified (and they are substantial: Clinton *et al* 2018), the issue of ecosystem disservices in the form of nutrient pollution should also be considered when intensification of UA is proposed. In order to avoid creating urban gardens as hotspots of nutrient pollution monitoring, education, and appropriate regulations are needed. Policies aimed at increasing UA should be accompanied by funding for monitoring (e.g. garden soil testing) and extension support by agronomists trained in nutrient management. As UA continues to expand, nutrient management will require an increased focus, in order to maximize desired outcomes and minimize unintentional consequences.

Acknowledgments

This study was supported in part by a National Science Foundation CAREER award (award number 1651361) to G E Small.

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