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Potential for high contribution of urban gardens to nutrient export in urban watersheds

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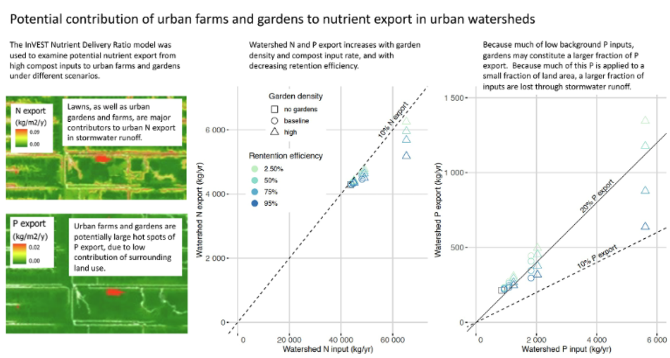
HIGHLIGHTS

- Compost application to gardens comprises a large phosphorus input to the urban landscape.
- Gardens can be significant contributors to watershed P export if compost inputs are high.
- N runoff is mostly from lawns near streets, but garden compost can be an important contributor.

GRAPHICAL ABSTRACT



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ABSTRACT

Urban gardens and farms typically use compost as a source of nutrients, often at levels that exceed crop nutrient demands. Although land dedicated to agriculture is a small fraction of urban land use, high input rates coupled with low nutrient use efficiencies suggest that export of nitrogen (N) and phosphorus (P) from this land could be potentially important contributors to urban nutrient budgets. We used the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) Nutrient Delivery Ratio model to examine the potential impact of garden density, compost input rates, and nutrient retention efficiency on N and P export from stormwater runoff for a 737-ha urban residential area in Saint Paul, Minnesota. Although gardens and farms accounted for 0.1-0.5% of land area in our scenarios, compost inputs accounted for as much as 33% of N inputs and 85% of P inputs to the urban landscape. The contribution of gardens to urban nutrient export through stormwater runoff is highly dependent on modeled maximum retention efficiency values. If retention efficiency is high, gardens with low compost inputs are similar to other vegetated land uses in contributions to nutrient export, but gardens become significant contributors to watershed P export if compost inputs are high, or if retention efficiency drops to 75%

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or lower. These results underscore mass-balance constraints inherent in urban nutrient recycling and highlight the importance of understanding the long-term fate of excess nutrients applied to urban landscapes.

1. Introduction

Land dedicated to crop production is a small fraction of total land area in cities. While nutrient budgets in rural agricultural watersheds are dominated by fertilizer inputs and crop harvest (e.g., Lowrance et al. 1985), urban watershed nutrient budgets are influenced by factors such as residential fertilizer use (mostly for lawns), leaf and other vegetation inputs, and pet waste (Hobbie et al. 2017). However, urban land dedicated to crop production may be an unrecognized contributor to urban nutrient budgets and eutrophication of downstream water bodies. Recycling of organic materials from municipal waste streams continues to increase (EPA, 2020), and compost application on urban gardens creates the potential to recycle waste nutrients back into the human food system. There are inherent biophysical constraints that pose challenges to recycling nutrients from urban compost back into the human food system. First, the mass of nutrients in compost generated in cities is much larger than the capacity of urban crops to assimilate those nutrients. For example, Metson and Bennett (2015) showed that land area of Montreal is not large enough to assimilate all the phosphorus (P) generated as organic waste by residents of the city, even if all urban land were used for crop production. Stoichiometric imbalance poses a second constraint. Compost typically has a low nitrogen:phosphorus (N:P) ratio relative to crop demands (Kleinman et al. 2007; Kleinman et al., 2011), and as a result, the application of compost to meet crop N demand results in excess application of P. Further contributing to nutrient overapplication, there is little or no economic and regulatory disincentive to limit compost application in small-scale urban agriculture, as there is in large-scale commercial agriculture.

We previously documented high compost inputs in urban gardens in Minneapolis and Saint Paul, Minnesota, with median application rates of 300 kg P/ha and 1400 kg N/ha (Small et al., 2019a), more than an order of magnitude higher than N and P application rates for conventional agriculture (IPNI, 2010; Swaney et al. 2018). Although urban food cultivation occupies only 0.1 % of the Capitol Region Watershed District in Saint Paul (Small et al., 2019a), this input rate of P in the form of compost was similar in magnitude to other sources of P inputs (household pet waste, atmospheric deposition, and weathering) estimated for this same watershed by Hobbie et al. (2017). Median nutrient use efficiency for these urban gardens was 2.5 % for P and 5 % for N (Small et al., 2019a), indicating that 20–40 times more nutrients are applied to gardens each year compared to the mass of nutrients recovered in harvested produce. Similarly low nutrient use efficiency was documented for P in Montreal (Metson and Bennett 2015), for N and P in several West African cities (Smith 2001, Cofie et al. 2003, Abdulkadir et al. 2015), and for many other locations (van de Vlasakker et al. 2022), suggesting that this is a common phenomenon, even if individual gardens within a city are highly variable.

Understanding the fate of these excess nutrients is important as urban vegetable gardens and farms that receive high compost inputs each year may have the potential to act as point-sources of nutrient pollution. Excess nutrients build up in garden soils; in our survey, median garden soil Bray-1P concentrations were 80 PPM (far higher than recommended values of 20–30 PPM for most field crops), and garden age was correlated with concentrations of plant-available P, indicating accumulation over time (Small et al., 2019a). The capacity of soil to retain excess P is finite; a long-term agricultural P addition experiment found that soils strongly retained P below a certain threshold, but above this threshold, P losses in drainage water increased with increasing soil P (Heckrath et al., 1995). From experimental garden plots in Saint Paul, we have documented that, although most excess P accumulates in the garden soil, nearly as much P can be exported as leachate or runoff as is recovered in harvested crops (Small et al. 2018), leading to a buildup of P in native soil below gardens (Small et al., 2019b). However, P loss can be greatly reduced with more targeted compost application (Shrestha et al. 2020).

Nutrient export from agricultural and urban landscapes is a primary driver of eutrophication in freshwater and coastal ecosystems (Smith et al. 2006). Legacy P along hydrologic flowpaths is known to contribute to long-term water quality impairment in rural agricultural watersheds (Sharpley et al., 2013; Van Meter et al., 2021), but this phenomenon is less well-understood in urban systems, with highly engineered hydrology and more heterogeneity of land use and nutrient inputs (Lintern et al. 2020). Nutrients lost from urban gardens could enter surface runoff, potentially leading to rapid export from the watershed via storm drains. Alternatively, nutrients lost from a garden via leachate could slowly build up in soil, eventually reaching the water table and subsequently being transported by groundwater flow on a much slower timescale. The relative magnitude of these hydrologic pathways depends in part on the land use history and spatial configuration of the garden site; we note for example that one urban farm in Minneapolis consists of 30 cm of soil atop an asphalt parking lot. Nutrient export from individual urban farms and gardens is likely to be highly heterogeneous and temporally variable, and as a result, estimating the potential contribution of this land use is a challenge.

Table 1
Area and corresponding NDR model parameters for each land use category for the 737-ha study area. Model parameters for all non-garden land use types were taken from Lonsdorf et al. (2021).

Land Cover	Area (ha)	N Input (kg/ha/y)	P Input (kg/ha/y)	Maximum N retention efficiency	Maximum P retention efficiency	Critical length (m)
Grass/Shrubs	126.0	94.1	1.46	0.93	0.44	20
Bare Soil	0.7	14.5	0.63	0.83	0.80	20
Buildings	134.1	14.5	0.49	0.05	0.05	1
Roads/Paved Surfaces	169.5	14.5	0.96	0.05	0.05	1
Lakes/Ponds	0.0	14.5	0.63	0.95	0.95	1
Deciduous Tree Canopy	275.6	94.1	1.46	0.93	0.44	20
Coniferous Tree Canopy	9.6	94.1	1.46	0.93	0.44	20
Emergent Wetlands	3.7	14.5	0.63	0.99	0.99	1
Forest/Shrub Wetland	0.0	14.5	0.63	0.99	0.99	1
Rivers	17.0	14.5	0.63	0.95	0.95	1
Extraction	0.0	14.5	0.63	0.05	0.05	1
Garden (low-input)	0.0–3.6	400	100	0.025–0.95	0.025–0.95	20
Garden (medium-input)	0.0–3.6	1400	300	0.025–0.95	0.025–0.95	20
[illegible]	[illegible]	[illegible]	1200	0.025–0.95	0.025–0.95	20

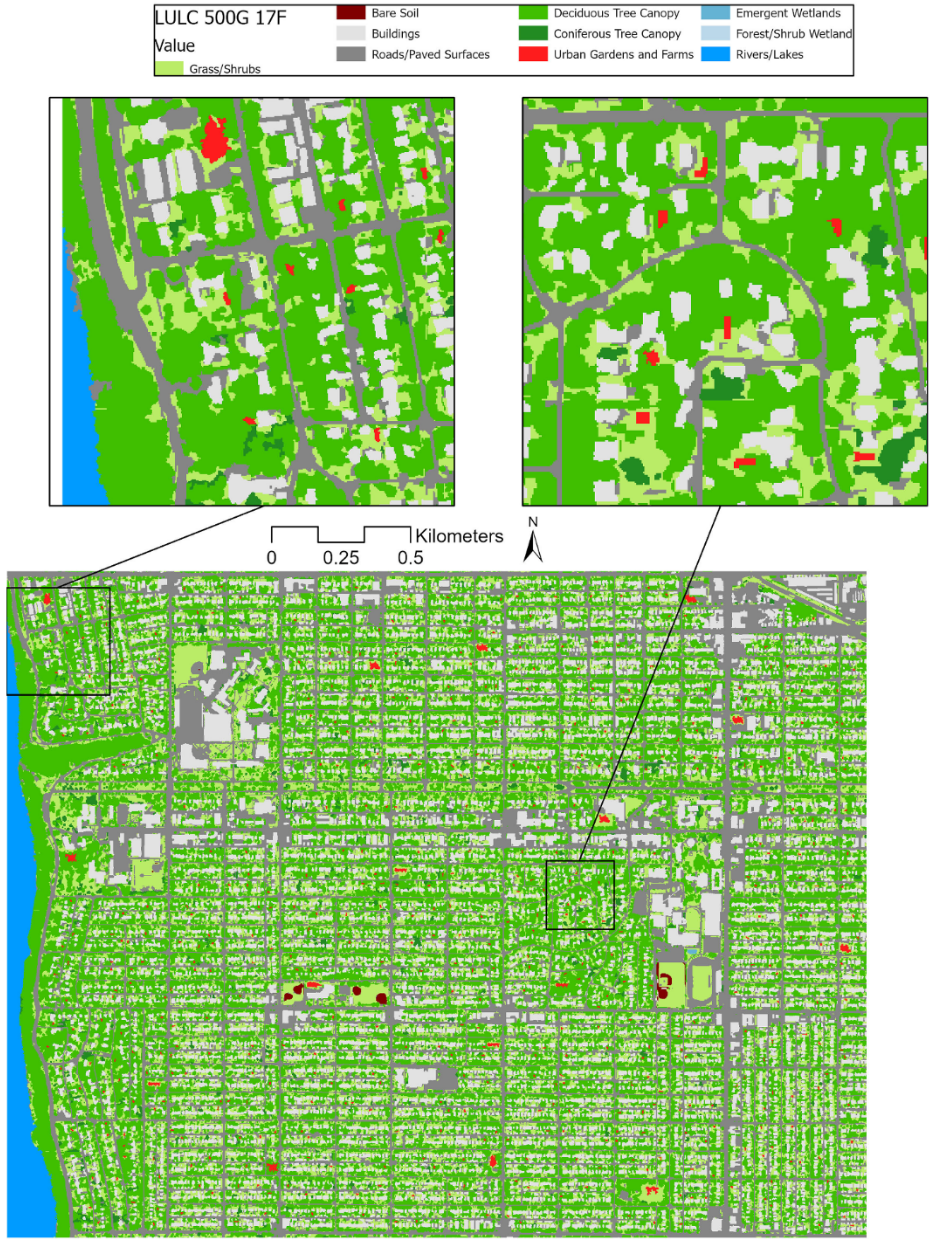

Fig. 1. Land use in the 737-ha study area, with the high-garden density scenario represented, containing 500 small gardens and 17 larger urban farms.

The objective of this study was to examine how the density of food-producing urban gardens and farms (hereafter referred to as "garden density"), compost input rates, and nutrient retention efficiency along hydrologic flowpaths affects the potential contribution of nutrient loss from urban gardens relative to other urban sources. We used the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) Nutrient Delivery Ratio (NDR) model, a tool designed to estimate the effects of land use changes and nutrient input and retention parameters on nutrient export that has been used in river basins across the United Kingdom (Redhead et al. 2018), and in the Southeastern (Benez-Secanho and Dwivedi, 2019) and Midwestern United States (Han et al. 2021). This model was recently used to estimate stormwater nutrient export from different land use scenarios in Minneapolis-St. Paul (Lonsdorf et al. 2021). Here, we apply the InVEST NDR model to compare scenarios with no gardens, baseline garden density, and high garden density, and with low, medium, and high compost input rates in St. Paul, Minnesota. Because nutrient retention is the least constrained parameter, we [illegible]ith the goal of exploring the conditions under which urban garden nutrient export becomes a significant contributor to urban runoff.

2. Methods

2.1. Site description and land cover

We selected a 737-ha urban residential neighborhood in Saint Paul, Minnesota, as this neighborhood was well-represented in our previous survey data Small et al., 2019a) which was used to parameterize the model. We used the Twin Cities Metro Area 1-Meter Land Cover Classification which has 12 distinct land cover types (Host et al. 2016). Land cover in the study area was dominated by deciduous tree canopy (37.4 %), roads and other paved surfaces (23.0 %), buildings (18.2 %), and grass and shrubs (17.1%) (Table 1). Gardens/farms were not identifiable on the land cover data set, so this map was used for the scenarios with no gardens. The relatively dense tree canopy cover does not allow for precise mapping of existing gardens using areal imagery, so instead we

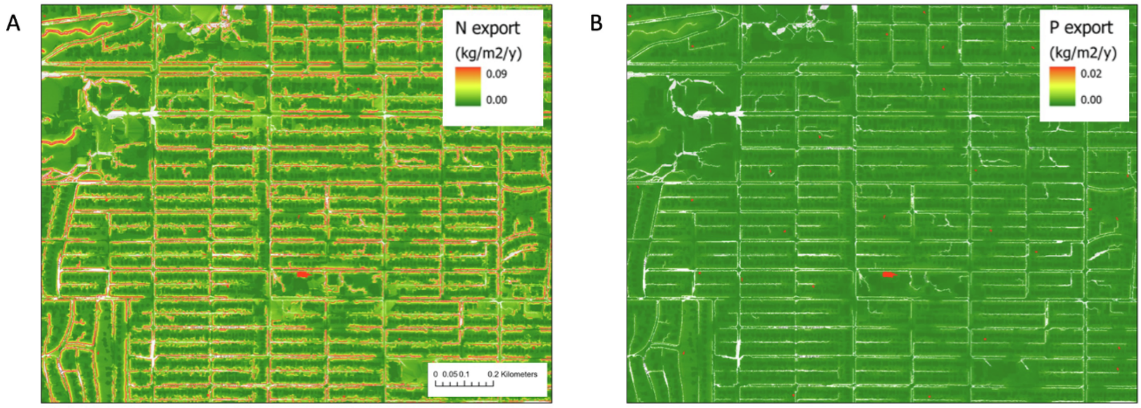

Fig. 2. Nitrogen (A) and phosphorus (B) export from a subset of the study area, under baseline garden density, medium compost inputs, and 50 % retention efficiency. Each pixel represents the mass of nutrients that are exported from 1 m^2 annually. One larger farm and twenty-eight small gardens are visible sources of N, but lawns adjacent to streets are cumulatively the largest contributor to N export. The gardens and farm are far greater contributors to P export compared to other land use.

manually added gardens and farms to the landscape at reasonable locations (i.e., replacing pixels designated as lawn or tree canopy) in ArcGIS Pro (Fig. A1) based on the density of farms and gardens that we previously calculated for this urban watershed using a combination of survey data and analysis of aerial imagery (Small et al., 2019a). Based on those data, this 737-ha area in Saint Paul would contain 100 small gardens and three larger urban farms. Backyard gardens were set at an area of 56 m^2 (0.0056 ha) and urban farms were set to be 721 m^2 (0.0721 ha), based in median values reported in Small et al. (2019a).

In the baseline garden density (Fig. A2) scenario, small backyard gardens accounted for a total of 0.56 ha, and larger urban farms accounted for 0.21 ha, so in total, cultivated land accounted for 0.77 ha, or 0.1 % of total land cover. To create the high-density garden scenario, a total of 500 backyard gardens and 17 larger urban farms were placed onto the landscape, so that cultivated land accounted for 3.88 ha, or 0.5 % of total land cover (Fig. 1). This scenario accounts for potential expansion in urban food cultivation and approximates the garden density documented in south Chicago (Taylor and Taylor Lovell 2012). As our focus is on food-producing gardens, we do not explicitly consider compost inputs to ornamental gardens and landscaping, but the high-density garden scenario may also be representative of a broader range of compost inputs to the urban landscape.

2.2. Model description and parameterization

The InVEST NDR model (version 3.10.2) simulates long-term, steady-state nutrient flows in a landscape. Each pixel is characterized by (1) an areal nutrient input, which is a function of land cover, and (2) a nutrient delivery ratio (nutrient export as fraction of nutrient inputs), which is a function of both the upslope area and the nutrient retention efficiencies of land cover types along the downslope flowpath (Sharp et al. 2020). Total nutrient inputs and export are calculated for a defined area.

For the 11 non-agricultural land-cover types, we used values for nutrient input, retention efficiency, and critical flowpath length that were previously parameterized for the Twin Cities area by Lonsdorf et al. (2021) (Table 1). Those nutrient input rates were taken from published estimates (Hobbie et al. 2017) or back-calculated based on published nutrient efficiencies and watershed-scale monitoring studies. Minimum N input values of 14.5 kg N/ha/y reflect background N deposition. Phosphorus inputs to vegetated categories primarily include pet waste, weathering, and atmospheric deposition, with contribution from fertilizers assumed negligible due to a statewide restriction on P-based lawn

...ut values correspond to median input values of N and P that Saint Paul gardeners reported applying to their gardens as compost, with low-input and high-input values corresponding to first and third quartiles, respectively (Small et al., 2019a). We considered a wide range of possible N and P nominal retention efficiencies, ranging from a maximum of 95 % (based on our previously documented N and P loss rates from leachate) to a minimum of 2.5 % (assuming that all excess N and P is ultimately exported from the garden as runoff). We also included simulations with nominal nutrient retention values of 50 % and 75 %. Parameter values for the 25 model runs are show in in Table A1.

The Threshold Flow Accumulation (number of upstream pixels that must flow into a pixel before it is classified as a stream, which the model assumes provides routing to the watershed outlet with no further nutrient retention) was set at 500 m^2 in these simulations, following Lonsdorf et al. (2021). The Critical Flowpath Length for a given land use category (the distance beyond which that LULC type retains the nutrient at its maximum capacity) also followed parameter values used by Lonsdorf et al. (2021). A sensitivity analysis was conducted using three different Threshold Flow Accumulation values (250 m^2, 500 m^2, and 1000 m^2) and for three different Critical Length values for grass/shrubs and farms/gardens (5 m, 20 m, 25 m).

2.3. Model output interpretation and model assumptions

Combining garden densities (no gardens, baseline density, and high density), garden compost input rates (low, medium, and high), and a wide range of nutrient retention efficiencies (2.5 %, 50 %, 75 %, 95 %), we created 25 scenarios with the goal of examining the relative effect of these parameters to bracket the potential contributions of garden compost inputs to urban nutrient export.

Due to the lack of information on subsurface flowpaths in urban soils, we followed Lonsdorf et al. (2021) in modeling surface flow only. As a result, these simulations only account for nutrient loss occurring via surface runoff. Flowpaths here were based on surface topography using a 1 m resolution digital elevation model, and therefore do not include flows through storm sewers. Although the within-watershed routing of stormwater flows may not be realistic, the model tended to identify roadways as "stream channels". Since these "channels" provide no additional nutrient retention (much like roadways and storm sewers), any flowpaths reaching streets and alleys will not provide additional nutrient retention, and therefore the model should still give a reasonable estimate of total surface nutrient export from the urban watershed. The model is also based on steady-state nutrient fluxes, and so does not account for gradual saturation of nutrient retention along flowpaths, or changes in precipitation frequency or intensity that could alter nutrient

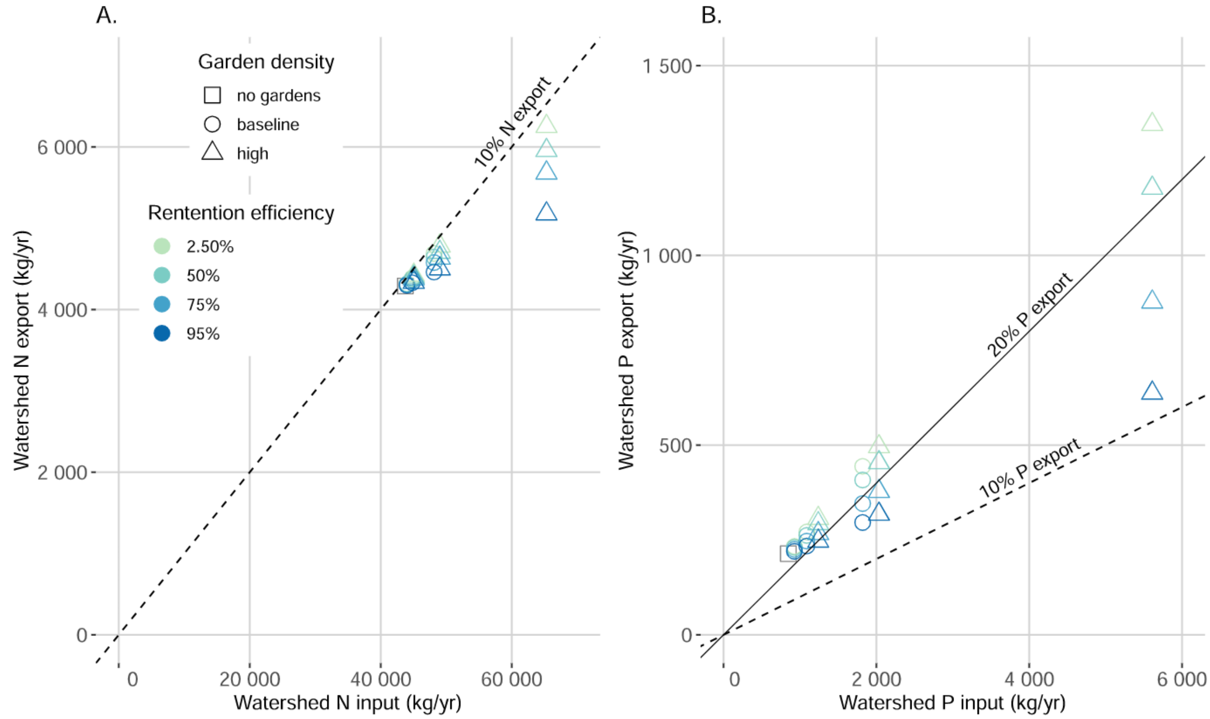

Fig. 3. Relationship between inputs and export of nitrogen (A) and phosphorus (B) for the 25 garden scenarios. Values are reported for the 737-ha study site in Saint Paul Minnesota. Garden density (distinguished by shape) combine with compost input rates to determine N or P input. Input rates combined with maximum nutrient retention efficiency (shown by color) determine watershed N or P export. Ratios between watershed input and export rates are shown as dashed (10%) or solid (20%) lines.

Table 2
Percent increase in N export relative to no-garden scenario, for gardens at different spatial densities, compost input levels, and N retention efficiency (NRE) levels.

Garden density	Compost inputs	NRE 95 %	NRE 75 %	NRE 50 %	NRE 2.5 %
baseline	low	0.2	0.4	0.5	0.6
high	low	1.1	2.1	2.6	3.1
baseline	medium	0.9	1.7	2.1	2.3
high	medium	4.9	11.9	8.0	11.5
baseline	high	4.0	6.7	8.3	9.3
high	high	20.7	32.4	38.9	45.8

Table 3
Percent increase in P export relative to no-garden scenario, for baseline garden density, at three different compost input levels and four P retention efficiency (PRE) levels.

Garden Density	Compost inputs	PRE 95 %	PRE 75 %	PRE 50 %	PRE 2.5 %
baseline	low	2.8	5.1	7.5	8.9
high	low	15.4	24.8	36.9	43.4
baseline	medium	9.3	15.4	22.4	26.6
high	medium	48.6	76.6	112.1	131.8
baseline	high	38.3	61.7	90.7	107.5
high	high	197.7	309.8	450.0	528.5

export. Even so, this model provides a useful first step in quantifying the potential contribution of an under-studied land use category (urban agriculture) to watershed nutrient export.

3. Results

3.1. The importance of gardens on watershed nutrient inputs

Across the scenarios we considered, inputs of N and P from compost to the urban watershed varied widely, with a higher relative contribution to the urban landscape P budget due to low background P inputs. Nitrogen inputs in the no-garden scenario are 43,754 kg N/year across the 737-ha study area (59.4 kg N/ha/y). Relative to the no-garden scenario, compost inputs at baseline garden density represent an increase of 0.6 % (low input), 2.4 % (medium input), and 10.0 % (high input) to total urban landscape N inputs. At high garden density, compost N inputs account for a 2.9 % (low input), 12.0 % (medium input), and 49.3 % (high input) increase in N inputs relative to the no-garden scenario (Fig. 2).

ario are 860 kg P/year

across the 737-ha study area (1.1 kg P/ha/y). At baseline garden density, compost inputs represent an increase of 9.4 % (low input), 28.6 % (medium input), and 115 % (high input) to the urban landscape. At high garden density, compost inputs represent an increase of 49.3 % (low input), 140 % (medium input), and 562 % (high input) in P input relative to the no-garden scenario (Fig. 3).

3.2. Nitrogen export

While gardens can be significant sources of N export, lawns located close to streets were the largest contributor to urban N export (Fig. 2). At each N input rate (determined based on garden density and compost input rate), N export varied according to maximum N retention efficiency (Fig. 2). Total N export from the 737-ha study area in the no-garden scenario was 4,290 kg N/y (5.8 kg N/ha/y). At baseline garden density, with low compost inputs and at 95 % N retention efficiency, gardens resulted in a very small (0.2 %) increase in landscape N export relative to the no-garden scenario (Table 2), as these gardens export nearly the same amount of N compared to the land uses that they replace (e.g., lawns). However, compost applied at high rates to these gardens

Table A1
Nitrogen and phosphorus input and export for the 25 model scenarios.

Garden density	Compost input	Nutrient Retention Efficiency	Nitrogen Input (kg/yr)	Nitrogen Export (kg/yr)	Phosphorus Input (kg/yr)	Phosphorus Export (kg/yr)
no gardens	–	–	43,754	4,290	847	214
baseline	low	2.50 %	44,002	4,317	927	233
baseline	middle	2.50 %	44,813	4,389	1,089	271
baseline	high	2.50 %	48,139	4,689	1,820	444
baseline	low	50 %	44,002	4,313	927	230
baseline	middle	50 %	44,813	4,378	1,089	262
baseline	high	50 %	48,139	4,645	1,820	408
baseline	low	75 %	44,002	4,308	927	225
baseline	middle	75 %	44,813	4,361	1,089	247
baseline	high	75 %	48,139	4,578	1,820	346
baseline	low	95 %	44,002	4,298	927	220
baseline	middle	95 %	44,813	4,330	1,089	247
baseline	high	95 %	48,139	4,461	1,820	296
high	low	2.50 %	45,044	4,423	1,239	307
high	middle	2.50 %	49,017	4,782	2,034	496
high	high	2.50 %	65,305	6,253	5,610	1,345
high	low	50 %	45,044	4,400	1,239	293
high	middle	50 %	49,017	4,705	2,034	454
high	high	50 %	65,305	5,957	5,610	1,177
high	low	75 %	45,044	4,378	1,239	267
high	middle	75 %	49,017	4,633	2,034	378
high	high	75 %	65,305	5,680	5,610	877
high	low	95 %	45,044	4,336	1,239	247
high	middle	95 %	49,017	4,501	2,034	318
high	high	95 %	65,305	5,177	5,620	637

results in a 4.0 % increase in N export form the landscape. Reducing the N retention efficiency to 50 % results in an 8.3 % increase, and N retention of only 2.5 % results in a 9.3 % increase in N export (Table 2).

High garden density with low compost inputs and a 95 % N retention efficiency resulted in 1.1 % more N export from the landscape relative to the no-garden scenario. Increasing compost inputs to high levels resulted in a 20.7 % increase in landscape N export relative to the no-garden scenario. Reductions in N retention efficiency (especially from 95 % to 75 %) resulted in relatively large increases in N export (Table 2). A high garden density with high compost inputs and very low N retention efficiency resulted in a 45.8 % increase in N export relative to the no-garden scenario.

3.3. Phosphorus export

Because of relatively low background P export (Fig. 2), watershed P export was highly sensitive to changes in P inputs (based on garden density and compost inputs) and to changes in maximum P retention efficiency. Total P export from the 737-ha study area in the no-garden scenario is 214 kg P/y (0.3 kg P/ha/y). Low compost inputs coupled with a 95 % P retention efficiency results in a 2.8 % increase in P export. Baseline garden density with high compost inputs and 95 % P retention results in a 38.3 % increase over the no-garden scenario, and high compost inputs at 2.5 % P retention results in a 107.5 % increase in total P export (Table 3). Total P export is more sensitive to the reduction from 95 % to 75 % P retention efficiency than to subsequent decreases (Fig. 3). The high garden-density scenario amplifies these observed effects. Low compost inputs combined with high P retention efficiency results in a 15.4 % increase in P export, whereas high compost inputs and low P retention efficiency result in a 528.5 % increase in P export relative to the no-garden scenario.

3.4. Sensitivity analyses with other model parameters

Changes to the threshold flow accumulation parameter, which defines the beginning of streams below which no further nutrient uptake take place, has a greater effect on N export, which is predominantly from

t, where P sources (urban

gardens set back from

streets. For the baseline garden density, medium compost application rate scenario at 75 % nutrient retention, reducing the threshold flow accumulation from 500 m^2 to 250 m^2 results in a 14 % increase of N export and a 4 % increase in P export. Increasing threshold flow accumulation from 500 m^2 to 1000 m^2 reduces N export by 14 % and P export by 3 % (Table A2).

The critical length parameter determines the number of pixels of a given land use type along a flow path before the maximum retention efficiency for that land use type is achieved. Watershed N export is sensitive to the critical length for grass, whereas P export is sensitive to the critical length for gardens. For the baseline garden density, medium compost application rate scenario at 75 % nutrient retention, decreasing the critical length for grass/shrubs from 20 m to 5 m results in a 16 % reduction in N export and a 2.8 % reduction in P export, whereas increasing this critical length from 20 m to 25 m results in a 3.6 % increase in N export and a 0.4 % increase in P export. Decreasing the critical length for farms or gardens from 20 m to 5 m results in a 0.2 % reduction in N export and a 3.6 % reduction in P export, whereas increasing the critical length from 20 m to 25 m results in a very small (<0.5 %) increase in N and P export.

4. Discussion

Our results indicate that compost application to urban gardens and farms has the potential to significantly alter P budgets for an urban residential landscape. At baseline garden density with medium compost inputs (our best estimate of what is currently occurring in this landscape), compost P inputs to gardens account for an estimated 29 % of P inputs to the urban landscape concentrated on 0.1 % of the land area. Depending on P retention efficiency, P loss from gardens under this scenario could result in anywhere from a 9.3–26.6 % increase in P export, relative to this same landscape in the absence of gardens. Increasing the density of gardens in the landscape, or the compost inputs to individual gardens, has significant implications for landscape P inputs and (under most scenarios) P export. By contrast, garden compost inputs and losses are smaller relative to other N fluxes in the urban landscape, with the notable exception of high garden density coupled with high compost inputs. These results illustrate that the relative importance of compost inputs to urban nutrient export depends greatly on other

Table A2
Sensitivity analysis with NDR model parameters. Sensitivity analyses were conducted using the baseline garden scenario, with medium compost input, and a maximum N and P retention value of 7 %. The Threshold Flow Accumulation (number of upstream pixels that must flow into a pixel before it is classified as a stream, with no nutrient retention), which was set at 500 m^2 in our reported simulations, was varied to 250 m^2, 750 m^2, and 1000 m^2. The Critical Length for a given land use category is the distance beyond which that LULC type retains the nutrient at its maximum capacity. Distances shorter than the Critical Length have a lower than maximum retention, based on an exponential function. In our reported simulations, Critical Length for gardens and for grass (which typically separates gardens from impervious surfaces) was set at 20 m, based on the parameter value used in Lonsdorf et al. (2021). We compared critical length values of 5 m, 10 m, and 25 m for both of these LULC types.

Sensitivity analysis	N export (kg/y)	P export (kg/y)
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, garden Critical Length = 20 m, grass Critical Length = 20 *m*	4 361	247
Baseline garden density, medium compost, 75 % nutrient retention, **Threshold Flow Accumulation = 250 m^2**, garden Critical Length = 20 m, grass Critical Length = 20 *m*	4 950	256
Baseline garden density, medium compost, 75 % nutrient retention, **Threshold Flow Accumulation = 750 m^2**, garden Critical Length = 20 m, grass Critical Length = 20 *m*	4 103	245
Baseline garden density, medium compost, 75 % nutrient retention, **Threshold Flow Accumulation = 1000 m^2**, garden Critical Length = 20 m, grass Critical Length = 20 *m*	3 865	240
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, garden Critical Length = 20 m, **grass Critical Length = 5 *m***	3 631	240
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, garden Critical Length = 20 m, **grass Critical Length = 10 *m***	3 935	243
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, garden Critical Length = 20 m, **grass Critical Length = 25 *m***	4 517	248
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, **garden Critical Length = 5 m,** grass Critical Length = 20 *m*	4 351	238
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, **garden Critical Length = 10 m,** grass Critical Length = 20 *m*	4 355	241
Baseline garden density, medium compost, 75 % nutrient retention, Threshold Flow Accumulation = 500 m^2, **garden Critical Length = 25 m,** grass Critical Length = 20 *m*	4 363	248

nutrient inputs to the landscape. It is notable that a statewide restriction in household lawn P fertilizer application since 2004 has amplified the relative importance of other inputs such as pet waste (Hobbie et al. 2017) and, as shown here, compost application. If lawn P fertilization were common, total P export would be much higher and likely would be dominated by lawns in close proximity to impervious surfaces, as we found for N (Fig. 2A).

[illegible], Minnesota, results are likely applicable to other urban areas. We note that our high-density garden scenario (0.5 % of land use) is equivalent to garden density documented in south Chicago (Taylor and Taylor Lovell 2012) and is much lower than the 3.6 % of land area in Montreal currently under cultivation (Metson and Bennett 2015). Similarly high compost inputs to gardens have also been documented in Chicago (Taylor & Lovell, 2015) and Montreal (Metson and Bennett 2015). Other recent studies have examined local (Grewal and Grewal 2012, Mcdougall et al. 2020) or global (Clinton et al. 2018) land use scenarios involving a substantial expansion of land dedicated to urban food production that would far exceed our high-density garden scenario. One interesting historic comparison is with the highly productive market gardens of Paris in the late 19th Century, which covered one-sixth of the city's area and produced more than 100,000 tons of salad crops annually from the application of 900,000 tons of horse manure (Stanhill 1977). The annual input rates of 400 tons P across 14 km^2 of cultivated land (260 kg P/ha) are similar to P application rates in our medium compost input scenarios, but in the Paris system, 10 % of the added P was recovered by salad crops and an additional 84 % of P inputs were recovered in the spent growing media ("terreau"), which was removed from gardens each year and sold as a valuable by-product of this system (Stanhill 1977). This physical removal of a large fraction of the excess P from these urban gardens would have limited the rate of P buildup and loss per ha of garden area, although the vast extent of these market gardens would have still created the potential for significant total P export. We also note that, while our analysis focused on land used for urban food production, the implication of these results extend to any urban land on which compost is applied on a regular basis, including ornamental gardens (Gouin 1998; Estévez-Schwarz et al., 2013). Therefore, the baseline garden density may underestimate the current urban land area that receives compost inputs and is subject to nutrient export through leachate.

While it is not surprising that nutrient retention efficiencies have an important influence on nutrient loss, these results underscore a larger question about the ultimate fate of excess nutrients applied to residential landscapes, and in particular, to urban gardens and farms. In an analysis of the P budget for the Minneapolis-St. Paul metropolitan region, Baker (2011) estimated that P buildup in soils to be 0.66 Gg/y across the 3011 km^2 urban watershed, accounting for the fate of 16 % of total P inputs to this urban watershed. This estimated P accumulation rate in soils for the larger watershed corresponds to annual inputs of 1600 kg P over our 737 ha highly urbanized section of this watershed, well within the range of these simulations (Fig. 3). In the same study (Baker 2011), stream P export was estimated to be 0.11 Gg/y, 10 % of the magnitude of P buildup in soils. Whether, or for how long, P can continue to accumulate in urban soils without increasing stream export is a question that is of fundamental importance for urban water quality, especially in cities such as Minneapolis and St. Paul, with an abundance of P-sensitive urban lakes.

The extremely heterogenous spatial application of this P to the urban landscape (with P-inputs ranging from 0.49 to greater than 100 kg P/ha/y; Table 1) creates the potential to saturate the soil's capacity for P retention and lead to increased P losses. One limitation of the NDR model is that it is based on annual steady state fluxes and assumes "retention" to be a fixed parameter, but the very process of retention entails increases in storage at various timescales along hydrologic flowpaths (Sharpley et al., 2013), and as this storage increases, the capacity for additional storage diminishes. We have previously documented P accumulation in native soils below raised-bed gardens within seven years of establishment (Small et al., 2019b), and P accumulation in soils along surface flowpaths may also be likely to gradually decrease retention. These temporal dynamics are compounded by the spatial heterogeneity in garden density, compost input rates, and other garden management practices (e.g., irrigation), and the highly engineered hydrology in urban landscapes. Hydrologic flowpaths tend to be relatively short in urban watersheds due to high connectivity of the landscape to streets and storm sewers (Shuster et al. 2005), reflected in our

Fig. A1. Land use in the 737-ha study area, with the baseline-garden density scenario represented.

parameterization of the model (Lonsdorf et al. 2021), so retention along flowpaths is likely to be small (Walsh et al. 2005). It is unlikely that any one nutrient retention efficiency value is correct; rather, we can antic- [illegible] ecrease over time as the [illegible] such, it is notable that our simulations showed that relatively small decreases in P retention efficiency (from 95 % to 75 %) could result in large increases in P export.

Another limitation of the NDR model is that we did not explicitly consider subsurface flows, even though the nutrient losses we have previously documented from experimental raised-bed garden plots

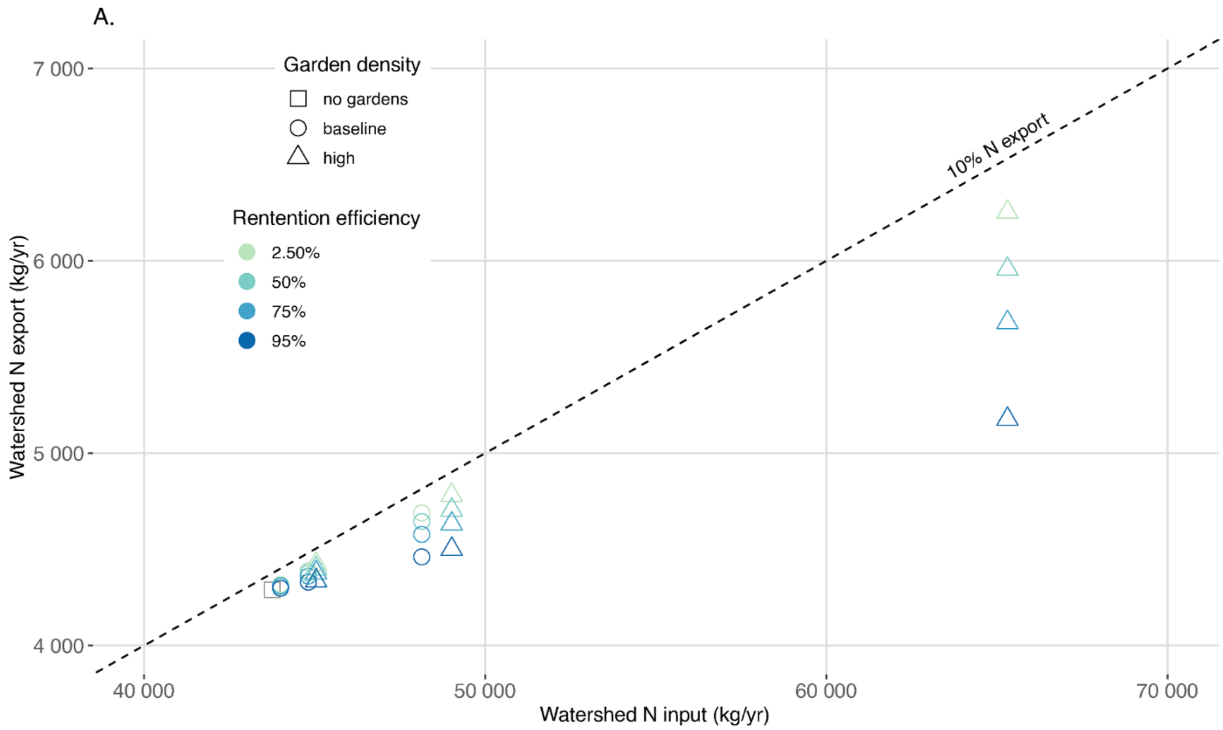

Fig. A2. re-scaled version of Fig. 3A, allowing for differentiation among data points.

(Small et al. 2018, Shrestha et al. 2020) have come from leachate captured in pan lysimeters. A previous analysis of nutrient export for the Twin Cities found that P loss was dominated by stormwater drainage (Hobbie et al. 2017), which included baseflow dominated by groundwater inputs in some larger watersheds (Janke et al. 2014). While the model allows for the simulation of subsurface nutrient retention (Sharp et al. 2020), we followed the approach used by Lonsdorf et al. (2021) in constraining the analysis to losses occurring via surface flows, because of the high uncertainty inherent in parameterizing subsurface processes in an urban watershed. Nutrients exported from gardens as leachate may have a much slower rate of movement (controlled by factors such as soil sorption kinetics and rate of groundwater flow) relative to nutrients transported in surface runoff, which may have more direct access to storm sewers. An analogous issue has been documented in rural row-crop agriculture, where the buildup of organic N in agricultural soils in the Mississippi River Basin will lead to a biogeochemical lag time of 35 years (Van Meter et al. 2016). Because the NDR model assumes steady-state dynamics, faster or slower pathways of loss would have the same effect on the modeled nutrient budget of the surficial urban landscape.

This NDR model is a useful first step in estimating the possible range of nutrient mass loss from the urban landscape, but understanding the impact on a particular water body is a more complicated question, requiring an understanding of time lags along hydrologic flow path, and nutrient limitation in a particular water body. Compared to average nutrient export rates from rural agriculture (14.2 kg N/ha/y and 2.2 kg P/ha/y; Harmel et al. 2008), the export of compost-derived N and P from urban gardens relative to the total area of the urban watershed is fairly low (0.024–1.204 kg N/ha/y; 0.008–1.535 kg P/ha/y). However, Minnesota lakes in urban and suburban watersheds are commonly co-limited by N and P (Bratt et al. 2020), and any additional nutrient loading could undermine efforts to protect nutrient-sensitive lakes, or to remediate already-impaired lakes (Abell et al. 2020). Because of time lags and homogenization of nutrients from different sources along flowpaths, it would be challenging to empirically determine the ... P export in urban water- ... d surface runoff along flowpaths downslope of potential nutrient sources (Lautz et al. 2020) would be a start, as would direct nutrient export measurements from hydrologically well-defined urban farms such as rooftop farms (Harada et al. 2018).

5. Conclusions

The results of this analysis illustrate the potential for urban gardens and farms to contribute significantly to nutrient export from the urban landscape, despite a small spatial footprint. We also acknowledge the numerous other social and environmental benefits of urban gardens and farms (Nogeire-McRae et al., 2018), and we emphasize that excessive nutrient loss is not inevitable. Compost application targeted to meet crop nutrient demand results in relatively high nutrient use efficiency and low nutrient loss via leachate (Shrestha et al. 2020). Many urban gardeners are motivated by sustainable food production and the environmental benefits of gardening (McDougall et al., 2019), suggesting that educational tools and support structures to reduce compost inputs to avoid P over-application could be effective. However, this underscores the challenge of how to use the increasingly large amounts of compost that are generated as cities recycle larger fractions of the organic waste stream. Careful consideration should be given as to how this material can be used without exacerbating urban nutrient pollution.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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Appendix

References

Abdulkadir, A., Leffelaar, P. A., Agbenin, J. O., & Giller, K. E. (2015). Nutrient flows and balances in urban and peri-urban agroecosystems of Kano, Nigeria. *Nutrient Cycling in Agroecosystems, 95*, 231–254.

Abell, J. M., Özkundakci, D., Hamilton, D. P., & Reeves, P. (2020). Restoring shallow lakes impaired by eutrophication: Approaches, outcomes, and challenges. *Critical Reviews in Environmental Science and Technology, 52*, 1199–1246. https://doi.org/10.1080/10643389.2020.1854564

Baker, L. A. (2011). Can urban P conservation help to prevent the brown devolution? *Chemosphere, 84*, 779–784. https://doi.org/10.1016/j.chemosphere.2011.03.026

Benez-Secanho, F. J., & Dwivedi, P. (2019). Does quantification of ecosystem services depend upon scale (resolution and extent)? A case study using the InVEST Nutrient Delivery Model in Georgia. *United States. Environments, 6*, 52. https://doi.org/10.3390/environments6050052

Bratt, A. R., Finlay, J. C., Welter, J. R., Vculek, B. A., & Van Allen, R. E. (2020). Co-limitation by N and P characterizes phytoplankton communities across nutrient availability and land use. *Ecosystems, 23*, 1121–1137.

Clinton, N., Stuhlmacher, M., Miles, A., Aragon, N. U., Wagner, M., Georgescu, M., Herwig, C., & Gong, P. (2018). A global geospatial ecosystem services estimate of urban agriculture. *Earth's Future, 6*, 40–60. https://doi.org/10.1002/2017EF000536

Cofie, O. O., Veenhuizen, R. V., & Dreschel, P. (2003). Contribution of urban and peri-urban agriculture to food security in sub-Saharan Africa. Africa session of 3rd WWF, 17 March, 2003, Kyoto.

Estévez-Schwarz, I., Seoane, S., Núñez, A., & López-Mosquera, M. E. (2013). Characterization and evaluation of compost utilized as ornamental plant substrate. *Compost Science & Utilization, 17*, 210–219.

Gouin, F. R. (1998). Using compost in the ornamental horticulture industry. In *Beneficial Co-Utilization of Agricultural, Municipal and Industrial by-Products* (pp. 131–138). Dordrecht: Springer.

Grewal, S. S., & Grewal, P. S. (2012). Can cities become self-reliant in food? *Cities, 29*, 1–11. https://doi.org/10.1016/j.cities.2011.06.003

Han, B., Reidy, A., & Li, A. (2021). Modeling nutrient release with compiled data in a typical Midwest watershed. *Ecological Indicators, 121*, Article 107213. https://doi.org/10.1016/j.ecolind.2020.107213

Harada, Y., Whitlow, T. H., Templer, P. H., Howarth, R. W., Walter, M. T., Bassuk, N. L., & Russell-Anelli, J. (2018). Nitrogen biogeochemistry of an urban rooftop farm. *Front. Ecol. Evol.* https://doi.org/10.3389/fevo.2018.00153.

Harmel, D., Qian, S., Reckhow, K., & Casebolt, P. (2008). The MANAGE database: Nutrient load and site characteristic updates and runoff concentration data. *Journal of Environmental Quality, 37*, 2403–2406. https://doi.org/10.2134/jeq2008.0079

Heckrath, G., Brookes, P. C., Poulton, P. R., & Goulding, K. W. T. (1995). Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk Experiment. *Journal of Environmental Quality, 24*, 904–910.

Hobbie, S. E., Finlay, J. C., Janke, B. D., Nidzgorski, D., Millet, D. B., & Baker, L. A. (2017). Contrasting nitrogen and phosphorus budgets in urban watersheds and implications for managing urban water pollution. *Proceedings of the National Academy of Sciences, 114*(16), 4177–4182. https://doi.org/10.1073/pnas.1618536114

Host, T. K., Rampi, L. P., & Knight, J. F. (2016). Twin Cities Metropolitan Area 1-Meter Land Cover Classification (Urban Focused). https://doi.org/10.13020/D6959B.

International Plant Nutrition Institute (IPNI). (2010). Nutrient Use Geographic Systems for the US 30-3270 (Norcross, GA: International Plant Nutrition Institute).

Janke, B. D., Finlay, J. C., Hobbie, S. E., Baker, L. A., Sterner, R. W., Nidzgorski, D., & Wilson, B. N. (2014). Contrasting influences of stormflow and baseflow pathways on nitrogen and phosphorus export from an urban watershed. *Biogeochemistry, 121*, 209–228.

Kleinman, P. J. A., Allen, A. L., Needelman, B. A., Sharpley, A. N., Vadas, P. A., Saporito, L. S., Folmar, G. J., & Bryant, R. B. (2007). Dynamics of phosphorus transfers from heavily manured coastal plain soils to drainage ditches. *Journal of Soil and Water Conservation, 62*, 225–235.

Kleinman, P. J. A., Sharpley, A. N., McDowell, R. W., Flaten, D. N., Buda, A. R., Tao, L., Bergstrom, L., & Zhu, Q. (2011). Managing agricultural phosphorus for water quality protection: Principles for progress. *Plant and Soil, 349*, 169–182.

Lautz, L. K., Ledford, S. H., & Beltran, J. (2020). Legacy effects of cemeteries on groundwater quality nitrate loads to a headwater stream. *Environmental Research Letters, 15*, Article 125012. https://doi.org/10.1088/1748-9326/abc914

Lintern, A., McPhilips, L., Winfrey, B., Duncan, J., & Grady, C. (2020). Best management practices for diffuse nutrient pollution: Wicked problems across urban and agricultural watersheds. *Environ. Sci. Technol., 54*(15), 9159–9174.

Lonsdorf, E. V., Nootenboom, C., Janke, B., & Horgan, B. (2021). Assessing urban ecosystem services provided by green infrastructure: Golf courses in the Minneapolis-St. Paul metro area. *Landscape and Urban Planning, 208*, Article 104022.

Lowrance, R. R., Leonard, R. A., Asmussen, L. E., & Todd, R. L. (1985). Nutrient budgets for agricultural watersheds in the southeastern coastal plain. *Ecology, 66*(1), 287–296. https://doi.org/10.2307/1941330

McDougall, R., Kristiansen, P., & Rader, R. (2019). Small-scale urban agriculture results in high yields but requires judicious management of inputs to achieve sustainability. *Proceedings of the National Academy of Sciences USA, 116*, 129–134.

McDougall, R., Rader, R., & Kristiansen, P. (2020). Urban agriculture could provide 15% of food supply to Sydney, Australia, under expanded land use scenarios. *Land Use Policy, 94*, Article 104554. https://doi.org/10.1016/j.landusepol.2020.104554

Metson, G. S., & Bennett, E. M. (2015). Phosphorus cycling in Montreal's Food and Urban Agriculture Systems. *PLoS ONE, 10*(3), e0120726.

Nogeire-McRae, T., Ryan, E. P., Jablonski, B. B. R., Carolan, M., Arathi, H. S., Brown, C. S., Saki, H. H., McKeen, S., Lapansky, E., & Schipanski, M. E. (2018). The role of urban agriculture in a secure, healthy, and sustainable food system. *BioScience, 68*(10), 748–759. https://doi.org/10.1093/biosci/biy071

Redhead, J. W., May, L., Oliver, T. H., Hamel, P., Sharp, R., & Bullock, J. M. (2018). National scale evaluation of the InVEST nutrient retention model in the United Kingdom. *Science of the Total Environment, 610–611*, 666–677.

Sharp, R., Douglass, J., Wolny, S., Arkema, K. K., Benhardt, J., Bierbower, W., … Wyatt, K. (2020). InVEST 3.10.0 Users Guide. The Natural Capital Project, Stanford University, University of Minnesota, The Nature Conservancy, and World Wildlife Fund. Retrieved from https://storage.googleapis.com/releases.naturalcapitalproject.org/invest-userguide/latest/index.html.

Sharpley, A., Jarvie, H. P., Buda, A., May, L., Spears, B., & Kleinman, P. (2013). Phosphorus legacy: Overcoming the effects of past management practices to mitigate future water quality impairment. *Journal of Environmental Quality, 42*, 1308–1326.

Shrestha, P., Small, G. E., & Kay, A. (2020). Quantifying nutrient recovery efficiency and loss from compost-based urban agriculture. *PLoS ONE, 15*(4), e0230996.

Shuster, W. D., Bonta, J., Thurston, H., Warnemuende, E., & Smith, D. R. (2005). Impacts of impervious surface on watershed hydrology: A review. *Urban Water J., 2*, 263–275. https://doi.org/10.1080/15730620500386529

Small, G. E., Osborne, S., Shrestha, P., & Kay, A. (2019). Measuring the fate of compost-derived phosphorus in native soil below urban gardens. *International Journal of Environmental Research and Public Health, 16*, 3998. https://doi.org/10.3390/ijerph16203998

Small, G. E., Shrestha, P., & Kay, A. (2018). The fate of compost-derived phosphorus in urban gardens. *International Journal of Design & Nature and Ecodynamics, 13*(4), 415–422. https://doi.org/10.2495/DNE-V13-N4-415-422

Small, G. E., Shrestha, P., Metson, G. S., Polsky, K., Jimenez, I., & Kay, A. (2019). Excess phosphorus from compost applications in urban gardens creates potential pollution hotspots. *Environmental Research Communications, 1*, Article 091007. https://doi.org/10.1088/2515-7620/ab3b8c

Smith, O. B. (2001). *Overview of urban agriculture and food security in West Africa*. Ottawa, Canada: International Development Research Centre.

Smith, V. H., Joy, S. B., & Howarth, R. W. (2006). Eutrophication of freshwater and marine ecosystems. *Limnology and Oceanography, 51*, 351–355. https://doi.org/10.4319/lo.2006.51.1_part_2.0351

Stanhill, G. (1977). An urban agro-ecosystem: The example of nineteenth century Paris. *Agro-Ecosystems, 3*, 269–284.

Swaney, D. P., Howarth, R. W., & Hong, B. (2018). Nitrogen use efficiency and crop production: Patterns of regional variation in the United States, 1987–2012. *Sci. Total Environ., 635*, 498–511.

Taylor, John R., & Lovell, S. T. (2015). Urban home gardens in the Global North: A mixed methods study of ethnic and migrant home gardens in Chicago, IL. *Renewable Agriculture and Food Systems, 30*, 22–32. https://doi.org/10.1017/S1742170514000180

Taylor, J. R., & Taylor Lovell, S. (2012). Mapping public and private spaces of urban agriculture in Chicago through the analysis of high-resolution aerial images in Google Earth. *Landscape and Urban Planning, 108*, 57–70. https://doi.org/10.1016/j.landurbplan.2012.08.001

United States Environmental Protection Agency (EPA). (2020). Advancing Sustainable Materials Management: 2018 Fact Sheet. Retrieved from: https://www.epa.gov/sites/default/files/2021-01/documents/2018_ff_fact_sheet_dec_2020_fnl_508.pdf.

van de Vlasakker, P. C. H., Tonderski, K., & Metson, G. S. (2022). A review of nutrient losses to waters from soil- and ground-based urban agriculture—more nutrient balances than measurements. *Frontiers in Sustainable Food Systems*. https://doi.org/10.3389/fsufs.2022.842930

Van Meter, K. J., Basu, N. B., Veenstra, J. J., & Burras, C. L. (2016). The nitrogen legacy: Emerging evidence of nitrogen accumulation in anthropogenic landscapes. *Environ. Res. Lett., 11*, Article 035014. https://doi:10.1088/1748-9326/11/3/035014.

Van Meter K J, McLeod M M, Liu J, Tenkouano G T, Hall R I, Van Cappellen P and Basu N B (2021) Beyond the Mass Balance: Watershed Phosphorus Legacies and the Evolution of the Current Water Quality Policy Challenge *Water Resources Research* **57** e2020WR029316.

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: Current knowledge and the search for a cure. *Journal of the North American Benthological Society, 24*, 706–723.