BIOGEOCHEMISTRY OF THE BROOKLYN GRANGE, AN URBAN ROOFTOP FARM

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BIOGEOCHEMISTRY OF THE BROOKLYN GRANGE, AN URBAN ROOFTOP FARM

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Intensive agriculture is an emerging theme of green roof technology, aiming to provide fresh and affordable vegetables for local consumption. The intensification of water and N management is likely to increase the discharge volume and N load to sewers, and atmospheric deposition of heavy metals could compromise food safety. This has not been measured in full-scale rooftop farms and there is little information for land use planning to estimate the environmental and economic returns from rooftop farming. This study is located at the Brooklyn Grange, a 0.6-ha operational rooftop farm atop an 11-story building in New York City. I report the Grange's balance for water, N, and heavy metals, and address how this balance could be improved in terms of environmental quality, resource use efficiency, and food safety. I found that the discharge volume to sewers was 1.1X the precipitation, and the Grange was a net source for water in the urban watershed. Depending on crop types, water use efficiency was lower than in-ground intensive vegetable production in dry regions, and over half of irrigation was wasted by drainage. Drainage N output was 11 X the atmospheric bulk deposition, and was 5.4X the estimated total atmospheric N deposition, which makes the Grange a net N source in the urban watershed. Fertilizer N input and N leaching loss from soil were similar, and efficiency of N management can be lower than in-ground intensive vegetable production. Efficiency of N and water management could be improved by increasing the soil water storage within the range of plant available water. Atmospheric deposition of Pb and Mn exceeded drainage outputs, and the Grange was a net sink for these metals in the urban environment respectively. There were unwashed vegetable samples exceeding guideline level for Pb, yet the Pb concentration for human consumption was likely to be lower because vegetables are machine-washed before sales. In order to reduce atmospheric deposition of Pb, it is important to cover the soil with mulch.

BIOGRAPHICAL SKETCH

Yoshiki Harada is a product of rural upbringing in the southwest region of Japan, and his early life was characterized by 2 achievements: being licensed as a first-class professional skier, and reading all architecture books in the largest library of the region. Yoshiki moved to Tokyo and studied in an architecture school, where he felt he had to design strong cities by analyzing maps of natural disasters and topography, because it was only 2 years after the major earthquake destroyed the city of Kobe, which was his favorite destination for family trips. However, Yoshiki could not really analyze maps until he received the formal training in geographic information science at the University of Tokyo. During his 3 years of deep immersion to maps and aerial photographs, Yoshiki led an advanced seminar based on the book "Road Ecology" written by Richard Forman. Further search for the topics of urban ecology made him realize that the design approach he tried to comprehend in the architecture school was called *landscape architecture* in the United States, where *green infrastructure* makes cities beautiful, resilient, and sustainable. During his academic and professional trainings in landscape architecture in Cambridge, MA and New York City, Yoshiki continued to explore his interests in urban ecology, yet felt unsatisfied because he had little idea how urban ecosystems could improve environmental quality. Yoshiki moved to New Haven, CT, and joined experiments with urban ecologists and other environmental scientists in a forestry school. The research team spent so much time trying to figure out how to study hydrologic performance of green roofs, and eventually consulted a scientist from Cornell University, who was performing air particulate sampling around a roadside park in New York City. This scientist was Thomas Whitlow, who suggested that Yoshiki's specific area of research interest might be *urban biogeochemistry*, which was the very beginning of this dissertation research.

I dedicate this work to my wife, Anchalee, and all other members of my family in Japan, Thailand, and the United States for always supporting me, valuing education, and having raised me to believe that unshakable devotion to pursuing my goals will make tomorrow better than today.

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1-1. Approach

Rooftop farming draws on many specialized fields, each with its own terminology as well as different definitions for the same term. As a prime example, one can argue that there is no soil on a roof, so why should there be a chapter on rooftop farming in a volume devoted to soil science? A simple definition of soil that is universally accepted by agriculturalists, geologists, ecologists and engineers is a perennial challenge, yet by calling the material placed on a roof in order to grow plants a *soil*, it is appropriate to follow Jenny's (1941) wise choice of leaving the definition open and inclusive.

Among the many subject areas informing rooftop farming are soil science, biogeochemistry, horticultural science, green roof design and management, and potting soils (Fig. 1). In the biophysical sciences, *soil science* provides the foundation for understanding the physics and chemistry of water and nutrient movement in a rooftop soil while *biogeochemistry* treats these fluxes at the ecosystem scale. In the realm of applied science, production *horticulture* has historically dealt primarily with crops grown in-ground in native soils. In contrast, roofs lack native soil, are disconnected from the underlying subsoil and parent material and are disconnected from upland watersheds. A rooftop farm resembles a *green roof* yet would probably require deeper soil and supplemental irrigation to achieve acceptable yield and quality. A rooftop farm devoted to vegetable production also draws on the technologies developed for *greenhouse production*, including supplemental irrigation and nutrients and the use of artificial soil-less or potting mixes. Since the 1950s an extensive literature has dealt with synthetic soil mixes for greenhouse and nursery production which provides information that could be used to develop soil for extensive outdoor landscapes like green roofs (Baker 1957; Dasberg 1999; Jozwik 2000; Newman 2008). Lastly, *urban*

planning and design lends understanding of how rooftop farming both drives and responds to the complex, coupled human and natural systems of modern cities. This literature is large and expanding rapidly. A recent search of bibliographic data bases found over 1,000 peer reviewed papers relevant to growing plants on roofs, while the intersection of the disciplines contains roughly 100 papers.

This chapter is intended for a wide audience, not just soil scientists. After a review of the context for rooftop farming, the focus is on the influence of soil composition and depth on the water and nutrient budgets of rooftop farms and conclude with a case study from New York City. (Figure 1-1 & 1-2)

Figure 1-1. Related fields of fooftop farming

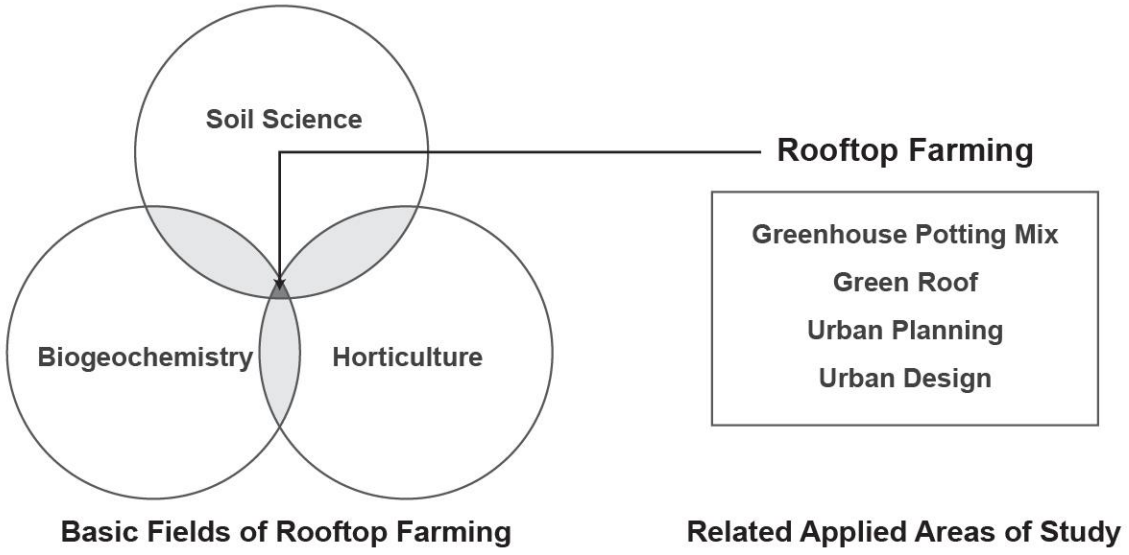
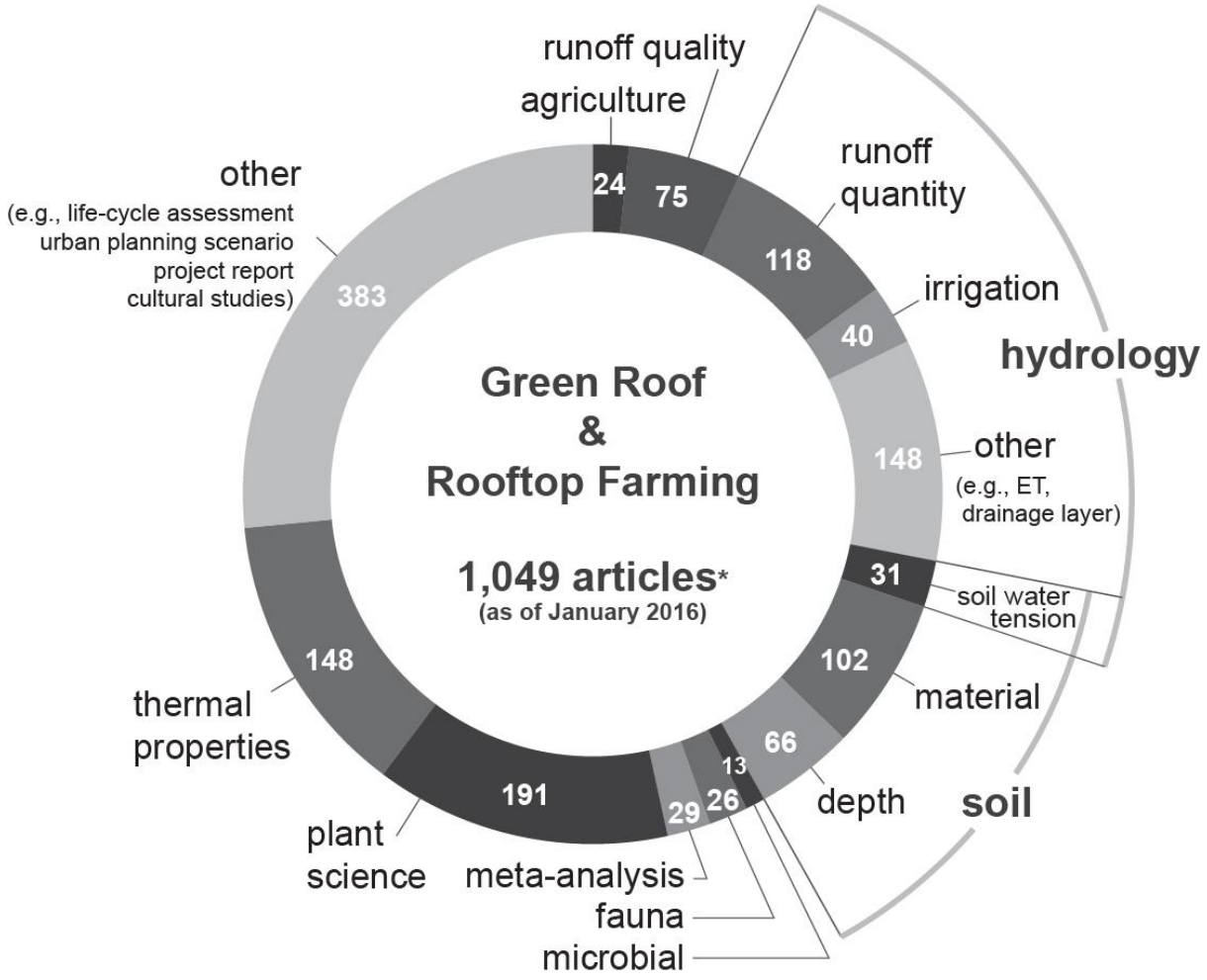


Figure 1-2. Topic groups in green roof & rooftop farming research *based on the Web of Science entry, and each article can be categorized into multiple topic groups.



1-2. Background

Cities are hotspots for biogeochemical cycles, making them ideal locations for developing and testing novel ecosystems to enhance sustainability (Palmer et al. 2004, Grimm et al. 2008). Among these are a wide variety of green infrastructure projects intended to manage stormwater, save energy, and manage waste. Social dimensions to these practices include environmental education, investment, green job employment, eco-justice, food security and building more cohesive communities.

Urban rooftop farms could potentially integrate many ecosystem services. These perceived services necessarily involve regulatory and investment sectors as well as public preferences (Plakias 2016). In this regard, urban rooftop farming could be viewed as one of the most creative components of 21st century planning for sustainable cities. The *"combining"* and *"stacking"* of multiple ecosystem services presents a unique opportunity for cross disciplinary research (Felson et al. 2005, Lovell 2010, Robertson et al. 2014)

In New York City, green infrastructure projects are supported by a 20-year $1.5-billion capital initiative to fund community-based projects across the city (NYC DEP 2010, Bloomberg 2011, De Blasio 2015). In April 2012, a new zoning code allows retrofitting rooftops to include vegetable farms (NYC DCP 2012), prompting an influx of public and private funds into rooftop farms. A prominent example is the Brooklyn Grange, a 70,000-square-foot (0.65 ha) commercial rooftop farm, constructed in 2012 with a $592,730 grant from the Community-Based Green Infrastructure Program of the NYC Department of Environmental Protection (Figure 1-3) (NYC DEP 2011). This grant was partly based on the expectation that the farm would reduce stormwater runoff and the resulting N pollution caused by combined sewer discharges into surface waters.

Figure 1-3. The Brooklyn Grange Farm at Brooklyn Navy Yard, Summer 2015

Since its inception, the Grange has linked to the local community through businesses, schools, non-profit organizations, and underrepresented populations by providing organic vegetables, collecting food wastes for composting, and offering educational and green-job training programs (Plakias 2016). However, the functional environmental performance of rooftop farms has received little attention from the scientific community and there is little quantitative information on the design and operation of rooftop farms from a resource subsidies perspective. Understanding the water and nutrient budgets could lay the foundation for optimizing the environmental, yield and economic return of the farm.

Opportunities. A rooftop is a simple watershed system analogous to the Hubbard Brook experimental watershed. Inputs and outflows of water and nutrients are easiest to measure in watersheds with clear boundaries and topographic gradients, shallow soil overlying impervious rock, and a centralized stream network (Likens 2013). This approach provides a foundation for science to inform policy and practice across cities, thereby allowing comparisons of water and nutrient budgets across geographic locations and with varying degrees of human influence (Howarth et al. 1996). The extended community of scientists involved in the Long Term Ecological Research (LTER) program supported by the US National Science Foundation affords an ideal opportunity extending and generalizing the state of knowledge for applying of knowledge (Fahey et al. 2015).

Challenges. Although the biogeochemical processes of rooftop farms resemble those of forested and agricultural ecosystems, because many components of in-ground ecosystems are either *absent* (e.g., ground water recharge), *simplified* (e.g., soil horizonation), or *replaced with artificial materials* (e.g., soilless media), it is difficult to apply knowledge from other systems directly to rooftops.

Even among the horticultural disciplines that contribute to the emerging practice of rooftop farming, the existing scientific understanding of system performance may not translate directly. For example, extensive green roofs typically use soil media that are designed to be light weight and drain

rapidly to minimize roof loads. Nutrient supply after establishment is also relatively unimportant. While these soils may supply adequate water for slow growing, drought tolerant *Sedum* species typically used on extensive green roofs, they are not optimal for a vegetable production system where yield and quality are important. In order to increase the nutrient and water holding capacity for vegetable cropping, green roof soil is amended with peat, coconut coir, biochar and spent mushroom media which are typical components used in greenhouse potting mixes managed for a single crop cycles. In contrast, rooftop farms use the same soil indefinitely under outdoor conditions with diurnal and seasonal cycles of temperature and moisture. Soils in rooftop farms are further amended with organic fertilizers for vegetable production (kelp meal, blood meal) and food-waste compost, which is low in many essential nutrients found in the native soil.

Ideally, each ingredient of a rooftop soil would contribute to optimizing the water and nutrient budgets. Optimum management of existing farms could aim at the steady state of nutrient budget in the traditional sense of biogeochemistry (Likens 2013), while "enhanced" steady state could be achieved by applying scientific understanding of site characteristics (e.g., substrates, irrigation systems) to improve performance (Fahey et al. 2015). Beyond rooftop farms, similar challenges confront all green infrastructure projects, including green roofs, bioswales, rain gardens and reclaimed urban parcels.

Environmental Quality. New York City alone has about 8,600 ha of flat rooftop surface (Acks 2006), about 25 times the size of the Central Park. Converting even a small fraction of this space to agriculture raises concern about increased N load to surface water bodies draining the city. The appearance of "dead zones" resulting from low oxygen concentration in the Chesapeake Bay and Gulf of Mexico are the legacy of fertilizer runoff from farming practices intended to maximize crop yield (Rabalais et al. 2002, Kemp et al. 2005). In the US, the passage of the Clean Water act in 1970 and its subsequent amendments reflects a growing recognition that management practices can have a

profound influence on downstream water quality. The establishment of Total Maximum Daily Loads (TMDLs) for each governmental jurisdiction in a watershed is intended to eliminate downstream pollution (NYS DEC 2000). The need to comply with increasingly stringent regulations has led to the development of Best Management Practices (BMPs) to reduce both concentration and runoff volume from all land uses (Meals et al. 2010). Had we anticipated the nature and extent of pollution from agricultural runoff in the 1950s, it is conceivable that today's problems could have been avoided. Because urban rooftop farming is in its infancy, there exists an unprecedented opportunity to develop and implement BMPs before problems arise. Soil science is central to engineering soils that satisfy both the concerns of roof bearing capacity and nutrient and water retention. Among the most important soil properties affecting these are depth, composition, and pore size distribution.

1-3. Soil Design

Overview. Over the past decade, reviews of the green roof literature report a wide range of water retention and nutrient loss (Mentens et al. 2006, Berndtsson 2010, Rowe 2011, Li et al. 2014, Driscoll et al. 2015). Reviews of green roof performance include both soil-less (Ampim, et al. 2010) and soil-based media (Best et al. 2015), and irrigation (Van Mechelen et al. 2015). The physical, chemical, and biological properties of the soil are the key factors which relate the design and management to water retention and nutrient leaching from green roofs (Berndtsson 2010, Rowe 2011, Buffam et al. 2015). It has proven difficult to develop standard specifications for rooftop soil that meet the competing demands of vegetable yield and quality, water retention, and nutrient leaching while keeping weight to a minimum. Green roofs with similar design and management vary widely in performance, yet the key information needed to explain such difference is often unknown or unreported. Studies often do not include statistics for both runoff quantity and quality, while coupled studies often lack details of soil composition. Insufficient detail stems in part from the difficulties of defining each component of green roofs soil mixes. Even the standardized industrial-

grade products like expanded shale would require laboratory testing to quantify their physical characteristics (e.g., pore-size distribution), and there are many components that are too variable for exact specification (e.g., composts), especially as their properties change over time in response to field conditions and the management history. Those factors present a challenge to synthesize the knowledge from different studies across disparate climactic zones and cultural contexts.

***Soil Composition*.** Rooftop farms use a variety of soils, including commercial potting mixes (e.g., planter mixes, garden mixes), commercial green roof mixes, experimental mixes using common / novel ingredients for a green roof, and commercial rooftop farming mixes (Table 1-1). Both commercial potting and green roof mixes use organic (e.g., compost, peat) and mineral (e.g., vermiculite, perlite) soilless materials, and both have been used in the studies on rooftop farming, while ESCS (expanded shale, clay, and slate) is the main mineral component typically used in commercial green roof mixes in order to meet the drainage guidelines and weight limitations (Ampim et al. 2010). Naturally-sourced soils (e.g., sand, loam) are sometimes used for green roofs with or without being blended with soilless materials (Best et al. 2015). The review of available literature indicated only one commercial formulation specified for the rooftop farming, Rooflite®, which is reported in the feasibility review of rooftop farming by NYSERDA (2013). It is a blend of heat-treated shale, spent mushroom media, and composts (Kong et al. 2015), and is used in the Brooklyn Grange, a rooftop farm in NYC, that is the subject of the "Case Study" later in this chapter. Given the variety of soils by rooftop farms, it is important to define the components in each study in order to obtain generalizable interpretations of the results.

In a broad sense, interest in rooftop farming is an outgrowth of ecological awareness, including adaptive re-use of waste products. Of 24 rooftop farming studies identified in this review, Grard et al. (2015) studied a rooftop farm in Paris, France that used locally sourced yard waste compost, crushed wood, and coffee grounds in its soil, growing lettuce (*Lactuca sativa*) and tomatoes (*Lycopersicum*

esculentum var. *chery*) with irrigation. This study reports satisfactory yields of both crops and heavy metal levels lower than European standard. However, the original soil depth of 300 mm soil decreased to 100 – 150 mm after the first growing season due to settling and decomposition of organic matter (OM). The volume reduction and consumption of urban wastes is an important ecosystem service, yet the changes in soil depth complicate management and point to the need to replenish OM frequently.

In green roof research, Ampim et al. (2010) reviewed the physical and chemical characteristics of recycled soil ingredients, and emphasize the need of combining soil material research with observations of plant response, and runoff quantity and quality. More recently, an increasing number of studies have reported satisfactory growth of grass, sedum and wildflowers in soils made from recycled construction materials (e.g., bricks, tiles) (Bates et al. 2015, Molineux et al. 2015), paper ash, and bark (Young et al. 2014, Molineux et al. 2015).

Biochar (Cao et al. 2014) and hydrogel products (Olszewski et al. 2010, Savi et al. 2014) have also been tested for their ability to increase the plant available water. Biochar could also improve the water and nutrient retention of soil (Beck et al. 2011), while effects of hydrogel was species-dependent (Farrell et al. 2013). Unlike typical green roofs, which do not produce food, recycled materials used for rooftop farms would need to be tested for toxic residues to insure public health.

Plant Growth Effects. Across a variety of soils, all 7 studies of rooftop farms (Table 1-1) report satisfactory yield in all species except pepper (*Capsicum annuum*) (Whittinghill et al. 2013). All studies used irrigation yet none reported the effect of soil composition on irrigation requirements, although Kong et al. (2015) reported that the addition of composted yard waste to Rooflite® resulted in higher yields of Swiss chard (*Beta vulgaris*) and reduced leaching loss of nitrogen.

Table 1-1. Soil Type, Depth, and Plant Growth in Rooftop Farming Research

Soil Type	Soil Detail	Crops	Yield[*1]	Irrigation[*2]	Soil Depth (mm)	Location	Author
commercial potting mix	Sunshine Mix #4, Sun-Gro Horticulture (55-65% peat, 35-45% perlite)	lettuce chicory[*3]	S	Y	50, 100, 200	Korea (rooftop)	Cho (2008)
commercial potting mix	Sunshine Mix #4, Sun-Gro Horticulture (55-65% peat, 35-45% perlite)	lettuce chicory[*4]	S	Y	150	Korea (rooftop)	Cho et al (2010)
commercial potting mix	Sunshine Mix Fisons (composition unspecified)	Kale[*5]	S	Y	102	VA USA (rooftop)	Elstein et al (2006)
commercial potting mix	1) Terre à planter, Brun brand (topsoil, blond sphagnum peat moss, composted bark, brown peat, horse manure and composted seaweed, ratio unspecified)	lettuce tomatoes	S	Y	300	Paris, France (rooftop)	Grard et al (2015)
experimental mix	2) 100% yard waste compost, underlain by 100 % crushed wood						
experimental mix	3) 100% yard waste compost, underlain by 100 % coffee ground layer, 100 % crushed wood layer						
experimental mix	4) 50 % yard waste compost, 50% crushed wood						
commercial green roof mix	Renewed Earth (50% expanded shale, 35% sand, 15% leaf compost)	Tomatoes[*6] beans[*7] cucumbers[*8] peppers chives[*9] basil[*10]	S (except pepper)	Y	105	MI USA (rooftop)	Whittinghill et al (2013)
experimental mix	expanded shale, sand + (0, 20, 40, 60, 80, 100% yard waste compost)	cucumbers[*8] peppers	S	Y	125	MI USA (rooftop)	Eksi et al (2015)
commercial rooftop farming mix	Rooflite, Skyland (lightweight mineral aggregates, mushroom compost, unspecified organic composted component, ratio unspecified) + 1) yard waste compost, 2) composted poultry manure, 3) vermicompost, 4) controlled release fertilizer	swiss chard	S	Y	110	NY USA (greenhouse)	Kong et al (2015)

[*1] S :satisfactory yield, [*2] Y :irrigated
[*3] *Cichorium intybus* var. *folisum*, [*4] *Cichorium endivia* var. *endivia*, [*5] *Brassica oleracea* var. *acephala*, [*6] *Solanum lycopersicum*, [*7] *Phaseolus vulgaris*, [*8] *Cucumis sativus*, [*9] *Allium schoenoprasum*, [*10] *Ocimum basilicum*

Soil Depth. Among the studies on rooftop farming summarized in Table 1-1, soil depth varied between 50 – 300 mm. Only Cho (2008) specifically tested different soil depths and reported positive but non-significant growth response to deeper soil. It is noteworthy that the manufacturer of Rooflite® specifies minimum depth of 8 inches (≈ 200 mm) (Skyland USA LLC 2016), yet 6 of 7 studies report satisfactory yields with soil less than 200 mm deep.

In addition to the studies of rooftop farms, green roof research includes an additional 60+ studies addressing the effect of soil depth both with and without irrigation. Standard extensive and intensive green roofs typically use drought-tolerant plants which require much less water and nutrients in

comparison to vegetable crops, hence soils designed for green roofs may not be optimal for vegetable production. However, these studies still report important information on soil properties which could reduce the evaporative loss while maintaining plant available water (see "Soil Moisture and ET" below). Ten studies using soil depths ranging between 20 – 400 mm reported that increasing depth increased available water and biomass across a wide range of species including **succulents** (VanWoert et al. 2005, Getter et al. 2009), **dry grassland species** (Dunnett et al. 2008), **turf grass species** (Nektarios et al. 2010, Ntoulas et al. 2012, Ntoulas et al. 2013), **drought-adapted shrubs** (Nektarios et al. 2011, Kotsiris et al. 2012, Savi et al. 2015), and **olive trees** (Kotsiris et al. 2013).

Soil Moisture and ET. In terms of the water budget of a rooftop farm, the ideal soil reduces runoff volume and minimizes the irrigation demand while achieving commercially viable yield and quality of crops. To this goal, it is important to balance ET demand and soil water holding capacity. In a Mediterranean climate, Nektarios et al. (2011) grew *Dianthus fruticosus* in either 750mm or 150 mm of pumice-based soil with irrigation. In this field experiment, shallow soil had higher evaporative losses presumably because the zone of capillary rise was closer to the soil surface, therefore both the diffusive resistance of the soil was less and the temperature gradient between the soil surface and capillary water was steeper. Pore size distribution and soil depth are key to controlling evaporative losses.

In a controlled greenhouse simulation of spring and summer conditions (Sheffield UK, average temperature 7.1 and 16.7 °C respectively) (Poë et al. 2015), ET was positively correlated with the water holding capacity of the soil, and was greatest in the soil with the highest OM and fine mineral fractions whose VWC was 25% at 33 kPa.

In a greenhouse simulation of summer conditions (Subtropical, New Zealand, average air temperature 22°C), Voyde et al. (2010) monitored ET of *Sedum mexicanum* and *Disphyma australe*, 2 drought hardy succulents, grown in 70 mm of pumice, zeolite, and compost. Over the first 9 days

of the experiment, ET from planted treatments was consistently higher than the evaporative loss from bare soil. After day 17 when ET essentially ceased due to the drought stress, both treatments lost equal amounts of water. In a rooftop farm, transpiration in the absence of water stress could be much higher because of the large leaf area, and could decline more rapidly as vegetables close their stomates in response to drought.

Water Retention. SWT (soil water tension) plays important roles in the water retention characteristics of a soil, including the field capacity, plant available water, and the effects of porosity and self-mulching. Compared to greenhouse cropping systems, management of water is more important outdoors because of diurnal and seasonal variation in water supply and demand experienced in the field. Therefore, SWT of green roof and rooftop faming mixes are much less articulated or understood, containing large amount of soilless mixes typical to greenhouse cropping system. In addition, the low range of tension (< 10 kPa) is particularly important for rooftop systems, because the soil depth is much shallower (< 500 mm) than the in-ground systems (> 1000 mm). Also, meta-analysis on SWT-based irrigation criteria reports 6 - 10 kPa for the mustard greens, collard, and leaf lettuce in sandy loam (Shock et al. 2011), and such leaf greens are important crops for rooftop farming. Therefore, for the effective management of water in rooftop farming, it is important to understand the water of soilless mixes in low range of tension (< 10 kPa).

SWT is sensitive to the composition of soil mixes, and it is difficult to characterize the SWT of soil mixes solely based on SWT of each ingredient, yet the key effect of each material could be generalized to some extent. Within the tension range between 2 and 10 kPa, mixes of pumice and thermally treated clay showed 2 – 7 % higher VWC (volumetric water content), in comparison to the mixes of crushed tile and pumice, which showed VWC between 15 and 20% (Ntoulas et al. 2015). Organic soilless materials and naturally-sourced soil could also change VWC. Below 10 kPa, peat often holds more water than compost at the volumetric addition of 15 and 20% (Ntoulas et al. 2012,

Ntoulas et al. 2013, Ntoulas et al. 2015), yet compost was more effective at the increased addition rates of 30% (Kotsiris et al. 2012, Kotsiris et al. 2013). Types of the compost also have influence on the moisture contents (Ntoulas et al. 2015). In the mixture of pumice, compost, and zeolite, the 15% volumetric addition of sandy loam increased the VWC above 6.5 kPa tension, while mixes without sandy loam had higher VWC below 2 kPa (Nektarios et al. 2011). Adding 30% sandy loam to peat or compost treatments reduced the VWC above 3 kPa tension (Ntoulas et al. 2013). In order to design the SWT, it is important to establish the critical composition of each material.

1-4. Runoff Volume

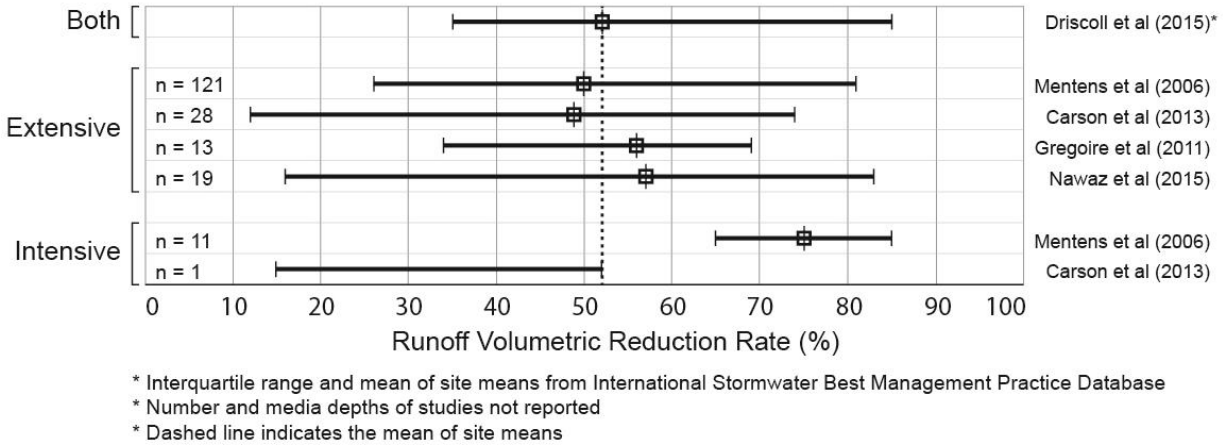
The effect of particle size distribution on water retention and drainage is well understood and relatively easy to take for granted in native mineral soils, as are their effects on plant growth. Soils retain the maximum amount of available water in the texture class known as silt loam, where particle size classes, and hence pore sizes, are more or less evenly distributed. However, soils intended for rooftop farms often lack the familiar sand, silt and clay size fractions and when organic materials(e.g., compost , biochar ,highly modified skeletal components like expanded shale) are substituted for these mineral components, designing an optimal soil for rooftop farming becomes quite challenging.

In rooftop farms, surface runoff would not occur under normal conditions, and “runoff” as used in this context indicates the export of drainage water following the lateral flow through the soil, underlying drainage layer and along the impervious roof membrane. Rooftop farms are irrigated and it is important to differentiate “volumetric runoff reduction” from “water budget”, which includes all fluxes of water including irrigation, while runoff reduction only compares precipitation to runoff. Irrigation can produce discharge. In green roofs and rooftop farms, irrigation could produce base

flow if it is frequent and maintains water content near field capacity. Elstein et al. (2006) report the volumetric runoff reduction of a rooftop farm of 69.2%, while Whittinghill et al. (2015) report a reduction over 85% for 0 - 10 mm of precipitation, and below 60% for precipitation events over 10 mm. The study does not report the overall volumetric reduction based on the cumulative precipitation and drainage during the entire study period.

In studies of non-production green roofs, over 100 studies report volumetric runoff reduction, and 5 reviews show large variation across individual studies between 12 – 85% (Figure 1-4) (Carson et al. 2013; Gregoir and Clausen 2011; Mentens et al. 2006; Nawaz et al. 2015; Driscoll et al. 2015). This wide variation defies simple generalizations applicable across climate zones, soils and plant cover types, but the average reduction is around 50%.

Figure 1-4. Runoff volumetric reduction of extensive and intensive green roofs

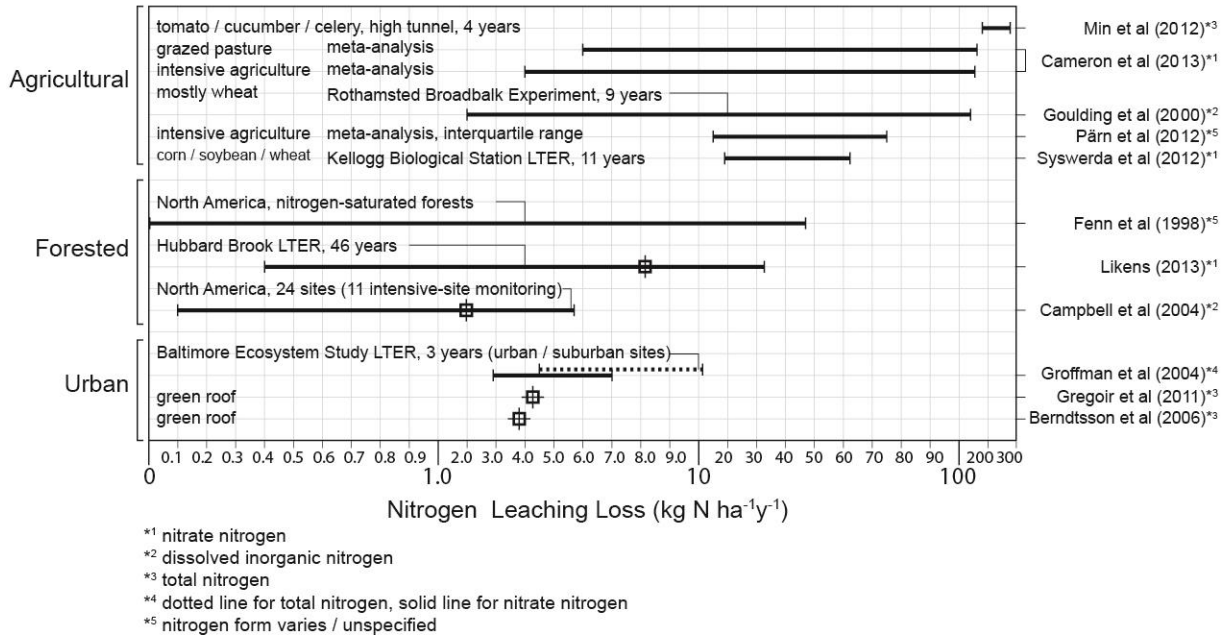


1-5. Runoff Quality

Overview. Through industry, agriculture, and urbanization, human activity is a major driver of the global N cycle, affecting the form and function of ecosystems across diverse scales (Howarth et al. 1996, Vitousek et al. 1997). It is useful to put N leaching from urban rooftop farming in the context of other land uses (Figure 1-5). Among 11 studies reporting N loss, including the NSF LTER projects

in agricultural, forested, and urban watersheds, N losses vary from <0.1 kg N ha^{-1} y^{-1} from a forested watershed, to 277 kg N ha^{-1} y^{-1} from a vegetable farm (Berndtsson et al. 2006; Cameron et al. 2013; Campbell et al. 2004; Fenn et al. 1998; Gregoir et al. 2011; Groffman et al. 2004; Goulding et al. 2013; Likens 2013; Min et al. 2012; Pärn et al. 2012; Syswerda et al. 2012). Green roof losses average between 4 - 5 kg N ha^{-1} y^{-1} (Berndtsson et al. 2006, Gregoire et al. 2011). Although 2 studies report the N concentration in leachate from rooftop farms as of January 2016, there have been no field observations on mass N leaching. Rooftop vegetable production could result in substantial N loss due to rapid soil drainage rates and fertility. Both urban design and policy need to consider N losses in order to quantify the environmental costs and benefits from urban rooftop farms.

Figure 1-5. Nitrogen leaching loss from forested, agricultural, and urban watersheds

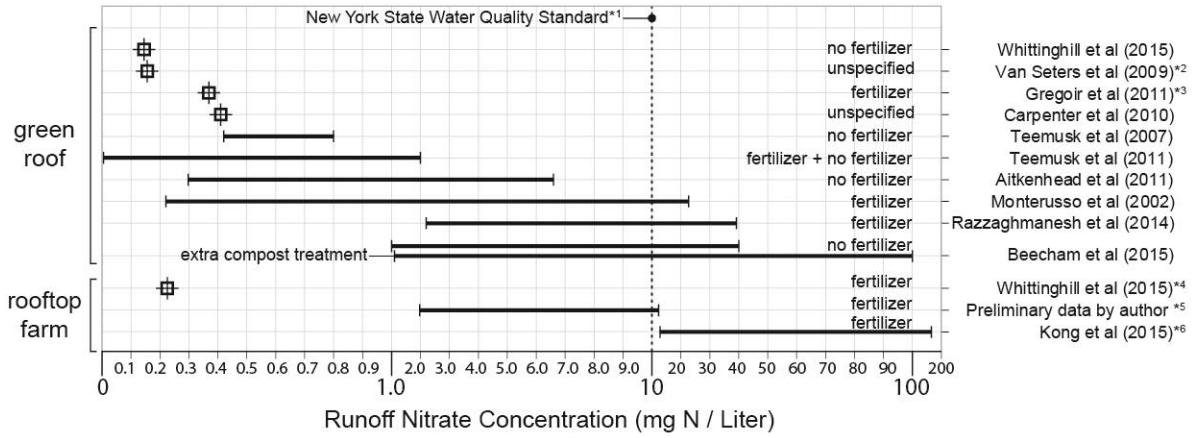


N Concentration. N concentration is a key metric used by environmental regulators to gauge surface water quality (Groffman et al. 2004), hence N concentration in runoff is a useful indicator of N loading from rooftop farms and green roofs. Two studies report the N concentration relevant to the

runoff from rooftop farming. Whittinghill et al. (2015) reported average runoff nitrate concentration of 0.22 mg L^{-1} from a field experiment while in a greenhouse experiment, Kong et al. (2015) reported maximum nitrate N concentrations of 12.8 mg L^{-1} from vermicompost and 165.8 mg L^{-1} using a controlled-release fertilizer. Different nitrate N concentrations between Whittinghill et al. (2015) and Kong et al. (2015) reflect nitrate levels present in the soil products (65 v.s. 118 mg L^{-1}), and N addition rates (35 v.s. 126 – 189 kg N ha^{-1}).

Figure 1-6 includes the runoff nitrate N concentrations from these studies as well as the preliminary results from the Brooklyn Grange farm at the Brooklyn Navy Yard in NYC that uses the same soil used by Kong et al. (2015). The results from the Brooklyn Grange farm (2 – 12 mg L^{-1}) fall between the values from other two studies. With the oversight of US EPA, New York State sets standard for nitrate N in rivers and streams as 10 mg L^{-1} (6 NYCRR Part 703) (NYS DEC 2016). Note that roof drains are not directly subject to regulation.

Figure 1-6. Runoff nitrate nitrogen concentration of green roofs and rooftop farms



*1 NYS regulation 6 NYCRR Part 703, Surface Water and Groundwater Quality Standards and Groundwater Effluent Limitations
*2 geometric mean
*3 nitrate + nitrite concentration, geometric mean
*4 vegetable cropping with commercial green roof mix
*5 preliminary data on the field sampling of drainage water at the Brooklyn Grange, using commercial rooftop farming mix
*6 pour-through sampling in greenhouse experiment, using commercial rooftop farming mix

Among 10 studies of non-production green roofs summarized in Figure 1-6, nitrate N concentration in runoff ranges from below 0.2 to 100 mg L^{-1} (Aitkenhead et al. 2011; Beecham et al. 2015;

Carpenter et al. 2010; Gregoir et al. 2011; Monterusso et al. 2002; Razzaghmanesh et al. 2014; Teemusk et al. 2007, 2011; Van Seters et al. 2009; Whittinghill et al. 2015). Teemusk et al. (2011) report higher N concentration in runoff from their fertilized green roof yet this is not reflected in the ranking of maximum nitrate concentrations across all of the studies. The study conducted by Gregoire et al. (2011) reported low N concentration in runoff despite using twice the rate of N application used in studies that yielded higher concentrations (Monterusso et al. 2002, Razzaghmanesh et al. 2014).

Composts are another potential source of N leached from green roofs (Hathaway et al. 2008, Toland et al. 2012). In Figure 1-6, the 2 highest nitrate concentrations were observed in unfertilized treatments that used compost (Beecham et al. 2015). Most commercial green roof mixes contain sources of organic N, including manure, blood meal, biosolids, and kelp even if it is not reported. In order to make meaningful comparisons among studies, it is important for each study to define the levels and sources of soil nutrient, in order to understand how fertilizer and compost contribute to the high N concentration in runoff.

N Sink. Urban green infrastructure is intended to be the N sink, reducing the N load to surface waters. Driscoll et al. (2015) reviewed the 7 types of green infrastructure projects (Bioretention, Media Filter, Detention Pond, Swale, Wetland, Green Roof), and report only green roofs behave as N source due to the fertilizer application, while rooftop farming was not within the scope of the review. As of January 2016, among 24 studies of rooftop farming, none reports the N source/sink relationship. In green roof literature, most studies on N source/sink are based on observations of runoff N concentrations alone.

Table 1-2. Comparison of N Concentration of green roof runoff & reference stormwater

Green Roof Runoff*[1]		REFERENCE STORMWATER*[2]	RUNOFF VOLUME REPORTED	SOIL DEPTH (mm)	PLANTS	LOCATION	AUTHOR
NO3	NH4						
LOW	LOW	RAIN	N	30	sedum	Sweden	Berndtsson et al (2009)
LOW	LOW	RAIN	N	400	shrub, tree	Japan	
LOW	LOW	RAIN	Y	30 - 40	sedum	Sweden	Berndtsson et al (2006)
LOW	LOW	ROOF RUNOFF	Y	140	wildflower	Toronto	Van Seters (2009)
SIMILAR	LOW	RAIN	Y	102	sedum	CT USA	Gregoire et al (2011)
SIMILAR	SIMILAR	ROOF RUNOFF	N	unspecified	sedum, moss	AR USA	Toland et al (2012)
SIMILAR	HIGH	RAIN	Y*[3]	100 - 300	Brachyscome, Chrysocephalum, Disphyma spp	South Australia	Beecham et al (2015)
VARIES	SIMILAR	RAIN	N	71	Sedum, Delosperma, Talinum spp	TX USA	Aitkenhead-Peterson et al (2011)
HIGH	HIGH	RAIN	Y*[4]	100	Sedum, Thymus, Dianthus, Cerastium spp	Estonia	Teemusk et al (2007)

*[1] comparison of green roof runoff N concentration to reference stormwater (precipitation or runoff from unvegetated roof)
*[2] precipitation is used as a reference stormwater if runoff from unvegetated roof is also reported
*[3] only retention rates reported
*[4] rainfall events not specified in concentration

Table 1-2 compares N concentration of green roof runoff to that of precipitation or runoff from control roof without vegetation (Aitkenhead-Peterson et al. 2011; Beecham et al. 2015; Berndtsson et al.2006, 2009; Gregoire et al. 2011; Teemusk et al. 2007; Toland et al. 2012; Van Seters et al. 2009). Results varied across the ranges of soil depths, vegetation and geographic locations. The first 4 studies reported lower nitrate and ammonia N concentrations in green roof runoff compared to precipitation or unvegetated roofs, hence green roofs are most likely a N sink. Among the remaining 5 studies, only Gregoire et al. (2011) compared mass N loading, reporting that the green roof was a net sink for N. The other 4 studies do not report runoff volume in terms that can be used to calculate loading rates. Because rooftop farms are likely sources of N due to the high soil infiltration and the application of irrigation and N fertilizer, it is necessary to measure both runoff volume and N concentration in order to calculate total load to downstream water bodies.

1-6. CASE STUDY

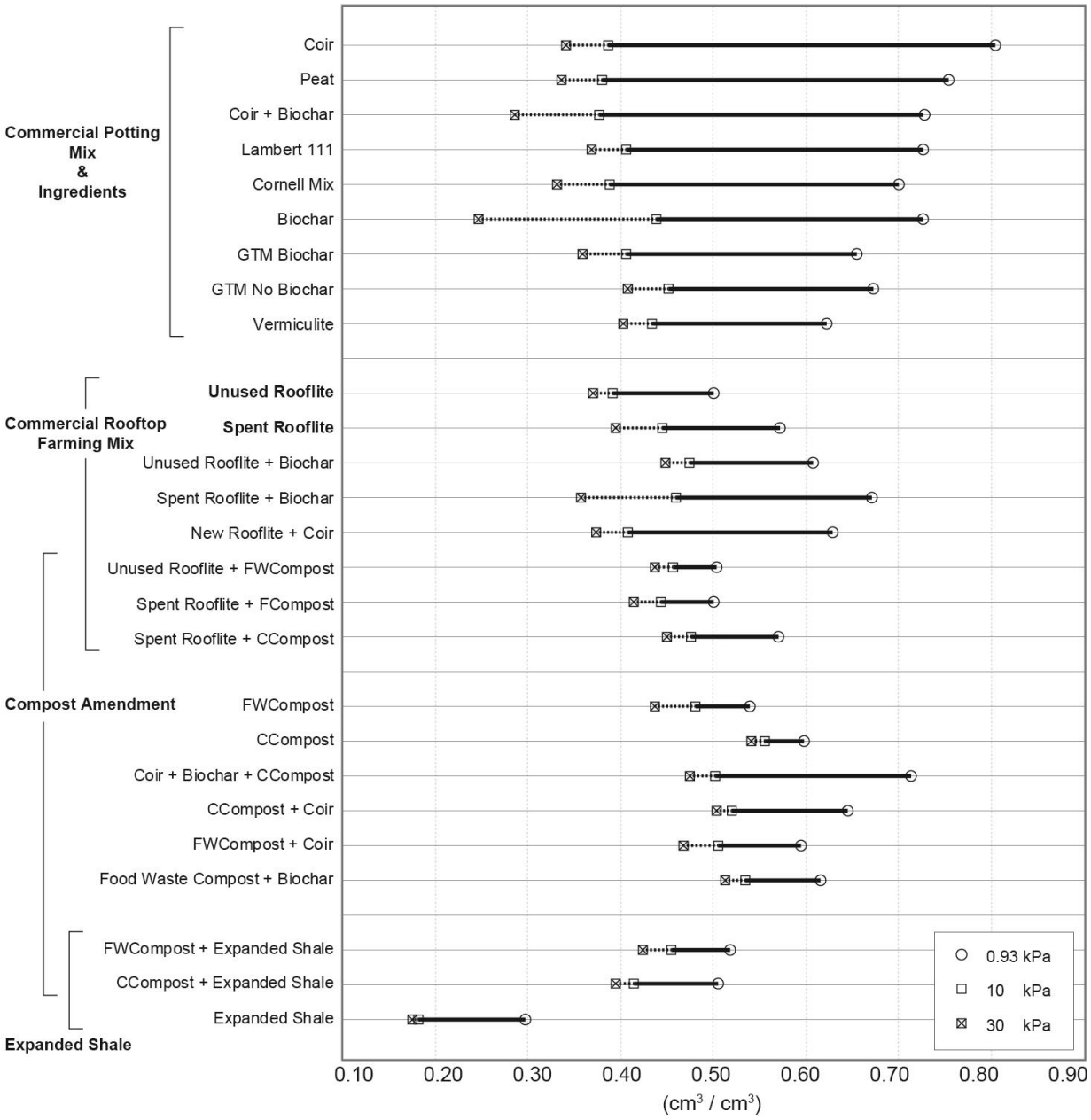
The literature review presented herein reveals that the water and N budgets of green roofs and rooftop farms varies widely in relation to soil composition, depth, vegetation, management regime and regional climate. In an effort to disentangle these variables, a multiyear study was initiated at the Brooklyn Grange, a 70,000-square-foot (0.65 ha) commercial rooftop farm in NYC. The Grange uses Rooflite®, presently the only commercial soil available for rooftop farming in the US. Observation included monitoring irrigation, ET, VWC, and runoff volume and quality and are experimentation with a variety of soil mixes in an effort to optimize the use of water and nutrient subsidies. Even though the farm is irrigated multiple times each day using a combination of drip tape and overhead sprinklers in order to maintain VWC between 25 – 35%, during the dry periods, VWC drops to 15 – 25%. Irrigation consistently produces base flow even below 25% VWC, suggesting that opportunities exist for optimizing the water and nutrient budgets of the farm by modifying the soil mix currently in use.

Soil Water Retention. Biogeochemical performance is not the only criterion for rooftop soils. They must also meet construction material guidelines for weight, wind, and fire set by the American National Standards Institute (ANSI). Also, decisions about including re-used waste products in soil mixes would ideally be informed by comprehensive life-cycle assessments, including the energy and carbon (C) emission by the transportation and manufacturing processes. The approach adopted in this study is to first narrow down the biogeochemically superior soil design through a controlled laboratory experiment, followed by field experiments, then consider regulatory and life-cycle perspectives.

Figure 1-7 compares the VWC of 26 potential rooftop soils and amendments obtained from replicated tension table experiments. Among 26 mixes, including 8 Rooflite® and 9 potting mixes, VWC at 0.93 kPa varied from < 30 % for expanded shale to 80% for coconut coir. The lowest VWC

of both unused and spent Rooflite® is ca 35%, exceeding field observations during rainless summer periods. This could be due to non-uniform field conditions caused by preferential flows and hysteresis or by variation in irrigation. Amending Rooflite® with biochar, coir and compost increased maximum VWC up to 60-70%, while 6 of 9 potting mix treatments had maximum VWM > 70%, which indicates the potential for designing lightweight rooftop soil by substituting organic amendments for expanded shale. Based on these lab results, we are field testing new soils that include coir and biochar.

Figure 1-7. Volumetric water content of soilless mixes and ingredients at 0.93, 10, 30kPa



Coir : Coconut Coir
Lambert 111 : Commercial Potting Mix
GTM : Commercial Potting Mix (uses Coir and Biochar tested in the study)
FWCompost : Food Waste Compost
CCompost : Cornell Compost
Rooflite : Commercial Rooftop Farming Mix (uses Expanded Shale in the study)

* each material is mixed in the same volume in the treatments with multiple materials

Yield. Despite the anticipated advantages afforded by local food production systems, if crop yield does not equal or exceed conventional agriculture, rooftop farming may do little to advance urban sustainability. A comparison of yields for four of the many vegetable crops grown at the Grange with California, New York, and New Jersey shows that the Grange out performs statewide averages for conventional farms (Table 1-3). On a per hectare basis, the Grange yielded 1.2 times more lettuce than California in 2015 and five times more tomatoes than NY in 2014. Note that California was experiencing a multiyear drought during this period, which could have depressed yields and skewed the data in favor of the Grange. In any case, it appears that rooftop vegetable production can produce high yields with appropriate inputs of water and nutrients.

Table 1-3. Vegetable yield of the Brooklyn Grange and in-ground agriculture

Crop	Year	Yield (metric ton / ha)			
		Brooklyn Grange	New York*2	New Jersey*2	California*2
Snap Beans	2014	20.40	7.07	3.70	12.35
	2015	19.79	7.30	3.59	13.47
Tomatoes	2014	68.58	13.47	24.14	35.36
	2015	47.62	14.59	25.26	34.80
Leaf Lettuce	2014	34.22*1	NA	NA	26.94
	2015	35.52*1	NA	NA	29.19
Bell Peppers	2014	128.26	NA	38.17	54.45
	2015	70.78	NA	34.24	51.08

*1 based on the yield of leafy greens mix
*2 based on the National Agricultural Statistics Service, USDA

Water and N Budget. Field monitoring of the farm's water budget indicates that 38% of the water supplied is lost to drainage (Table 1-4). Because irrigation relies on potable water from Upstate New York reservoirs, it would be desirable to reduce drainage losses. This could be accomplished by

varying the soil mix and depth to maximize water retention in the range of plant available soil moisture.

Mass N input to the Grange through the fertilizer application and atmospheric N deposition (Table 1-5) is about 120 kg N ha^{-1} y^{-1}, while mass N contained in vegetables leaving the farm as food crops is over 60% of N input. However, the sampled N loss by soil leachate (Table 1-5) is 10X the initial estimation by this N balance model (Fertilizer application + Atmospheric deposition – Vegetable harvest ≈ 40 kg N ha^{-1} y^{-1}). Organic N in soil imported in the original soil mix could account for these high leaching losses. Further research will examine the accuracy of the sampling method, and denitrification potential of soil leachate in the drainage layer of the Grange.

Table 1-4. Water Budget of the Brooklyn Grange

Water Flux Type	Water Flux (10^6 Liter y^{-1})	Method
Irrigation input	≈ 1.5	Flow meter
Precipitation input	> 5	Rain gauge
ET loss	> 4	Penman–Monteith equation
Drainage loss	≈ 2.5	Water Balance Model (Irrigation + Precipitation – ET)

Preliminary data on the water budget of the Brooklyn Grange, a rooftop farm at Brooklyn Navy Yard, NYC 2014 - 2015

Table 1-5. Nitrogen Budget of the Brooklyn Grange

N Flux Type	N Flux (kg N ha^{-1} y^{-1})	N Form	Method
Fertilizer Application	> 110	Total N	Inventory analysis (farming record)
Atmospheric Deposition	< 10	Dissolved Inorganic N	Bulk collector (with Ion-exchange resin)
Soil Leachate	> 400	Dissolved Inorganic N	Soil mesh bag (with ion-exchange resin)
Vegetable Harvest	> 80	Total N	Inventory analysis (farming record)

Preliminary data on the N budget of the Brooklyn Grange, a rooftop farm at Brooklyn Navy Yard, NYC 2014 - 2015

Conclusion

Rooftop farming requires the synthesis of knowledge from many fields, including soil science, biogeochemistry, horticulture, and urban planning and design, among which soil science is central to both understanding and improving practices. This chapter reviewed the intersection of those fields with an emphasis on their relevance to rooftop farming. The perspective of soil science is central to both understanding and improving rooftop farming.

Limitations of current research

As of January 2016, more than 1000 papers relevant to green roof design and rooftop ecosystems have been published. Of these studies, the bulk are focused on hydrologic responses, media and plant performance as well as climate effects. Green roof practices have been driven by the perceived need to reduce weight, drain rapidly and contribute to building insulation. Few even attempt comprehensive integration of the many topics shown in Figure 1-2. As of yet, only 24 studies address rooftop agriculture. This review has identified these key gaps in knowledge:

- Commercial/custom potting mixes, conventional roofing ballast, and mineral soil could be important functional components roof infrastructure, but are not systematically studied in green roof or rooftop vegetable cropping systems.

- While there is increasing interest in soil composition, depth, and moisture, key factors are often unreported, preventing understanding of what is actually driving soil water and nutrient dynamics.

- Even when studies include both soil performance and plant growth, they do not include inputs and losses, or runoff volume, quality and variation necessary to calculate loading rates and other ecosystem-level responses.

Future work

Rooftop farms are ideal for investigating urban biogeochemical processes because they are simple enough to be studied in detail but complex enough to yield insights into the way cities function in the global context for food security, environmental quality and waste management. Studies like the one at the Brooklyn Grange reveal how variables like soil composition, physical characteristics, depth, and application of fertilizer and irrigation subsidies affect the ability of a roof to deliver ecosystem services while at the same time actively informing daily management practices to improve the rapid development of BMPs. Future research should include:

- Detailed studies of engineered soils to conserve water, reduce leachate, and optimize partitioning of water into transpiration
- Systematic analysis of novel/different soil components, including repurposed waste and native soil
- How plant growth and quality can be optimized along with ecosystem-level responses.
- The small science approach of studying individual farms should be expanded to a research network including cities in different climate zones in order to develop a comprehensive framework of best management practices.

References

Aitkenhead-Peterson, J. A., B. D. Dvorak, A. Volder and N. C. Stanley (2011). "Chemistry of growth medium and leachate from green roof systems in south-central Texas." Urban Ecosystems 14(1): 17-33.

Acks, K. (2006). "A framework for cost-benefit analysis of green roofs: initial estimates." Green Roofs in the Metropolitan Region: Research Report. Columbia University Center for Climate Systems Research.

Ampim, P. A., J. J. Sloan, R. I. Cabrera, D. A. Harp and F. H. Jaber (2010). "Green roof growing substrates: types, ingredients, composition and properties." Journal of Environmental Horticulture 28(4): 244.

Baker, K. F. 1957 (ed). The U. C. System for Producting Healthy Container-grown Plants. Univ. of California Division of Agricultural Sciences.

Bates, A. J., J. P. Sadler, R. B. Greswell and R. Mackay (2015). "Effects of recycled aggregate growth substrate on green roof vegetation development: A six year experiment." Landscape and Urban Planning 135: 22-31.

Beck, D. A., G. R. Johnson and G. A. Spolek (2011). "Amending greenroof soil with biochar to affect runoff water quantity and quality." Environmental Pollution 159(8-9): 2111-2118.

Beecham, S. and M. Razzaghmanesh (2015). "Water quality and quantity investigation of green roofs in a dry climate." Water research 70: 370-384.

Berndtsson, J. C., T. Emilsson and L. Bengtsson (2006). "The influence of extensive vegetated roofs on runoff water quality." Science of the Total Environment 355(1): 48-63.

Berndtsson, J. C., L. Bengtsson and K. Jinno (2009). "Runoff water quality from intensive and extensive vegetated roofs." Ecological Engineering 35(3): 369-380.

Berndtsson, J. C. (2010). "Green roof performance towards management of runoff water quantity and quality: A review." Ecological Engineering 36(4): 351-360.

Best, B. B., K. R. Swadek and L. T. Burgess (2015). Soil-Based Green Roofs. Green Roof Ecosystems. K. R. Sutton. Cham, Springer International Publishing: 139-174.

Bloomberg, M. (2011). PlaNYC: A Greener, Greater New York, City of New York.

Buffam, A. P. D. I. and M. E. Mitchell (2015). Nutrient cycling in green roof ecosystems. Green Roof Ecosystems. Cham, Switzerland: Springer International Publishing, 107–137.

Cameron, K., H. Di and J. Moir (2013). "Nitrogen losses from the soil/plant system: a review." Annals of Applied Biology 162(2): 145-173.

Campbell, J. L., J. W. Hornbeck, M. J. Mitchell, M. B. Adams, M. S. Castro, C. T. Driscoll, J. S. Kahl, J. N. Kochenderfer, G. E. Likens, J. A. Lynch, P. S. Murdoch, S. J. Nelson and J. B. Shanley (2004). "Input-output budgets of inorganic nitrogen for 24 forest watersheds in the northeastern United States: A review." Water Air and Soil Pollution 151(1-4): 373-396.

Cao, C. T., C. Farrell, P. E. Kristiansen and J. P. Rayner (2014). "Biochar makes green roof substrates lighter and improves water supply to plants." Ecological Engineering 71: 368-374.

Carpenter, D. D. and P. Kaluvakolanu (2010). "Effect of roof surface type on storm-water runoff from full-scale roofs in a temperate climate." Journal of Irrigation and Drainage Engineering 137(3): 161-169.

Carson, T., D. Marasco, P. Culligan and W. McGillis (2013). "Hydrological performance of extensive green roofs in New York City: observations and multi-year modeling of three full-scale systems." Environmental Research Letters 8(2): 024036.

Cho, Y. Y. (2008). "Soil depth affects the growth of lettuce and chicory in extensive green roofs." HORTICULTURE ENVIRONMENT and BIOTECHNOLOGY 49(6): 434-438.

Cho, Y. Y., K. Y. Choi and Y.-B. Lee (2010). "Effects of irrigation methods on the growth, water holding capacity of substrate and nutrient uptake of Lettuce, Chicory and Endive grown in an extensive green roof system." Horticulture Environment and Biotechnology 51(4): 348-354.

Dasberg, S. (1999) The root medium. In Stanhill, G. and H. Z. Enoch (1999) (eds) Greenhouse Ecosystems. Elsevier Science Publishers, 1999.

De Blasio, W. W. (2015). "One New York: The Plan for a Strong and Just City [Online]." New York City, NY. Accessed on 3 Apr. 2016. <http://www.nyc.gov/html/onenyc/downloads/pdf/publications/OneNYC.pdf>.

Driscoll, C. T., C. G. Eger, D. G. Chandler, C. I. Davidson, B. K. Roodsari, C. D. Flynn, K. F. Lambert, N. D. Bettez and P. M. Groffman (2015). "Green Infrastructure: Lessons from Science and Practice. A publication of the Science Policy Exchange." 32.

Dunnett, N., A. Nagase and A. Hallam (2008). "The dynamics of planted and colonising species on a green roof over six growing seasons 2001–2006: influence of substrate depth." Urban Ecosystems 11(4): 373-384.

Eksi, M., D. B. Rowe, R. Fernández-Cañero and B. M. Cregg (2015). "Effect of substrate compost percentage on green roof vegetable production." Urban Forestry & Urban Greening 14(2): 315-322.

Elstein, J., G. E. Welbaum, D. A. Stewart and D. R. Borys (2006). Evaluating growing media for a shallow-rooted vegetable crop production system on a green roof. IV International Symposium on Seed, Transplant and Stand Establishment of Horticultural Crops; Translating Seed and Seedling 782.

Fahey, T. J., P. H. Templer, B. T. Anderson, J. J. Battles, J. L. Campbell, C. T. Driscoll, A. R. Fusco, M. B. Green, K.-A. S. Kassam and N. L. Rodenhouse (2015). "The promise and peril of intensive‐site‐based ecological research: insights from the Hubbard Brook ecosystem study." Ecology 96(4): 885-901.

Farrell, C., X. Q. Ang and J. P. Rayner (2013). "Water-retention additives increase plant available water in green roof substrates." Ecological Engineering 52: 112-118.

Felson, A. J. and S. T. Pickett (2005). "Designed experiments: new approaches to studying urban ecosystems." Frontiers in Ecology and the Environment 3(10): 549-556.
Fenn, M. E., M. A. Poth, J. D. Aber, J. S. Baron, B. T. Bormann, D. W. Johnson, A. D. Lemly, S. G. McNulty, D. F. Ryan and R. Stottlemyer (1998). "Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies." Ecological Applications 8(3): 706-733.
Getter, K. L. and D. B. Rowe (2009). "Substrate depth influences Sedum plant community on a green roof." HortScience 44(2): 401-407.
Goulding, K. W. T., Poulton, P. R., Webster, C. P., and Howe, M. T. (2000). Nitrate leaching from the Broadbalk Wheat Experiment, Rothamsted, UK, as influenced by fertilizer and manure inputs and the weather. Soil Use and Management 16(4): 244–250.
Grard, B. J.-P., N. Bel, N. Marchal, F. Madre, J.-F. Castell, P. Cambier, S. Houot, N. Manouchehri, S. Besancon and J.-C. Michel (2015). "Recycling urban waste as possible use for rooftop vegetable garden." Future of Food: Journal on Food, Agriculture and Society 3(1): 21-34.
Gregoire, B. G. and J. C. Clausen (2011). "Effect of a modular extensive green roof on stormwater runoff and water quality." Ecological Engineering 37(6): 963-969.
Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai and J. M. Briggs (2008). "Global change and the ecology of cities." science 319(5864): 756-760.
Groffman, P. M., N. L. Law, K. T. Belt, L. E. Band and G. T. Fisher (2004). "Nitrogen fluxes and retention in urban watershed ecosystems." Ecosystems 7(4): 393-403.
Hathaway, A. M., W. F. Hunt and G. D. Jennings (2008). "A field study of green roof hydrologic and water quality performance." Transactions of the ASABE 51(1): 37-44.
Howarth, R. W., G. Billen, D. Swaney, A. Townsend, N. Jaworski, K. Lajtha, J. Downing, R. Elmgren, N. Caraco and T. Jordan (1996). Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Nitrogen cycling in the North Atlantic Ocean and its watersheds, Springer: 75-139.
Jenny, H. (1941) Factors of Soil Formationm. McGraw-Hil, New York.
Jozwik, F. X. (2000). The Greenhouse and Nursery Handbook. Andmar Press, Mill, Wyoming.
Kemp, W. M., W. R. Boynton, J. E. Adolf, D. F. Boesch, W. C. Boicourt, G. Brush, J. C. Cornwell, T. R. Fisher, P. M. Glibert and J. D. Hagy (2005). "Eutrophication of Chesapeake Bay: historical trends and ecological interactions." Marine Ecology Progress Series 303(21): 1-29.
Kong, A. Y., C. Rosenzweig and J. Arky (2015). "Nitrogen Dynamics Associated with Organic and Inorganic Inputs to Substrate Commonly Used on Rooftop Farms." HortScience 50(6): 806-813.
Kotsiris, G., P. A. Nektarios, N. Ntoulas and G. Kargas (2013). "An adaptive approach to intensive green roofs in the Mediterranean climatic region." Urban forestry & urban greening 12(3): 380-392.
Kotsiris, G., P. A. Nektarios and A. T. Paraskevopoulou (2012). "Lavandula angustifolia growth and physiology is affected by substrate type and depth when grown under Mediterranean semi-intensive green roof conditions." HortScience 47(2): 311-317.
Li, Y. and R. W. Babcock Jr (2014). "Green roofs against pollution and climate change. A review." Agronomy for Sustainable Development 34(4): 695-705.
Likens, G. E. (2013). Biogeochemistry of a forested ecosystem, Springer Science & Business Media.
Lovell, S. T. (2010). "Multifunctional urban agriculture for sustainable land use planning in the United States." Sustainability 2(8): 2499-2522.
Meals, D. W., S. A. Dressing and T. E. Davenport (2010). "Lag time in water quality response to best management practices: A review." Journal of environmental quality 39(1): 85-96.
Mentens, J., D. Raes and M. Hermy (2006). "Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century?" Landscape and urban planning 77(3): 217-226.
Min, J., H. Zhang and W. Shi (2012). "Optimizing nitrogen input to reduce nitrate leaching loss in greenhouse vegetable production." Agricultural water management 111: 53-59.
Molineux, C. J., A. C. Gange, S. P. Connop and D. J. Newport (2015). "Using recycled aggregates in green roof substrates for plant diversity." Ecological Engineering 82: 596-604.
Monterusso, M., D. Rowe, C. Rugh and D. Russell (2002). Runoff water quantity and quality from green roof systems. XXVI International Horticultural Congress: Expanding Roles for Horticulture in Improving Human Well-Being and Life Quality 639.
Nawaz, R., A. McDonald and S. Postoyko (2015). "Hydrological performance of a full-scale extensive green roof located in a temperate climate." Ecological Engineering 82: 66-80.
Nektarios, P., N. Ntoulas and E. Nydrioti (2010). Turfgrass use on intensive and extensive Green roofs. XXVIII International Horticultural Congress on Science and Horticulture for People (IHC2010): International Symposium on 938.
Nektarios, P. A., I. Amountzias, I. Kokkinou and N. Ntoulas (2011). "Green roof substrate type and depth affect the growth of the native species Dianthus fruticosus under reduced irrigation regimens." HortScience 46(8): 1208-1216.
Newman, J. 2008. Protecting water quality. In. Newman, J. 2008 (ed). Greenhouse and Nursery management Practices to protect Water Quality. Univ of California Pub. 3508.
New York City Department of City Planning (2012). "Zone Green Text Amendment [Online]." New York City, NY. Accessed on 3 Apr. 2016. <http://www1.nyc.gov/assets/planning/download/pdf/plans/zone-green/zone_green.pdf>.

New York City Department of Environmental Protection (2010). "NYC Green Infrastructure Plan: A sustainable strategy for clean waterways [Online]." New York City, NY. Accessed on 3 Apr. 2016. <http://www.nyc.gov/html/dep/html/stormwater/nyc_green_infrastructure_plan.shtml>.

New York City Department of Environmental Protection (2011). "DEP Awards $3.8 Million in Grants for Community-Based Green Infrastructure Program Projects [Online]." New York City, NY. Accessed on 3 Apr. 2016. <http://www.nyc.gov/html/dep/html/press_releases/11-46pr.shtml#.VvIn6fkrKUk>.

New York State Department of Environmental Conservation, Connecticut Department of Environmental Protection (2000). "A total maximum daily load analysis to achieve water quality standards for dissolved oxygen in Long Island Sound." Connecticut Department of Environmental Protection, Hartford.

New York State Department of Environmental Conservation (2016). "Water Quality Standards and Classifications [Online]." Accessed on 25 Jun. 2016. <http://www.dec.ny.gov/chemical/23853.html>

New York State Energy Research and Development Authority (2013). "Sustainable Urban Agriculture: Confirming Viable Scenarios for Production, Final Report [Online]." Accessed on 3 Apr. 2016. <http://urbandesignlab.columbia.edu/files/2015/04/2_Sustainable-Urban-Agriculture_NYSERDA.pdf>.

Ntoulas, N., P. A. Nektarios, E. Charalambous and A. Psaroulis (2013). "Zoysia matrella cover rate and drought tolerance in adaptive extensive green roof systems." Urban forestry & urban greening 12(4): 522-531.

Ntoulas, N., P. A. Nektarios, T.-E. Kapsali, M.-P. Kaltsidi, L. Han and S. Yin (2015). "Determination of the Physical, Chemical, and Hydraulic Characteristics of Locally Available Materials for Formulating Extensive Green Roof Substrates." HortTechnology 25(6): 774-784.

Ntoulas, N., P. A. Nektarios and E. Nydrioti (2013). "Performance of Zoysia matrella 'Zeon'in shallow green roof substrates under moisture deficit conditions." HortScience 48(7): 929-937.

Ntoulas, N., P. A. Nektarios, K. Spaneas and N. Kadoglou (2012). "Semi-extensive green roof substrate type and depth effects on Zoysia matrella 'Zeon'growth and drought tolerance under different irrigation regimes." Acta Agriculturae Scandinavica, Section B-Soil & Plant Science 62(sup1): 165-173.

Olszewski, M. W., M. H. Holmes and C. A. Young (2010). "Assessment of physical properties and stonecrop growth in green roof substrates amended with compost and hydrogel." HortTechnology 20(2): 438-444.

Palmer, M., E. Bernhardt, E. Chornesky, S. Collins, A. Dobson, C. Duke, B. Gold, R. Jacobson, S. Kingsland, R. Kranz, M. Mappin, M. L. Martinez, F. Micheli, J. Morse, M. Pace, M. Pascual, S. Palumbi, O. J. Reichman, A. Simons, A. Townsend and M. Turner (2004). "Ecology for a Crowded Planet." Science 304(5675): 1251-1252.

Pärn, J., G. Pinay and Ü. Mander (2012). "Indicators of nutrients transport from agricultural catchments under temperate climate: A review." Ecological indicators 22: 4-15.

Plakias, A. C. (2016). The Farm on the Roof: What Brooklyn Grange Taught Us about Entrepreneurship, Community, and Growing a Sustainable Business, Penguin Publishing Group.

Poë, S., V. Stovin and C. Berretta (2015). "Parameters influencing the regeneration of a green roof's retention capacity via evapotranspiration." Journal of Hydrology 523: 356-367.

Rabalais, N. N., R. E. Turner and W. J. Wiseman Jr (2002). "Gulf of Mexico hypoxia, AKA" The dead zone"." Annual Review of ecology and Systematics: 235-263.

Razzaghmanesh, M., S. Beecham and F. Kazemi (2014). "Impact of green roofs on stormwater quality in a South Australian urban environment." Science of the Total Environment 470: 651-659.

Robertson, M., T. K. BenDor, R. Lave, A. Riggsbee, J. Ruhl and M. Doyle (2014). "Stacking ecosystem services." Frontiers in Ecology and the Environment 12(3): 186-193.

Rowe, D. B. (2011). "Green roofs as a means of pollution abatement." Environmental Pollution 159(8): 2100-2110.

Savi, T., D. Boldrin, M. Marin, V. L. Love, S. Andri, M. Tretiach and A. Nardini (2015). "Does shallow substrate improve water status of plants growing on green roofs? Testing the paradox in two sub-Mediterranean shrubs." Ecological Engineering 84: 292-300.

Savi, T., M. Marin, D. Boldrin, G. Incerti, S. Andri and A. Nardini (2014). "Green roofs for a drier world: effects of hydrogel amendment on substrate and plant water status." Science of the Total Environment 490: 467-476.

Shock, C. C. and F.-X. Wang (2011). "Soil water tension, a powerful measurement for productivity and stewardship." HortScience 46(2): 178-185.

Skyland USA LLC (2016). "Rooflite Intensive Ag [Online]" Accessed on 3 Apr. 2016. <http://www.rooflitesoil.com/products/intensive-ag>.

Syswerda, S., B. Basso, S. Hamilton, J. Tausig and G. Robertson (2012). "Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA." Agriculture, Ecosystems & Environment 149: 10-19.

Teemusk, A. and Ü. Mander (2007). "Rainwater runoff quantity and quality performance from a greenroof: The effects of short-term events." Ecological engineering 30(3): 271-277.

Teemusk, A. and Ü. Mander (2011). "The influence of green roofs on runoff water quality: a case study from Estonia." Water resources management 25(14): 3699-3713.

Toland, D., B. Haggard and M. Boyer (2012). "Evaluation of nutrient concentrations in runoff water from green roofs, conventional roofs, and urban streams." Transactions of the ASABE 55(1): 99-106.

United States Environmental Protection Agency (2016). "Clean Water Act (CWA) Compliance Monitoring [Online]." Accessed on 3 Apr. 2016. <https://www.epa.gov/compliance/clean-water-act-cwa-compliance-monitoring>.

Van Mechelen, C., T. Dutoit and M. Hermy (2015). "Vegetation development on different extensive green roof types in a Mediterranean and temperate maritime climate." Ecological Engineering 82: 571-582.
Van Seters, T., L. Rocha, D. Smith and G. MacMillan (2009). "Evaluation of green roofs for runoff retention, runoff quality, and leachability." Water Quality Research Journal of Canada 44(1): 33.
VanWoert, N. D., D. B. Rowe, J. A. Andresen, C. L. Rugh and L. Xiao (2005). "Watering regime and green roof substrate design affect Sedum plant growth." HortScience 40(3): 659-664.
Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger and D. G. Tilman (1997). "Human alteration of the global nitrogen cycle: sources and consequences." Ecological applications 7(3): 737-750.
Voyde, E., E. Fassman, R. Simcock and J. Wells (2010). "Quantifying evapotranspiration rates for New Zealand green roofs." Journal of Hydrologic Engineering 15(6): 395-403.
Whittinghill, L. J., D. B. Rowe, J. A. Andresen and B. M. Cregg (2015). "Comparison of stormwater runoff from sedum, native prairie, and vegetable producing green roofs." Urban Ecosystems 18(1): 13-29.
Whittinghill, L. J., D. B. Rowe and B. M. Cregg (2013). "Evaluation of vegetable production on extensive green roofs." Agroecology and Sustainable Food Systems 37(4): 465-484.
Young, T., D. D. Cameron, J. Sorrill, T. Edwards and G. K. Phoenix (2014). "Importance of different components of green roof substrate on plant growth and physiological performance." Urban Forestry & Urban Greening 13(3): 507-516.

Chapter 2. Hydrology of the Brooklyn Grange, an Urban Rooftop Farm

Abstract

Intensive agriculture is an emerging theme of green roof technology. Production of affordable, nutritious vegetables for local consumption and stormwater management are among the ecosystem services emphasized in studies of rooftop farming. However, intensive vegetable production requires irrigation, which can reduce stormwater retention, and there is little empirical data from full-scale operational rooftop farms. In this paper, we report the hydrologic performance of the Brooklyn Grange Navy Yard Farm, a 0.61-ha rooftop farm atop an 11-story building in New York City USA. We monitored soil water content, precipitation, irrigation, drainage, and crop yields to provide water balance and water use efficiency. We found cumulative discharge exceeded precipitation by 11% during the entire study period, hence the farm was a net source of water in the urban hydrologic cycle. Depending on crop types, water use efficiency at the Grange was lower than in-ground agriculture in dry regions with high irrigation demands. For the Brooklyn Grange to integrate stormwater management and water-efficient crop production, it will be important to increase soil water storage, to reduce irrigation demands and drainage loss of water while maintaining average soil moisture levels in the range of readily available water between 10-100 kPa of soil water tension.

Keywords: green roof; urban agriculture; green infrastructure; ecosystem service; stormwater management; water use efficiency

2-1. Introduction

Since the late 20th century, over 1,000 peer-reviewed papers have reported on the potential for green roofs to enhance the sustainability of the built environment through improved stormwater management, energy conservation, air pollution abatement, and increasing biodiversity (Harada et al. 2017; Lundholm 2015; Mentens et al. 2006; Oberndorfer et al. 2007; Rowe 2011). A recent extension of green roof technology is intensive vegetable production. Rooftop farming could improve urban food security, waste management by using compost, and social outcomes like environmental education, green job employment, and building more cohesive communities (Ackerman et al. 2014; Ackerman et al. 2013; Harada et al. 2017; Lovell 2010; Specht et al. 2016; Thomaier et al. 2015; Whittinghill and Starry 2016). Despite these anticipated benefits, it is challenging to accomplish multiple ecosystem services with a single intervention in a complex urban system. For example, irrigation can have negative effect on stormwater reduction. In intensive vegetable production, irrigation maintains soils at higher mean moisture levels for the satisfactory crop yield and quality. During storm events, this higher antecedent moisture causes irrigated soils to discharge more water than they would otherwise. Rooftops are underused impervious surfaces, which makes them logical places to manage stormwater, mitigate flood risks, and reduce combined sewer overflows (CSOs) to surface waters (Rosenzweig et al. 2006). Yet the economic viability of urban rooftop agriculture requires the production of high value crops like lettuce, mustard greens, tomatoes, and peppers (Ackerman et al. 2013; Lovell and Taylor 2013), which have high irrigation demands.

Soil plays an important role in precision water management, and could balance the negative effect irrigation has on stormwater retention. Ideally, precision water management would use irrigation to maintain optimum soil water for satisfactory crop yield and quality. Field capacity can be estimated by laboratory measurements of moisture release at varying tensions, while the optimum soil water tension for a specific crop is determined empirically in the field (Shock and Wang 2011). These

variables are the foundation for precision water management. Other important factors include the effects of soil hydrophobicity, preferential flow, irrigation methods (e.g., sprinkler, drip tape), and mulching (Rowe et al. 2014; Stovin et al. 2015; Van Dam et al. 1990; Whittinghill et al. 2016b).

Table 2-1. Field studies on rooftop farming (modified from Harada et al. 2017) *Köppen Climate Classification System, **S: satisfactory yield

Location	KCC*	Crop	Yield**	Max. Irrigation (mm / day)	Irrigation Method	Soil Depth (mm)	Hydrologic Performance	Soil Type (composition)	Author
BlacksburgVA USA	Cfa	Kale	S	as needed	drip tape	102	stormwater retention	commercial potting soil (unspecified)	Elstein et al (2006)
Bologna, Italy	Cfa	lettuce, tomato, eggplant, pepper, melon, watermelon	S	11.7	-	-	-	commercial potting soil (unspecified)	Sanyé - Mengual et al (2015)
Bologna, Italy	Cfa	lettuce, cabbage, chicory, tomato, eggplant, pepper, melon, watermelon	S	-	-	-	-	-	Orsini et al (2014)
New York, NY USA	Cfa	-	-	-	drip	25 - 250	-	commercial green roof soil (unspecified)	Whittinghill et al (2016)[a]
Suwon Korea	Cfb	lettuce, chicory	S	-	wick	50, 100, 200	-	commercial potting soil (55-65% peat, 35-45% perlite)	Cho (2008)
Suwon Korea	Cfb	lettuce, chicory	S	-	wick, drip reservoir-drainage	150	-	commercial potting soil (55-65% peat, 35-45% perlite)	Cho et al (2010)
Paris France	Cfb	lettuce, tomato	S	-	drip	300	-	1) commercial potting soil (unspecified) 2) experimental potting soil (yard waste compost, crushed wood, coffee ground)	Grard et al (2015)
East Lansing MI USA	Dfb	cucumber, pepper	S	1.6	overhead sprinklers	125	-	experimental green roof soil (0, 20, 40, 60, 80, 100% yard waste compost treatments, expanded shale, sand,)	Eksi et al (2015)
East Lansing, MI USA	Dfb	tomato, beans, cucumber, pepper chive, basil	S (except pepper)	11.0	micro-emitters	105	-	commercial green roof soil (50% expanded shale, 35% sand, 15% leaf compost)	Whittinghill et al (2013)
East Lansing, MI USA	Dfb	tomato, beans, cucumber, pepper, chive, basil	S	9.0	micro-emitters	105	stormwater retention	commercial green roof soil (unspecified)	Whittinghill et al (2015)
East Lansing, MI USA	Dfb	tomato, beans, cucumber, pepper, chive, basil	S	13.5	micro-emitters	127	-	commercial green roof soil (unspecified)	Whittinghill et al (2016)[b]

In concept, precision water management could be easily accomplished in rooftop farms. Soil can be engineered to optimal design specifications at the time of construction, and it is easier to amend or replace soil in comparison to in-ground agriculture. Also, rooftops are simple and discreet hydrologic compartments in which input and output of water can be monitored precisely. The potential for precise hydrologic observation and soil manipulation makes rooftop farms ideal models for developing adaptive hydrologic management of urban areas. To date, no studies have reported on the hydrologic performance of the full-scale rooftop farm. Among 11 studies on rooftop farming (Table

2-1), all except Whittinghill et al. (2016a) used small microcosms, while Whittinghill et al. (2016a) focused only on nutrient loss, not hydrology.

Rooftop Farm Soils. Large particles of expanded shale, clay, and silt (ESCS) are common constituents of green roof soils, while potting mixes generally use peat, coconut coir, and other organic base materials which have higher water holding capacity and lower bulk density (Ampim et al. 2010; Ntoulas et al. 2015). Among the studies of rooftop farming summarized in Table 2-1, 5 used potting soils, and 5 used soils with ESCS materials. Five studies report volumetric water contents (VWC) of soil during field experiments, or laboratory measurements of field capacity (VWC at 10 kPa) (Cho 2008; Cho et al. 2010; Eksi et al. 2015; Whittinghill et al. 2015; Whittinghill et al. 2016b). It is difficult to quantify the effect of soil design on hydrologic performance of rooftop farms because none compares physical properties of soil with drainage losses and soil water under field conditions.

Irrigation and evapotranspiration. Among the studies of rooftop farming summarized in Table 2-1, 5 report maximum daily irrigation rates ranging 1.6-13.5 mm / day. None provide a formal water balance including evapotranspiration rates, or discharge generated by irrigation. In rooftop farms, irrigation needs to target a shallow root zone because crop cycles can be short (4 to 5 weeks for salad greens), which require episodes of germination and establishment throughout the growing season. In a soil with ESCS particles (e.g., > 5mm, 50% mass), water tends to accumulate near the bottom of the soil column due to the large pore size (Berretta et al. 2014). Deeper soil makes it difficult to maintain sufficient water near the surface where it is needed for germination and root growth in seedlings. Also, field capacity in the field can be lower than predicted from laboratory measurements due to preferential flow and hydrophobicity (Stovin et al. 2015). These factors could increase irrigation demands and decrease stormwater retention, thereby producing drainage discharge even during the dry periods.

Watershed Engineering. To relate rooftop farming to the practices of watershed engineering, it is useful to determine the runoff curve number (CN), which is broadly used to estimate runoff volume from ungauged watersheds. The methods outlined in Technical Release 55 (TR55) by Natural Resources Conservation Service (NRCS 1986) could be used to estimate drainage, which for rooftop farms is functionally equivalent to runoff. While no studies report CNs for rooftop farms, Fassman-Beck et al. (2015) report CN for unirrigated green roofs. When estimating discharge in this way, CN cannot be determined from an existing hydrologic soil group because more stormwater infiltration can result in more discharge hence greater CN for green roofs. Instead, Fassman-Beck et al. (2015) proposed CN = 84 for extensive (soil depth < 150 mm) green roofs when individual rainfall depth > 20-30 mm, while maximum water storage is calculated as soil depth (mm) X plant available water (%) for smaller precipitation events.

To quantify reduction of stormwater discharge, stormwater discharge must be disaggregated from irrigation discharge, since these two outflows are combined in rooftop farms. A useful approach follows that used to disaggregate base flow from surface runoff. These two outflows are distinguished by (1) discharge path (e.g., ground water vs. surface runoff), (2) discharge response time (e.g., slow flow vs. quick flow), or (3) both, although this distinction has been ambiguously applied to hydrograph analysis (Schwartz 2007). In rooftop farms, distinguishing among discharge sources is relatively unimportant because precipitation and irrigation are the only sources of discharge, and can be easily monitored. In contrast, discharge response time is important in stormwater management practices. To separate quick flow from slow flow, potentially useful methods include automated base flow separation algorithms (Eckhardt 2005; Lim et al. 2005; Schwartz 2007).

Stormwater reduction. Among the studies on rooftop farming summarized in Table 2-1, Whittinghill et al. (2015) report stormwater reduction of 87.7% in light precipitation (< 2 mm), 85.6% in medium

precipitation (2-10 mm), and 58.9% in heavy precipitation (> 10 mm) during the growing seasons. Another study by Elstein et al. (2006) only report stormwater reduction (69.2%) relative to control gravel plot, and cannot be compared with other studies. Driscoll et al. (2015) reviewed the studies on non-production green roofs, and report the mean stormwater reduction of 53% during summer, which is lower than Whittinghill et al. (2015). Spolek (2008) reports stormwater reduction of 12-25% by specifically testing irrigated full-scale green roofs growing sedums, grasses, and wildflowers. Such low stormwater retention could be due to the fact that observations included winter periods. In addition, Spolek (2008) reports stormwater reduction by comparing precipitation to total discharge including irrigation discharge. While literature on rooftop farming and non-production green roofs provides important insights on the hydrologic performance of rooftop farms, scales and metrics of water balance differ across studies. Also, knowledge from the plot-scale experiments or non-production green roofs may not translate directly to the full-scale rooftop farm. The objective of this paper is to quantify the hydrologic performance of a rooftop farm in New York City, USA, in a humid temperate climate with hot summers and no dry season. We pose the following *a priori* questions which this study will address:

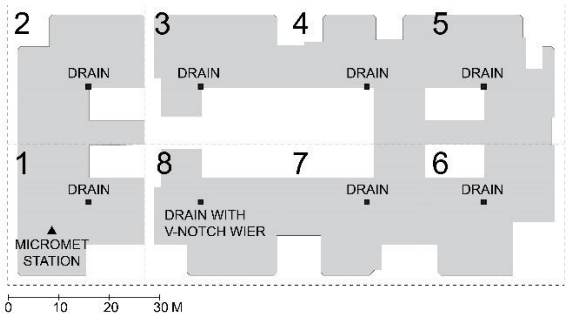
1. How much stormwater is reduced by the rooftop farm, and how does this compare with unirrigated green roofs?
2. Do tension-based estimates of field capacity provide reasonable estimates of stormwater retention for the rooftop farm?
3. Does and how much does irrigation contribute to discharge?
4. How much water is lost to evapotranspiration?
5. How can water use efficiency be improved in rooftop farming?

2-2. Material and Methods

Site Description. The study site is the Brooklyn Grange Navy Yard farm, an irrigated rooftop vegetable farm on an 11-story building at the former Brooklyn Navy Yard, New York City, USA (Figure 2-1). The Community-Based Green Infrastructure Program of the NYC Department of Environmental Protection funded $592,730 for the construction, anticipating that stormwater management would be an equally important goal of the farm in addition to the vegetable production (NYC DEP 2011). As shown in Figure 2-1, the site has a total of 8 catchments in a 2 (north-south) by 4 (east-west) grid arrangement, and each catchment has a single roof drain. While the farm grows over 60 crops, leaf lettuce (*Lactuca sativa*), mustard greens (*Brassica juncea*), arugula (*Eruca sativa*), and tomato (*Solanum lycopersicum*) can account for over the half of planting bed area, sales, and yield in fresh weight. Of the site area, 33% is impervious and consists of rooftop structures (e.g., elevator rooms, water tanks, building top lights) and walkways between soiled areas, whereas 47% of the site (70% of area covered with soil) consists of flat tops of planting beds, which are mounded without sidewalls (Figure 2-1). On these planting beds, vegetables are grown during the growing seasons, receiving drip tape and sprinkler irrigation. The remaining 20% of the site (30% of area covered with soil) is bare soil between flat tops of planting beds which receives sprinkler irrigation even though it lacks vegetation. The ratio of flat tops of the planting beds and in-between bare soil is approximately the same in each catchment.

Figure 2-1. Site Information of the Brooklyn Grange Navy Yard Farm, NYC. In the site plan, grey area shows the area covered with soil, dotted lines show grade breaks, and white area shows impervious areas such as elevator rooms, water tanks, building top lights, and walkways. Aerial site image ©google 2014

Location	Brooklyn Navy Yard, Brooklyn, NY	
Geographic coordinate	latitude: 40.698325, longitude: -73.97263	
Elevation above sea level	46 m (building height: 41m, ground level: 5m)	
Area (entire farm)	6122 m^2	4082 m^2 (area covered with soil)
		2040 m^2 (impervious area)
Area (catchment 8 with v-notch weir)	760 m^2	489 m^2 (area covered with soil)
		271 m^2 (impervious area)
Catchment slope	≈ 2%	
Completion of construction	June 01, 2012	
Growing season	April to November	



***Soil Description*.** The soil used on the site is Rooflite® Intensive Ag (Rooflite hereafter), a commercial soil blend of expanded shale, animal manure compost, and the spent substrate from mushroom production, which is commonly made of animal manure and lignocellulosic materials such as cereal straws and saw dusts (Kong et al. 2015; Paredes et al. 2009; Phan and Sabaratnam 2012; Skyland USA LLC 2015). The flat tops of planting beds (250 mm deep) are typically 1000-mm wide. Bare soil between planting beds (25-50 mm deep) are typically 430-mm wide. Soil is underlain by filter fabric, a drainage layer (100-150 mm deep) filled with expanded shale, and a water-proofing membrane.

***Laboratory Measurements of Moisture Release Characteristics*.** Rooflite was sampled from the site on March 31, 2015 after the 3 years of use. In the laboratory, water contents of the sample were measured at tensions between 0-1500 kPa in 3 stages. A pressure plate extractor was used to measure gravimetric water content within the high-tension range (33, 500, 1000, 1500 kPa, replicated 4 times). Volumetric water contents (VWC) within the middle-tension range (6.5, 10, 20, 30 kPa, replicated 36 times) were measured using a sand table and metal soil rings (height: 62 mm, internal diameter: 76 mm) filled with samples. Then, the same 36 soil rings were assembled into 6 stacks of 6 which were saturated for 48 hours, and freely drained until drainage stopped, to measure VWC within the low-tension range (0.31, 0.93, 1.55, 2.17, 2.79, 3.41 kPa, replicated 6 times). Rooflite was compared with native mineral soils (the top 80 mm of Hudson silty clay loam, Collamer silt loam, Arkport sandy loam) collected from experimental plots at Cornell University in December 2016. The same soil ring was used to measure VWC at 3.1 kPa without stacking, followed by the measurement by the pressure plate extractor (33, 500, 1000, 1500 kPa, replicated 4 times). The average bulk density in the measurements of volumetric water contents were used to convert the results of the pressure plate extractor, from gravimetric to volumetric water contents.

***Soil Water Monitoring*.** Soil VWC was monitored in catchments 1, 2, 4, 5, 6, and 8 (Figure 2-1). Wireless soil VWC sensors (Model W-SMC, Onset Computer Corporation) were deployed at 100 mm below the flat tops of planting beds and 25 mm above the bottom of planting beds. The probes were calibrated before the study period. Measurements were made every minute and uploaded to the local computer via a wireless connection. Initially, sensors were replicated 3 times in each of 6 catchments.

Weather Monitoring. A micrometeorological station by Onset Computer Corporation (Bourne MA) was installed in catchment 1 (Figure 2-1) to monitor precipitation temperature and relative humidity for calculating reference ET. The station included a tipping bucket rain gauge (Model: S-RGA), temperature and relative humidity sensors (Model S-THB-M002), solar radiation sensor (Model: S-LIB-M003), and wind speed sensor (Model S-WSA-M003). Windspeed and other sensors were 2.0-m and 1.7-m above the flat tops of planting bed respectively. The data were aggregated every minute, uploaded to the remote server hourly via a data logger (Model HOBO U30 ETH).

***Irrigation Monitoring*.** The farm uses both drip tape and sprinkler systems. Flow rates of both systems were monitored by using in-line flow meters (Model: SPWM-075, EKM Metering, Inc. Santa Cruz, CA) in the piping. The data were aggregated hourly and uploaded to the local laptop computer via a wireless connection. Hand watering was negligible, yet the hoses for the hand watering were connected to the piping for sprinkler, and were also monitored as a part of the sprinkler system.

Drainage Discharge Monitoring. Drainage discharge rate was monitored only in catchment 8 (Figure 2-1). A small V-notch weir was fabricated based on Carson et al. (2013), and installed to the roof drain outfall with a 152 mm (6 inch) in internal diameter. The water depth at the notch was measured by a sonic distance sensor (Model TSPC-30S1), and the V-notch weir was calibrated in the

laboratory before the field installation. Measurements were logged every 15 minutes as averages of 90 10-sec point measurements.

2-3. Models

We used 7 independent models gleaned from the literature to interpret our data. Discharge simulation used a daily water balance model based on Sherrard and Jacobs (2011) and 3 empirical models of unirrigated non-production green roofs by Carson et al. (2013). In the daily water balance model, discharge from impervious areas and bare soil areas between planting beds were estimated by using the runoff curve numbers 98 and 96 respectively based on NRCS (1986). Slow flow was disaggregated from quick flow by Eckhardt's (2005) method, using the Web-based Hydrograph Analysis Tool (WHAT) (Lim et al. 2005) as the calculator. Hourly ET from a reference short grass surface was estimated by the Penman-Monteith method from American Society of Civil Engineers, Environmental Water Resources Institute (ASCE-EWRI 2004), using a Microsoft Excel spreadsheet by Snyder and Eching (2008) as the calculator. Variables and models are summarized in Table 2-2 and 3 respectively. In addition to these 6 models, we also summarized the calculations we used to express water flux for either the entire surface covered with soil, or just the area occupied by planting beds (Table 2-3).

Table 2-2. Variables of the models used in this study

I	infiltration into soil	mm	Sherrard and Jacobs (2011)
S_0	initial soil water	mm	
S_{fc}	field capacity	mm	
Rf	rainfall	mm	Carson et al. (2013)
Ro	runoff	mm	
Q	runoff	inch	NRCS (1986)
P	rainfall	inch	
S	potential maximum retention after runoff begins	inch	
CN	runoff Curve number	0-100	
b_t	baseflow in time step t	m^3	Eckhardt (2005)
BFI_{max}	long term ratio of baseflow to total streamflow	m^3/m^3	
a	filter parameter		
Q_t	the total streamflow in time step t	m^3	
ET_r	hourly evapotranspiration from 0.12-m tall grass reference surface	mm	ASCE-EWRI (2004)
R_n	hourly net radiation	$MJm^{-2}h^{-1}$	
γ	psychrometric constant	$kPa°C^{-1}$	
T_M	hourly mean air temperature	°C	
u_2	hourly mean wind speed	ms^{-1}	
Δ	slope of the saturation vapor pressure curve at hourly mean air temperature	$kPa°C^{-1}$	
e_s	saturation vapor pressure at hourly mean air temperature	kPa	
e_a	saturation vapor pressure at hourly mean dew point temperature		
C_h	denominator constant		
A_{TC}	Total area of catchment 8	759.8 m^2	The Brooklyn Grange (this study)
A_{TS}	Total area covered with soil in catchment 8	489.2 m^2	
A_{BED}	Planting bed area in catchment 8	342.4 m^2	
W_{TC}	Water flux expressed for the total area of catchment 8	mm	
W_{TS}	Water flux expressed for the area covered with soil in catchment 8	mm	
W_{BED}	Water flux expressed for planting bed area in catchment 8	mm	

Table 2-3. Models used in this study

Eq 1	$R =$	0 $I + S_0 - S_{fc}$	if $I + S_0 \leq S_{fc}$ if $I + S_0 > S_{fc}$	Sherrard and Jacobs (2011)
Eq 2	$Ro =$	0 $0.0012(Rf)^2 + 0.6852\,(Rf) - 3.1200$	if $Rf \leq 4.6$ if $Rf > 4.6$	Carson et al. (2013)
Eq 3	$Ro =$	0 $0.0020(Rf)^2 + 0.5821(Rf) - 3.6024$	if $Rf \leq 6.1$ if $Rf > 6.1$	Carson et al. (2013)
Eq 4	$Ro =$	0 $0.0049(Rf)^2 + 0.2540(Rf) - 0.1497$	if $Rf \leq 0.6$ if $Rf > 0.6$	Carson et al. (2013)
Eq 5	$Q =$	0 $(P - 0.2S)^2/(P + 0.8S)$	if $R \leq 0.2S$ if $R > 0.2S$	NRCS (1986)
Eq 6	$S =$	$\frac{1000}{CN} - 10$		NRCS (1986)
Eq 7	$b_t =$	$\frac{(1 - BFI_{max}) \times a + b_t + (1 - a) \times BFI_{max} \times Q_t}{1 - a \times BFI_{max}}$		Eckhardt (2005)
Eq 8	$ET_r =$	$\frac{0.408\Delta(R_n - G) + \left(\frac{37\gamma}{T_M + 273}\right) u_2(e_s - e_a)}{\Delta + \gamma(1 + C_h u_2)}$		ASCE-EWRI (2004)
Eq 9	$W_{TS} =$	$\frac{W_{TC} \times A_{TC}}{A_{TS}}$		This study
Eq 10	$W_{BED} =$	$\frac{W_{TC} \times A_{TC}}{A_{BED}}$		This study

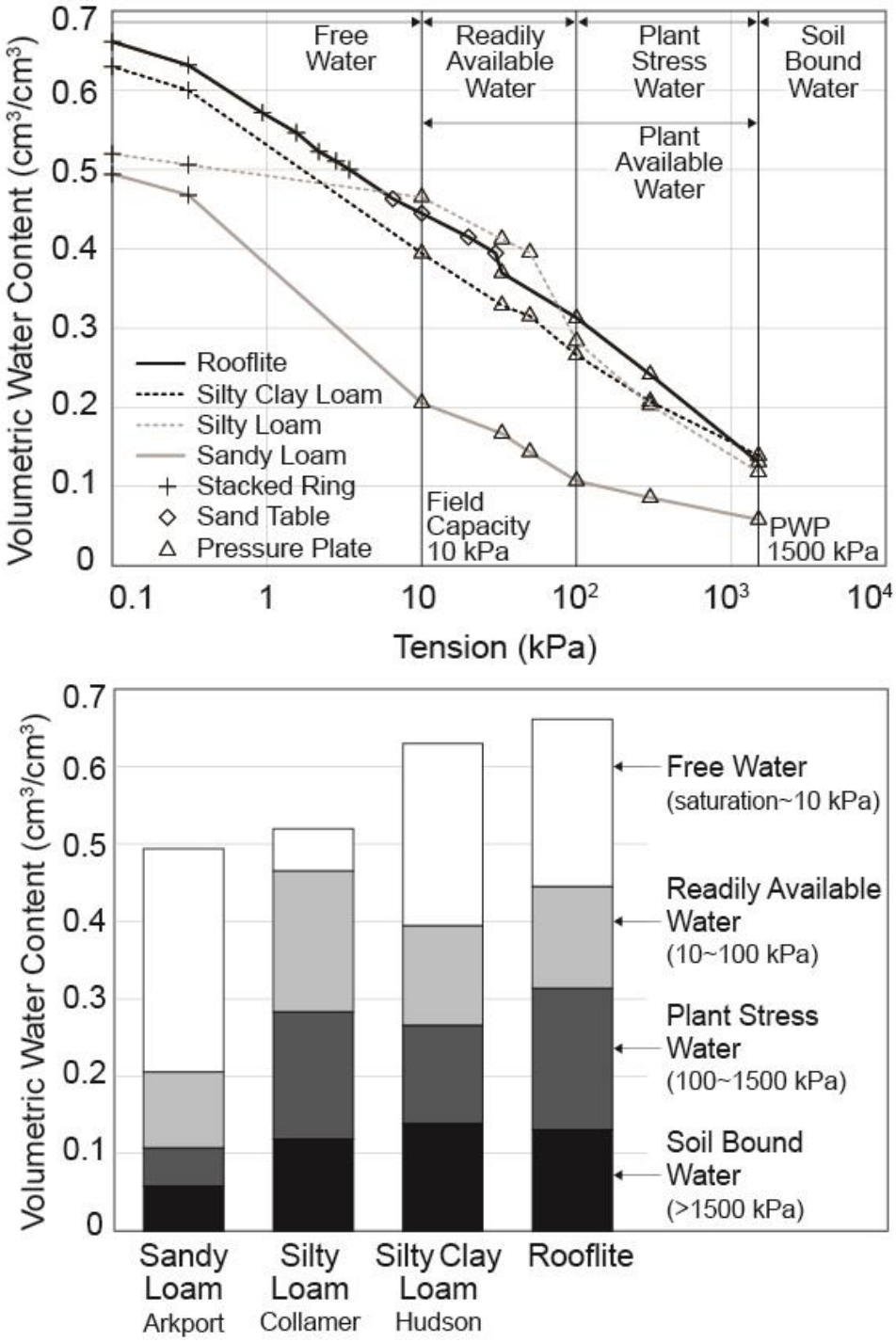
2-4. Results

Moisture Release Curve. VWC was 13.1% at the permanent wilting point (PWP, 1500 kPa) (Figure 2-2). During the growing season, irrigation must keep the soil above the PWP because in concept, once soil water reaches PWP, plants cannot recover from drought stress. VWC of the Rooflite at 10 kPa was 44.5%. VWC at 10 kPa is used to determine water retained at field capacity for sandy soils (Romano and Santini 2002), and is also used as field capacity in green roof research (Fassman-Beck and Simcock 2012; Rowe et al. 2014; Whittinghill et al. 2015).

Gradwell and Birrell (1979) generalized the pressure-based soil water storage and plant availability of water into 4 categories: *free water* (saturation -10 kPa), *readily available water* (10-100 kPa), *plant stress water* (100-1500 kPa), and *soil-bound water* (1500 kPa - oven dry). Readily available water (10-100 kPa) is the water retained against the force of gravity, and can be readily taken up by plants. Plant stress water (100-1500 kPa) is water held against gravity which, although available for plant uptake, requires plants to adjust osmotically and thereby involves a physiological cost (Gradwell and Birrell 1979). Readily available water plus plant stress water account for the water storage, commonly known as plant available water (10-1500 kPa)

Mineral soils retain the maximum amount of plant available water in the texture class known as silt loam. In terms of plant available water (10-1500 kPa) and readily available water (10-100 kPa), Rooflite was within the range of loam soils (Figure 2-2). The ranking of VWC of plant available water was Collamer silty loam (35%), followed by Rooflite (31%), Hudson silty clay loam (26%), and Arkport sandy loam (15%). This ranking remained the same in readily available water; Collamer silty loam (18%), followed by Rooflite (13%), Hudson silty clay loam (13%), and Arkport sandy loam (10%).

Figure 2-2. Moisture release characteristics of Rooflite, silty clay loam, silty loam, and sandy loam. Volumetric water content of each sample at saturation is plotted at 0.1 kPa.



Soil VWC. Soil VWC for the entire farm was monitored from May 27, 2014 – December 16, 2016, which includes 653 days during the growing seasons and 259 days during the off seasons. Due to equipment problems, 3, 15, and 7 days in 2014, 2015, and 2016 were excluded respectively.

At the farm scale, average soil VWC never reached PWP during the growing seasons; during the off seasons, average soil VWC reached PWP for a total of 55 days due to the soil freezing (Figure 2-3). The highest VWC 100 mm below the top of planting beds was 32.1%, but this stratum never reached 44.5% (10 kPa), which is the pressure-based estimate of field capacity. This VWC value was exceeded only once at the soil bottom (25 mm above the bottom of the planting beds) (Figure 2-3).

The spikes of soil VWC in Figure 2-3 correspond to precipitation. During the growing seasons, the daily maximum VWC increased in proportion to precipitation at both shallow and deep strata in planting beds. The effect of precipitation on the daily maximum VWC is less consistent in off seasons, in part because the dates with soil freezing or snowmelt are included (Figure 2-4).

Rainless days accounted for 71% of the growing seasons. During those dry periods, the range of average minimum and maximum VWC was 20.4-22.0% at the soil top and 24.4-25.7% at the soil bottom (Figure 2-4). These VWC ranges occurred during dry periods, reflecting the net effects of irrigation, ET, and drainage on soil water during the growing season.

Figure 3. Farm-wide average daily maximum and minimum soil volumetric water content at the soil top (100 mm below the tops of planting beds) and the soil bottom (25 mm above the soil bottom) during May 27, 2014 – December 16, 2016, at the Brooklyn Grange Navy Yard Farm, NYC.

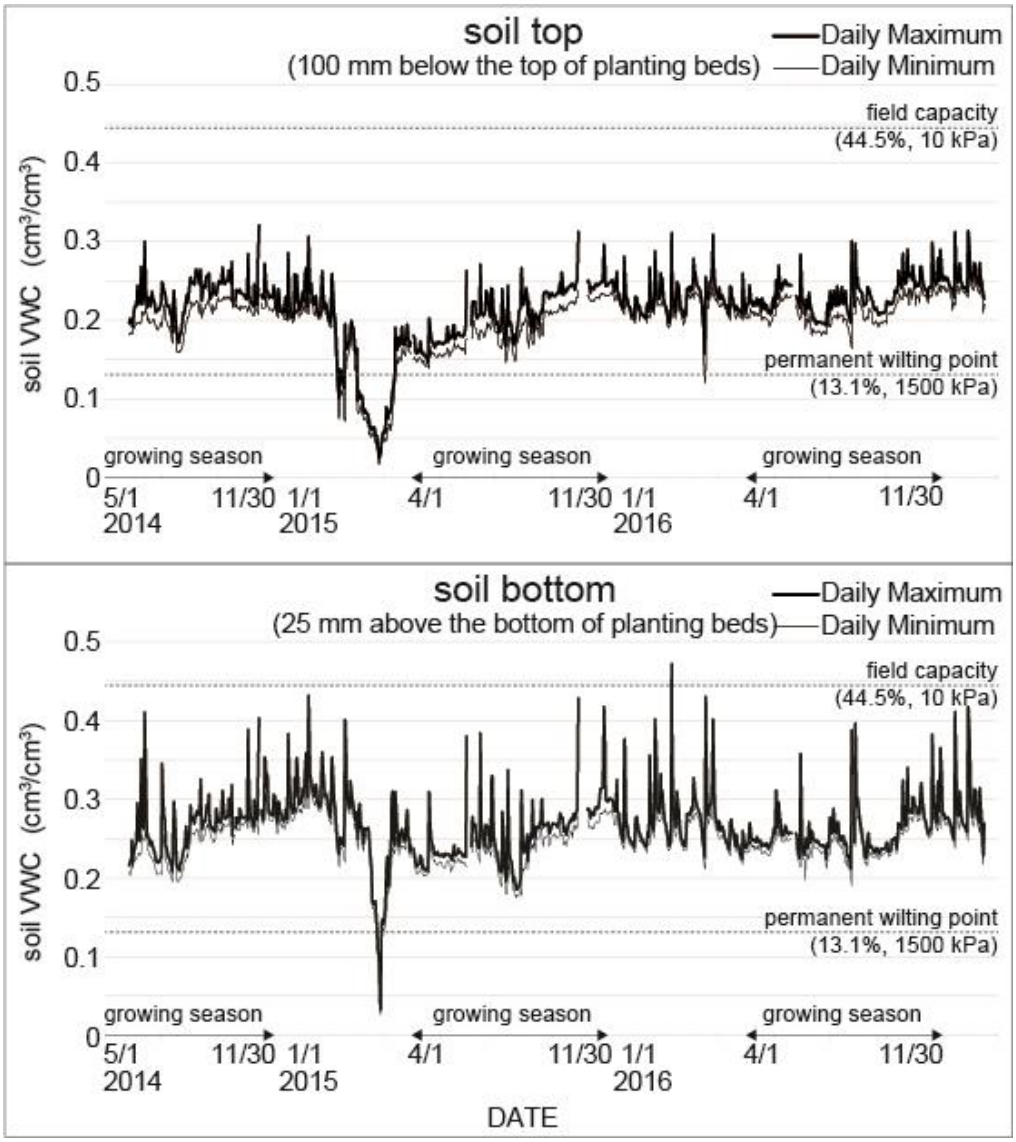
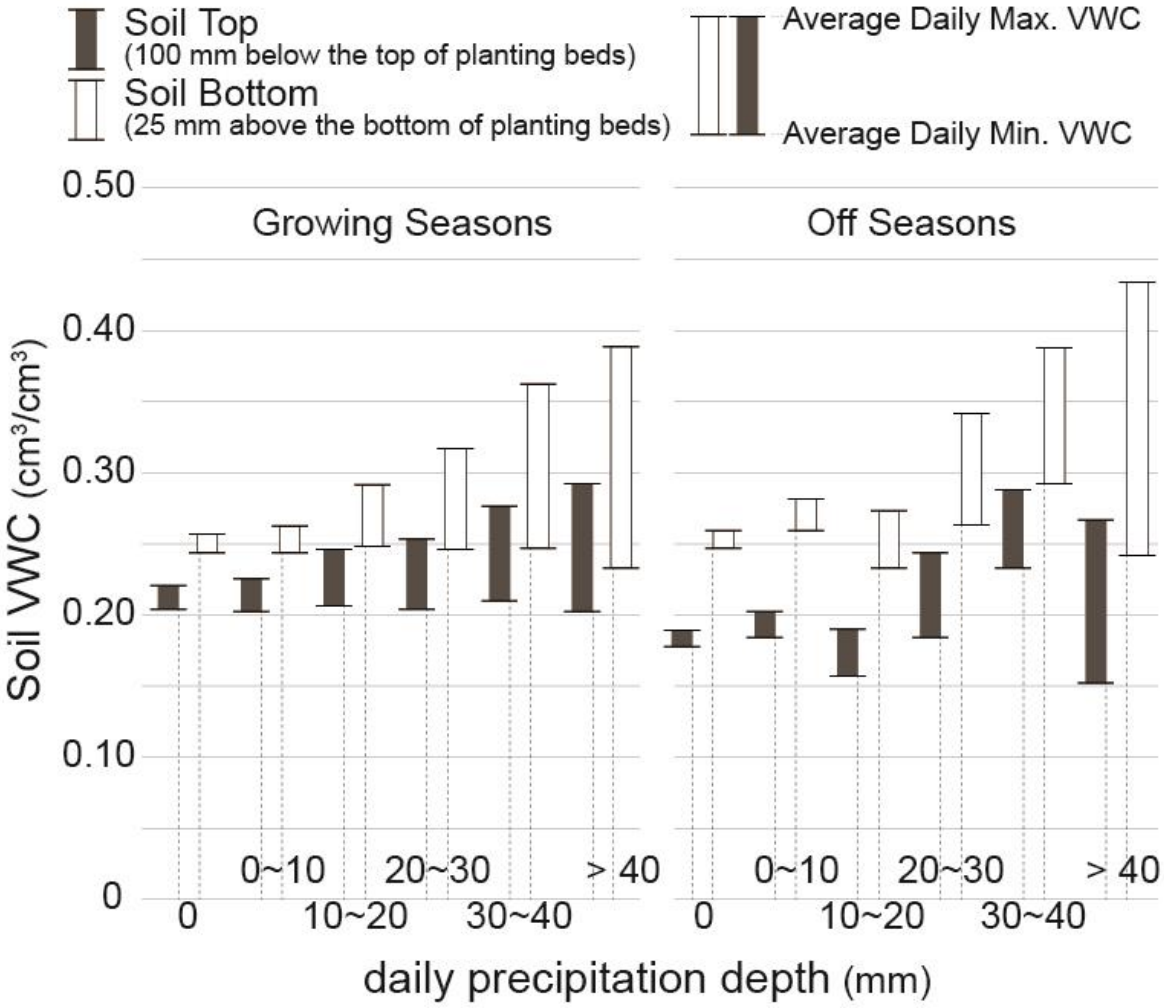


Figure 2-4. Farm-wide average daily maximum and minimum soil volumetric water contents by daily precipitation depth during May 27, 2014 – December 16, 2016, at the Brooklyn Grange Navy Yard Farm, NYC. Each growing season was between April 1 – November 30.



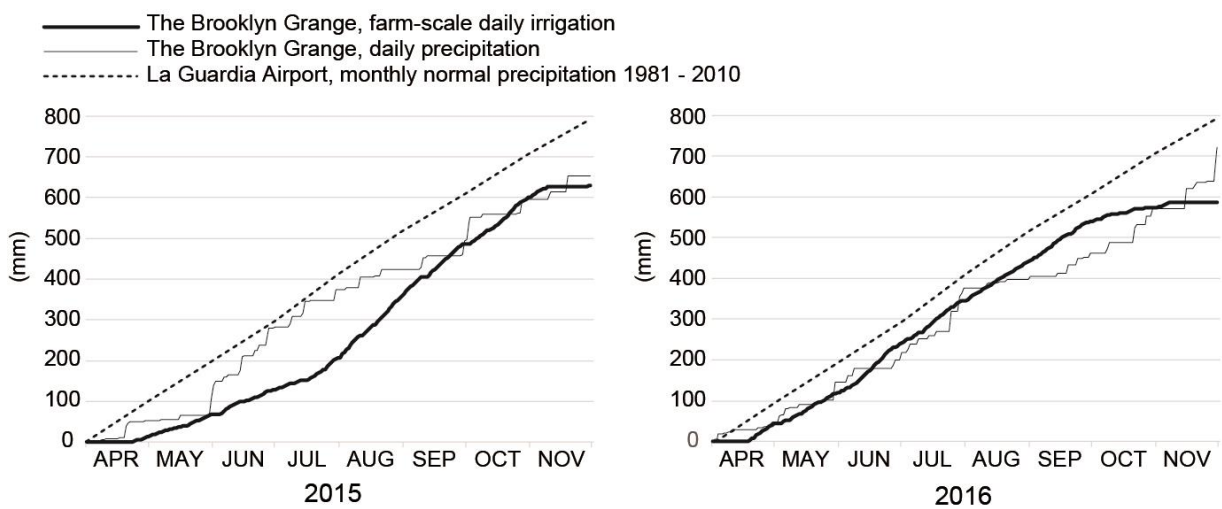
Irrigation. Irrigation is expressed for the total soil area of the farm excluding impervious areas. The daily irrigation depth was calculated as the daily total irrigation (sprinkler + drip tape) volume divided by the total area covered with soil in all 8 catchments, including both planting beds (1000-mm wide) and bare soil areas (430-mm wide). While drip tape applies water to just planting beds, sprinklers apply water to the total area covered with soil. Irrigation was monitored during the growing season (April 1 – November 30) in 2015 and 2016.

In monthly steps, irrigation rates ranged between 0.4-4.9 mm d^{-1} during the 2015 and 2016 growing seasons (Table 2-4). The average daily irrigation rate during the growing season was 2.6 mm d^{-1} in 2015 and 2.4 mm d^{-1} in 2016 (Table 2-4). During peak irrigation months, the daily irrigation rate was 4.9 mm d-1 in 2015 and 4.0 mm d-1 in 2016. During the growing seasons, the total water input (precipitation + irrigation) to the soil area was about 1300 mm in both 2015 and 2016 (Table 2-4) with irrigation accounting for approximately half (45-49%) of the total. Precipitation during the growing season was 91% in 2015, and 82% in 2016 of the 30-year normal observed at La Guardia Airport, NYC (Figure 2-5) [National Oceanic and Atmospheric Administration, National Climatic Data Center (NOAA NCDC)], which is 12 km away from our study.

Table 2-4. Farm-wide irrigation and precipitation in the growing season 2015 and 2016, at the Brooklyn Grange Navy Yard Farm, NYC.

	Growing Season		Highest Irrigation Month		Lowest Irrigation Month	
Year	2015	2016	2015	2016	2015	2016
Month(s)	Apr - Nov	Apr – Nov	Aug	Jun	Apr	Nov
Total Irrigation (mm)	626.7	586.9	152.3	119.5	13.2	13.1
Total Precipitation (mm)	653.6	720.7	50.7	56.0	52.8	149.5
Total Irrigation + Precipitation (mm)	1280.3	1307.6	203.0	175.5	66.0	162.6
Daily Average Irrigation (mm d^{-1})	2.6	2.4	4.9	4.0	0.4	0.4
Daily Average Precipitation (mm d^{-1})	2.7	3.0	1.6	1.9	1.8	5.0
Daily Average Irrigation + Precipitation (mm d^{-1})	5.2	5.4	6.5	5.9	2.2	5.4

Figure 2-5. Farm-wide cumulative daily irrigation and precipitation during the growing seasons 2015 and 2016 at the Brooklyn Grange Navy Yard Farm, NYC. 30-year (1981 - 2010) monthly normal precipitation at La Guardia Airport (NOAA NCDC). The irrigation depth was calculated as the farm-scale irrigation volume divided by the total area covered with soil.



Water Balance. The water balance was calculated for catchment 8, the only catchment equipped with a drainage monitor. Drainage volume includes impervious areas, planting beds, and bare soil between planting beds. A total of 475 days was used in the analysis. The irrigation depth was calculated as the total irrigation volume applied to the catchment divided by its total area, not just the area covered with soil (Table 2-5). Actual evapotranspiration (AET) was calculated as the total water input (precipitation + irrigation) minus drainage (Table 2-5). Except for the reference ET (ETr) from the planting beds shown for general comparison, values shown in Table 2-5 are the average of the entire catchment 8.

Over the entire study period including off seasons, discharge was 1.10 × precipitation, and 2.53 × irrigation (Figure 2-6). Discharge was 77% of the total water input (precipitation + irrigation) (Figure 2-6). Based on linear regression, response of daily discharge to daily water input was about 0.61 whether precipitation alone or with irrigation was used as water input (Figure 2-7). Based on NRCS's (1986) method (Table 2-3) a runoff curve number of 96 had the minimum root mean square error (RMSE) between observed and estimated daily discharges in response to daily precipitation with or without irrigation. In terms of the daily water balance, drainage was insensitive to irrigation even though irrigation accounted for 30% of catchment-scale water input (Table 2-5).

In the cumulative daily water flux during the partial growing season 2015, discharge was 1.66 × precipitation, and was 77% of the total water input (Table 2-5). Monthly precipitation of August 2015 was 50.7 mm at the Grange, and was about half of the 30-year normal monthly precipitation (104.7 mm) observed at La Guardia Airport, NYC (Figure 2-5). In response to this low precipitation, the Grange farmers increased the irrigation rate across the farm, leading to an artificially large discharge volume during the partial growing season 2015. During the growing season 2016, drainage equalled 96% of precipitation and 64% of the total water input.

Table 2-5. Water balance of catchment 8 during July 17, 2015 – December 16, 2016, the Brooklyn Grange Navy Yard Farm, NYC. Actual evapotranspiration (AET) was calculated as irrigation + precipitation - discharge. ET_r for planting bed area is shown for general comparison, and is the 0.12-m tall grass reference surface by ASCE Penman-Monteith method. Except ETr, each flux is the average of the entire catchment 8 including impervious area.

Period	Days	Precipitation mm (mm d^{-1})	Irrigation mm (mm d^{-1})	Discharge mm (mm d^{-1})	AET mm (mm d^{-1})	ET_r mm (mm d^{-1})
Partial Growing Season 2015 (7/17 – 11/30 2015)	104	190 (1.8)	217 (2.1)	316 (3.0)	92 (0.9)	357 (3.4)
		408 (3.9)				
Off Season 2015-16 (12/1 2015 – 3/31 2016)	122	347 (2.8)	0 (0)	370 (3.0)	NA	NA
		347 (2.8)				
Growing Season 2016 (4/1 – 11/30 2016)	233	691 (3.0)	337 (1.4)	664 (2.8)	365 (1.6)	902 (3.9)
		1028 (4.4)				
Partial Off Season 2016 (11/30 – 12/16 2016)	16	37 (2.3)	0 (0)	55 (3.4)	NA	NA
		37 (2.3)				
Entire Study Period (7/17, 2015 – 12/16, 2016)	475	1265 (2.7)	554 (1.2)	1404 (3.0)	NA	NA
		1819 (3.8)				

Figure 2-6. Cumulative daily precipitation, irrigation, discharge, and total water input (precipitation + irrigation) at catchment 8 during July 17, 2015 – December 16, 2016, the Brooklyn Grange Navy Yard Farm, NYC. All fluxes are expressed for the total area of catchment 8 including impervious area.

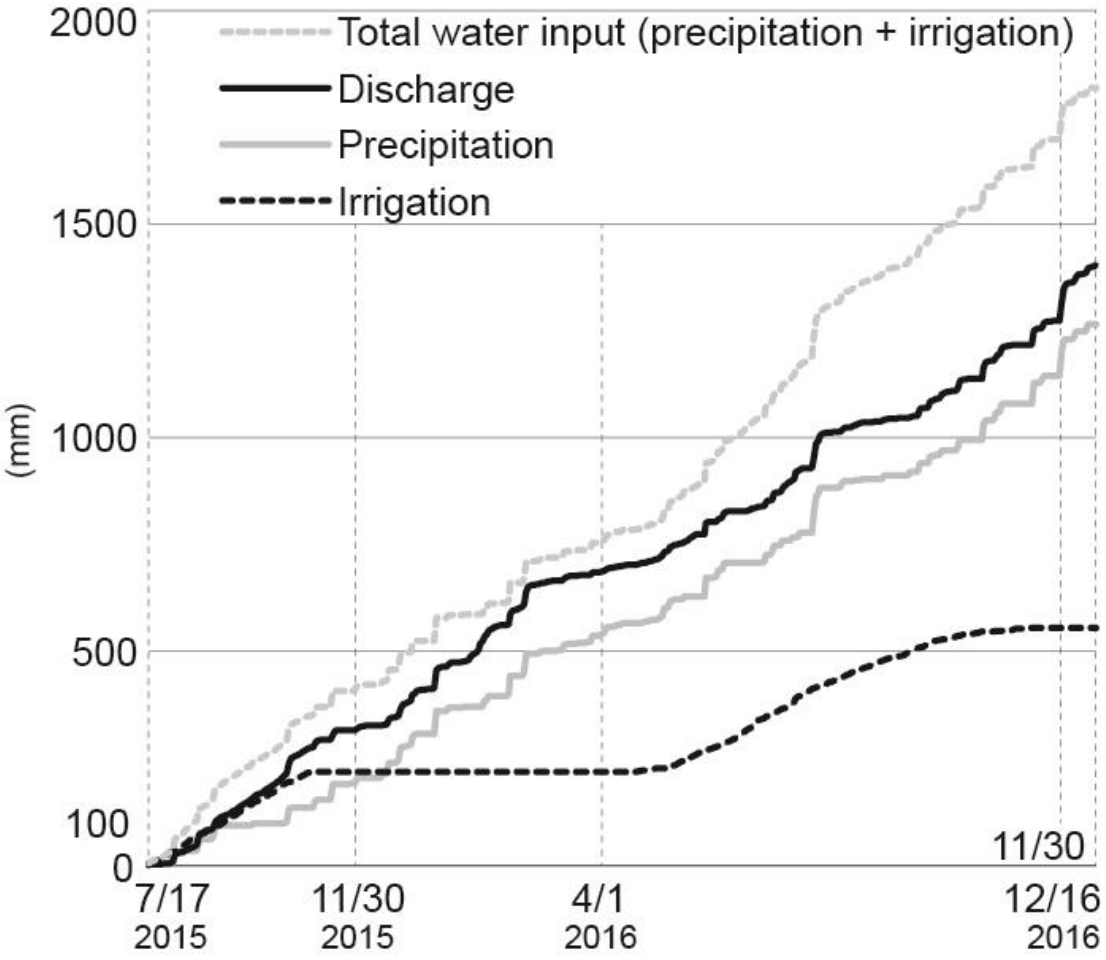
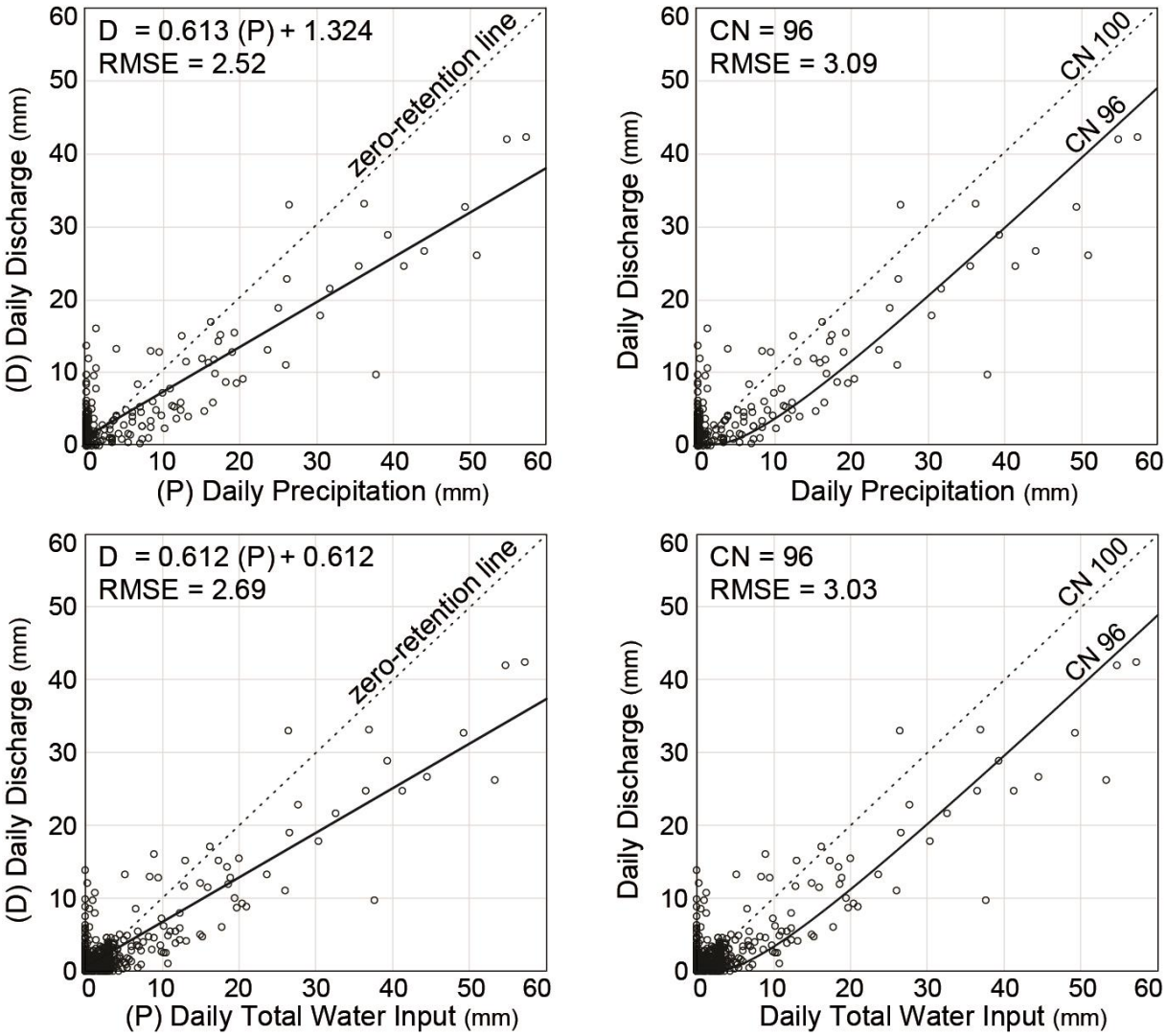


Figure 2-7. Daily discharge and precipitation with and without irrigation during July 17, 2015 – December 16, 2016, the Brooklyn Grange Navy Yard Farm, NYC. Daily total water input is daily precipitation + daily irrigation.



Historic Context of Precipitation. Precipitation patterns during the growing season (April 1 – November 30) were compared between the 50-year (1967 – 2016) averages and 2016, between the Grange, Central Park, La Guardia Airport, and JFK Airport (NOAA NCDC), all of which are within 17 km from each other. Only the Grange lacks a 50-year observation record.

For the 3 regional reference sites, precipitation during the growing season 2016 (583-745 mm) was lower than 50-year (1967 – 2016) average of the growing season (743-879 mm) (Figure 2-8). During the growing season 2016, the precipitation at the Grange (691 mm) was within the range of 3 reference sites (583-745 mm) (Figure 2-8). Assuming that the precipitation patterns are normally similar between the Grange and the 3 reference sites in NYC, the Grange received less precipitation during our study period in comparison to the 50-year (1967 – 2016) normal.

In terms of total water input, irrigation made the Grange different from unirrigated systems in NYC. Including irrigation, the total water input at the Grange during the growing season 2016 was 1028 mm, exceeding the 50-year average precipitation of the growing seasons in each of the 3 reference sites (743-879 mm) (Figure 2-8). Among the 3 reference sites, days when precipitation < 10 mm contributed 158-201 mm of precipitation during the growing season 2016, while that of the Grange was 414 mm, if irrigation was included (Figure 2-8).

The precipitation return frequency was calculated by using the 50-year observations at Central Park, La Guardia Airport, and JFK Airport (Figure 2-9). During the entire study period (475 days, including off seasons) of water balance at the Grange, only 7 days had precipitation exceeding a 1-year return frequency at one or more of the reference sites, and none exceeded a return frequency of 2 years.

Figure 2-8. Total precipitation and contribution of daily precipitation depths during the growing season (April 1 – November 30) at the Brooklyn Grange Navy Yard Farm, Central Park, La Guardia Airport, and JFK Airport, between the growing season 2016 and average 50-year observations (1967 – 2016) (NOAA NCDC).

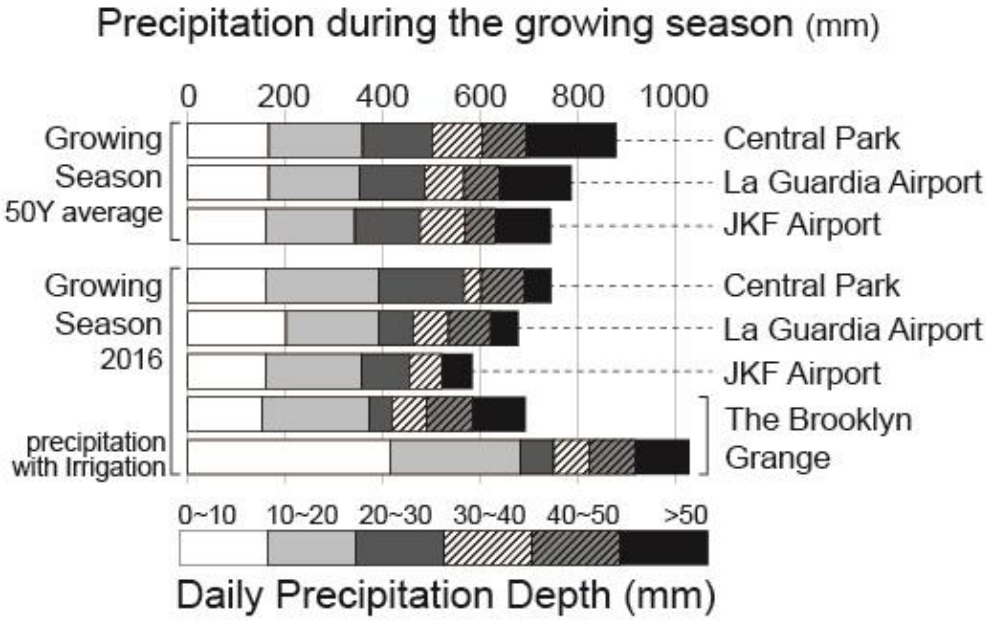
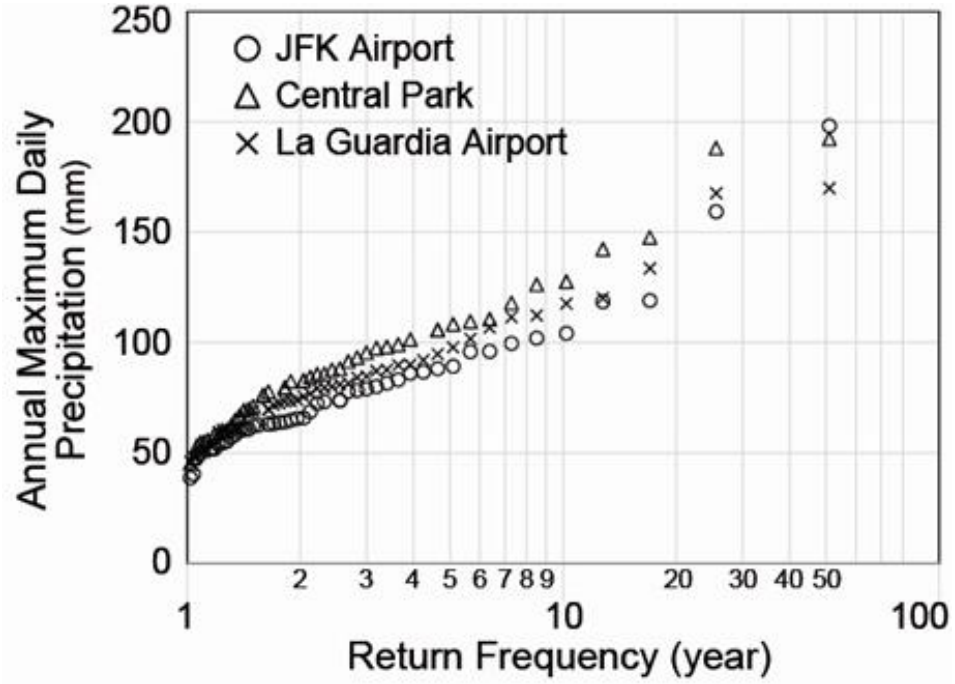


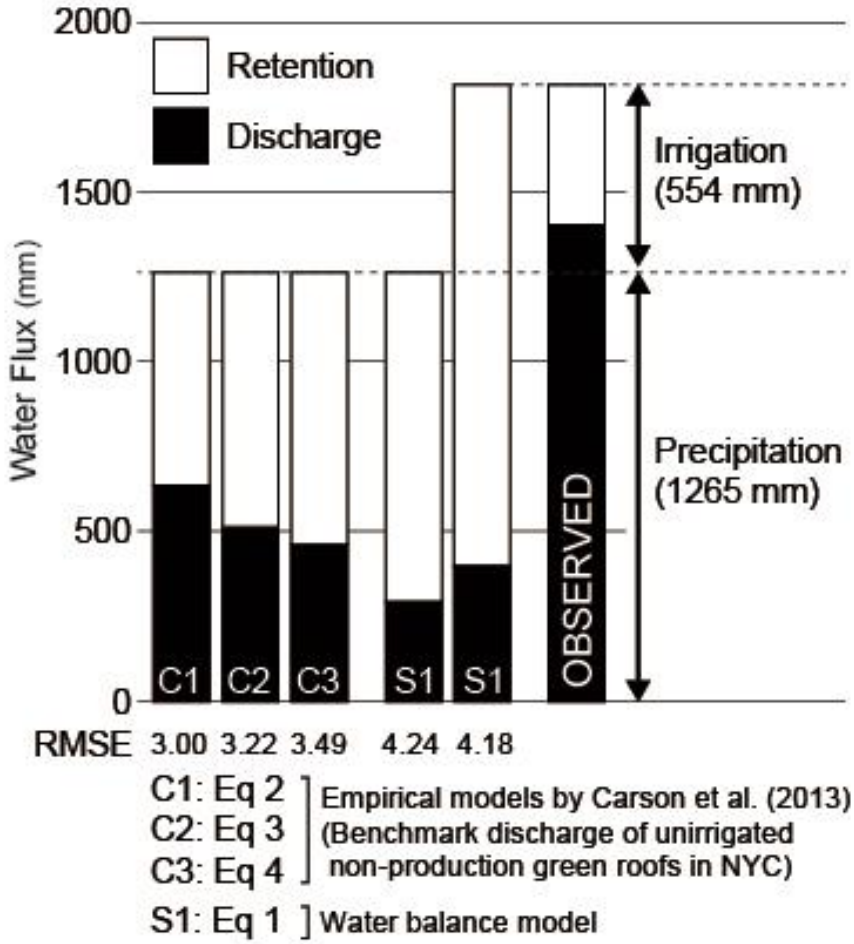
Figure 2-9. Precipitation return frequency at JFK Airport, Central Park, and La Guardia Airport, NYC by using 50-year (1967 – 2016) (NOAA NCDC).



Discharge Simulation. Moisture release curves provided a pressure-based estimate of field capacity of 44.5% at 10 kPa. To see whether this estimate explains observed discharge in the field, we simulated discharge by a daily water balance model (Table 2-3) which calculates daily water retention capacity as field capacity minus soil water at midnight (Sherrard and Jacobs 2011). Discharges from impervious area and bare soil between planting beds were estimated by NRCS's (1986) method using runoff curve number (CN) 98 and 96 respectively (Table 2-3). When daily precipitation was used as the only water input, simulated cumulative discharge was 21% of the observed discharge (Figure 2-10), increasing to 28% of the observed discharge when daily precipitation + irrigation was used as water input (Figure 2-10). These overestimations of retention capacity indicate the soil (Rooflite) under field conditions could reach field capacity at lower moisture levels than the pressure-based estimates. To calibrate field capacity in the model, we tested varying hypothetical VWC values for field capacity in the water balance model (Eq 1), using daily precipitation with irrigation as water input. Between observed and simulated discharges, the minimum RMSE (2.53) was attained when 21.9% was used as VWC at field capacity.

In the practice of stormwater management, drainage from the Grange is likely to be compared with the discharge from unirrigated green roofs receiving only precipitation. To estimate this comparative benchmark, we used observed daily precipitation data at the Grange, and simulated discharges from 3 unirrigated green roofs growing sedums, all of which had been empirically modeled in unrelated study by Carson et al. (2013) within 12 km from the Grange (Eq 2, 3, 4). Among these 3 models, simulated cumulative discharge during 475 days of study period was all below 50% of precipitation (Figure 2-10).

Figure 2-10. Simulated and observed cumulative discharge at the Brooklyn Grange Navy Yard Farm, NYC. Precipitation with and without irrigation was used as water input for the daily water balance model (Eq 1). Only precipitation was used as water input for empirical models (Eq 2, 3, 4) by Carson et al (2013) providing the comparative benchmark stormwater reduction for unirrigated sedum green roofs. Root mean square error (RMSE) is between observed and simulated discharges.

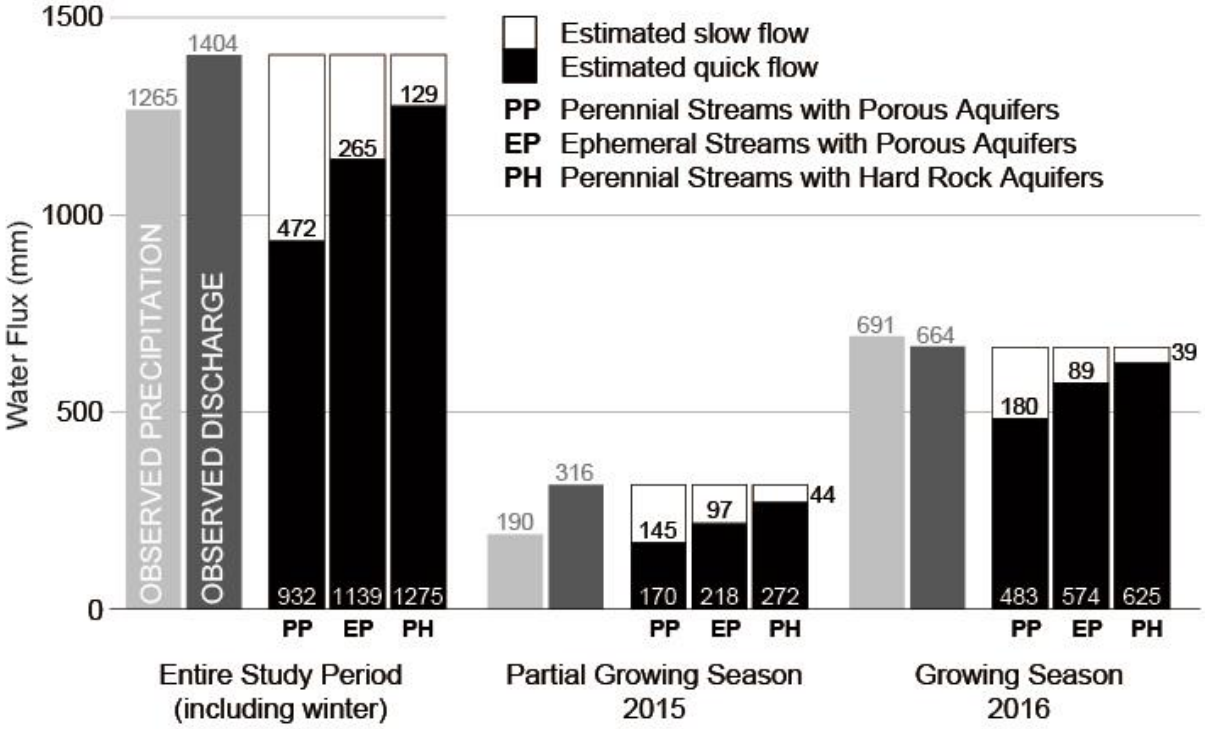


Depression Storage and Slow Flow Separation. Small irregularities in impervious surfaces contain pockets of water known as depression storage. With direct exposure to atmosphere, surface evaporation can effectively recover the stormwater retention capacity of depression storage in ballasted (gravel) and asphalt rooftops, which can retain 9-58% of stormwater (Berghage et al. 2009; Carpenter and Kaluvakolanu 2010; DeNardo et al. 2005; Mentens et al. 2006; VanWoert et al. 2005). Even with restricted exposure to the atmosphere, depression storage in the drainage layer increased the stormwater retention of unirrigated extensive green roofs with a total depth of vegetated sedum mat and geo-composite drainage material of 32-mm (Carson et al. 2013).

At the Grange, 440 out of 475 days had drainage discharge, suggesting that depression storage in the drainage layer was saturated for at least 90% of the time. In comparison to unirrigated green roofs with shallow (< 150 mm) soil, depth over the water-proofing membrane was insulated from ET by a deeper drainage layer (100-150 mm, filled with expanded shale), filter fabric, and deeper soil (250 mm), which also received irrigation during the growing seasons. The depression storage of the drainage layer seemed to contribute little to the stormwater reduction at the Grange.

The deep drainage layer does appear to extend the time during which discharge occurred. Discharge immediately increased in response to precipitation while discharge over longer periods continued even during rainless periods due to irrigation. To separate these slow and quick flows by discharge response time, we compared Eckhardt's (2005) digital filter methods (Table 2-3) for a perennial stream with a porous aquifer (PP), an ephemeral stream with a porous aquifer (EP), and a perennial stream with a hard-rock aquifer (PH). Although none of these is an exact analogue to a rooftop, it sets a context comparing the Grange with natural watersheds which have been studied in hydrologic engineering. Among these 3 methods, cumulative quick flow was at least 70% of precipitation, during the partial growing season 2015 (89-143%), the growing season 2016 (70-90%), or the entire study period (74-101%) including winter (Figure 2-11).

Figure 2-11. Estimated slow flow and quick flow at catchment 8 at the Brooklyn Grange Navy Yard Farm, NYC. Eckhardt's (2005) digital filter was used for flow disaggregation.



2-5. Discussion

Stormwater Reduction. The Grange reduced little stormwater. Cumulative discharge was 96% of precipitation during the growing season 2016. Over the entire study period including winter, cumulative discharge was 1.1 X precipitation, indicating that the Grange was a net source of discharge to the sewer system, not a reservoir for stormwater. Simulated cumulative discharge from unirrigated green roofs were less than 50% of observed precipitation at the Grange, which is less than the cumulative quick flow (over 70% of precipitation) separated by Eckhardt's (2005) digital filter. Even if only quick flow is counted as stormwater discharge for the Grange, unirrigated green roofs are likely to be more effective at reducing stormwater inputs to the sewer system than an irrigated farm.

A runoff CN of 96 best fit the observed discharge and precipitation both with and without irrigation at the Grange. However, linear regression of daily estimates against observations had smaller RMSE values and CNs are not to express discharge patterns of the Grange due to the continuous discharge. A CN of 96, while among the highest reported for 21 unirrigated green roofs, is still in this range. The best lesson to be learned from all these studies, including ours, is that green roofs may do little, if anything, to reduce stormwater discharge. It must be recognized that curve numbers do not reflect varying discharge response of green roofs due to the curve shapes and the forced initial abstractions inherent in the CN model (Fassman-Beck et al. 2015).

***Field Capacity*.** Based on the sand table measurements, we found that a VWC of 44.5 % at 10 kPa overestimated the storage. The VWC at field capacity calibrated by the same daily water balance model was 21.9%. VWC at 10 kPa is used to estimate field capacity for sandy soils (see Romano and Santini 2002). This method is also used in green roof research (see Rowe et al. 2014; Whittinghill et

al. 2015), and Fassman-Beck and Simcock (2012) reports VWC at 10 kPa provided reasonable estimates of soil water storage for unirrigated sedum green roofs. In Fassman-Beck and Simcock (2012), however, soil (100-150 mm) overlays filter fabric and plastic drainage layer ("egg-crate" and "rigid cell" type, depths < 30 mm). At the Grange, deeper soil (250 mm) overlays filter fabric and deeper drainage layer filled with expanded shale (100-150 mm). Unlike a plastic drainage layer, expanded shale in drainage layer could reduce the effect of a capillary barrier at the soil bottom, and depth of the entire profile (350-400 mm) increases the gravitational head that generates drainage. Our findings suggest the important lesson that the hydrologic design of a soil for green roofs, including rooftop farms, must incorporate the subsoil components, not just the soil layer. Further research is needed on the effects of hydrophobicity, soil particle size, and preferential flow on the water storage of ESCS-based soils.

Forschungsgesellschaft Landschaftsentwicklung Landschaftsbau (FLL) specifies the testing protocols for materials used in green roof industry. FLL (2002) defines field capacity as the VWC after 2 hours of free drainage following saturation. However, Fassman-Beck and Simcock (2012) found FLL (2002) method overestimated water storage of ESCS-based green roof mixes in field condition. Based on the FLL (2002) method, the manufacturer reports that Rooflite has a field capacity between 45-65% VWC (Skyland USA LLC 2015), which is higher than pressure-based estimate in our study (VWC 44.5% at 10 kPa), and overestimates up to 3 X the soil water storage calibrated by the daily water balance model.

Irrigation. During the partial growing season of 2015, cumulative discharge (361 mm) exceeded precipitation (190 mm), and at least 171 mm of the discharge was contributed by irrigation. This period began when drainage was instrumented during two days without discharge. Cumulative discharge in the partial growing season 2015 is less likely to include delayed discharge from the previous period.

To separate slow flow from quick flow, we used 3 models of Eckhardt's (2005) digital filter (PP: Perennial streams with porous aquifers, EP: Ephemeral streams with porous aquifers, PH: Perennial streams with hard rock aquifers). Digital filters disaggregate discharge by discharge response time, and not by discharge source. Among these 3 models, discharge composition (i.e., slow flow vs. quick flow) by the PP model most closely parallels the relative contributions of irrigation and precipitation (Table 2-6). The percentage of cumulative irrigation in total water input (irrigation + precipitation) decreased from 53% in 2015 to 33% in 2016. Based on PP model, percentage of cumulative slow flow to total discharge decreased from the 2015 and 2016 growing seasons. If slow flow separation by the PP model was sensitive to irrigation, over half of irrigation was lost to drainage at the Grange during the partial 2015 and 2016 growing seasons (Table 2-6). In summary, much water could be wasted as discharge when green roofs are irrigated.

Table 2-6. Comparison of cumulative irrigation, total water input, slow flow, and total discharge during the growing seasons at the Brooklyn Grange Navy Yard Farm, NYC. Total water input* = irrigation + precipitation. Eckhardt's (2005) digital filter methods** are PP (perennial streams with porous aquifers), EP (ephemeral streams with porous aquifers), and PH (perennial streams with hard rock aquifers).

	Eckhardt's (2005)** Digital Filter Methods	Partial Growing Season 2015	Growing Season 2016
$\frac{\text{cumulative irrigation}}{\text{cumulative total water input*}}$	-	53 %	33 %
$\frac{\text{cumulative slow flow}}{\text{cumulative total discharge}}$	PP	46 %	27 %
	EP	31 %	13 %
	PH	14 %	6 %
$\frac{\text{cumulative slow flow}}{\text{cumulative irrigation}}$	PP	67 %	54 %
	EP	45 %	27 %
	PH	20 %	12 %

Evapotranspiration. It is important to understand the contribution of AET to the water balance due to its implications for the urban heat island effect. To facilitate comparison with studies of unirrigated green roofs in the Northeast U.S., Table 2-7 reports AET at the Grange during the 2016 growing season in terms of both the total area covered with soil and just the area occupied by planting beds. It is difficult to directly compare AET from extensive green roofs and a production farm, but assuming that the unit of analysis is the total area covered with soil, not just planting beds, AET from unirrigated extensive green roofs in the northeastern US ranges between 0.84-3.15 mm (Gregoire and Clausen 2011; Sherrard and Jacobs 2011), which brackets the 2.97 mm d^{-1} observed at the Grange (Table 2-7).

The crop coefficient for sedum should be lower than for vegetables (0.5 vs 1.0) because Crassulacean Acid Metabolism (CAM) allows sedums to keep stomata closed during the day, thus reducing water loss due to transpiration (Bryla et al. 2010; Sherrard and Jacobs 2011). During the April-September growing period, however, Marasco et al. (2015) report crop coefficient for sedums of 1.02, slightly exceeding the 0.94 observed at the Grange even when expressed for the area occupied by planting beds, instead of the total area covered with soil. The lesson to be learned from all these studies and our findings is that ET from a rooftop farm is not necessarily higher than from unirrigated green roofs.

Table 2-7. Comparison of the average daily AET of the Brooklyn Grange Navy Yard Farm, NYC and the unirrigated green roofs growing sedums in the Northeast U.S. *AET is expressed both for the total area covered with soil (planting bed area + bare soil area) and for just planting bed area. Mixed-crop coefficients are shown for the Brooklyn Grange. **Dynamic chamber method directly measured water vapor flux in enclosed sections of green roofs in Marasco et al (2015)

Study		Average ET (mm d^{-1})	Crop Coefficient (Kc)	Method	Vegetation	Location	Study Period (MM / YY)
Gregoir et al (2011)	including off season	1.6	unreported	Lysimeter	Sedum spp.	Storrs, CT	09/09 – 01/10
Sherrard et at (2011)	partial growing season	0.84	0.53			Durham, NH	08/09 – 11/09
Marasco et al (2015)	including off season	1.93	1.02	Dynamic** chamber		New York, NY	04/12 – 10/13
	partial growing season 2013 (APR – SEP)	3.15					04/13 – 09/13
Brooklyn Grange (this study)	partial growing season 2016 (APR - SEP)	2.97 area covered with soil*	0.66	Catchment water balance	vegetable crops	Brooklyn, NY	04/16 – 09/16
		4.24 planting beds*	0.94				
	full growing season 2016 (APR - NOV)	2.43 area covered with soil*	0.63				04/16 – 11/16
		3.47 planting beds*	0.90				

***Efficiency of Water Management*.** Harada et al. (2017) reported that the yield of snap beans, tomatoes, leaf lettuce, and bell peppers at the Grange exceeded the state-wide average yields of in-ground agriculture in New York, New Jersey, or California. However, the Grange abstracted little stormwater, and there was continuous discharge even during rainless days indicating that irrigation exceeded demand. We compared the Grange with in-ground agriculture from 2 viewpoints: Yield per Water Supply (YRW) and water use efficiency (WUE), given by Eq 11 and 12 respectively.

$$\text{YWS (Yield per Water Supply, kg ha}^{-1}\text{ mm}^{-1}) = \frac{\text{yield (kg ha}^{-1})}{\text{cumulative (irrigation + precipitation) (mm)}} \quad \text{(Eq 11)}$$

$$\text{WUE (Water Use Efficiency, kg ha}^{-1}\text{ mm}^{-1}) = \frac{\text{yield (kg ha}^{-1})}{\text{cumulative AET (mm)}} \quad \text{(Eq 12)}$$

We used irrigation, precipitation, and AET (irrigation + precipitation – discharge) observations from our water balance observations in catchment 8 along with the farm-wide yield data from the 2016 growing season. Water flux were expressed both for the total area covered with soil (planting bed + bare soil), and for just the planting bed area occupied by plants (Table 2-3). Both expressions are shown in Figure 2-12.

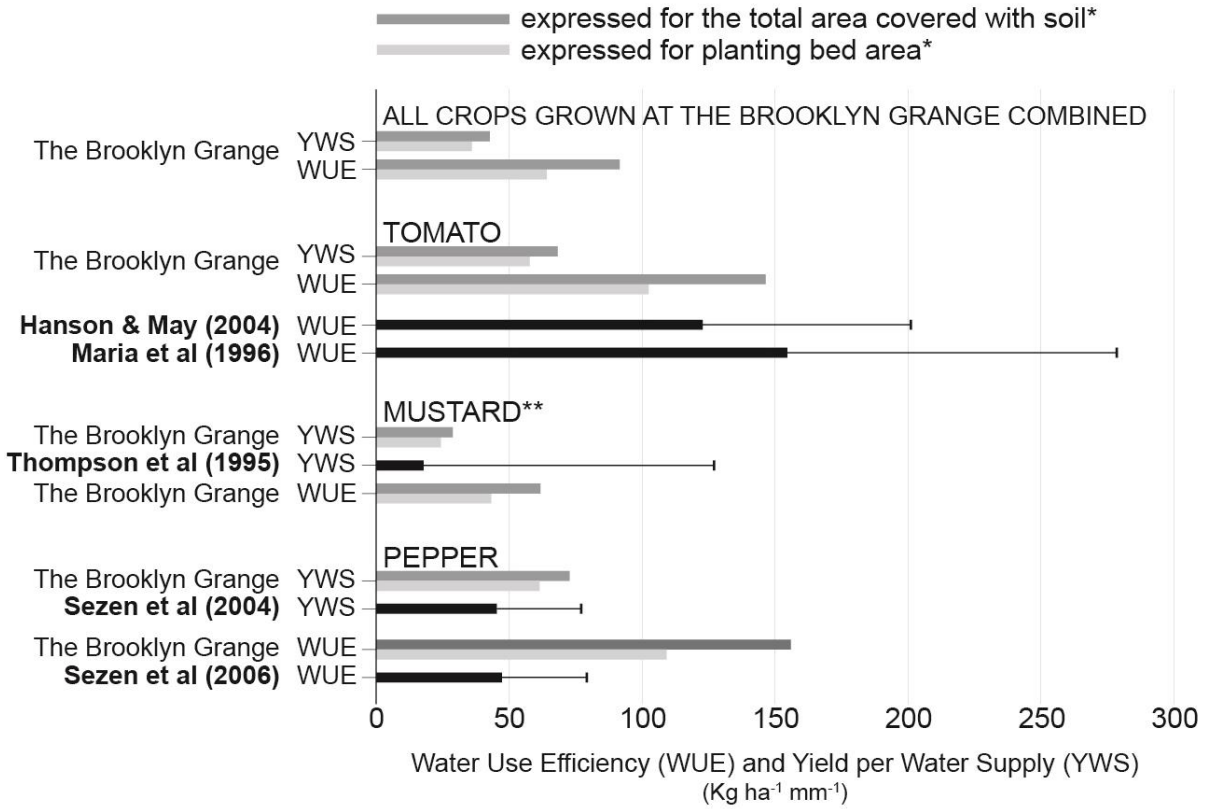
WUE of tomato at the Grange (< 150 Kg ha^{-1} mm^{-1}) was lower than the maximum values reported by Hanson and May (2004) (201 Kg ha^{-1} mm^{-1}) in southern California or by Maria do Rosário et al. (1996) (279 Kg ha^{-1} mm^{-1}) in Coruche Portugal (Figure 2-12). At the Grange, average VWC in the planting beds during rainless periods was in the range of plant stress water, which may have reduced WUE.

YWS of mustard (< 30 Kg ha^{-1} mm^{-1}) at the Grange was less than 23% of the maximum YWS of mustard (127 Kg ha^{-1} mm^{-1}) reported by Thompson and Doerge (1995) in southern Arizona (Figure 2-12). Unlike WUE, YWS can be lowered by reducing drainage losses. At the Grange, water lost to drainage was 65% of total water supplied during the 2016 growing season (Table 2-5).

YWS of pepper at the Grange was similar to the maximum value reported by Sezen et al. (2006) in Turkey while WUE exceeds the maximum reported in the same study (Figure 2-12). Whether the Grange or in-ground agriculture is more efficient in water management, likely depends on crop selection.

In summary, 65% of total water input was lost as discharge during the 2016 growing season, suggesting that there are opportunities to either reduce or reuse this discharge. Although there are studies of the potential benefits from recycling stormwater in rooftop vegetable production (see Specht et al. 2014; Thomaier et al. 2015; Ackerman et al. 2013), empirical studies from full-scale operational rooftop farms are needed.

Figure 2-12. Water use efficiency (WUE) and yield per water supply (YWS) compared between in-ground agriculture and the Brooklyn Grange Navy Yard Farm during growing season 2016. YWS was calculated as yield / (irrigation + precipitation). WUE was calculated as yield / (irrigation + precipitation – discharge). *WUE and YWS are expressed both for the total area covered with soil (planting beds + bare soil area) and for just planting bed area. **Yield of greens mix is used as yield of mustard. Minimum and maximum values among treatments of irrigation and / or nutrient application rates are shown for studies by others.



2-6. Conclusions

This study reports the hydrologic performance of the Brooklyn Grange, an operational rooftop farm in NYC. We found that drainage exceeded precipitation, which makes the Grange a net source of water and less effective than unirrigated non-production green roofs in reducing stormwater runoff. When the scope of hydrologic planning includes rooftop farming, traditional methods from the watershed engineering must be applied with caution, because curve numbers did not effectively express discharge patterns of the Grange. Further, pressure-based estimation of a field capacity overestimated soil water storage in the field. In terms of the water use efficiency, the Grange appears to be less efficient than in-ground vegetable farms. The Grange uses municipal potable water for irrigation, and over the half of this water was discharged to combined sewers. This means that the rooftop farms could compete with direct human consumption of water in cities. Importantly, rooftop farming is in its infancy and there are opportunities to incorporate water use efficiency into standard farm operations. Fortunately, rooftop farms are simple enough to be studied in detail, yet complex enough to provide insights into the way constructed ecosystems should be designed and managed to enhance sustainability. Further research is needed on the effects of engineered soils and subsoil components on the site-scale hydrology across different site configurations, management practices, and climate zones.

References

Ackerman K, Conard M, Culligan P, Plunz R, Sutto M-P, Whittinghill L (2014) Sustainable food systems for future cities: The potential of urban agriculture The Economic and Social Review 45:189-206

Ackerman K, Dahlgren E, Xu X (2013) Sustainable Urban Agriculture: Confirming Viable Scenarios for Production, Final Report No.13-07 vol NYSERDA No. 13-07 [Online] <http://urbandesignlab.columbia.edu/files/2015/04/2_Sustainable-Urban-Agriculture_NYSERDA.pdf> (Accessed on April 01, 2016).

Ampim PA, Sloan JJ, Cabrera RI, Harp DA, Jaber FH (2010) Green roof growing substrates: types, ingredients, composition and properties Journal of Environmental Horticulture 28:244

ASCE-EWRI (2004) Technical Committee Report of the Task Committee on Standardization of Reference Evapotranspiration. Environmental and Water Resources Institute, American Society of Civil Engineers, ASCE, Reston, Va

Berghage RD, Beattie D, Jarrett AR, Thuring C, Razaei F, O'Connor TP (2009) Green roofs for stormwater runoff control. EPA 600-R-09-026. National Risk Management Research Laboratory Office of Research and Development U.S. Environmental Protection Agency, Cincinnati, OH. Citeseer,

Berretta C, Poë S, Stovin V (2014) Moisture content behaviour in extensive green roofs during dry periods: The influence of vegetation and substrate characteristics Journal of Hydrology 511:374-386

Bryla DR, Trout TJ, Ayars JE (2010) Weighing lysimeters for developing crop coefficients and efficient irrigation practices for vegetable crops HortScience 45:1597-1604

Carpenter DD, Kaluvakolanu P (2010) Effect of roof surface type on storm-water runoff from full-scale roofs in a temperate climate Journal of Irrigation and Drainage Engineering 137:161-169

Carson T, Marasco D, Culligan P, McGillis W (2013) Hydrological performance of extensive green roofs in New York City: observations and multi-year modeling of three full-scale systems Environmental Research Letters 8:024036

Cho YY (2008) Soil depth affects the growth of lettuce and chicory in extensive green roofs HORTICULTURE ENVIRONMENT and BIOTECHNOLOGY 49:434-438

Cho YY, Choi KY, Lee Y-B (2010) Effects of irrigation methods on the growth, water holding capacity of substrate and nutrient uptake of Lettuce, Chicory and Endive grown in an extensive green roof system Horticulture Environment and Biotechnology 51:348-354

DeNardo JC, Jarrett AR, Manbeck HB, Beattie DJ, Berghage RD (2005) STORMWATER MITIGATION AND SURFACE TEMPERATURE REDUCTION BY GREEN ROOFS 48 doi:10.13031/2013.19181

Driscoll CT et al. (2015) Green Infrastructure: Lessons from Science and Practice. A publication of the Science Policy Exchange:32

Eckhardt K (2005) How to construct recursive digital filters for baseflow separation Hydrological processes 19:507-515

Eksi M, Rowe DB, Fernández-Cañero R, Cregg BM (2015) Effect of substrate compost percentage on green roof vegetable production Urban Forestry & Urban Greening 14:315-322

Elstein J, Welbaum GE, Stewart DA, Borys DR Evaluating growing media for a shallow-rooted vegetable crop production system on a green roof. In: IV International Symposium on Seed, Transplant and Stand Establishment of Horticultural Crops; Translating Seed and Seedling 782, 2006. pp 177-184

Fassman-Beck E, Hunt W, Berghage R, Carpenter D, Kurtz T, Stovin V, Wadzuk B (2015) Curve number and runoff coefficients for extensive living roofs Journal of Hydrologic Engineering 21:04015073

Fassman-Beck E, Simcock R (2012) Moisture Measurements as Performance Criteria for Extensive Living Roof Substrates Journal of Environmental Engineering 138:841-851 doi:doi:10.1061/(ASCE)EE.1943-7870.0000532

Gradwell M, Birrell K (1979) Methods for physical analysis of soils. NZ Soil Bureau, Department of Scientific and Industrial Research,

Grard BJ-P et al. (2015) Recycling urban waste as possible use for rooftop vegetable garden Future of Food: Journal on Food, Agriculture and Society 3:21-34

Gregoire BG, Clausen JC (2011) Effect of a modular extensive green roof on stormwater runoff and water quality Ecological Engineering 37:963-969

Hanson B, May D (2004) Effect of subsurface drip irrigation on processing tomato yield, water table depth, soil salinity, and profitability Agricultural Water Management 68:1-17 doi:http://dx.doi.org/10.1016/j.agwat.2004.03.003

Harada Y, Whitlow TH, Bassuk NL, Russell-Anelli J (2017) Biogeochemistry of Rooftop Farm Soils. In: Lal R, Stewart BA (eds) Urban Soils. Advances in Soil Science Series. Taylor & Francis Group, Portland, United States,

Kong AY, Rosenzweig C, Arky J (2015) Nitrogen Dynamics Associated with Organic and Inorganic Inputs to Substrate Commonly Used on Rooftop Farms HortScience 50:806-813

Landschaftsbau FL (2002) Guidelines for the planning, execution and upkeep of green-roof sites Forschungsgesellschaft Landschaftsentwicklung Landschaftsbau, Bonn, Germany

Lim KJ, Engel BA, Tang Z, Choi J, Kim KS, Muthukrishnan S, Tripathy D (2005) Automated web GIS based hydrograph analysis tool, WHAT JAWRA Journal of the American Water Resources Association 41:1407-1416

Lovell ST (2010) Multifunctional urban agriculture for sustainable land use planning in the United States Sustainability 2:2499-2522

Lovell ST, Taylor JR (2013) Supplying urban ecosystem services through multifunctional green infrastructure in the United States Landscape Ecology 28:1447-1463 doi:10.1007/s10980-013-9912-y
Lundholm JT (2015) The ecology and evolution of constructed ecosystems as green infrastructure Frontiers in Ecology and Evolution 3:106
Marasco DE, Culligan PJ, McGillis WR (2015) Evaluation of common evapotranspiration models based on measurements from two extensive green roofs in New York City Ecological Engineering 84:451-462
Maria do Rosário GO, Calado AM, Portas CAM (1996) Tomato root distribution under drip irrigation Journal of the American Society for Horticultural science 121:644-648
Mentens J, Raes D, Hermy M (2006) Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? Landscape and urban planning 77:217-226
Natural Resources Conservation Service (1986) Urban hydrology for small watersheds. U.S. Dept. of Agriculture, Conservation Engineering Division, Washington, DC.,
Ntoulas N, Nektarios PA, Kapsali T-E, Kaltsidi M-P, Han L, Yin S (2015) Determination of the Physical, Chemical, and Hydraulic Characteristics of Locally Available Materials for Formulating Extensive Green Roof Substrates HortTechnology 25:774-784
NYC DEP (2011) DEP Awards $3.8 Million in Grants for Community-Based Green Infrastructure Program Projects NYC DEP Webpage [Online] <http://www.nyc.gov/html/dep/html/press_releases/11-46pr.shtml#.VvIn6fkrKUk> (Accessed on April 01, 2016)
Oberndorfer E et al. (2007) Green roofs as urban ecosystems: ecological structures, functions, and services BioScience 57:823-833
Orsini F et al. (2014) Exploring the production capacity of rooftop gardens (RTGs) in urban agriculture: the potential impact on food and nutrition security, biodiversity and other ecosystem services in the city of Bologna Food Security 6:781-792
Paredes C, Medina E, Moral R, Pérez-Murcia MD, Moreno-Caselles J, Angeles Bustamante M, Cecilia JA (2009) Characterization of the different organic matter fractions of spent mushroom substrate Communications in soil science and plant analysis 40:150-161
Phan C-W, Sabaratnam V (2012) Potential uses of spent mushroom substrate and its associated lignocellulosic enzymes Applied Microbiology and Biotechnology 96:863-873 doi:10.1007/s00253-012-4446-9
Romano N, Santini A (2002) Water retention and storage: Field Methods of Soil Analysis, Part 4, Physical Methods (JH Dane and GC Topp, eds) 4:721-738
Rosenzweig C, Gaffin S, Parshall L (2006) Green roofs in the New York metropolitan region: research report Columbia University Center for Climate Systems Research and NASA Goddard Institute for Space Studies New York:1-59
Rowe DB (2011) Green roofs as a means of pollution abatement Environmental Pollution 159:2100-2110
Rowe DB, Kolp MR, Greer SE, Getter KL (2014) Comparison of irrigation efficiency and plant health of overhead, drip, and sub-irrigation for extensive green roofs Ecological Engineering 64:306-313
Sanyé-Mengual E, Orsini F, Oliver-Solà J, Rieradevall J, Montero JI, Gianquinto G (2015) Techniques and crops for efficient rooftop gardens in Bologna, Italy Agronomy for Sustainable Development 35:1477-1488 doi:10.1007/s13593-015-0331-0
Schwartz SS (2007) Automated Algorithms for Heuristic Base-Flow Separation JAWRA Journal of the American Water Resources Association 43:1583-1594
Sezen SM, Yazar A, Eker S (2006) Effect of drip irrigation regimes on yield and quality of field grown bell pepper Agricultural Water Management 81:115-131
Sherrard JA, Jacobs JM (2011) Vegetated roof water-balance model: experimental and model results Journal of Hydrologic Engineering 17:858-868
Shock CC, Wang F-X (2011) Soil water tension, a powerful measurement for productivity and stewardship HortScience 46:178-185
Skyland USA LLC (2015) "Rooflite Intensive Ag Product Specification" <http://www.rooflitesoil.com/products/intensive-ag>accessed June 01, 2016
Snyder R, Eching S (2008) Penman-Monteith (hourly) reference evapotranspiration equations for estimating ETos and ETrs with hourly weather data http://biomet ucdavis edu/Evapotranspiration/PM hrXLS/PMhrDoc pdf> Accessed on June 12 2014
Specht K et al. (2014) Urban agriculture of the future: an overview of sustainability aspects of food production in and on buildings Agriculture and Human Values 31:33-51 doi:10.1007/s10460-013-9448-4
Specht K, Siebert R, Thomaier S (2016) Perception and acceptance of agricultural production in and on urban buildings (ZFarming): a qualitative study from Berlin, Germany Agriculture and Human Values 33:753-769
Spolek G (2008) Performance monitoring of three ecoroofs in Portland, Oregon Urban Ecosystems 11:349-359
Stovin V, Poë S, De-Ville S, Berretta C (2015) The influence of substrate and vegetation configuration on green roof hydrological performance Ecological Engineering 85:159-172
Thomaier S, Specht K, Henckel D, Dierich A, Siebert R, Freisinger UB, Sawicka M (2015) Farming in and on urban buildings: Present practice and specific novelties of Zero-Acreage Farming (ZFarming) Renewable Agriculture and Food Systems 30:43-54 doi:10.1017/S1742170514000143
Thompson TL, Doerge TA (1995) Nitrogen and water rates for subsurface trickle-irrigated collard, mustard, and spinach HortScience 30:1382-1387

Van Dam J, Hendrickx J, Van Ommen H, Bannink M, Van Genuchten MT, Dekker L (1990) Water and solute movement in a coarse-textured water-repellent field soil Journal of Hydrology 120:359-379

VanWoert ND, Rowe DB, Andresen JA, Rugh CL, Fernandez RT, Xiao L (2005) Green roof stormwater retention: effects of roof surface, slope, and media depth Journal of environmental quality 34:1036-1044

Whittinghill L, Starry O (2016) Up on the Roof: Considerations for Food Production on Rooftops. In: Brown S, McIvor K, Hodges Snyder E (eds) Sowing Seeds in the City: Ecosystem and Municipal Services. Springer Netherlands, Dordrecht, pp 325-338. doi:10.1007/978-94-017-7453-6_23

Whittinghill LJ, Hsueh D, Culligan P, Plunz R (2016a) Stormwater performance of a full scale rooftop farm: Runoff water quality Ecological Engineering 91:195-206

Whittinghill LJ, Rowe DB, Andresen JA, Cregg BM (2015) Comparison of stormwater runoff from sedum, native prairie, and vegetable producing green roofs Urban Ecosystems 18:13-29

Whittinghill LJ, Rowe DB, Ngouajio M, Cregg BM (2016b) Evaluation of nutrient management and mulching strategies for vegetable production on an extensive green roof Agroecology and Sustainable Food Systems 40:297-318 doi:10.1080/21683565.2015.1129011

Chapter 3. Nitrogen Biogeochemistry of the Brooklyn Grange, An Urban Rooftop Farm

Abstract

Intensive agriculture represents a recent extension of green roof technology. Perceived ecosystem services provided by rooftop farming include stormwater management and the production of affordable and nutritious vegetables for local consumption. However, intensive agriculture can increase nutrient loads to surface water, yet there is little empirical data from full-scale operational rooftop farms. This study reports the N balance and N management efficiency of the Brooklyn Grange Navy Yard Farm, a 0.61-ha farm atop an 11-story building in New York City USA. We monitored atmospheric N deposition, soil N concentration, N output by harvest, N leaching from soil, and drainage N output, in addition to estimating net N mineralization and the N load to sewers during the combined sewer overflows. We found that the annual drainage N output was 1100% of the atmospheric bulk N deposition, and was 540% of the estimated total atmospheric N deposition, which makes the Brooklyn Grange a net N source in the urban environment. Annual N leaching from soil was 97% of fertilizer N input, and the efficiency of N management can be lower than of in-ground intensive vegetable production. For the Brooklyn Grange to integrate stormwater management and intensive agriculture, it will be important to use soil with larger storage capacity within readily available water, and recycle drainage. This case study shows how the intensification of agriculture on rooftops should be managed for both the yield and quality of crops, as well as for the potential for N to leave them in stormwater drainage, which can have implications for aquatic ecosystems and water quality.

3-1. Introduction

As hotspots for biogeochemical cycles, cities offer various opportunities for developing novel ecosystems to enhance sustainability (Grimm et al. 2008; Kaye et al. 2006; Lundholm 2015; Palmer et al. 2004; Pataki et al. 2011). These opportunities include green roofs to manage stormwater runoff, save energy, reduce air pollution, and preserve biodiversity (Berndtsson 2010; Mentens et al. 2006; Oberndorfer et al. 2007; Rowe 2011). More recently, rooftop farming has been explored as a way to achieve these goals and to produce food for local communities (Ackerman et al. 2013; Harada et al. 2017; Thomaier et al. 2015; Whittinghill and Starry 2016). Ideally, these diverse ecosystem services could be integrated within the limited spaces available in urban environments and to offer social-cultural benefits such as environmental education, eco-justice, and building more cohesive communities (Lovell 2010; Lovell and Taylor 2013; McPhearson et al. 2013; Specht et al. 2014). However, it can be challenging to integrate ecosystem services if each requires different design and management of ecosystems. For example, intensive agriculture can require higher nutrient inputs (and conceivably drainage outputs) to maintain yield and quality, while stormwater management aims to reduce nutrient loads to surface water by managing vegetation with limited supplemental nutrients. Drainage output of nutrients from intensive agriculture, such as nitrogen (N) from application of high amounts of fertilizer, has led to multiple negative consequences, including contamination of groundwater and development of “dead zones” such as the Chesapeake Bay, Gulf of Mexico, and Long Island Sound (Howarth 2008b; Howarth et al. 2000; Kemp et al. 2005; Rabalais et al. 2002). These nutrient loadings into waterways are regulated by the Clean Water Act (CWA) through the total maximum daily load (TMDL) and best management practices (BMPs) (Wainger 2012). Some estuaries are more sensitive to nutrient loads than others. For instance, the longer mean residence time of water in Long Island Sound (1100 days) compared to Delaware Bay (60 days) and Chesapeake Bay (250 days) makes N management particularly important to New York City, which is

a major source of N for Long Island Sound (Howarth et al. 2006; Nixon et al. 1996). In the infancy of rooftop farming practices, there are opportunities to establish and implement BMPs to limit the contribution of urban farming to nutrient loading of nearby waterways.

A variety of studies have reported rates of drainage N output from agricultural, forested, and urban land uses. Whittinghill et al. (2016) reports the rate of drainage N output from an operational rooftop farm in NYC (33.6 kg N $ha^{-1}y^{-1}$) to be about three times greater than rates of maximum drainage N output from urban watersheds (11.4 kg N $ha^{-1}y^{-1}$, except for wastewater treatment effluents), and within the range of N-saturated forests (0.04 - 47 kg N $ha^{-1}y^{-1}$) (Berndtsson et al. 2006; Campbell et al. 2004; Fenn et al. 1998; Gregoire and Clausen 2011; Groffman et al. 2004; Howarth et al. 2002; Likens 2013; Whittinghill et al. 2016). The rate of N in the rooftop farm (33.6 kg N $ha^{-1}y^{-1}$) falls within the range of in-ground agriculture (2 – 353.7 kg N $ha^{-1}y^{-1}$), in-ground organic farming (25 – 36 kg N $ha^{-1}y^{-1}$), and in-ground farming using BMPs for no-tillage system with legume cover crops (4.1 – 33.9 kg N $ha^{-1}y^{-1}$), and the low end of in-ground intensive vegetable production (32.8 – 353.7 kg N $ha^{-1}y^{-1}$) (Cameron et al. 2013; Eriksen et al. 2004; Goulding et al. 2000; Jayasundara et al. 2007; Min et al. 2012; Pärn et al. 2012; Syswerda et al. 2012; Whittinghill et al. 2016; Zhang et al. 2017; Zhao et al. 2010). These observations suggest that relatively low drainage N output from rooftop farming could be accomplished, but the best approaches to achieve this goal have not been determined.

There are many potential management practices urban farmers could use to reduce drainage output of nutrients into nearby waterways. For example, reducing drainage volume can reduce drainage N output from soil-plant systems on roofs (Cameron et al. 2013; Di and Cameron 2002; Hartz 2006). However, this effort can be difficult when the ESCS (expanded shale, clay and slate) is used as base materials in rooftop farm soils, because ESCS holds less water and nutrients than field soils, amplifying the potential for drainage output (Ampim et al. 2010; Best et al. 2015; Emilsson et al.

2007; Harada et al. 2018). Even in non-production green roofs where plant yield and quality are arguably less important, fertilizer and irrigation inputs during the initial establishment period can increase drainage output of nutrients (Berndtsson 2010; Driscoll et al. 2015; Emilsson et al. 2007; Rowe 2011). In comparison to vegetables, however, non-crop vegetation such as *Sedum* spp. requires less water and nutrients, thereby drainage output of nutrients from non-production green roof can be reduced after establishment by reducing or terminating water and nutrient subsidies (see Congreves and Van Eerd, 2015; Shock and Wang, 2011; Van Mechelen et al., 2015; Emilsson et al., 2007). In rooftop farms, however, economic viability relies on the yield and quality of diverse vegetables with high market value, such as salad greens, tomatoes, peppers, and eggplants, which have high requirements for water and nutrients (Ackerman et al. 2013; Congreves and Van Eerd 2015; Shock and Wang 2011). Since not all of the added nutrients are taken up by these vegetables, greater rates of fertilizer application can lead to greater drainage output of nutrients (Cameron et al. 2013).

Potentially mineralizable N (PMN) can be measured by the laboratory incubation of soil samples, indicating the potential of microbial activity converting soil organic N to NO_3-N and NH_4-N (Drinkwater et al. 1996). Kong et al. (2015) report PMN of a rooftop farm soil and found that rates of net mineralization fall within the ranges observed in rural forests, agricultural sites, and urban green space (Benedetti and Sebastiani 1996; Drinkwater et al. 1995; Kong et al. 2015; Poudel et al. 2002; Scharenbroch et al. 2005; Szlavecz et al. 2006). Mineralization of organic N can increase drainage N output from soil plant systems, including intensive agriculture and non-production green roofs (Buffam et al. 2016; Cameron et al. 2013; Di and Cameron 2002; Drinkwater et al. 1996). Many soil products used in rooftop farms and non-production green roofs are made of ESCS materials amended with compost, peat, coconut coir, and other sources of soil organic matter (SOM) (Ampim et al. 2010; Buffam et al. 2016; Eksi et al. 2015; Whittinghill et al. 2016). In comparison to these ESCS-based soils, potting mixes have much higher SOM, and are also used for rooftop farming (Harada et

al. 2017). Higher SOM may enhance water holding capacity of soils, reducing drainage output of water and N, but the higher SOM and the enhanced mineralization rates from the greater air and water contents, may increase drainage N output from rooftop farms. This relative contribution of added SOM to drainage N output from rooftop farms is currently unknown.

Despite the potential for rooftop farming to induce drainage N output from roofs, the recent increase in rooftop farming in places such as New York City provides the opportunity to evaluate and determine the best management practices that minimize drainage nutrient output, while maintaining crop yield and quality. Compared to traditional agricultural practices that occur on the ground, rooftop farming provides a relatively simpler and more discreet hydrologic regime with centralized drainage systems that allow for easier quantification of nutrient sources, cycling, and outputs. The creation of these novel urban ecosystems allows for precise observation and manipulation over time, making an ideal model for improving management practices for sustainable intensification of urban agriculture.

Although multiple studies have documented rates of soil mineralization and drainage N output from rooftop farming, we are not aware of any studies that have completed a total ecosystem N balance for rooftop farms. The quantification of total inputs from fertilizer and atmospheric deposition, and total outputs by drainage and harvest of vegetables, as well as internal microbial processes such as denitrification and mineralization, would provide the understanding necessary to determine controls on nutrient retention and loss that will enable creation of BMPs for rooftop farms. The objective of this paper is to quantify the N balance of a full-scale operational rooftop farm in New York City, USA. We pose the following a priori questions that this study addresses:

1. Are rooftop farms a net N sink or source?
2. What is the contribution of atmospheric deposition to total N inputs of roof top farms?
3. Is N leaching from these soils controlled by mineralization of SOM or drainage volume?

4. How much drainage N output occurs during combined sewer overflows (CSOs)?
5. What lessons can we learn from creating a total ecosystem N budget for writing BMPs for management of rooftop farms?

3-2. Materials and Methods

Site description. The study was located at the Brooklyn Grange Navy Yard farm (the Grange hereafter), atop an 11-story building in the former Brooklyn Navy Yard, NYC. The entire site area (6,122 m^2) is divided into 8 catchments (Figure 3-1), each with a drain connected to the municipal combined sewer system. Of the entire site area, 33% (2,040 m^2) of the surface is impervious and 67% (4,082m^2) is covered with soil underlain by a drainage layer (Figure 3-1).

Fertilizer is applied to only the planting bed area, which is 60% of the total area covered with soil (Figure 3-2). Over the 3-year study, annual fertilizer N input increased twofold (Figure 3-3). Synthetic N (nitrate, ammoniacal, and urea N) was applied only during the 2015 growing season and accounted for only 5% of the total N during this period. Among non-synthetic N sources summarized in Figure 3-3, the mined $NaNO_3$ was the only N source that was nearly 100% mineral N. All other N sources are organic N derived from either plant, animal, or seafood by-products, which contained NH_4-N less than 3% of the total N mass. Over the 2014 and 2015 growing seasons, 11 organic N fertilizer products were used, which were narrowed down to only Nature Safe 13-0-0 ® and Nature Safe 10-2-8 ® (Darling Ingredients Inc. Irving, TX) in the 2016 growing season, and both were made of animal by-products (Figure 3-3).

Figure 3-1. Site information for the Brooklyn Grange Navy Yard farm in New York City, NY. The top figure shows aerial photo of the roof from © google 2014. In the bottom figure, dotted lines indicate the 8 catchments on the roof, and circled numbers indicate the height in meters of atmospheric deposition collectors above the soil surface.

Location	Brooklyn Navy Yard, Brooklyn, NY		
Geographic coordinate	latitude: 40.698325, longitude: -73.97263		
Elevation above sea level	46 m (building height: 41m, ground level: 5m)		
Site Area	6122 m^2 entire site	4082 m^2 area covered with soil	2429 m^2 planting bed area
			1653 m^2 uncropped area
		2040 m^2 impervious	
Catchment slope	≈ 2%		
Completion of construction	June 01, 2012		
Growing season	April to November		

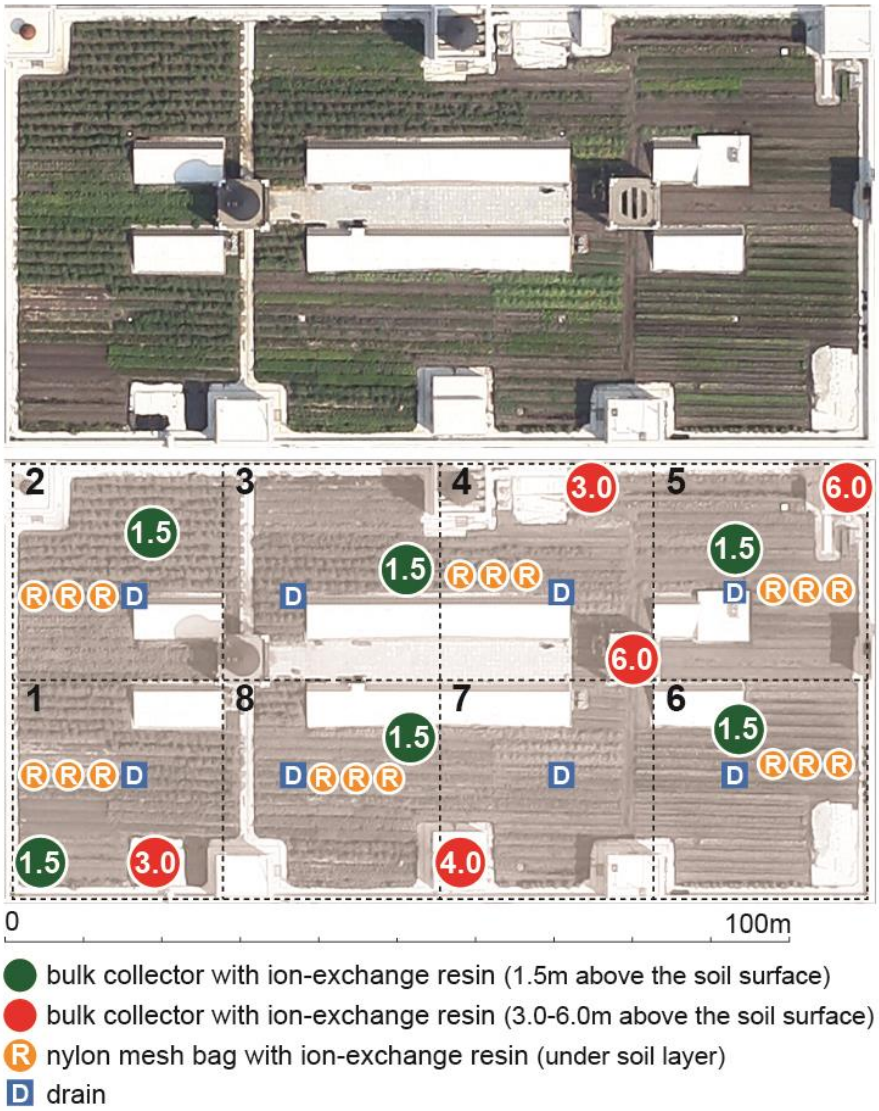


Figure 3-2. Diagramatic site section showing N fluxes of the Brooklyn Grange Navy Yard farm, NYC.

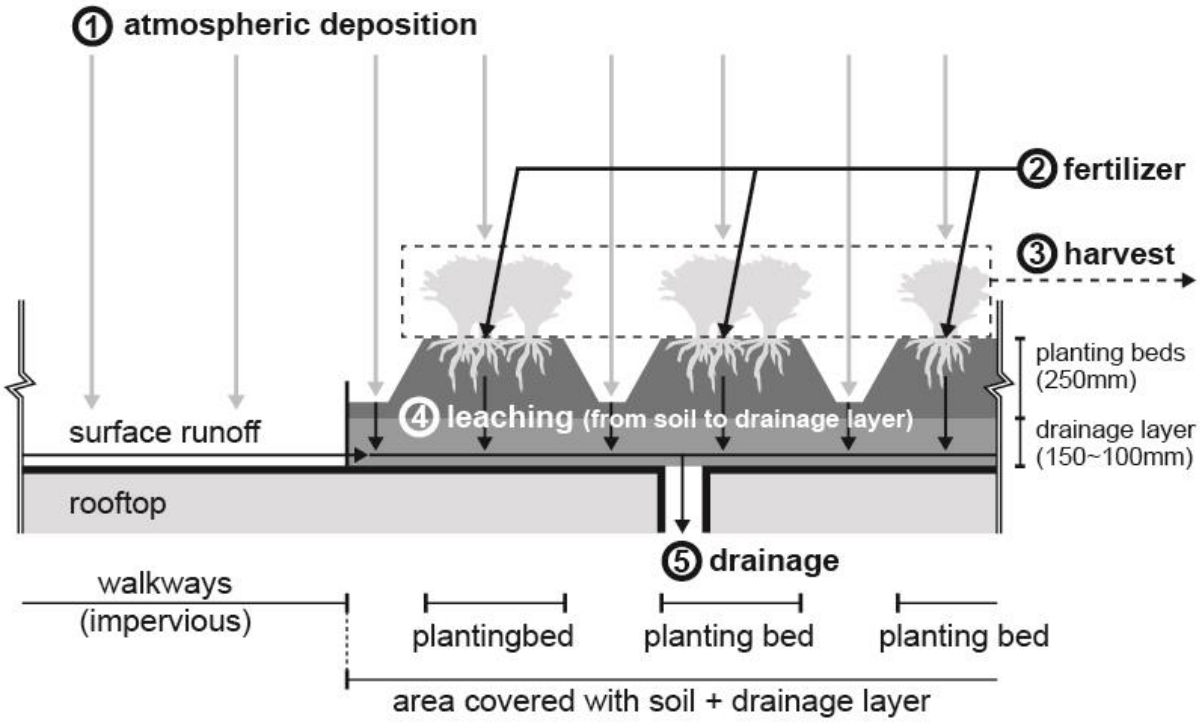
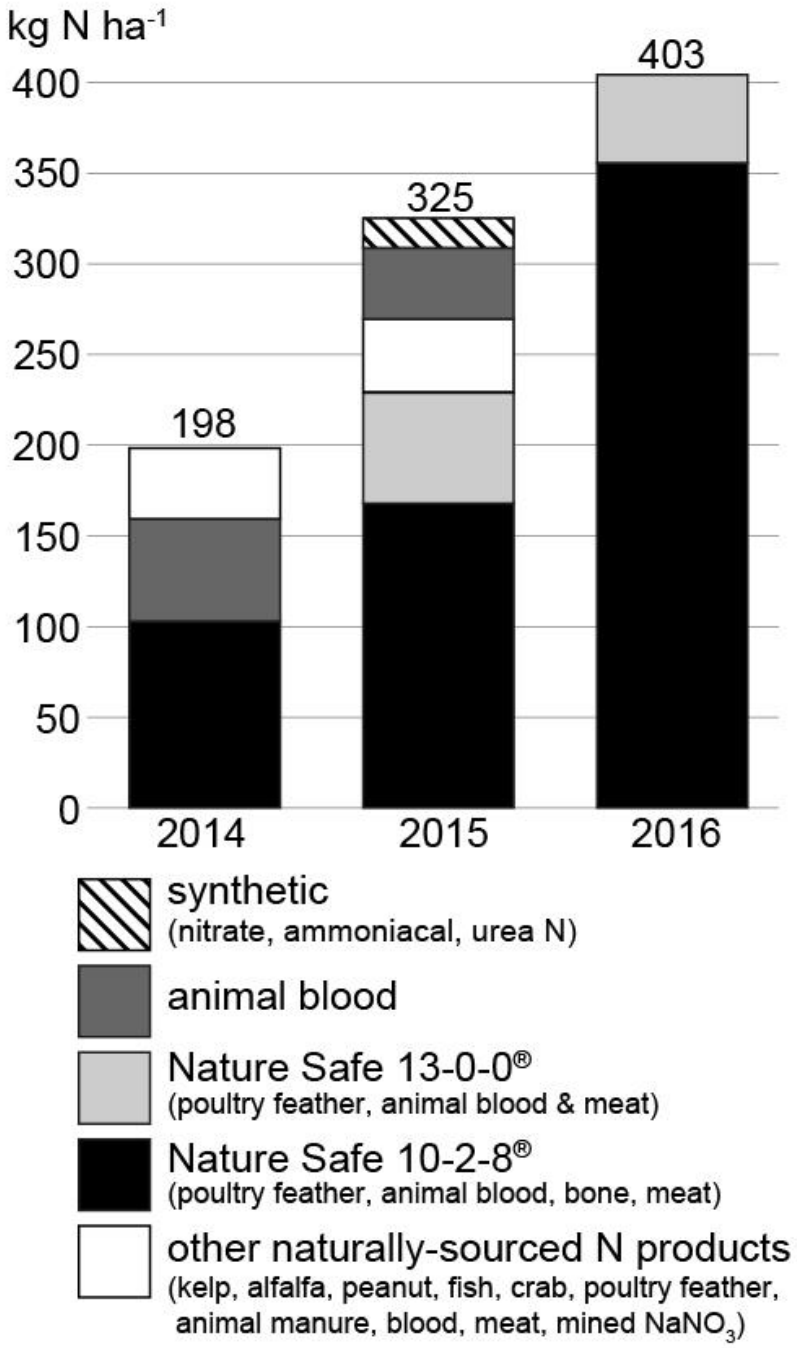


Figure 3-3. N input by fertilizer application and N sources at the Brooklyn Grange Navy Yard farm, NYC. All values are expressed for the planting bed area.



Soil is Rooflite® intensive ag, a commercial soil blend of expanded shale, animal manure, and re-used mushroom substrate (Kong et al. 2015; Skyland USA LLC 2015). The drainage layer consists of filter fabric covering gravel 100 – 150 mm in depth (Figure 3-2). Within each catchment, the lateral export of stormwater runoff from impervious surfaces and the leachate from soil enters the drainage layer, then flows laterally to the drain (Figure 3-2). The Grange grows over 60 crops, with leaf lettuce (*Lactuca sativa*), mustard greens (*Brassica juncea*), arugula (*Eruca sativa*), kale (*Brassica oleracea*), pepper (*Capsicum annuum*), and tomato (*Solanum lycopersicum*) accounting for over 60% of planting bed area and fresh weight.

Atmospheric N deposition. Like most rooftops, our site lacks trees so we only measured bulk deposition, which was compared with total N deposition measured by the National Atmospheric Deposition Program (NADP; nearest site 20 km away) and estimated for our site using the Community Multiscale Air Quality (CMAQ) model (NADP Total Deposition Science Committee 2017; Schwede and Lear 2014). Bulk deposition was measured at 11 locations on the roof between 8/30 2014 – 11/12 2016. Six collectors were 1.5 m above the soil surface and the remaining 5 collectors were placed 3-6 m above the roof plane on rooftop structures (Figure 3-1). Each bulk collector consisted of a 900 mm polypropylene Büchner funnel inserted into a disposable 20ml chromatography column (BioRad, Hercules, CA) filled with a mixed bed ion-exchange resin (Dowex Amberlite IRN150). These resin columns were replaced every 6 weeks and extracted and shaken with 150ml 2M KCl solution and filtered through one filter. Nitrate (NO_3^-) and ammonium (NH_4^+) concentrations were analyzed by microplate reader (VersaMax, Molecular Devices, Sunnyvale, CA) following the colorimetric method as described in Sims et al. (1995) and Doane and Horwáth (2003). Concentrations in sample solutions were converted to flux by taking into account the total volume of KCl used to extract samples (150 ml), the surface area of the top of the funnel (9 cm diameter), and the amount of time the columns incubated in the field.

N leaching from soil. N leaching from soil is the vertical export of N from soil to the drainage layer (Figure 3-2). Between 8/30 2014 – 11/12 2016, nylon mesh bags filled with 100 ml of ion-exchange resin (Dowex Amberlite IRN150) were used to monitor N leaching from soil. Three bags were placed in each of 6 planting beds (Figure 3-1). Bags were clamped in a circular 100 mm plastic frames to ensure a uniform area through which vertical flow could occur, and buried under the soil layer at a depth of 250 mm, atop the filter fabric covering the drainage layer. Every 6 weeks, ion exchange resin bags were sampled from soils upon which 20 ml of ion exchange resin was extracted with 150 ml 2M KCl and analyzed in the same fashion as bulk deposition samples described above. Concentrations in resin extracts were converted to flux by taking into account the total volume of KCl used to extract samples (150 ml), the cross-sectional area of the plastic frame (10 cm diameter), and the amount of time the resin bags incubated in the field.

N in vegetables. Over the 2014, 2015, and 2016 growing seasons, leaves of the greens mix (leaf lettuce and mustard greens) and arugula were collected from the same 6 planting beds where the resin bags were located (Figure 3-1), and total nitrogen content of 10 – 15 mg dried samples were analyzed by Vario El Cube CHNOS Elemental Analyzer at Cornell Nutrient Analysis Laboratory (CNAL). The ratio of dry to fresh weight was measured by drying these samples by oven at 60 °C for 48 hours. Mass N leaving the farm via harvest of these leafy vegetables were calculated as dry-weight N concentration multiplied by dry-to-fresh weight ratio, and the fresh weight from the harvest record (B. Flanner, personal communication). Mass N of all other crops were estimated as fresh weight from the same harvest record multiplied by the fresh weight N concentrations from National Nutrient Database for Standard Reference (USDA 2016). Vegetable sampling, the harvest record, and the information retrieved from USDA database focused on the edible portion of each crop, which leaves the Grange for sales, while all remaining parts of the crops are composted in the field, and used as amendments for planting beds.

N in soil. Soil samples were collected from the same 6 planting beds where the resin bags were located. Between 8/2014 – 11/2016, soil samples were collected to the total depth of soil layer (25 cm), before and after each growing season, with two additional samples collected in 8/2014 and 8/2015. Total nitrogen content was analyzed by the same method as the aforementioned vegetable sample. Soil N pool was calculated by Eq 1.

$$N_{soil} = N_{concentration} \times S_{density} \times S_{depth} \times 10^4 \qquad \text{(Eq 1)}$$

where N_{soil} = soil N (kg N ha^{-1})

$N_{concentration}$ = soil N concentration (kg N kg soil^{-1})

$S_{density}$ = average soil bulk density (624.57 kg m^{-3})

S_{depth} = planting bed depth (0.25m)

Potential N mineralization. Between 12/2014 – 11/2016, soil samples to the total depth of soil layer (25 cm) were collected every 6 weeks from each of the 6 beds where the resin bags were located, and each sample was sieved (2 mm mesh) and divided into 6 subsamples. Ten grams from three sub-samples each were extracted with 40 ml 2M KCl within 24 hours of collection and the remaining three soil samples were extracted after a 28 day aerobic incubation in the laboratory (Drinkwater et al. 1996; Robertson 1999). KCl extract solutions were filtered through 1 filter. Net N mineralization was calculated as the extracted TIN (NO_3-N + NH_4-N) on day 28 minus the extracted TIN on day 0, divided by the number of days soils incubated in the lab. TIN concentration of extracts were analyzed by the aforementioned colorimetric microplate. Concentrations were converted to mass of N mineralized per unit dry soil after determining gravimetric soil moisture content. Soil moisture was determined by drying samples by oven at 60 °C for 24 hours.

Drainage N output. Lateral flow through the drainage layer contains leachate from soil layer and surface runoff from impervious area, and is lost to drains (Figure 3-2). This drainage N output is the N load to municipal sewer system and surface water, including Long Island Sound. Drainage samples were collected from all 8 drains, at least once in each sampling period for atmospheric N deposition and N leaching from soil between 2/2015 – 11/2016. Unfiltered drainage samples were analyzed by the aforementioned colorimetric microplate method. Drainage N output for each sampling period was calculated as average concentration multiplied by cumulative drainage volume estimated from the direct measurement from catchment 8 with a V-notch weir (see Harada et al. 2018 for details). Average concentration for each sampling period was the daily average concentrations averaged across each sampling period.

To estimate the drainage N output during CSOs, the drainage N output was modeled as a monomial function of water input depth, calibrated by the results from the monitoring campaign in the 2016 growing season. This monitoring campaign continued for 14 hours following 5.4mm irrigation on 6/22, 20.5 mm rainfall on 7/4-5, and 36.1 mm rainfall on 7/29 respectively. Each of these water input depths and 14-hour loads were used as daily water input and drainage N output for calibration respectively. Drainage samples were taken hourly from 2 drains (catchment 2 and 8) on 6/22, and from all 8 drains on 7/4-5 and 29. Hourly drainage N output was estimated as hourly drainage N concentration multiplied by hourly drainage depth. Drainage sample analysis and drainage depth estimation were same as the aforementioned method for annual drainage N output.

N balance and efficiency of N management. Fertilizer N input, N output via vegetable harvest, and N leaching from soil are fluxes occurring in planting bed area only, whereas atmospheric N deposition and the drainage area for the roof applies to the entire roof, including impervious area. To allow for comparison between these N fluxes, each flux was expressed both for just the planting bed (N mass divided by time and the planting bed area), and for the entire site (N mass divided by time

and the entire site area) in the annual N balance. For the efficiency of N management, we compared the 2015 and 2016 growing seasons, and fluxes were expressed for the planting bed area with N leaching from soil treated as the outflow N loss from the production system.

3-3. Results

N balance. Drainage N output (104 kg N ha^{-1} y^{-1}) was 1100% of cumulative bulk deposition (9.1 kg N ha^{-1} y^{-1}) (Figure 3-4), and was 540% of the estimated total deposition (19.40 kg N ha^{-1} y^{-1}) (Table 3-1) (NADP Total Deposition Science Committee 2017; Schwede and Lear 2014), which makes the Grange a net N source in the urban environment. The major inputs were fertilizer (160 kg N ha^{-1} y^{-1}) and atmospheric deposition (9.1 kg N ha^{-1} y^{-1}), but these inputs were offset by N lost via harvest of vegetables (91 kg N ha^{-1} y^{-1}) and drainage output (104 kg N ha^{-1} y^{-1}), and the annual N budget of the ecosystem was -26 kg N ha^{-1} y^{-1} (Figure 3-4). The annual N budget of the soil layer (-76 kg N ha^{-1} y^{-1}) suggests that equivalent to 83% of N output via harvest was lost from the total N pool of the soil layer over time, including both growing and non-growing seasons. The annual N budget of the drainage layer (+50 kg N ha^{-1} y^{-1}) suggests that 32% of N leaching from soil (154 kg N ha^{-1} y^{-1}) to the drainage layer was retained within the drainage layer, instead of leaving the system as drainage N output. This net effect of the drainage layer combines denitrification, microbial immobilization, and the exchangeable nutrient pool of the drainage layer.

Figure 3-4. Annual N balance of the Brooklyn Grange Navy Yard farm, NYC, between 11/12/2015 – 11/12/2016. Average daily fluxes of each sampling period were aggregated annually. The bottom figure shows relative magnitude of fluxes expressed for the entire site area. * Bulk collector 1.5m above the soil surface.

		expressed for planting bed area (Kg N ha^{-1} y^{-1})	expressed for entire site area (Kg N ha^{-1} y^{-1})
input	atmospheric deposition*	9.1	9.1
	fertilizer	403	160
	total	412	169
sub-system flux	leaching (from soil to drainage layer)	389	154
output	harvest	229	91
	drainage	262	104
	total	491	195
N budget	**Ecosystem** (total input – total output)	-79	-26
	Drainage layer (leaching from soil – drainage)	+127	+50
	Soil layer (total input - harvest - leaching)	-206	-76

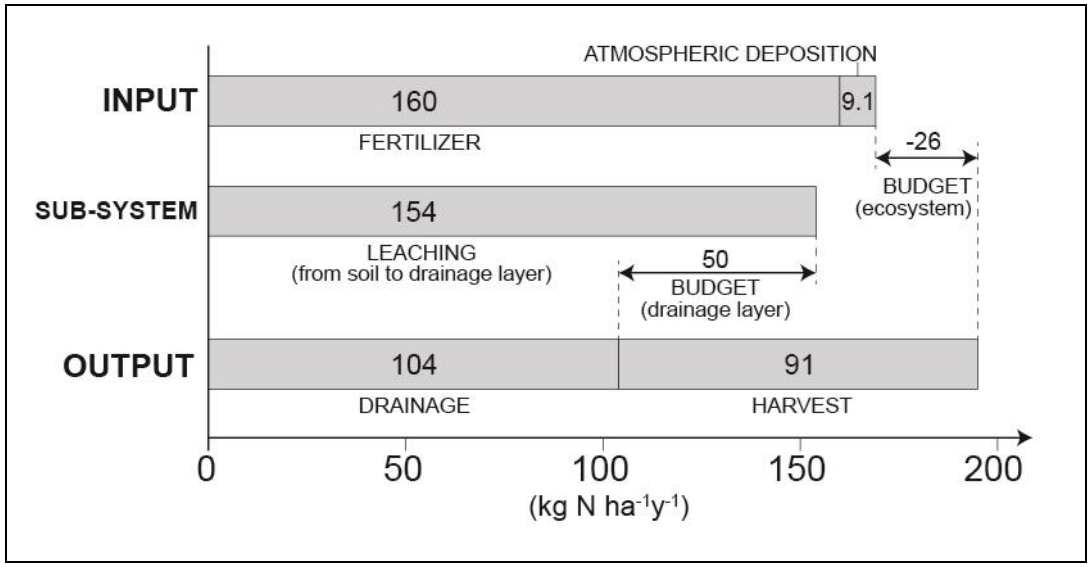


Table 3-1. Atmospheric N deposition at the Brooklyn Grange Navy Yard farm, NYC, between 11/12/2014 – 11/12/2015 and 11/12/2015 – 11/12/2016, compared with other studies.

atmospheric deposition by sample type	2014-2015 (Kg N ha^{-1} y^{-1})	2015-2016 (Kg N ha^{-1} y^{-1})
1.5m above the soil surface	8.8	9.1
3.0-6.0m above the soil surface	8.0	11.0

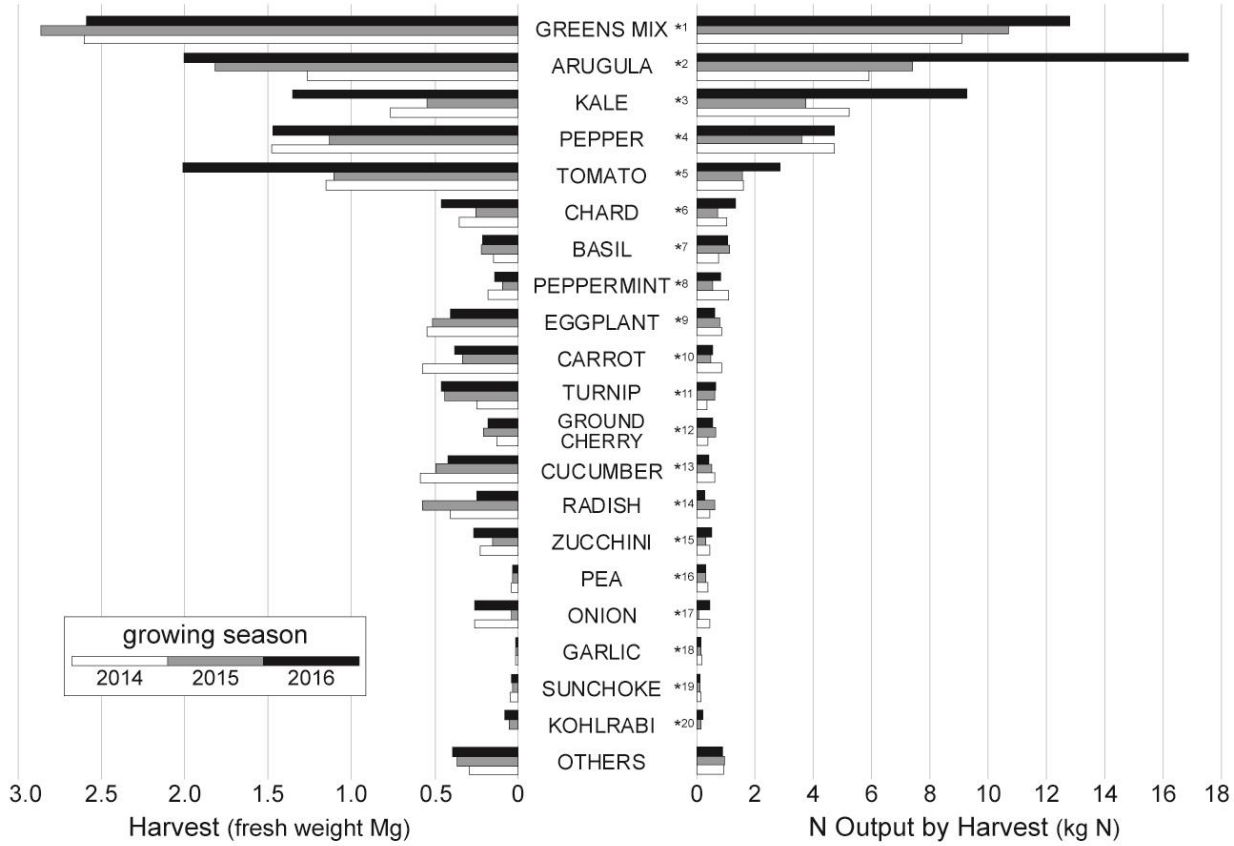
comparison of urban atmospheric N deposition			
location	sample type	deposition rate (Kg N ha^{-1} y^{-1})	author
Baltimore (2010 - 2011)	bulk (2010-2011)	6.3	Bettez et al. (2013)
	throughfall (2010-2011)	13.3	
The Brooklyn Grange	bulk (2014-2015)	8.8	This study
	bulk (2015, wet + dry particulate)	4.81	Total Deposition Map by NADP Total Deposition Science Committee (2017)
	total (2015)	19.40	

Soil N and N output via harvest. Over the 3-year study, average soil total N concentration was 7.9 g N kg soil^{-1}, which translates into the total farm N pool of 12,000 kg N ha^{-1} expressed for the planting bed area, or 4,900 kg N ha^{-1} expressed for the entire site area. On average, extractible TIN was 48 ± 47 mg N kg^{-1}, or only 0.6% of the total N pool, and PMN was 1.9 mg N kg^{-1}d^{-1}. Over the 3 growing seasons, a total of 126 kg N was exported from the Grange through the harvest of 36,200 kg vegetables in fresh weight (Figure 3-5). Annual harvest N output was 35 kg N in 2014 and 2015, and increased to 56 kg N in 2016 (Figure 3-5). Pea (*Pisum sativum*) and bush bean (*Phaseolus vulgaris*) were the only legume crops grown, which accounted for only 1% of total N output by harvest over the 3 growing seasons.

Efficiency of N management. For the production of leafy vegetables during growing seasons (3/31 – 11/12, 226 days), the major N supplies were fertilizer, atmospheric deposition, and the extractable soil N, but these N supplies were offset by the harvest and N leaching from soil. The total N balance ranged between + 1 kg N ha^{-1} and - 160 kg N ha^{-1} (Table 3-2). The total N supply over the 2016 growing season (510 kg N ha^{-1}) was 123% of the 2015 growing season (414 kg N ha^{-1}), whereas leaching from soil over the 2016 growing season (336 kg N ha^{-1}) was 112% of the 2015 growing season (300 kg N ha^{-1}). This means that although N leaching loss increased, N supply increased more than N leaching loss. As a result, N supply efficiency increased from 28% to 34%.

From the 2015 to 2016 growing seasons, N recovery rate of arugula increased from 47% to 66%, while N recovery rate of greens mix increased only from 32% to 34% respectively (Table 3-2). However, fresh weight yield for arugula over the 2016 growing season (40 Mg ha^{-1}) was only 83% of the 2015 growing season (48 Mg ha^{-1}), and fresh weight yield for greens mix over the 2016 growing season (35 Mg ha^{-1}) was only 97% of the 2015 growing season (36 Mg ha^{-1}) (Figure 3-5) . This was due to the dry-weight N concentrations, which increased from 4.5% to 6.4% for arugula, and from 3.5% to 6.2% for greens mix, from the 2015 to 2016 growing seasons respectively.

Figure 3-5. Harvest fresh mass and N mass output over the 2014, 2015, and 2016 growing seasons at the Brooklyn Grange Navy Yard farm, NYC. Crops are listed in order of the harvest N output over three growing seasons combined.



*1 *Lactuca sativa* & *Brassica juncea*
*2 *Eruca sativa*
*3 *Brassica oleracea* (mostly var. *laciniata* & var. *palmifolia*)
*4 *Capsicum annuum*
*5 *Solanum lycopersicum*
*6 *Beta vulgaris*
*7 *Ocimum basilicum*
*8 *Mentha × piperita*
*9 *Solanum melongena*
*10 *Daucus carota* subsp. *sativus*
*11 *Brassica rapa* var. *rapa*
*12 *Prunus fruticosa*
*13 *Cucumis sativus*
*14 *Raphanus raphanistrum* subsp. *sativus*
*15 *Cucurbita pepo*
*16 *Pisum sativum*
*17 *Allium cepa*
*18 *Allium sativum*
*19 *Helianthus tuberosus*
*20 *Brassica oleracea* var. *gongylodes*

Table 3-2. Efficiency of N management for leafy vegetable production at the Brooklyn Grange Navy Yard farm, NYC, in the 2015 and 2016 growing seasons (3/31-11/12, 226 days). All N fluxes are in kg N ha^{-1} 226 d^{-1} expressed for the planting bed area. * 2M-KCl extractable TIN at the beginning of the growing season. ** calculated as total N supply – N leaching from soil – N output by harvest.

		growing season 2015	growing season 2016
N supply	fertilizer N input	325	403
	initial extractable soil N *	83	100
	atmospheric N deposition	6.2	6.9
	total N supply	414	510
total N outflow output	N leaching from soil	300	336
	NSE (Eq 4)	28%	34%
arugula production	N output by harvest	196	334
	N budget**	-82	-160
	NRR (Eq 5)	47%	66%
greens mix production	N output by harvest	133	173
	N budget**	-19	+1
	NRR (Eq 2)	32%	34%

$$N\ supply\ efficiency\ (NSE) = \frac{total\ N\ supply - total\ N\ outflow\ loss}{total\ N\ supply} \quad \text{(Eq 2)}$$

$$N\ recovery\ rate\ (NRR) = \frac{Yield\ N}{total\ N\ supply} \quad \text{(Eq 3)}$$

Drainage N output during combined sewer overflows (CSOs). Daily drainage N output was expressed as a monomial function of daily water input, and calibrated (RMSE = 0.06) by the results from the hourly monitoring campaign in the 2016 growing season, where 5.4mm irrigation, 20.5mm rainfall, and 36.1mm rainfall resulted in 0.3 kg N ha^{-1}, 2.1 kg N ha^{-1}, and 4.9 kg N ha^{-1} of drainage N outputs respectively (Figure 3-6 g, h, i). The model is given by Eq 4.

$$N_{load} = 0.0304 \times W^{1.4135} \qquad \text{(Eq 4)}$$

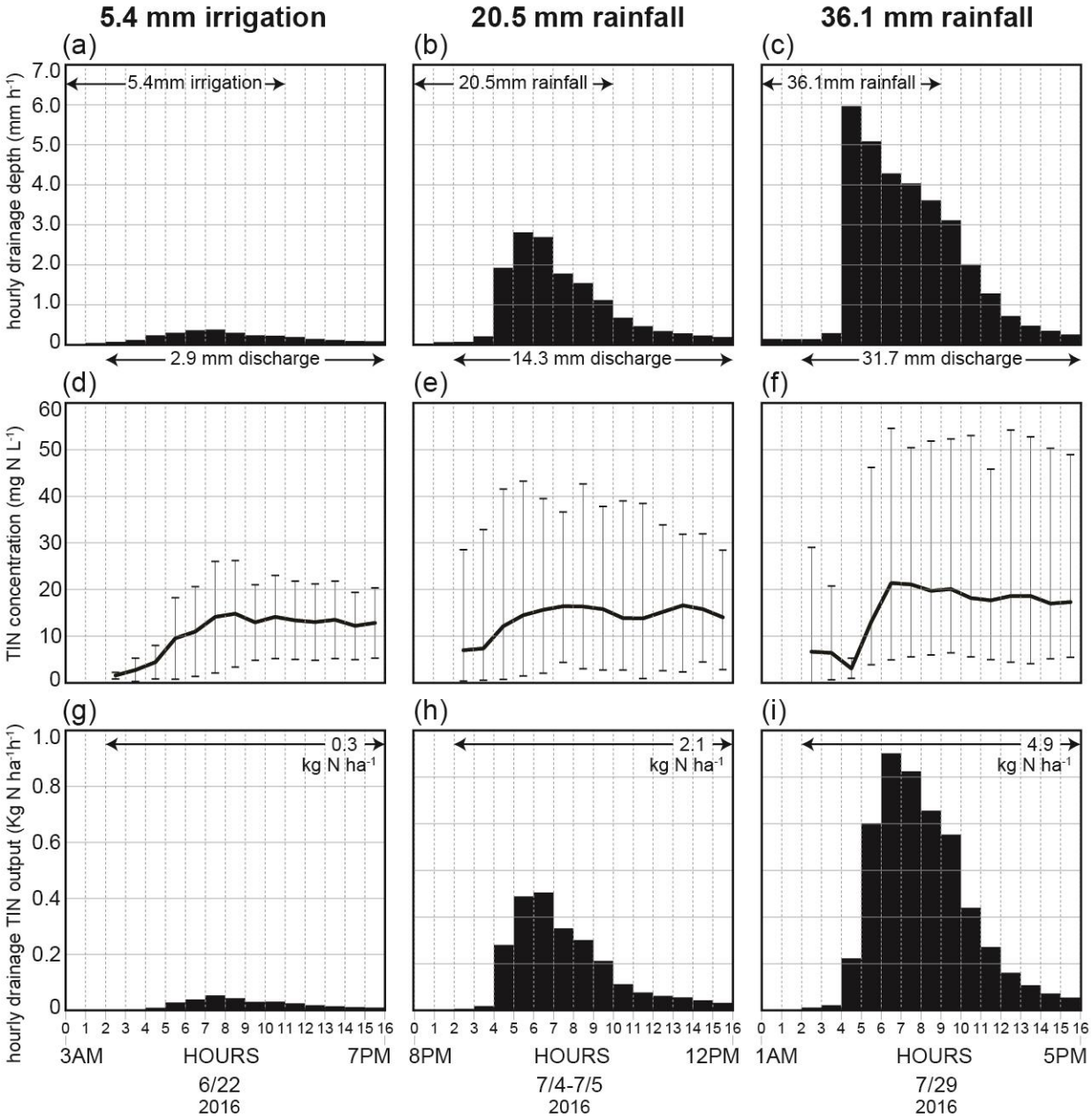
where N_{load} = daily drainage N output (kg N ha^{-1} d^{-1}) expressed for the entire site area; and

W = daily total water input (rainfall + irrigation) (mm d^{-1}) expressed for the entire site area.

During the 2016 growing season (3/31 – 11/12, 226 days), the cumulative drainage N output estimated by the model (79 kg N ha^{-1}) exceeded the measured N load (77 kg N ha^{-1}) by 3%, and RMSE between the measured N load and model estimate in each sampling period was 9.6. This result suggests that the model could express the general trend of the Grange's drainage N output over the 2016 growing season.

A minimum of 0.4 inch (10 mm) daily rainfall can trigger combined sewer overflows (CSOs) from the watershed of Red Hook WWTP, which receives N load from the Grange while discharging untreated sewage to the East River and Long Island Sound during CSOs (NYC DEP 2015; NYC DEP 2016; ODMHED NYC 2011). Daily rainfall depth was 10mm or deeper for 9% (20 days) of the 2016 growing season. For these 20 days, the cumulative daily N load estimated by the model was 67% (51 kg N ha^{-1}) of the cumulative drainage N output over the 2016 growing season, which was likely to be discharged from the Grange to the East River and Long Island Sound during CSOs.

Figure 3-6. Hourly drainage depth, TIN concentration, TIN output during the monitoring campaign on 6/22, 7/4-5, 7/29 in 2016 at the Brooklyn Grange Navy Yard farm, NYC.



3-4. Discussion

Drainage N output. The annual drainage N output (104 kg N ha^{-1} y^{-1}) was 310% of another study (33.60 kg N ha^{-1} y^{-1}) of operational full-scale rooftop farming by Whittinghill et al. (2016), perhaps because fertilizer N input in this study was 730% of Whittinghill et al. (2016) (22 kg N ha^{-1} y^{-1}, calculated by using the reported values). For the direct comparison with other agricultural systems, drainage N output must be expressed for just the planting bed area (262 kg N $ha^{-1}y^{-1}$), which falls within the range of in-ground intensive vegetable production (Min et al. 2012; Zhang et al. 2017; Zhao et al. 2010).

Drainage N output from the Grange is collected by Red Hook Wastewater Treatment Plant (WWTP), which discharges effluent to the East River and the Long Island Sound (NYC DEP 2007; ODMHED NYC 2011). In 2016, the Grange's mass N load (0.064 Mg N y^{-1}) to sewer was only 0.01% of effluent N load (634 Mg N y^{-1}) from the Red Hook WWTP (US EPA 2017a). Also, New York City alone has 8599 ha of flat rooftop surface, of which 14% (1246 ha) is reported as suitable for commercial rooftop farming larger than 0.09 ha (10,000 ft^2) (Ackerman et al. 2013; Acks 2006). If all of these suitable sites are converted to rooftop farms like the Grange (104 kg N $ha^{-1}y^{-1}$), the citywide N load from these rooftop farms would be 129 Mg N y^{-1}, which is only 0.6% of the citywide effluent N load (23049 Mg N y^{-1}) from all 14 N-point-source WWTPs combined (NYC DEP 2007; US EPA 2017a).

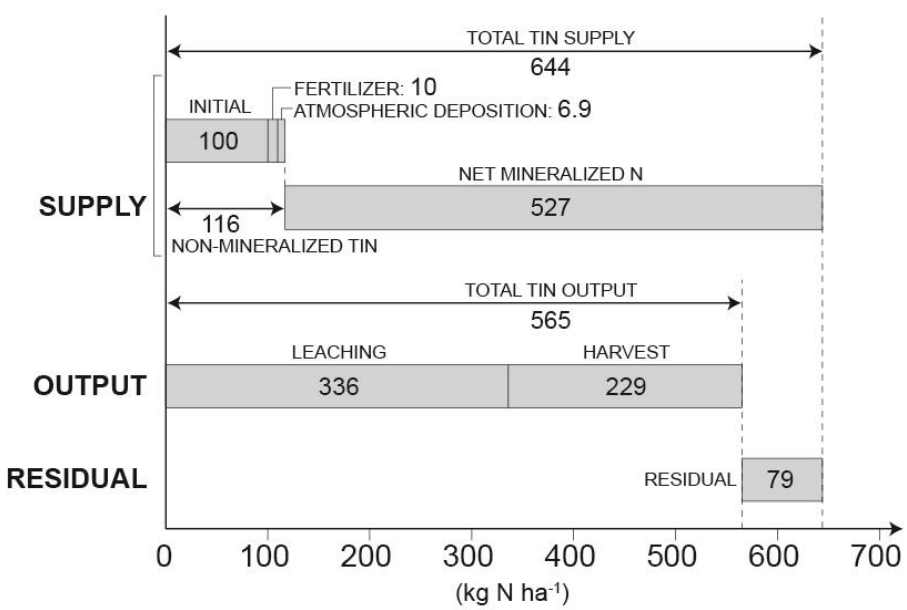
Urban atmospheric N deposition. Between 11/2014 – 11/2015, bulk N deposition was only 45% of the estimated total N deposition from the Total Deposition Map (NADP Total Deposition Science Committee 2017; Schwede and Lear 2014), perhaps because bulk collectors undersampled cloud and gaseous N deposition. In Baltimore, Bettez and Groffman (2013) report bulk N deposition was only 47% of throughfall N deposition, which is used as a surrogate for total N deposition in many studies. These findings suggest that cloud and gaseous N deposition could account for over the half of total N

deposition in urban environments. In our study, possible N gas emission sources include the Brooklyn Navy Yard Cogeneration Facility, which is a 286-megawatt gas-fired power plant 300m west from the site, annually releasing over 32 Mg NH_3-N gas between 2014 – 2016 (EPA 2017). Also, Brooklyn / Queens Expressway is 250m south from the site, and had annual average daily traffic (AADT) of 123,631 cars day^{-1} in 2015 (NYS DOT 2015). Atmospheric deposition monitoring networks in the United States are generally located away from the urban and industrial emission sources, yet there is a growing body of work in urban areas (Decina et al. 2017; Holland et al. 2005; Howarth 2008a; Rao et al. 2014).

N mineralization. Over the 2016 growing season, TIN represented only 2% of fertilizer N inputs, which suggests mineralization of organic N contributed much of TIN leaching loss from soil. In the mass balance over the 2016 growing season (Figure 3-7), the estimated rates of net mineralization were 527 kg N ha^{-1} 226 d^{-1}. Total non-mineralized TIN supply was 116 kg N ha^{-1}, which is equivalent to only 35% of the 336 kg ha^{-1} leached from soil. This would indicate that at least 65% of TIN leached from soil came from mineralization during the 2016 growing season. However, the mass balance also shows that over the half of the total TIN supply (non-mineralized TIN + net mineralized N = 644 kg N ha^{-1}) was lost by leaching from soil. This result suggests both the drainage volume and mineralization were important drivers increasing TIN leaching loss.

Figure 3-7. Estimation of N mineralization over the 2016 growing season (226 days between 3/31 - 11/12). All values are expressed for the planting bed area. All fluxes are in kg N ha^{-1} 226 d^{-1}, and extractable soil TIN values are in kg N ha^{-1}. The bottom figure shows relative magnitude of N values used for the estimation.

TIN supply	non-mineralized TIN supply	initial extractable soil TIN at the beginning of the growing season	100
		cumulative atmospheric deposition	6.9
		fertilizer TIN input	10
		total non-mineralized TIN	116
	net mineralized N	total TIN output + residual TIN - total non-mineralized TIN	527
	total TIN supply	net mineralized N + non-mineralized TIN supply	644
TIN output		N output by harvest	229
		cumulative TIN leaching from soil	336
		total TIN output	565
residual TIN		extractable soil TIN at the end of the growing season	79



Implications of drainage N concentrations for environmental planning. For the Eastern coastal plain including New York City, US EPA has Ambient Water Quality Criteria (AWQC) total nitrogen concentrations for rivers and streams (0.71 mg N L^{-1}) and lakes and reservoirs (0.32 mg N L^{-1}), but not for salt water (US EPA 2000; US EPA 2001). Because salt water ecosystems can be more N limited than fresh water ecosystems (see Howarth 2008b), these AWQC could be too permissive for salt water systems like Long Island Sound, yet are exceeded by the annual average (12.0 ± 6.5 mg NO_3-N L^{-1}) and annually flow-weighted average (10.7 mg NO_3-N L^{-1}) drainage concentration of the Grange by an order of magnitude. Among N drainage concentrations from various land uses, rainwater, CSOs, surface water, and surface runoff in urban environments were around or below the AWQC, whereas effluent from WWTPs and agriculture, including rooftop farming, can exceed these AWQC by an order of magnitude or more (Bakhsh et al. 2010; Benotti et al. 2007; Gold et al. 1990; Goulding et al. 2000; Gundersen et al. 2006; Likens 2013; NADP 2017; Pimentel et al. 2005; Randall et al. 1997; Steuer et al. 1997; US EPA 2017a; US NPS 2017; Whittinghill et al. 2016; Zhao et al. 2010).

WHO (2004), US EPA (2017b), and NYS DEC (2017) has guideline levels for drinking water at 10 mg NO_3-N L^{-1}. Because drinking water below this guideline level can still pose human health risks (see Townsend et al. 2003), this guideline level could be too permissive, yet is exceeded by in-ground agriculture and this study. In short, rooftop farming could be similar to in-ground agriculture and wastewater to the perspective of environmental planning based on drainage N concentrations.

Efficiency of N management. Since the N concentrations of leafy vegetables were around or above the reference range for vegetable production (arugula: 2.86% – 3.97%, mustard greens: 2.97%-3.85%) reported by Mills and Jones (1996), the production of these leafy vegetables was unlikely to be limited by N. This suggests the further increase in fertilizer N input is unlikely to increase fresh weight yield effectively, which is important to the economic viability of fresh vegetable production.

Between 11/2015 – 11/2016, the annual N leaching from the planting beds (389 kg N $ha^{-1}y^{-1}$) was 97% of the N input by fertilizer application (403 kg N $ha^{-1}y^{-1}$), which suggests a great opportunity for improving the efficiency of N management. Among the studies of in-ground intensive vegetable production, Min et al. (2012) reports that the fertilizer N input (1104 kg N $ha^{-1}y^{-1}$) was 270% of the Grange, yet only 25% of this fertilizer N was lost by leaching. This is perhaps because annual drainage depth (234.7 mm y^{-1}) was only 25% of the Grange (954.0 mm y^{-1}). Also, Zhao et al. (2010) reports that the fertilizer N input (1480 kg N $ha^{-1}y^{-1}$) was 370% of the Grange, yet only 24% of this fertilizer N was lost by leaching. Zhao et al. (2010) reports annual drainage depth (786 mm y^{-1}) was only 39% of the annual total water input (irrigation + rainfall, 2036mm y^{-1}), whereas, annual drainage depth (954.03 mm y^{-1}) was 62% of the annual total water input at the Grange (1543.6mm y^{-1}). These findings suggest that the suboptimal management of water at the Grange increased the drainage volume and N leaching from soil.

The efficiency of N management could be enhanced by using soil with larger storage capacity within the range of readily available water (10-100 kPa) (Harada et al. 2018). For example, shallow (< 150 mm) potting mixes using coconut coir or other organic materials as base materials, could increase water in the effective root zone, while reducing the drainage by the gravitational force (Harada et al. 2017; Harada et al. 2018). Such soil-based approach could be combined with drainage recycle system, while issues of salt accumulation, plant pathogen, and the need of frequent organic amendments must be addressed. Further empirical studies are needed for the cost-benefit relationship between the enhanced N use efficiency and the system construction and maintenance, while comparing soil-based vs. hydroponic systems, and outdoor vs. controlled-environment systems.

3-5. Conclusions

This study reports the N balance and efficiency of N management of the Brooklyn Grange, a full-scale operational rooftop farm in NYC, USA. We found that the drainage N output to sewer was 1100% of the atmospheric bulk deposition, and was 540% of the estimated total deposition, which makes the Grange a net N source in the urban environment. Both the drainage volume and soil N mineralization were the important factors increasing N discharge, because fertilizer N input was dominantly organic N, and much of the mineralized N was lost to drains. In particular, suboptimal management of water increased the overall drainage N output, and drainage N output during combined sewer overflows, while decreasing the efficiency of N management to be lower than of in-ground intensive vegetable production. Recycling drainage, and using the soil with enhanced water holding capacity in the range of readily available water, could reduce N load to sewers, while maintaining satisfactory yield and quality. Rooftop farming is in its infancy, and there are unprecedented opportunities to develop best management practices in advance to the large-scale implementation. Further empirical studies are needed for N use efficiency and cost benefit relationship across different designs of soil, irrigation methods, types and rates of N applied, and climate zones while comparing soil-based vs. hydroponic systems, and outdoor vs. controlled-environment systems.

REFERENCE

Ackerman K, Dahlgren E, Xu X (2013) Sustainable Urban Agriculture: Confirming Viable Scenarios for Production, Final Report No.13-07 vol NYSERDA No. 13-07 [Online] <http://urbandesignlab.columbia.edu/files/2015/04/2_Sustainable-Urban-Agriculture_NYSERDA.pdf> (Accessed on April 01, 2016).

Acks K (2006) A framework for cost-benefit analysis of green roofs: initial estimates Green Roofs in the Metropolitan Region: Research Report Columbia University Center for Climate Systems Research

Ampim PA, Sloan JJ, Cabrera RI, Harp DA, Jaber FH (2010) Green roof growing substrates: types, ingredients, composition and properties Journal of Environmental Horticulture 28:244

Bakhsh A, Kanwar RS, Baker J (2010) N-application methods and precipitation pattern effects on subsurface drainage nitrate losses and crop yields Water, Air, & Soil Pollution 212:65-76

Benedetti A, Sebastiani G (1996) Determination of potentially mineralizable nitrogen in agricultural soil Biology and fertility of soils 21:114-120

Benotti MJ, Abbene M, Terracciano S (2007) Nitrogen Loading in Jamaica Bay, Long Island, New York: Predevelopment to 2005 Scientific Investigations Report 2007–5051 U.S. Department of the Interior and U.S. Geological Survey

Berndtsson JC (2010) Green roof performance towards management of runoff water quantity and quality: A review Ecological Engineering 36:351-360

Berndtsson JC, Emilsson T, Bengtsson L (2006) The influence of extensive vegetated roofs on runoff water quality Science of the Total Environment 355:48-63

Best BB, Swadek KR, Burgess LT (2015) Soil-Based Green Roofs. In: Sutton KR (ed) Green Roof Ecosystems. Springer International Publishing, Cham, pp 139-174. doi:10.1007/978-3-319-14983-7_6

Bettez ND, Groffman PM (2013) Nitrogen deposition in and near an urban ecosystem Environmental science & technology 47:6047-6051

Buffam I, Mitchell ME, Durtsche RD (2016) Environmental drivers of seasonal variation in green roof runoff water quality Ecological Engineering 91:506-514 doi:http://dx.doi.org/10.1016/j.ecoleng.2016.02.044

Cameron K, Di H, Moir J (2013) Nitrogen losses from the soil/plant system: a review Annals of Applied Biology 162:145-173

Campbell JL et al. (2004) Input-output budgets of inorganic nitrogen for 24 forest watersheds in the northeastern United States: A review Water Air Soil Pollut 151:373-396 doi:10.1023/b:wate.0000009908.94219.04

Congreves K, Van Eerd L (2015) Nitrogen cycling and management in intensive horticultural systems Nutrient cycling in agroecosystems 102:299-318

Decina SM, Templer PH, Hutyra LR, Gately CK, Rao P (2017) Variability, drivers, and effects of atmospheric nitrogen inputs across an urban area: Emerging patterns among human activities, the atmosphere, and soils Science of The Total Environment 609:1524-1534

Di H, Cameron K (2002) Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies Nutrient cycling in agroecosystems 64:237-256

Doane TA, Horwáth WR (2003) Spectrophotometric determination of nitrate with a single reagent Analytical Letters 36:2713-2722

Drinkwater L, Letourneau D, Workneh F, Van Bruggen A, Shennan C (1995) Fundamental differences between conventional and organic tomato agroecosystems in California Ecological Applications 5:1098-1112

Drinkwater LE, Cambardella CA, Reeder JD, Rice CW (1996) Potentially mineralizable nitrogen as an indicator of biologically active soil nitrogen Methods for assessing soil quality:217-229

Driscoll CT et al. (2015) Green Infrastructure: Lessons from Science and Practice. A publication of the Science Policy Exchange:32

Eksi M, Rowe DB, Fernández-Cañero R, Cregg BM (2015) Effect of substrate compost percentage on green roof vegetable production Urban Forestry & Urban Greening 14:315-322

Emilsson T, Berndtsson JC, Mattsson JE, Rolf K (2007) Effect of using conventional and controlled release fertiliser on nutrient runoff from various vegetated roof systems Ecological engineering 29:260-271

EPA U (2017) The Toxics Release Inventory (TRI) Search System [Online] <https://www.epa.gov/enviro/tri-search> (Accessed on December 03, 2017)

Eriksen J, Askegaard M, Kristensen K (2004) Nitrate leaching from an organic dairy crop rotation: the effects of manure type, nitrogen input and improved crop rotation Soil Use Manage 20:48-54

Fenn ME et al. (1998) Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies Ecological Applications 8:706-733

Gold AJ, DeRagon WR, Sullivan WM, Lemunyon JL (1990) Nitrate-nitrogen losses to groundwater from rural and suburban land uses Journal of soil and water conservation 45:305-310

Goulding KWT, Poulton PR, Webster CP, Howe MT (2000) Nitrate leaching from the Broadbalk Wheat Experiment, Rothamsted, UK, as influenced by fertilizer and manure inputs and the weather Soil Use Manage 16:244-250

Gregoire BG, Clausen JC (2011) Effect of a modular extensive green roof on stormwater runoff and water quality Ecological Engineering 37:963-969
Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu J, Bai X, Briggs JM (2008) Global change and the ecology of cities science 319:756-760
Groffman PM, Law NL, Belt KT, Band LE, Fisher GT (2004) Nitrogen fluxes and retention in urban watershed ecosystems Ecosystems 7:393-403
Gundersen P, Schmidt IK, Raulund-Rasmussen K (2006) Leaching of nitrate from temperate forests effects of air pollution and forest management Environmental reviews 14:1-57
Harada Y, Whitlow TH, Bassuk NL, Russell-Anelli J (2017) Biogeochemistry of Rooftop Farm Soils. In: Lal R, Stewart BA (eds) Urban Soils. Advances in Soil Science Series. Taylor & Francis Group, Portland, United States,
Harada Y, Whitlow TH, Todd Walter M, Bassuk NL, Russell-Anelli J, Schindelbeck RR (2018) Hydrology of the Brooklyn Grange, an urban rooftop farm Urban Ecosystems doi:10.1007/s11252-018-0749-7
Hartz T (2006) Vegetable production best management practices to minimize nutrient loss HortTechnology 16:398-403
Holland EA, Braswell BH, Sulzman J, Lamarque J-F (2005) Nitrogen deposition onto the United States and Western Europe: synthesis of observations and models Ecological applications 15:38-57
Howarth R (2008a) Estimating Atmospheric Nitrogen Deposition in the Northeastern United States: Relevance to Narragansett Bay. doi:10.1007/978-0-387-35299-2_3
Howarth RW (2008b) Coastal nitrogen pollution: a review of sources and trends globally and regionally Harmful Algae 8:14-20
Howarth RW et al. (2000) Nutrient pollution of coastal rivers, bays and seas vol 7.
Howarth RW, Marino R, Swaney DP, Boyer EW (2006) Wastewater and watershed influences on primary productivity and oxygen dynamics in the lower Hudson River estuary The Hudson River Estuary 136
Howarth RW, Sharpley A, Walker D (2002) Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals Estuaries 25:656-676
Jayasundara S, Wagner-Riddle C, Parkin G, von Bertoldi P, Warland J, Kay B, Voroney P (2007) Minimizing nitrogen losses from a corn–soybean–winter wheat rotation with best management practices Nutrient Cycling in Agroecosystems 79:141-159 doi:10.1007/s10705-007-9103-9
Kaye JP, Groffman PM, Grimm NB, Baker LA, Pouyat RV (2006) A distinct urban biogeochemistry? Trends in Ecology & Evolution 21:192-199
Kemp WM et al. (2005) Eutrophication of Chesapeake Bay: historical trends and ecological interactions Mar Ecol-Prog Ser 303:1-29
Kong AY, Rosenzweig C, Arky J (2015) Nitrogen Dynamics Associated with Organic and Inorganic Inputs to Substrate Commonly Used on Rooftop Farms HortScience 50:806-813
Likens GE (2013) Biogeochemistry of a forested ecosystem. Springer Science & Business Media,
Lovell ST (2010) Multifunctional urban agriculture for sustainable land use planning in the United States Sustainability 2:2499-2522
Lovell ST, Taylor JR (2013) Supplying urban ecosystem services through multifunctional green infrastructure in the United States Landscape Ecology 28:1447-1463 doi:10.1007/s10980-013-9912-y
Lundholm JT (2015) The ecology and evolution of constructed ecosystems as green infrastructure Frontiers in Ecology and Evolution 3:106
McPhearson T, Kremer P, Hamstead ZA (2013) Mapping ecosystem services in New York City: Applying a social–ecological approach in urban vacant land Ecosystem Services 5:11-26 doi:http://dx.doi.org/10.1016/j.ecoser.2013.06.005
Mentens J, Raes D, Hermy M (2006) Green roofs as a tool for solving the rainwater runoff problem in the urbanized 21st century? Landscape and urban planning 77:217-226
Mills HA, Jones JB (1996) Plant analysis handbook II: A practical sampling, preparation, analysis, and interpretation guide. vol 581.13 M657.
Min J, Zhang H, Shi W (2012) Optimizing nitrogen input to reduce nitrate leaching loss in greenhouse vegetable production Agricultural water management 111:53-59
NADP (2017) National Trend Network Online Database [Online] <http://nadp.sws.uiuc.edu/data/NTN/> (Accessed on August 21, 2017)
NADP Total Deposition Science Committee (2017) Total Deposition Maps [Online] <http://nadp.sws.uiuc.edu/committees/tdep/tdepmaps/> (Accessed on August 21, 2017)
Nixon S et al. (1996) The fate of nitrogen and phosphorus at the land-sea margin of the North Atlantic Ocean Biogeochemistry 35:141-180
NYC DEP (2007) New York City's Wastewater Treatment System [Online] <http://www.nyc.gov/html/dep/pdf/wwsystem.pdf> (Accessed on April 01, 2016)
NYC DEP (2015) Combined Sewer Overflow Long Term Control Plan for Gowanus Canal Capital Project No. WP-169 Long Term Control Plan 2 [Online] <http://www.nyc.gov/html/dep/pdf/cso_long_term_control_plan/gowanus-canal-ltcp-201506.pdf> (Accessed on August 21, 2016).
NYC DEP (2016) New York City CSO Advisory Web Page

[Online] <http://www.nyc.gov/html/dep/pdf/harbor/cso_waterbody_advisories_about.pdf> (Accessed on August 21, 2016).

NYS DEC (2017) 6 CRR-NY 703.5 TITLE 6. CHAPTER X SUBCHAPTER A. GENERAL ARTICLE 2 PART 703. SURFACE WATER AND GROUNDWATER QUALITY STANDARDS AND GROUNDWATER EFFLUENT LIMITATIONS vol [Online] <http://www.dec.ny.gov/chemical/23853.html> (Accessed on August 01, 2016).
New York State Department of Transportation Traffic Data Viewer (2015) https://www.dot.ny.gov/tdv. Accessed August 30, 2017
Oberndorfer E et al. (2007) Green roofs as urban ecosystems: ecological structures, functions, and services BioScience 57:823-833
ODMHED NYC (2011) Water and Sewer Infrastructure Admirals Row Plaza Environmental Impact Statement CEQR No. 11DME001K [Online] <https://a002-ceqraccess.nyc.gov/ceqr/> (Accessed on August 21, 2016).
Palmer M et al. (2004) Ecology for a Crowded Planet Science 304:1251-1252 doi:10.1126/science.1095780
Pärn J, Pinay G, Mander Ü (2012) Indicators of nutrients transport from agricultural catchments under temperate climate: A review Ecological indicators 22:4-15
Pataki DE et al. (2011) Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions Front Ecol Environ 9:27-36
Pimentel D, Hepperly P, Hanson J, Douds D, Seidel R (2005) Environmental, energetic, and economic comparisons of organic and conventional farming systems BioScience 55:573-582
Poudel D, Horwath W, Lanini W, Temple S, Van Bruggen A (2002) Comparison of soil N availability and leaching potential, crop yields and weeds in organic, low-input and conventional farming systems in northern California Agriculture, ecosystems & environment 90:125-137
Rabalais NN, Turner RE, Wiseman Jr WJ (2002) Gulf of Mexico hypoxia, AKA" The dead zone" Annual Review of ecology and Systematics:235-263
Randall G, Huggins D, Russelle M, Fuchs D, Nelson W, Anderson J (1997) Nitrate losses through subsurface tile drainage in conservation reserve program, alfalfa, and row crop systems Journal of Environmental Quality 26:1240-1247
Rao P, Hutyra LR, Raciti SM, Templer PH (2014) Atmospheric nitrogen inputs and losses along an urbanization gradient from Boston to Harvard Forest, MA Biogeochemistry 121:229-245
Robertson GP (1999) Standard soil methods for long-term ecological research vol 2. Oxford University Press on Demand,
Rowe DB (2011) Green roofs as a means of pollution abatement Environmental Pollution 159:2100-2110
Scharenbroch BC, Lloyd JE, Johnson-Maynard JL (2005) Distinguishing urban soils with physical, chemical, and biological properties Pedobiologia 49:283-296 doi:http://dx.doi.org/10.1016/j.pedobi.2004.12.002
Schwede DB, Lear GG (2014) A novel hybrid approach for estimating total deposition in the United States Atmospheric Environment 92:207-220
Shock CC, Wang F-X (2011) Soil water tension, a powerful measurement for productivity and stewardship HortScience 46:178-185
Sims G, Ellsworth T, Mulvaney R (1995) Microscale determination of inorganic nitrogen in water and soil extracts Communications in Soil Science & Plant Analysis 26:303-316
Skyland USA LLC (2015) "Rooflite Intensive Ag Product Specification" <http://www.rooflitesoil.com/products/intensive-ag>accessed June 01, 2016
Specht K et al. (2014) Urban agriculture of the future: an overview of sustainability aspects of food production in and on buildings Agriculture and Human Values 31:33-51 doi:10.1007/s10460-013-9448-4
Steuer J, Selbig W, Hornewer N, Prey J (1997) Sources of contamination in an urban basin in Marquette, Michigan and an analysis of concentrations, loads, and data quality. US Geological Survey,
Syswerda S, Basso B, Hamilton S, Tausig J, Robertson G (2012) Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA Agriculture, Ecosystems & Environment 149:10-19
Szlavecz K, Placella SA, Pouyat RV, Groffman PM, Csuzdi C, Yesilonis I (2006) Invasive earthworm species and nitrogen cycling in remnant forest patches Applied Soil Ecology 32:54-62 doi:http://dx.doi.org/10.1016/j.apsoil.2005.01.006
Thomaier S, Specht K, Henckel D, Dierich A, Siebert R, Freisinger UB, Sawicka M (2015) Farming in and on urban buildings: Present practice and specific novelties of Zero-Acreage Farming (ZFarming) Renewable Agriculture and Food Systems 30:43-54 doi:10.1017/S1742170514000143
Townsend AR et al. (2003) Human health effects of a changing global nitrogen cycle Front Ecol Environ 1:240-246
US EPA (2000) Ambient Water Quality Criteria Recommendations for Rivers and Streams in Nutrient Ecoregion XIV, Office of Water 4304, EPA 822-B-00-022 [Online] <https://www.epa.gov/sites/production/files/documents/rivers14.pdf> (Accessed on March 20, 2018).
US EPA (2001) Ambient Water Quality Criteria Recommendations for Lakes and Reservoirs in Nutrient Ecoregion XIV, Office of Water 4304, EPA 822-B-01-011 [Online] <https://www.epa.gov/sites/production/files/documents/lakes14.pdf> (Accessed on March 20, 2018).
US EPA (2017a) Enforcement and Compliance History Online website [Online] <https://echo.epa.gov/> (Accessed on December 6, 2017).
US EPA (2017b) National Primary Drinking Water Regulations vol [Online] <https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations> (Accessed on August 01, 2016).
US NPS (2017) Jamaica Bay Water Quality Data from The NPS-USGS Water Quality Partnership program [Online] <https://www.epa.gov/waterdata/water-quality-data-wqx> (Water Quality Data Warehouse, Accessed on December 6, 2017).

USDA (2016) National Nutrient Database for Standard Reference [Online] <https://ndb.nal.usda.gov/ndb/> (Accessed on August 01, 2016)
Van Mechelen C, Dutoit T, Hermy M (2015) Adapting green roof irrigation practices for a sustainable future: A review Sustainable Cities and Society 19:74-90
Wainger LA (2012) Opportunities for reducing total maximum daily load (TMDL) compliance costs: Lessons from the Chesapeake Bay Environmental science & technology 46:9256-9265
Whittinghill L, Starry O (2016) Up on the Roof: Considerations for Food Production on Rooftops. In: Brown S, McIvor K, Hodges Snyder E (eds) Sowing Seeds in the City: Ecosystem and Municipal Services. Springer Netherlands, Dordrecht, pp 325-338. doi:10.1007/978-94-017-7453-6_23
Whittinghill LJ, Hsueh D, Culligan P, Plunz R (2016) Stormwater performance of a full scale rooftop farm: Runoff water quality Ecological Engineering 91:195-206
WHO (2004) Guidelines for Drinking-water Quality: Recommendations. World Health Organization,
Zhang B, Li Q, Cao J, Zhang C, Song Z, Zhang F, Chen X (2017) Reducing nitrogen leaching in a subtropical vegetable system Agriculture, Ecosystems & Environment 241:133-141
Zhao C, Hu C, Huang W, Sun X, Tan Q, Di H (2010) A lysimeter study of nitrate leaching and optimum nitrogen application rates for intensively irrigated vegetable production systems in Central China Journal of Soils and Sediments 10:9-17

Chapter 4. Heavy Metal Balance of the Brooklyn Grange, an Urban Rooftop Farm

Abstract

Goals of green roof technology recently expanded to include intensive agriculture, aiming to produce fresh and healthy vegetables for local consumption. However, air pollution abatement is among the important ecosystem services emphasized in green roof research. If vegetables and soil in rooftop farms remove air pollutants, then food safety could be compromised by heavy metal contamination, and there is little empirical data from the operational rooftop farms. This study reports the heavy metal balance of the Brooklyn Grange Navy Yard Farm, a 0.6-ha rooftop vegetable farm on an 11-story building in New York City, USA. We measured heavy metal concentration of soil, atmospheric deposition, harvest output, leaching from soil, and drainage output. Drainage Pb and Mn outputs were 4% and 18% of atmospheric deposition respectively, and the Grange was a net sink for Pb and Mn in the urban environment. Heavy metal concentration of soil never exceeded the guideline level, which left the atmospheric deposition the only factor likely to impact food safety. Thirty-three percent of the unwashed leafy vegetable samples exceeded a toxicity-based guideline level for Pb, yet actual heavy metal concentration for human consumption could be lower because all marketable crops are machine-washed upon harvest. In order to reduce the atmospheric deposition of heavy metals onto vegetables and soil, it is important to cover the soil with mulch and discard the used mulch and unmarketable portion of vegetables without recycling them via composting for soil amendments.

4-1. Introduction

Urban planning in the 21st century requires diverse means to restore ecosystem services and enhance sustainability of built environments. Green roofs are increasingly promoted as a way to achieve the specific objectives of storm water management, energy saving, air pollution abatement, biodiversity conservation (Berndtsson 2010; Oberndorfer et al. 2007; Rowe 2011). Recently, the practice of rooftop farming has been promoted as a way to provide healthy and affordable vegetables for local consumption (Ackerman et al. 2013; Harada et al. 2017; Whittinghill and Starry 2016). However, there are likely to be undesirable combinations of ecosystem services that work at cross purposes to each other. For example, if plants and soil can capture atmospheric contaminants, then food safety can be compromised by heavy metal contamination. Particulate matter (PM) can contain heavy metals like Pb, and can deposit directly onto vegetables (Kim et al. 2015; McBride et al. 2014). For relatively soluble heavy metals like As, over the half of the total deposition can occur via precipitation in dissolved forms (Wenzel 2013), and easily reach soil covered by vegetation. If heavy metals accumulate in soil over time, they can be taken up by vegetables and pose hazard to human health (Nabulo et al. 2012; Sanchez-Camazano et al. 1994). However, unlike urban outdoor vegetable production using existing soil, engineered soil products are imported to the entire site for the construction of rooftop farms, which could be less vulnerable to legacy contamination from waste incineration, coal combustion, lead paint and leaded gasoline (Mitchell et al. 2014; Nabulo et al. 2012; Sanchez-Camazano et al. 1994). Also, since irrigation on roof relies on potable water, food security of rooftop farming can be unaffected by the heavy metal contamination of ground water. Atmospheric deposition is the single major flux that could increase the heavy metal concentration of vegetables and soils in rooftop farming, which must be investigated by empirical studies prior to large-scale implementation of rooftop farming systems.

Atmospheric heavy metal deposition to rooftops. To date, we are not aware of any study that reports the rates of atmospheric heavy metal deposition to rooftop farms, but 2 studies provide useful insights on PM deposition on rooftops. In New York City, NY, Tong et al. (2016) monitored $PM_{2.5}$ concentration along a building elevation from curb to roof and found that a rooftop farm 26m above the ground level had 7 – 33% lower $PM_{2.5}$ concentrations than at ground level. In Manchester UK, Speak et al. (2012) studied the PM_{10} captured by vegetation and found PM_{10} deposition rates were different between grass (32.1 kg $ha^{-1}y^{-1}$) and sedum (4.2 kg $ha^{-1}y^{-1}$) species. Heavy metal deposition on rooftops can be less than at ground level, yet can vary by an order of magnitude among plants with different canopy roughness.

Food safety regulations. Public health and trade organizations set guideline levels for heavy metal concentrations of food, which can be used for evaluating food safety of vegetables in rooftop farming. European Union (EU), Food and Agriculture Organization (FAO) and World Health Organization (WHO) of the United Nations have set toxicity-based guidelines for Pb and Cd in vegetables, but not for other heavy metals (EC 2006; FAO WHO 1995) (Table 4-1). There are no accepted toxicity-based heavy metal guideline in the United States, although the Total Diet Study (TDS) by US Food and Drug Administration (FDA) can be used as “market-basket” levels, which are normal ranges of elemental concentrations found in foods in trade including vegetables (US FDA 2007; US FDA 2010). It is noteworthy that all those heavy metal concentrations of vegetable tissue are expressed on a fresh-weight basis.

Table 4-1. Fresh-weight heavy metal concentrations in leafy vegetables and fruits grown in rooftop farms and in-ground community gardens, and comparative concentrations reported by government agencies. *Heavy metal concentrations were converted by the present study from dry to fresh weight using an assumed moisture content for romaine lettuce of 94.9% (US EPA ,1997).

Leafy Vegetables											
TYPE	CROP TYPE	PROTOCOL	YEAR	Pb (mg / kg f.w.)		Cd (mg / kg f.w.)		Cr (mg / kg f.w.)		Region	Author
				MIN	MAX	MIN	MAX	MIN	MAX		
Rooftop Farms	lettuce (1 site)	WASHED	2012 + 2013	0.04	0.08	0.006	0.02	-	-	Paris France	Grard et al (2015)
	Chinese Cabbage + lettuce (18 sites) n = 54	WASHED	2012 + 2013	0.05	0.13	-	-	0.14	1.04	Seoul Korea	Kim et al (2015)*
		UNWASHED	2012 + 2013	0.06	0.22	-	-	0.23	1.13		
In-ground Community Gardens	leafy vegetable (17 sites) n = 67	WASHED	2011 + 2012	0.010	0.59	<0.013	0.20	-	-	NY USA	McBride et al (2014)
Relevant Guideline / Reference Levels											
EU Standard	leafy vegetables	NA	NA	0.3		0.2		NA		EU	EC (2006)
FAO WHO Standard				0.3		0.2				NA	FAO WHO (1995)
US FDA Market Basket	lettuce			0	0.017	0.012	0.175			USA	US FDA (2010)

Heavy metal contents of vegetables from rooftop farms. Among studies on rooftop farming and in-ground community gardens summarized in Table 4-1, the maximum Pb concentration of washed leafy vegetable in rooftop farming (0.13 mg Pb kg^{-1}) was 22% of in-ground community gardens (0.21 mg Pb kg^{-1}), while the maximum Pb concentration of washed fruit in rooftop farming (0.11 mg Pb kg^{-1}) was 52% of in-ground community gardens (0.21 mg Pb kg^{-1}) (Grard et al. 2015; Kim et al. 2015; McBride et al. 2014). In the 2 studies of rooftop farming summarized in Table 4-1, Pb concentrations of all samples were above the market-basket concentration reported by the US FDA (2010), but below the EU guideline levels except for Pb concentrations on washed tomato (0.11 mg Pb kg^{-1}) which exceeded the EU Pb guideline level for fruit (0.1 mg Pb kg^{-1}) (EC 2006; Grard et al. 2015; Kim et al. 2015). Average Pb concentrations in soil were below local guidelines and remained

similar over 2 years of study in both Grard et al. (2015) and Kim et al. (2015), which suggests Pb in vegetables was likely to directly deposited from the atmosphere. Kim et al. (2015) further report unwashed vegetable had 23 – 29% more Pb contents and 4 - 32% more Cr contents than washed samples indicating the direct atmospheric deposition of these metals onto vegetables (Table 4-1). However, atmospheric deposition must be directly monitored by collectors because a fraction of particulates on vegetable surface can be easily washed off by precipitation and another fraction of particulates could remain even after washing in the laboratory (McBride et al. 2014; Schreck et al. 2012; Uzu et al. 2010).

Drainage heavy metal outputs. Drainage output of heavy metals could reduce the residence time of heavy metals within a farming system, thereby reducing the risk of heavy metal uptake by vegetables. To date, however, only Whittinghill et al. (2016) reports the chemistry of drainage water from an operational full-scale rooftop farm, and all heavy metal concentrations were below the limit of detection over a 12 month study, which suggests that either heavy metal atmospheric deposition was negligible, or that metals accumulated in soil and / or crops. Heavy metals have been detected in drainage water from non-production green roofs, yet the determination of source or sink varied for Cr, Cu, Pb, and Zn among the 4 studies we found (Berghage et al. 2009; Berndtsson et al. 2006; Gregoire and Clausen 2011; Speak et al. 2014; Van Seters et al. 2009). For example, Speak et al. (2014) found that a non-production green roof could be a source of Pb, while others found that they functioned as sinks retaining 20-110 g Pb $ha^{-1}y^{-1}$ (Berndtsson et al. 2006; Gregoire and Clausen 2011; Van Seters et al. 2009).

Although multiple studies provide useful insights on heavy metal fluxes in rooftop farms, the current findings from non-production green roofs may not translate directly to rooftop farms due to irrigation, tillage, crop cycles, and frequent amendments. No study reports the formal balance of heavy metals for rooftop farming, including atmospheric deposition, leaching from soil, drainage

output, and output via harvest of vegetables, all of which are necessary for understanding the food safety and environmental benefits of rooftop farming. This study reports the heavy metal balance of a full-scale operational rooftop farm atop an 11-story building in an industrial district of New York City in the US. Our study is motivated by the following questions:

1) What is the heavy metal concentration of vegetables harvested from a rooftop farm?
2) What are the atmospheric deposition rates of heavy metals?
3) What are the heavy metal concentrations in the soil, and are they increasing over time?
4) Does the rooftop farm reduce heavy metals in the urban environment?
5) Are there any hot spots or hot moments of heavy metal fluxes in the rooftop farm, and how this information help farmers improve food safety?

4-2. Material and Methods

Site description. Between 8/2014 – 11/2016, we studied a rooftop farm located at the Brooklyn Grange Navy Yard (the Grange hereafter) atop an 11-story building in the former Brooklyn Navy Yard in New York City, USA. As summarized in Figure 4-1, the roof has a 0.6 ha footprint, of which 0.4 ha is covered with a synthetic soil (Rooflite™ Intensive Agriculture). Each of 8 drains has drainage area dividing the entire farm. Over 60 vegetable crops are produced by the farm.

Among the possible sources of heavy metals is the adjacent 60m-tall building to east of the site, where renovation of this old building continued throughout the study period. The Brooklyn / Queens Expressway (interstate 278) is 250m south from the site, and has annual average daily traffic (AADT) of 123,631 cars day^{-1} (NYS DOT 2015) (Figure 4-2). Within 5km radius, there are 12 major PM_{10} and $PM_{2.5}$ point-source entries in US EPA's (2017) National Emission Inventory, among which a 286-megawatt gas-fired electric power plant (Brooklyn Navy Yard Cogeneration Facility) is 300m

away from the site (Figure 4-2). The predominating wind is from the North, followed by Southwest (Figure 4-2). Wind data were taken every minute by an on-site anemometer (resolution: 0.38 ms^{-1}, accuracy ±1.1 ms^{-1}, model: S-WSA-M003) and wind direction sensor (resolution: 1.4 degrees, accuracy ±5 degrees, model: S-WSA-M003) 2.0 m above the soil surface in the study site.

Figure 4-1. Site information for the Brooklyn Grange Navy Yard farm, NYC.

Location	Brooklyn Navy Yard, Brooklyn, NY		
Geographic coordinate	latitude: 40.698325, longitude: -73.97263		
Elevation above sea level	46 m		
Area	entire site 6122 m^2	covered with soil 4082 m^2	planting bed area: 2429 m^2
			uncropped area: 1653 m^2
		Impervious 2040 m^2	
average catchment slope	2%		
Completion of construction	6/01/2012		
Growing season	April to November		

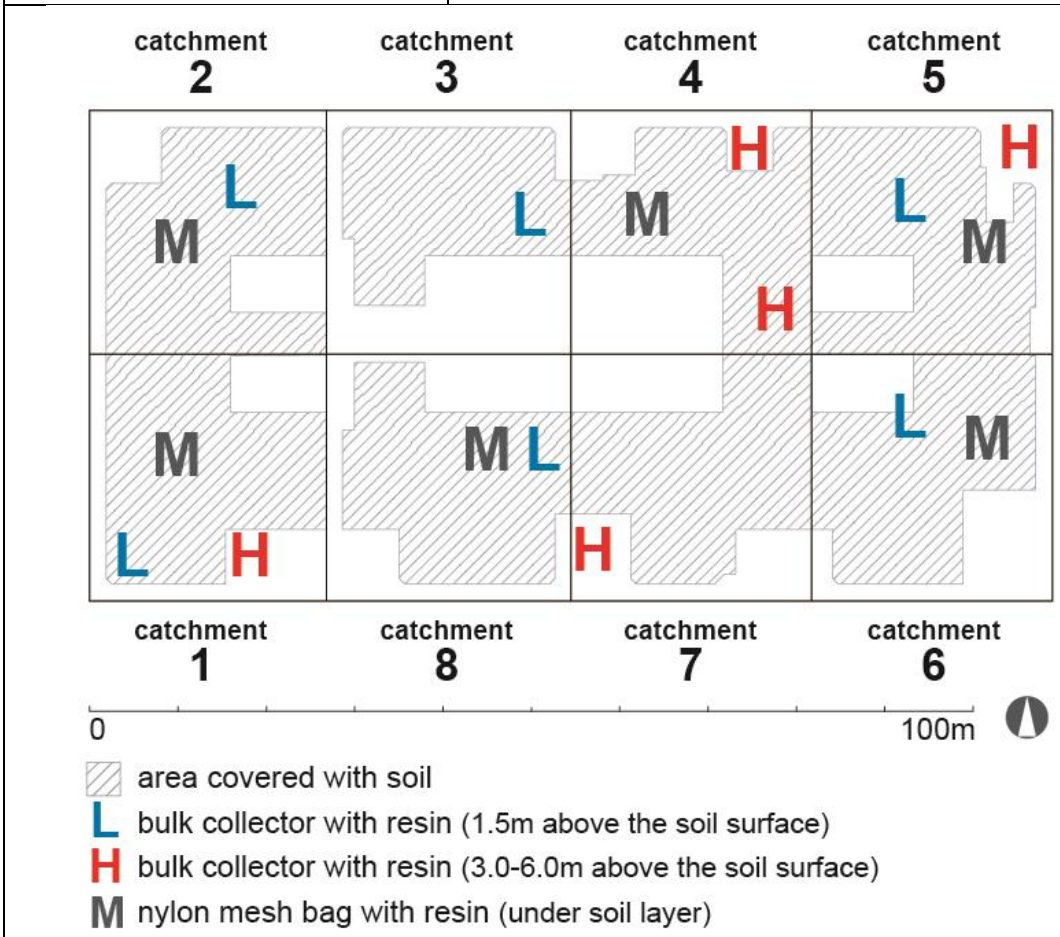
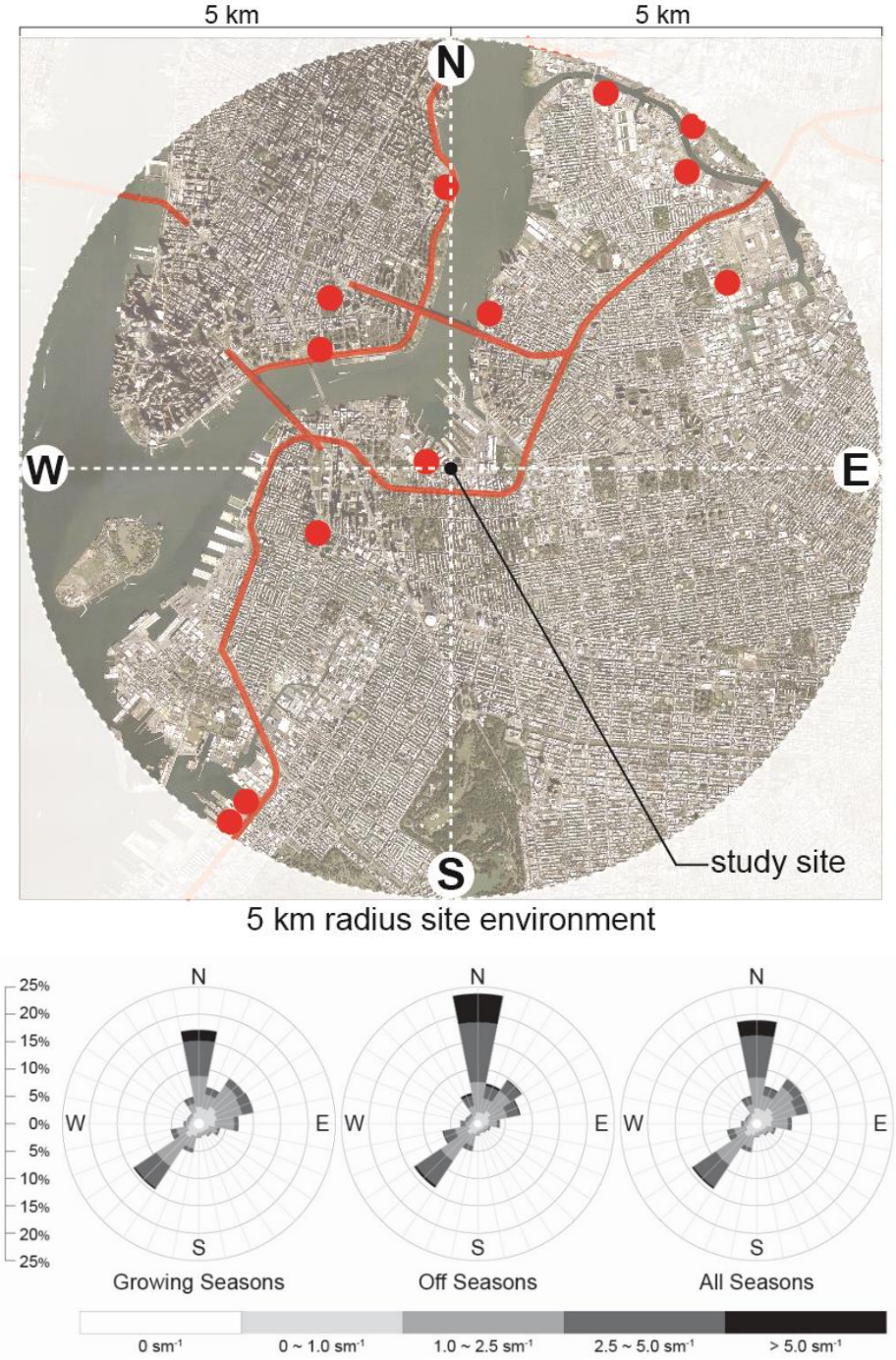


Figure 4-2. The site environment of the Brooklyn Grange Navy Yard farm, NYC. In the top figure, red circles indicate major PM_{10} and $PM_{2.5}$ emission sources identified by the National Emission Inventory US EPA (2017), and red lines indicate roads with annual average daily traffic > 75,000 cars day^{-1} reported by NYS DOT (2015) (image © google 2014). In the bottom figure, wind direction and speed were monitored at the study site between 5/2014 – 11/2016.



Sampling and Analysis. We collected 7 types of samples as summarized in Table 4-2, and heavy metal concentrations of all samples were analyzed by ICP-AES at Cornell Nutrient Analysis Laboratory. Values below the limit of detection were treated as zero.

Table 4-2. Summary of sample type and analyses.

Samples		Sampling Method	Sample for Analysis	Method reference	Analytical Equipment
Atmospheric Deposition	bulk deposition	collector with resin	2M KCl extract of resin	-	ICP-AES at Cornell Nutrient Analysis Laboratory (CNAL)
	filtered deposition	collector with resin	2M KCl extract of resin	-	
	insoluble particulates	cellulosic filter inserted to collector	cellulosic filter	-	
Soil		soil cores		EPA 3050 3051 3052	
Vegetable Tissue		unwashed leaves			
Soil Leachate		mesh bag with resin	2M KCl extract of resin	-	
Drainage Loss		drainage water		-	

Bulk collectors. Atmospheric heavy metal deposition was monitored by bulk collectors at 6 locations 1.5 m above the soil surface, and another 5 locations on rooftop structures located 3.0 – 6.0m above the soil surface (Figure 4-1). Each bulk collector consists of polypropylene Büchner funnel (90 mm diameter) inserted to disposable chromatography column (20 ml) filled with ion-exchange resin (Dowex Amberlite IRN150). Between 8/2014 – 11/2016, resin columns were replaced every 6 weeks, and extracted by a total of 150 ml 2M KCl solution. Concentrations in the solution were converted to flux by taking into account the solution volume, the cross-sectional area of the top of the funnel (9cm diameter), and the sampling period.

Filtered collectors. Each bulk collector was paired with a filtered collector in order to estimate deposition of dry insoluble particulates. Bulk and filtered collectors used the same components, except for cellulosic filters (Grade 42 Whatman quantitative filter papers: WHA1442110) inserted into the Büchner funnels. Resin columns of filtered collectors were replaced and analyzed in the same fashion as bulk collectors. Cellulosic filters were digested using a 50 / 30 volumetric ratio of nitric to perchloric acid at 180 ^{o}C, followed by ICP-AES analysis. Concentrations in the solution

were converted to flux by taking into account the solution volume, the cross-sectional area of the top of the funnel (9cm diameter), and the amount of time the paper filters were incubated in the field (one year).

Leachate from soil. Heavy metal leaching from soil is the vertical export of heavy metals from soil to the drainage layer, and was monitored at 6 locations (Figure 4-2) using nylon mesh bags containing mixed bed ion-exchange resin (Dowex Amberlite IRN150). As shown in Figure 4-3, approximately 100 ml of resin was loaded to each mesh bag which was placed in the rigid circular plastic frame (100 mm diameter), then buried under soil layer (250 mm deep) directly on the filter fabric covering the drainage layer. Every 6 weeks, ion exchange resin bags were sampled from soils upon which 20 ml of ion exchange resin was subsampled, and extracted with 150 ml 2M KCl. Concentrations in resin extracts were converted to flux by taking into account the total volume of KCl used to extract samples (150 ml), the cross-sectional area of the plastic frame (10 cm diameter), and the sampling period.

Figure 4-3. Nylon mesh bags loaded with ion-exchange resin were placed in circular rigid plastic frames, and buried under the soil layer of planting beds, and directly on the filter fabric.

Soil and leafy vegetables. Soil and vegetable samples were collected from the same beds where the resin bags were located (Figure 4-1). Soil sample to the total depth of soil layer (250 mm) was collected before and after each growing season from 8/2014 – 11/2016. Unwashed leaves of arugula and greens mix were collected and analyzed during the 2014, 2015, and 2016 growing seasons. Fresh to dry weight ratio of vegetable sample was estimated by drying samples in oven at °C 60 for 48 hours. Analysis of soil and vegetable samples followed US EPA SW-846 Method 3050, 3051, and 3052 (US EPA 2012). Annual average heavy metal concentration in vegetable samples was converted to flux by taking into account the cropping area and the fresh weight yield from the harvest record (B. Flanner, personal communication), and fresh-to-dry weight ratio.

Drainage output. Lateral flow through the drainage layer contains leachate from soil layer and surface runoff from impervious area, and is lost to drains. Unfiltered drainage samples were collected at least once in each of sampling period (same as bulk collectors and resin bags) between 6/2015 – 11/2016 from all 8 drains and analyzed by ICP-AES. Drainage heavy metal output for each sampling period was calculated as average heavy metal concentration multiplied by cumulative drainage volume estimated from the direct measurement from catchment 8 with a V-notch weir (see Harada et al. 2018 for details). Average concentration for each sampling period was calculated as the daily average concentrations of each sampling date averaged across each sampling period.

Calculation and expression of the annual fluxes. Heavy metal output via harvest of vegetables and leaching from soil are fluxes occurring in planting bed area only, whereas atmospheric heavy metal deposition and the drainage area for the roof applies to the entire roof, including impervious area. To allow for comparison between these fluxes, each flux was expressed for the entire site area (heavy metal mass divided by time and the entire site area) in the annual balance. The annual heavy metal output via harvest of vegetables was calculated by scaling up the estimate for the leafy vegetables.

All other annual fluxes were estimated by aggregating the average daily flux of each sampling period (6 weeks).

4-3. Results

Heavy metal concentration in soil. In the New York State's Environmental Remediation Program, Residential Soil Cleanup Objective specifies the guideline levels of heavy metal concentration of soil for residential land use, which includes vegetable gardening (NYSDEC 2006). Over the study period, the farm-wide average heavy metal concentrations in the soil ranged between 0 – 54% of NYSDEC's (2006) guideline level (Table 4-3, Figure 4-4). Among 6 sampling locations, only catchment 8 (47 mg Cr kg soil^{-1}) before the 2015 growing season, and catchment 5 (49 mg Cr kg soil^{-1}) before the 2016 growing season, exceeded NYSDEC's (2006) guideline level for Cr (36 mg Cr kg soil^{-1}) respectively. All heavy metal concentrations of all other samples ranged between 0-50% of the NYSDEC's (2006) guideline levels (Table 4-3). Highest spatial and temporal fluctuation was found in Pb concentrations, among which the maximum concentration (177 mg Pb kg soil^{-1}) was found in catchment 4 before the 2015 growing season, and was 14 folds of the average (13 mg Pb kg soil^{-1}), whereas the maximum concentrations of all other heavy metals were less than 4 folds of the average concentrations respectively (Table 4-3).

Heavy metal concentrations in leafy vegetables. Over the 3 growing seasons, no sample exceeded toxicity-based Cd guideline level of EU (EC 2006) (0.2 mg Cd kg^{-1} f.w.), while 33% of samples exceeded the Pb guideline (0.3 mg Pb kg^{-1} f.w.), and sample average Pb concentration (0.58 mg Pb kg^{-1} f.w.) was 190% of this guideline level (Table 4-4, Figure 4-5). Compared to the US FDA's (2010) Total Diet Study as a reference, Pb concentrations in 83% of samples exceeded average market basket Pb concentration for raw leaf lettuce (0.005 mg Pb kg^{-1} f.w.), and average Pb

concentration (0.58 mg Pb kg^{-1} f.w.) was 12000% of the average market basket concentration (Table 4-4). The present study reports heavy metal concentrations of unwashed vegetable samples whereas the Grange machine-washes all vegetables with tap water upon harvests, thereby our reported values are likely to be higher than heavy metal concentrations for human consumption, which was not determined by this study.

Table 4-3. Heavy metal concentrations of soil samples collected between 8/2014 – 11/2016 at the Brooklyn Grange Navy Yard farm, NYC. NYSDEC (2006) guideline levels indicate soil cleanup objective used for residential land use including vegetable gardening in the New York State's Environmental Remediation Programs.

		As	Ba	Cd	Cr	Cu	Mn	Pb	Zn
Minimum	mg kg $soil^{-1}$	1.3	29	0	0.82	37	200	0	51
Maximum	mg kg $soil^{-1}$	4.2	66	0.47	49	120	990	180	150
Average	mg kg $soil^{-1}$	2.6	44	0.18	15	68	370	13	92
Median	mg kg $soil^{-1}$	2.7	42	0.16	14	63	340	5.3	88
Standard deviation	mg kg $soil^{-1}$	0.7	10	0.14	10	23	160	30	27
Total number of samples		36	36	36	36	36	36	36	36
Number of samples below the limit of detection		0	0	8	0	0	0	4	0
NYSDEC (2006) Guidance Level	mg kg $soil^{-1}$	16	350	2.5	36	270	2000	400	2200
Number of samples exceeded NYSDEC (2006) Guidance Level		0	0	0	2	0	0	0	0
Percentage of average sample concentration to NYSDEC (2006) Guidance Level		17%	13%	7%	42%	25%	19%	3%	4%
Percentage of maximum sample concentration to NYSDEC (2006) Guidance Level		26%	19%	19%	140%	43%	50%	44%	7%

Figure 4-4. Heavy metal concentrations in soil samples collected between 8/2014 – 11/2016 at the Brooklyn Grange Navy Yard farm, NYC. NYSDEC (2006) guideline levels indicate soil cleanup objective used for residential land use including vegetable gardening in the New York State's Environmental Remediation Programs.

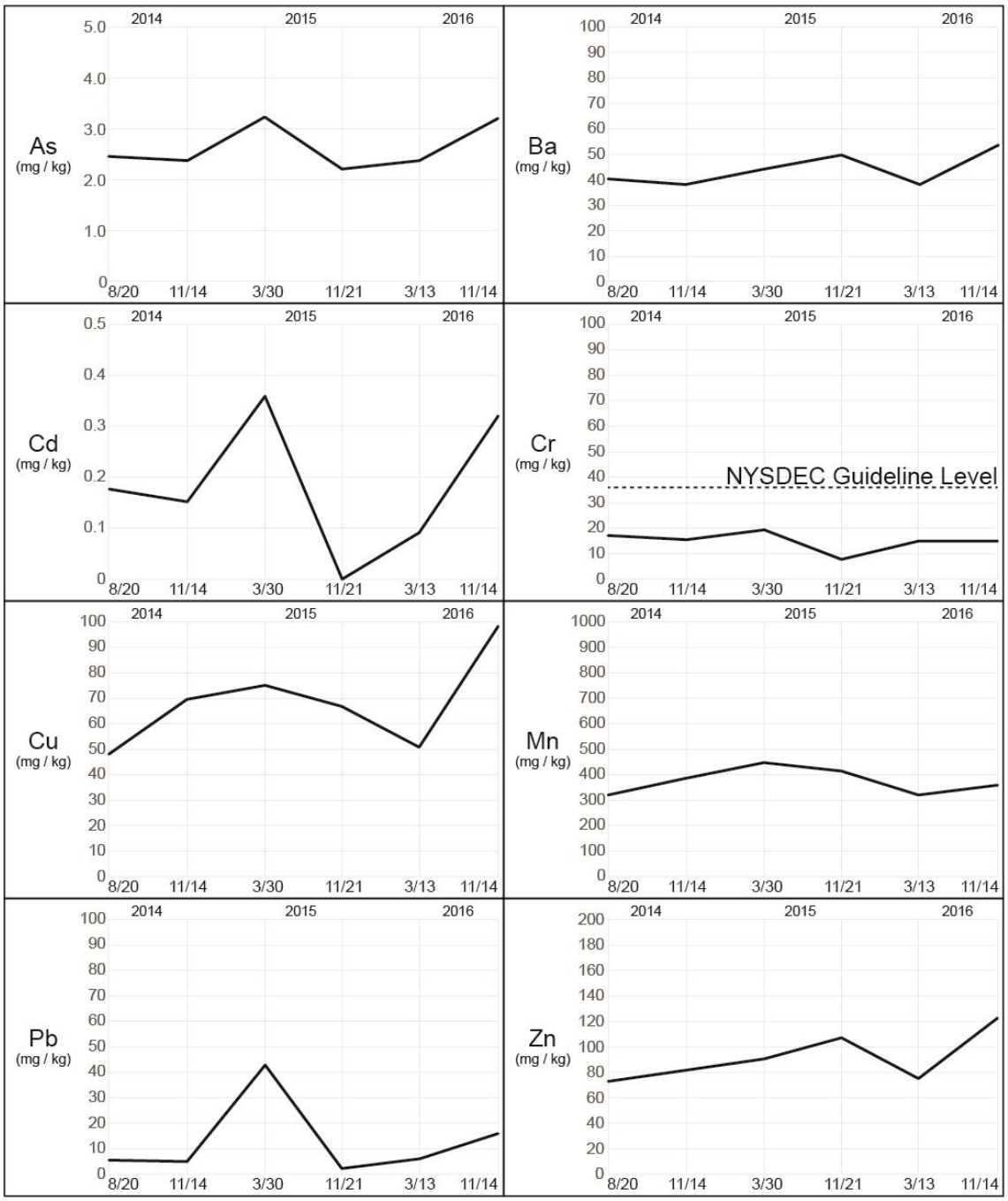
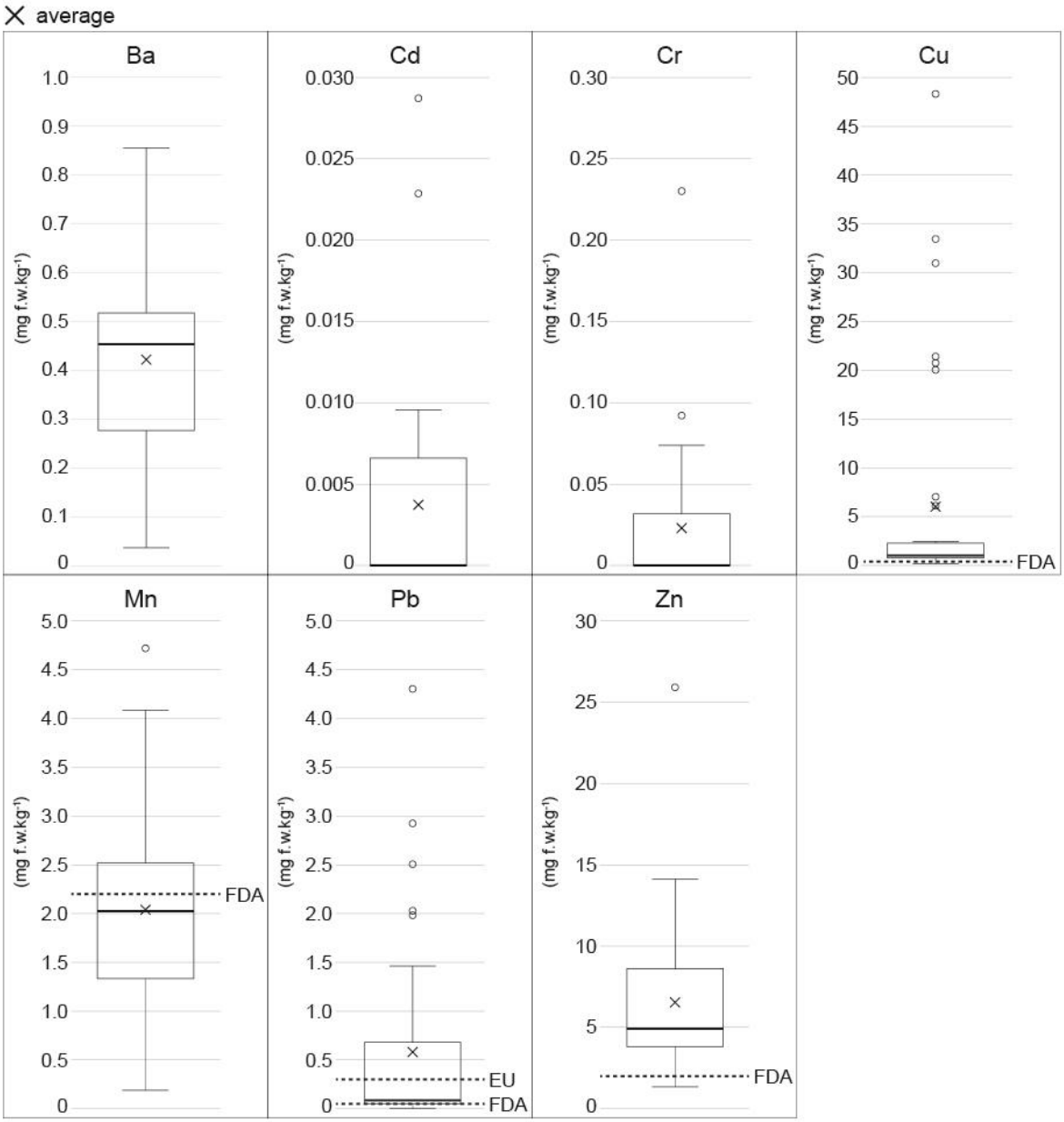


Table 4-4. Summary of fresh-weight heavy metal concentrations of unwashed leafy vegetable samples over the 2014, 2015, and 2016 growing seasons at the Brooklyn Grange Navy Yard farm, NYC. Results for As were unreported due to the equipment problems.

		Ba	Cd	Cr	Cu	Mn	Pb	Zn
The Brooklyn Grange	Minimum (mg / f.w.kg)	0.04	0	0	0.23	0.19	0	1.3
	Maximum (mg / f.w.kg)	0.86	0.03	0.23	48	4.7	4.3	26
	Average (mg / f.w.kg)	0.42	0.004	0.02	6.0	2.0	0.58	6.5
	Median (mg / f.w.kg)	0.45	0	0	1.0	2.0	0.09	4.9
	Standard deviation (mg / f.w.kg)	0.18	0.01	0.04	11	0.95	1.0	4.6
	Total number of samples	36	36	36	36	36	36	36
	Number of samples below the limit of detection	0	21	20	0	0	6	0
Market Basket US FDA (2010) (raw leaf lettuce)	Minimum (mg / f.w.kg)	NA	0.012	NA	0	0.71	0	1.00
	Maximum (mg / f.w.kg)		0.175		0.8	3.90	0.017	3.10
	Average (mg / f.w.kg)		0.066		0.4	2.20	0.005	2.00
	Number of samples exceeded the Average Market Basket levels		0		34	14	30	34
	Percentage of average sample concentration to average Market Basket level		6%		1500%	93%	12000%	330%
EU Guidance Level (leaf vegetables)	EU Guidance Level (mg / f.w.kg)	NA	0.2	NA			0.3	NA
	Number of samples exceeded the EU guidance level		0				12	
	Percentage of average sample concentration to EU guidance level		2%				190%	
	Percentage of maximum sample concentration to EU guidance level		14%				1400%	

Figure 4-5. Heavy metal concentrations of unwashed leafy vegetable samples over the 2014, 2015, and 2016 growing seasons at the Brooklyn Grange Navy Yard farm, NYC. Results for As were unreported due to the equipment problems. EU guideline level (EC, 2006) and average concentration in trade (US FDA, 2010) are shown as dotted lines.



Heavy metal balance. In the annual balance of heavy metals between 11/12 2015 – 11/12 2016, the drainage output exceeded atmospheric deposition for As, Ba, Cd, Cu, and Zn respectively, which suggests that the Grange could be a net source for these 5 heavy metals in the urban environment (Table 4-5). For Pb and Mn, drainage outputs were 6% and 14% of atmospheric deposition respectively, which suggests that the Grange was a net sink of Mn and Pb in the urban environment (Table 4-5). Among 8 metals studied, the balance for Pb shows the highest percentage (70%) of harvest output to the total output (harvest output + drainage output) (Table 4-5). The balance for Zn shows the highest percentage (130%) of harvest output to atmospheric deposition (Table 4-5).

Table 4-5. Annual heavy metal balance between 11/12 2015 – 11/12 2016 for the Brooklyn Grange Navy Yard farm, NYC. All fluxes are expressed for the entire site area. Harvest output for As is unreported due to the equipment problems.

Sample Type		Annual Flux (g $ha^{-1}y^{-1}$)							
		As	Ba	Cd	Cr	Cu	Mn	Pb	Zn
input	deposition*1	29	82	28	3	62	101	53	54
sub-system flux	leaching*2 (from soil to drainage layer)	25	136	35	2	14	296	7	5
output	harvest*3	NA	5 (12)	0.09 (0.2)	0.3 (1)	53 (134)	26 (65)	7 (18)	70 (176)
	drainage	157	101	126	3	315	14	3	167

Compartment	Annual Budget (g $ha^{-1}y^{-1}$)							
	As	Ba	Cd	Cr	Cu	Mn	Pb	Zn
Drainage Layer (leaching - drainage)	-132	+35	-91	-1	-301	+283	+4	-161
Soil Layer (deposition - harvest - leaching)	NA	-58	-7	+1	-5	-222	+38	-21
Atmosphere-to-Sewer continuum (deposition - drainage)	-128	-19	-98	0	-253	+87	+50	-113
Ecosystem (deposition - drainage - harvest)	NA	-24	-98	0	-306	+61	+43	-183

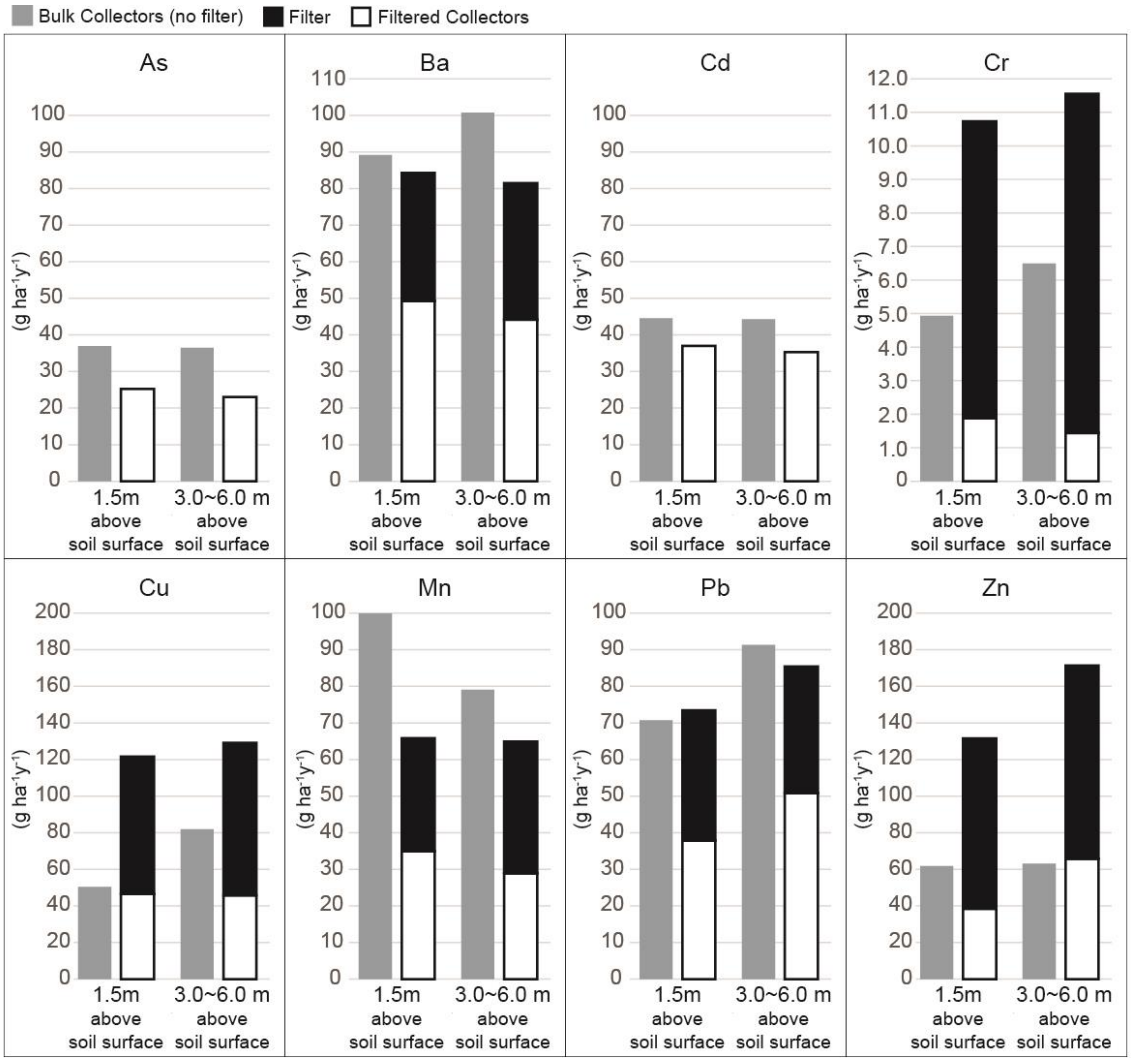
*1 bulk collector with ion-exchange resin 1.5m above the soil surface

*2 soil mesh bag with ion exchange resin.

*3 values in brackets are expressed for just the planting bed area.

Insoluble particulate deposition. Cellulosic filters inserted to filtered collectors and incubated between 4/1/2015 - 4/1/2016 were analyzed for heavy metal concentrations in order to estimate the insoluble particulate of atmospheric deposition. On the farm-wide annual average, filtered deposition plus deposition absorbed in filters were higher than bulk deposition for Cr, Cu, Zn, and were lower than bulk deposition for Ba and Mn in both collector heights (Figure 4-6). Filtered samples plus deposition absorbed in filters were closest to bulk deposition for Pb, and was 100% (1.5m above the soil surface) and 94% (3.0-6.0m above the soil surface) of bulk deposition respectively (Figure 4-6).

Figure 4-6. Annual heavy metal deposition collected by cellulosic filters, filtered samples, and bulk samples at the Brooklyn Grange Navy Yard farm, NYC, between 4/1 2015 – 4/1 2016. Results of filters for As and Cd were unreported due to the equipment problems.



4-4. Discussion

Heavy metals in soil. The farm-wide average heavy metal concentrations in soil were below 54% of the NYSDEC's (2006) guideline for field soils, yet this translates to an even lower concentration if we account for the fact that the average bulk density of the soil was only 0.62 ± 0.03 g cm^{-3}. This is less than 60% of bulk density generally found in field soil (Gee 2005; Gregory et al. 2006; Jim 1998; Pouyat et al. 2007; Scharenbroch et al. 2005), thereby the heavy metal concentrations at the Grange would fall below 32% of the guideline. In an univariate analysis summarized in Figure 4-7, concentrations of Ba, Cu, and Zn do increase slightly over time, yet not for Pb and Cd. In summary, we found little need of soil decontamination or replacement at the Grange regarding the heavy metal concentration.

Heavy metals in leafy vegetable. The heavy metal concentrations of unwashed leafy vegetables over the 3 growing seasons was highly variable, and the ratio of average to maximum concentrations ranged from 200% (Ba), up to 1200% (Cr) (Table 4-4). On a dry-weight basis, the Grange's average Pb concentration was 310% of average Pb concentration reported by Kim et al. (2015) from urban rooftop farms in Seoul, Korea (Table 4-6). Also, Kim et al. (2015) reports annual average Pb concentration from each of 18 sites over 2 years were all on the same order of magnitude, whereas annual average varied by two orders of magnitude at the Grange over the 3 years (Table 4-6). This suggests that the Pb concentration of vegetables at the Grange could be increased by the atmospheric deposition from relatively temporal sources, perhaps close to our study site, yet the source determination requires further research.

Figure 4-7. Univariate analysis of heavy metal concentrations in soil as a function of time since the first sampling in 8/2014, at the Brooklyn Grange Navy Yard farm, NYC.

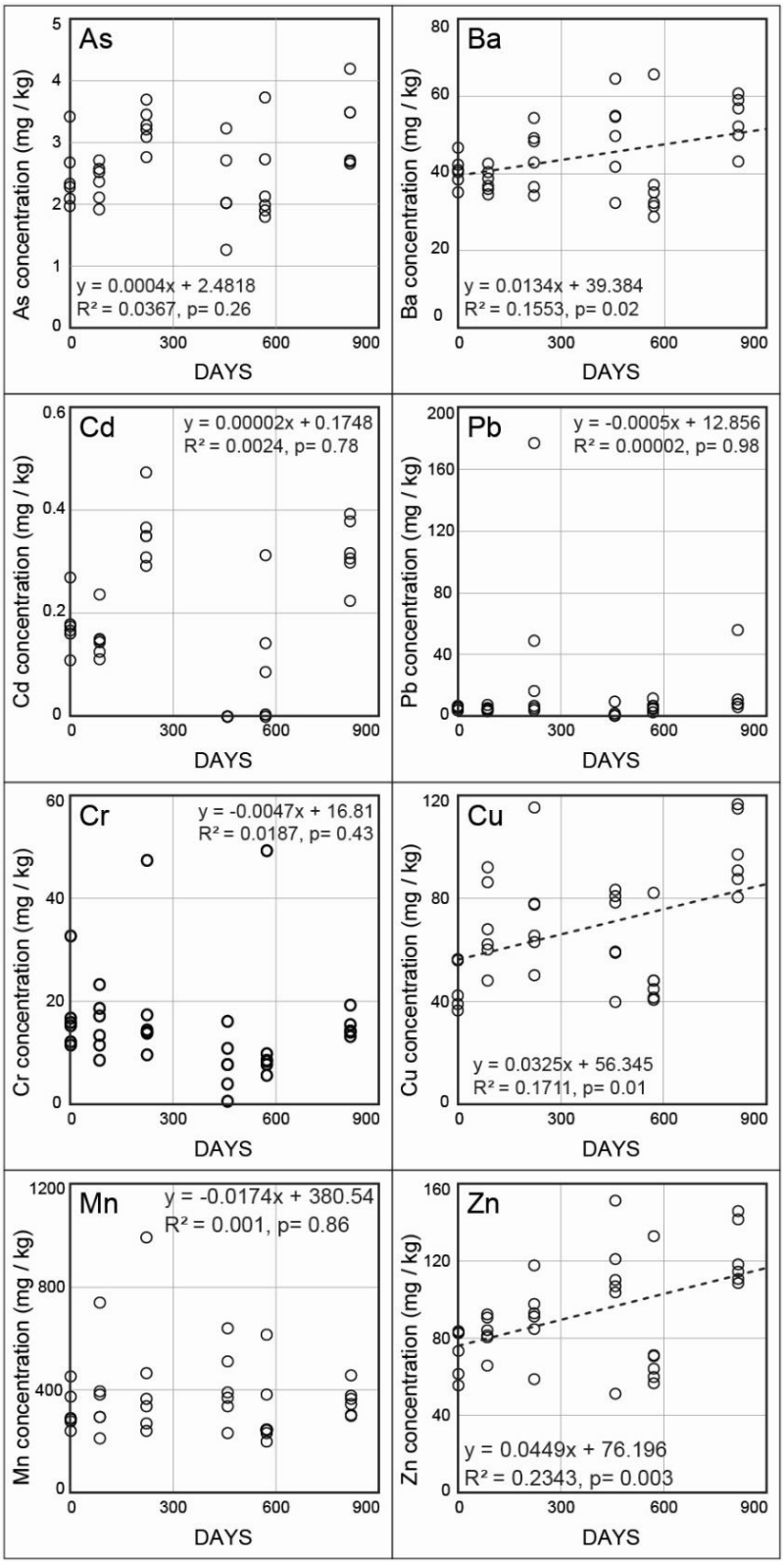


Table 4-6. Dry-weight heavy metal concentrations in unwashed samples of leafy vegetables grown in urban rooftop farms.

Year	average heavy metal dry-weight concentrations (mg / kg)							location	author
	Ba	Cd	Cr	Cu	Mn	Pb	Zn		
2014	5.7	0.05	0.20	120	29	11	100	NYC USA	this study
2015	3.7	0.01	0.40	8.4	14	0.32	37		
2016	3.7	0.07	0.21	41	20	5.7	54		
overall	**4.8**	**0.04**	**0.26**	**71**	**23**	**6.8**	**74**		
2012	-	-	19.56	4.96	-	2.49	33.12	Seoul Korea	Kim et al. (2015)
2013	-	-	6.53	5.71	-	1.95	60.33		
overall			**13.05**	**5.34**		**2.22**	**46.73**		

Atmospheric deposition of heavy metals. Atmospheric deposition of heavy metals can be highly variable and defies simple generalization across different site environments, yet the comparison within urban sites, as well as between urban and non-urban sites, could provide a useful context. In comparison to the studies of non-urban sites summarized in Table 4-7, the As and Cd deposition were greater at the Grange by an order of magnitude, whereas the deposition of Cr, Cu, Mn, Pb, and Zn were on the same order of magnitude as non-urban sites respectively (Golomb et al. 1997; Pike and Moran 2001; Sweet et al. 1998). Among urban sites summarized in Table 4-7, the Zn deposition at the Grange is an order of magnitude lower than the others, and the Pb and Cu deposition were bracketed by the others respectively (Gregoire and Clausen 2011; Sabin et al. 2005; Shahin et al. 2000). In summary, whether deposition is greater in the Grange or in non-urban sites depends on the heavy metal element, and further research is needed in urban sites, especially for As and Cd.

Table 4-7. Annual atmospheric deposition of heavy metals (4/1/2015 – 4/1/2016) at the Brooklyn Grange Navy Yard farm, NYC, compared with other studies in USA. Bulk deposition is shown as the total deposition.

deposition type		deposition flux (g ha^{-1} y^{-1})								site type	location	author
		As	Ba	Cd	Cr	Cu	Mn	Pb	Zn			
1.5m above soil surface	total	**37**	**89**	**45**	**4.9**	**51**	**100**	**71**	**62**	urban	New York City NY	this study
3.0-6.0m above soil surface	total	**36**	**101**	**44**	**6.5**	**82**	**79**	**92**	**63**			
	dry	-	-	-	4.4	32	-	20	130		Los Angeles CA	Sabin et al (2005)
	wet	-	-	-	0.18	2.0	-	0.29	15			
	total	-	-	-	**4.6**	**34**	-	**20**	**145**			
	dry	-	-	-	-	230	-	140	440		Chicago IL	Shahin et al (2000)
	wet	-	-	-	-	20	-	110	380		Storrs CT	Gregoir et al (2011)
	dry	0.19	-	1.3	12	20	27	13	51	non-urban (urban Influenced)	Massachusetts Bay	Golomb et al (1997)
	wet	0.035	-	1.4	15	5.0	7.5	5.7	27			
	total	**0.22**	-	**2.7**	**27**	**25**	**34**	**18**	**78**			
	dry	0.88	-	4.0	10	33	21	7.8	110		Lake Erie	Sweet et al (1998)
	wet	0.94	-	0.94	0.63	8.5	24	10	55			
	total	**1.8**	-	**4.9**	**11**	**42**	**45**	**18**	**170**			
	dry	0.91	-	3.8	1.3	24	19	9.2	53	non-urban	Lake Superior	
	wet	0.78	-	0.78	0.78	7.0	23	5.5	35			
	total	**1.7**	-	**4.6**	**2.2**	**31**	**42**	**15**	**88**			
	dry	0.66	-	3.8	1.3	13	9.1	9.5	18		Lake Michigan	
	wet	0.72	-	0.72	0.72	5.7	19	6.4	42			
	total	**1.4**	-	**4.5**	**2.0**	**19**	**28**	**16**	**60**			
	dry	0.56	-	0.13	0.41	1.6	-	3.7	10		Gulf of Maine	Pike & Moran (2001)
	wet	2.7	-	1.2	0.78	6.7	-	7.8	83			
	total	**3.2**	-	**1.3**	**1.2**	**8.3**	**-**	**12**	**94**			

Heavy metal balance. The ecosystem budget of heavy metals suggests that only Mn and Pb increased in the Grange. The annual increase of Pb was 1400% of drainage output, which translate that it could take 14 years to purge the annual increase from the system, and it is important to reduce the atmospheric deposition of Pb. Comparison of atmospheric deposition with drainage output could suggest whether the Grange was a net source or sink of heavy metals in the urban environment. Among the goals of stormwater management is the reduction of heavy metals in the urban

environment, and it is useful to compare the heavy metal reduction between the Grange with non-production green roofs and other stormwater management systems. Among the 5 studies of stormwater management systems summarized in Table 4-8, the percentage of drainage output to control was largest from the Grange for Cd (450%), Cu (510%), and Zn (310%) (Berghage et al. 2009; Berndtsson et al. 2006; Driscoll et al. 2015; Gregoire and Clausen 2011; Speak et al. 2014; Van Seters et al. 2009). For these 3 heavy metals, the Grange may not be useful for stormwater management. Conversely, only for the Grange and non-production green roofs from 2 studies, the percentage of drainage Pb output to control was less than 10% (Berndtsson et al. 2006; Gregoire and Clausen 2011). The relatively high retention rate of Pb at the Grange could be useful for stormwater management, yet may compromise the food safety.

In addition to the heavy metal drainage output, heavy metal concentrations of drainage water is important for environmental regulators. As summarized in Table 4-9, all 200 drainage samples from the Grange exceeded the US EPA's (2017) guideline for freshwater maximum concentration for Cd (1.8 μg Cd L^{-1}) by an order of magnitude. In comparison to the urban surface runoff summarized in Table 4-9, the average concentration of drainage water from the Grange was the highest for Cd (12 μg Cd L^{-1}), yet lowest for Cr (0.4 μg Cr L^{-1}), Pb (0.4 μg Pb L^{-1}), and Zn (18 μg Zn L^{-1}) respectively (Davis et al. 2001; Göbel et al. 2007). In summary, when stormwater management is based on the drainage concentration of heavy metals, the Grange could be useful for the reduction of Cr, Pb, and Zn, yet not for Cd.

Leaching from the soil layer and surface runoff from impervious walkway are collected by the drainage layer, followed by the lateral flow to the drains. The annual heavy metal budget for the drainage layer (Table 4-5) suggests that Ba, Mn, and Pb were likely to be retained in the drainage layer. For the rest of 5 heavy metals studied, the drainage layer is the source if these 5 heavy metals were released from the components of the drainage layer, such as expanded-shale gravel and the

water-proofing membrane. Another factor that can affect the heavy metal budget for the drainage layer is the surface runoff from the impervious walkways. In conventional roofs and non-productive green roofs, maintenance is the only human activity, whereas the Grange offers environmental educational programs and other cultural events in addition to the farming activities. All these human activities could have transported heavy metals from the ground level, which requires further research.

Table 4-8. Annual drainage heavy metal output from various stormwater management systems compared with the Brooklyn Grange Navy Yard farm, NYC. *Maximum observed values.

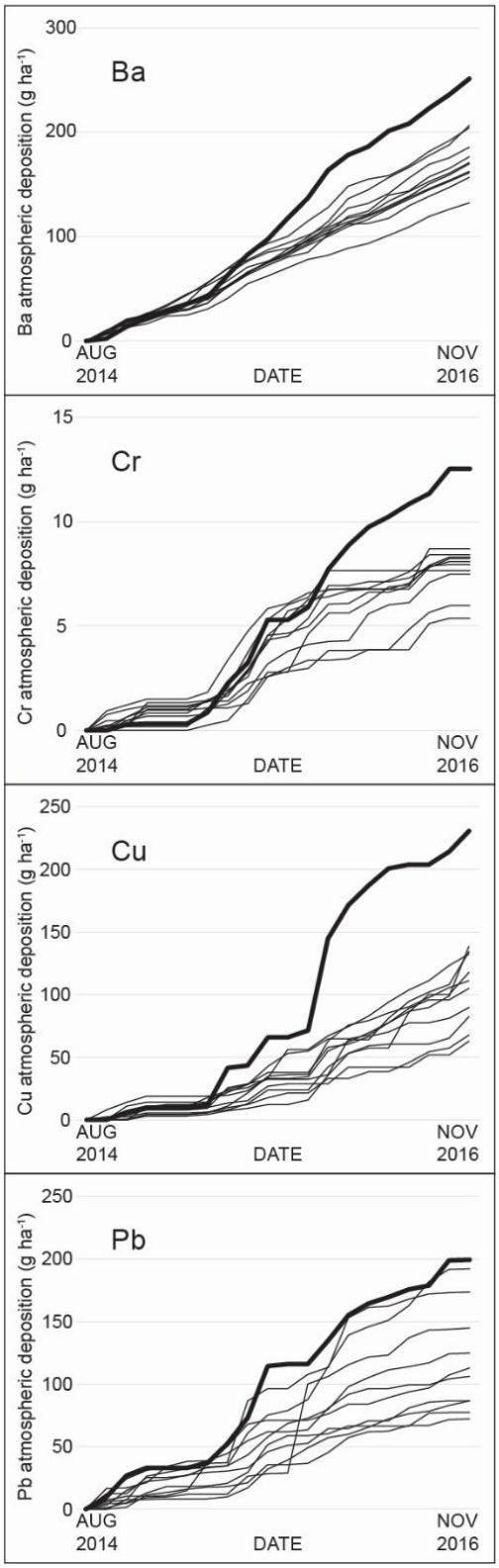
system type and control	flux ($g / ha^{-1}y^{-1}$)					drainage output / control (%)					location	author
	Cd	Cr	Cu	Pb	Zn	Cd	Cr	Cu	Pb	Zn		
green roof	-	0.7	57.2	1.5	71.1	-	39	397	7	92	Malmo Sweden	Berndtsson et al (2006)
rainfall	-	1.8	14.4	21.6	77.6							
green roof	-	-	30	0	130	-	-	150	0	34	Storrs CT, USA	Gregoir et al (2011)
rainfall	-	-	20	110	380							
green roof	-	-	-	-	-	3.6	134.9	11.1	734.4	16.6	Manchester UK	Speak et al (2014)*
rainfall	-	-	-	-	-							
green roof	-	-	-	-	70	-	-	-	-	189	University Park PA, USA	Berghage et al (2009)
asphalt	-	-	-	-	37							
green roof	1.75	3.02	156	16.2	33.7	48	41	14	47	31	Toronto Canada	Van Seters et al (2009)
asphalt	3.64	7.39	1108	34.5	110.2							
rooftop farm	126	3	315	3	167	450	100	510	6	310	NYC NY, USA	this study
bulk deposition	28	3	62	53	54							
green roof	NA					40.6	NA		38.9	NA	Mostly USA	Driscoll et al (2015)
bioretention cell						53.5			50.1			
swale						70.8			77.3			
media filter						71.8			59.9			
retention pond						75.2			82.4			
detention pond						75.8			68.2			
porous pavement						87.2			87.9			
wetland						112.2			83.1			

Table 4-9. Heavy metal concentrations in 200 drainage samples collected between 6/2015 - 11/2016 at the Brooklyn Grange Navy Yard farm, NYC, compared with runoff from various urban and industrial land uses. *1National Recommended Water Quality Criteria for freshwater. *2Guideline level for chromium (VI). *3Number of samples exceeding the guideline levels specified as maximum concentration in EPA (2017).

		heavy metal concentration (μg L^{-1})								author
		As	Ba	Cd	Cr	Cu	Mn	Pb	Zn	
US EPA (2017) guideline levels*1	Continuous concentration	150	-	0.72	11*2	-	-	2.5	120	US EPA (2017)
	Maximum concentration	340	-	1.8	16*2	-	-	65		
Rooftop Farm (The Brooklyn Grange)	Average (± SD)	16 (± 2)	11 (± 7)	12 (± 2)	0.4 (± 0.3)	34 (± 7)	1.6 (± 5.1)	0.4 (± 0.8)	18 (± 9)	this study
	maximum	21	39	15	2.4	59	67	6.5	75	
	number of exceedance*3	0	-	200	0	-	-	2	0	
Roof (residential)	average	-	-	0.12	-	7.5	-	1.5	100	Davis et al. (2001)
Roof (commercial)	average	-	-	1.3	-	200	-	62	1100	
Roof (institutional)	average	-	-	0.6	-	5000	-	64	1100	
Roof (no zinc gutters and downpipes)	representative values determined by meta-analysis	-	-	0.8	4	153	-	69	370	Göbel et al. (2007)
Roof (with zinc gutters and downpipes)		-	-	0.8	4	153	-	69	1851	
Green roof		-	-	0.1	3	58	-	6	468	
Gardens, grassed areas, cultivated land		-	-	0.7	3	11	-	9	80	
Pedestrian / cycle way		-	-	0.8	-	23	-	107	585	
Parking lot		-	-	1.2	-	80	-	137	400	
service road		-	-	1.6	10	86	-	137	400	
main road		-	-	1.9	11	97	-	170	407	
Motorway (highway)		-	-	3.7	13	65	-	224	345	

Hot spots and hot moments. The soil sample with the maximum Pb concentration (180 mg kg soil^{-1}), and 2 other samples exceeding the NYSDEC guideline for Cr were all collected at the end of the winter fallow periods when the soil was bare. During the fallow periods, atmospheric heavy metal deposition to soil could be reduced by mulching and the use of cover crops, both of which must be removed from the site, instead of being used as amendments via composting. Among 11 locations for bulk collectors, cumulative atmospheric deposition was highest in northeast corner of the farm for Ba, Cr, Cu, and Pb, which was 140%, 160%, 200%, and 160 % of the farm-wide average respectively (Figure 4-8). Possible sources of these heavy metals include the adjacent 60m-tall building east of the site where renovations were occurring. In the locations with relatively high rates of atmospheric heavy metal deposition, we recommend growing root vegetables and ornamentals and covering the soil with mulch. Currently, planting beds receive composted unmarketable parts of vegetables, such as leaves of root vegetables and stems and leaves of fruit-bearing vegetables. Because theses may receive deposition to leaves, we would advise that these unmarketable portions be discarded, and not used for amendments via composting.

Figure 4-8. Cumulative atmospheric deposition of each of 11 bulk collectors between 8/2014 – 11/2016 at the Brooklyn Grange Navy Yard farm, NYC. Bold lines indicate the bulk collector at Northeast corner of the site 6.0m above the soil surface.



4-5. Conclusions

This study reports the heavy metal balance of the Brooklyn Grange, an operational rooftop vegetable farm in New York City, USA. We found that the atmospheric deposition of Pb and Mn exceeded the drainage output to the municipal sewer system, hence the Grange was a net sink for Pb and Mn respectively. For As, Ba, Cd, Cu, and Zn, the Grange could be a net source in the urban environment. There were unwashed vegetable samples exceeding the guideline level for Pb. This exceedance is unlikely to be caused by the vegetables' metal uptake from soil because heavy metal concentration in the soil never exceeded the guideline level, which leaves atmospheric deposition as the only factor affecting food safety. It is important to recognize that the heavy metal concentrations in vegetables for human consumption are probably lower than our findings because vegetables are machine-washed prior to sale. Food safety could be improved by reducing atmospheric deposition of heavy metals by covering the soil with mulch. Further research is needed on the effectiveness of mulch to improve food safety, the heavy metal transport from the ground level to rooftop by human activity, and the atmospheric deposition of heavy metals in different site environments and climate zones.

REFERENCE

Ackerman K, Dahlgren E, Xu X (2013) Sustainable Urban Agriculture: Confirming Viable Scenarios for Production, Final Report No.13-07 vol NYSERDA No. 13-07 [Online] <http://urbandesignlab.columbia.edu/files/2015/04/2_Sustainable-Urban-Agriculture_NYSERDA.pdf> (Accessed on April 01, 2016).

Berghage RD, Beattie D, Jarrett AR, Thuring C, Razaei F, O'Connor TP (2009) Green roofs for stormwater runoff control. EPA 600-R-09-026. National Risk Management Research Laboratory Office of Research and Development U.S. Environmental Protection Agency, Cincinnati, OH. Citeseer,

Berndtsson JC (2010) Green roof performance towards management of runoff water quantity and quality: A review Ecological Engineering 36:351-360

Berndtsson JC, Emilsson T, Bengtsson L (2006) The influence of extensive vegetated roofs on runoff water quality Science of the Total Environment 355:48-63

Davis AP, Shokouhian M, Ni S (2001) Loading estimates of lead, copper, cadmium, and zinc in urban runoff from specific sources Chemosphere 44:997-1009

Driscoll CT et al. (2015) Green Infrastructure: Lessons from Science and Practice. A publication of the Science Policy Exchange:32

EC (2006) Commission regulation (EC) no 1881/2006 of 19 December 2006, Setting maximum levels for certain contaminants in foodstuffs Official Journal of the European Union L 364

FAO WHO (1995) GENERAL STANDARD FOR CONTAMINANTS AND TOXINS IN FOOD AND FEED (CODEX STAN 193-1995).

Gee GW (2005) TEXTURE A2 - Hillel, Daniel. In: Encyclopedia of Soils in the Environment. Elsevier, Oxford, pp 149-155. doi:https://doi.org/10.1016/B0-12-348530-4/00428-8

Göbel P, Dierkes C, Coldewey W (2007) Storm water runoff concentration matrix for urban areas Journal of contaminant hydrology 91:26-42

Golomb D, Ryan D, Eby N, Underhill J, Zemba S (1997) Atmospheric deposition of toxics onto Massachusetts Bay—I. Metals Atmospheric Environment 31:1349-1359 doi:https://doi.org/10.1016/S1352-2310(96)00276-2

Grard BJ-P et al. (2015) Recycling urban waste as possible use for rooftop vegetable garden Future of Food: Journal on Food, Agriculture and Society 3:21-34

Gregoire BG, Clausen JC (2011) Effect of a modular extensive green roof on stormwater runoff and water quality Ecological Engineering 37:963-969

Gregory JH, Dukes MD, Jones PH, Miller GL (2006) Effect of urban soil compaction on infiltration rate Journal of soil and water conservation 61:117-124

Harada Y, Whitlow TH, Bassuk NL, Russell-Anelli J (2017) Biogeochemistry of Rooftop Farm Soils. In: Lal R, Stewart BA (eds) Urban Soils. Advances in Soil Science Series. Taylor & Francis Group, Portland, United States,

Jim CY (1998) Urban soil characteristics and limitations for landscape planting in Hong Kong Landscape and Urban Planning 40:235-249 doi:https://doi.org/10.1016/S0169-2046(97)00117-5

Kim H-S, Kim K-R, Lim G-H, Kim J-W, Kim K-H (2015) Influence of airborne dust on the metal concentrations in crop plants cultivated in a rooftop garden in Seoul Soil Science and Plant Nutrition 61:88-97

McBride MB et al. (2014) Concentrations of lead, cadmium and barium in urban garden-grown vegetables: the impact of soil variables Environmental Pollution 194:254-261

Mitchell RG et al. (2014) Lead (Pb) and other metals in New York City community garden soils: factors influencing contaminant distributions Environmental Pollution 187:162-169

Nabulo G, Black C, Craigon J, Young S (2012) Does consumption of leafy vegetables grown in peri-urban agriculture pose a risk to human health? Environmental pollution 162:389-398

New York State Department of Transportation Traffic Data Viewer (2015) https://www.dot.ny.gov/tdv. Accessed August 30, 2017

NYSDEC (2006) 6 NYCRR Part 375-Environmental Remediation Programs. New York State Department of Environmental Conservation.,

Oberndorfer E et al. (2007) Green roofs as urban ecosystems: ecological structures, functions, and services BioScience 57:823-833

Pike SM, Moran SB (2001) Trace elements in aerosol and precipitation at New Castle, NH, USA Atmospheric Environment 35:3361-3366 doi:https://doi.org/10.1016/S1352-2310(00)00525-2

Pouyat RV, Yesilonis I, Russell-Anelli J, Neerchal N (2007) Soil chemical and physical properties that differentiate urban land-use and cover types Soil Sci Soc Am J 71:1010-1019

Rowe DB (2011) Green roofs as a means of pollution abatement Environmental Pollution 159:2100-2110

Sabin LD, Lim JH, Stolzenbach KD, Schiff KC (2005) Contribution of trace metals from atmospheric deposition to stormwater runoff in a small impervious urban catchment Water research 39:3929-3937

Sanchez-Camazano M, Sanchez-Martin M, Lorenzo L (1994) Lead and cadmium in soils and vegetables from urban gardens of Salamanca (Spain) Science of the Total Environment 146:163-168

Scharenbroch BC, Lloyd JE, Johnson-Maynard JL (2005) Distinguishing urban soils with physical, chemical, and biological properties Pedobiologia 49:283-296 doi:http://dx.doi.org/10.1016/j.pedobi.2004.12.002

Schreck E et al. (2012) Metal and metalloid foliar uptake by various plant species exposed to atmospheric industrial fallout: mechanisms involved for lead Science of the Total Environment 427:253-262

Shahin U, Yi S-M, Paode RD, Holsen TM (2000) Long-term elemental dry deposition fluxes measured around Lake Michigan with an automated dry deposition sampler Environmental science & technology 34:1887-1892

Speak A, Rothwell J, Lindley S, Smith C (2012) Urban particulate pollution reduction by four species of green roof vegetation in a UK city Atmospheric Environment 61:283-293

Speak A, Rothwell J, Lindley S, Smith C (2014) Metal and nutrient dynamics on an aged intensive green roof Environmental Pollution 184:33-43

Sweet CW, Weiss A, Vermette SJ (1998) Atmospheric deposition of trace metals at three sites near the Great Lakes Water, Air, and Soil Pollution 103:423-439

Tong Z, Whitlow TH, Landers A, Flanner B (2016) A case study of air quality above an urban roof top vegetable farm Environmental Pollution 208:256-260

SW-846 Waste e Hazard Waste Test Methods, Online Document URL (http://www.epa.gov/osw/hazard/testmethods/sw846/online/index.htm), Accessed 09.24.2017 (2012). Accessed 09.24.2017

Facility Emission Summaries Top PM2.5 and PM 10 emitting sectors (2017a) US EPA. https://www3.epa.gov/air/emissions/where.htm. Accessed August 30, 2017

National Recommended Water Quality Criteria, Online Document URL (http://www.epa.gov/osw/hazard/testmethods/sw846/online/index.htm), Accessed 09.24.2017 (2017b). Accessed 09.24.2017

US FDA (2007) Total Diet Study Statistics on Element Results Revision 4.1, Market Baskets 1991-3 through 2005-4. US Food and Drug Administration, College Park, MD.

US FDA (2010) Total Diet Study Statistics on Element Results Market Baskets 2006-1 through 2008-4. US Food and Drug Administration, Center for Food Safety and Applied Nutrition, College Park, MD.

Uzu G, Sobanska S, Sarret G, Munoz M, Dumat C (2010) Foliar lead uptake by lettuce exposed to atmospheric fallouts Environmental Science & Technology 44:1036-1042

Van Seters T, Rocha L, Smith D, MacMillan G (2009) Evaluation of green roofs for runoff retention, runoff quality, and leachability Water Quality Research Journal of Canada 44:33

Wenzel WW (2013) Arsenic. In: Alloway BJ (ed) Heavy Metals in Soils: Trace Metals and Metalloids in Soils and their Bioavailability. Springer Netherlands, Dordrecht, pp 241-282. doi:10.1007/978-94-007-4470-7_9

Whittinghill L, Starry O (2016) Up on the Roof: Considerations for Food Production on Rooftops. In: Brown S, McIvor K, Hodges Snyder E (eds) Sowing Seeds in the City: Ecosystem and Municipal Services. Springer Netherlands, Dordrecht, pp 325-338. doi:10.1007/978-94-017-7453-6_23

Whittinghill LJ, Hsueh D, Culligan P, Plunz R (2016) Stormwater performance of a full scale rooftop farm: Runoff water quality Ecological Engineering 91:195-206