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The Effects of Annual Compost Addition to Agricultural Green Roofs on Runoff Water Quality

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Abstract. **Space availability is one of the largest barriers to urban agriculture. One way around this issue that urban farmers in some parts of the world are exploring is moving their farming activities to building rooftops. One method of rooftop farming in use is row agriculture using green roof technology. Vegetable crop plants, which typically require more water and more nutrients than the ornamental species found on green roofs, require irrigation and the use of fertilizers. One nutrient management practice that some rooftop farmers are using is the addition of compost, which could lead to changes over time in the water-holding capacity, organic matter content, and weight of green roof media. This practice and its long-term implications have not been well-studied. Green roof platforms were created to examine how the annual additions of compost in quantities of 0, 0.33, 0.66, and 1 kg/m^2 affect runoff water quality and green roof media properties. Runoff water samples were collected and analyzed for pH, conductivity, color, turbidity, and nitrate nitrogen, ammonia nitrogen, total phosphorus, and potassium contents. Compost treatment had no effect on any water quality metric except for color, which had slightly different changes over time in the different compost treatments. The lack of difference among the treatments may be attributed to the low nutrient content of the compost and continued use of fertilizers to provide nutrients. Most samples observed in this study exceeded the US Environmental Protection Agency water quality guidelines for nitrate nitrogen and phosphorus and were similar to values observed in the green roof literature regarding agricultural and ornamental green roofs. This has potential implications for surface water quality and eutrophication, especially as green roof agriculture increases.**

Space availability is one of the largest barriers to urban agriculture. This is partly caused by competition between urban agriculture and other land uses, especially development (Graefe et al. 2008; Vagneron 2007), which typically leaves little unused land in urban centers. One solution to this issue is to move agricultural activities to rooftops. This is a practice growing in popularity among urban farmers that reduces competition with development for space; however, it can be more expensive, more labor-intensive, and require higher input than farming in the ground (Appolldoni et al. 2021; Buehler and Junge 2016; Specht et al. 2014; Thomaier et al. 2014; Whittinghill and Rowe 2012). Several different rooftop agriculture practices are in use, including container production, hydroponics (both open air and inside greenhouses), and the use of green roof technology (Appolloni et al. 2021; Buehler and Junge 2016).

Green roofs are rooftops that support plant life using engineered technology. These are composed of a drainage layer over the roof waterproofing, filter fabric, engineered media, and vegetation (Whittinghill et al. 2013). Green roofs vary in the amount of filter fabric and media used and the plant communities found on them. The species chosen for planting are generally determined by the roof load capacity (which restricts media depth), media composition, water use needs, budget, and intent of the project (Dvorak and Volder 2010). Ornamental green roofs also provide a variety of ecosystem services, including stormwater mitigation, energy savings, and mitigation of the urban heat island (Alsup et al. 2013; Fassman-Beck et al. 2013; Gong et al. 2019; Jadaa et al. 2019; Karczmarczyk et al. 2020; Rowe 2011; Saadatian et al. 2013). What can be planted on such a roof and the exact benefits provided depend on the depth of the growing media, which is dictated by the load capacity of the underlying rooftop. Additional limitations on plant selection may exist because of the limited nutrient content and fast-draining nature of green roof media; however, these can be overcome with irrigation and nutrient management (Bates et al. 2015; Clark and Zheng 2013, 2014; Whittinghill et al. 2016b). This has enabled crop plants to be included on the list of plant types found on green roofs.

Crop plants require more nutrition than the *Sedum* spp. typically found on ornamental green roofs. The University of Kentucky recommends that 2.24 to 16.81 g/m^2 of nitrogen should be applied each year for vegetable crop production (Saha et al. 2016). This is much greater than the 7.03 g/m^2 of nitrogen recommendation for ornamental green roofs (FFL 2002). Rooftop farmers compensate for this difference in nutrient needs by adding nutrients to the rooftop in the form of compost and/or fertilizers (Grard et al. 2015; Harada et al. 2018a; Whittinghill et al. 2016a). The effects of fertilizer use on ornamental green roofs are well-studied. The use of fertilizers can result in higher nutrient concentrations in runoff from the roof, and even more so when highly soluble fertilizers are used (Buffam et al. 2016; Clark and Zheng 2013, 2014; Czemiel Berndtsson 2010; Rowe 2011). Nutrient concentrations in green roof runoff have also been linked to higher organic matter content, with a positive dose-response (Czemiel Berndtsson 2010; Hathaway et al. 2008; Ntoulas et al. 2015; Rowe 2011). This effect has been demonstrated when organic matter is added to the roof after installation (Czemiel Berndtsson 2010), as would be the case with regular compost additions. Ornamental green roofs also tend to have lower concentrations of nutrients in runoff as they age, which is thought to be linked to reduced media organic matter and nutrient content (Buffam and Mitchell 2015; Czemiel Berndtsson 2010; Mitchell et al. 2017; Rowe 2011).

Few studies have examined the impact of runoff water quality with nutrient management on green roof farms. Of the few that exist, even fewer have compared green roof farms with ornamental green roofs. However, they have demonstrated that nutrient concentrations in runoff from agricultural green roofs are higher than those of runoff from ornamental green roofs (Whittinghill et al. 2015, 2016a). Agricultural green roofs are considered a source of nutrients, even though studies have not compared them to ornamental green roofs (Harada et al. 2017, 2018b). Controlled experiments examining the use of compost amendments to green roof media that include crop or herb plants showed that base media (Elstein et al. 2008; Grard et al.

2018; Harada et al. 2020), compost quantity and type (Kong et al. 2015; Matlock and Rowe 2017), and crop plants (Aloisio et al. 2016; Matlock and Rowe 2017) have an impact on nutrient leaching. Compost with more soluble nutrients, such as vermicompost, resulted in higher levels of nutrient runoff, but not throughout the study (Matlock and Rowe 2017). Both Matlock and Rowe (2017) and Kong et al. (2015) showed a decrease in nitrogen leaching over time after compost or fertilizer addition; neither tracked the nitrogen dynamics after a second compost addition. Kong et al. (2015) found that compost treatments had lower nitrogen concentrations in leachate than the synthetic fertilizer treatment, but not across all time points.

No long-term studies have examined the effects of annual compost additions to green roofs, which is a common practice for agricultural green roofs (Almaaitah and Joksimovic 2022; Harada et al. 2018a; Whittinghill et al. 2016a). The continued use of this practice and the nature of nutrient leaching on green roofs raise questions about whether soil building on such roofs is taking place or if nutrients from the compost are lost to the system in too short of a timeframe to achieve soil building. At the very least, the practice could disrupt the typical reduction in nutrient concentrations that occurs as roofs age (Mitchell et al. 2017).

As previously stated, the configuration of a green roof relies, at least in part, on how much weight the underlying building structure can support. Therefore, green roof media is engineered to be light. In general, green roofs are designed to match the load capacity of the building. A main factor in weight is the water-holding capacity, which is affected by the media depth, texture, and organic matter content and roof age (FFL 2002; Dvorak and Volder 2010). An agricultural green roof with approximately 20 cm of growing media would weigh approximately 961 to 1121 kg/m^3 when fully saturated (Skyland LLC 2017). If 0.64 kg/m^2 of compost is added to the roof on an annual basis, as is performed for some agricultural roofs (Whittinghill et al. 2016a), then this could lead to a build-up of organic matter, an increase in the water-holding capacity (Eksi et al. 2015; Matlock and Rowe 2017), and an increase in the saturated weight of the green roof. Although this increase in roof weight may be small over the course of 1 or 2 years, no long-term studies have examined a system in which compost is continually added over the life of the roof. An alternative scenario is that the compost breaks down over the course of 1 year, thus releasing nutrients that may not be used by the crops and leading to a potential environmental problem. A recent study has supported this latter scenario by demonstrating that substrate depth does not change over time; however, media and runoff nutrient content were not the focus of that study (Almaaitah and Joksimovic 2022).

This study was designed to determine which of these two outcomes was likely on agricultural green roofs and evaluate the impact of annual compost addition on plant productivity, nutrient leaching from the green roof media, and green roof media properties. This work focuses on the impacts on nutrient leaching. Runoff water properties were analyzed at regular intervals throughout the 2-year study period, thus enabling an examination of the impact of the compost additions as well as seasonal changes in management, crop cover, and weather.

Materials and Methods

Sixteen green roof platforms were constructed at Kentucky State University's Harold R. Benson Research and Demonstration Farm in Frankfort, KY, USA, in May 2018. These consisted of 1.22-m × 1.22-m decks topped with 30-cm-deep raised beds filled with 5.08 cm of Rooflite® Drain media, topped with Rooflite® Separation Fabric, and finally filled with 20.32 cm of Rooflite® intensive green roof media (Skyland, LLC, Landenberg, PA, USA), as outlined in Whittinghill and Poudel (2020). A 5.08-cm gap was left at the front of each raised bed and covered with a Phifer super solar charcoal fiberglass replacement screen (Tuscaloosa, AL, USA) to allow for water flow but retain the media. Covered gutters fitted with down spouts were attached to the front of each bed at this gap. Down spouts drained into 7.57-L buckets for runoff collection.

A randomized complete block design was used with four replicates of each nutrient management treatment. Green roof media was amended with one of four compost treatments at the beginning of the growing season in 2018 and 2019 (Table 1). The compost treatments consisted of 0, 0.33, 0.66, and 1 kg/m^2 of Garden Magic® compost and manure (0.1–0.1–0.1) (Michigan Peat Company, Houston, TX, USA). These treatment rates were selected because one full-scale rooftop farm supplied approximately 0.64 kg/m^2 of compost to their roof in 2013 (Whittinghill et al. 2016a). The remaining plant nutrients were supplied using the following three organic fertilizers: tomato tone (3–4–6), bone meal (4–12–0), and blood meal (12–0–0) (The Espoma Company, Milleville, NJ, USA). Fertilizers were applied at each planting (Table 1) to meet the following nutrient recommendations for greens according to the College of Agricultural and Environmental Sciences at the University of Georgia: 19.61 g N/m^2, 16 g P_2O_5/m^2, and 16 K_2O/m^2. Seven greens were planted and harvested (Table 1). Greens were selected for their prevalence in urban production, including rooftop settings (Cho et al. 2010; Grard et al. 2015; Harada et al. 2017; Mowrer et al. 2019; Whittinghill et al. 2016a; Whittinghill and Sarr 2021), their suitability for production in small areas and raised beds (Berle and Westerfield 2013), high nutritive value (Bojilov et al. 2020; Carillo et al. 2020; Casajús et al. 2021; Cheng et al. 2021; Corrado et al. 2020; Reda et al. 2020), and responsiveness to changes in nutrient sources and application rates (Chakwizira et al. 2015; Coria-Cayupán et al. 2009; de Barros Sylvestre et al. 2019; Iheshiulo et al. 2017; Pereira et al. 2020; Stagnari et al. 2007, 2015). Irrigation was supplied to the crops through drip irrigation to meet the recommended 2.5 cm of water per week, including rain. More details about crop production can be found in the work by Whittinghill and Poudel (2020).

Runoff samples were collected from each plot once each month, weather permitting, from Jul 2018 through Feb 2020. A water quality analysis was not performed during four separate months during the study period (Dec 2018, Mar 2019, Sep 2019, and Nov 2019). No water was collected during these months because there was no rain (Sep 2019), insufficient rain, or larger rainstorms occurred when samples could not be collected. When storms early in the month did not result in enough runoff water, further attempts were made to collect water later in the month. Samples of 250 mL each were collected from each plot with a functional gutter and downspout and transported to Kentucky State University's Aquaculture Research Center for laboratory analysis. The pH values were measured using a Fisherbrand™ Accumet™ AP110 Meter Kit (Thermo Fisher Scientific, Inc., Waltham, MA, USA). Conductivity values were obtained using an Oakton® Con 6 + meter kit (Vernon Hills, IL, USA). Color was analyzed using a LaMotte Smart3 colorimeter (LaMotte Company, Chestertown, MD, USA), and turbidity was determined using a LaMotte 2020we turbidity meter (LaMotte Company). Nitrate nitrogen (NO_3^-), ammonium nitrogen (NH_4^+), total phosphorus, and potassium were analyzed using Hach Water Quality Testing Protocols and a Hach DR600 Spectrophotometer (Hach, Loveland, CO, USA). Before the nutrient analysis, samples were filtered according to vacuum filtration techniques using Whatman® paper filters (grade 40; Global Life Sciences Solutions USA, LLC, Marlborough, MA, USA), which had a diameter of 4.7 cm.

Occasional issues with downspout or collection bucket failure occurred and resulted in lower than necessary water collection volumes. In these cases, no analysis was performed or some of the analyses were performed and chosen based on water volume requirements to

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Table 1. Nutrient management, planting, and harvesting timeline with calendar dates and days after compost addition for the successional planting of *Lactuca sativa* (Encore lettuce mix), *Eruca sativa* (Astro arugula), *Brassica rapa* (Mizuna Asian greens), *Brassica juncea* (Red giant mustard greens), *Beta vulgaris* (Fordhook giant Swiss chard), *Brassica napus* (Red Russian kale), and *Spinacia oleracea* (Covair spinach) during both study years.

	Year 1 (2018)		Year 2 (2019)	
Activity	Date	Days after compost addition	Date	Days after compost addition
Compost added	16 May	0	4 Apr	0
Fertilizer added and lettuce planted			11, 12 Apr	7–8
Lettuce harvest			23, 24 May	49–50
Fertilizer spread and arugula planted	4 Jun	19	28 May	54
Arugula harvest	11 Jul	56	19–21 Jun	76–78
Fertilizer spread and mizuna planted	11 Jul	56	21 Jun	78
Mizuna harvest	6 Aug	82	11 Jul	98
Fertilizer spread and mustard planted	6 Aug	82	11, 12 Jul	98–99
Mustard harvest	30 Aug	106	5 Aug	123
Fertilizer spread and Swiss chard planted			6 Aug	124
Swiss chard harvest			12, 13 Sep	161–162
Fertilizer spread and kale planted	30 Aug	106	13 Sep	162
Kale harvest	26 Sep	133	16, 17 Oct	197–198
Fertilizer spread and spinach planted	26 Sep	133		
Spinach harvest	22 Oct	159		

maximize the number of analyses that could be performed. Water collection system failures increased during the second study year as downspouts aged. Laboratory equipment failure resulted in the inability to accurately test for potassium in runoff samples from most container plots in Sep 2018. Furthermore, NO_3^- could not be tested in any plots in Jul 2019 because the laboratory did not have enough staff, and samples needed to be stored for longer than recommended for that analysis. This, along with occasional downspout failure or low runoff volume, resulted in missing data points for some of the plots in each growing system and each fertilizer treatment at least once. Because sampling took place on different days after compost (DAC) addition during study years 1 and 2, comparisons between years were limited to samplings that occurred within 7 d of each other and the study year main effect, as appropriate. The following four pairs of sampling dates were within 7 d of each other: 61 and 67, 145 and 152, 184 and 186, and 281 and 286 (Table 2).

Average air temperature and total precipitation data were obtained for the Franklin County MESONET weather station located at the Harold R. Benson Research and Demonstration Farm, which is 250 m from the experimental site. Climatic normal air temperature and precipitation data based on a time period from 1981 to 2010 were obtained from the National Oceanographic and Atmospheric Administration (NOAA) Climatic Data Online for the weather station at the Capital City Airport in Frankfort, KY, USA.

Statistical analyses were performed using R (version 4.1.1; The R Project for Statistical Computing, Vienna, Austria). Data that did not conform to a normal distribution were transformed using Box-Cox transformation [pH and ammonia nitrogen (NH_3) ($\lambda = -2$); log transformation (NO_3^-, phosphorus), cube root transformation (conductivity, color), and inverse transformation (turbidity)]. All water quality measures were analyzed using a linear mixed effects model (lme4 package in R) with compost treatment, DAC addition, and study year as fixed effects, and plot was used as the random effect for repeated measures. Significant differences among means of compost treatments, DAC addition, and significant interactions were separated using least-square means (using the emmeans and multcomp packages in R), with an α level of 0.05.

Results

Weather data. Temperatures during 2018 were higher than those during 2019 in May and June; however, they were lower during 2018 than during 2019 in July and September (Fig. 1A). Temperatures were close to the climatic normal during both years of the study, but they were slightly higher than normal in September and lower than normal in November during both years. Temperatures in 2019 were also higher than normal during January, February, April, and May; however, they were lower than normal in March. Total precipitation was 1341 mm for the study period of 2018, and 1483 mm for 2019. Precipitation in 2018 was higher than normal and highest in August and September (Fig. 1B). Precipitation in 2019 was highest in February and October and higher than normal in November and December. Precipitation during the summer months was lower than normal in 2019. This included a drought in Kentucky from 28 Aug to 6 Oct, during which time there was only 10 mm of precipitation.

pH. The two-way interactions between compost treatment and DAC addition and compost treatment and study year were not significant (F = 0.906; P = 0.439 and F = 1.000; P = 0.394, respectively). Compost treatment had no significant effect on runoff water pH (F = 1.713; P = 0.210) (Table 3, Fig. 2A). There was a general trend of decreasing runoff water pH with time after compost addition (Fig. 2B). One exception to this trend was at 55 DAC addition (mean ± *SE*: 6.84 ± 0.06), with pH that was significantly lower than that at all other sampling days except 243, 286, and 328 DAC addition. Two other exceptions were 145 and 186 DAC addition (8.19 ± 0.23 and 8.17 ± 0.22, respectively), which had the highest average pH and higher variability than other sampling days. There was only one sampling date pair with a significant difference between study years 1 and 2: 184 and 186 DAC addition. Runoff water pH was significantly higher during study year 2 (186 DAC addition) than during study year 1 (184 DAC addition, 7.32 ± 0.03).

Conductivity. The two-way interactions between compost treatment and DAC addition and compost treatment and study year were not significant (F = 1.671; P = 0.174 and F = 1.181; P = 0.909, respectively). Compost

Table 2. Water sampling dates and their corresponding days after compost addition during both study years.

	Year 1		Year 2	
Sampling month	Date	Days after compost addition	Date	Days after compost addition
April			8 Apr 2019	4
May			29 May 2019	55
June			10 Jun 2019	67
July	16 Jul 2018	61	18 Jul 2019	105
August	8 Aug 2018	84	27 Aug 2019	145
September	24 Sep 2018	131		
October	15 Oct 2018	152	7 Oct 2019	186
November	16 Nov 2018	184		
December			3 Dec 2019	243
January	24 Jan 2019	253	14 Jan 2020	286
February	21 Feb 2019	281	25 Feb 2020	328
March				

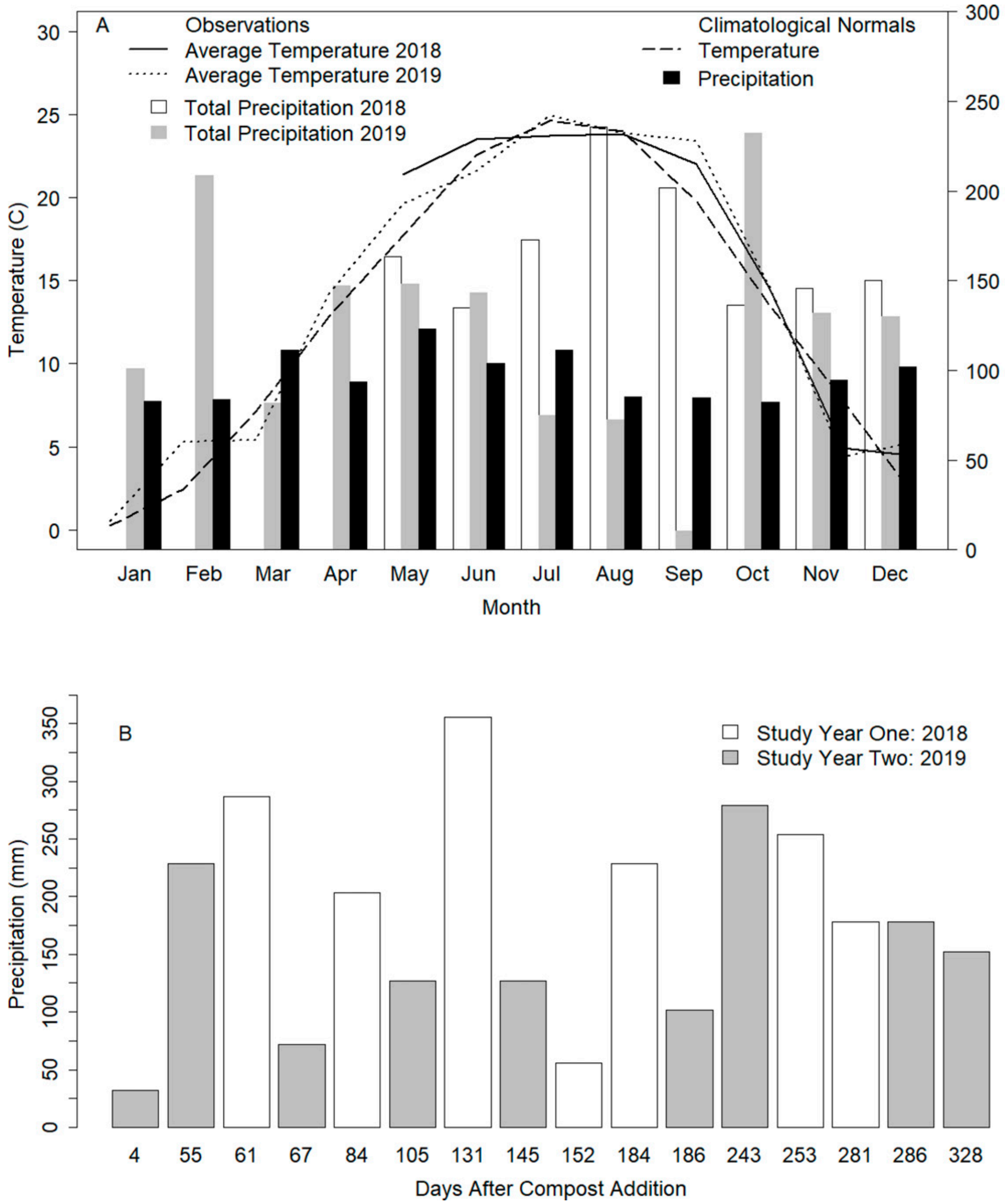

Fig. 1. (**A**) Monthly total precipitation (mm) and average temperature (°C) during the period of study between May 2018 and Dec 2019 and 30-year (1981–2010) climatological normal (NOAA 2020). (**B**) Total precipitation between sampling times during the first (2018) and second (2019) years of the study.

treatment had no effect on the conductivity of runoff water ($F = 0.402$; $P = 0.754$) (Table 3, Fig. 3A). There was a general trend of higher conductivity from growing season sampling dates (approximately 152 DAC addition and earlier) than from sampling dates during winter months (approximately 184 DAC addition and later) (Fig. 3B). Conductivity was highest approximately midway through the growing season (e.g., 67 DAC addition, 1092 ± 91; 152 DAC addition, 933 ± 144 µs/cm). Exceptions to the general pattern were observed at 55 and 105 DAC addition (173 ± 14 µs/cm and 10 ± 0 µs/cm, respectively), with significantly lower conductivity than that at many growing season sampling dates. Conductivity at 105 DAC addition was significantly lower than that at all other sampling dates, except at 243 DAC addition (67 ± 35 µs/cm), and had very low variability. Variability was highest at 152 DAC addition, followed by that at 4 and 84 DAC addition. There was only one sampling date pair with a significant difference between study years 1 and 2: 145 and 152 DAC addition. Conductivity was significantly

Table 3. Means and *SE* of runoff water quality from green roof platforms with 0, 0.33, 0.66, and 1 kg/m^2 of compost added annually.

	Compost treatment			
Water quality metric	0 kg/m^2	0.33 kg/m^2	0.66 kg/m^2	1 kg/m^2
pH	7.52 (0.07)	7.63 (0.06)	7.48 (0.07)	7.68 (0.08)
Conductivity (µs/cm)	421.2 (57.3)	461.9 (55.4)	375.0 (49.7)	403.4 (53.6)
Turbidity (NTU)	2.26 (0.27)	2.53 (0.26)	2.80 (0.50)	2.36 (0.27)
Nitrate nitrogen (mg/L)	18.59 (3.56)	32.84 (8.85)	36.10 (9.47)	19.35 (3.67)
Ammonia nitrogen (mg/L)				
Total phosphorus (mg/L)	3.04 (0.35)	3.99 (0.41)	3.50 (0.37)	3.48 (0.35)
Potassium (mg/L)	51.74 (11.08)	66.96 (11.63)	65.83 (11.87)	64.36 (13.57)

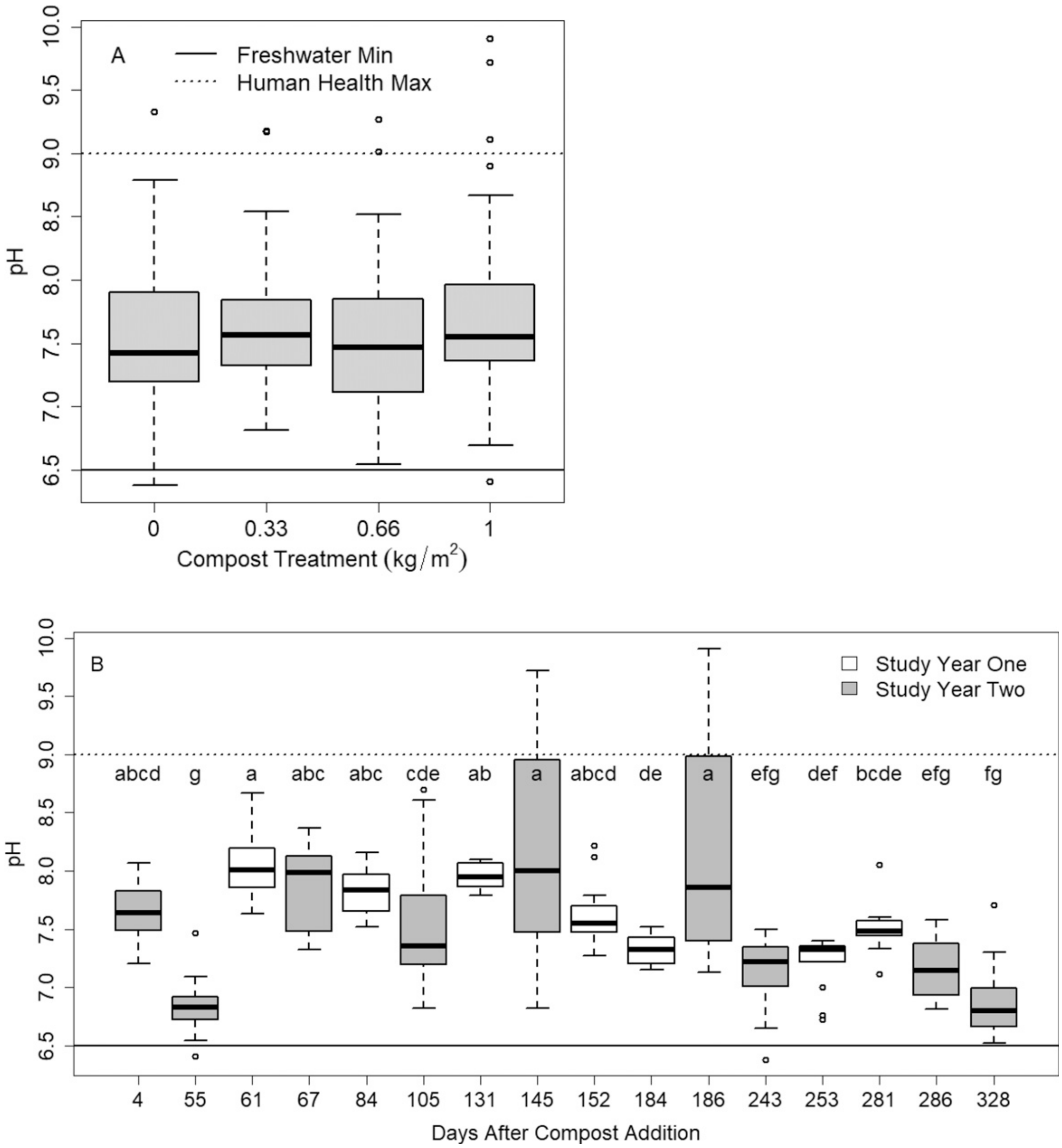

Fig. 2. The pH of runoff water for each (**A**) compost treatment and (**B**) sampling time in days after compost addition for study years 1 and 2. US Environmental Protection Agency freshwater minimum (6.5) (USEPA 2022a) and human health maximum (9.0) pH (USEPA 2022b) thresholds are included as solid and dashed lines, respectively. Letters denote significant differences among sampling time means.

higher during study year 1 (152 DAC addition) than during study year 2 (145 DAC addition, 101 ± 24 μs/cm).

Color. The two-way interaction between compost treatment and DAC addition was not significant (F = 2.391; P = 0.069); however, color was only slightly higher than the α level of 0.05 and could suggest a trend. There were no significant differences among compost treatments within any sampling date. Significant differences among sampling dates within compost treatments did vary slightly among the compost treatments; however, they followed the same general pattern (Fig. 4). Color was highest on 84 and 105 DAC addition, but not significantly higher than that at most other sampling times. With the control treatment, 84 and 105 DAC addition (925 ± 57 and 1041 ± 639 PtCo, respectively) had significantly higher color than that at 145, 243, 286 and 328 DAC addition (43 ± 6, 24 ± 22, 38 ± 15, and 33 ± 22 PtCo, respectively), which were not significantly different from each other (Fig. 4A). With the 0.33 kg/m^2 compost treatment, color at 84 and 105 DAC addition (959 ± 19 and 748 ± 203 PtCo, respectively) was only significantly higher than that at 145 DAC addition (90 ± 40 PtCo) (Fig. 4B). With the 0.66 kg/m^2 compost treatment 4, color at 84, 105, and 131 DAC addition (588 ± 38, 942 ± 31, 797 ± 230, and 388 ± 107 PtCo, respectively) was significantly higher than that at 328 DAC addition (69 ± 32 PtCo) (Fig. 4C). With the 1 kg/m^2 compost treatment, color at 84, 105 and 131 DAC addition (1052 ± 69, 1113 ± 134, and 718 ± 28 PtCo, respectively) was significantly higher than that at 145, 186, 243, 286, and 328 DAC addition (43 ± 12, 43 ± 13, 37 ± 9, 124 ± 76, and 50 ± 7 PtCo, respectively) (Fig. 4D). In that compost treatment, color at 145 and 184 DAC addition was also significantly lower than that at 67 DAC addition (457 ± 7 PtCo). There were no significant differences between study years for any sampling date pair within any compost treatment. The two-way interaction between compost treatment and study year was not significant (F = 1.130; P = 0.337). Overall, color was higher during study year 1 (477 ± 26 PtCo) than during study year 2 (350 ± 33 PtCo).

Turbidity. The two-way interactions between compost treatment and DAC application and study year were not significant (F = 0.863; P = 0.46 and F = 0.770; P = 0.512, respectively). Compost treatment had no effect on the turbidity of runoff water (F = 1.128; P = 0.370)

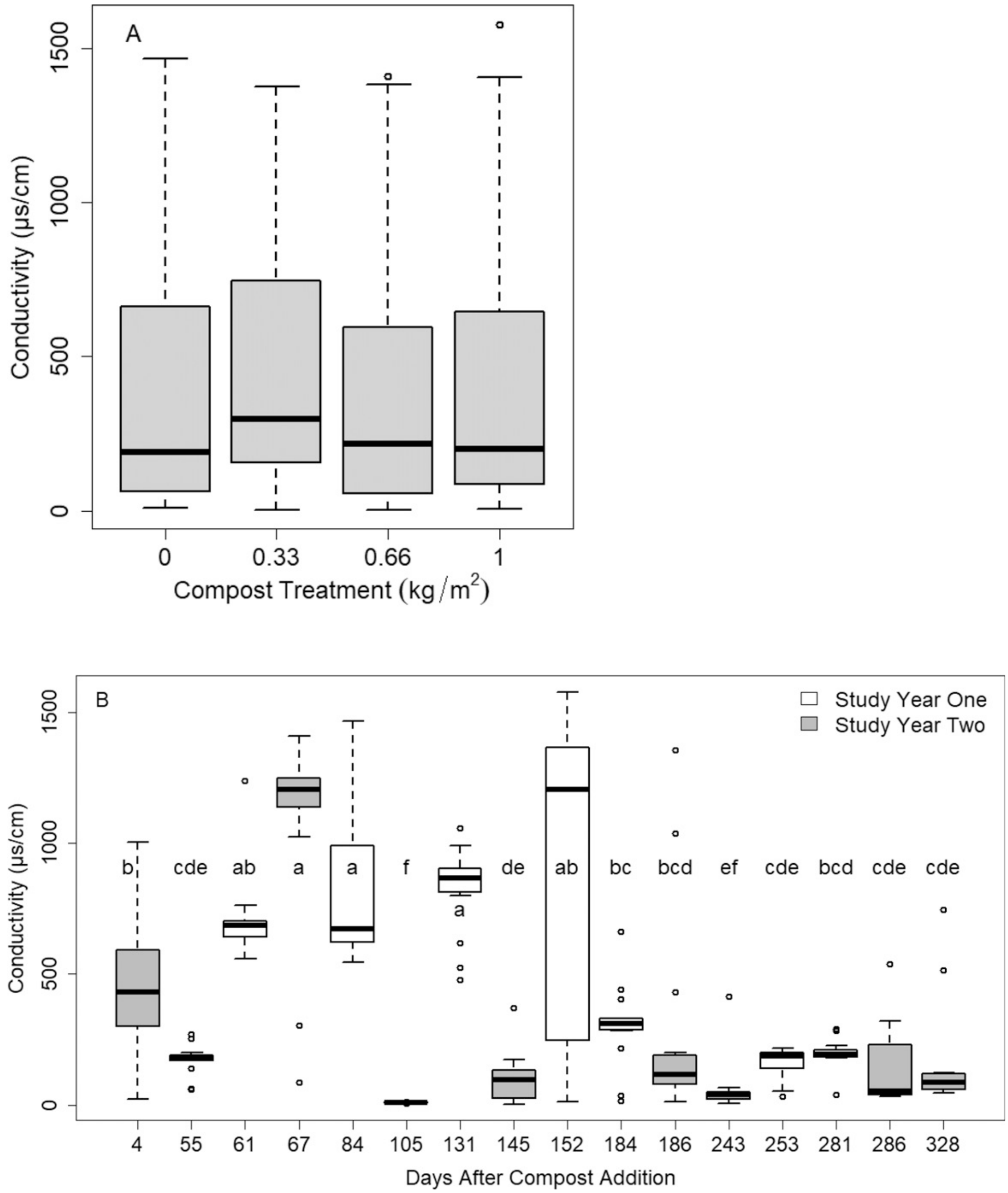

Fig. 3. The conductivity (µs/cm) of runoff water for each (**A**) compost treatment and (**B**) sampling time in days after compost addition for study years 1 and 2. Letters denote significant differences among sampling time means.

(Table 3, Fig. 5A). Turbidity of runoff water did not follow an obvious pattern with increasing time since the addition of compost (Fig. 5B). Two peaks were visible, the first at 84, 105, and 131 DAC addition (5 ± 0, 6 ± 1, 4 ± 0 NTU, respectively), and the second at 184 DAC addition (4 ± 0 NTU). There was a third, although shorter, peak at 253 and 281 DAC addition (2 ± 0 and 2 ± 0 NTU, respectively). Turbidity was higher during the first study year than during the second study year in 184 and 186 (1 ± 0 NTU) DAC addition and 281 and 286 (1 ± 0 NTU) DAC addition samplings. The variability of turbidity readings was highest at 105 and 184 DAC addition (Fig. 5B).

NO_3^-. Three outliers were removed from the analysis of the NO_3^- data (Table S1). Discussion of these outliers and the sway they had on graphing and interpreting results can be found in the Supplemental Materials (Fig. S1). The two-way interactions between compost treatment and DAC application and study year were not significant ($F = 0.226$; $P = 0.878$ and $F = 0.378$; $P = 0.769$, respectively). Compost treatment had no effect on the NO_3- content of runoff water ($F = 2.377$; $P = 0.114$) (Table 3, Fig. 6A). The NO_3^- concentrations were highest at 67 DAC addition (74.1 ± 7.0 mg/L), but not significantly higher than those at 84, 131, 152, and 184 DAC addition (43.7 ± 6.1, 48.0 ± 1.9, 48.5 ± 7.9, and 30.9 ± 2.3 mg/L, respectively) (Fig. 6B). The average NO_3^- concentration at 145 DAC addition (18.2 ± 16.9 mg/L) was significantly lower than that other sampling times during that part of the growing season. NO_3^- was low early during the growing season (4–61 DAC addition) and after the growing season (186–286 DAC addition); additionally, it was lowest at 186 DAC addition (1.5 ± 0.5 mg/L). The NO_3^- concentrations in runoff water were significantly higher during the first study year for the first DAC addition sampling pairs of 145 and 152 and 184 and 186, but they were significantly lower for the 61 (2.8 ± 0.5 mg/L) and 67 DAC addition pair (Fig. 6B).

NH_3. The two-way interactions between compost treatment and DAC application and

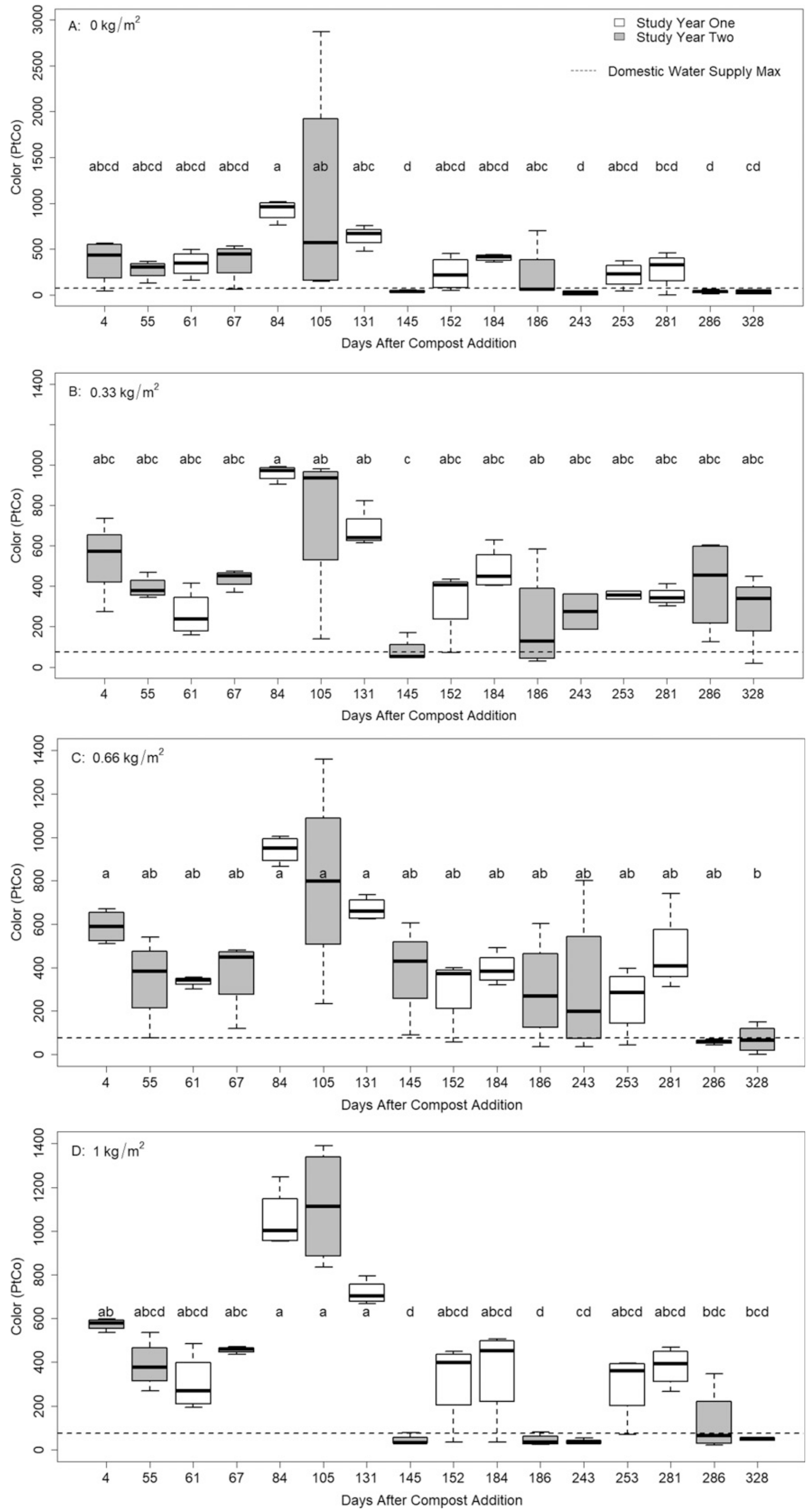

Fig. 4. The color (PtCo) of runoff water from green roof platforms for each sampling period of each study year by days after compost addition with compost treatments comprising (**A**) 0, (**B**) 0.33, (**C**) 0.66, and (**D**) 1 kg/m^2. Letters indicate a statistically significant difference among means within each compost treatment. The US Environmental Protection Agency domestic water supply maximum (USEPA 1986) threshold (75 PtCo) is included as a dashed line.

study year were not significant ($F = 0.579$; $P = 0.629$ and $F = 1.040$; $P = 0.376$, respectively). Compost treatment, number of DAC addition, and study year had no effect on the NH_3 content of runoff water ($F = 0.367$; $P = 0.778$, $F = 0.548$; $P = 0.460$, and $F = 0.013$; $P = 0.911$, respectively) (Table 3, Figs. 7 and 8). The range of observed NH_3 values was similar for all compost treatments (Fig. 7A) but higher during study year 2 than during study year 1 (Fig. 7B). The range of observed values was considerably higher at 105 and 243 DAC addition (Fig. 8A). The NH_3 levels during most other sampling times were lower than 1.5 mg/L, with little variation, except for that at 61 DAC addition (Fig. 8B).

Total phosphorus. The two-way interactions between compost treatment and DAC application and study year were not significant ($F = 1.732$; $P = 0.161$ and $F = 0.924$; $P = 0.430$, respectively). Compost treatment and number of DAC addition had no effect on the total phosphorus content of runoff water ($F = 0.862$; $P = 0.479$ and $F = 228.98$; $P = 0.188$, respectively) (Table 3, Fig. 9). The variability of total phosphorus was highest at 105 DAC addition and high again during the period of 243 to 286 DAC addition (Fig. 9C). These time periods also represent times of slightly elevated, although not significantly higher, phosphorus levels. Total phosphorus was significantly higher during study year 1 (4.23 ± 0.23 g/L) than during study year 2 (2.99 ± 0.27 mg/L) (Fig. 9B).

Potassium. The two-way interactions between compost treatment and DAC application and study year were not significant ($F = 0.273$ and $P = 0.845$ and $F = 0.363$ and $P = 0.779$, respectively). Compost treatment had no effect on the potassium content of runoff water ($F = 1.118$ and $P = 0.373$) (Table 3, Fig. 10A). Potassium in runoff water increased with time after compost addition until 105 DAC addition (256 ± 42 mg/L), after which time it decreased again (Fig. 10B). This peak approximately corresponds to the growing season. There was no difference between the earliest sampling times, 4 and 55 DAC addition, and sampling at 186 DAC addition and later during the study year. Variability was also higher for some of the sampling times during the growing season. Mean potassium in runoff was lower during study year 1 than during study year 2 for two date pairs, 61 and 67 DAC addition (44 ± 12 and 110 ± 16 mg/L, respectively) and 145 and 152 DAC addition (98 ± 40 and 130 ± 17 mg/L, respectively) (Fig. 10B).

Discussion

Compost treatment

Compost treatment had no effect on any water quality metric except color (Table 3). The general lack of effect of compost treatment was likely attributable to the relatively small quantities of compost used. The nutrient contributions of the compost treatments, even with the 1 kg/m^2 compost treatment, were relatively low compared with the nutrient contributions of the fertilizer used (Table 4). Compost treatments also had few differences when comparing values observed during this study to relevant water quality thresholds. Only two samples were lower than the USEPA freshwater minimum pH of 6.5 (USEPA 2022a), and eight samples exceeded the USEPA human

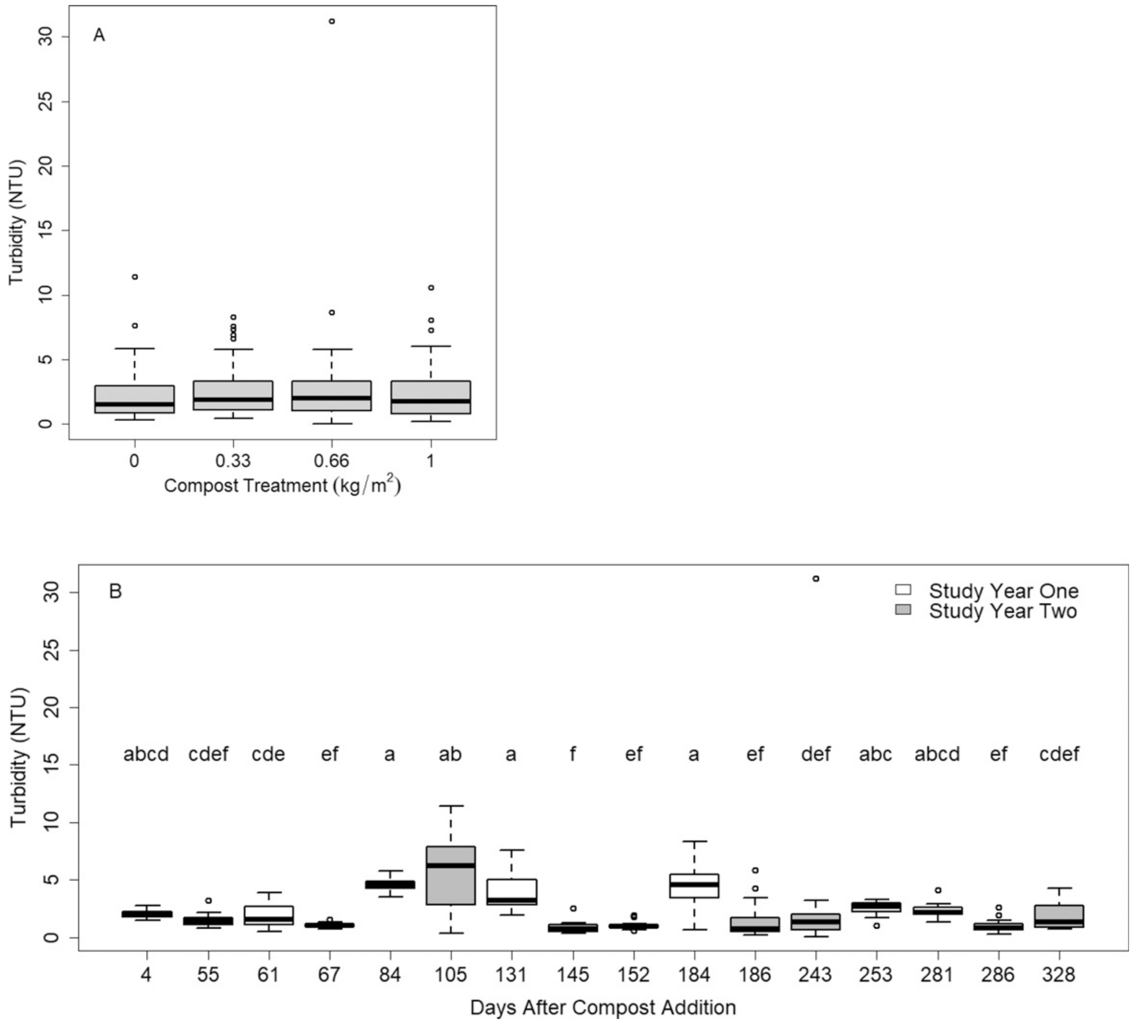

Fig. 5. The turbidity of runoff water for each (**A**) compost treatment and (**B**) sampling time in days after compost addition for study years 1 and 2. Letters denote significant differences among sampling time means.

health maximum pH of 9.0 (USEPA 2022b), but these were spread across all compost treatments (Fig. 2A). The effect on color was primarily changes in color over time that differed slightly for each compost treatment. However, at any specific sampling time, there was no significant difference among compost treatments (Fig. 4). There were, however, slight differences in the number of samples that exceeded the USEPA drinking water supply maximum of 75 PtCo (USEPA 1986) (χ^2 = 11.07; P = 0.011), with slightly more observations exceeding the limit in the 0.33 and 0.66 kg/m^2 compost treatments.

Runoff pH observed during this study tended to be higher than that observed at the Brooklyn Grange rooftop farm, where the maximum pH was 7.75, and closer to the pH of runoff from the ornamental green roof reported in another study (Whittinghill et al. 2016a). The results of this study are closer to the range of runoff pH reported in the literature for ornamental green roofs (Fig. 2, Table 5). The conductivity of runoff water observed during this study was also higher than that observed on both roofs from that study, which had maximums of 473.3 and 233 μS/cm, respectively (Whittinghill et al. 2016a). The range of runoff conductivity values of ornamental green roofs found in the literature is now much larger than that reported by Whittinghill et al. (2016a) and closer to that observed during this study, with some values considerably exceeding those observed during this study (Fig. 3, Table 5). For most of the sampling times, the range of runoff water color observed in all compost treatments for this study was similar to that observed at the Brooklyn Grange (152–813 PtCo) (Whittinghill et al. 2016a) and that of ornamental green roofs reported in the literature (Fig. 4, Table 5). The exceptions are the peaks in color at 84 and 105 DAC addition (Fig. 4), which were either higher than or extended beyond the range observed. The range of turbidity values observed during this study was also similar to that observed at the Brooklyn Grange (0.576–4.152 NTU) (Whittinghill et al. 2016a), with the exception of the same peaks at 84, 105, and 131 DAC addition and at 184 DAC addition (Fig. 5B). However, those peaks in turbidity were still relatively low compared with some ornamental green roof values reported in the literature, even after excluding values reported for bare media (Fig. 5, Table 5).

Similar to color, the 0.33 and 0.66 kg/m^2 compost treatments had slightly more observations that exceeded the National Pollutant Discharge Elimination System (NPSES) stormwater discharge maximum for nitrate concentrations (0.68 mg/L) (USEPA 2015) (χ^2 = 10.607; P = 0.014). Most observed values exceeded the USEPA drinking water maximum for NO_3^- (Fig. 6A) and phosphorus thresholds to prevent the development of biological nuisances and control eutrophication (0.05 mg/L) (USEPA 1986) and the stormwater discharge maximum for phosphorus (2 mg/L) (USEPA 2015) without differences in the proportion of observations that exceeded the thresholds among compost treatments (χ^2 = 1.987 and P = 0.575, χ^2 = 1.919 and P = 0.589, and χ^2 = 6.224 and P = 0.101, respectively). The NO_3^- concentrations in runoff from this study were much higher than some reported in the literature regarding agricultural green roofs, but they were also much lower than others. Some studies reported NO_3^- concentrations between approximately 0.3 and 16 mg/L, including

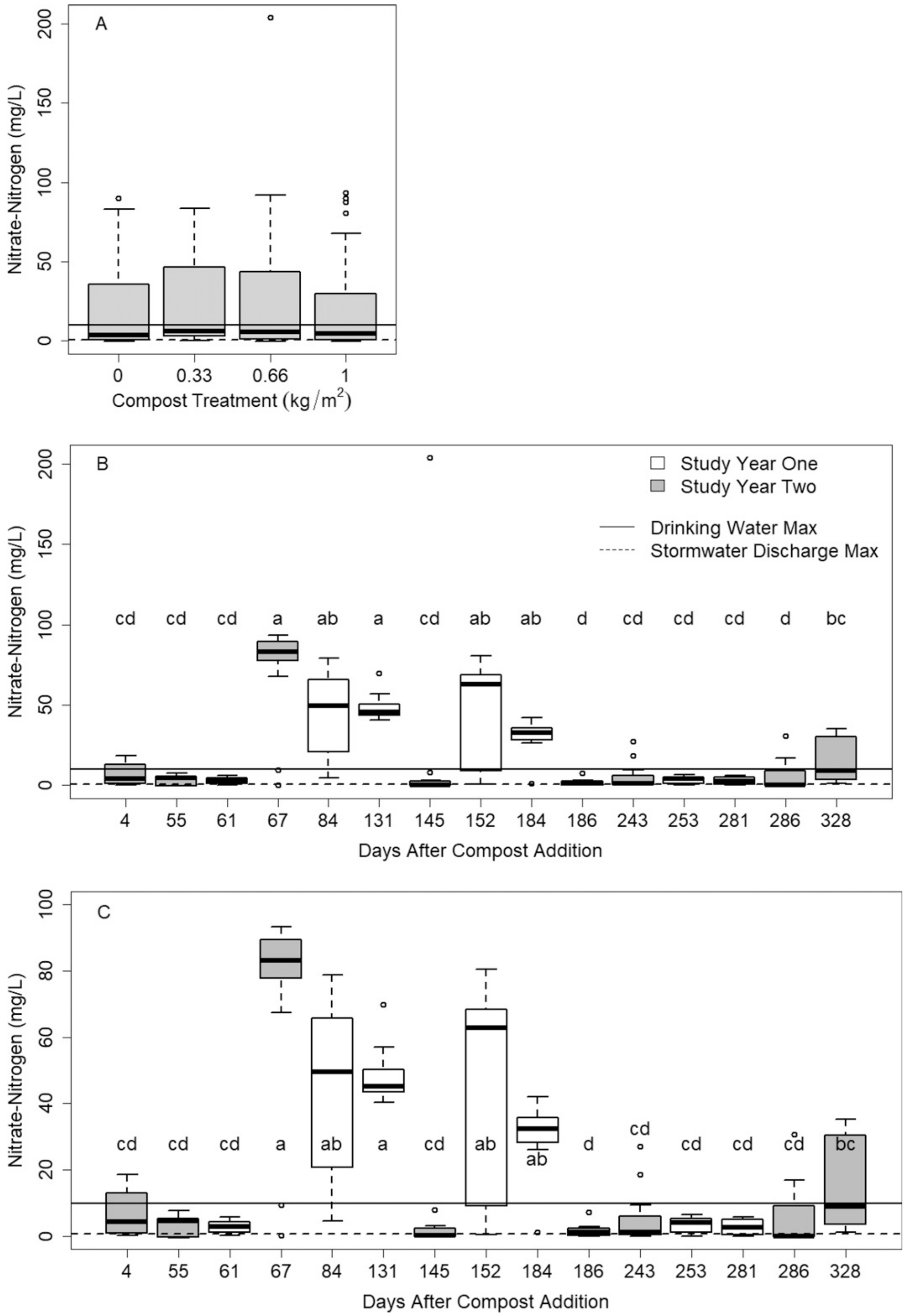

Fig. 6. Nitrate nitrogen concentrations (mg/L) of runoff water for each (**A**) compost treatment, (**B**) sampling time in days after compost addition for study years 1 and 2 for the full range of observations, and (**C**) sampling time in days after compost addition for study years 1 and 2 for observations up to 150 mg/L. Letters denote significant differences among sampling time means. US Environmental Protection Agency Drinking Water maximum (10 mg/L) (USEPA 2022b) and national pollutant discharge elimination system (NPSES) stormwater discharge maximum (0.68 mg/L) (USEPA 2015) thresholds are included as solid and dashed lines, respectively.

observations from two operational rooftop farms and one controlled experiment using green roof platforms (Harada et al. 2017; Whittinghill et al. 2015, 2016a). Other controlled experiments using a variety of media have reported NO_3^- concentrations as high as 130 to 1207.7 mg/L (Harada et al. 2020; Kong et al. 2015; Matlock and Rowe 2017), with some indication that lower levels can be observed from higher plants using nitrogen, such as herbs (Matlock and Rowe 2017). When compared with NO_3^- concentrations of runoff water reported in the literature regarding ornamental green roofs, those observed during this study were similar (Fig. 6, Table 5). Only 11 observations were greater than that range, and they were within the range if extended to include the study that used media with 15% compost (Ntoulas et al. 2015).

Most of the observed ammonia concentrations were below the USEPA acute (17 mg/L)

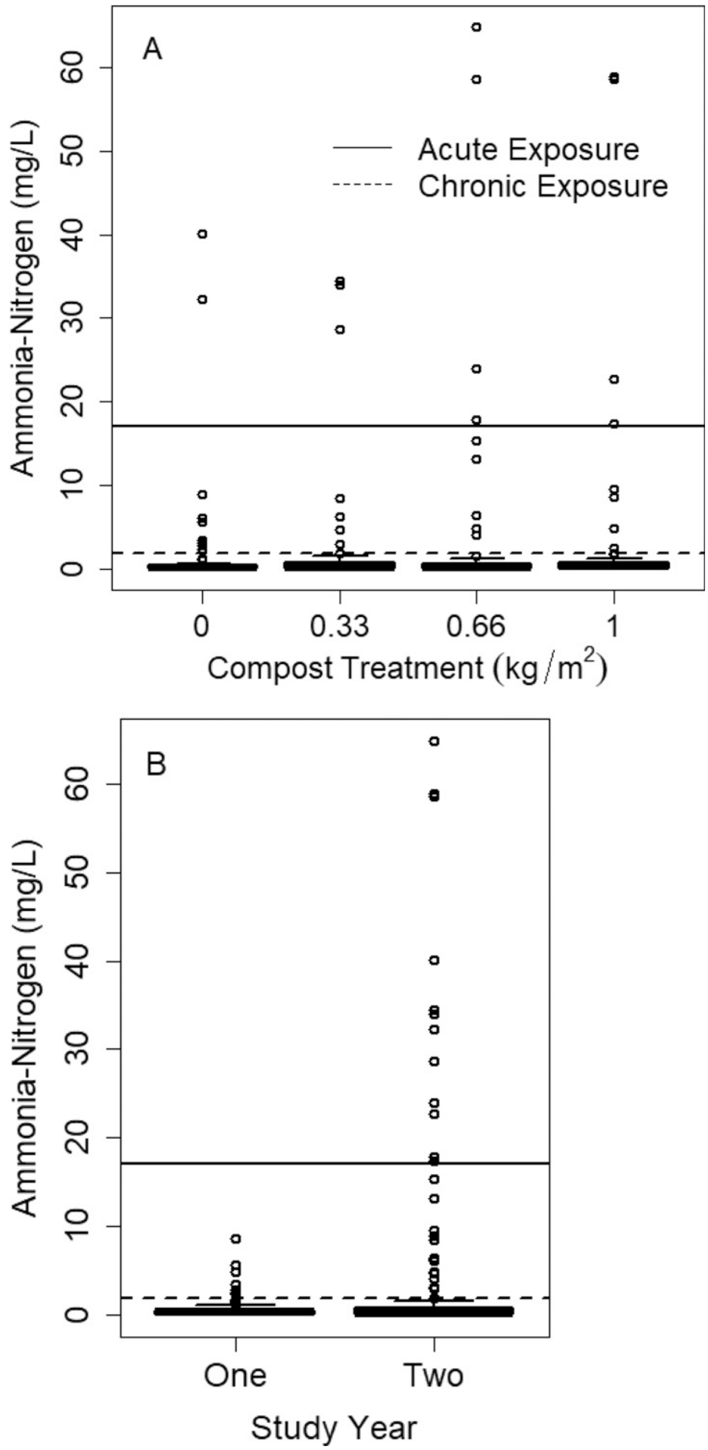

Fig. 7. Ammonia nitrogen concentrations (mg/L) of runoff water for each (**A**) compost treatment and (**B**) study year. US Environmental Protection Agency acute (17 mg/L) and chronic (1.9 mg/L) exposure thresholds (USEPA 2013) are included as solid and dashed lines, respectively.

and chronic (1.9 mg/L) exposure thresholds (USEPA 2013) (Fig. 8), and there was no difference among compost treatments for the proportion of observations that exceeded those thresholds ($\chi^2 = 0.744$; $P = 0.863$ and $\chi^2 = 0.056$; $P = 0.997$, respectively). The bulk of NH_3 concentrations observed during this study were also similar to those observed in the agricultural green roof literature, which were mostly between 0 and approximately 18 mg/L (Harada et al. 2020; Kong et al. 2015; Whittinghill et al. 2016a). Although there were several observations during this study between 22 and 65 mg/L, there was also a high concentration noted during another study that examined at different growing media, including a coconut coir-based potting mix with a biochar amendment, which had an observed NH_3 runoff concentration of 58.8 mg/L (Harada et al. 2020). The range of NH_3 concentrations of ornamental green roofs reported in the literature was closer to the lower range of agricultural green roofs (Table 5); therefore, most of the values observed during this study were also within that range (Figs. 7 and 8).

Runoff phosphorus concentrations reported in the agricultural green roof literature are between 0.14 and approximately 6 mg/L (Aloisio et al. 2016; Matlock and Rowe 2017; Whittinghill et al. 2015, 2016b). This range is similar to most of the observations during this study; however, a few were higher and exceeded 10 mg/L. Interestingly, the range of runoff phosphorus concentrations observed during this study (Fig. 9) and reported in the agricultural green roof literature are both on the low side compared with the range reported for ornamental green roofs (Table 5). There are several possible explanations for this. Either the vegetable crops use more phosphorus than the ornamental crops, as suggested by Matlock and Rowe (2017) regarding the differences in nitrogen leaching, or there were other differences in management, media composition, and climate that cause higher phosphorus concentrations in the runoff from ornamental green roofs. This highlights the need for a greater understanding of the phosphorus dynamics of green roofs so that phosphorus leaching from agricultural and ornamental green roofs can be minimized. This may become especially important as the number of agricultural and ornamental green roofs continues to grow, and leaching phosphorus could start to contribute more significantly to eutrophication in water bodies that receive stormwater runoff.

A single study that monitored an operational rooftop farm reported potassium concentrations in runoff with a range of 0.256 to 89.193 mg/L (Whittinghill et al. 2016a). Approximately 23% of the observed potassium concentrations from this study exceeded that range during the growing season across all compost treatments (Fig. 10). The range of values reported by Whittinghill et al. (2016a) was similar to that reported in the literature regarding ornamental green roofs (Table 5); 23% of the values observed during this study also exceeded that range. Relatively few studies of ornamental and agricultural green roofs reported potassium leaching when compared with the number of studies that reported nitrogen and phosphorus nutrients. Although potassium is not considered a water quality issue in the same way that nitrogen and phosphorus are because of their role in eutrophication, it represents a loss of important nutrients to crop plants on agricultural green roofs. These are nutrients that rooftop farmers pay for as inputs; therefore, they potentially spend much effort getting them to the rooftop, thus representing an economic loss.

Sample timing

Fertilizer applications. Although some fertilizer applications were performed in close proximity to runoff water sampling, they did

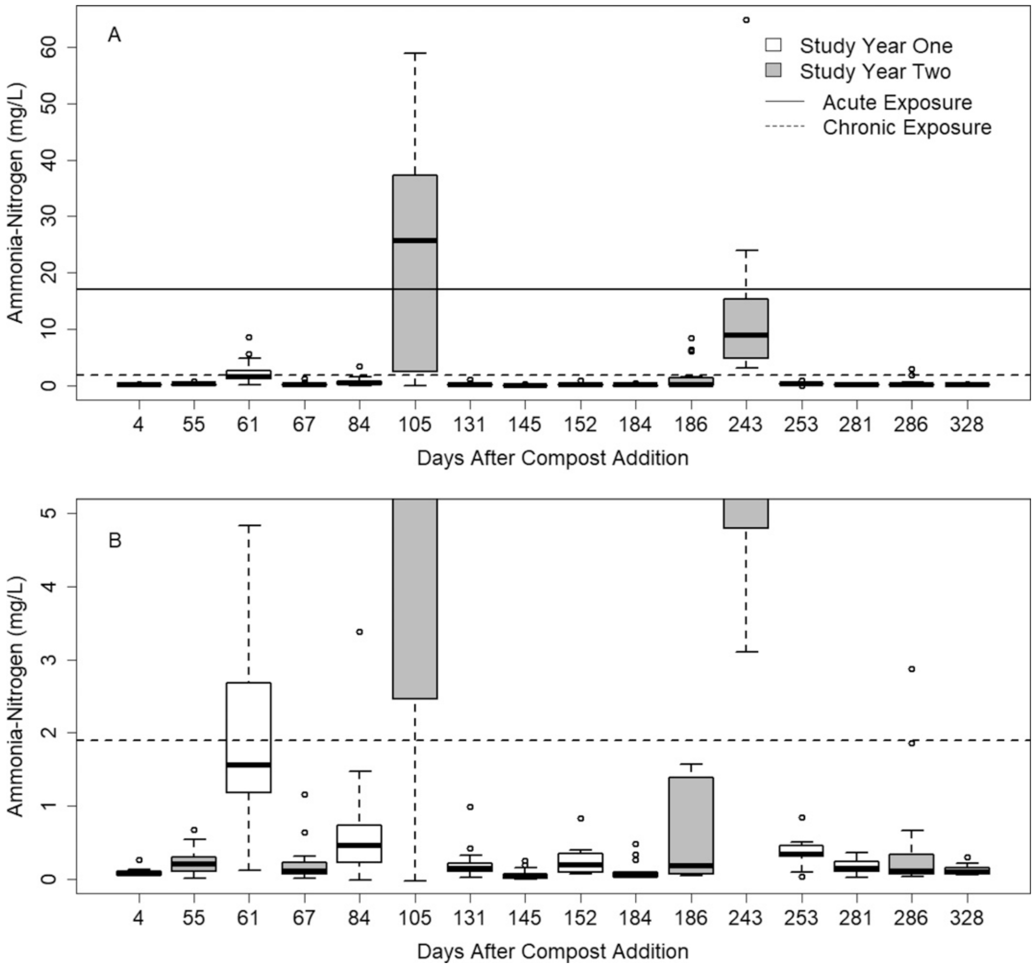

Fig. 8. Ammonia nitrogen concentrations (mg/L) of runoff water for each (**A**) sampling time in days after compost addition for study years 1 and 2 for the whole observed range and (**B**) sampling time in days after compost addition for study years 1 and 2 up to 5 mg/L. US Environmental Protection Agency acute (17 mg/L) and chronic (1.9 mg/L) exposure thresholds (USEPA 2013) are included as solid and dashed lines, respectively.

not appear to have an impact on runoff water quality beyond slightly higher values of pH, conductivity, NO_3^-, and potassium during the growing season and some higher variability of those metrics during the growing season (Figs. 2B, 3B, 6B, 6C, and 10B, respectively). Four sampling dates were close to fertilizer applications, 55, 61, 84, and 105 DAC addition, which were 1, 5, 2, and 6 to 7 d after fertilizer applications, respectively (Tables 1 and 2). However, these sampling dates are not represented by significant increases or decreases in water quality metrics when compared with those at 131, 145, 184, and 186 DAC addition, which were during the growing season but 21 to 27 d after fertilizer applications (Tables 1 and 2). In all cases, there were either no significant differences between the former and latter groups of dates or no pattern of significant differences. For all four metrics, the former group of dates, those close to fertilizer applications, also encompass the highest and lowest metrics. For the metrics with a less obvious pattern associated with the growing season, turbidity, color, NH_3, and total phosphorus (Figs. 4, 5B, 8, and 9C, respectively), both groups of dates had means that were not significantly different from the highest and lowest means.

Weather conditions. The amount of rainfall between sampling events can have an impact on runoff water quality (Hoskins et al. 2014; Teemusk and Mander 2007). During this study, there were several sampling dates with high (>200 mm) precipitation and several with low (≤100 mm) precipitation (Fig. 1B). These were at 55, 61, 131, 184, 243, and 253 DAC addition and at 4, 47, 152, and 184 DAC addition, respectively. Among the water quality metrics measured, pH, conductivity, and potassium were the only ones that showed slight differentiation between sampling dates with high or low precipitation leading up to sampling. For these metrics, sampling dates with high precipitation were among both the highest and lowest observations and, in many cases, were not significantly different from sampling dates with low precipitation before sampling (Figs. 2B, 3B, and 10B). These findings are contrary to the findings of previous studies. Teemusk and Mander (2007), for example, noted higher pH when rainfall was low.

Study year

The few differences between study years suggest that successive additions of compost may not have an additive effect on runoff water quality. In some instances, study year 1 had significantly higher effects; in others, study year 2 had higher effects. For the four pairs of sampling dates that were compared between study year 1 and study year 2, the 281 and 286 sampling dates only had significantly different turbidity; in which case, study year 1 had higher turbidity. For those with multiple significant differences across water quality metrics, study year 2 had consistently higher differences according to the 61 (year 1) and 67 (year 2) comparison [nitrate (Fig. 6B and C) and potassium (Fig. 10B)], and study year 1 had higher differences according to the 145 (year 2) and 152 (year 1) comparison [conductivity (Fig. 3B), NO_3^- (Fig. 6B and C), and potassium (Fig. 10B)]. Although this seems inconsistent, the year with the higher value was the year with less rainfall before the sampling date (67 DAC addition and 152 DAC addition) (Fig. 1B). For the comparison of 184 (year 1) and 186 (year 2) DAC addition sampling dates, higher rainfall before sampling appeared to have increased turbidity and NO_3^- but decreased pH (Figs. 5B, 6B, 6C, and 2B, respectively). Regarding color, only the main effect of the study year was significant; however, study year 1, which was the year with more precipitation, had higher color than study year 2. Differences between study years

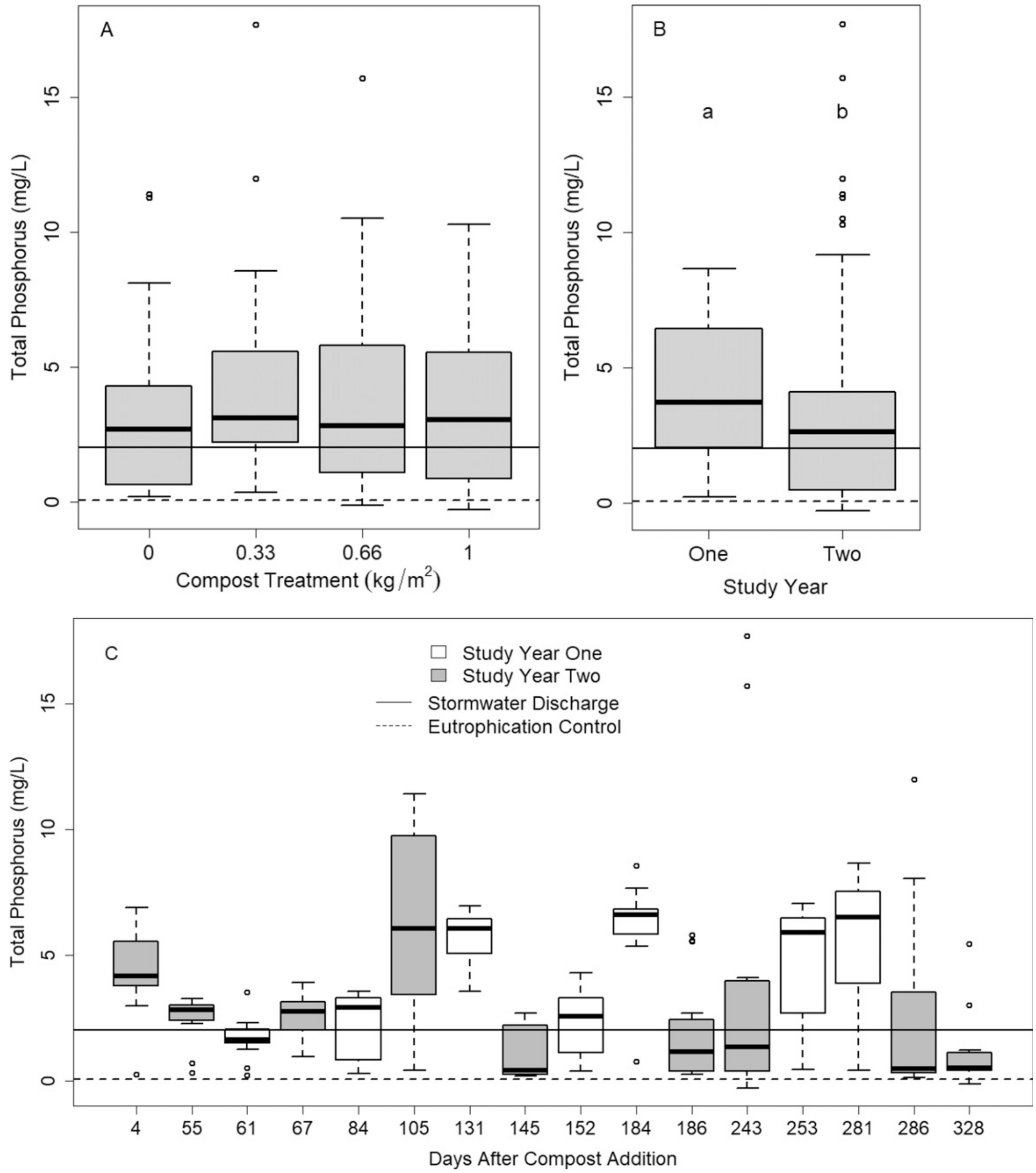

Fig. 9. Total phosphorus concentrations (mg/L) of runoff water for each (**A**) compost treatment and (**B**) study year, and (**C**) sampling time in days after compost addition for study years 1 and 2. US Environmental Protection Agency stormwater discharge maximum (2 mg/L) (USEPA 2015) and threshold to prevent the development of biological nuisances and to control eutrophication (0.05 g/L) (USEPA 1986) are included as solid and dashed lines, respectively.

appeared to be related more to weather patterns than successive additions of compost.

Although the results of this study show clear seasonal patterns associated with runoff water quality from a vegetable-producing green roof, 2 years of data collection may not be enough to fully clarify the effect of annual compost additions over many years. Disruptions in laboratory work attributable to the pandemic caused by severe acute respiratory syndrome coronavirus 2 (SARS-CoV-2) and coronavirus disease 2019 (COVID-19) occurred; therefore, water quality analyses could not be performed during study years 3 (Spring 2020–Spring 2021) and 4 (Spring 2021–Spring 2022). Most other studies performed on agricultural green roofs have collected runoff from experiments within a period as short as 5 weeks (Harada et al. 2020), but most have only covered a single growing season or calendar year (Aloisio et al. 2016; Matlock and Rowe 2017; Whittinghill et al. 2015, 2016a). One project did involve monitoring for a longer period, 2.5 years, but it focused more on annual mass loading than the dynamics of runoff water quality over the course of a growing season or the study period (Harada et al. 2017, 2018). More long-term monitoring that includes runoff water quality from both full-scale rooftop farms that use the practice of annual compost addition and long-term controlled experiments is necessary and would help to clarify the impact of repeated compost additions. Finer-scale data collection could also help determine how changes in water quality relate to specific management events like compost or fertilizer applications and storm events of different sizes.

Conclusions

Despite the addition of compost to green roofs for agricultural production being a popular practice intended to improve production, this practice does not appear to positively impact runoff water quality when compost is added in small quantities compared with the use of fertilizers alone. The lack of differences

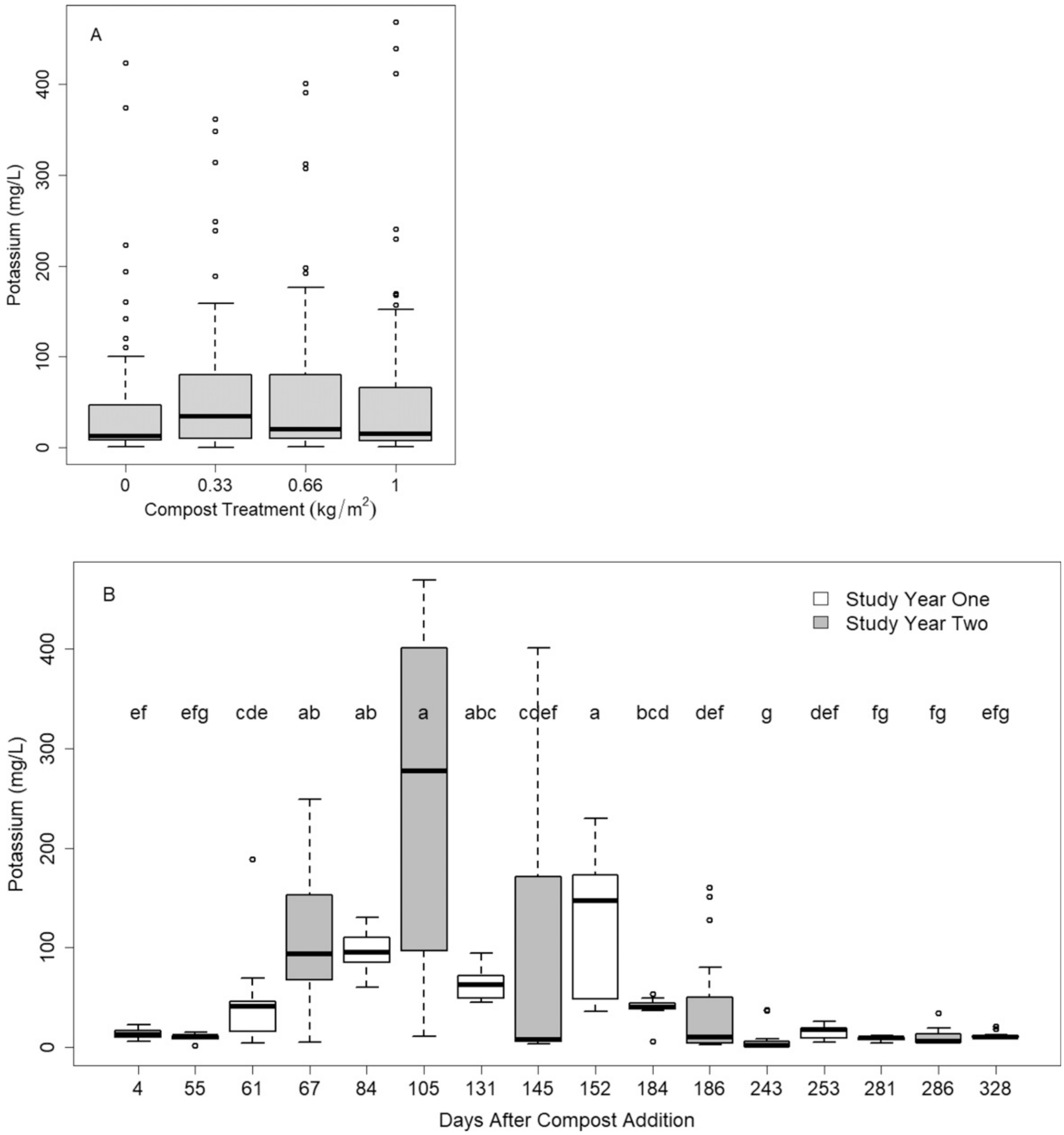

Fig. 10. Potassium concentrations (mg/L) of runoff water for each (**A**) compost treatment and (**B**) sampling time in days after compost addition for study years 1 and 2. Letters denote significant differences among sampling time means.

observed during this study may be attributable, in part, to the low nutrient contributions of the compost used. More extensive studies of this practice with additional types of compost and greater compost additions reflective of the practices used by more green roof farmers will help determine the long-term effects of compost addition. Although there were few differences in the two years when this study was conducted, data were not collected during a sufficiently long timeframe to rule out a compounding effect of annual compost additions on runoff water quality or alterations to how runoff water quality changes with roof aging when compared with ornamental green roofs.

Table 4. Absolute and relative nutrient contributions of compost and fertilizers used in each compost treatment.

		Compost		Fertilizer	
Treatment	Nutrient	Quantity (g/m^2)	Percent	Quantity (g/m^2)	Percent
0 kg/m^2	N	0	0	19.61	100
	P_2O_5	0	0	16	100
	K_2O	0	0	16	100
0.33 kg/m^2	N	0.18 (at most)	0.9	19.42	99.1
	P_2O_5	1.01	6.3	14.99	93.7
	K_2O	0.47	2.9	15.53	97.1
0.66 kg/m^2	N	0.36 (at most)	1.8	19.25	98.2
	P_2O_5	2.02	12.6	13.98	87.4
	K_2O	0.95	5.9	15.05	94.1
1 kg/m^2	N	0.55 (at most)	2.8	19.06	97.2
	P_2O_5	3.07	19.2	12.93	80.8
	K_2O	1.43	8.9	14.57	91.1

Table 5. Water quality metrics and their values reported in the green roof literature.

Water quality metric	Range	References
pH	4.5–8.75	Beecham and Razzaghmanesh 2015; Berghage et al. 2009; Bliss et al. 2009; Buccola et al. 2008; Buffam et al. 2016; Castro et al. 2020; Clark and Zheng 2014; Czemiel Berndtsson et al. 2009; De Cuyper et al. 2005; Gnecco et al. 2013; Mitchell et al. 2017; Razzaghmanesh et al. 2014; Teemusk and Mander 2007; Teemusk and Mander 2011; Van Seters et al. 2009; Vijayaraghavan and Badavane 2017; Vijayaraghavan et al. 2012; Vijayaraghavan and Joshi 2015; Woods 2009; Zhang et al. 2015
Conductivity (uS/cm)	83.82–1200 (up to 8000*)	Berghage et al. 2009; Buffam et al. 2016; *Clark and Zheng 2014; De Cuyper et al. 2005; Gnecco et al. 2013; Van Seters et al. 2009; Vijayaraghavan and Badavane 2017; Vijayaraghavan et al. 2012; Vijayaraghavan and Joshi 2015
Apparent color (PtCo)	219.04–878.19	Berghage et al. 2009; De Cuyper et al. 2005
Turbidity (NTU)	3.6–310 (to 568 nonvegetated)	Beecham and Razzaghmanesh 2015; Berghage et al. 2009; Bliss et al. 2009; Castro et al. 2020; Clark and Zheng 2014; Morgan et al. 2011; Van Seters et al. 2009
Nitrate (mg/L)	0.005–80 (160 with 15% compost in media)	Beecham and Razzaghmanesh 2015; Berghage et al. 2009; Buccola et al. 2008; Buffam et al. 2016; Carpenter and Kaluvakolanu 2011; Castro et al. 2020; Czemiel Berndtsson et al. 2008; Emilsson et al. 2007; Gong et al. 2019; Gregoire and Clausen 2011; King and Torbert 2007; Johnson 2014; Ntoulas et al. 2015; Razzaghmanesh et al. 2014; Seidl et al. 2013; Teemusk and Mander 2007; Toland et al. 2012; Van Seters et al. 2009; Vijayaraghavan et al. 2012; Whittinghill et al. 2015; Woods 2009; Zhang et al. 2015
Ammonium (mg/L)	0–20	Beecham and Razzaghmanesh 2015; Buffam et al. 2016; Castro et al. 2020; Czemiel Berndtsson et al. 2006, 2008, 2009; Emilsson et al. 2007; Gong et al. 2019; Gregoire and Clausen 2011; Johnson 2014; Teemusk and Mander 2007; Toland et al. 2012; Van Seters et al. 2009; Zhang et al. 2015
Total phosphorus (mg/L)	0–40	Beck et al. 2011; Berghage et al. 2009; Bliss et al. 2009; Buccola et al. 2008; Clark and Zheng 2014; Castro et al. 2020; Czemiel Berndtsson et al. 2006, 2008, 2009; De Cuyper et al. 2005; Emilsson et al. 2007; Gong et al. 2019; Gregoire and Clausen 2011; Harper et al. 2015; Hathaway et al. 2008; Kouppamäki et al. 2016; Liu et al. 2019; Malcolm et al. 2014; Monterusso et al. 2004; Teemusk and Mander 2007; Teemusk and Mander 2011; Toland et al. 2012; Van Seters et al. 2009; Whittinghill et al. 2015; Zhang et al. 2015
Potassium (mg/L)	0.2–85	Beecham and Razzaghmanesh 2015; Berghage et al. 2009; Buffam et al. 2016; Czemiel Berndtsson et al. 2006, 2008, 2009; Emilsson et al. 2007; Gnecco et al. 2013; Razzaghmanesh et al. 2014; Van Seters et al. 2009; Vijayaraghavan and Badavane 2017; Vijayaraghavan et al. 2012; Vijayaraghavan and Joshi 2015; Zhang et al. 2015

However, this study does support the findings of the few other published studies of runoff water quality from agricultural green roof systems. This strengthens the overall conclusion that although there is greater nutrient runoff than that from some ornamental roof types, the nutrient runoff observed during this study was within the range observed across the literature regarding ornamental green roofs.

The results of this study also highlight some areas in which nutrient runoff is concerning and the impact of the agricultural growing season on nutrient losses. The NO_3^- and total phosphorus concentrations in runoff during the growing season were often much higher than the water quality thresholds designed to help prevent eutrophication. These observed high concentrations do not appear to correlate directly with compost or fertilizer applications, but they were high throughout the growing seasons observed. This suggests that further work can be performed to examine how to reduce this runoff through different nutrient application practices and areas of agricultural green roofs that have not yet been explored. High levels of other nutrients in runoff during the growing season, such as potassium, support this idea of nutrient use inefficiency in the green roof system. How this might impact the productivity and long-term economic sustainability of this system is unknown. The clear seasonality associated with the growing season and agricultural activity observed during this study highlight the need for finer-scale data collection for this system when examining its impact on the ecosystem services typically ascribed to ornamental green roofs.

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