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# Urban forests near municipal solid waste incinerators do not show elevated trace metal or rare earth element concentrations across three cities in the northeast USA

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## Abstract

With the World Bank estimating global municipal solid waste MSW to increase to 3.4 billion tonnes by 2050, municipal solid waste incineration (MSWI) needs to be considered as part of sustainable waste management. In spite of major advancements in air pollution control systems and emission regulations, emissions of trace metals (TMs) (As, Cd, Cu, Ga, Mn, Ni, Pb, V, W, Zn) and rare earth elements (REEs) (La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Yb) may occur. Here, leaves, wood, forest floor, and A horizons in urban forests were assessed to determine if TM and REE concentrations show enrichment near MSWI relative to other urban areas for three cities in the northeastern USA. Vegetation and soil concentrations measured were generally comparable to concentrations observed in previous studies in non-urban sites. Overall, higher TM and REE in forest floor, A horizons, leaves, and wood were not observed consistently for MSWI sites when compared to city sites. Although pooled forest floor Ni and V and A horizon La and Nd concentrations were significantly different, they were not consistent but driven by a large difference at one of the three cities. This indirect study suggests air pollution control systems have prevented or have reduced TM and REE emissions to local urban forests to negligible amounts. Further studies are needed to evaluate potential interferences from other urban air pollution sources, and conduct direct emission measurements to validate MSWIs are not emitting TMs and REEs.

**Keywords** Urban soils · Industrial land use · Forest soils · Toxic metals · Heavy metal pollution

## Introduction

Municipal solid waste incinerators (MSWIs) are important for sustainable waste management and energy co-generation globally. The World Bank estimates global municipal solid waste (MSW) to be 2.0 billion tonnes per year in 2017 and expects to increase to 3.4 billion tonnes by 2050 (Kaza et al. 2018). Across the world, most MSW (37%) end up in landfills (Kaza et al. 2018) but an increasing proportion is incinerated.

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Waste incineration reduces volume required for landfilling by > 90%, bottom ash can be used as a construction material, heat from combustion can be used as an energy source for steam and electricity, and incinerators can be located closer to residential areas than landfills due to minimal noxious odors (Cheng and Hu 2010). Combustion of MSW is widely practiced; Japan, Denmark, Netherlands, Norway, and Sweden combusted > 50% of their MSW in 2013 while Switzerland, France, UK, Germany, and South Korean combusting ≥ 25% of their MSW in 2015 (OECD 2015). The United States Environmental Protection Agency (USEPA) estimates that only 12.7% of the 243 million tonnes of MSW produced domestically was incinerated in 2017 (USEPA 2019). China has substantially increased MSWI from 3% in 2004 to 30% in 2014 (Mian et al. 2017) and aims to further increase the total volume and proportion of MSW incinerated over the next 5 years.

One issue with waste to energy MSWI facilities is its potential environmental impacts (whether actualized, such as Loppi et al. 2000, or perceived, see Budris 2019) for

atmospheric pollution. Globally, MSWI facilities have improved from unregulated combustion with few pollution management practices (NRC 2000) emitting large quantities of metals to local environments (e.g., Walsh et al. 2001) to a cleaner practice through implemented air pollution control (APC) systems and requirements of MSWI facilities to meet the USEPA's Maximum Achievable Control Technology standards (US 40 Code of Federal Regulation §Part 63). A large portion of macro and trace metals (TMs) within the waste flow entering the MSWI are retained in the bottom ash, such as Si, Al, Fe, Ba, Cu, Co, Ga, Ni, W, and V (e.g., Morf et al. 2013; Funari et al. 2015). The bottom ash can be a useful resource for recovery of TMs (Allegrini et al. 2014; Funari et al. 2015). Although rare earth elements (REEs) can be enriched in the bottom ash compared with the fly ash, their concentrations are far below REE ore concentrations and REE containing materials should be removed prior to incineration (McGill 2012; Allegrini et al. 2014; Binnemans et al. 2015). Other macro and TMs are enriched in the fly ash, such as Ca, K, Na, Ag, As, Bi, Cd, Cr, Cs, Pb, Sb, Sn, and Zn (Funari et al. 2016). APC systems are largely effective in collection of TMs in flue gas and fly ash, which are trapped then disposed of in specialized hazardous waste landfills (Cheng and Hu 2010; Funari et al. 2016).

Despite regulations and advancements in APC effectiveness commonly > 99.99% in fly ash and flue gas capture (e.g., Takaoka et al. 2016; Ashworth et al. 2016), waste to energy MSWI facilities continues to be a perceived and realized source of atmospheric pollution globally. Due to costs associated with acquiring and maintaining APC systems (Mian et al. 2017) and operational effects from combustion conditions (i.e., temperature, MSW composition, chlorine abundance, moisture content) that can cause increased evaporation, emission of TMs can still occur (Belevi and Moench 2000; Hu et al. 2003; Zhang et al. 2008). Most of the literature on the release of metals have focused on macro metals and TMs and have not quantified REE release from MSWI facilities. This is potentially due to the very low REE concentrations found in fly ash because of their preference for bottom ash retention (Allegrini et al. 2014) and that REEs are rarely at toxic concentrations in terrestrial environments not directly associated with mining (e.g., Migaszewski et al. 2016). Thus, REEs may be a novel tool to observe MSWI pollution as they are unlikely to be sourced from other anthropogenic pollution such as Pb, Cu, and Zn and have a low natural abundance.

Assessment of TM and REE concentrations in plants and soils surrounding a suspected point source emitter, such as a MSWI, is a well-established and passive approach for monitoring pollutant emissions (Loppi et al. 2000; NRC 2000; Steinnes and Friedland 2006). The surface soils and above-ground vegetation of urban forests can serve for monitoring pollution, as leaves and soils can accumulate particulate and

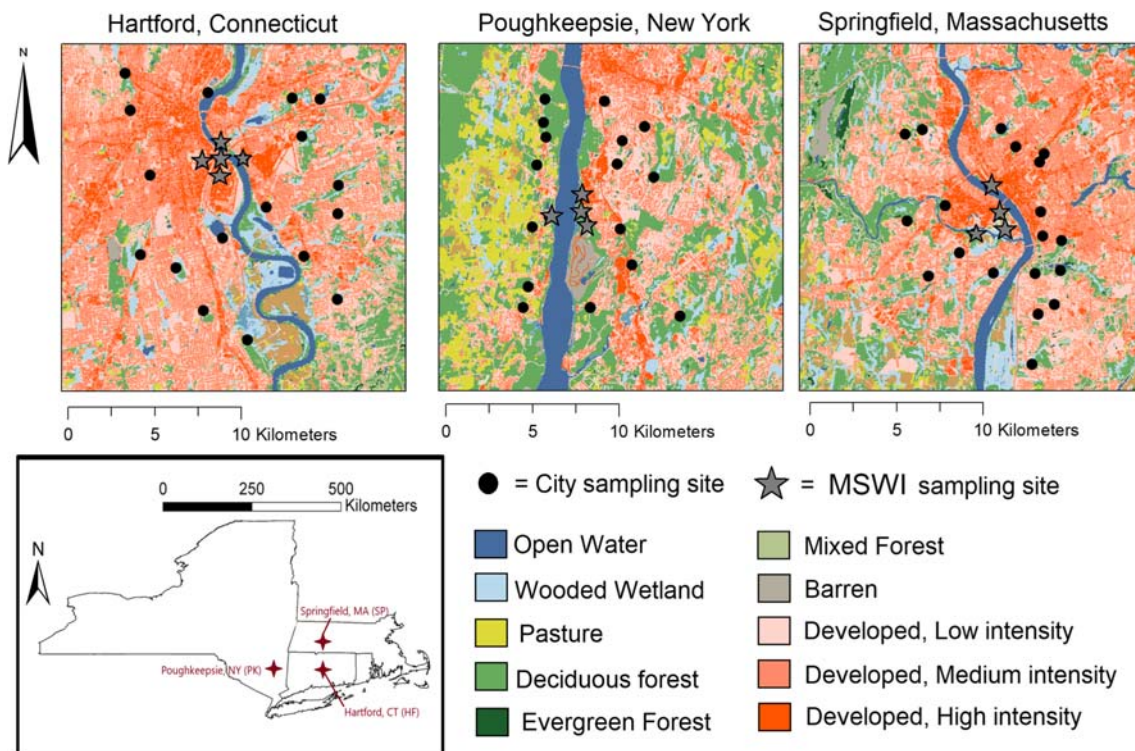
gaseous metal emissions from local point sources (Beckett et al. 1998; NRC 2000; Pickett et al. 2011; Livesley et al. 2016). Urban forests are patchworks of woody plants, with minimal management, existing between commercial lots, riparian zones, and residential corridors, either intentionally or unintentionally reforested (Livesley et al. 2016). However, the accumulations of TMs and REEs by urban forests are also integrated signals from municipal sources (automobiles, weathering building materials), industrial (refining and smelting), and commercial activities (manufacturing). Thus, establishing a baseline of urban city pollution and sampling near the point source (typically with < 2 km of a point source, e.g., Loppi et al. 2000; Hu et al. 2003; Zhan et al. 2014, Derome and Lindroos 1998) is important for characterizing enrichment specifically from a MSWI.

In New England, the Conservation Law Foundation argues that local waste incinerators pose unknown health hazards to local communities and due to their age, need to be removed as costs associated with updates are too large (Budris 2019). However, no independent study has sought to assess the impact of MSWI facilities on their potential or perceived emissions. The objective of this study was to determine if vegetation and soils of urban forests in close proximity to MSWIs have elevated TM and REE concentrations compared to surrounding urban areas in three cities. Few studies have been conducted to evaluate the current state of pollution from MSWIs in the USA, and exploring REEs can be used for identifying MSWI pollution. First, it was hypothesized that if MSWIs are an important source for TMs and REEs, surface soils and vegetation (leaves and bole wood) in close proximity (0.1 to 1.4 km) would have elevated concentrations compared to other developed areas (> 2 km away) in the same city. Second, it was hypothesized that soil properties (specifically soil organic matter content, pH, or clay) or vegetation properties (genera specific physiology) may control trace metal and rare earth element accumulation over proximity to MSWI facilities. Exploring the contribution of MSWI to urban forests is essential for our overall understanding and valuation of urban ecosystems, and protecting human and wildlife health.

## Methods

### Selection of sites

Three cities in the northeastern USA were studied: Hartford, Connecticut; Poughkeepsie, New York; and Springfield, Massachusetts. These cities were chosen because of their comparable climates, similar populations, and distance of < 200 km of each other (Fig. 1). Each city has a MSWI in or near it. The facility in Hartford, CT commenced operation in 1988 and combusts up to 2580 t (2850 US tonnes) per day, can generate up to 90 MW of electricity, and is equipped with



**Fig. 1** Location of municipal solid waste incinerator (MSWI) sites in close proximity to a waste incinerator, and city sampling sites mapped with land-use data from NLCD (2016) for the three cities

and APC that meet the USEPA Maximum Available Control Technology (MACT) emissions standards (<https://www.naes.com/locations/mid-connecticut/> Accessed March 1, 2020). The facility in Poughkeepsie, NY commenced operation in 1989 and combusts up to 410 t (450 US tonnes) per day, can generate up to 9.2 MW of electricity, and is equipped with and APC that meet the USEPA Maximum Available Control Technology (MACT) emissions standards (<http://www.dutchessny.gov/test101/17459.htm>, Accessed March 1, 2020). The facility in Springfield, MA commenced operation in 1988 and combusts 370 t (408 US tonnes) per day, can generate up to 9.4 MW of electricity, and is equipped with activated carbon and dry lime injection with fabric filter baghouse as its APC (<https://www.covanta.com/Our-Facilities/Covanta-Springfield>, Accessed March 1, 2020).

All three cities are on the border between Dfa and Dfb Köppen climate classifications, with humid continental mild to hot summers, wet all year. Urban forest sites were sampled during July and August of 2017, 2018, and 2019. To select urban forest sites, orthorectified satellite imagery from 2016 was used to find forested areas. Sites were ground inspected on foot to ensure forests were unmanaged with respect to litter removal and no-grass present. Additional factors influenced sampling locations, including accessibility by foot and property laws. At each site, a sampling location was chosen > 10 m from a road, > 10 m from any human structures, < 8° slope, upland position without hydrophytic vegetation or hydric

soils, and active urban forest understory dynamics in which leaves decompose naturally to form a forest floor. Only urban forests with hardwood tree genera (*Acer*, *Alnus*, *Carya*, *Fagus*, *Fraxinus*, *Prunus*, *Quercus*, *Rhus*, *Ulmus*, etc.) with native and exotic understory plants but without grasses were studied.

To capture the effect of waste incineration, four or five urban forest sites within a 1.4-km distance of the MSWI were sampled in each city. Loppi et al. (2000) observed enrichment of TMs within 2 km of a MSWI in Italy. Separate urban forest stands meeting the desired requirements were limited near the Springfield and Poughkeepsie facilities. This assumes that pollution from a MSWI generates a measurable effect on trace metals in urban forests in close proximity < 1.5 km. Hu et al. (2003) observed that air sampling within 2 km had significantly higher metal concentrations than sites 4 km away from a MSWI in Taiwan. Further, this atmospheric deposition distance mimics point source pollution from smelters (see Zhan et al. 2014; Chen et al. 2016). To capture the effect of urban pollution throughout each city, 18 to 20 urban forests sites were identified in ~ 10 km × ~ 10 km grid surrounding the MSWI for each city. These city sites were expected to capture background enrichment of trace metals and rare earth elements from other pollution sources in urban environments: automobiles, exhaust and combustion of fuels, dust, commercial activities, and municipal pollution (Steinnes and Friedland 2006; Adriano 2001).

## Soil and vegetation sample collection

One soil monolith collected by each master horizon (Forest floor, A, and B horizons) was collected at each sampling location. First, a 15 cm × 15 cm wooden block was randomly placed on the ground and forest floor (Oi + Oe + Oa horizons) was separated from the A horizon with a clean stainless-steel handsaw (see Richardson et al. 2014, 2015 for further details). Mineral soil was removed from around the template using handsaws, tile-spades and trowels, leaving the soil below the template intact. The exposed undisturbed rectangular soil monolith was described according to U.S. Soil Taxonomy. Each soil monolith ranged in depth from 18 to 27 cm, depending on the depth of the restricting layer and the cohesiveness of the soil monolith placed the entire mass of each master horizon into separate LDPE bags (full method explained in Dobson et al. 2017; Richardson et al. 2018). Soil samples were dried in an oven at 70 °C for 24 to 48 h to a constant weight. Soil samples were then sieved to < 2 mm to remove rock fragments, human materials (e.g., glass), and roots.

Leaf and wood samples from the three to five dominant tree genera present at the sampling site above the soil monolith collection point. Trees needed to be > 10 cm in diameter to avoid sampling very young trees. Branches and leaf samples located 4 to 9 m above ground were collected using a stainless-steel pole saw. It should be noted that sampling the lower branches of trees can cause biases in metal concentrations; metals sourced from atmospheric deposition would be expected to be highest on the leaves directly exposed to dry deposition in the upper canopy (12 to 18 m in height). Plant material was dried to a constant weight for accurate weighing. Plant samples were air-dried at ~35 °C in an oven for 24 h. Dried leaves had their main veins removed to avoid the dilution of leaf blade material by the woody material. Leaf blade material was then ground to < 2 mm in a stainless-steel grinder.

## Soil physicochemical analyses

To measure soil pH, 3.0 ± 0.2 g of sieved soil material was weighed into 50-mL centrifuge tubes and extracted with 7.5 mL of 0.01 M CaCl<sub>2</sub> and shaken for 1 h. The soil was allowed to settle and pH was measured in the supernatant using a pH Meter (8015 VWR, Radnor, PA, USA). Loss-on-ignition (LOI) was measured to estimate soil organic matter content (%SOM), with ~ 10 g subsamples of soil were combusted at 550 °C in a muffle furnace for 8 h. Soil particle size distribution was measured using a modified Bouyoucos hydrometer method, with ~ 30 g of dried soil dispersed with 100 mL of 1 M sodium hexametaphosphate (HMP) solution for at least 8 h. The HMP-soil slurry was washed out into a 1-L graduated cylinder with deionized water. After mixing, hydrometer readings were taken at 60 s and 1.5 h to the closest

0.5 g L<sup>-1</sup>. For soil pH, LOI, and particle size analyses, every batch of 20 samples included one blank and a duplicate.

## Soil and plant elemental analyses

For trace element and rare earth element analyses in soils, 0.5 g was weighed into 50 mL centrifuge tubes for acid digestion. Soil samples were extracted using a strong acid, pseudo-total digestion with 5 mL of 9:1HNO<sub>3</sub>:HCl (reverse aqua regia) acid heated to 90 °C for 45 min. This method allows for quantification of metals that are sorbed to organic matter and secondary Al and Fe oxides but not within crystalline silicates, providing an estimate of metals that are bioavailable or potentially mobile (Chen et al. 2016). Plant samples were ashed at 550 °C in a muffle furnace and digested with 5 mL of reverse aqua regia heated to 90 °C for 45 min. With every 20 samples, a preparation blank, a duplicate, and a standard reference material (SRM) were included. Montana Soil 2711a and Peach leaves 1547a from the National Institute of Standards and Technology (NIST) were used as SRMs for soil and plant samples, respectively. The digests were then diluted to 50 mL using de-ionized water then further diluted with 2.5% nitric acid at 1:2 ratio and analyzed for a suite of trace metals (As, Cd, Cu, Ga, Mn, Ni, Pb, V, W, Zn) and rare earth elements (La, Ce, Pr, Nd, Sm, Eu, Gd, Tb, Dy, Ho, Yb) with an Agilent 7700x Inductively Coupled Plasma Mass Spectrometer (Agilent Technology, Santa Clara, CA, USA). Recoveries for pseudo-total digests of As, Cd, Ce, Co, Cr, Cu, Ga, La, Mn, Nd, Ni, Pb, V, W, Zn in Montana Soil 2711a were 62–109% of their certified, reference, or informational values (Mackey et al. 2010). The low recovery of elements such as Mn and Nd was due to non-dissolution of silicate minerals, which is expected for pseudo-total digestions (Chen et al. 2016). Recoveries for pseudo-total digests of As, Cd, Ce, Cu, Gd, Pb, La, Mn, Nd, Ni, Sm, Tb, V, Yb, and Zn in Peach Leaves 1547a were 81 to 105% of their certified and informational values. The metal concentration coefficient of variation between intra-sample duplicates was < 7%, and metal concentrations in the preparation blank samples were < 1% of their analyte concentrations.

## Statistical analyses

Descriptive statistics were calculated in Matlab (Mathworks, Natick, MA, USA). Average values are presented in text and in figures ± 1 standard error. Data were tested for normal distribution using the Kolmogorov-Smirnov test and logarithmically transformed when necessary to establish normality. Trace metal and rare earth element concentrations were tested for correlation with each other using linear regressions on untransformed data. Forest floor and A horizon soil concentrations were compared by their MSWI proximity using analysis of variation (ANOVA) tests with posthoc *t* tests. Tree

leaves and wood concentrations were compared across MSWI proximity and genera using analysis of variation (ANOVA) tests with posthoc *t* tests.

## Results

### Forest floor TM and REE concentrations

Overall, the majority of the TM concentrations for forest floor were within typical ranges of soil concentrations. Median forest floor TM concentrations were as follows: As = 1.5 mg kg<sup>-1</sup>, Cd = 0.2 mg kg<sup>-1</sup>, Cu = 10.3 mg kg<sup>-1</sup>, Ga = 1.4 mg kg<sup>-1</sup>, Mn = 421 mg kg<sup>-1</sup>, Ni = 8.5 mg kg<sup>-1</sup>, Pb = 61 mg kg<sup>-1</sup>, V = 20.3 mg kg<sup>-1</sup>, W = 14.3 mg kg<sup>-1</sup>, and Zn = 72 mg kg<sup>-1</sup>. The forest floor TM concentrations were comparable (greater or lower than 50%) to those measured in non-urban areas by Richardson and Friedland (2016), Gandois et al. (2010) and Landre et al. (2010). When comparing forest floor TM concentrations at city sites among the three cities, most TMs were not significantly different (*p* > 0.10; Table 1). Springfield had significantly lower forest floor Mn and Ni concentrations and Poughkeepsie had lower forest floor Pb and W concentrations than Hartford (*p* < 0.05; Table 1). Median forest floor REE concentrations were La = 3.1 mg kg<sup>-1</sup>, Ce = 7.6 mg kg<sup>-1</sup>, Pr = 0.7 mg kg<sup>-1</sup>, Nd = 2.8 mg kg<sup>-1</sup>, Sm = 0.6 mg kg<sup>-1</sup>, Eu = 0.2 mg kg<sup>-1</sup>, Gd = 0.6 mg kg<sup>-1</sup>, Tb =

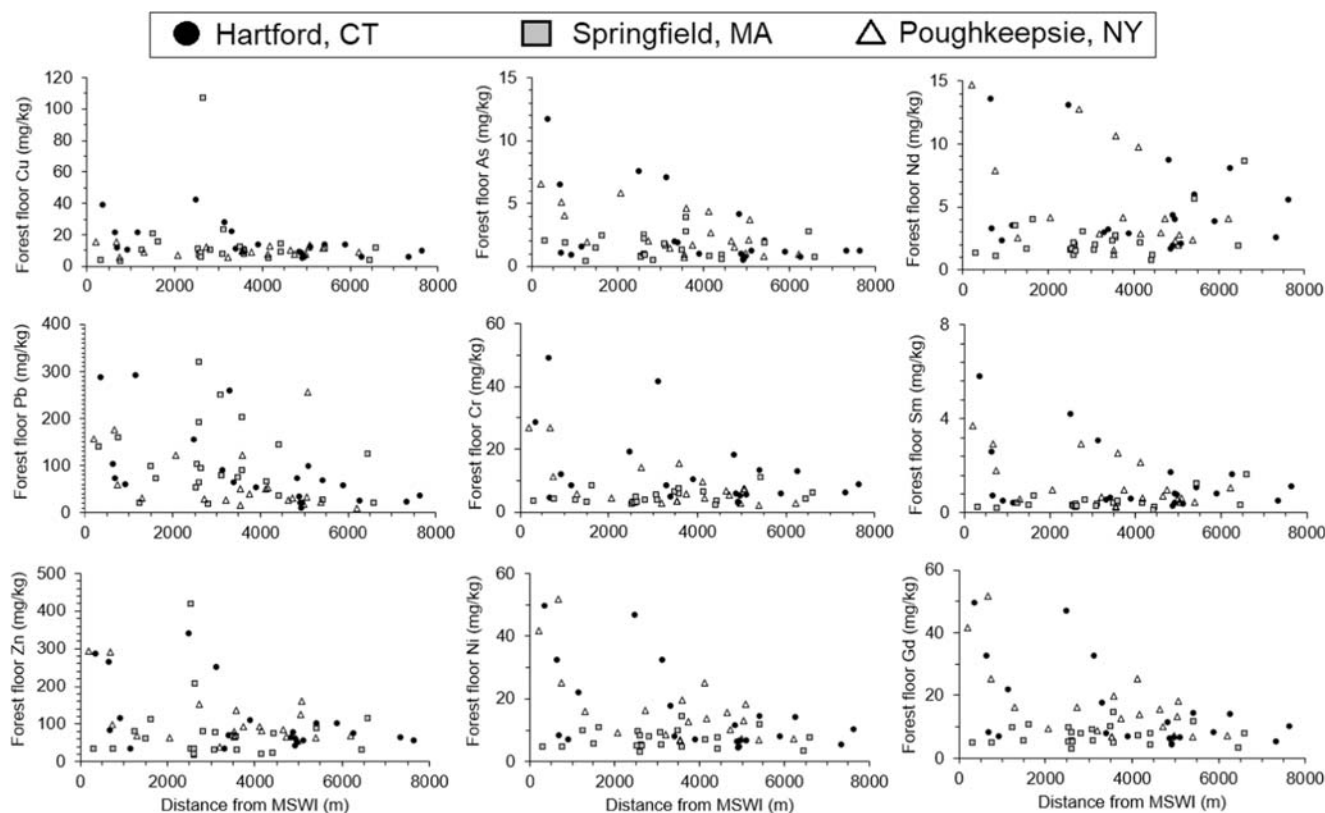
38.2 mg kg<sup>-1</sup>, Dy = 0.5 mg kg<sup>-1</sup>, Ho = 0.1 mg kg<sup>-1</sup>, and Yb = 0.2 mg kg<sup>-1</sup>. REE concentrations were comparable (greater or lower than 50%) to litter and forest floor concentrations in southern Sweden (Tyler 2004, 2005) and South Korea (Gautam et al. 2019), except for Tb which was 4 to 20 times higher than the other studies. When comparing forest floor REE concentrations for city sites among the three cities, REE concentrations were similar for Hartford and Poughkeepsie but Springfield had significantly lower concentrations of REE compared to Hartford (*p* < 0.05; Table 1).

MSWI sites and city sites were not significantly different at Hartford for any metal. In Springfield, forest floor W and Zn concentrations were greater at MSWI sites than city sites. Further, Poughkeepsie had forest floor Cd, Ga, Mn, Ni, Pb, V, W, and Zn concentrations greater at MSWI sites than city sites. Moreover, forest floor REE concentrations at MSWI sites and city sites were not significantly different at Hartford for any element. In Springfield, forest floor La concentrations were greater at MSWI sites than city sites while, in Hartford, all REE concentrations were greater at MSWI sites than city sites (Table 1). When considering all MSWI sites and city sites pooled together across the three cities, TMs exhibited a few significant differences for Ni and V but REEs were not significantly different (Supplemental Figure 2). When examining a select subset of TM and REEs in Fig. 2, there does not appear to be any consistent visible trends in concentrations with distance away from the MSWI facility. Forest floor

**Table 1** Forest floor TM and REE concentrations at waste incinerator sites compared to sites distributed throughout each city. WI sites are urban forests < 1.3 km in proximity to the waste incinerator while city sites are >

2 km away from the waste incinerator. HF = Hartford CT, PK = Poughkeepsie NY, and SP = Springfield MA. (\*) indicates a significant difference between waste incinerator and city sites *p* < 0.05

	As	Cd	Cu	Ga	Mn	Ni	Pb	V	W	Zn		
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
HF city sites	2.4 ± 0.7	0.4 ± 0.1	14 ± 3	2.3 ± 0.4	826 ± 344	14 ± 3	77 ± 17	28 ± 6	46 ± 14	98 ± 20		
HF MSWI sites	1.7 ± 1.1	0.3 ± 0.1	13 ± 3	1.9 ± 0.7	394 ± 177	11 ± 4	67 ± 26	39 ± 7	57 ± 24	87 ± 25		
PK city sites	2.3 ± 0.3	0.3 ± 0.0	10 ± 1	1.7 ± 0.2	795 ± 88	12 ± 1	55 ± 14	23 ± 4	11 ± 3	88 ± 8		
PK MSWI sites	4.4 ± 0.4	0.5 ± 0.0*	11 ± 1	3.8 ± 0.4*	2760 ± 513*	34 ± 4*	106 ± 16*	105 ± 16*	5 ± 1*	187 ± 27*		
SP city sites	1.6 ± 0.2	0.2 ± 0.0	15 ± 5	1.3 ± 0.2	242 ± 48	7 ± 1	106 ± 19	22 ± 3	38 ± 7	81 ± 22		
SP MSWI sites	1.4 ± 0.2	0.1 ± 0.0	10 ± 2	1.3 ± 0.1	189 ± 43	6 ± 1	104 ± 14	19 ± 2	14 ± 1*	52 ± 5*		
	La	Ce	Pr	Nd	Sm	Eu	Gd	Tb	Dy	Ho	Yb	
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	
HF City sites	6.4 ± 1.2	14.8 ± 2.8	1.4 ± 0.3	5.8 ± 1.3	1.3 ± 0.3	0.3 ± 0.1	1.3 ± 0.3	77 ± 21	1.0 ± 0.3	0.18 ± 0.05	0.4 ± 0.1	
HF MSWI sites	6.0 ± 1.9	13.4 ± 4.8	1.2 ± 0.4	4.8 ± 2.0	0.9 ± 0.5	0.2 ± 0.1	0.9 ± 0.6	53 ± 33	0.7 ± 0.4	0.12 ± 0.07	0.3 ± 0.1	
PK City sites	4.6 ± 0.8	9.1 ± 1.7	1.0 ± 0.2	4.6 ± 0.8	1.0 ± 0.2	0.3 ± 0.0	1.1 ± 0.2	64 ± 11	0.8 ± 0.1	0.14 ± 0.02	0.3 ± 0.1	
PK MSWI sites	9.2 ± 1.1*	27.4 ± 3.5*	2.3 ± 0.3*	10.6 ± 1.5*	2.3 ± 0.3*	0.7 ± 0.1*	2.9 ± 0.5*	228 ± 34*	2.7 ± 0.5*	0.45 ± 0.07*	0.8 ± 0.1*	
SP City sites	3.4 ± 0.7	6.6 ± 0.8	0.6 ± 0.1	2.5 ± 0.4	0.5 ± 0.1	0.1 ± 0.0	0.5 ± 0.1	28 ± 6	0.4 ± 0.1	0.07 ± 0.01	0.2 ± 0.0	
SP MSWI sites	1.9 ± 0.1*	5.7 ± 0.6	0.5 ± 0.1	1.9 ± 0.3	0.3 ± 0.0	0.1 ± 0.0	0.3 ± 0.1	22 ± 2	0.3 ± 0.0	0.04 ± 0.01	0.1 ± 0.0	



**Fig. 2** Comparison of select TMs and REEs concentrations in the forest floor by distance from MSWI sites across each city. TMs selected are common urban pollutants: As, Cr, Cu, Pb, Ni, and Zn. The REEs chosen

are Nd, Sm, and Gd, which have lower abundances than other REEs and may be more sensitive to MSWI emissions.  $N = 68$  soils total

concentrations of Cu, Pb, Nd, and Sm are similar across distances away from the MSWI (Fig. 2). However, there are fewer high Zn, Cr, and Ni concentration outliers as the distance from MSWI increases (Fig. 2).

### Mineral soil a horizon TM and REE concentrations

Overall, A horizon TM and REE concentrations were within typical ranges of soil concentrations. Median A horizon TM concentrations were as follows: As =  $0.5 \text{ mg kg}^{-1}$ , Cd =  $0.8 \text{ mg kg}^{-1}$ , Cu =  $21 \text{ mg kg}^{-1}$ , Ga =  $4.2 \text{ mg kg}^{-1}$ , Mn =  $284 \text{ mg kg}^{-1}$ , Ni =  $15.5 \text{ mg kg}^{-1}$ , Pb =  $72 \text{ mg kg}^{-1}$ , V =  $34.7 \text{ mg kg}^{-1}$ , W =  $15.5 \text{ mg kg}^{-1}$ , and Zn =  $55 \text{ mg kg}^{-1}$ . The A horizon TM concentrations were comparable (greater or lower than 50%) to those measured in non-urban areas by Richardson and Friedland (2016) and Gandois et al. (2010) in northeastern USA and France, respectively. However, A horizon TM concentrations were generally lower than concentrations observed by Landre et al. (2010). When comparing A horizon TM concentrations for city sites among the three cities, Poughkeepsie had significantly higher concentrations of As, Cd, Cu, Mn, Ni, V, and Zn than Hartford and Springfield ( $p < 0.05$ ; Table 2). Median A horizon REE concentrations were La =  $8.2 \text{ mg kg}^{-1}$ , Ce =  $18.5 \text{ mg kg}^{-1}$ , Pr =  $2.7 \text{ mg kg}^{-1}$ , Nd =  $9.0 \text{ mg kg}^{-1}$ , Sm =  $2.6 \text{ mg kg}^{-1}$ , Eu =  $0.5 \text{ mg kg}^{-1}$ , Gd =  $2.0 \text{ mg kg}^{-1}$ , Tb =

$1.8 \text{ mg kg}^{-1}$ , Dy =  $2.3 \text{ mg kg}^{-1}$ , Ho =  $0.3 \text{ mg kg}^{-1}$ , and Yb =  $0.6 \text{ mg kg}^{-1}$ . REEs were comparable (greater or lower than 50%) to surface soil concentrations in southern Sweden (Tyler 2004, 2005) and France and Germany (Brioschi et al. 2013). When comparing A horizon REE concentrations for city sites among the three cities, REE concentrations were not significantly different for Poughkeepsie and Springfield ( $p > 0.10$ ) but Hartford had significantly lower concentrations of REE compared to Poughkeepsie ( $p < 0.05$ ; Table 2).

In Hartford, A horizon Pb concentrations were greater at MSWI sites than city sites. In Poughkeepsie, A horizon Cd and Ni concentrations were greater at MSWI sites than city sites. However, A horizon REE concentrations at MSWI sites and city sites were not significantly different for any element for Hartford. In Springfield, A horizon La and Nd concentrations were greater at MSWI sites than city sites (Table 2). MSWI sites had A horizons La concentrations greater than city sites at Poughkeepsie (Table 2). When considering all MSWI sites and city sites pooled together across the three cities, TMs exhibited a no significant differences but REEs had significant differences for La and Nd (Supplemental Figure 2). Using a select subset of TM and REEs in Fig. 3, there does not appear to be any consistent visible trends in concentrations with distance away from the MSWI facility. Concentrations of Cu, Cr, Ni, Sm, and Gd are similar across distances away from the MSWI (Fig. 3). However, there are



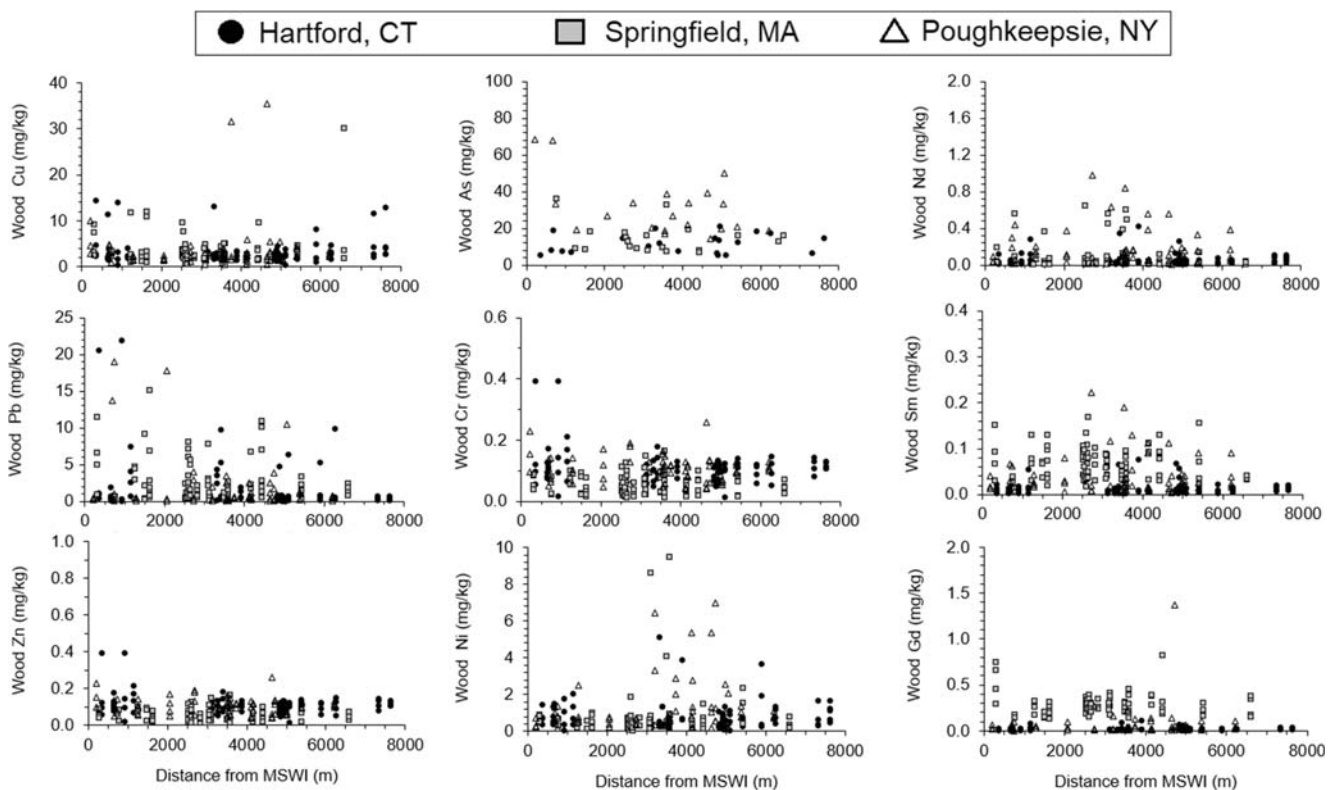
**Table 2** A horizon-TM concentrations at waste incinerator sites compared to sites distributed throughout each city. WI sites are urban forests < 1.3 km in proximity to the waste incinerator while city sites are > 2 km away from the waste incinerator. HF = Hartford CT, PK = Poughkeepsie NY, and SP = Springfield MA. (\*) indicates a significant difference between waste incinerator and city sites  $p < 0.05$

	As	Cd	Cu	Ga	Mn	Ni	Pb	V	W	Zn	
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	
HF city sites	0.34 ± 0.06	0.63 ± 0.05	17 ± 2	4.6 ± 0.4	202 ± 46	11 ± 1	56 ± 7	35 ± 4	20 ± 3	35 ± 6	
HF MSWI sites	0.30 ± 0.15	0.46 ± 0.08	19 ± 4	3.1 ± 0.6	87 ± 43	9 ± 2	82 ± 15*	29 ± 2	18 ± 4	31 ± 14	
PK city sites	0.93 ± 0.09	1.18 ± 0.10	31 ± 3	5.4 ± 0.5	789 ± 93	27 ± 2	116 ± 22	54 ± 5	14 ± 3	96 ± 9	
PK MSWI sites	1.64 ± 0.54	2.01 ± 0.58*	50 ± 16	8.5 ± 2.4	968 ± 212	47 ± 12*	206 ± 62	90 ± 24	21 ± 9	162 ± 49	
S city sites	0.64 ± 0.14	0.79 ± 0.07	23 ± 3	4.4 ± 0.4	292 ± 54	14 ± 1	96 ± 27	34 ± 3	27 ± 9	66 ± 15	
SP MSWI sites	0.80 ± 0.36	1.12 ± 0.38	28 ± 13	8.5 ± 3.8	636 ± 313	26 ± 10	70 ± 23	51 ± 21	16 ± 2	81 ± 37	
	La	Ce	Pr	Nd	Sm	Eu	Gd	Tb	Dy	Ho	Yb
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
HF city sites	7 ± 1	17 ± 2	2.6 ± 0.2	7 ± 1	2.3 ± 0.2	0.4 ± 0.1	1.6 ± 0.2	1.5 ± 0.2	1.9 ± 0.2	0.2 ± 0.1	0.4 ± 0.1
HF MSWI sites	8 ± 1	15 ± 3	2.1 ± 0.2	8 ± 2	2.1 ± 0.2	0.4 ± 0.1	1.7 ± 0.3	1.4 ± 0.4	1.6 ± 0.2	0.2 ± 0.1	0.5 ± 0.1
PK city sites	9 ± 1	20 ± 2	3.3 ± 0.3	11 ± 2	3.5 ± 0.4	0.8 ± 0.1	3.2 ± 0.3	3.1 ± 0.3	3.9 ± 0.4	0.5 ± 0.1	0.9 ± 0.1
PK MSWI sites	17 ± 5*	33 ± 10	5.4 ± 2.0	21 ± 7	6.3 ± 2.3	1.0 ± 0.4	5.7 ± 2.0	5.2 ± 1.8	6.5 ± 2.5	0.7 ± 0.3	1.3 ± 0.5
SP sites	11 ± 2	20 ± 3	3.2 ± 0.4	10 ± 1	2.6 ± 0.3	0.5 ± 0.1	2.1 ± 0.3	2.0 ± 0.3	2.5 ± 0.4	0.3 ± 0.1	0.7 ± 0.1
SP MSWI sites	29 ± 12*	43 ± 18	6.5 ± 3.0	25 ± 11*	5.4 ± 2.4	1.1 ± 0.6	4.9 ± 2.3	4.9 ± 2.4	5.3 ± 2.7	0.8 ± 0.4	1.7 ± 0.9

fewer high Pb, Zn, As, Cr, and Nd concentration outliers as the distance from MSWI increases (Fig. 3).

Soil pH was similar for Hartford ( $4.22 \pm 0.13$ ), Poughkeepsie ( $4.38 \pm 0.16$ ), and Springfield ( $4.11 \pm 0.17$ ). Similarly, %SOM was similar for Hartford ( $12\% \pm 1\%$ ),

Poughkeepsie ( $14\% \pm 2\%$ ), and Springfield ( $10\% \pm 2\%$ ). Soils at Poughkeepsie had higher %Clay ( $6.0\% \pm 0.7\%$ ) than Springfield ( $3.9\% \pm 0.6\%$ ), but Hartford CT %Clay ( $5.3\% \pm 0.6\%$ ) was similar to both cities. To explore the second hypothesis that master variables for soil metal sorption (%Clay,



**Fig. 3** Comparison of select TMs and REEs concentrations in the A horizon by distance from MSWI sites across each city. TMs selected are common urban pollutants: As, Cr, Cu, Pb, Ni, and Zn. The REEs

chosen are Nd, Sm, and Gd, which have lower abundances than other REEs and may be more sensitive to MSWI emissions.  $N = 68$  soils total

%SOM, and soil pH) could mask the effect of proximity to the MSWI, linear regressions for soil properties with TM and REE concentrations. Soil pH and %Clay were not significantly between MSWI and city sites for all three cities. However, %SOM was significantly different for Hartford ( $12\% \pm 1\%$  for city sites and  $8\% \pm 1\%$  for MSWI sites) and Poughkeepsie ( $16\% \pm 2\%$  for city sites and  $8\% \pm 2\%$  for MSWI sites). In spite of these differences, results of the linear regressions found that  $< 10\%$  of the variation in any TM or REE in the A horizon could be attributed to variations in %Clay, %SOM, and soil pH, suggesting they did not exert an appreciable amount of control on variations among the sites.

### Leaf TM and REE concentrations and tree genera

Overall, median leaf TM concentrations were as follows: As =  $0.1 \text{ mg kg}^{-1}$ , Cd =  $0.07 \text{ mg kg}^{-1}$ , Cu =  $5.4 \text{ mg kg}^{-1}$ , Ga =  $0.04 \text{ mg kg}^{-1}$ , Mn =  $279 \text{ mg kg}^{-1}$ , Ni =  $1.5 \text{ mg kg}^{-1}$ , Pb =  $0.6 \text{ mg kg}^{-1}$ , V =  $0.2 \text{ mg kg}^{-1}$ , and Zn =  $32.5 \text{ mg kg}^{-1}$ . These concentrations were comparable (greater or lower than 50%) to other studies in non-polluted systems of Canada (Landre et al. 2010), northeastern USA (Richardson and Friedland 2016), southern Sweden (Tyler 2005), and South Korea (Gautam et al. 2019). When comparing TMs in city sites among the three cities, Springfield city sites had significantly higher Ga and Pb leaf concentrations than Poughkeepsie and Hartford city sites ( $p < 0.05$ ; Table 3). Median leaf REE concentrations were as follows: La =  $0.35 \text{ mg kg}^{-1}$ , Ce =  $0.12 \text{ mg kg}^{-1}$ , Pr =  $0.06 \text{ mg kg}^{-1}$ , Nd =  $0.09 \text{ mg kg}^{-1}$ , Sm =  $0.06 \text{ mg kg}^{-1}$ , Eu =  $0.03 \text{ mg kg}^{-1}$ , Gd =  $0.08 \text{ mg kg}^{-1}$ , Tb =  $0.43 \text{ mg kg}^{-1}$ , Dy =  $0.05 \text{ mg kg}^{-1}$ , Ho =  $0.012 \text{ mg kg}^{-1}$ , and Yb =  $0.02 \text{ mg kg}^{-1}$ . For REEs in city sites among the three cities, Springfield city sites had significantly higher leaf La, Pr, Eu, Gd, Ho, and Yb than Hartford and Poughkeepsie city sites ( $p < 0.05$ ; Table 3). Further Springfield and Poughkeepsie city sites had significantly higher leaf Sm, Gd, Tb, Dy, Ho, and Yb concentrations than Hartford ( $p < 0.05$ ; Table 3). Evaluating a select subset of TM and REEs in Fig. 4, there does not appear to be any visible trends in concentrations with distance away from the MSWI facility for Cu, Pb, and As. Leaf concentrations of Zn and Cr are greatest within 2000 m of the MSWI and between 2000 and 6000 m distance away from the MSWI (Fig. 3). This suggests there may be two sources of Cr and Zn across the three cities, which roughly correspond with one being the MSWI and nearby industrial areas and the urban centers for the three cities. Foliar concentrations of Ni, Nd, and Sm appear to peak 2000 to 6000 m away from the MSWI. However, there are fewer high Pb, Zn, As, Cr, and Nd concentration outliers as the distance from MSWI increases (Fig. 4).

Overall, 23 different tree genera were sampled across the three cities but were not evenly distributed among the cities. *Populus* was only sampled at Hartford; *Tilia*, *Gleditsia*, and *Oxydendrum* were only sampled at Springfield; and *Platanus* and the invasive *Ailanthus altissima* were only present in Poughkeepsie. Uncommon tree genera sampled from at least two cities include *Aesculus*, *Catalpas*, *Celtis*, *Cornus*, *Salix*, *Sassafras*, *Magnolia*, and *Juglans*. Five the six most common tree genera (present  $> 10$  sites) were found across the three cities (*Acer*, *Fagus*, *Prunus*, *Quercus*, and *Ulmus*). *Alnus* was present at Hartford and Springfield, but not Poughkeepsie. To test if tree genera influenced foliar concentrations, leaf TM and REE concentrations of the six most common trees from all sites were compared by ANOVA (Fig. 5). As shown in Fig. 6, Cd, Pb, Pr, and Tb were significantly different among the six tree genera ( $p < 0.05$ ) while elements such as Cu, Ni, V, Sm and Ho had narrow variations and were not significantly different ( $p > 0.10$ ). More importantly, ANOVA with posthoc *t* tests revealed there were significant differences among genera for As, Cd, Mn, Pb, La, Pr, Nd, Eu, Tb, Dy, and Yb ( $P < 0.05$ ) and genus *Fagus* consistently had lower TM and REE concentrations compared to the other genera ( $p < 0.05$ ; Fig. 6).

### Wood TM and REE concentrations

Overall, median wood TM concentrations were as follows: As =  $0.04 \text{ mg kg}^{-1}$ , Cd =  $0.04 \text{ mg kg}^{-1}$ , Cu =  $2.4 \text{ mg kg}^{-1}$ , Ga =  $0.02 \text{ mg kg}^{-1}$ , Mn =  $65 \text{ mg kg}^{-1}$ , Ni =  $0.5 \text{ mg kg}^{-1}$ , Pb =  $0.5 \text{ mg kg}^{-1}$ , V =  $0.09 \text{ mg kg}^{-1}$ , and Zn =  $15.5 \text{ mg kg}^{-1}$ . These concentrations were comparable to other studies outside of urban systems (Landre et al. 2010; Richardson and Friedland 2016). When comparing TMs in city sites among the three cities, Springfield city sites had significantly higher wood Ga and Mn concentrations than Poughkeepsie and Hartford city sites (Table 4). Median wood REE concentrations were as follows: La =  $0.13 \text{ mg kg}^{-1}$ , Ce =  $0.05 \text{ mg kg}^{-1}$ , Pr =  $0.03 \text{ mg kg}^{-1}$ , Nd =  $0.04 \text{ mg kg}^{-1}$ , Sm =  $0.02 \text{ mg kg}^{-1}$ , Eu =  $0.02 \text{ mg kg}^{-1}$ , Gd =  $0.03 \text{ mg kg}^{-1}$ , Tb =  $0.21 \text{ mg kg}^{-1}$ , Dy =  $0.02 \text{ mg kg}^{-1}$ , Ho =  $0.007 \text{ mg kg}^{-1}$ , and Yb =  $0.01 \text{ mg kg}^{-1}$ . Comparing REEs in city sites among the three cities, Springfield city sites had significantly higher wood La, Pr, Eu, Gd, Ho, and Yb than Hartford and Poughkeepsie city sites ( $p < 0.05$ ; Table 4). Further, Springfield city sites had significantly higher wood Sm, Tb, and Dy concentrations than Hartford city sites ( $p < 0.05$ ; Table 4).

When comparing city sites to MSWI sites, wood TM and REE concentrations did not show any significant differences

**Table 3** Leaf-TM concentrations at waste incinerator sites compared to sites distributed throughout each city. WI sites are urban forests < 1.3 km in proximity to the waste incinerator while city sites are > 2 km away from the waste incinerator. HF = Hartford CT, PK = Poughkeepsie NY, and SP = Springfield MA. (\*) indicates a significant difference between waste incinerator and city sites  $p < 0.05$

	As	Cd	Cu	Ga	Mn	Ni	Pb	V	Zn			
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg			
HF city sites	0.1 ± 0.0	0.2 ± 0.0	6 ± 0	0.04 ± 0.00	499 ± 70	2.2 ± 0.4	0.5 ± 0.1	0.22 ± 0.02	62 ± 10			
HF MSWI sites	0.1 ± 0.0	0.5 ± 0.5	6 ± 1	0.05 ± 0.02	377 ± 282	2.2 ± 0.3	2.8 ± 2.8	0.31 ± 0.09	123 ± 116			
PK city sites	0.1 ± 0.0	0.1 ± 0.0	5 ± 0	0.03 ± 0.00	393 ± 66	2.4 ± 0.3	0.4 ± 0.1	0.17 ± 0.01	31 ± 3			
PK MSWI sites	0.1 ± 0.1	0.1 ± 0.1	6 ± 2	0.03 ± 0.01	284 ± 181	2.0 ± 0.8	0.7 ± 0.4	0.21 ± 0.12	26 ± 8			
SP city sites	0.2 ± 0.1	0.2 ± 0.1	8 ± 3	0.17 ± 0.05	380 ± 96	1.9 ± 0.9	7.7 ± 3.5	0.17 ± 0.02	80 ± 22			
SP MSWI sites	0.3 ± 0.3	0.3 ± 0.3	13 ± 10	0.30 ± 0.19	454 ± 159	1.3 ± 0.3	7.4 ± 5.0	0.25 ± 0.08	74 ± 48			
	La	Ce	Pr	Nd	Sm	Eu	Gd	Tb	Dy	Ho	Yb	
	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	
HF city sites	0.3 ± 0.1	0.3 ± 0.2	0.1 ± 0.0	0.2 ± 0.1	0.04 ± 0.02	0.03 ± 0.01	0.05 ± 0.02	0.3 ± 0.1	0.04 ± 0.02	0.01 ± 0.00	0.02 ± 0.01	
HF MSWI sites	0.2 ± 0.1	0.3 ± 0.1	0.1 ± 0.0	0.1 ± 0.1	0.03 ± 0.01	0.03 ± 0.01	0.03 ± 0.01	0.4 ± 0.3	0.03 ± 0.01	0.01 ± 0.00	0.01 ± 0.00	
PK city sites	0.6 ± 0.2	0.5 ± 0.2	0.1 ± 0.0	0.5 ± 0.2	0.15 ± 0.06	0.05 ± 0.01	0.19 ± 0.07	4.3 ± 1.6	0.14 ± 0.06	0.03 ± 0.00*	0.05 ± 0.02*	
PK MSWI sites	0.3 ± 0.1	0.3 ± 0.1	0.1 ± 0.0	0.3 ± 0.1	0.08 ± 0.03	0.03 ± 0.01	0.09 ± 0.03	6.7 ± 3.4	0.06 ± 0.03	0.01 ± 0.00	0.02 ± 0.01	
SP city sites	4.8 ± 2.1	0.2 ± 0.1	2.2 ± 1.6	0.2 ± 0.1	0.15 ± 0.02	0.11 ± 0.02	0.66 ± 0.03	1.7 ± 1.2	0.13 ± 0.01	0.08 ± 0.00	0.38 ± 0.05	
SP MSWI sites	6.1 ± 5.9	0.3 ± 0.3	2.6 ± 3.9	0.2 ± 0.2	0.17 ± 0.05	0.14 ± 0.07	0.78 ± 0.19	1.9 ± 2.4	0.17 ± 0.04	0.10 ± 0.05	0.45 ± 0.12	

between MSWI sites and those distributed in other city sites. For the REEs, MSWI sites did not have significantly higher concentrations in their leaves than city sites (Table 4). Moreover, MSWI sites had lower leaf REE concentrations than city sites at Poughkeepsie (Table 4). When considering the leaf data pooled across the three cities, TMs and REEs did not vary significantly. When investigating a select subset of TM and REEs in Fig. 5, there does not appear to be any consistent visible trends in concentrations with distance away from the MSWI facility. Wood concentrations of Cu, Zn, As, Cr, Sm, and Gd were comparable from 0 to 6000 m away from the MSWI. Wood concentrations of Ni, Nd, and Sm appear to peak 2000 to 6000 m away from the MSWI. However, there are fewer high Pb concentration outliers as the distance from MSWI increases (Fig. 5).

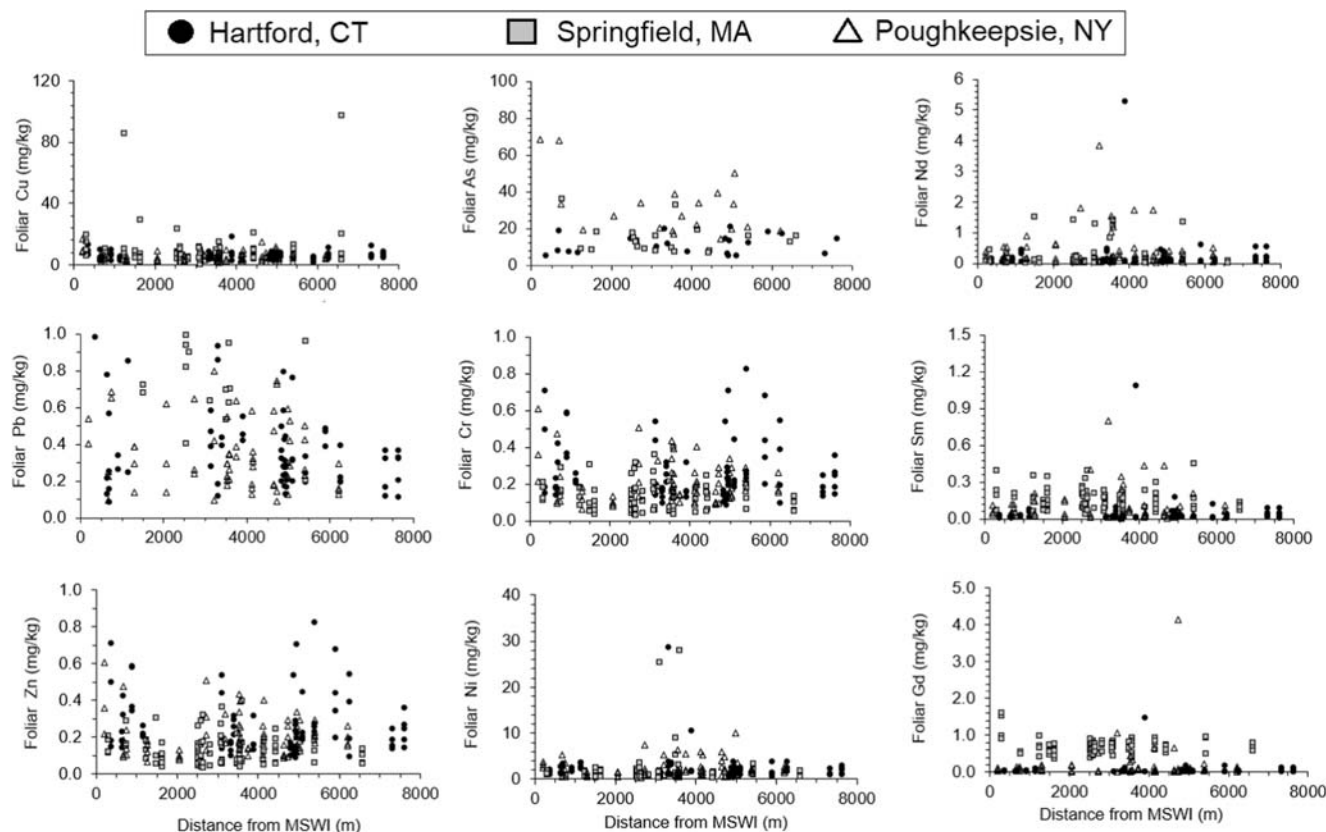
## Discussion

### Hypothesis one: no consistent enrichment of TM and REE in urban forests near MSWI facilities

The main objective of this study was to determine if urban forests in close proximity to MSWIs have elevated TM and REE concentrations compared to surrounding areas for three cities. Overall, higher TM and REE in forest floor, A horizons, leaves,

and wood were not observed consistently for MSWI sites when compared to city sites (Figs. 2, 3, 4, and 5; Supplemental Figures 2 and 3). We observed more outliers near the MSWI in the forest floor, A horizons, leaves, and wood, but there was no overall trend of decreasing metal concentrations away from the MSWI. In fact, we observed higher foliar and wood TM and REE concentrations away from the MSWI near the center of urban development. Although pooled forest floor Ni and V and A horizon La and Nd concentrations were significantly different, these differences were driven by a difference at one of cities. For example, Poughkeepsie caused overall differences in forest floor TMs while no significant differences were observed for Springfield and Hartford CT. These results suggest that MSWIs did not significantly enrich soils in close proximity with significantly greater TMs and REEs.

There are four most likely mechanisms that may have generated these results. The first and best-case scenario is that advances in air pollution control systems for MSWI is highly effective at reducing solid and gaseous phase releases to surrounding urban systems, leading to undetectable differences in urban pollution concentrations. This largely agrees with previous studies that found TMs (Cd, Cu, Mn, Ni, Pb, and Zn) from the combusted MSW are trapped as bottom ash, boiler ash, or fly ash (Belevi and Moench 2000; Chang and Ku 1998; Kuo et al. 2007; De Boom and Degrez 2012; Morf et al. 2013; Funari et al.



**Fig. 4** Comparison of select foliar TMs and REEs concentrations in the leaves by distance from MSWI sites across each city. TMs selected are common urban pollutants: As, Cr, Cu, Pb, Ni, and Zn. The REEs chosen

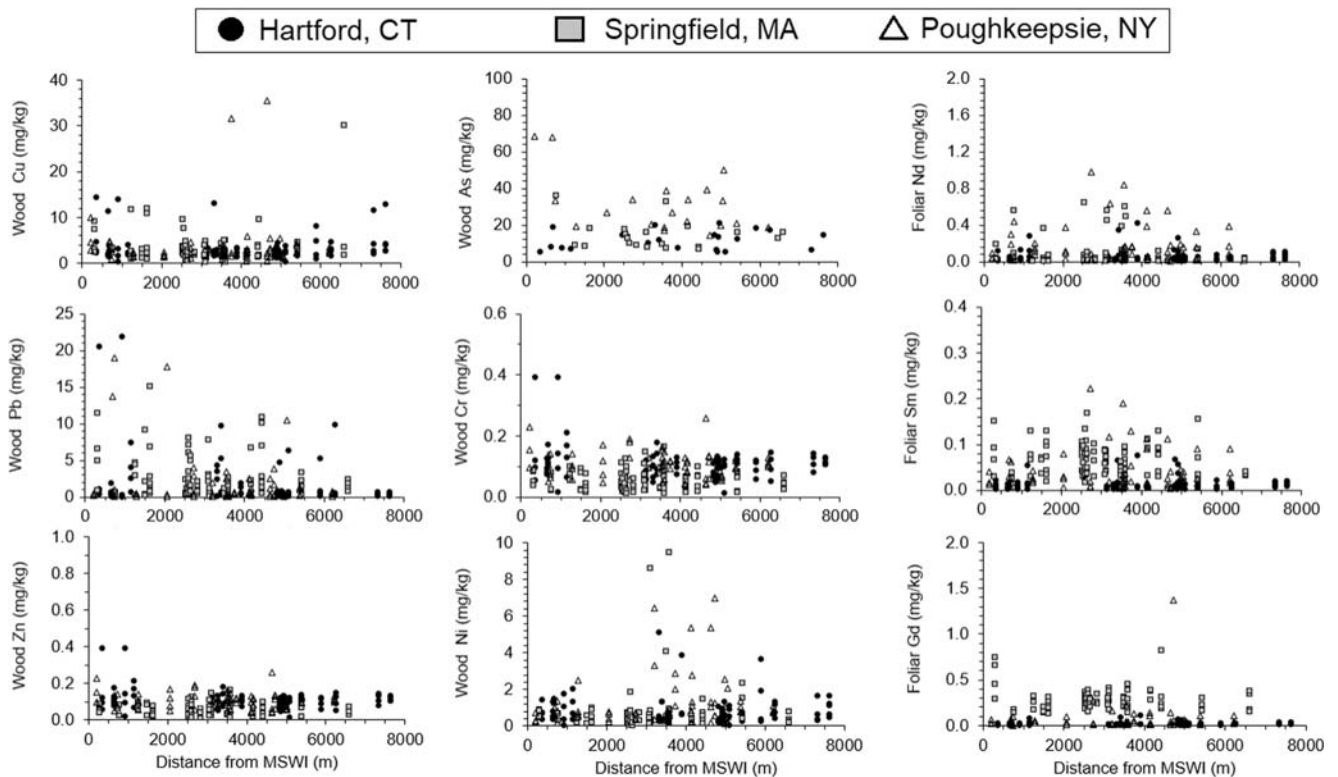
are Nd, Sm, and Gd, which have lower abundances than other REEs and may be more sensitive to MSWI emissions.  $N = 260$  trees total

2015). Moreover, the MSWI can be following best management practices of removing items containing high concentrations of TMs and REEs prior to incineration (McGill 2012; Allegrini et al. 2014; Binnemans et al. 2015). Furthermore, this agrees with Setyan et al. (2017) in which macro metals (K, Mg, Ca) emissions from two MSWIs in Switzerland were either below detection or  $< 3 \mu\text{g m}^{-3}$ . This disagrees with Loppi et al. (2000) which found elevated TMs within 2 km of a MSWI in Italy, this may be due to more effective APC systems in place at the three MSWI facilities studied.

Second, TM and REE pollution from MSWI facilities may be negligible as other forms of current urban pollution (automobiles emissions, domestic fuel combustion, industrial manufacturing, metal foundries and plating Hutzell and Luecken 2008 assessment of the USA and Huang et al. 2018 assessment in China) or historical pollution (rusting metal debris, weathering of construction materials, small-scale smelting) may be masking any TM and REE pollution. As shown in Figs. 2, 3, 4, and 5, trees and soils 2000 to 6000 m away from the MSWI but in or near the center of urban developments can have similar or even higher TM and REE concentrations. Since only deciduous tree leaves were studied and most pollutant TMs are actively discriminated against during plant root uptake and vascular transport (see Clemens

2006; Peralta-Videa et al. 2009; Richardson and Friedland 2016), it may be surmised that the tree leaf data should effectively monitor current atmospheric deposition of pollution and the overall background of urban pollution masks any potential signal from the MSWI. In this scenario, TM and REE pollutions from a MSWI would pose a lesser health hazard than other urban sources. However, organic contaminants potentially emitted from MSWI could be a more important health concern than TM and REEs such as Dioxins (Chao et al. 2003; Hu and Shy 2001). It must be noted that meta-analyses on epidemiological studies on human communities near MSWIs have found little evidence of adverse health effects (e.g., Porta et al. 2009 and Giusti 2009).

Third, localized effects from geologic and pedogenic processes could also mask pollution from MSWI facilities. Glacial till, glaciofluvial, and fluvial materials comprise the soil parent materials in the three cities and are heterogeneous mixtures of rocks, ranging from low to medium grade metamorphic rocks in Poughkeepsie to weathered granitic and basaltic rocks in Springfield and Hartford. These variations in soil parent materials could mask effects from known point source pollution. For example, surficial geology decreased detecting and identification of inputs of Pb-Zn smelter by Stafilov et al. (2010). Moreover, many of the soils in the urban



**Fig. 5** Comparison of select TMs and REEs concentrations in branch bole wood by distance from MSWI sites across each city. TMs selected are common urban pollutants: As, Cr, Cu, Pb, Ni, and Zn. The REEs

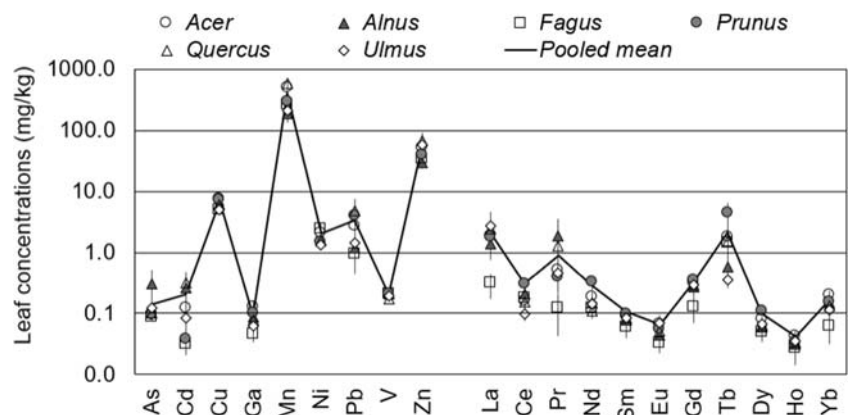
chosen are Nd, Sm, and Gd, which have lower abundances than other REEs and may be more sensitive to MSWI emissions.  $N = 260$  trees total

forests were technosols, containing remnants of construction materials, building gravel, and metal food-related debris. Due to the high degree of weathering and sorting in glaciofluvial and fluvial systems, the sediments are dominated with weathering resistant quartz and feldspars containing low TM and REE concentrations (Marek and Richardson 2020). Thus, it may be argued that geologic materials are less likely to influence metal concentrations as anthropogenic alterations.

Lastly, the experimental design assumed that TM and REE from a MSWI would behave similarly to emissions from a smelter and would be best detected within 1.5 km of the MSWI. This was largely based upon Hu et al. (2003) and

Loppi et al. (2000) where they observed the greatest air TM concentrations were within 2 km of the MSWI while sites 4 km away were nearly indistinguishable from background air TM concentrations. One plausible situation is that MSWI emissions mixed by atmospheric dispersion would release very low TM and REE concentrations, indistinguishable from background urban pollution from other sources. In a study on a Cu smelter in Spain, Chen et al. (2016) modeled As, Cu, and Zn emissions were relatively small  $\sim 10 \text{ ng m}^{-3}$  and tightly constrained within 4 km of the emission site but an even smaller emission of  $< 1 \text{ ng m}^{-3}$  but broader influence greater of  $> 40 \text{ km}$ , largely depending on wind speed.

**Fig. 6** A comparison of TMs and REEs within leaves of the six most common trees sampled in city and waste incinerator sites among the three cities



**Table 4** Branch bole wood TM concentrations at waste incinerator sites compared to sites distributed throughout each city. WI sites are urban forests < 1.3 km in proximity to the waste incinerator while city sitesare > 2 km away from the waste incinerator. HF = Hartford CT, PK = Poughkeepsie NY, and SP = Springfield MA. No significant differences between waste incinerator and city sites were observed  $p < 0.05$ 

	As mg/kg	Cd mg/kg	Cu mg/kg	Ga mg/kg	Mn mg/kg	Ni mg/kg	Pb mg/kg	V mg/kg	Zn mg/kg			
HF city sites	0.07 ± 0.01	0.2 ± 0.2	3 ± 0	0.01 ± 0.00	126 ± 20	0.8 ± 0.1	1 ± 0	0.13 ± 0.02	30 ± 5			
HF MSWI sites	0.08 ± 0.03	0.3 ± 0.3	4 ± 2	0.01 ± 0.00	86 ± 73	0.9 ± 0.2	3 ± 3	0.20 ± 0.07	41 ± 24			
PK city sites	0.05 ± 0.01	0.1 ± 0.1	4 ± 1	0.01 ± 0.00	80 ± 15	1.2 ± 0.2	2 ± 1	0.10 ± 0.01	20 ± 2			
PK MSWI sites	0.07 ± 0.02	0.1 ± 0.1	3 ± 1	0.01 ± 0.00	60 ± 37	0.8 ± 0.3	3 ± 3	0.16 ± 0.07	20 ± 6			
SP city sites	0.07 ± 0.05	0.1 ± 0.1	3 ± 1	0.07 ± 0.02	203 ± 53	0.7 ± 0.3	2 ± 1	0.09 ± 0.01	33 ± 9			
SP MSWI sites	0.12 ± 0.16	0.1 ± 0.1	4 ± 2	0.08 ± 0.03	167 ± 92	0.4 ± 0.1	3 ± 2	0.12 ± 0.05	49 ± 30			
	La mg/kg	Ce mg/kg	Pr mg/kg	Nd mg/kg	Sm mg/kg	Eu mg/kg	Gd mg/kg	Tb mg/kg	Dy mg/kg	Ho mg/kg	Yb mg/kg	
HF city sites	0.09 ± 0.02	0.08 ± 0.02	0.02 ± 0.00	0.05 ± 0.01	0.01 ± 0.00	0.02 ± 0.00	0.02 ± 0.00	0.1 ± 0.0	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	
HF MSWI sites	0.07 ± 0.04	0.09 ± 0.04	0.01 ± 0.01	0.06 ± 0.03	0.01 ± 0.01	0.02 ± 0.01	0.02 ± 0.01	0.2 ± 0.2	0.01 ± 0.00	0.01 ± 0.00	0.01 ± 0.00	
PK city sites	0.26 ± 0.08	0.21 ± 0.07	0.05 ± 0.02	0.23 ± 0.08	0.05 ± 0.02	0.02 ± 0.02	0.07 ± 0.02	0.9 ± 0.2	0.04 ± 0.02	0.01 ± 0.00	0.01 ± 0.01	
PK MSWI sites	0.14 ± 0.07	0.11 ± 0.04	0.03 ± 0.01	0.13 ± 0.06	0.03 ± 0.01	0.02 ± 0.01	0.04 ± 0.01	1.9 ± 0.9	0.02 ± 0.01	0.01 ± 0.00	0.01 ± 0.00	
SP city sites	2.52 ± 1.03	0.06 ± 0.04	0.83 ± 0.58	0.07 ± 0.03	0.06 ± 0.01	0.05 ± 0.02	0.28 ± 0.02	1.9 ± 1.4	0.06 ± 0.00	0.03 ± 0.01	0.21 ± 0.03	
SP MSWI sites	4.12 ± 3.41	0.15 ± 0.15	1.39 ± 1.23	0.10 ± 0.08	0.06 ± 0.02	0.05 ± 0.02	0.29 ± 0.09	1.7 ± 2.4	0.06 ± 0.02	0.04 ± 0.02	0.27 ± 0.07	

However, Cu and Zn concentrations of 5 to 30 ng m<sup>-3</sup> are typical for urban air in the northeastern USA (Gao et al. 2002). In this case, localized pollution is unlikely to occur at a detectable concentration in soils and plants as uptake and weathering would likely exceed atmospheric inputs. This may be the case as leaf TM and REE concentrations at the MSWI sites were comparable to concentrations to rural forests of Canada, Vermont, and New Hampshire (Landre et al. 2010; Richardson and Friedland 2016).

### Hypothesis two: limited effects from soil properties and tree genera

In the second hypothesis, it was postulated that soil properties, such as %SOM, %Clay, or soil pH, and tree genera-specific physiology could impact how TMs and REEs are captured or retained in soils and vegetation, respectively. Significant differences were not observed for %Clay or soil pH for A horizons between MSWI and city sites for all three cities. However, %SOM in A horizons was significantly greater at city sites than MSWI sites for Poughkeepsie and Hartford, potentially increasing the relative sorption and retention of TMs and REEs. This could cause greater concentrations at city sites than MSWI sites but %SOM, %Clay, and soil pH

was poorly correlated with all A horizon TM and REE concentrations ( $p > 0.05$ ,  $R^2 < 0.10$ ). Thus, it can be concluded that soil properties did not enhance accumulation of TM and REE concentrations. Furthermore, these results highlight anthropogenic inputs and potentially legacy effects of land-use exert a greater impact on TM and REE concentrations than soil properties traditionally known to control sorption and retention of elements.

Tree genera significantly impacted TM and REE concentrations in leaves. There was one order of magnitude difference for As, Cd, Mn, Pb, La, Pr, Nd, Eu, Tb, Dy, and Yb for the six commonly sampled tree genera. *Fagus grandifolia* exhibited a pattern of consistently had lower TM and REE concentrations, which has been observed before (e.g., Pardo et al. 2005; Richardson and Friedland 2016) as their leaves are thin, waxy, and nutrient poor. However, most tree genera like *Acer*, *Prunus*, *Quercus*, and *Ulmus* did not exhibit a consistent pattern above or below the pooled mean. Moreover, three to five tree samples were collected from each site, diminishing the effect of any one genus effect on leaf TM and REE concentrations report for each site. Thus, while tree genera did impact foliar TM and REE concentrations, the lack of a consistent pattern and sampling of several tree genera at each site should prevent against any bias.

## Conclusions

The objective of this study was to determine if TM and REE concentrations MSWIs were greater in urban forest soils and vegetation near MSWI facilities compared to background urban pollution. These results failed to find consistent evidence that MSWI was increasing TMs and REEs. There were outliers near the MSWI with elevated TM concentrations in the forest floor, A horizons, tree leaves, and tree wood in close proximity < 2000 m when compared to urban pollution elsewhere in three cities. But overall, average concentrations in TM and REE concentrations near the MSWI were not significantly different from city sites in other urban areas of the city. This indirect study provides evidence that APC systems are effectively removing TMs and REES from being emitted to local urban forests. Many of the tree leaves in the urban forests near the MSWI had lower TM and REE concentrations than rural areas of Canada and the northeastern USA. However, further studies are needed to assess the efficacy of APC systems and eliminate potential interferences of alternative air pollution sources to validate MSWIs that are not emitting TMs and REEs to local communities. This would require direct monitoring of potential ventilation and emission sources.

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## Compliance with ethical standards

**Conflict of interest** The authors declare that they have no conflict of interest.

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