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RESEARCH ARTICLE

Uptake of toxic and nutrient elements by foraged edible and medicinal mushrooms (sporocarps) throughout Connecticut River Valley, New England, USA

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Abstract
Foraging for edible and medicinal mushrooms is a cultural and social practice both globally and in the United States. Determining the toxic and nutrient element concentrations of edible and medicinal mushrooms is needed to ensure the safe consumption of this food source. Our research examined wild, foraged mushrooms in New England, USA to assess nutrient (Ca, K, Mg, P) and toxic (As, Hg, Pb, Cd) element relationships between mushrooms, substrates, and soils. We examined a gradient in nutrient and toxic elements from more rural Mountain and Hill Zones in Massachusetts, Vermont, and New Hampshire to more developed and urban Valley and Coastal Zones in Connecticut. Substrates and mineral soils were moderate to weak predictors of mushroom tissue concentrations. We found significant differences in nutrient and toxic element concentration among the five common genera: *Ganoderma*, *Megacollybia*, *Pluteus*, *Pleurotus*, and *Russula*. In particular, *Pluteus* had consistently higher toxic element concentrations while *Pleurotus* and *Russula* had the highest Bioaccumulation Factors (BAFs). We found that the urban areas of the Valley and Coastal zones of Connecticut had Cd Target Hazard Quotient (THQ) values and ΣTHQ values > 1.0, indicating potential non-carcinogenic health hazard. However, the trend was largely driven by the > 2.0 Cd THQ for *Pluteus*. Our results suggest that foraging in more urban areas can still yield mushrooms with safe concentrations of toxic elements and abundant nutrients. Further research of this kind needs to be conducted within this region and globally to ensure humans are consuming safe, foraged mushrooms.

Keywords Forest fungi · Macronutrients · Risk assessment · Trace elements · Pollution

Introduction

Foraging wild edible mushrooms is a cultural and social practice many humans partake in, both globally and in New England, USA. Mushrooms, which are the fruiting bodies of fungi, are valuable non-timber forest products. Due to the bioactive components within mushrooms which have been shown to decrease diseases such as some cancers, heart disease, viral infections and more, this has caused the global market to expand (Sahoo et al. 2022; Breene 1990). According to Malone et al. (2022), a certified forager in Michigan, USA harvesting and selling wild *Morchella* mushrooms may earn an average of $36 per pound. However, on a larger scale, wild *Morchella* mushrooms are part of a multi-billion-dollar industry, where in most cases, local mushroom hunters sell their finds to larger markets for export (Raut et al. 2019). Thus, there is a demand for foraged mushrooms in the global market, and this demand will continue to grow with time.

For human and animal consumers, mushrooms can provide inorganic nutrients essential to health (such as Ca, K, Mg, and P), but may also serve as vectors for toxic elements (such as As, Cd, Hg, Pb) that can also be found in soil and substrates. Nutrient elements are essential for multicellular

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life (e.g., chemical signaling, biochemical structures, proteins) and are sourced from decomposing plant material (e.g., forest floor, woody debris), weathering of non-silicate minerals (e.g., dolomite, apatite) and silicate minerals (e.g., epidote, feldspar, amphibolite). Toxic elements are not needed for multicellular life and can be sourced from natural weathering of rocks enriched in trace elements (e.g., sulfide-bearing rocks and ultramafic rocks) or human pollution (e.g., mining, fuel combustion emissions, smelting emissions, other industrial emissions). Fungi are capable of acquiring nutrients from both organic (e.g., plant debris) and inorganic (e.g., mineral rock) sources. Unfortunately, hyphae of fungi can uptake toxic elements from these substrates, accumulating toxic elements bound to proteins inside fruiting bodies (Ab Rhaman et al. 2022). Toxic and nutrient element uptake is of particular importance for individuals foraging medicinal and edible mushrooms for extensive personal consumption or the commercial market, but it is unclear if urbanized areas or even rural areas with historical non-point source pollution may yield contaminated mushrooms. Substrates in historically polluted sites are likely to directly transfer pollutants but may vary with litter layer and woody debris colonization (Kalač et al. 1996; García et al. 1998; Falandysz et al. 2001).

Nutrient and toxic element accumulation in fruiting bodies of soil-dwelling fungi may depend on whether they are saprotrophic, obtaining their energy from the decomposition of organic matter, or mycorrhizal, obtaining their energy from living plant hosts. Saprotrophic wild foraged mushrooms have preferred substrates, e.g. some saprotrophic mushrooms are decomposers of sapwood on fallen trees while others consume hardwood branches and large woody debris (Baroni 2017). Mushroom forming ectomycorrhizal fungi are symbionts with trees and other woody plants and obtain their energy from their hosts in exchange for inorganic nutrients acquired from soil. Due to their role as nutrient extractors, mycorrhizal fungi may likely be more effective in nutrient uptake from soil (Talbot et al. 2013) and thus possibly toxic element uptake as well (Tekaya et al. 2017), but this may depend on the element in question (Berthelsen et al. 1995). Saprotrophic fungi are more likely to favor forest woody debris as their substrate (Baldrian 2008) and are more competitive for C-rich compounds (Talbot et al. 2013), which may be less likely to have accumulated toxic elements due to the low metal concentrations in northern hardwood woody parts (Richardson and Friedland 2016). Lastly, bioaccumulation of toxic elements is commonly driven by low abundance of nutrient elements and subsequent uptake of toxic elements with similar ionic charge and radius (Snyder et al. 1990). As a prime example, Se uptake was shown to diminish Pb and Cd uptake by oyster mushrooms (*Pleurotus ostreatus* and *Pleurotus djamor*) (de Oliveira et al. 2022). When considering forests within the Northeastern USA, it is important to note that the limiting nutrients of N and P may impact bioaccumulation factors, with the potential for more nutrients to become limiting as the stands grow in age (Naples and Fisk 2010).

Previous studies have revealed three important factors that may govern nutrient and toxic element transfer to mushrooms: (1) the substrate consumed, (2) their ecological function, such as decomposers (saprotrophic fungi) or symbiotic organisms (mycorrhizal fungi), and (3) biogeochemical competition. The cumulative effect of these factors is likely to determine the potential nutritive quality as well as safety of a given wild foraged mushroom species. Much of the current literature focuses on assessing element concentrations within edible mushrooms in countries other than the United States including China (Fu et al. 2020; Chen et al. 2009), Czech Republic (Svoboda et al. 2005), Turkey (Cayir et al. 2009; Isildak et al. 2007; Turkekul et al. 2004; Tuzen et al. 2007; Yamaç et al. 2007), Poland (Podlasińska et al. 2015), and Greece (Ouzouni et al. 2009; Kokkoris et al. 2019). The existing studies that have measured concentrations of nutrient elements present in wild edible mushrooms have found concentrations comparable to many grains and vegetables (Woldegiorgis et al. 2015; Singdevsachan et al. 2014). A review by Kalač (2012) reports that the usual content of nutrient elements Ca at 0.1 – 0.4 g/kg, K at 20 – 40 g/kg, Mg at 0.8 – 1.8 g/kg, and P at 5 – 10 g/kg in wild-growing mushrooms. Toxic elements in mushrooms, however, have been measured at levels approaching or exceeding safety thresholds for food. Wong et al. (2022) compiled data from the FDA and the EPA regarding heavy metal reference values for As, Cd, Hg, and Pb at 0.3 ug/kg/day, 1 ug/kg/day, 0.1 ug/kg/day, and 0.16 ug/kg/day respectively. Kalač (2012) additionally reports that the usual content of toxic elements As at 0.5 – 5 mg/kg, Cd at 1 – 5 mg/kg, Hg at < 0.5 – 5 mg/kg, and Pb at < 1 – 5 mg/kg in wild-growing mushrooms. Thus, the average concentrations of wild-growing mushrooms can exceed the daily reference values of an adult.

The overall objective of this study was to assess the nutrient and toxic element accumulations in wild foraged mushrooms from forests in the Connecticut River Valley of southern New England, USA. The first goal was to determine regional estimates for nutrient and toxic elements concentrations, bioaccumulation factor (BAF), and target hazard quotient (THQ) in foraged mushrooms, in particular, assessing if urbanized areas with historical pollution or nutrient-rich soil parent material impact nutrient and toxic element concentrations. The second goal was to assess if the substrate (organic soil horizon vs. woody debris) being consumed by the mushroom or background mineral soil concentrations are controlling elemental concentrations within mushrooms. The third goal was to evaluate element accumulation across common mushroom genera and between ecological niches, as elemental extraction rates may be greater by mycorrhizal

fungi. By unravelling some of these effects, this work may prove useful for optimizing the safety and nutritional quality of foraging wild mushrooms.

Materials and methods

Field sampling methods

Twenty sites were chosen across an urban development gradient from the developed, coastal forests of southern Connecticut to the montane, rural forests of southern Vermont and New Hampshire (Fig. 1). The transect covers humid continental (Dfa) to humid subtropical (Cfa) according to the Köppen climate types, with mean annual precipitation 1050 to 1300 mm/yr and mean annual temperature of 10 °C. Summers are warm to hot (30 °C) with humidity while winters are cold (− 7 °C) and can be bitter (− 15 °C) with no dry season. Native forests are dominated by northern hardwoods of american beech (*Fagus grandifolia*), maples (*Acer saccharum* and *Acer rubrum*), oaks (*Quercus* spp.), birches (*Betula* spp.), pine (*Pinus* spp.), hickory (*Carya* spp.), ashes (*Fraxinus* spp.), tulip poplar (*Liriodendron tulipifera*), poplars (*Populus* spp.), northern catalpa (*Catalpa speciosa*), eastern hemlock (*Tsuga canadensis*) but more urban forests are impacted by exotic and invasive trees such as tree of heaven (*Ailanthus altissima*), norway maple (*Acer platanoides*), and black locust (*Robinia pseudoacacia*).

Each sampling site was a ~ 2 ha section of a town or state forest, which are public areas available for wild mushroom foraging. The twenty sites were sampled twice, first June 2022 for early summer mushroom sampling and then revisited in August and early September 2022 for late summer mushroom sampling. At each site, edible mushrooms and their respective substrates (forest floor or woody debris) were collected. Linking the relationship between concentrations of elements between the mushrooms and their growth mediums (substrates), along with soils from the forest to examine soil cycling of toxic and nutrient elements.

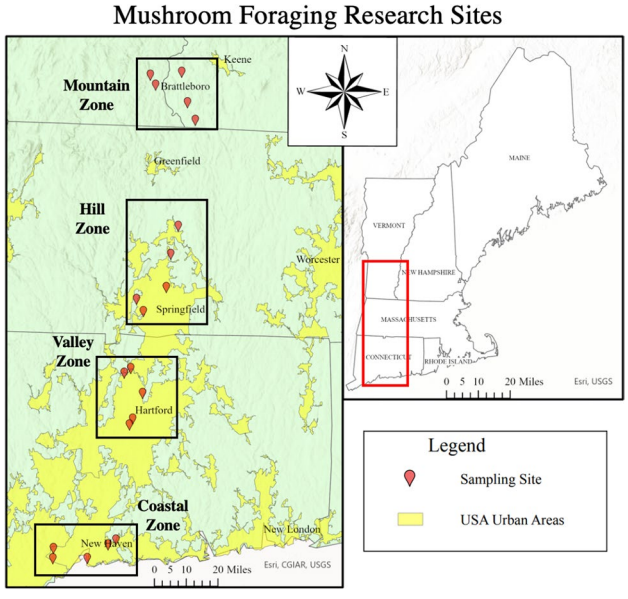

Fig. 1 Mushroom foraging sites ($n = 20$) throughout the Connecticut River Valley in New England, USA

To assess the level of ambient nutrient and toxic elements abundant at each site, we collected five mineral soil samples. The soil samples were spaced at least 50 m apart to capture the variability in elevation and hydrology at each site. Soil samples were obtained by removing the forest floor layer and collecting mineral soils from the 0 to 5 cm depth, placing them into labeled polypropylene bags.

Once back in the laboratory, physiological characteristics of the mushroom were recorded, such as gill pattern, attachment of gills to the stipe, cap and gills color, stipe characteristics, and bruising coloration. Mushroom spore prints were also generated to aid in identification (Lincoff 1981). The stipe was then removed from the mushroom, and the cap was placed spore-side down onto a labeled spore print card, placing a cup over the sample. Substrate samples were also characterized to determine if the material was decomposing leaves (forest floor) or woody material.

Mushroom and substrate digestion and analyses

After identification, mushroom samples were weighed for their fresh weight, frozen at -50 °C, and then freeze-dried for 48 h or longer to a constant mass. Mushroom samples were then re-weighed to capture the dry weight before being ground into a fine powder using a mortar and pestle for storage. The substrates and soils were placed into an oven at 50 °C for 72 h to allow the samples to dry. Substrates were then ground into a fine powder using a milling machine. The mushrooms, substrates, and soils were weighed (0.001 g scale) and ashed in a muffle furnace at 550 °C for 10—14 h to remove carbon from the samples prior to digestion. Samples were then prepared for digestions by placing ashed material into centrifuge tubes. To measure soil pH, 5.00 g of each soil and 12.5 mL 0.01 M $CaCl_2$ were added to labeled 50 mL centrifuge tubes. Samples were placed onto a shaker table for 1 h and supernatant was analyzed for pH using a calibrated VWR pH benchtop probe and meter (VWR International, Radnor, PA, USA) in triplicate.

During the digestion process, 5 mL of aqua regia (9:1 HNO_3:HCl) was added into each tube. After degassing overnight, tubes were heated to 90 °C using a tube rack heater for 1 h. Each sample was then diluted to approximately 50 g using 18.2 MΩ de-ionized water. The mass of each sample digest and dilution were recorded. Samples were then ready to be analyzed for trace elements using an Agilent

7700 × Inductively Coupled Plasma—Mass Spectrometer (ICP-MS; Agilent Technologies Inc, Santa Clara, CA, USA) and macroelement concentrations using an Agilent 5110 Inductively Coupled Plasma Optical Emission Spectrometer (ICP-OES). For the ICP-MS, 12-point calibration curves using power regressions were used. Linear regressions are overly sensitive to large values and power regressions provide models that better model the 5 orders of magnitude. Continuing Calibration Verification Standard and Initial Calibration were used every 15 samples. The limit of quantification (LOQ) for As, Cd, and Pb in diluted digests by the ICP-MS were 0.15, 0.02, and 0.08 ng/mL. For the ICP-OES, 9-point calibration curves using power regressions were used. Continuing Calibration Verification Standard and Initial Calibration were used every 15 samples. The LOQ for P, Ca, Mg, and K in diluted digests by the ICP-MS were 0.01, 0.04, 0.01, and 0.20 µg/mL. Batches of samples included standard reference materials (SRMs), procedural blanks, and sample duplicates every 30 samples for both instruments. All SRMs, Peach leaves SRM 1547, Montana soil SRM 2711, and Oyster tissue SRM 1566b, are from the National Institute of Standards and Technology (NIST, National Institute of Standards and Technology, Gaithersburg, MD, USA). Peach leaves SRM 1547 and Montana soil SRM 2711 were used as certified reference materials for As, Cd, Pb, P, Ca, Mg, and K concentrations in soil and substrate samples. For mushroom samples, Peach leaves SRM 1547 and Oyster tissue SRM 1566b were used as reference As, Cd, Pb, P, Ca, Mg, and K concentrations. Certified reference material recovery rates were 96 to 105% of the certified values for all toxic and nutrient elements. To assess Hg concentrations, the Milestone DMA-80 evo (Direct Mercury Analyzer, Milestone SRL, Sorisole, BG, Italy) was used with Peach leaves SRM 1547 were used as reference Hg values along with blank samples for mushroom, substrate, and soil samples. The LOQ for Hg was 0.05 ng/g and recovery rates were averaged 94% for certified values of SRMs.

Data visualization and data analyses

Matlab R2022b (MATLAB 9.13) was used for data analysis and production of figures including linear regressions. Statistical analyses were performed within Matlab using the Kruskal–Wallis test to assess significant differences in the data (e.g., sampling zones, mushroom genera, mushroom ecology, soils, substrates, and substrate types). The most abundant mushroom genera were *Ganoderma, Megacollybia, Pleurotus, Pluteus,* and *Russula* and thus intra-genus comparisons only focused on these five genera. ArcGIS Pro software was used to map field sites. A USA urban area layer accessed from the U.S. Census Bureau, Department of Commerce aided to visualize where sampling sites intersect with urban development.

Bioaccumulation factors (BAF) were calculated to determine the mushroom's ability to uptake both toxic and nutrient elements from their respective substrates. BAF values were calculated as follows in Eq. 1:

$$BAF = \frac{Cm}{Cs} \tag{1}$$

where, *Cm* is the concentration of a toxic or nutrient element within a mushroom sample, and *Cs* is the concentration of toxic or nutrient elements within respective substrate samples. The BAF was calculated separately for each element analyzed. BAF values > 1 indicate that a mushroom sample is bioaccumulating an element from the respective substrate.

To assess the safety for human consumption of the mushrooms sampled, Target Hazard Quotient (THQ) values were calculated for each toxic element of each mushroom sample. THQ is the ratio of the toxic element exposure to the highest oral reference dose which poses no adverse health complications to consumers. THQ values were calculated in Eq. 2:

$$THQ = \frac{Efr * ED * ADC * CE}{RfD * BW * ATn} * 10^3 \tag{2}$$

where, *Efr* represents the frequency of exposure during the summer foraging months (90 days), *ED* represents the exposure duration for a US citizen (77.28 years), ADC represents the average daily consumption of dry weight mushrooms in the US (15 g/day), *CE* represents the average concentration of each toxic element within mushroom samples (mg/kg), and *RfD* represents the oral reference dose of each respective toxic element. *BW* represents the average body weight (80 kg) and *ATn* represents the average exposure time (90 days × 77.28 years = 6955.2 days). THQ values > 1 indicate the possibility for adverse health effects after consumption. ΣTHQ refers to the addition of THQ values of each toxic element sampled (As, Cd, Hg, Pb).

Results

Mushroom genera and species collected

There were 18 different species of mushrooms collected in the early summer: *Artomyces pyxidatus* ($n = 4$), *Auricularia auricula-judae* ($n = 2$), *Butyriboletus frostii* ($n = 1$), *Cerioporus squamosus* ($n = 2$), *Ganoderma tsugae* ($n = 8$), *Grifola frondosa* ($n = 1$), *Hymenopellis radicata* ($n = 2$), *Inonotus obliquus* ($n = 1$), *Leratiomyces percevalii* ($n = 5$), *Megacollybia rodmani* ($n = 47$), *Mycena leaiana* ($n = 4$), *Neolentinus Lepideus* ($n = 1$), *Pluteus cervinus* ($n = 22$), *Polyporus alveolaris* ($n = 4$), *Russula* ($n = 3$), *Trametes versicolor* ($n = 4$), *Tylopilus alboater* ($n = 1$), and *Volvariella volvacea* ($n = 2$). Saprotrophic mushroom samples ($n = 109$) exceeded

mycorrhizal mushroom samples ($n=5$) in early summer sampling (Table 1).

There were 36 different species of mushrooms found in the late summer collection across the 20 sites: *Amanita vaginata* ($n=1$), *Artomyces pyxidatus* ($n=5$), *Aureoboletus auriporus* ($n=1$), *Auricularia auricula-judae* ($n=1$), *Boletus auripes* ($n=1$), *Cerioporus squamosus* ($n=4$), *Cyanoboletus pulverulentus* ($n=1$), *Desarmillaria caespitosa* ($n=5$), *Fomitopsis betulina* ($n=1$), *Ganoderma tsugae* ($n=3$), *Gyroporus castaneus* ($n=3$), Hymenopellis radicata ($n=4$), *Inonotus obliquus* ($n=1$), *Laetiporus sulphureus* ($n=2$), *Leccinum versipelle* ($n=1$), *Leratiomyces Percevalii* ($n=4$), *Leucopaxillus giganteus* ($n=1$), *Lycoperdon pyriforme* ($n=5$), *Megacollybia rodmani* ($n=7$), *Mycena leaiana* ($n=5$), *Neolentinus Lepideus* ($n=6$), *Pleurotus pulmonarius* ($n=7$), *Pluteus cervinus* ($n=10$), *Pseudomerulius curtisii* ($n=1$), *Russula aeruginea* ($n=2$), *Russula amoenolens* ($n=2$), *Russula Aurea* ($n=1$) *Russula brevipes* ($n=6$), *Russula claroflava* ($n=1$), *Russula compacta* ($n=2$), *Russula cyanoxantha* ($n=6$), *Russula mariae* ($n=1$), *Russula paludosa* ($n=8$), *Russula virescens* ($n=1$), *Trametes versicolor* ($n=8$), *Tremella mesenterica* ($n=5$). Saprotrophic mushroom samples ($n=89$) again exceeded mycorrhizal mushroom samples ($n=36$) in late summer sampling (Table 1).

Nutrient and toxic elements in mushrooms

Mushroom toxic elemental concentrations differed among the most abundant genera (Fig. 2; Supplemental Table 1). Mushroom tissue As concentrations were significantly higher for *Ganoderma, Megacollybia, and Pluteus* than *Pleurotus* and *Russula*. Mushroom Cd concentrations were significantly higher for *Pluteus* than *Megacollybia* and *Russula*. Mushroom Hg concentrations were significantly higher for *Megacollybia, Pluteus,* and *Russula* than *Pleurotus*. Mushroom tissue Pb concentrations exhibited no significant differences in toxic element concentrations between genera.

Mushroom nutrient concentrations also differed among the genera collected (Fig. 2). Mushroom Ca concentrations in mushroom tissues were significantly higher for *Ganoderma, Pleurotus, and Russula* than *Megacollybia* and *Pluteus*. Mushroom K concentrations in mushroom tissues were significantly higher for *Ganoderma, Pluteus, Pleurotus, and Russula* than *Megacollybia*. Mushroom Mg and P concentrations in mushroom tissues were significantly higher for

Table 1 Species of most common mushroom genera foraged ($n=5$; *Ganoderma*, *Megacollybia*, *Pluteus*, *Pleurotus*, and *Russula*) are shown per site ($n=20$) comparing early and late summer

Zone	Site #	Site Name	Latitude	Longitude	Site State	Early Summer Mushrooms	Late Summer Mushrooms
Hill	1	Agawam	42.08994	-72.64478	MA	M, R	-
Hill	2	Bear Hole	42.13097	-72.67578	MA	M	R
Hill	3	Chicopee	42.17105	-72.54037	MA	M	-
Hill	4	Bachelor	42.28238	-72.52025	MA	M	P
Hill	5	Amethyst	42.37648	-72.48612	MA	G	P
Mountain	6	Pulpit	42.73112	-72.40873	NH	M	R
Mountain	7	Pisgah	42.78991	-72.44430	NH	M	M, R, P
Mountain	8	Friedsam	42.89136	-72.47042	NH	M	M, R, P
Mountain	9	Evergreen	42.84842	-72.58893	VT	M	M, R
Mountain	10	Brattleboro	42.88165	-72.61287	VT	M	R, P
Valley	11	Northwest	41.89768	-72.70194	NoCT	G, M, P	G, M, R
Valley	12	Superfund	41.88195	-72.73129	NoCT	G, P	L, R
Valley	13	Windsor	41.81341	-72.64801	NoCT	P	P
Valley	14	Cemetery	41.72644	-72.69305	NoCT	G, P	G, L
Valley	15	Cedar Hill	41.70669	-72.70781	NoCT	P	L, M, P
Coastal	16	North Farms	41.31534	-72.76922	SoCT	M, P	L, P
Coastal	17	Branford	41.29855	-72.80487	SoCT	P	L
Coastal	18	Lighthouse	41.25212	-72.89949	SoCT	M	-
Coastal	19	Eisenhower	41.25156	-73.05564	SoCT	P, R	-
Coastal	20	Turkey	41.2863	-73.05305	SoCT	M, P, R	P

* MA = Massachusetts, NH = New Hampshire, VT = Vermont, NoCT = Northern Connecticut, SoCT = Southern Connecticut

* M = *Megacollybia rodmani*, R = *Russula*, G = *Ganoderma tsugae*, L = *Pleurotus pulmonarius*, P = *Pluteus cervinus*

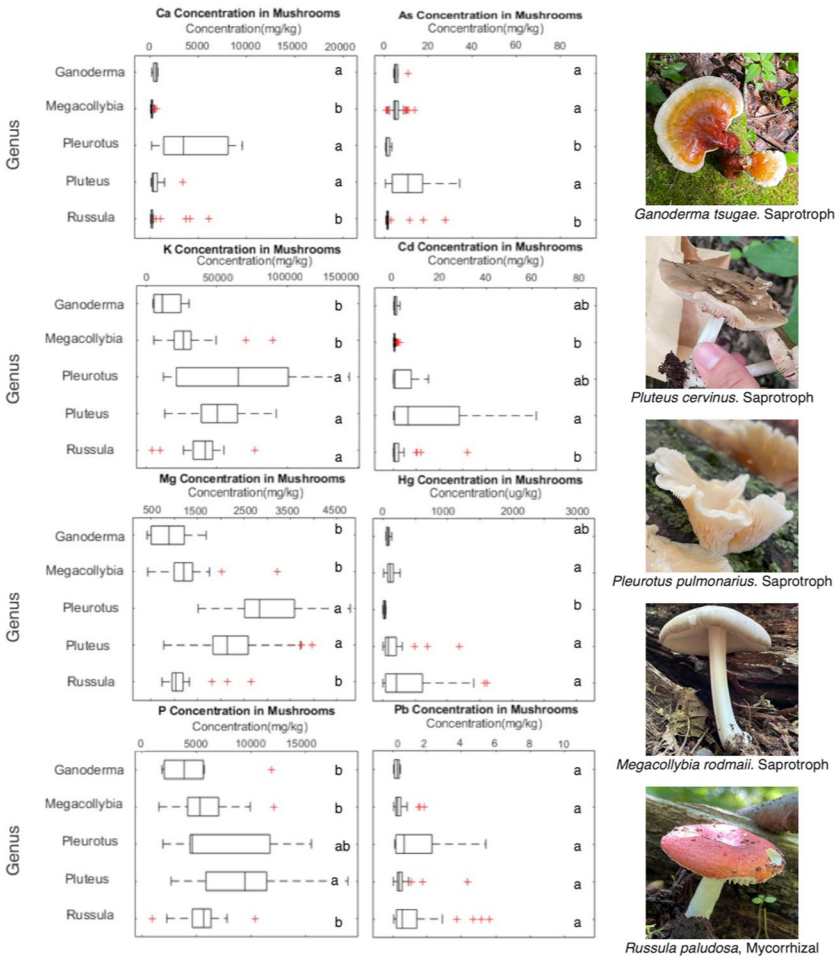

Fig. 2 Mushroom tissue dry weight concentrations by genera for toxic (As, Cd, Hg, Pb) and nutrient (Ca, K, Mg, P) elements. Significant differences ($p < 0.05$) among genera are depicted using different letters. Photos of the 5 common genera (*Ganoderma*, *Megacollybia*, *Pluteus*, *Pleurotus*, and *Russula*) are shown to the right

Pluteus and *Pleurotus* than *Megacollybia, Ganoderma,* and *Russula*.

Mushroom toxic element concentrations differed across the four zones. Mushroom As and Cd tissue concentrations were significantly higher for the Valley zone than the Coastal, Hill, and Mountain zones. Mercury concentrations in mushroom tissues were significantly higher for the Mountain zone than the Coastal, Hill, and Valley zones. Mushroom Pb tissue concentrations exhibited no significant difference ($p = 0.74$) in mushroom nutrient element concentrations between zones. Nutrient elemental concentrations in mushrooms also differed across the four zones. Calcium, Mg, and K concentrations in mushroom tissues were significantly higher for the Valley and Coastal zones than Hill and Mountain zones. Phosphorous exhibited no significant difference ($p = 0.06$) in mushroom nutrient element concentrations between zones.

Saprotrophic mushrooms had minimal significant differences to mycorrhizal mushrooms considering both nutrient and toxic element concentrations (Fig. 3). Mushroom Pb and K concentrations were significantly higher for mycorrhizal mushrooms than saprotrophic mushrooms. Conversely,

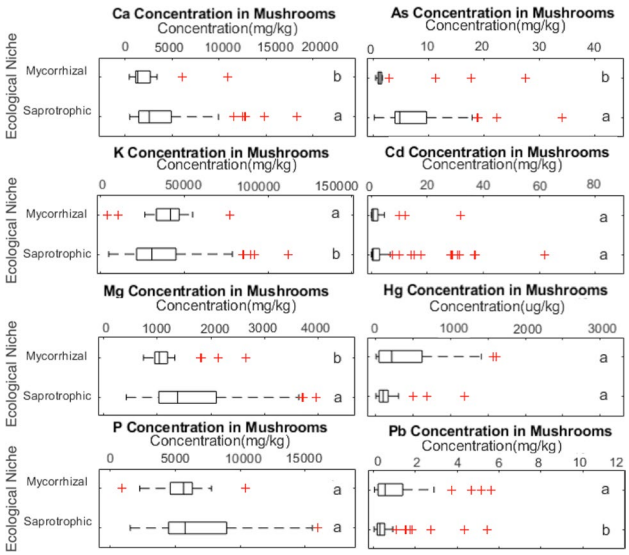

Fig. 3 Concentrations of nutrient (Ca, K, Mg, P) and toxic (As, Cd, Hg, Pb) elements between mycorrhizal vs. saprotrophic dry weight mushrooms are shown in respect to ecological niche. Significant differences ($p < 0.05$) among ecological niches are depicted using different letters

mushroom As, Ca, and Mg concentrations were significantly higher for saprotrophic mushrooms than mycorrhizal mushrooms. There were no significant differences ($p > 0.05$, $p = 0.40$, and $p = 0.22$ respectively) in Mushroom Hg, Cd, and P concentrations between saprotrophic and mycorrhizal mushrooms.

Mineral soil properties

Soil pH regarding mineral soils differed among the four zones. Mountain zone soils ($n = 25$) had an average soil pH of 3.22 ± 0.35. Valley zone soils ($n = 25$) had an average soil pH of 4.18 ± 0.98. Coastal zone soils ($n = 25$) had an average soil pH of 4.47 ± 0.60. Valley and Coastal zone soil pH values were significantly higher than Mountain and Hill zones. Loss on Ignition (LOI) of mineral soils differed among the four zones. Mountain zone soils ($n = 25$) had an average LOI of $20.9\% \pm 6.6\%$. Hill zone soils ($n = 25$) had an average LOI of $16.5\% \pm 14.8\%$. Valley zone soils ($n = 25$) had an average LOI of $15.6\% \pm 12.0\%$. Coastal zone soils ($n = 25$) had an average LOI of $25.1\% \pm 18.7\%$.

For mineral soils, As concentrations were significantly higher for the Mountain zone than the Hill zone (Supplemental Table 2). Mineral soil Cd and Hg concentrations were significantly higher for the Coastal and Valley zones than the Hill zone. Mineral soil Ca and K concentrations in soils were significantly higher for the Valley and Coastal zones than Hill zone. Mineral soil P and Mg concentrations in mineral soils were significantly higher for the Coastal zone than the Hill, Mountain, and Valley zones. Mineral soil Pb exhibited no significant difference ($p = 0.48$) among zones.

Substrate properties

All mycorrhizal mushrooms sampled ($n = 30$) were collected from forest floor (FF) substrates, compared to the saprotrophic mushrooms sampled ($n = 98$) which were collected on both woody debris (WD; $n = 55$) and FF ($n = 43$). There were some significant differences in toxic and nutrient element concentrations between FF and WD substrates. However, substrate Hg and Pb concentrations were significantly higher for FF than WD. Substrate Mg and P concentrations were significantly higher for FF than WD substrates. Conversely, Ca concentrations were significantly lower for FF than WD substrates. Substrate As, Cd, and K concentrations exhibited no significant difference ($p = 0.22$, $p = 0.06$, and $p > 0.05$ respectively).

Substrate toxic element concentrations differed across zones (Supplemental Table 3). Substrate As and Cd concentrations were significantly higher for the Hill zone than the Coastal and Mountain zones. Substrate Hg and Pb concentrations in substrates were significantly higher for the Mountain zone than the Valley and Coastal zones. Substrate Ca concentrations were significantly higher for the Hill and Valley zones than the Coastal and Mountain zones. Substrate P concentrations were significantly higher for the Hill and Mountain zones than the Coastal and Valley zones. Substrate K and Mg concentrations exhibited no significant difference ($p = 0.90$ and $p = 0.51$ respectively) between zones.

Relationships between mushrooms, substrates, and mineral soils

Linear regressions were performed to assess if toxic metal and nutrient concentrations in substrates or mineral soils were associated with mushroom uptake, either through availability driving uptake or if nutrient availability suppressed toxic metal uptake (Fig. 4). Starting with the substrate-mushroom relationship, linear regressions show that mushroom As, Pb, and Hg concentrations were poorly correlated with their respective substrate concentrations, with $R^2 < 0.14$ for linear regressions. However, mushroom Cd was significantly correlated with substrate Cd, with $R^2 = 0.30$ ($p < 0.01$). For nutrients, mushroom concentrations of Ca, K, Mg, and P were poorly correlated with their respective substrate concentrations with $R^2 < 0.13$ for linear regressions. Comparing mineral soil samples with mushroom concentrations, there were no significant correlations between soil and mushroom concentrations for As, Cd, Hg, and Pb based on linear regression models ($p > 0.05$). When considering nutrients, only K had a significant mushroom to mineral soil relationships ($p = 0.05$, $R^2 = 0.28$).

Lastly, linear regressions were performed to assess if mineral soil toxic element or nutrient concentrations were related among the 20 sites (Fig. 4). Mineral soil As was significantly correlated to Pb ($p < 0.01$, $R^2 = 0.08$), Cd ($p < 0.01$, $R^2 = 0.10$), Hg ($p < 0.01$, $R^2 = 0.20$), and P ($p < 0.01$, $R^2 = 0.01$) concentrations in soil. Mineral soil Pb was significantly correlated to mineral soil Hg ($p < 0.01$, $R^2 = 0.8$) and mineral soil P ($p < 0.01$, $R^2 = 0.1$) concentrations in soil. Mineral soil Cd was significantly correlated to mineral soil As ($p < 0.01$, $R^2 = 0.1$), mineral soil Hg ($p < 0.01$, $R^2 = 0.01$), and mineral soil P ($p < 0.01$, $R^2 = 0.01$) concentrations in soil. Mineral soil Mg was significantly correlated to mineral soil Ca ($p < 0.01$, $R^2 = 0.01$), mineral soil K ($p < 0.01$, $R^2 = 0.7$) and mineral soil P ($p < 0.01$, $R^2 = 0.01$) concentrations in soil. All regression slopes were positive indicating that in each elemental relationship both elements had increasing concentrations as the other element increased.

Lastly, linear regressions were performed to assess if nutrient availability in substrates and mineral soils suppressed toxic element uptake by mushrooms. Mineral soil P was positively correlated with Cd ($p < 0.05$, $R^2 = 0.06$) concentrations in mushrooms. Substrate Ca was positively correlated with As ($p < 0.05$, $R^2 = 0.03$) concentrations in

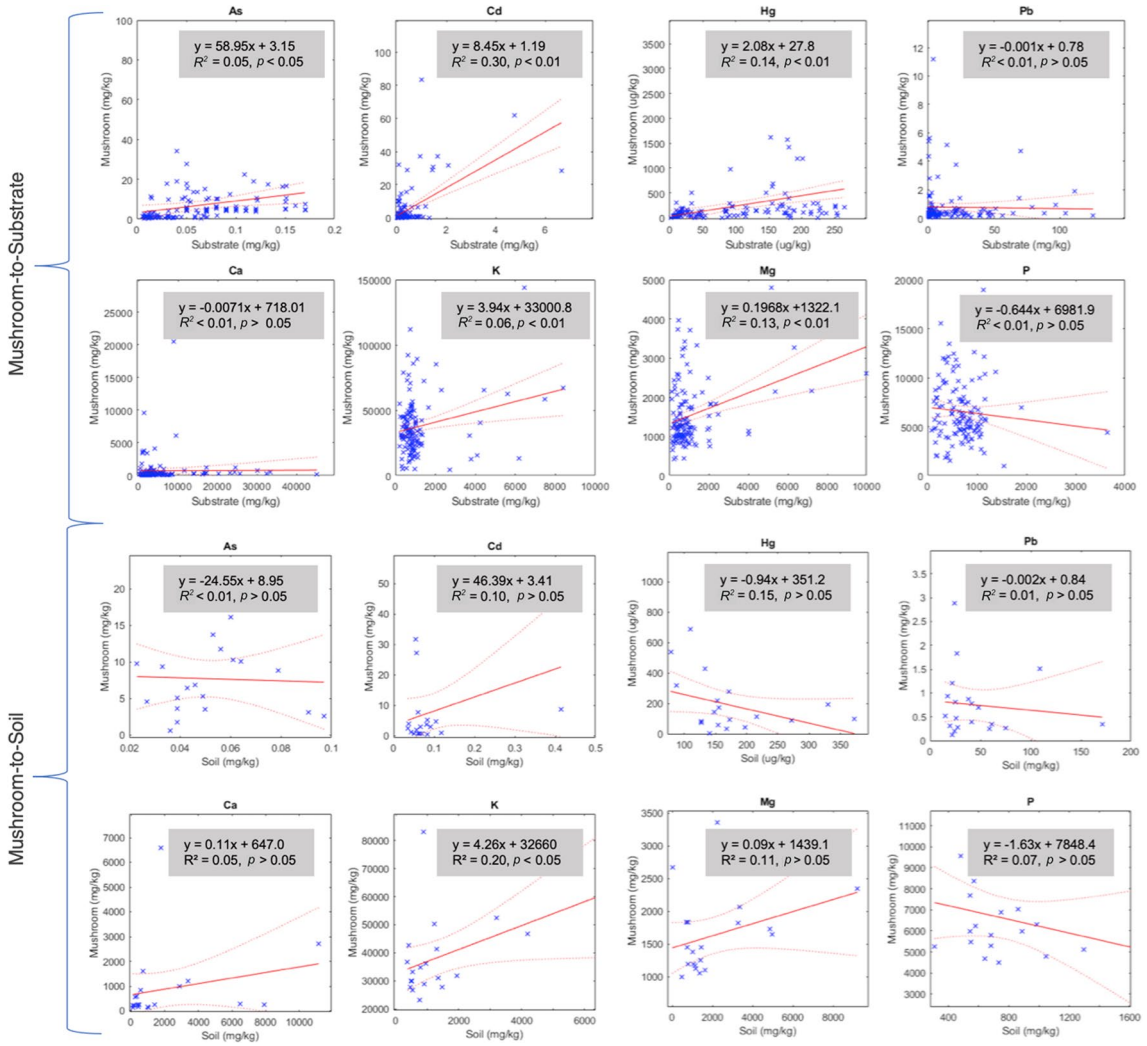

Fig. 4 Correlations between mushroom-to-substrate and mushrooms-to-soil for nutrient (Ca, K, Mg, P) and toxic element (As, Cd, Hg, Pb) concentrations

mushrooms. Substrate P was negatively correlated to As ($p < 0.01$, $R^2 = 0.06$) and Cd ($p < 0.05$, $R^2 = 0.03$) concentrations in mushrooms, both with negative slopes.

Bioaccumulation factors for mushrooms

BAF values for mushrooms were compared between genera (Table 2). Mushroom Cd BAF values were significantly higher for *Pluteus* and *Russula* than *Megacollybia*. Mushroom As and Hg BAF values were significantly higher for *Pluteus* than *Megacollybia* and *Russula*. Mushroom Pb BAF values were significantly higher for *Pleurotus* and *Pluteus* than *Megacollybia*. When considering nutrients, Ca BAF values in mushroom tissues were significantly higher for *Pleurotus* than *Megacollybia*. Mushroom tissue K BAF values were significantly higher for *Pluteus* and *Russula* than *Ganoderma, Megacollybia,* and *Pleurotus*. Mushroom P BAF values were significantly higher for *Pluteus* than *Megacollybia* and *Russula*. Mushroom Mg BAF values exhibited no significant difference ($p = 0.12$) in BAF values between genera.

Comparing mycorrhizal and saprotrophic mushrooms, mushroom Cd BAF values were significantly higher for mycorrhizal than saprotrophic mushrooms. Mushroom As, Hg, and Pb BAF values exhibited no significant differences ($p = 0.31$, $p = 0.13$, and $p = 0.96$, respectively) between the two ecological niches. Mushroom K BAF values were significantly higher for mycorrhizal than saprotrophic mushrooms. Conversely, P BAF values were significantly lower for mycorrhizal than saprotrophic mushrooms. Mushroom Ca and Mg BAF values exhibited no significant difference ($p = 0.10$ and $p = 0.23$ respectively) in nutrient element BAF values between ecological niches.

Table 2 BAF values (mean ± standard deviation) among the 5 most commonly foraged mushroom genera (*Ganoderma*, *Megacollybia*, *Pluteus*, *Pleurotus*, and *Russula*), saprotrophs vs. mycorrhizal, and between zones for nutrient (Ca, K, Mg, P) and toxic elements (As, Cd, Hg, Pb)

	As mg/kg	Cd mg/kg	Hg ug/kg	Pb mg/kg	Ca mg/kg	Mg mg/kg	K mg/kg	P mg/kg
Ganoderma	103 ± 25	6.3 ± 5.0	4.3 ± 2.2	0.2 ± 0.2	0.2 ± 0.1	2.9 ± 1.6	26 ± 26	18 ± 13
Megacollybia	72 ± 61	2.7 ± 3.3	2.0 ± 2.2	0.2 ± 0.4	0.1 ± 0.1	2.1 ± 1.5	39 ± 25	12 ± 13
Pleurotus	110 ± 126	42 ± 81	3.5 ± 2.4	2.3 ± 4.2	2.3 ± 2.2	1.6 ± 1.5	31 ± 52	17 ± 20
Pluteus	237 ± 297	22 ± 26	5.9 ± 6.1	1.0 ± 3.5	0.3 ± 0.7	4.4 ± 3.9	70 ± 45	22 ± 11
Russula	125 ± 188	32 ± 76	3.2 ± 4.1	0.5 ± 1.4	0.3 ± 0.5	1.9 ± 1.1	67 ± 49	7.1 ± 3.0
Saprotrophs	129 ± 189	12 ± 27	3.5 ± 4.2	0.6 ± 2.3	0.3 ± 0.9	2.8 ± 2.7	47 ± 38	16 ± 14
Mycorrhizal	125 ± 188	32 ± 76	3.2 ± 4.1	0.5 ± 1.4	0.3 ± 0.5	1.9 ± 1.1	67 ± 49	7.1 ± 3.0
Mountain Zone	56 ± 42	6.9 ± 14	2.0 ± 2.6	0.3 ± 1.0	0.1 ± 0.2	2.1 ± 1.5	48 ± 36.8	8.0 ± 5.3
Hill Zone	82 ± 129	5.5 ± 8.7	3.7 ± 3.8	0.1 ± 0.1	0.1 ± 0.1	2.0 ± 1.7	43 ± 32	12 ± 10
Valley Zone	122 ± 98	22 ± 32	3.9 ± 3.0	0.4 ± 0.7	0.3 ± 0.7	3.4 ± 3.1	59 ± 50	21 ± 15
Coastal Zone	158 ± 105	13 ± 13	5.4 ± 4.2	1.5 ± 3.1	1.3 ± 1.9	3.7 ± 3.8	60 ± 44	24 ± 17.8
All Mushrooms	128 ± 188	17 ± 45	3.5 ± 4.1	0.6 ± 2.1	0.3 ± 0.8	2.6 ± 2.4	52 ± 41	14 ± 13

Table 3 THQ values (mean ± standard deviation) among the 5 most commonly foraged mushroom genera (*Ganoderma*, *Megacollybia*, *Pluteus*, *Pleurotus*, and *Russula*), ecological niche (saprotroph or mycorrhizal), and zones

	As THQ	Cd THQ	Hg THQ	Pb THQ	Σ THQ
Ganoderma	0.2 ± 0.1	0.2 ± 0.2	0.01 ± 0.01	0.1 ± 0.0	0.5 ± 0.3
Megacollybia	0.2 ± 0.1	0.1 ± 0.1	0.02 ± 0.02	0.1 ± 0.0	0.3 ± 0.2
Pleurotus	0.1 ± 0.0	0.1 ± 0.1	0.02 ± 0.02	0.0 ± 0.0	0.2 ± 0.1
Pluteus	0.6 ± 0.5	2.8 ± 2.5	0.03 ± 0.02	0.1 ± 0.1	3.5 ± 2.6
Russula	0.1 ± 0.0	0.2 ± 0.2	0.07 ± 0.11	0.3 ± 0.4	0.5 ± 0.5
Saprotrophs	0.3 ± 0.5	1.0 ± 2.4	0.02 ± 0.02	0.1 ± 0.1	1.4 ± 2.6
Mycorrhizal	0.1 ± 0.0	0.2 ± 0.2	0.07 ± 0.11	0.3 ± 0.4	0.5 ± 0.5
Mountain Zone	0.1 ± 0.1	0.2 ± 0.2	0.04 ± 0.09	0.2 ± 0.3	0.5 ± 0.4
Hill Zone	0.2 ± 0.1	0.2 ± 0.2	0.02 ± 0.04	0.1 ± 0.1	0.5 ± 0.2
Valley Zone	0.4 ± 0.6	2.2 ± 3.7	0.02 ± 0.02	0.1 ± 0.1	2.6 ± 3.7
Coastal Zone	0.5 ± 0.9	1.3 ± 2.1	0.04 ± 0.05	0.1 ± 0.1	1.7 ± 2.3
All Mushrooms	0.3 ± 0.5	0.8 ± 2.2	0.03 ± 0.06	0.1 ± 0.2	1.2 ± 2.3

Lastly, we considered differences in mushroom BAF values among zones. We found As, Hg, and Pb BAF values were significantly higher for the Valley and Coastal zones than the Mountain zone. Mushroom BAF values for Cd were significantly higher for the Valley zone than the Mountain Zone.

Target hazard quotient (THQ) for human consumption

THQ values for As did not exceed one across genera, ecological niches, and zones (Table 3). THQ values for As were significantly higher for *Ganoderma*, *Megacollybia*, and *Pluteus* than *Russula* and *Pleurotus*. THQ values for Cd were > 1 for the genus *Pluteus*, but not the other genera. Moreover, Cd THQ values were > 1 for saprotrophs as well as the Valley and Coastal zones. THQ values for Hg were not > 1 for any genera, ecological niche, or sampling zone but Hg THQ were significantly higher for *Megacollybia, Pluteus,* and *Russula* than *Pleurotus.* THQ values for Pb were not > 1 for any genera, ecological niche, or sampling zone but Pb THQ were significantly higher for *Russula.* When considering ΣTHQ values of As, Cd, Hg, and Pb, only the genus *Pluteus* ΣTHQ values > 1 while the other genera were < 1. Furthermore, ΣTHQ values > 1 for saprotrophic mushrooms but not mycorrhizal mushrooms and ΣTHQ values > 1 for mushrooms from the Valley and Coastal Zones but not the Mountain and Hill zones.

Discussion

Nutrient and toxic elements in foraged mushrooms

The first goal of this study was to determine if nutrient and toxic element concentrations, bioaccumulation factor (BAF), and target hazard quotient (THQ) in foraged mushrooms varied along the urbanization gradient or due to differences in mineral soil concentrations. Despite background levels of soil contamination having no major differences between the zones, higher BAF and THQ values were observed in both the Valley and Coastal zones. Differences in enzyme production and release among the genera analyzed may have affected nutrient and toxic element uptake, potentially diminishing the relationship between substrate concentrations with mushroom concentrations. Additionally, Gebrelibanos et al. (2016) recognizes that the substrate-to-mycelium transfer may be affected by the relationship of mycelium to symbiotic plant species which can directly affect element absorption. *Russula* was the only mycorrhizal mushroom out of the most common genera within our study, thus only

offering concern of this phenomena for one of the five genera.

Average toxic element concentrations for As and Cd in our study fall within the range of values from mushroom foraging literature conducted in other countries (Table 4). However, Hg and Pb concentrations were lower in the present study compared with similar surveys conducted in China (Fu et al. 2020; Chen et al. 2009), Czech Republic (Svoboda et al. 2005), Turkey (Cayir et al. 2009; Isildak et al. 2007; Turkekul et al. 2004; Tuzen et al. 2007; Yamaç et al. 2007), Poland (Podlasińska et al. 2015), and Greece (Ouzouni et al. 2009; Kokkoris et al. 2019). Podlasińska et al. (2015) reported that Hg was accumulated in the highest level by *Pluteus cervinus* at an average concentration of 0.38 mg/kg, compared to the other 7 genera they collected. Our study found average *Pluteus cervinus* Hg concentrations of 0.18 mg/kg, thus showing that *Pluteus cervinus* mushrooms collected in our study have lower Hg concentrations than they did in Poland. The lower toxic element concentrations in our study likely result from improvements made by policies diminishing pollution releases in the United States (e.g., the Clean Air Act, Clean Water Act, etc.) and lack of intensive mineral mining within the New England region. In addition, our study was conducted in state forests away from point-source polluters, decreasing the possibility for direct exposure to substantially toxic element concentrations.

Average nutrient element Ca and P concentrations from our study fall within the range in values from mushroom foraging literature from other countries (Table 4). However, average K and Mg concentrations were higher in our present study than studies in China (Chen et al. 2009) and Poland (Pająk et al. 2020; Malinowski et al. 2021). These findings were surprising as historical acid rain, deforestation, and current tree species shifts have caused losses of nutrients from the forest ecosystems (Vitousek et al. 1979; Cincotta et al. 2019). However, deforestation and increased woody biomass deposition from historical clear-cutting may have increased availability of soil nutrient pools, or acidification can increase soil nutrient release rates at our study sites.

Our study found that as elemental concentrations varied across the zones, there were significant influences on mushroom BAFs and THQs. Overall, mushroom As, Cd, Ca, and Mg concentrations were greater in the Valley and Coastal zones. The Valley and Coastal zones are closer to greater historical and modern industrial processing. As shown by Richardson (2020), central Connecticut forest soils in and near Hartford CT had nearly double the As, Cd, and Ni concentrations than western Massachusetts forest soils in and near Springfield MA. Thus, the higher levels of urbanization and proximity to air pollution likely enriched the mushrooms with higher concentrations of As, Hg, Cd, and Pb. Additionally, Liu et al. (2015) sampled wild edible mushrooms from Yunnan Province, China, which is heavily polluted by industry and exploitation of raw materials such as coal, natural gas, nonferrous metals, and more. The mushrooms Liu et al. (2015) sampled had As and Cd concentrations that exceeded safe limits within China and had the highest bioaccumulation values among the toxic metals they tested, further showing that in areas with more industrial processes result in mushrooms with high As and Cd concentrations.

THQs varied significantly among mushroom genera and metals, yielding a somewhat complex result at the site and zone level. When THQ were summed among toxic elements (As, Cd, Hg, Pb), sites within the Valley and

Table 4 Mean and range toxic element concentrations (As, Cd, Hg, Pb; mg/kg) and nutrient element concentrations (Ca, K, Mg, P; g/kg) of mushrooms are compared across studies

	As		Cd		Hg		Pb	
	mg/kg		mg/kg		mg/kg		mg/kg	
	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Ouzouni et al. (2009)	-	-	0.23	0.08–0.41	-	-	0.81	0.05–1.37
Soylak et al. (2007)	-	-	0.66	0.14–0.95	-	-	1.28	0.75–1.99
Yamaç et al. (2007)	-	-	0.99	0.26–3.24	-	-	2.39	0.3–11.7
Isildak et al. (2007)	-	-	1.33	0.3–3	-	-	2.57	2.1–3.5
Cayir et al. (2009)	-	-	0.72	0.18–4.23	-	-	1.53	0.59–3.05
Fu et al. (2020)	6.75	0.16–34.5	3.96	0.21–21	1.07	0.07–3.59	1.09	0.21–3.56
Svoboda et al. (2005)	-	-	13.44	0.04–166	2.24	0.03–22.4	6.88	0.1–37.6
Present Study (2023)	6.98	0.24–91.7	4.93	0.05–83	0.23	0.002–3.12	0.76	0.04–11.2
	Ca		K		Mg		P	
	g/kg		g/kg		g/kg		g/kg	
	Mean	Range	Mean	Range	Mean	Range	Mean	Range
Pająk et al. (2020)	0.48	0.1–1.5	36.4	21.8–62.4	0.8	0.4–1.4	5.7	2.8–9.1
Malinowski et al. (2021)	0.14	0.10–0.20	6.4	0.69–12.1	1.1	0.68–2.20	8.3	6.9–10.0
Chen et al. (2009)	2.66	0.85–3.83	22.2	12–34	1.2	0.57–1.90	5.2	3.4–8.0
Present Study (2023)	0.73	0.05–20.55	37.5	4.22–144	1.5	0.42–4.79	6.5	0.9–18.9

Coastal zones had average ΣTHQ values > 1.0, which is considered generally unsafe for consumption. Moreover, the Mountain zone had two sites with ΣTHQ values > 1.0. At a coarse view, this suggests that mushroom foraging within the Mountain, Valley, and Coastal zones can yield mushrooms that are unsafe but there are important caveats in this finding. First, THQ for Cd was mainly responsible for this trend while As, Hg, and Pb THQ values were < 0.5. A second key aspect is that the *Pluteus cervinus* and *Pleurotus pulmonarius* mushrooms influenced the unsafe THQ values. *Pluteus cervinus* and *Pleurotus pulmonarius* were more abundant in the Valley and Coastal zones and had significantly higher As, Cd, and Hg concentrations. Thus, consumption of other mushrooms with THQ values < 1.0 from the Valley and Coastal zones can be done to minimize toxic element exposure, particularly by consuming *Russula* or *Megacollybia* for food or medicinal uses of *Ganoderma*. Alternatively, decreasing frequency or mass of mushrooms consumed can alter the non-carcinogenic hazard since THQ calculations assume a certain average daily intake and frequency. Here, we utilized the assumption of exposure frequency of the entire summer of foraging (90 days) and daily consumption rate of 15 g of mushroom dry mass. THQ values may be much higher for individuals consuming more mushrooms or THQ values may be much lower for individuals foraging for shorter intervals throughout the summer season (Fig. 5).

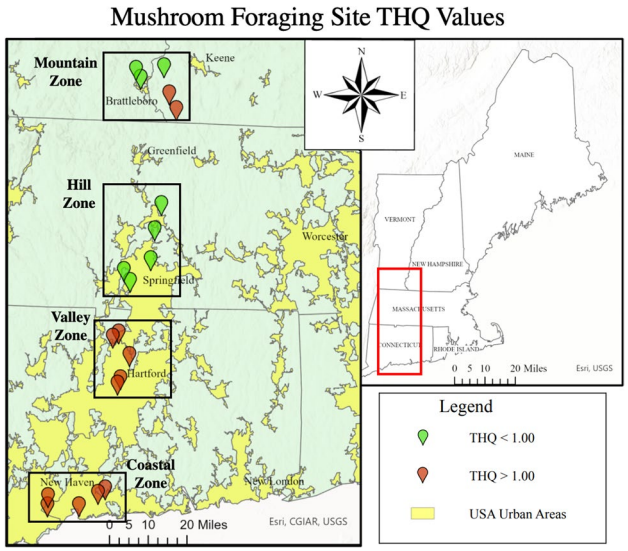

Fig. 5 ΣTHQ values (As THQ + Cd THQ + Hg THQ + Pb THQ) averaged across 5 most common genera (*Ganoderma*, *Megacollybia*, *Pluteus*, *Pleurotus*, and *Russula*) foraged at research sites ($n = 20$) are shown with green indicating low risk, and red indicating high risk to human health by consumption

Substrates effects on mushroom elemental concentrations

From our second goal, we initially expected that mushroom elemental concentration could be explained by soil or substrate concentrations. Mineral soils were not well-correlated with mushroom concentrations suggesting differences in geology and soil parent materials were unlikely driving mushroom concentrations. As further evidence, the Valley and Coastal zones had significantly higher Cd, Ca, and Mg in their mineral soil than the Hill and Mountain zones, but this was not reflected in the mushrooms. Interestingly, mineral soil As concentrations were greater in the Hill and Mountain zones yet mushroom As concentrations were significantly lower for the Valley and Coastal Zones. Therefore, our results do not support this causal relationship for toxic elements between soils and mushrooms. This lack of relationship between mushroom and mineral soil concentrations was consistent for nutrients Ca, Mg, and P as well. Only the nutrient K had a significant mushroom to mineral soil relationship based on linear regression models. This may be that K limitations at the forest ecosystem level may be reflected in the plant and microorganism levels and agrees with findings by Haro and Benito (2019) that fungi are important for sourcing K both as symbionts and decomposers.

Alternatively, it was hypothesized that the substrates fungi were directly consuming may strongly influence their mushroom elemental concentrations. Toxic elements Hg, Cd, and As had significant positive relationships between mushroom-to-substrate concentrations with R^2 values between 0.13 to 0.39 from linear regression models. Further, nutrient elements K and Mg had significant positive mushroom-to-substrate relationships with R^2 values of 0.28 and 0.35, respectively. These results show that substrate, not mineral soil concentrations, are better predictors for toxic and nutrient elements in mushrooms. This is due to the fungal hyphae directly consuming and obtaining elements from their food source while toxic and nutrient elements in the mineral soils are separated from mushrooms by the action of trees. The discriminant uptake of nutrients and toxic elements by trees controls element abundance in woody debris (WD) and the forest floor (FF), which alters their availability for uptake by mushroom. WD and FF had a few significant differences, with higher Hg, Pb, Mg, and P in the forest floor than WD. This finding agrees with results from Richardson and Friedland (2016) with trees bioaccumulating nutrients and the forest floor retaining pollutants like Pb. The higher nutrient concentrations in the forest floor are likely due to the lower proportion of lignin, cellulose, and other carbohydrate-rich, metal-poor substances in woody debris (McClaugherty and Berg 1987). Thus, mushrooms consuming WD are exposed to proportionally lower nutrients and

toxic elements but factors other than their growth substrate are likely more important for wild foraged mushrooms.

Mycorrhizal vs saprotrophic fungi and differences among genera

The third goal was to compare elemental concentrations, BAF, and THQ among genera and between mycorrhizal versus saprotrophs, with the expectation of higher elemental concentrations by mycorrhizal fungi. Out of the five different mushroom genera analyzed, *Russula* were the only mycorrhizal fungi compared with the four saprotrophic genera. Toxic element concentrations were mixed across the genera, with only *Pluteus* having the highest concentrations or not significantly lower than other genera. One potential explaination is that *Pluteus* was more abundant at sites with higher toxic element concentrations likely by chance. The higher abundance of *Pluteus* at Coastal and Valley zones were sites with higher concentrations of substrate and mineral soil Pb and Cd. An additional effect is that saprotrophic mushrooms consuming WD with lower toxic metals may cloud this comparison as well. This was true for *Ganoderma,* which was exclusively found on decomposing Eastern Hemlock snags and had the lowest toxic metal concentrations. The WD effect was not consistent for *Pluteus* and *Pleurotus*, as they were also commonly found on decomposing hardwood logs. This could suggest that softwood substrates (e.g., *Tsuga canadensis*) absorb less toxic metal concentrations than hardwoods (e.g., *Betula, Fraxinus, Acer,* etc.). However, it may likely be an ecophysiological specific attribute of *Ganoderma* as it is has a variety of detoxification routes to enhance heavy metal resistance (e.g., binding to fungus cell walls, complexation, vaporization, and compartmentalization; Sharma and Kumar 2021).

Ultimately, we did not observe a significant difference in toxic and nutrient element concentrations between saprotrophic and mycorrhizal. Instead, we did observe significant differences among specific genera. The saprotrophic mushroom *Pluteus* consistently had higher toxic and nutrient element concentrations, BAF, and THQ values compared with *Russula,* a mycorrhizal mushroom. In addition, THQ values were statistically highest for *Pluteus* for elements Pb and Hg. However, as this was only observed for one saprotrophic mushroom, we cannot generalize to saprotrophs at large and highlights the intrinsic differences among mushroom genera and likely species (e.g., Li et al 2023). Our data suggests that another factor may be important, such as ecophysiology, anatomy, or preference for WD with higher element concentrations (e.g., *Fagus sylvatica*; Pecina et al. 2022).

Conclusions

In summary, our findings suggest that urban areas of the Coastal and Valley areas of Connecticut in New England can be safe for foraging for mushrooms, if avoiding *Pluteus cervinus* due to their Cd THQ and subsequent ΣTHQ values being > 1. It should be noted that THQ values were based upon 90 days of foraging and consumption of 15 g dry weight per day, and lower consumption rates and avoiding *Pluteus cervinus* can allow for safe diets of mushrooms even in urban areas. Further, our research showed that substrate concentrations were moderate to weak predictors for nutrient and toxic elements in mushrooms. Substrates being consumed (WD and FF materials) were more important than bulk mineral soils at a location for predicting nutrient and toxic elements in mushrooms. This is likely due to plants and trees acting as intermediators. Toxic and nutrient element BAF values were statistically higher for *Pluteus* for most elements, suggesting it may be the most effective bioaccumulator out of the five genera collected or by chance was more abundant at the more polluted sites studied. *Russula* had elevated Hg concentrations but was not bioaccumulating Hg based upon the < 1 BAF value showing site specific effects. Fortunately, *Russula* had THQ values < 1 for all toxic metals. With diverse ecotones in the United States with complex land use histories, it is essential that more research is conducted in the different regions of the United States in order to identify safe and unsafe mushroom foraging areas.

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Authors contributions Marissa L. Hanley led conceptualization, field and laboratory methodology, manuscript writing, data visualization, statistical analyses, and manuscript editing for the study. Eric Vukicevich contributed to sample collection, mushroom identification, field methodology, conceptualization of experimental design, manuscript writing, and manuscript editing. Alexandrea Rice contributed to sample collection, mushroom identification, field methodology, experimental design, manuscript writing, and manuscript editing. Justin B. Richardson contributed through project conceptualization, field and laboratory methodology, project administration, manuscript writing, data visualization, statistical analyses, manuscript writing, and manuscript editing.

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Data Availability Mushroom, substrate, and soil data are available in the supplementary materials.

Declarations

Ethical approval Not Applicable.

Consent to participate and publish Not Applicable.

Competing interests The authors declare no competing interests.

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