



# A cross-regional ecological assessment of contaminants in soils intended for agri-food initiatives on Indigenous Peoples' lands in Australia and Canada

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**Abstract** Globally, soil contamination threatens ecosystems and food security. This study examines the contamination of soils intended for agri-food initiatives by Indigenous communities across New South Wales and Queensland, Australia, and Newfoundland and subarctic Ontario, Canada. Soils from 47 sites were tested for metals, metalloids, organochlorine (OC) pesticides, and polychlorinated biphenyls (PCBs) to assess ecological risks and compare against national guidelines. Australian soils were

primarily contaminated with lead (Pb) and to a lesser extent with other metals and metalloids, whereas subarctic Ontario soils were heavily contaminated with OC pesticides, and to a lesser extent metals and metalloids. Newfoundland soils contained arsenic (As) concentrations exceeding agricultural soil guidelines with limited OC levels. The contaminants from these sites stem from both anthropogenic activities and natural geological sources; however, their precise origins—whether from the historical use of banned substances or mineral extraction—are not fully elucidated for all sites. This article specifically highlights the need to assess soil to be used in agri-food initiatives for contaminants in rural and remote landscapes in Indigenous homelands worldwide and, in general, for all soil to be used in any agri-food initiative.

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## Introduction

Agricultural lands are used for activities such as cultivating crops or raising livestock, providing habitat for local and migrating wildlife, and a natural environment for local plant species (Canadian Council of Ministers of the Environment [CCME], 2006). The risk of contamination by toxicants—such as chemical fertilizers, pesticides, industrial runoff, and

hazardous waste—in soil used for agricultural activities is a significant global concern (Arao et al., 2010; Bondareva & Fedorova, 2021; Plaza-Bolaños et al., 2012), especially in local-scale agri-food initiatives. These initiatives aim to enhance community development and sustainability through local food production to improve food security (Sonnino, 2013). Over time, contaminants can accumulate in the soil, compromising its quality and safety and posing risks to human health through direct ingestion or consumption of contaminated crops (Mishra et al., 2021; Stefan et al., 2018). In urban agri-food initiatives, the primary sources of agricultural soil contamination are industrial activities and incorrect waste disposal (Laidlaw et al., 2018; Paltseva et al., 2022; Tume et al., 2019). Additionally, in urban locations, it is relatively easier and less costly to remediate contaminated soil to acceptable concentrations—or to remove the contaminants—compared to rural and remote locations, due to greater accessibility of resources such as topsoil and heavy equipment (Dixon et al., 2006; Klaus & Kiehl, 2021).

Soil contamination in rural and remote regions may arise from both external and local sources. External sources may include long-range atmospheric transport of natural or liberated (e.g., metals) and historically used contaminants (e.g., organochlorine (OC) pesticides), while local sources include the seepage of contaminants from abandoned industrial sites (e.g., polychlorinated biphenyls (PCBs)) and naturally occurring metals (Sanscartier et al., 2010; Tsuji et al., 2006; Tzanetou & Karasali, 2022; Yu et al., 2015). Thus, soils intended for agri-food initiatives in rural and remote areas are not necessarily less contaminated than those in urban regions and can still pose significant threats to the ecosystem or human health (Reyes et al., 2015; Tsuji et al., 2005).

Indigenous Peoples frequently reside in rural and remote regions, where soil contamination remains overlooked despite their reliance on the land for small-scale agricultural activities. These regions offer a unique opportunity to assess soil contamination levels (Niu et al., 2018), as they are often assumed to be less impacted and more pristine due to their remote nature. Furthermore, communities in these regions have increasingly focused on cultivating and consuming local foods (e.g., community gardens, food forests) as part of climate change adaptation strategies (Tsuji et al., 2019). However,

when soil contaminants pose a risk to food safety or ecosystem health (Shafiee et al., 2022; Tsuji et al., 2019; Weiler et al., 2015; Wilton et al., 2023), the benefits of agri-food initiatives may be undermined. The legacy of colonialization presents a double-edged sword for some Indigenous communities: not only has it forced them onto less viable or degraded lands with poor soil quality through restrictive land governance (Domínguez & Luoma, 2020; Nightingale & Richmond, 2022), but these lands have also been further compromised by soil contamination from practices like historical pesticide application, making agricultural initiatives even more challenging. This compounded legacy of limited access to viable lands and ongoing soil contamination necessitates environmental evaluations.

Ecological assessments serve an important role in identifying environmental contaminants that impact the environment (Jensen et al., 2001; Luo et al., 2010). These assessments aim to evaluate pollutants and stressors that could impact the ecosystem health, biodiversity, and long-term sustainability of the environment. By offering a comprehensive understanding of local environmental pollutants, these assessments play a crucial role in guiding effective management and remediation strategies. Unlike human health risk assessments, which focus on the effects of contaminants on human populations, ecological assessments concentrate on the risks posed to ecosystems and environmental resources.

The objective of this exploratory study was, therefore, to investigate contamination concentrations in soils intended for agricultural use by Indigenous Peoples aiming to enhance food security in their communities. We conducted a cross-regional ecological assessment comparing soil contamination in Australia and Canada, focusing on how geographical, geological, and non-agricultural land use histories influence regional ecological risk. The primary focus was identifying the key differences in soil contamination between the regions. By examining the distribution and pollution indices of metals and metalloids, OC pesticide residues, and PCBs from sites located in rural and remote Indigenous communities, this study informs strategies for cultivating safe and reliable food sources in soil intended for agri-food initiatives. Likewise, sharing these results provides valuable insights for those involved in self-guided food

security initiatives with Indigenous, rural, and remote communities.

**Methods**

*Study regions*

Soil sampling was carried out in seven rural or remote Indigenous communities located in the subtropics of New South Wales and the tropics of Queensland, Australia, and temperate marine Newfoundland, and sub-arctic Ontario, Canada, between 2010 and 2022. To protect the identity of participating communities, the exact dates and locations of soil sample collections were anonymized at the request of community representatives and only relative locations were published. Each community invited and permitted researchers to collect soil samples from sites selected by Indigenous representatives for potential agricultural development or for extracting soil to be relocated and utilized for future agricultural purposes.

Multiple factors were considered for selecting soil sampling sites for intended agri-food initiatives, including but not limited to accessibility, proximity to water, and openness (whether natural or due to tree removal). Traditional land management practices, such as selective tree clearing or controlled burns,

were also taken into account in certain communities. Furthermore, these considerations are similar to constraints faced in some urban and peri-urban gardening spaces (Drake & Lawson, 2015).

*Australia*

In Queensland, eight sites were selected for soil sampling (Figure 1). Six sampling sites encompassed a variety of land types, including residential areas, a cattle station paddock, ravine areas, and a regenerated savannah that was historically used as a segregation fringe camp for Indigenous Peoples, in and around Community A. One of the residential sites was a former gas filling station that had since been remediated due to historical contamination. The eighth sample site was a soil pile managed by a small local business that sold agricultural supplies. There was interest for agricultural activities to occur on these lands by Indigenous Peoples and organizations, with the request for soil contaminant testing to examine agricultural suitability. The soils tested in this region were clay loam and sandy loam textured soils depending on the site, and the sites experienced mean annual rainfall ranging from 600 to over 2000 mm, with temperatures between 18 and 30 °C (Bureau of Meteorology, 2024).

**Figure 1** Relative location map of soil sample sites in Australia. Queensland (Community A) and New South Wales (Communities B, C, and D)



In New South Wales, a total of nine sites were selected for soil sampling (Figure 1). These sites represented different terrains, including flood plains and tablelands, where the areas were either grass-dominated or used for agricultural activities. They also had a history of being areas of cultural importance. The textures of soils tested in this region were silty clay loams and sandy clay loams. The region of the sites had a mean annual temperature of approximately 12 to 35 °C with mean annual rainfall amounts of 500 to 1200 mm (Bureau of Meteorology, 2024).

#### *Newfoundland, Canada*

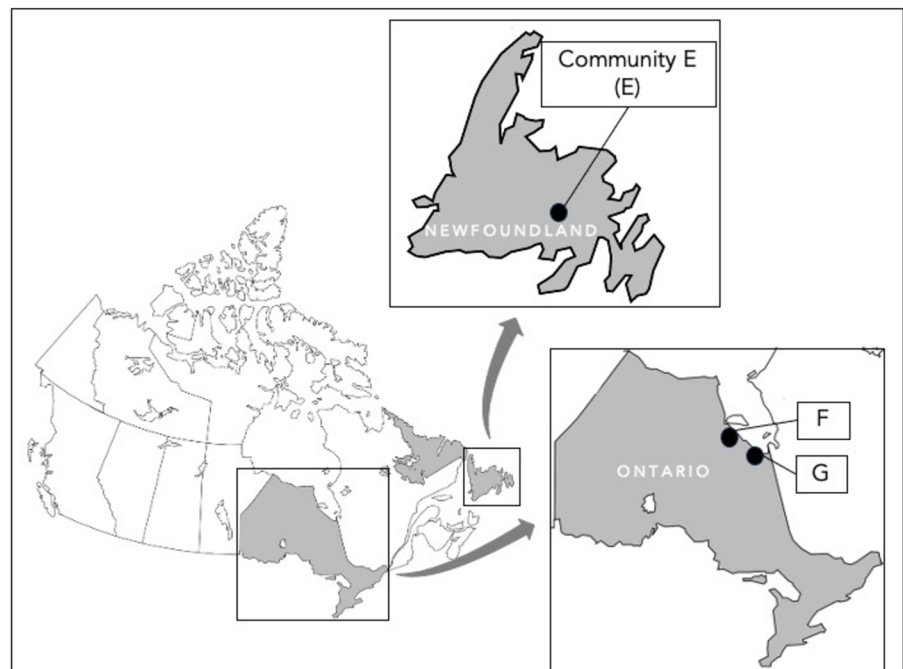
In Newfoundland (Figure 2), a total of 15 soil sampling sites were selected within Community E. Ten sites had in-ground soil samples taken as they were assessed for the development of agricultural activities, and one soil sample from a mixed pile to be potentially used as a local soil medium at a selected site(s) for agricultural development. These potential agricultural site locations were on lands controlled and historically utilized by the Indigenous community, and were within areas of park grounds, forests, campsites, and areas near regional and highway roads. Soils were also sampled from sites with known prior agricultural activities and timber extraction

histories. One sample site under study was a densely forested region with peat-rich soil, while another aligned with a hydro line corridor that was managed without known chemical usage. Additionally, samples from three soil distributors were examined as potential soil sources for agricultural use in Community E after it was determined that soil samples from community sites were elevated in arsenic (As) (Figure 2). The region experiences mean annual rainfall between 1000 and 1600 mm, with temperatures ranging from 5 to 10 °C (Environment & Climate Change Canada, 2024), and soils are peat, sandy, or rocky depending on local geology.

#### *Subarctic Ontario, Canada*

In subarctic Ontario, 24 soil sampling sites were selected in two remote Indigenous communities, Community F ( $n = 21$ ) and G ( $n = 3$ ), approximately 155 km apart (Figure 2). Nearby Community F, there was an abandoned Mid-Canada Radar Line (MCRL) that was remediated for historical contamination of both organic and metal contaminants. There are no conventional roads in this region, with the exception of ice roads that become accessible during winter months. Soils in this region had a silt loam texture with high soil organic matter content that was

**Figure 2** Relative location map of soil sample sites in Canada. Newfoundland (Community E) and subarctic Ontario (Community F and G)



below and above the 17% threshold for defining a soil as mineral or organic (Government of Canada, 2024). The sites experience mean annual temperatures between  $-19$  and  $16$  °C, with rainfall between 600 and 1000 mm (Environment & Climate Change Canada, 2024).

### Soil sampling

For all study locations, Indigenous community representatives were present for the sampling process. Composite soil samples were used for the analysis and were comprised of a minimum of ten individual soil samples for each site location. In-ground soil sampling occurred in a random grid pattern with the use of a nickel-plated chromoly steel soil probe. Locations in Australia used a 35.5-cm soil probe with a 30.5-cm sample window that had an inner diameter of 2.5 cm (AMS, American Falls, ID, USA), and locations in Canada used an 83.8-cm open-end soil probe measuring that had a 38.1-cm sampling slot with a 2.2-cm inner diameter (AMS, American Falls, ID, USA). The decision for sampling depth was region dependent. Sites within Australia had in-ground sampling to 0–15 cm soil depth, as recommended by Australian soil laboratories for deep-rooted crops (Environmental Analysis Laboratory, 2019; Government of New South Wales, 2019). In Newfoundland, the shallowness of bedrock dictated the decided sample depth per site to be 0–10 cm or 0–20 cm depth. In-ground sampling in subarctic Ontario was 0–20 cm soil depth for the considerations of deep-rooted crops and historical agricultural plowing practices. Obtaining a composite sample from mixed soil piles involved using a spade to collect the soil at various positions and a shovel to access deeper sections of the pile. When collecting soil for the composite samples per site (both for in-ground and mixed piles), individual samples were collected in a 20-L food-grade plastic bucket, with plant debris and stones manually removed. Composite samples from Australia sites were sieved and well mixed, divided and stored into labelled Ziploc® bags, and then kept cool at a temperature of 4 °C for 24 h to up to 1 week until submitted for analysis. The composite samples were divided in two, as separate National Association of Testing Authorities accredited laboratories were used in Australia: NSW Department of Primary Industries (Wollongbar, NSW) for metals and metalloid analysis, and

Symbio Laboratory (Brisbane, QLD) for organohalogen and pesticide analysis. Composite samples collected in Canada had one sample per site stored in labelled Ziploc® bags and frozen, until submitted to the Analytical Services Unit at Queen's University (Kingston, ON), a Canadian Association for Laboratory Accreditation accredited laboratory that analyzed the samples for all metal/metalloid and organic contaminants. In total, 30 metals and metalloids and 27 organohalogens (20 OC pesticides and seven PCBs as Aroclors) were analyzed for this study (Table 1); As was speciated following initial soil analysis. The selection of the contaminants is related to their persistence and ability to bioaccumulate, which may pose a risk to both ecosystem health and food security.

### Soil contaminant analysis

Soil samples were analyzed at the respective laboratories for metals and metalloids (hereafter metal(loid)s) concentrations using inductively coupled plasma mass spectrometry (ICP-MS) and for organohalogens using gas chromatography with gas chromatography-mass spectrometry (GC-MS).

### Analytical Methods for Australia

This assessment used diethylenetriaminepentaacetic acid (DTPA) to extract and measure micronutrient cations (i.e., Cu, Zn, Mn, and Fe) from the solid phase of soils. Air-dried subsamples were weighed and equilibrated for 2 h with an extractant solution of 0.005 M DTPA, 0.10 M triethanolamine, and 0.01 M  $\text{CaCl}_2$  at a 1:2 ratio of solution to dry soil. The buffered solution ensured Zn-DTPA complex formation and restricted trace element dissolution in strongly alkaline soils. Post-extraction, the leachates were centrifuged, filtered, diluted, and preserved. Finally, analysis was performed using inductively coupled plasma-optical emission spectrometer (ICP-OES) methods noted below.

To estimate total recoverable elements per unit mass in soils and sediments, the U.S. Environmental Protection Agency (USEPA) digestion procedure (Method 200.2) was used with some modifications. These modifications were in digestion time, tube shape, drying temperature, and lowered sample mass. Elements were extracted by hot dilute mineral acid refluxing, and the amounts of nutrients and heavy

**Table 1** Contaminants in soils intended for agricultural purposes analyzed in this study

Contaminants			
Metals ( <i>n</i> = 30)	Organohalogenes ( <i>n</i> = 27)		
Aluminum (Al)	Molybdenum (Mo)	2,4-Dichlorodiphenyldichloroethane (2,4-DDD)	Beta-benzene hexachloride ( $\beta$ -BHC)
Antimony (Sb)	Nickel (Ni)	2,4-Dichlorodiphenyldichloroethylene (2,4-DDE)	Delta-benzene hexachloride ( $\delta$ -BHC)
Arsenic (As)	Phosphorus (P)		Dieldrin
Barium (Ba)	Potassium (K)	2,4-Dichlorodiphenyltrichloroethane (2,4-DDT)	Endosulfan I
Beryllium (Be)	Selenium (Se)	4,4-Dichlorodiphenyldichloroethane (4,4-DDD)	Endosulfan II
Boron (B)	Silver (Ag)	4,4-Dichlorodiphenyldichloroethylene (4,4-DDE)	Endosulfan sulfate
Cadmium (Cd)	Sodium (Na)		Endrin
Calcium (Ca)	Strontium (Sr)	4,4-Dichlorodiphenyltrichloroethane (4,4-DDT)	Endrin aldehyde
Chromium (Cr)	Sulfur (S)	Aldrin	Gamma-benzene hexachloride ( $\gamma$ -BHC)
Cobalt (Co)	Thallium (Tl)	Alpha-benzene hexachloride ( $\alpha$ -BHC)	Heptachlor
Copper (Cu)	Tin (Sn)	3,3',5-Trichlorobiphenyl (Aroclor 1016)	Heptachlor epoxide isomer B
Iron (Fe)	Titanium (Ti)	2,2',4,4',5-Pentachlorobiphenyl (Aroclor 1221)	Methoxychlor
Lead (Pb)	Uranium (U)	2,2',4,4',5,6'-Hexachlorobiphenyl (Aroclor 1232)	
Magnesium (Mg)	Vanadium (V)	2,2',4,4',5,5'-Hexachlorobiphenyl (Aroclor 1242)	
Manganese (Mn)	Zinc (Zn)	2,2',3,4,4',5,5'-Heptachlorobiphenyl (Aroclor 1248)	
		2,2',4,4',5,5'-Hexachlorobiphenyl (Aroclor 1254)	
		2,2',3,4,4',5,5'-Heptachlorobiphenyl (Aroclor 1260)	

metals recovered in the solution were the same as those recovered by hot concentrated acids.

The amounts and proportions of major, minor, and trace elements were determined in solution using an Agilent 5110 ICP-OES according to USEPA Method 6010D. Quality control measures included analysis of analytical reagent blanks, spiked replicate samples, isotopic and elemental in-house standards, and certified reference materials. Blank levels were evaluated for all analytical runs, and accuracy and precision were assessed through matrix spikes, replicate analysis, and continuing calibration verification standards.

Sample preparation was performed using the QuEChERS (Quick, Easy, Cheap, Effective, Rugged, and Safe) extraction protocol, which was slightly modified in-house to maximize extraction efficiency. Modifications to the QuEChERS method included reducing the amount of solvent used by 20%, as noted by Perestrelo et al. (2019). The modification was aimed at improving extraction efficiency for specific soil types analyzed in this study. A Thermo-Scientific Triple Quadrupole GC-MS was used to assess PCBs in soil using Symbio's ENV146 methodology. A high-resolution LC-MS Orbitrap (liquid chromatography-mass spectrometry) was used to analyze multiple pesticides in soil using Symbio's ENV104 protocols. Samples were homogenized, weighed into

extraction tubes, spiked with appropriate solvents and salts, centrifuged to separate the organic phase carrying the analytes, and then submitted to a cleanup step in order to remove impurities. Homogenization of the soil samples was carried out by thoroughly passing the soil through a 2-mm sieve to ensure even distribution. This procedure minimizes sampling errors and ensures that each subsample represents the entire sample composition. Two sets of calibration curves to cover the expected concentration ranges of analytes and a quality control check every 20 samples were employed to keep track of the performance of the instrument and the extraction procedure. The acceptable recovery of the spike was in the range of 70–130%.

#### *Analytical methods for Newfoundland and subarctic Ontario*

For metal analysis, soil samples were prepared by drying, grinding, and acid digestion. The samples were then sub-sampled to ensure representativeness, dried in large weigh-boats, and ground to a fine homogenous consistency. The digestion process involved using a combination of nitric and hydrochloric acids, with samples heated in a DigiPrep™ apparatus to achieve complete digestion. The digested

soil samples were then filtered and diluted prior to ICP-MS analysis (Agilent 7700x), a process similar to USEPA Method 200.7, but with additional dilution for ICP-MS analysis due to the soil matrix.

For organohalogen analysis, soil samples were initially dried, ground, and homogenized before weighing 2–4 g into a clean glass thimble. These samples were spiked with the surrogate decachlorobiphenyl (DCBP) and extracted via Soxhlet extraction with dichloromethane. Post-extraction, the samples underwent gel permeation chromatography (GPC) cleanup, followed by concentration and final cleanup using a Florisil extraction column prior to analysis by GC-MS.

Speciation of As was carried out using liquid chromatography-inductively coupled plasma mass spectrometry (LC-ICP-MS; Agilent 1200series HPLC coupled to an Agilent 7700 ICPMS) by ALS Laboratories (Waterloo, Canada). Standard quality control methods included the use of a reference material certified for the As. More specifically, the NIST 1568B-Rice Flour standard, certified for dimethylarsinic acid (DMA) and monomethylarsonic acid (MMA), was used. The quality control methods also involved running blanks, laboratory control standards, and matrix spikes.

#### Ecological assessment of metal(loid)s in soil

There are no uniform global guidelines for assessing soil contamination in soils intended for agri-food initiatives, as different regions have unique geological and environmental conditions that influence contaminant levels. In this study, we applied region-specific geochemical backgrounds to account for these variations, while global ecological pollution indices were employed for the ecological risk assessment across regions. This method allowed for a more accurate reflection of the ecological risks specific to each region.

Contaminant threshold values for pollution indices, such as background concentration, only exist for certain metal(loid)s; therefore, we are limited to an ecological assessment of those contaminants which meet all variable criteria for each pollution index. For each study region, we employed the following methods to assess metal(loid) contamination in soil intended for agri-food initiatives for As, cobalt (Co), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), and zinc

(Zn): the degree of contamination ( $C_d$ ), potential ecological risk ( $RI$ ), and contamination severity index ( $CSI$ ). These are useful and important indices for soil contaminant assessment and estimating environmental risk (Sezgin & Temelli, 2023; Teng et al., 2022; Wieczorek & Baran, 2021). Combined, these assessment methods provide a more complete picture of the soil contamination in our study areas. Importantly, Australian and Canadian guidelines for agricultural soil contamination differ between governmental bodies—therefore, the pollution index analysis was carried out with respect to the regulatory standards of their respective location.

#### Degree of contamination ( $C_d$ )

To assess the enrichment of metal(loid)s in sampled soils beyond the existing pre-industrial background concentrations, we used a method devised by Hakanson (1980) to calculate the degree of contamination ( $C_d$ ) using Equation 1:

$$C_d = \sum CF, CF = \frac{C_a}{C_{pi}} \tag{1}$$

where  $CF$  is the contamination factor,  $C_a$  is the soil concentration of the contaminant under assessment, and  $C_{pi}$  is the pre-industrial reference value of the metal in the earth, both in mg/kg; values for  $C_{pi}$  are given in Table S1. The  $C_d$  employs a categorical hierarchy of contamination: low ( $C_d < 8$ ), moderate ( $8 \leq C_d \leq 16$ ), high ( $16 \leq C_d \leq 32$ ), and very high ( $C_d \geq 32$ ).

#### Potential ecological risk ( $RI$ )

The quality of soil and degree of ecological risk from toxic metal(loid) concentrations can be assessed by using the  $RI$  (Hakanson, 1980) and calculated using Equation 2:

$$RI = \sum \left( T \times \frac{C_a}{B_a} \right) \tag{2}$$

where  $T$  is the global toxicity response coefficient for each toxic metal (Table S1),  $C_a$  is the soil concentration of the assessed contaminant, and  $B_a$  is the geochemical background. The  $RI$  is classified as low ( $< 90$ ), moderate (90–180), strong (180–360), very strong (360–720), and extreme ( $\geq 720$ ).

### Contamination severity index (CSI)

In order to conservatively evaluate the concentration of toxic metal(loid)s in soil, we employed the *CSI* (Pejman et al., 2015). Sediment quality guidelines (Table S1) as the effects range low (ERL), where toxic effects are rarely observed, and effects range median (ERM), where toxic effects are often observed (Long et al., 1995; Luo et al., 2010), were used in the calculation for the *CSI* (Equation 3):

$$CSI = \sum w \left( \sqrt{\frac{C_a}{ERL}} + \left( \frac{C_a}{ERM} \right)^2 \right), w = \frac{VF \times \lambda}{\sum (VF \times \lambda)} \quad (3)$$

where  $C_a$  is the concentration of the toxic metal(loid) in soil,  $w$  is the weight of each metal,  $VF$  (varifactor) is a value derived from principal components in principal component analysis (PCA) and further refined through factor analysis (FA) to represent latent variables underlying the metal(loid) data, and  $\lambda$  is the eigenvalue computed by the PCA/FA carried out for the  $VF$  (Pejman et al., 2015). No ERL or ERM for Co exists; therefore, it was not included in the *CSI* analysis. The value of  $w$  for each toxic metal is presented in Table S2. There are nine classifications of the *CSI* (Table S3), but generally, if the *CSI* is less than 0.5, the soil is not contaminated, if the *CSI* is between 2 and 2.5, the soil contamination is of moderate severity, and if the *CSI* is greater than 5, the soil contamination is of ultra-high severity.

### Ecological assessment of organohalogenes in soils for agricultural activities

Organic contaminants are abundant in the environment due to anthropogenic activities, and like metal(loid)s, organic contaminants have environmental background levels because of their widespread use and translocation (Food and Agriculture Organization of the United Nations [FAO], 2021). Additionally, there is a gap in the literature for pollution indices that assess organic compounds. Therefore, we extrapolate methods of soil assessment from Qingjie et al. (2008) to assess soil organic residues in soils intended for agricultural use including the mean single pollution index ( $\overline{PI}$ ) and the Nemerow pollution index ( $PI_N$ ). Pesticide background thresholds were set at

0.078 mg/kg for *p,p'*- and *o,p'*-DDT and 0.05 mg/kg for DDT metabolites (Government of Ontario, 2015).

### Mean single pollution index

To assess the single organic compound contamination of the sampled soils in the study regions, we used the  $\overline{PI}$  as a basic measure of the contaminant concentration ( $C_a$ ), the background concentrations ( $B_a$ ) of the soil, and sample size ( $n$ ) of assessed detectable pesticides in Equation 4:

$$\overline{PI} = \frac{\sum PI}{n} = \frac{\sum \left( \frac{C_a}{B_a} \right)}{n} \quad (4)$$

A  $\overline{PI}$  that exceeds 1.0 indicates a lower quality and higher contamination in soil (Kolawska et al., 2018).

### Nemerow pollution index

The overall degree of soil contamination of pesticides requires a more complex assessment, and we used the  $PI_N$  to evaluate this (Equation 5):

$$PI_N = \sqrt{\frac{\overline{PI}^2 + PI_{\max}^2}{2}} \quad (5)$$

where  $\overline{PI}$  is the mean single pollution index and  $PI_{\max}^2$  is the maximum value of the pollution index for each assessed organic contaminant. Of the five classes of the  $PI_N$ , soil is clean ( $\leq 0.70$ ), approaching a warning limit (0.70–1.00), slightly polluted (1.00–2.00), moderately polluted (2.00–3.00), or heavily polluted ( $\geq 3.00$ ) based on this five-class scale.

### Statistical analysis

Descriptive statistics were calculated for all metal(loid)s and pesticides. Correlograms with dendrograms (clustering diagrams) were generated using the pheatmap package (Kolde, 2019) to assess the nature of the relationships and clustering of the toxic metals in soil for each study area. A strong strength of association was set at  $r \geq 0.70$  for metal(loid)s and  $r \geq 0.90$  for pesticides, respectively.

To carry out the PCA/FA for the toxic metal(loid)s *CSI*  $w$ , we imputed values for missing contaminant concentrations below the level of detection (LOD) using

50% of the minimum detectable contaminant (LOD/2) concentrations resulting from the region-specific soil analysis (Environmental Protection Agency [EPA], 2023). This resulted in 47 inputs for missing toxic metal or metalloid contaminant concentrations and represented 11% (47/392) of the toxic metal(loid) data used in the PCA/FA. Since the OC pesticides were all positively and significantly correlated ( $r \geq 0.91$ ,  $p < 0.001$ ), a PCA was also conducted because it is effective at finding underlying correlations that are not often detectable otherwise (Kherif & Latypova, 2020). However, we could not directly impute contaminant concentrations for the detectable OC pesticides, as 54% (125/230) of the samples were below the LOD, and carrying out a PCA with more than 30% of the imputed data as LOD/2 increases the variance to unacceptable levels (Farnham et al., 2002). Therefore, to overcome this problem, we imputed the missing contaminant concentration using the missMDA package (Josse & Husson, 2016) to estimate contaminant concentrations using actual data and 10,000 iterations to model contaminant concentrations below the LOD for dieldrin, endosulfan sulfate, and total DDE, DDD, and DDT, and to also overcome the probability of extreme variance with more than 30% of the contaminant samples below the LOD (Farnham et al., 2002).

Using Bartlett's test of sphericity (herein Bartlett's test) in the performance package (Lüdtke et al., 2021) and the Kaiser, Meyer, and Olkin (KMO) measure criterion in the psych package (Revelle, 2023), we assessed the suitability of the data for the PCA/FA for the *CSI w*. Bartlett's test assesses the correlation in the data for PCA and deems it suitable if the null hypothesis is rejected and the KMO assesses if the data are appropriate for FA by examining the strength of partial correlation between the variables. Following the suitability assessment, we used the factoextra (Kassambara & Mundt, 2020), psych, and gparotation (Bernards & Jennrich, 2005) packages to extract *VF* and  $\lambda$  (Table S2) to calculate the *CSI w* as outlined in Equation 4. All statistical analysis was carried out using R version 4.3.1 (R Core Team, 2023), and statistical significance was set at  $p < 0.05$ .

## Results

### Site presence of soil metal(loid)s

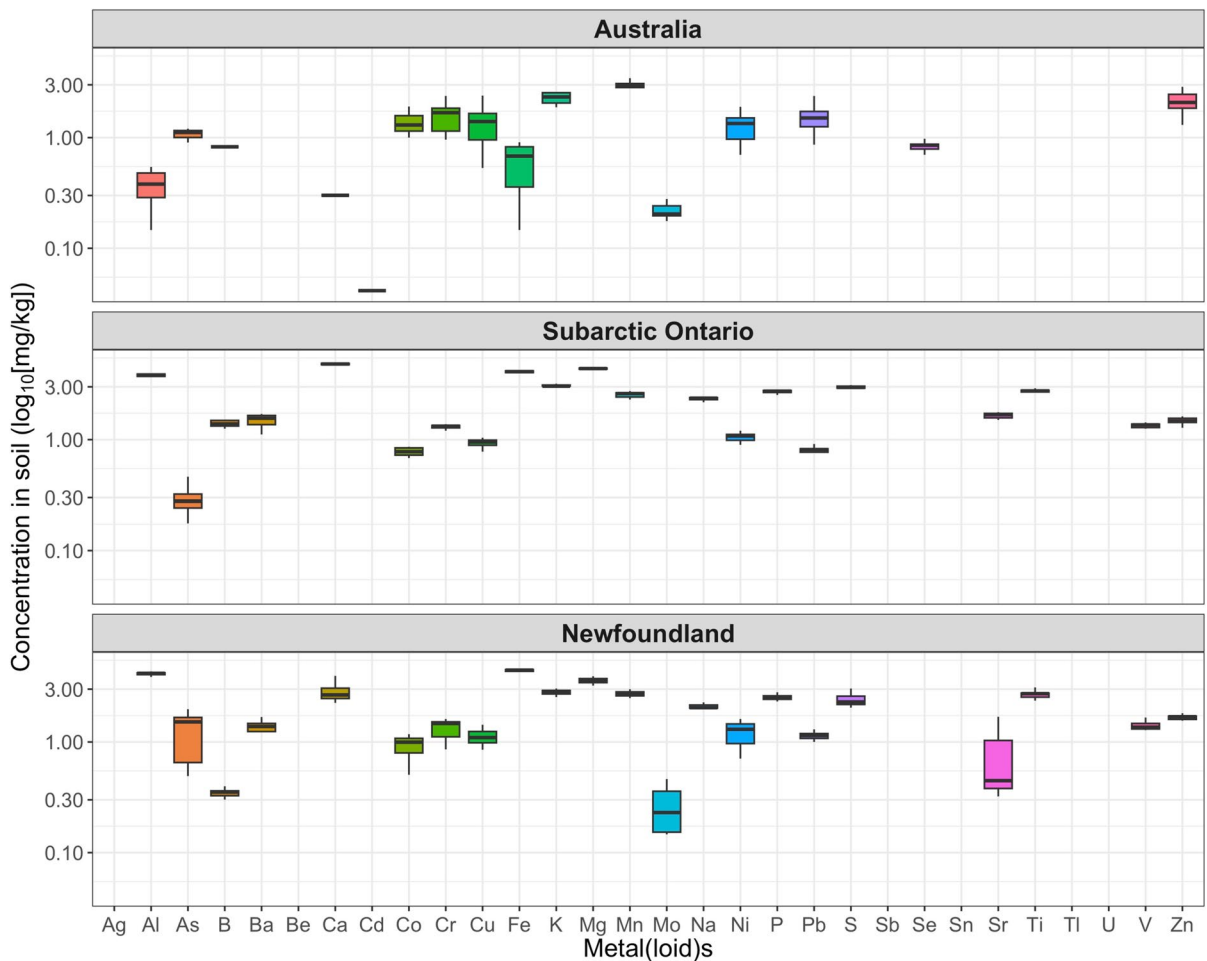
The descriptive statistics of all metals tested in all regions are presented in Figure 3 and Table S4. There are detectable concentrations of all metal(loid)s in soil in all study regions except for Sb, Sn, Tl, and U, and there is only one detectable concentration of Ag in Newfoundland. Detectable concentrations of metals ( $n$  detects; Table S4) ranked descending from highest to lowest across all regions, which are as follows: Al, Ca, Fe, K, Mg, Mn, P, S > Zn > Cu, Ni > Co > Cr > Pb > Na > As > Ba > Sr > Ti > V > Be > B > Mo > Se > Cd > Ag. The highest mean concentration of any metal(loid) was Ca (69,558.33 mg Ca/kg) and the lowest (with more than one detect) was Be (0.31 mg Be/kg), and the highest mean concentration of a toxic metal(loid) was Cr (63.14 mg Cr/kg) and the lowest was Cd (0.55 mg Cd/kg).

Mean concentrations of As, B, and S surpassed the Canadian Council of Ministers of the Environment (CCME) guidelines for agricultural soil quality, and Mn and Zn surpassed the National Environment Protection Council (NEPC) guidelines (Table S1). The contamination of sampled soils by metal(loid)s differed across Australia and subarctic Ontario, and Australia and Newfoundland, but not subarctic Ontario and Newfoundland, except for As.

### Ecological assessment of metal(loid)s

We assessed As (except for Australia due to a low number of detects), Co, Cr, Cu, Ni, Pb, and Zn across the study regions. The results of the pollution indices for all regions are presented in Table 2.

Bartlett tests were significant ( $p < 0.001$ ) for all study regions, with the KMO being 0.62 for Australia and 0.71 for Newfoundland indicating compatibility for PCA/FA for the *CSI w*, but not for subarctic Ontario (KMO = 0.38). Following examination of the data, it was found that the pairwise association of Cu with Zn ( $r > 0.90$ ) reduced the KMO; thus, the removal of one of these from the PCA/FA was necessary to ensure a reliable measure for the *CSI w* for subarctic Ontario. Therefore, Zn was removed and it improved the overall KMO for subarctic Ontario to 0.50. The *VF* and eigenvalue representative ( $\lambda > 1.00$ ) of the input into soil contamination were then



**Figure 3** Concentration of metal(loid)s across Australia, subarctic Ontario, and Newfoundland. The boxplots illustrate the variability of metal(loid) concentrations in soil samples from each region, with a  $\log_{10}$  transformation applied for visualization. Key: Ag, silver; Al, aluminum; As, arsenic; B, boron; Ba, barium; Be, beryllium; Ca, calcium; Cd, cadmium; Co, cobalt;

Cr, chromium; Cu, copper; Fe, iron; K, potassium; Mg, magnesium; Mn, manganese; Mo, molybdenum; Na, sodium; Ni, nickel; P, phosphorus; Pb, lead; S, sulfur; Sb, antimony; Se, selenium; Sn, tin; Sr, strontium; Ti, titanium; Tl, thallium; U, uranium; V, vanadium; Zn, zinc

used to calculate the  $CSI_w$ . All  $VF$  were above 0.70 for Australia, subarctic Ontario, and Newfoundland, except for Cu in subarctic Ontario ( $VF = 0.16$ ) and Pb in Newfoundland ( $VF = 0.41$ ), and explained 83.87%, 75.49%, and 80.68% of the total variance, respectively. The  $VF$  and  $\lambda$  for each region are presented in Table S2.

#### Site presence of soil metal(loid)s in Australia

The mean concentrations of detected metal(loid)s, from highest to lowest,  $Mn > Zn > K > Cr > Pb >$

$Cu > Ni > Co > As > Se > Fe > Al > Mo > Cd > Ca > Mg > P > S > Na$ ; Ag, Ba, Be, Sb, Sn, Sr, Ti, Tl, U, and V, are below the LOD (Table S4). There is a strong positive correlation in soil between Co and Cr ( $r = 0.81$ ,  $p < 0.01$ ), Co and Ni ( $r = 0.79$ ,  $p < 0.01$ ), Cr and Ni ( $r = 0.86$ ,  $p < 0.01$ ), and Cu and Zn ( $r = 0.75$ ,  $p < 0.05$ ); Cu and Pb are significantly but not strongly correlated ( $r = 0.69$ ,  $p < 0.05$ ) (Figure 4A). Cr and Ni and Cu and Zn are clustered together, respectively, where Co and Pb are in separate clusters (Figure 4A).

**Table 2** Pollution index results for metal(loid)s

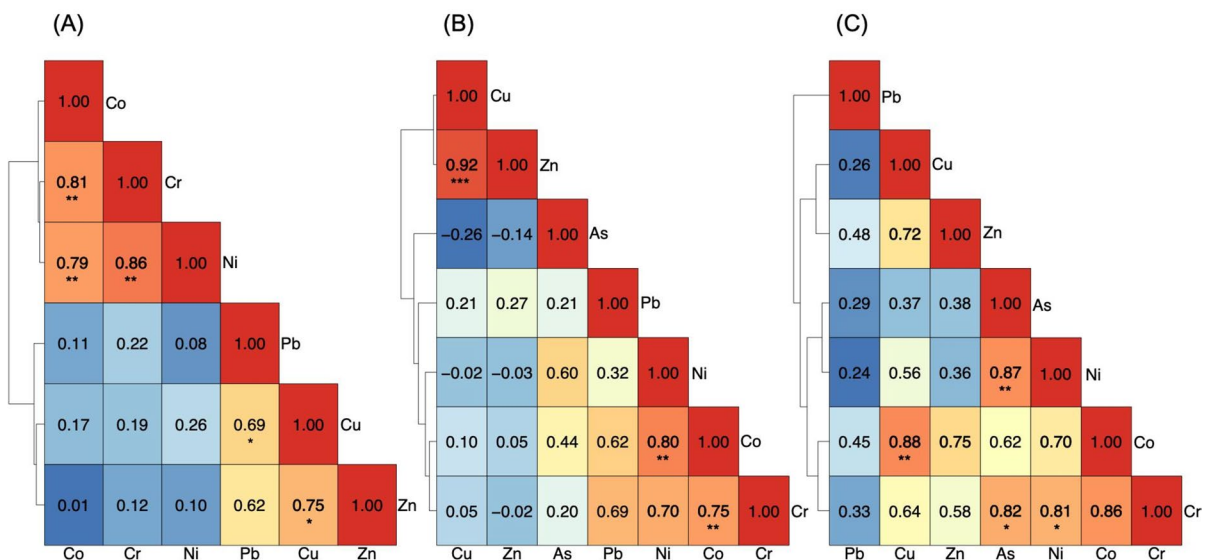
Location	Metal(loid)	$C_d$	$RI$	$CSI$
Australia	Co	17.68	350.79	--
	Cr	11.93	90.96	3.08
	Cu	14.49	323.48	3.63
	Ni	20.61	241.89	5.65
	Pb	14.15	685.80	3.37
	Zn	20.7	275.51	5.21
Subarctic Ontario	As	2.97	40.50	2.44
	Co	4.62	30.37	--
	Cr	3.67	9.85	2.24
	Cu	4.12	16.63	0.57
	Ni	10.7	33.24	3.81
	Pb	1.24	9.61	1.11
Newfoundland	Zn	4.08	4.92	--
	As	31.29	312.87	6.21
	Co	5.51	6.89	--
	Cr	3.85	6.93	1.25
	Cu	3.95	19.76	1.55
	Ni	13.18	21.66	3.23
	Pb	2.91	51.00	0.08
	Zn	3.72	13.03	1.47

Key:  $C_d$ , degree of contamination;  $RI$ , potential ecological risk index;  $CSI$ , contamination severity index; *As*, arsenic; *Co*, cobalt; *Cr*, chromium; *Cu*, copper; *Ni*, nickel; *Pb*, lead; *Zn*, zinc

The  $C_d$  indicates considerable soil contamination for Co (17.68) and moderate contamination of Cr, Cu, Ni, Pb, and Zn (range 11.93 to 20.70) (Figure 5; Table 2). The  $RI$  indicates very strong enrichment in sampled soils for Pb (685.80); strong enrichment for Co, Cu, Ni, and Zn (241.89 to 350.79); and low to moderate soil enrichment for Cr (90.96). Lastly, the  $CSI$  indicates ultra-high contamination of soils with Ni (5.65) and Zn (5.21), high contamination of Cu (3.63) and Pb (3.37), and moderate to high contamination of Cr (3.08) (Figure 5; Table 2).

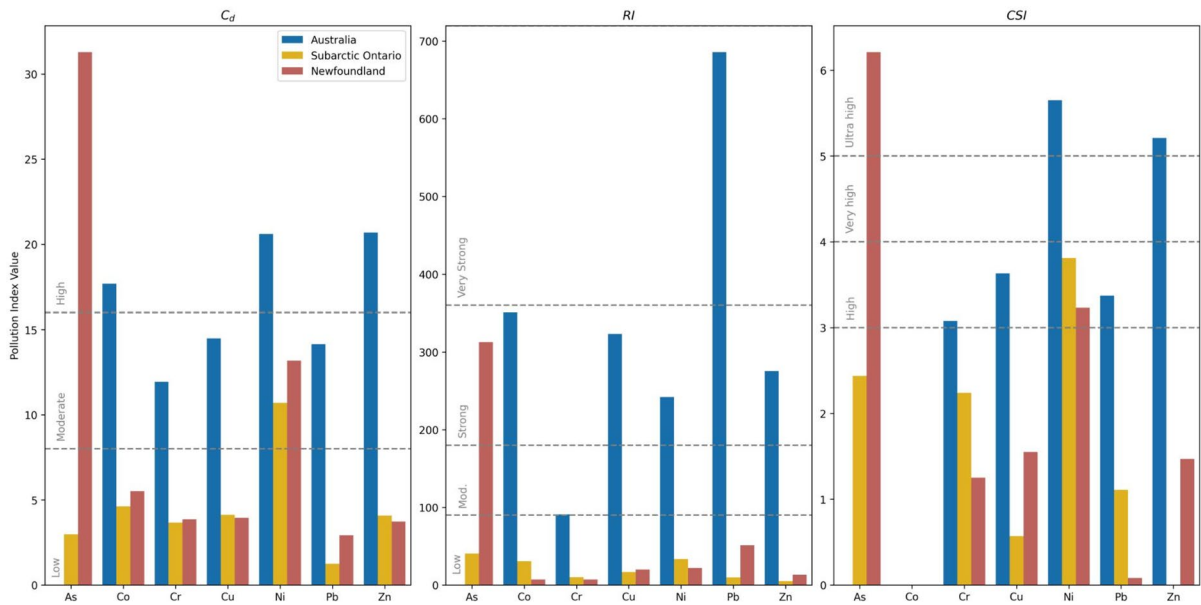
*Site presence of soil metal(loid)s in subarctic Ontario*

The mean concentrations of metal(loid)s from highest to lowest,  $Ca > Mg > Fe > Al > K > S > P > Ti > Mn > Na > Sr > Ba > Zn > B > V > Cr > Ni > Cu > Pb > Co > As > Be; Ag, Cd, Mo, Sb, Se, Sn, Tl, \text{ and } U$ , are below the LOD (Table S4). There is a strong positive correlation between Cu and Zn ( $r = 0.92, p < 0.001$ ), and they are clustered in the dendrogram along with Co and Ni ( $r = 0.80, p < 0.01$ ) and Co and Cr ( $r = 0.75, p < 0.01$ ) (Figure 4B). The remaining metals As, Pb, and Ni are in their own clusters with Co and Cr. The  $C_d$  (range 1.24 to 4.62) and  $RI$  (range 4.92 to 40.95) suggest that soil contamination by metal(loid)s, except Ni ( $C_d = 10.7; RI = 33.24$ ), is



**Figure 4** Correlations among metal(loid)s with corresponding cluster analysis dendrogram in **A** Australia; **B** subarctic Ontario; and **C** Newfoundland. Statistical significance is

denoted by asterisks (\* $p < 0.05$ , \*\* $p < 0.01$ , \*\*\* $p < 0.001$ ). Key: *As*, arsenic; *Co*, cobalt; *Cr*, chromium; *Cu*, copper; *Ni*, nickel; *Pb*, lead; *Zn*, zinc



**Figure 5** Pollution index values, identified by title, for metal(loid)s in tested soils from Australia, subarctic Ontario, and Newfoundland. Left panel: the degree of contamination

( $C_d$ ). Middle panel: potential ecological risk ( $RI$ ). Right panel: contamination severity index ( $CSI$ ). Index levels are categorized vertically in each panel

low (Figure 5; Table 2) and unpolluted, respectively. The  $CSI$  indicates a moderate to high contamination of Ni (3.81), a low to moderate contamination of As (2.44), and low to very low contamination of Cr (1.26), Pb (1.11), and Cu (0.57), respectively.

for Ni (3.23), low to moderate for Zn (1.47), low for Cr (1.25) and Cu (1.55), and very low for Pb (0.08) (Figure 5; Table 2).

#### Site presence of soil metal(loid)s in Newfoundland

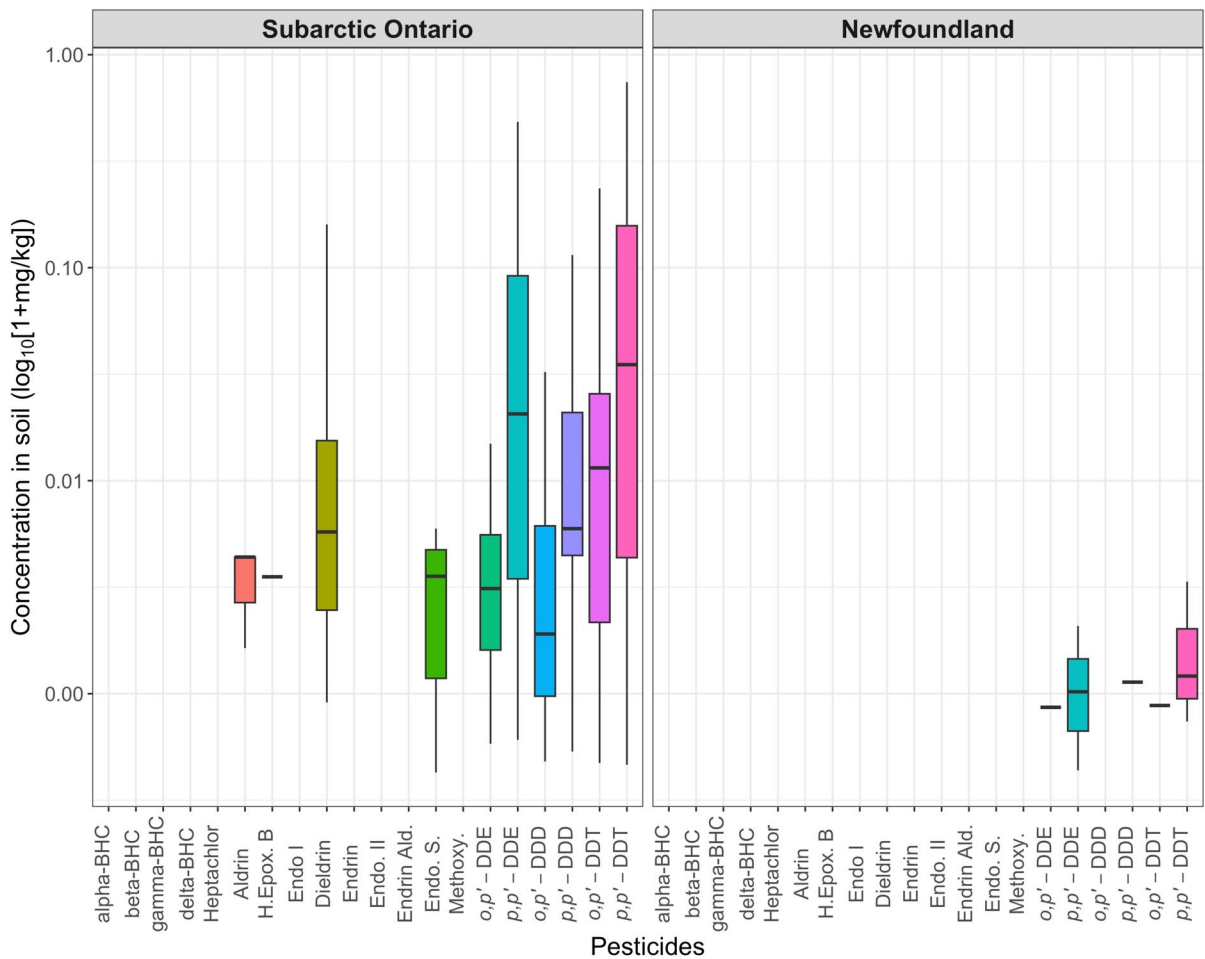
The mean metal(loid) concentrations, from highest to lowest, Fe > Al > Mg > Ca > K > Mn > Ti > P > S > Na > Zn > As > Cr > V > Ba > Ni > Cu > Pb > Sr > Co > B > Mo > Be > Ag; Cd, Sb, Se, Sn, Tl, and U, are below the LOD (Table S4). There is a strong, positive correlation between As and Ni ( $r = 0.87$ ,  $p < 0.01$ ), As and Cr ( $r = 0.82$ ,  $p < 0.05$ ), Cr and Ni ( $r = 0.81$ ,  $p < 0.05$ ), and Co and Cu ( $r = 0.88$ ,  $p < 0.01$ ) (Figure 4C). Pb is distant and does not cluster with any other metal; meanwhile, As and Ni are clustered with Cr, while Co and Cu are clustered with Zn (Figure 4C). The  $C_d$  is bordering on very high contamination (31.29) for As, moderate contamination of Ni (13.18), and low for Co, Cr, Cu, Pb, and Zn (range 2.91 to 5.51) (Figure 5; Table 2). The  $RI$  shows a strong enrichment factor of As (312.87), but not for the other metals or metalloids (range 6.89 to 51.00). The  $CSI$  is considered ultra-high for As (6.21), high

Site presence of organohalogen and organochlorine pesticides

No PCB or PCB-like compounds were detected (< LOD) in Australia or Newfoundland, and there was a single measure of Aroclor 1260 (0.26 mg/kg) compound from a site in subarctic Ontario (Figure 6). Therefore, they were not included in any pollution indices and are not reported hereafter. Additionally, no pesticides were detected in Australia; however, there were nine detects in Newfoundland and 158 total pesticide detects in subarctic Ontario. This region had soils that were the most contaminated with pesticides compared to the other regions tested (Table S5).

A breakdown of the detectable DDT and its metabolites is shown for subarctic Ontario (Figure 7).

Detectable levels of aldrin; heptachlor epoxide isomer B; dieldrin; endosulfan sulfate; *o,p'*-DDE, DDD, DDT; and *p,p'*-DDE, DDD, DDT were found in subarctic Ontario. Detects of *p,p'*-DDE, DDD, DDT and *o,p'*-DDE and DDT were also found in

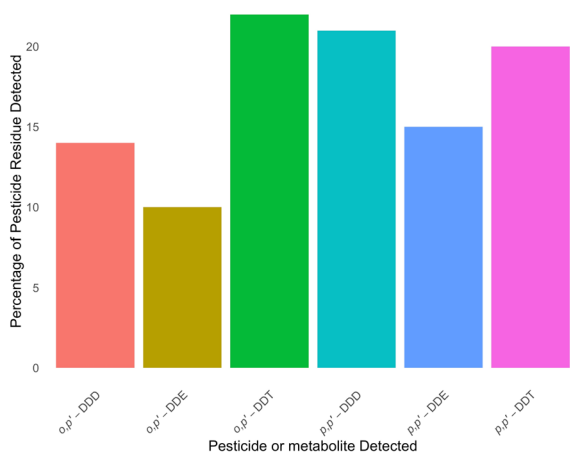


**Figure 6** Concentrations of various pesticides (1+ mg/kg) in soil samples from subarctic Ontario and Newfoundland with a  $\log_{10} +1$  transformation applied for visualization. Key: alpha-BHC, alpha-benzene hexachloride; beta-BHC, beta-benzene hexachloride; gamma-BHC, gamma-benzene hexachloride; delta-BHC, delta-benzene hexachloride; Heptachlor, heptachlor; H.Epox.B, heptachlor epoxide isomer B; Aldrin, aldrin; Endo I, endosulfan I; Dieldrin, dieldrin; Endrin, endrin; Endo

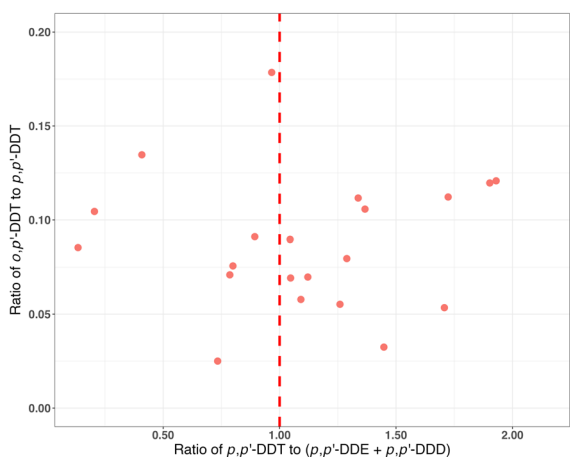
II, endosulfan II; Endo S., endosulfan sulfate; Endo Ald., endosulfan aldehyde; Methoxy, methoxychlor; *o,p'*-DDE, *o,p'*-dichlorodiphenyldichloroethylene; *p,p'*-DDE, *p,p'*-dichlorodiphenyldichloroethane; *o,p'*-DDD, *o,p'*-dichlorodiphenyldichloroethane; *p,p'*-DDD, *p,p'*-dichlorodiphenyldichloroethane; *o,p'*-DDT, *o,p'*-dichlorodiphenyltrichloroethane; *p,p'*-DDT, *p,p'*-dichlorodiphenyltrichloroethane

Newfoundland. Mean concentrations of the OC pesticides ranked highest to lowest: *p,p'*-DDT > *p,p'*-DDE > *o,p'*-DDT > *p,p'*-DDD > Dieldrin > *o,p'*-DDE, DDD, endosulfan sulfate, aldrin, heptachlor epoxide isomer B. A plot (Figure 8) differentiating the historical and recent use of technical DDT (*p,p'*-DDT) suggests that of the 21 complete pairwise observations for subarctic Ontario, 13 (62%) are a more recent input into the soil and 8 (38%) suggest a historical input of technical DDT.

The PCA (Bartlett’s test,  $p < 0.001$ ) on the detectable pesticides reduced the data to two PCA factors with  $\lambda > 1.00$  (Table S6). Together, they explained a total variance of 95.21% (Figure 9). Individually, PC1 and PC2 captured 75.20% and 20.01% of the total variance, respectively. PC1 displayed strong positive loadings for dieldrin, DDE, DDD, and DDT, while PC2 demonstrated strong negative loading for endosulfan sulfate.



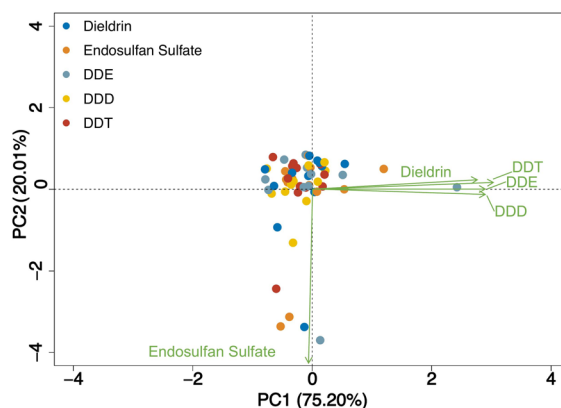
**Figure 7** Chemical composition of DDT and its metabolites in subarctic Ontario, Canada. Key, *o,p'*-DDE, *o,p'*-dichlorodiphenyldichloroethylene; *p,p'*-DDE, *p,p'*-dichlorodiphenyldichloroethylene; *o,p'*-DDD, *o,p'*-dichlorodiphenyldichloroethane; *p,p'*-DDD, *p,p'*-dichlorodiphenyldichloroethane; *o,p'*-DDT, *o,p'*-dichlorodiphenyltrichloroethane; *p,p'*-DDT, *p,p'*-dichlorodiphenyltrichloroethane



**Figure 8** A ratio plot showing the relative historical and recent use of technical DDT (*p,p'*-DDT) in subarctic Ontario, Canada. Key: *p,p'*-DDD, *p,p'*-dichlorodiphenyldichloroethane; *p,p'*-DDE, *p,p'*-dichlorodiphenyldichloroethylene; *o,p'*-DDT, *o,p'*-dichlorodiphenyltrichloroethane; *p,p'*-DDT, *p,p'*-dichlorodiphenyltrichloroethane

#### Site pollution indices for pesticides in Canada

There were not enough detects to assess the  $\bar{PI}$  and  $PI_N$  for Newfoundland; therefore, the results presented here reflect the subarctic Ontario region only



**Figure 9** The results of the PCA analysis for organochlorine pesticides in subarctic Ontario, Canada. Key: DDD, *p,p'*-dichlorodiphenyldichloroethane; DDE, *p,p'*-dichlorodiphenyldichloroethylene; DDT, *p,p'*-dichlorodiphenyltrichloroethane

**Table 3** Pollution indices for pesticides in subarctic Ontario

Pesticide	$\bar{PI}$	$PI_N$
Dieldrin	0.10	0.45
Endosulfan sulfate	0.19	0.28
<i>o,p'</i> -DDE	0.19	0.51
<i>p,p'</i> -DDE	4.60	29.17
<i>o,p'</i> -DDD	0.27	1.11
<i>p,p'</i> -DDD	0.87	4.32
<i>o,p'</i> -DDT	1.25	6.59
<i>p,p'</i> -DDT	6.35	41.40
Total DDT	0.74	5.51

Key:  $\bar{PI}$ , mean single pollution index;  $PI_N$ , Nemerow pollution index; *o,p'*-DDE, *o,p'*-dichlorodiphenyldichloroethylene; *p,p'*-DDE, *p,p'*-dichlorodiphenyldichloroethylene; *o,p'*-DDD, *o,p'*-dichlorodiphenyldichloroethane; *p,p'*-DDD, *p,p'*-dichlorodiphenyldichloroethane; *o,p'*-DDT, *o,p'*-dichlorodiphenyltrichloroethane; *p,p'*-DDT, *p,p'*-dichlorodiphenyltrichloroethane

(Table 3). We assessed *o,p'*- and *p,p'*-DDT, DDE, DDD, dieldrin, and endosulfan sulfate for the  $\bar{PI}$  and  $PI_N$ . The mean  $\bar{PI}$  indicates soils are polluted ( $\bar{PI} > 1.00$ ) with *p,p'*-DDE, *o,p'*-DDT, and *p,p'*-DDT. The  $PI_N$  indicates that the soils are heavily polluted with *p,p'*-DDE, DDD, DDT, and *o,p'*-DDT. There is slight contamination with *o,p'*-DDD, and the remaining pesticides do not excessively contaminate the soils.

## Discussion

There are detectable concentrations of metal(loid)s or pesticides in soils intended for Indigenous agri-food initiatives that exceed the established threshold for soil contamination in Australia and Canada. However, the concentrations and type of contaminants, and guidelines establishing threshold values, vary widely across the global regions. Metal(loid) concentrations are increased in Australian agricultural soils (i.e., Pb, Ni, and Zn) but are devoid of detectable pesticides and organohalogenes. On the contrary, the soils selected from sites in Canada are either contaminated with pesticides (i.e., DDT) or toxic metal(loid)s (i.e., As), but not both, albeit from anthropogenic and natural sources, respectively.

The discussion is locally targeted, even though our study investigated global regions, to provide insights specific to the regions under study. Expanding the scope to include a global perspective would risk diverting attention from the localized ecological assessments for the affected Indigenous communities in Australia and Canada. However, the findings of this research may serve as a foundation for future international studies on soil contamination in Indigenous and rural areas that may foster global collaboration on addressing contamination issues for agri-food initiatives. While this study focuses on ecological and soil contamination risks, potential human health risks from exposure to these contaminants will be explored in future research. At present, the strength of this study lies in its focused regional approach, ensuring that the discussion and any recommendations are directly relevant to the unique environmental and agricultural context of the regions involved.

### Soil contamination by metal(loid)s

#### *Australia*

Australia exhibited the highest metal(loid) contamination concentrations compared to the other study regions and the pollution indices (i.e.,  $C_d$ ,  $RI$ ,  $CSI$ ) consistently identified soil contamination by metal(loid)s which signaled a potential ecological risk. All the pollution indices indicated that Pb is present as a pollutant-contaminant in the soil, with the  $RI$  being the strongest indicator, followed by the  $CSI$  and  $C_d$  (Figure 5; Table 2), even though the mean

concentration of Pb in the soil (58.25 mg Pb/kg) does not exceed the Australian guidelines for agricultural or garden soils (Tables S1 and S4). The contamination of soils with Pb in rural sites in Australia and the high pollution index values (Figure 5; Table 2) raise concerns about environmental impacts from anthropogenic activities, such as mining, smelting, or deposition of historical leaded gasoline use—leaded gas was not banned until 2002 and its use is estimated to have resulted in the release of more than 240,000 tonnes of Pb across Australia over seven decades (Kristensen, 2015).

Given that the pollution indices used in this study (Figure 5; Table 2) estimate anthropogenic and/or natural deposition of metal(loid)s, we must also consider local geology as some regions in Australia naturally contain high levels of Pb in soils as a result of various geological processes. A recent study of Pb isotopes in Australia (Desem et al., 2024) indicated that there is abundant Pb in soils in the region of Communities B, C, and D, and this could also be responsible for the increased pollution indices. Nonetheless, Australia has some of the highest thresholds for Pb (Barry, 1997; National Environmental Protection Council [NEPC], 2011) in agricultural soil (300 mg/kg) compared to other countries like Canada (70 mg/kg; CCME, 2023). The maximum measured concentration of Pb (240 mg/kg) in soil tested in Australia would have exceeded the CCME (2023) guideline by almost 3.5 times; thus, some guidelines would recommend not to grow crops in this soil (Kansas State, 2017).

Regardless of the source of Pb, the implications for agri-food initiatives could be significant. While plants tend to limit Pb uptake (Castro-Bedriñana et al., 2021), there is still some uptake and the contamination of crops by soil with Pb that comes into contact with food may pose health risks. Although this study focuses on ecological risks, elevated soil concentrations of Pb can lead to human exposure through ingestion of contaminated soil or surface-contaminated (unwashed) crops, particularly posing a risk to children, who are vulnerable to developmental delays and cognitive impairments from Pb exposure (Kumar et al., 2020). For example, Schupp et al. (2020) found that a soil Pb concentration of approximately 5 mg Pb/kg correlated with an intake of 10  $\mu$ g Pb/day per person. Without remediation, soils in these areas may not be suitable for agri-food initiatives, potentially

undermining local efforts to enhance food security through sustainable agriculture.

Beyond Pb, the  $C_d$  is high and the *CSI* is ultra-high (Figure 5; Table 2) for Zn, and this could be indicative of nearby industrial activities, such as mining or manufacturing, in which Zn may be liberated directly or indirectly as a Cu-Zn ore (Evans, 2009), as the mineral potential for Zn is low in our study regions (Cloutier et al., 2023). As a possible source, there is active Zn mining in northwest Queensland (Queensland Government, 2017) and Cu mining in Queensland (Ibid) and New South Wales (New South Wales Government, 2022) northwest and west of the sites where the soil was sampled. These metals (Cu and Zn) are clustered in our analysis, and significantly and strongly correlated to each other (Figure 4A). We speculate that they could possibly be from a mined ore source.

High concentrations of Zn, Cd, and Ni, as indicated by the *CSI* and *RI* indices (Figure 5; Table 2), can adversely affect agricultural productivity and local agri-food initiatives. Zn, in particular, can disrupt soil microbial communities, soil structure, and enzymes and nutrient cycling processes, and can also leach into groundwater, posing risks to aquatic ecosystems (Liu et al., 2020; McLaughlin & Smolders, 2001). Additionally, the *Cd* for Co (17.68) and Ni (20.61) suggests high levels of pollution relative to the background concentrations, and the *RI* (Figure 5; Table 2) suggests a strong input of these metals into the soil; the *CSI* indicates an ultra-high input for Ni (6.19), and this could have an impact on ecological receptors similar to Zn. The strong correlations between metals such as Co, Cr, and Ni (Figure 4A) suggest that these contaminants could have industrial or agricultural origins, further compounding the risks to soils intended for agri-food initiatives.

### Newfoundland

Soils from sites in the Newfoundland study region are contaminated with As, primarily from the weathering of bedrock, where As occurs mainly as inorganic arsenite (As(III)) or arsenate (As(V)), with minimal concentrations of organic As compounds. These forms of As are associated with other metals or compounds, such as arsenopyrite (FeAsS), gold (Au), antimony (Sb), and iron (Fe) (Serpa et al., 2009). The region's geological profile consists of bedrock

containing high concentrations of pyrite, which includes FeAsS, Fe, and Cu, likely contributing to elevated As concentrations in the soil (Government of Newfoundland, 2019). A post hoc correlation analysis revealed that Fe was positively and significantly correlated with As ( $r = 0.73$ ,  $p = 0.002$ ), suggesting that high concentrations of Fe are associated with high concentrations of As in our study region, consistent with the geology of Newfoundland (Serpa et al., 2009). The mean As concentration (31.29 mg/kg) surpasses the CCME threshold for metals in agricultural soils by 2.6 times, raising concerns for agri-food initiatives in Newfoundland. In a comparable geological area in Newfoundland, mean As concentrations in agricultural (tilled) soils were 24.70 mg/kg (Serpa et al., 2009). Furthermore, the pollution indices (Figure 5; Table 2) characterize the soil as highly polluted, with a high *Cd* (31.29), ultra-high *CSI* (5.40), and strong *RI* (312.87). Other detected metal(loid)s in the soil (Table S4) were likely of natural origin due to weathering rather than anthropogenic activities, with little to no concern according to the pollution indices, except for nickel (Ni), which had a moderate *Cd* (13.18) and a high *CSI* (3.24), possibly due to contamination or measurement error.

The elevated As levels present significant risks to agri-food initiatives by disrupting plant growth and nutrient uptake. The dominant form of As in this region, As(V), competes with phosphate absorption in plants, impairing critical biological functions such as energy transfer and photosynthesis, while As(III) exhibits toxicity through its interaction with cellular proteins (Abbas et al., 2018). These disruptions can reduce crop yields and food quality, with the potential to jeopardize food security in affected regions. More so, the island of Newfoundland already faces limitations for agri-food initiatives due to rugged and often steep terrain, and approximately 40 to 50% of the island has detectable inorganic As contamination due to bedrock minerals (Government of Newfoundland, 2023).

### Soil contamination by organochlorine pesticides

Assessing OC pesticides revealed varying concentrations across the study regions. While Australia showed no detection of these compounds, Newfoundland had nine detects of five pesticides, and the subarctic Ontario region showed higher amounts of

anthropogenic pesticide residues with a total of 158 detects of 10 pesticide residues (Table S5).

DDT breaks down to DDE aerobically and converts to DDD anaerobically; the ratio (range 0.13–1.93) presented in Figure 8 indicates environmental input (Chattopadhyay & Chattopadhyay, 2015). If the ratio exceeds 1.00, then this suggests a more recent use or input (e.g., atmospheric transport). Note that this ratio (Figure 8) is an inference and not a definitive indicator of historical versus modern use, but it is an acceptable reference for DDT input into soil. We also infer that there was more recent use of endosulfan sulfate in subarctic Ontario (Chattopadhyay & Chattopadhyay, 2015). It was typical for endosulfan sulfate to be used (Weber et al., 2010) following the outright use ban of DDT (c. 1985) and dieldrin (c. 1987; Government of Canada, 1995, 2009). The use of endosulfan sulfate was banned in Canada in 2011 (Government of Canada, 2011).

DDT and its metabolites continue to be an environmental contaminant in subarctic Canada (AMAP, 2017), despite being banned in North America for nearly 40 years and not used in this study region for more than 50 years (Environmental Sciences Group [ESG], 1999). DDT was used globally to help control insect pests and vectors of infectious diseases (e.g., malaria), and there continues to be atmospheric transport of DDT. Both DDT and its main metabolite, DDE, are highly persistent organic compounds and have a long half-life in almost all environmental milieu. In subarctic Ontario, DDT was used historically as a chemical spray to combat mosquitoes and black flies (ESG, 1999), especially in the vicinity of the MCRL Site 050 (Tsuji et al., 2001, 2006). Aside from the long-range transport of pesticides (Shen et al., 2005) and the MRCL Site 050, we note that other point sources of DDT in soil in this region are not fully elucidated to date.

The  $\overline{PI}$  and  $PI_N$  results for pesticides in subarctic Ontario highlighted the soil pollution of DDT and its metabolites indicating the soil was polluted or heavily polluted with these pesticides. Soils in this region are heavily contaminated (Table 3) with *p,p'*-DDT and both of its metabolites (*p,p'*-DDE, DDD) and *o,p'*-DDT; the  $PI_N$  result (41.40) for *p,p'*-DDT exceeds the upper classification of the  $PI_N$  (3.00) by nearly 14 times, and its metabolite *p,p'*-DDE exceeds it by almost 10 times. This indicates that these pesticides continue to persist in the environment, raising

concerns about their long-term impact on the integrity of agri-food initiatives in the region.

The presence of detectable levels of OC pesticides in this region raises concerns about potential cumulative impacts on soil health and ecosystem integrity, and poses risks to agri-food initiatives, as their persistence in the environment can compromise both soil health and crop yields. Pesticides, once absorbed into the soil, can significantly reduce microbial diversity and disrupt enzymatic activity, both of which are crucial for maintaining soil fertility and supporting healthy crop growth (Mishra et al., 2021; Singh & Patel, 2021). For example, DDT's persistence in the soil disrupts microbial processes and nutrient cycling, posing a threat to the sustainability of agri-food initiatives (Megharaj et al., 2000). Furthermore, the bioaccumulation of DDT in food chains endangers biodiversity and food security, further compromising agri-food initiatives (Holm et al., 2006; Megharaj et al., 2000).

## Recommendations

Based on the findings of this study, several recommendations are proposed to address the ecological risks posed by soil contamination on Indigenous lands in both Australia and Canada. Regular soil monitoring programs should be established in Indigenous communities engaged in agricultural activities to detect and manage pervasive contaminants like As, Pb, and DDT, ensuring long-term soil health and food safety. Where contamination exceeds acceptable levels, appropriate remediation strategies—such as soil replacement, phytoremediation, or chemical stabilization—should be implemented when and where possible, with a focus on cost-effective solutions for rural and remote areas. For example, the high cost of soil contamination analysis, sometimes exceeding \$1000 per sample, poses a significant barrier to assessing the soil quality for agri-food initiatives. Indigenous Peoples sharing knowledge of their land prior to soil sampling may lower the overall cost of soil analysis by identifying locations or expressing which toxicants are suspected or of greatest concern (Berkes & Berkes, 2009; Borthakur & Singh, 2021), which could focus the investigation and reduce costs. Additionally, contaminant threshold levels could be re-evaluated and adapted to reflect the specific geological and environmental conditions of rural Indigenous

lands, as existing standards may not account for regional variations. Soil remediation or locally tailored agri-food initiatives are also essential to ensure the long-term sustainability and ensure food security (Moriarity et al., 2024).

Governments and environmental agencies must offer greater support for Indigenous-led agri-food initiatives, providing technical assistance and access to funding for remediation efforts to enhance food security and health equity. These recommendations could help to empower Indigenous communities to protect ecosystem health and food security in the face of soil contamination.

### Limitations

Myriad factors that encompass soil contamination (e.g., historical land use, soil profiles, past mining, geology, and airborne pollutants) may skew contamination levels, and our analysis does not reflect the individual input of contaminants in the soil. The use of cross-sectional data also limits insights into how contamination changes over time, since it provides a measure of contamination from one time point rather than a full continuous time series. The uncertainty of the results is not directly addressed due to the cross-sectional data, limiting temporal assessment. While standard practices were used, some inherent uncertainty exists, especially for low-concentration contaminants. The SARS-CoV-2 (COVID-19) pandemic was also a factor that limited travel and data collection for nearly 3 years in all study regions, and we cannot account for soil contaminant changes during that time. Additionally, pollution indices oversimplify complex environmental data and cannot absolutely differentiate between anthropogenic and natural sources of pollution, potentially obscuring significant variations in pollution sources. Despite these limitations, we are confident that our findings show original results for a global assessment of agricultural soil contamination on Indigenous lands.

### Conclusion

Soil contamination is a concern for Indigenous Peoples and their agri-food initiatives, and has the potential to negatively impact ecosystem health and local agricultural productivity. In this study, there is soil

contamination in rural and remote Indigenous lands, but they differ by global region: metal(loid) concentrations are elevated in Australian soils, and soils studied in Canada are either contaminated with pesticides or toxic metal(loid)s. These regional differences call for the development of localized strategies for soil monitoring and remediation. Given the results of this study, and the increasing awareness of legacy and emerging contaminants in rural and remote regions globally, we recommend vigilance and the testing of all soils and sites intended for agri-food initiatives.

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**Author contribution** R.M. designed the methodology, conducted the data analysis and prepared the figures. M.W. and L.T. collected and curated the data. R.M. wrote the initial draft of the manuscript, while M.W., A.S., E.N., and L.T. contributed to reviewing and editing the manuscript. All authors reviewed and approved the final manuscript.

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**Data availability** No datasets were generated or analysed during the current study.

### Declarations

**Ethical approval** Not applicable

**Competing interests** The authors declare no competing interests.

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