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## Metal concentrations in rural topsoil in South Carolina: Potential for human health impact

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### Abstract

Rural areas are often considered to have relatively uncontaminated soils; however few studies have measured metals in surface soil from low population areas. Many metals, i.e., arsenic (As), lead (Pb), and mercury (Hg), have well-documented negative neurological effects, and the developing fetus and young children are particularly at risk. Using a Medicaid database, two areas were identified: one with no increased prevalence of mental retardation and developmental delay (MR/DD) (Strip 1) and one with significantly higher prevalence of MR/DD (Strip 2) in children compared to the state-wide average. These areas were mapped and surface soil samples were collected from 0–5 cm depths from nodes of a uniform grid laid out across the sampling areas. Samples were analyzed for As, barium (Ba), beryllium (Be), chromium (Cr), copper (Cu), Pb, manganese (Mn), nickel (Ni), and Hg. Inverse distance weighting (IDW) was used to estimate concentrations throughout each strip area, and a principal component analysis (PCA) was used to identify common sources. All metal concentrations in Strip 2, the MR/DD cluster area, were significantly greater than those in Strip 1 and similar to those found in more urban and highly agricultural areas. Both Strips 1 and 2 had a high number of significant correlations between metals (33 for Strip 1 and 24 for Strip 2), suggesting possible similar natural or anthropogenic sources which was corroborated by PCA. While exposures were not assessed and direct causation between environmental soil metal concentrations and MR/DD cannot be concluded, the high metal concentrations in areas with an elevated prevalence of MR/DD warrants further consideration.

### Keywords

Heavy metals; rural surface soils; spatial distribution; mental retardation and developmental delay; neurotoxicity; soil contamination

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## 1. Introduction

Environmental surface soil contamination often is examined in populated urban areas with numerous industrial facilities (Chatterjee and Banerjee, 1999; Imperato et al., 2003; Duzgoren-Aydin et al., 2006) or in areas with a high density of agricultural land use (Möller et al., 2005; Rodríguez Martín et al., 2006). While more people are potentially at risk from metals in densely populated industrialized areas, risk from environmental soil contamination in less populated and less industrialized areas may exist and individuals in these rural areas may have greater exposure to soils than their urban counterparts.

Relatively few published studies have measured soil metal concentrations in rural and small-scale agricultural areas. While a source of metal contamination can be identified in some residential locations, i.e., chromated copper arsenate (CCA)-treated wood structures are associated with high concentrations of arsenic (As), chromium (Cr), and copper (Cu) in adjacent soils (Townsend et al., 2003), non-point sources are often more difficult to quantify. Industrial, atmospheric, and other non-point sources may contribute to soil contamination in rural locations, in addition to agricultural and natural inputs.

As part of an on-going study we identified a cluster of mental retardation and developmental delay (MR/DD) prevalence in children that was significantly higher than that of the state-wide population in South Carolina (SC). Data included only MR/DD cases of unknown cause, and were spatially analyzed based on the address location of the mothers during each month of pregnancy (Zhen et al., 2008). Fifty percent of mental retardation (MR) cases have unknown causes (McDermott et al., 2007). One possible contributing factor is maternal and childhood environmental exposure to neurotoxic metals. Several metals including As, lead (Pb), and Cr can be present in surface soils and are of particular concern because they have all been shown to negatively impact neurological development of children (Factor-Litvak et al., 1999; Stewart et al., 2003; Wasserman et al., 2004). Even metals that are essential to human biological function, such as manganese (Mn) and Cu can be toxic when ingested or inhaled at elevated concentrations. Other metals, such as beryllium (Be), can remain in the body for years and their association with neurological impacts is unknown. In all cases, bioavailability of the metals is important in determining toxicity (Caussy et al., 2003).

Associating a specific environmental contaminant with a specific health outcome is difficult, and we do not assume the metals measured in this study are the cause of the MR/DD outcomes in the cases with unknown causes. This study did not measure specific exposures to individual mother-child pairs by residence during pregnancy. Instead we measured metal concentrations in soils sampled on a grid system throughout two areas; a MR/DD non-cluster and a MR/DD cluster area to determine whether there were differences in concentrations and spatial distribution of metals, and whether the source of the metals, i.e., natural or anthropogenic, was different between areas. Metals in surface soils serve as good indicators of general environmental insult because they are stable and long-lived in the environment, accumulate in soils over time, and are from both natural and anthropogenic sources. While direct causation cannot be established from chemical data alone, information on metal concentrations in surface soils may help identify whether unfavorable environmental conditions exist, and whether individual metals or co-occurrence of specific metals should be investigated in future studies for potential associations with MR/DD.

## 2. Materials and Methods

### 2.1. Site Description

South Carolina Medicaid data were used to identify maternal residence during each month of pregnancy. These data and the follow-up health outcome of the infants born to these women

were used to identify areas with a higher incidence of MR/DD outcome than the state-wide norm. Children with known causes of MR/DD were excluded from this study. Advanced spatial analysis, which incorporated Bayesian kriging and local likelihood clustering methodology, was used to identify the irregularly shaped non-cluster and the cluster areas, and gradients of MR/DD risk within these cluster areas (Zhen et al., 2008). Thus the size of each area was not identical.

In both areas, the identity of the specific residences was known in order to carry out the spatial statistical clustering analysis. However due to issues of confidentiality, neither the identity of the specific individuals, nor the specific location of the residences was known for the purposes of soil sampling. Therefore no soil was measured at a specific residence with a specific health outcome identified. Instead a uniform grid was established in the non-cluster and cluster areas from which soil samples were collected.

The non-cluster area, Strip 1, was approximate 440 km<sup>2</sup>. Two small towns (total population ~15,000) were present in Strip 1, containing both residential and industrial areas. Outside these towns, the majority of land cover and land use was forests and crops, and pasture, respectively. The cluster area, Strip 2, was approximately 145 km<sup>2</sup>. A moderately-sized town (population 22,000) was located within Strip 2. Land cover and land use consisted primarily of some deciduous and evergreen (mixed) forests, as well as residential, commercial, and industrial areas. Both Strips 1 and 2 had similar land cover and use.

An “urban” area is defined as an area located within the boundaries of either an “urbanized area” or “urban cluster” (U.S. Census Bureau, 2002). The Census Bureau’s definition of an urbanized area is a land area that has a residential population of at least 50,000, an overall population density of at least 1,000 people per square mile, and surrounding census blocks that have an overall density of at least 500 people per square mile. Urban clusters encompass similar population densities as urbanized areas, but contain less than 50,000 people. A rural area is defined as an area not classified as an urbanized area or urban cluster. Both Strips 1 and 2 are located entirely in “rural” areas as defined by the U.S. Census Bureau.

## 2.2. Experimental Design

The coordinates for each strip area were mapped using ArcGIS® (Version 9.1, ESRI, Redlands, CA, 2005). Layer files including cities, counties, roads, land cover, and land use were also mapped. A uniform grid with 120 nodes was laid out over each strip area. The grid nodes were, therefore, independent of the locations of MR/DD cases due to confidentiality requirements as previously described. Some grid nodes fell in areas at which a soil sample could not be collected, i.e., inaccessible due to highways, water bodies, etc. In these cases the soil sample was collected as close to the node location as possible and thus soil sampling locations were not evenly spaced in all cases. In each strip, 119 soil samples were collected as close as possible to each grid sampling node, approximately 1.5–3 km apart for Strip 1 and 1–1.6 km apart for Strip 2. A portable GPS device (Garmin Etrex, Olathe, KS) was used to determine the actual latitude/longitude for each sampling location and these data were also mapped with ArcGIS®.

Prior to soil collection, leaf litter or other organic debris was removed from the ground surface. Approximately 30-g grab samples of topsoil were then collected from 0 to 5-cm depth using a sterile sampling spatula (Bel-Art Scienceware, Pequannock, NJ) and placed in a sterile Whirl-Pak® bag (Nasco, Fort Atkinson, WI) (Aelion and Davis, 2007). The samples were stored on ice and refrigerated in the lab upon return. Duplicate samples were collected at 10 % of the sampling locations for QA/QC.

After sampling, 60 samples from Strip 1 and 119 samples from Strip 2 were analyzed for metals by an independent analytical laboratory (Pace Analytical, Huntersville, NC). For both strips,

the samples were digested with nitric and hydrochloric acids (EPA method 3050B) and analyzed for As, barium (Ba), Be, Cr, Cu, Pb, Mn, and nickel (Ni) (EPA method 6010B) and Hg (EPA method 7471A) using inductively coupled plasma-emission spectroscopy (ICP-ES). In addition, Strip 1 was sampled for antimony (Sb) and cadmium (Cd) for which 98% and 100% of the samples, respectively, were below minimum detection limits. Since Sb and Cd were not detected in almost all samples in Strip 1, these metals were not measured in Strip 2 soil samples. Samples that were below detection limit were handled conservatively, and quantified as 0 mg kg<sup>-1</sup> for data analysis. Eight duplicate sets of soil samples were analyzed for As, Ba, Be, Cr, Cu, Pb, Mn, and Ni and nine duplicate sets were analyzed for Hg; Pace also analyzed blank surrogate soil samples of clean filter sand, and method blank, lab control and matrix spiked samples for laboratory QA/QC. Detection limits varied for each analyte, but were approximately 0.5 mg kg<sup>-1</sup> for As, Ba, Cr, Cu, Pb, Mn, and Ni, 0.1 mg kg<sup>-1</sup> for Be, and 0.005 mg kg<sup>-1</sup> for Hg.

Analysis of variance (ANOVA) was used to compare Strip 1 and 2 data for each metal, and Pearson correlations (non-parametric) were used to compare metals within each strip using SAS® Version 9.1 (SAS Institute, Cary, NC, 2002–2003). An alpha ( $\alpha$ ) level of 0.05 was used to determine significance.

Maps of estimated metal concentrations for each strip individually were generated using inverse distance weighted (IDW) analysis using ArcGIS® software, a spatial interpolation method that estimates values for a surface by weighting sampling points based on their proximity throughout the sampled area. Contour plots identify potential hot spots and spatial variability in the measured concentrations of each metal. For the IDW analysis, the power (value of the significance of nearby points) was optimized for each metal, and the standard analysis was used with at least 15 neighbor points, one sector, no ellipse angle, and an anisotropy factor of one. To verify the validity of predicted values and assess uncertainty, the root mean square prediction error (RMSPE), the mean prediction error (MPE), the mean standardized prediction error (MSPE) and the average standard error (ASE) were calculated using ArcGIS®. The RMSPE quantifies the error of the predicted surface. The MPE assesses whether the data are unbiased and should be a value near 0. Because the MPE is a function of the scale of the data, the MPE is standardized to the MSPE by dividing the prediction errors by their prediction standard errors. The MSPE also should be a value near 0. The variability of the prediction is assessed by comparing the ASE and the RMSPE. If the ASE is greater than the RMSPE, the variability of the predictions are overestimated; the reverse holds true if the ASE is less than the RMSPE. Finally, a principal component analysis (PCA) was carried out with SAS using the metal data from each strip separately, to cluster variables that behaved similarly to identify potential similar source. Only factors with eigen values greater than 1 were used in the PCA analysis.

### 3. Results

For all metals, the mean values, concentration ranges, and quartile ranges were greater for Strip 2 than for Strip 1 soils (Table 1). In all cases, the individual metal concentrations from Strip 2 were significantly greater than Strip 1 ( $p < 0.05$ ). Most metal mean concentrations for Strip 2 were at least two times as high as those for Strip 1. The largest difference in means was for Mn, which was six times higher for Strip 2 than for Strip 1. For all metals except As, Pb, and Hg, the 25 % quartile value for Strip 2 was greater than the 75 % quartile value for Strip 1 (Table 1). Strip 1 had a higher percent of soil samples in which As, Be, Ni, and Hg were not detected than Strip 2. All other metals were detected in 100 % of the soil samples in both strips.

The EPA Region 9 has set suggested Preliminary Remediation Goals (PRG) for residential soil limits (RSL) and industrial soil limits (ISL) that are used by the state of South Carolina. Of the

nine metals measured in the rural soils, 82% and 93 % of the soil samples exceeded the EPA PRG RSL for As ( $0.39 \text{ mg kg}^{-1}$ ) in Strip 1 and Strip 2, respectively. 30 % of samples from Strip 1 and 47 % of samples from Strip 2 also exceeded the EPA PRG ISL for As ( $1.6 \text{ mg kg}^{-1}$ ) (data not shown). Manganese concentrations varied widely at our study sites, and the highest Mn concentration measured in Strip 2 was approximately three times greater than the EPA PRG for residential soils (Table 1); 4% of Strip 2 soil samples exceeded the EPA PRG RSL ( $1800 \text{ mg kg}^{-1}$ ) for Mn.

City names and MR/DD cases and controls cannot be shown due to confidentiality considerations, and only generalized city boundaries and roads are shown in the maps (Figures 1–Figures 3). Results of IDW of 3 of the 9 metals were selected because they represent 1) high concentrations that exceeded the EPA PRG RSL (As), and 2) differences in potential sources indicated by results of the PCA analysis (Hg and Mn). Concentration intervals chosen for the metals were based on quartile values for combined Strip 1 and 2 data for each metal. For Strip 1, the RMSPE, ASE, and MSPE values for As were 1.31, 1.305, and 0.00903, respectively; for Hg and Mn, the values were 0.0242, 0.0236, 0.00544 and 249.9, 264, 0.0268, respectively. Strip 2 RMSPE, ASE, and MSPE values for As were 2.37, 2.01, and  $-0.00243$ , respectively, and values for Hg and Mn were 0.0258, 0.0277, 0.00989, and 635.7, 529,  $-0.0148$ , respectively.

Overall, metal concentrations varied spatially though some had higher localized concentrations near cities. The area of Strip 1 was larger than that of Strip 2 so the sizes of the IDW hot spots are not directly comparable. Strip 1 appeared to have a few hotspots of high concentrations but generally had low estimated metal concentrations throughout the majority of the strip area, while Strip 2 had more areas of high metal concentrations that were located near city boundaries for most metals. For Strip 1, As (Figure 1a), Hg (Figure 2a), and Mn (Figure 3a) had localized hotspots located in two common areas: one just outside the southeastern end of the city boundary, and one outside the city boundary in the western portion of the strip. Most estimated concentrations were in the two lowest ranges for all three metals in Strip 1, although a large area in the eastern part of Strip 1 had estimated As concentrations in the  $1.41\text{--}2.1 \text{ mg kg}^{-1}$  range (Figure 1a). For Strip 2, estimated As (Figure 1b) concentrations were high within city boundaries, with some hotspots located in the strip as well as a large area in the highest range in the northern strip area. For Hg (Figure 2b), high concentrations were located almost exclusively within or near city boundaries, with only a few additional hotspots. A decrease in estimated concentrations can be observed from south to north. Manganese concentrations in Strip 2 (Figure 3b) were high in the city boundaries but a large area with concentrations in the highest range ( $> 485 \text{ mg kg}^{-1}$ ) was also located north of city boundaries. Overall the presence of roads did not indicate hotspots in either strip.

In general, Strip 1 had more significant correlations between metals and a higher maximum correlation coefficient than Strip 2 (data not shown). Based on Pearson correlation coefficients ( $r_s$ ) and p values, Strip 1 had 33 significant correlations ( $p \leq 0.05$ ) between metals. The maximum correlation coefficient was 0.8 for As-Cr and for Be-Cr. The 6 metal pair correlations  $\geq 0.7$  in Strip 1 in addition to As-Cr and Be-Cr included As-Ba, As-Be, Ba-Cu, Ba-Mn, Ba-Ni, and Cu-Ni. All metals were correlated with at least one other metal. Strip 2 had 24 significant correlations between metals (data not shown). The maximum correlation coefficient for this strip was 0.76 for Ba-Be. The 3 metal pair correlations of  $\geq 0.7$  in Strip 2 in addition to Ba-Be included Be-Ni, Cr-Ni and Cu-Ni, and as for Strip 1, all metals in Strip 2 were significantly correlated with at least one other metal.

Due to the high numbers of correlated metals for both strips, PCA was run for each strip separately to assess potential common sources (Figure 4). For Strip 1, three factors were retained based on an eigen value  $> 1$ , and the first two factors explained 69 % of the variance (Figure 4a). Results of PCA indicate three separate groupings of metals: Ba, Be, Cr and Mn;



As and Cr; and Ni, Hg, and Pb. For Strip 2, three factors were again retained based on an eigen value > 1, and the first two accounted for 52 % of the variance (Figure 4b). For Strip 2 six of the metals were loosely clustered and included Ba, Be, Cr, Cu, Mn, and Ni. A second grouping included As, Hg, and Pb for which Hg and Pb were tightly associated. In both strips, Ba, Be, Cu, and Mn were grouped similarly, as were Hg and Pb. However, since As, Cr, and Ni were grouped differently in Strips 1 and 2, which suggests different sources for these metals in each strip.

#### 4. Discussion

Limited information is available on background metal concentrations of surface soils in rural areas. Two studies carried out in South Carolina on a much large spatial scale (Shacklette and Boerngen, 1984; Canova, 1999) reported concentrations similar to our measured ranges (Table 2). However, some of our Strip 1 and most Strip 2 maximum metal concentrations exceeded their reported ranges for As, Ba, Cr, Cu, Mn, Ni, and Hg. Both Strips 1 and 2 had maximum Cr, Mn, and Ni concentrations that exceeded Shacklette and Boerngen's (70, 500, and 30 mg kg<sup>-1</sup>, respectively). Strip 2 also had maximum As and Cu concentrations that exceeded those reported by Shacklette and Boerngen (10 and 50 mg kg<sup>-1</sup>, respectively). Strip 2 maximum Ba and Cu exceeded the maximum concentrations of 370 and 92 ppm, respectively reported by Canova. In addition, the maximum Mn concentration (5100 mg kg) in Strip 2 was approximately two times greater than Canova's maximum (2400 ppm) and the Strip 2 mean Mn (520 mg kg<sup>-1</sup>) was four times greater than Canova's statewide average (120 ppm). Canova also reported higher percents of soils with non-detectable concentrations of As, Ba, Be, Cr, Cu, Pb, and Ni as compared to the current study.

While urban and highly agricultural areas are often assumed to have much higher soil metal concentrations as compared to rural areas due to the high densities of potential sources (people, cars, industries, etc.), our rural soil metal concentrations in Strip 2 were comparable to concentrations found in urban and high-density agricultural soils. Our As concentrations were greater than would be expected in non-industrialized areas, and exceeded the EPA PRG RSL, and the ISL in some cases. Our Pb mean and maximum and our Hg maximum from Strip 2 were greater than those measured by Rodríguez Martín et al. (2006) in agricultural soils in Spain. Chromium and Cu concentration ranges in Strip 2 soils were similar to and our Pb range and mean concentrations in Strip 2 were higher than those measured by Möller et al. (2005) in urban and agricultural Syria (Table 2). Li et al. (2004) measured mean metal concentrations in urban soils of Hong Kong, which were only 1.3 times the Cr, 1.9 times the Cu, 2.2 times the Ni and 3.2 times the Pb measured in Strip 2. In addition our metal concentration ranges were similar to theirs and our maximum Cr concentration in Strip 2 was greater than their maximum Cr concentration. However As, Cd, Pb, and Hg concentrations reported by Diawara et al. (2006) in Pueblo, CO, USA, an area which has had a long history of mining and smelting operations, had greater minimum and maximum concentrations than those measured in our study (Table 2).

Metals in surface soils may be distributed over large areas or localized in smaller areas. The spatial scale of the current study did not evaluate heterogeneity of small-scale (on the order of meters) local variability or large-scale regional variability due to differences in geologic setting. Our spatial scale was similar to those investigated by Diawara et al. (2006) and Li et al. (2004), on the order of kms. Both our strips were in similar regional areas and thus can be compared to each other.

Using the IDW, the uncertainty, bias and variability of the predicted concentrations can be assessed which addresses the short-scale variability of the estimated concentrations. Our IDW statistical results indicate that the models were unbiased, as the MSPE for As, Hg, and Mn by

strip were all close to 0. Also, the RMSPE and ASE values were similar for each metal, which indicates that the variability in the predictions was assessed correctly in the models. In all cases except for Mn in Strip 1, the RMSPE was essentially equal (~1% difference) to the ASE, which suggests no under or over estimation of predicted values for As and Hg. For Strip 1 Mn, the ASE>RMSPE which indicates slight overestimation while for Strip 2 Mn, the opposite occurred; however RMSPE and ASE values were within 6% of each other for Mn for both strips.

The mapping of estimated concentrations by strip indicated that more localized hotspots were located throughout Strip 1, while Strip 2 had large areas with estimated concentrations in the highest ranges, some of which fell near or within the city boundaries as was evident for As and Hg. In contrast, Mn had some hotspots of high concentrations similar to As and Hg in Strip 1, but in Strip 2 a large area in the northern part of the strip outside the city boundaries had estimated Mn concentrations in the highest range. Li et al. (2004) concluded that anthropogenic metal concentrations for Cu, Ni, Zn and Pb were located at the junctions of major roads or next to major roads with large amounts of traffic; our sites had limited road infrastructure and roads did not appear to account for the hot spots.

Correlations between metals may indicate common sources, natural or anthropogenic, and multiple metal co-occurrence may be an important consideration for assessing the possibility of health effects from complex mixtures of metals (Carpenter et al., 2002). We found significant correlations (up to 0.8) in both strips as did Rodríguez Martín et al. (2006) in agricultural top soils in Spain. Most of the metals measured in our study could potentially have natural and anthropogenic sources. Our PCA results for Strips 1 and 2 suggest that the metals fell into two specific groups for each strip: one that contained Ba, Be, Cu and Mn, and one that contained Hg and Pb. It is probable that Ba, Be, Cu, and Mn are naturally occurring since they clustered together similarly in both strips, although Cu has been associated with anthropogenic sources by Möller et al. (2005) and Li et al. (2004). Mercury and Pb appear to be from anthropogenic sources in both our strips, as was concluded by several of the previously cited studies (Rodríguez Martín et al. 2006, Duawara et al. 2006, Möller et al. 2005, Li et al. 2004).

Unlike the previous metals which followed a similar clustering pattern in both strips, As, Cr, and Ni did not follow the same clustering patterns in Strips 1 and 2. Arsenic and Cr were tightly clustered only with each other in Strip 1, as might be found in CCA-treated wood structures (Townsend et al., 2003). In Strip 2, As was associated with Hg and Pb, suggesting a different anthropogenic source as compared to Strip 1. Diawara et al. (2006) determined that As was not associated with the anthropogenic metals Hg or Pb at their mining-impacted site, and that As was ubiquitous. Arsenic may be an example of a chemical with both natural and anthropogenic sources depending on the geographical location of the study site.

Nickel was loosely associated with the anthropogenic metals Hg and Pb in Strip 1, and more tightly associated with naturally occurring metals in Strip 2, while Cr was associated with naturally occurring metals in Strip 2 and clearly not associated with As. Other studies have associated Cr with natural sources (Rodríguez Martín et al. 2006), and Ni with either natural (Möller et al. 2005) or anthropogenic (Li et al. 2004) sources. Differences in land use, geology and spatial scale existed in the above-mentioned studies, and this may be reflected in the differences in the results. Also although similar patterns of spatial distributions and hot spots may indicate where potential sources are likely, additional research would be required to identify specific sources of metals.

Most of the studies associating MR/DD with metal exposure have been carried out based on high concentrations and direct exposures (Thornton et al., 1990; Wasserman et al., 2004; Factor-Litvak et al., 1999) and have not examined the role of low environmental concentrations.

We did not assess direct exposure of the mother-child pairs, but instead used environmental concentrations of metals as a surrogate for environmental insult and compared these concentrations over areas identified as a MR/DD non-cluster and MR/DD cluster area. Significantly higher concentrations of all the metals were measured in the MR/DD cluster area compared to the non-cluster area, and certain metals were higher than expected, particularly As and Mn. The MR/DD cluster area also had widespread higher estimated metal concentrations than the non-cluster area. Though the direct connection between residential, environmental concentrations of metals in soils and MR/DD has not yet been established, rural areas may have significant concentrations of metals and further investigations of metal toxicity, environmental distribution and bioavailability will help determine their potential human health effects.

## Acknowledgments

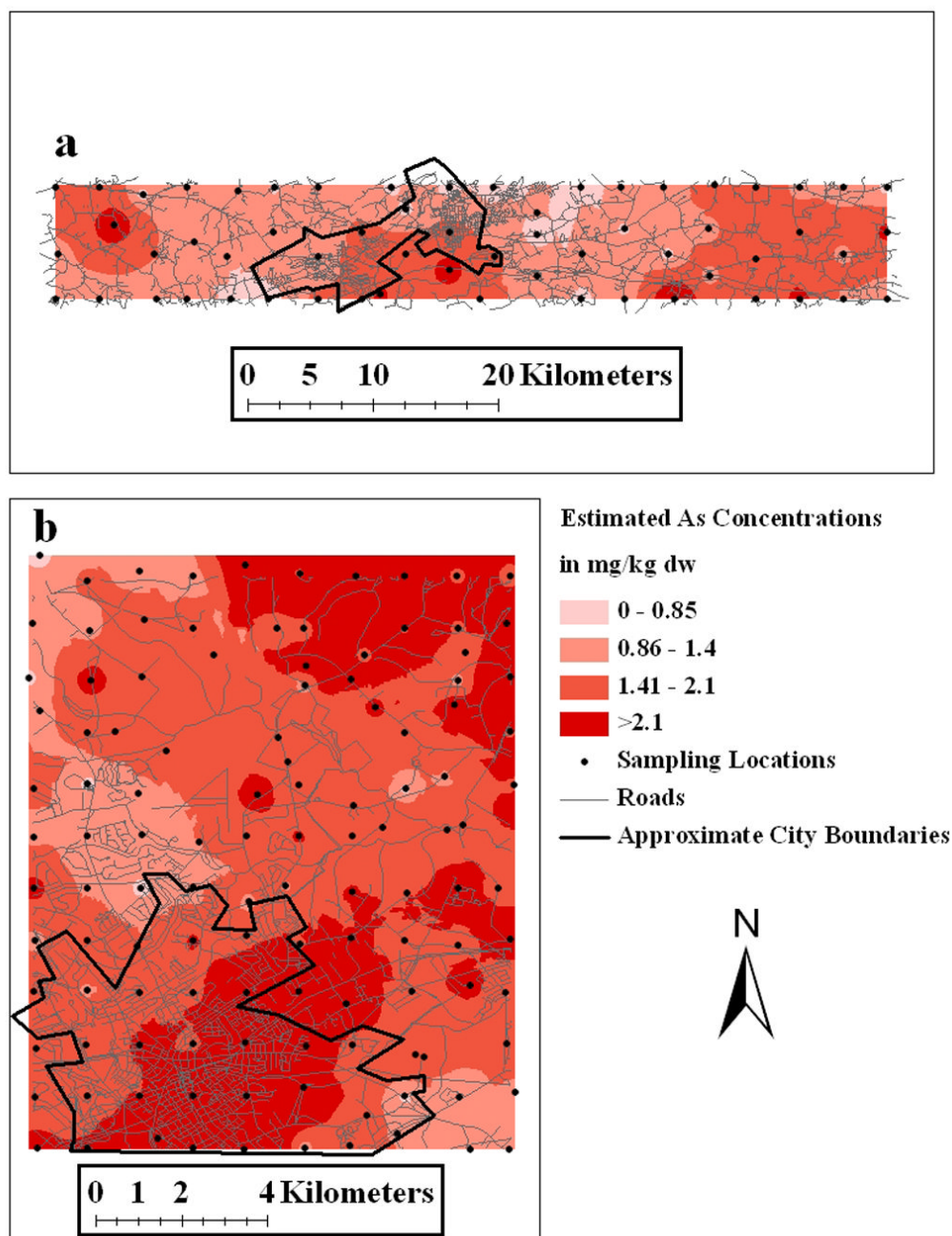
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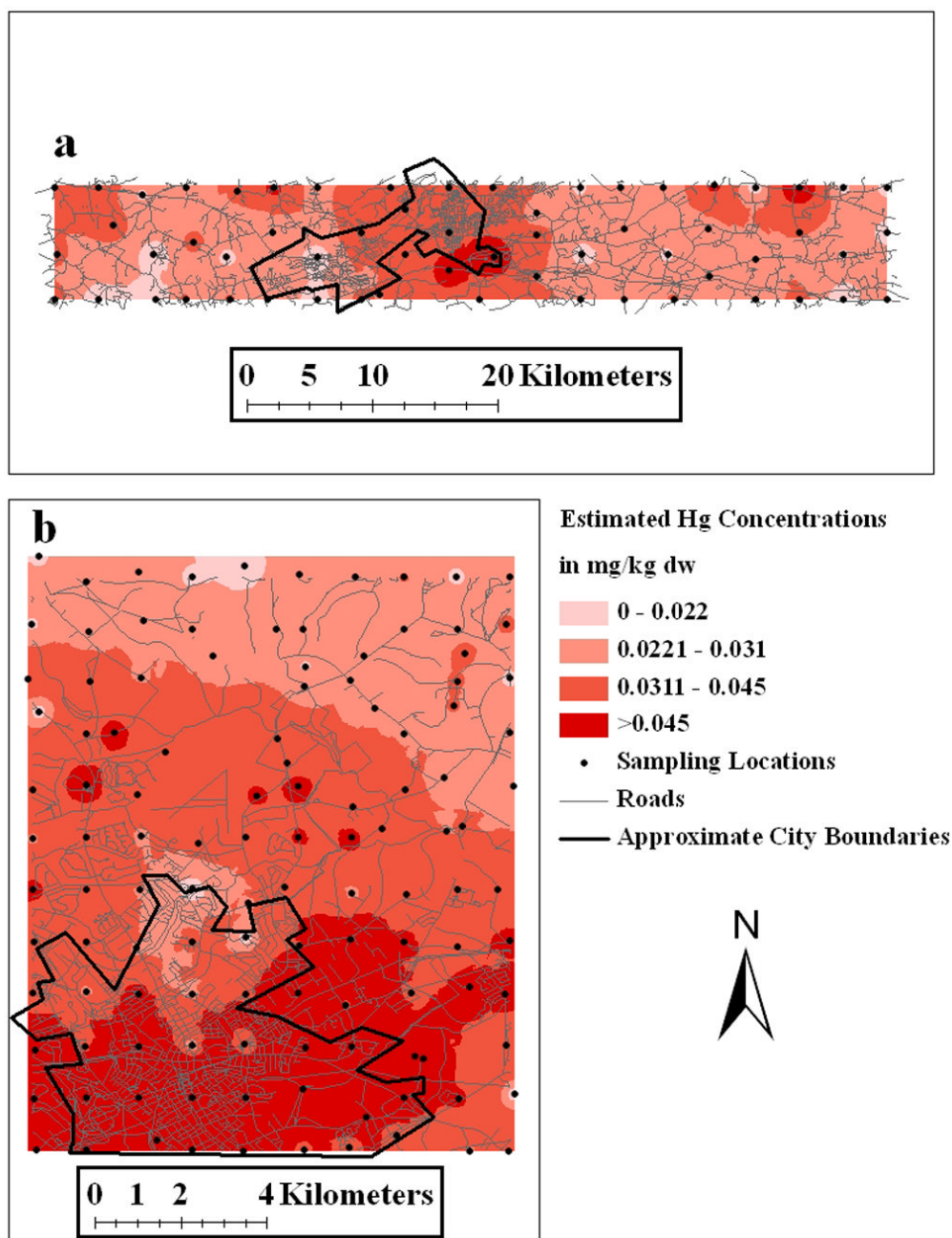
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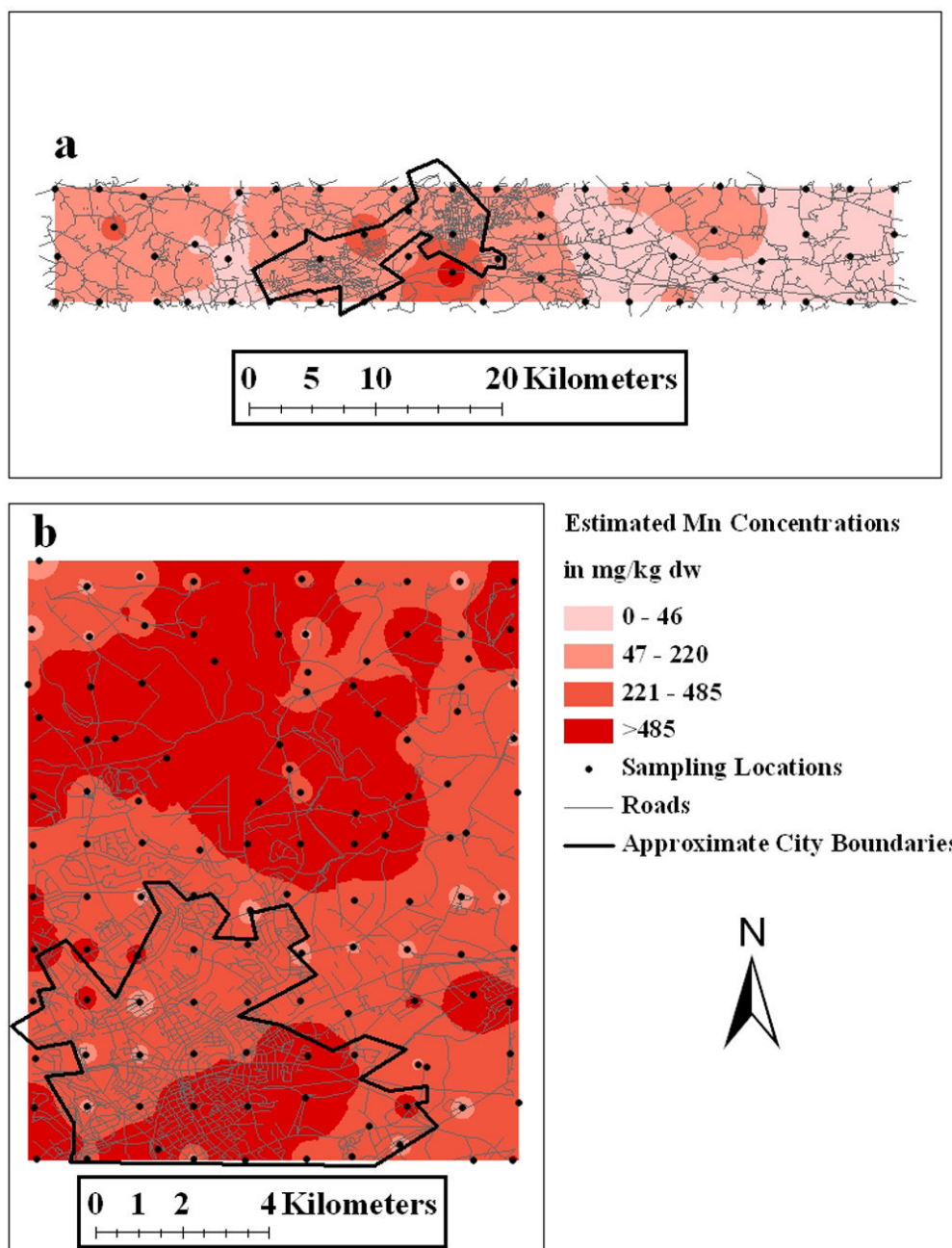
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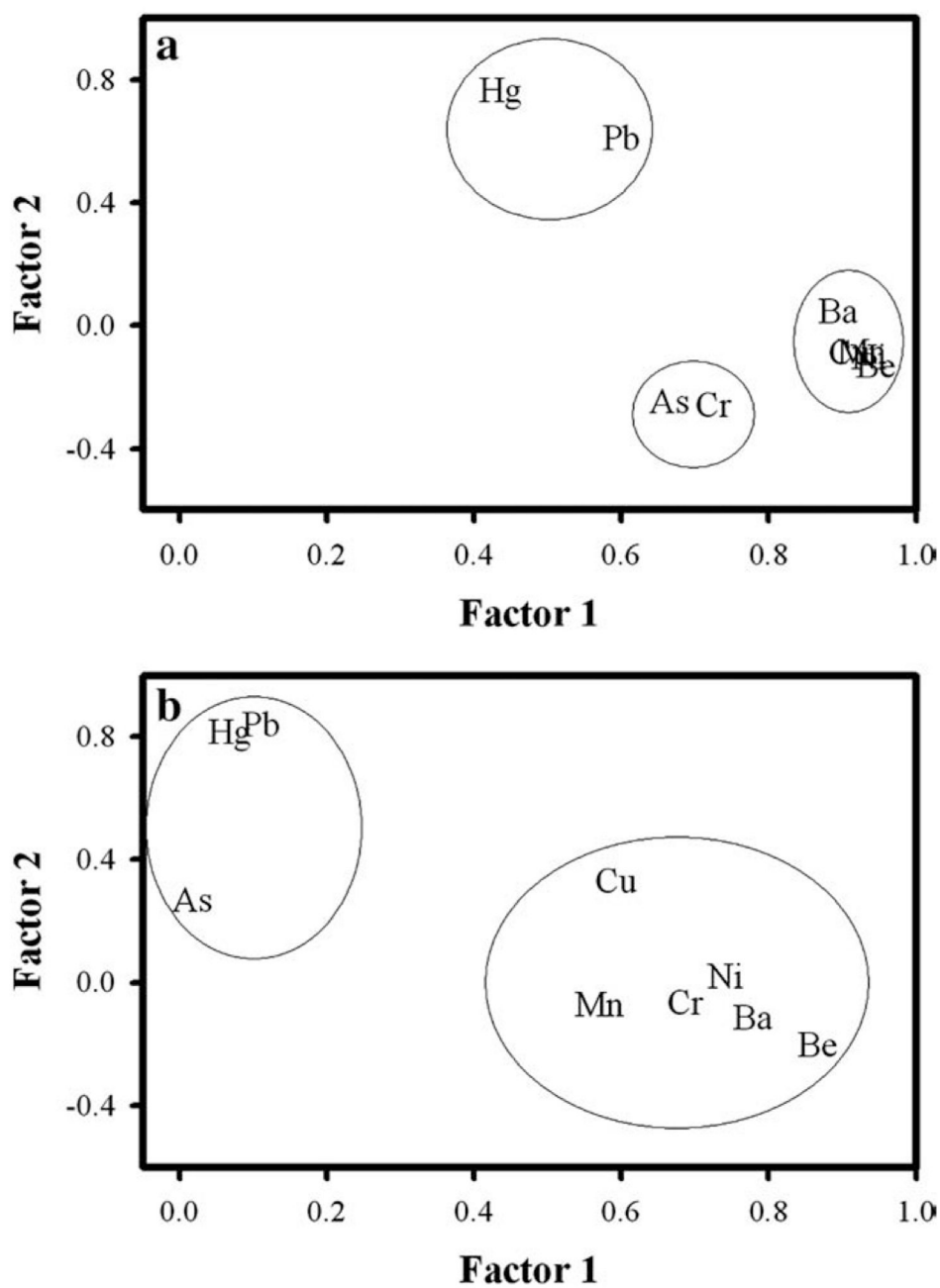
**Figure 1.**  
Estimated As concentrations ( $\text{mg kg}^{-1}$ ) in surface soils based on Inverse Distance Weighting in a) Strip 1 and b) Strip 2.



**Figure 2.** Estimated Hg concentrations ( $\text{mg kg}^{-1}$ ) in surface soils based on Inverse Distance Weighting in a) Strip 1 and b) Strip 2.



**Figure 3.** Estimated Mn concentrations ( $\text{mg kg}^{-1}$ ) in surface soils based on Inverse Distance Weighting in a) Strip 1 and b) Strip 2.



**Figure 4.** Principal component analysis (PCA) of the 9 metals plotted by the two major factors for a) Strip 1 and b) Strip 2.



Table 1

Strips 1 and 2 chemical concentration ranges, means, quartile ranges, non-detects, and p-values for comparisons between strips (ranges and means in mg kg<sup>-1</sup>) and remediation goals.

	Strip 1 <sup>1</sup>			Strip 2 <sup>2</sup>			Strip 3 <sup>3</sup>			Strip 4 <sup>4</sup>				
	Conc. Range	Mean	Quartile Range <sup>3</sup>	ND <sup>4</sup> (%)	Conc. Range	Mean	Quartile Range	ND (%)	Conc. Range	Mean	Quartile Range	ND (%)	ANOVA P-Value	EPA PRG RSL <sup>5</sup>
As	0-6.4	1.3	0.69-1.6	17	0-20	2.0	1.1-2.3	7	0-20	2.0	1.1-2.3	7	0.03	0.39
Ba	2.1-210	20	6.8-23	0	2.2-390	65	34-89	0	2.2-390	65	34-89	0	<0.0001	5400
Be	0-1.4	0.13	0-0.15	48	0-1.3	0.41	0.22-0.56	3	0-1.3	0.41	0.22-0.56	3	<0.0001	150
Cr	1-89	7.02	2.1-6.4	0	1.1-120	18	8.7-21	0	1.1-120	18	8.7-21	0	<0.0001	210
Cu	0.45-23	3.2	1.2-3.5	0	0.69-100	12	4.7-15	0	0.69-100	12	4.7-15	0	<0.0001	3100
Pb	2.1-53	12	5.1-14	0	6.5-200	30	13-34	0	6.5-200	30	13-34	0	<0.0001	400
Mn	2.7-1600	86	10-62	0	15-5100	520	200-570	0	15-5100	520	200-570	0	<0.0001	1800
Ni	0-53	2.8	0.7-1.6	10	0-39	5.5	1.7-6.1	2	0-39	5.5	1.7-6.1	2	0.02	1600
Hg	0-0.12	0.03	0.017-0.041	12	0-0.19	0.04	0.025-0.051	5	0-0.19	0.04	0.025-0.051	5	0.02	23

<sup>1</sup> n=60

<sup>2</sup> n=119

<sup>3</sup> 25 % to 75 % quartile range

<sup>4</sup> Not detected (% below method minimum detection limit)

<sup>5</sup> EPA Region 9 Preliminary Remediation Goals (PRG) residential soil limits (RSL)

Table 2  
Means and ranges from literature data for comparisons (all units in mg kg<sup>-1</sup> or ppm).

	Shacklette and Boerngen (1984) <sup>1</sup>	Canova (1999) <sup>2</sup>	Li et al.(2004) <sup>3</sup>	Möller et al.(2005) <sup>4</sup>	Rodríguez-Martínez et al. (2006) <sup>5</sup>	Diawara et al.(2006) <sup>6</sup>
As	<0.1 – 10	6.1 (ND <sup>7</sup> – 210)	NA <sup>8</sup>	NA	NA	12.5 (1.8 – 66.5)
Ba	10 – 500	59 (ND – 370)	NA	NA	NA	NA
Be	<1 – 15	0.6 (ND – 3.8)	NA	NA	NA	NA
Cr	1 – 70	29 (ND – 140)	23 (8.03 – 92)	57 (12 – 116)	20 (<0.1 – 170)	NA
Cu	<1 – 50	13 (ND – 92)	23 (0.76 – 117)	34 (16 – 97)	17 (2 – 207)	NA
Pb	<10 – 10	16 (ND – 200)	95 (12 – 360)	17 (<5 – 108)	18 (4 – 147)	88 (18 – 316)
Mn	<2 – 500	120 (ND – 2400)	NA	NA	NA	NA
Ni	<5 – 30	6 (ND – 52)	12 (5.3 – 40)	39 (24 – 58)	21 (2 – 523)	NA
Hg	<0.01 – 0.2	0.18 (ND – 0.38)	NA	NA	0.036 (0.001 – 0.72)	0.13 (<0.02 – 2.45)

<sup>1</sup> SC background soils, only ranges shown and n≈6–9 depending on metal

<sup>2</sup> SC background soils and n≈189–254 depending on metal

<sup>3</sup> Urban Hong Kong soils and n= 152

<sup>4</sup> Urban and agricultural Syrian soils and n=51

<sup>5</sup> Agricultural soils in Spain and n=624

<sup>6</sup> Urban, mining soils in Colorado and n=68

<sup>7</sup> Not detected

<sup>8</sup> Not applicable