

Article

Elemental Mercury Contamination in Soil Leading to Vapor Intrusion Impacts on an Occupied Building: A Detailed Case Study

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Abstract

Elemental mercury has been used in the production of bleach since at least 1892 and continues to be utilized in some manufacturing processes today. This case study examines a former bleach manufacturing facility in the western United States, where elemental mercury was utilized as an electrical conductor in the chlor-alkali process to produce chlorine and sodium hydroxide, essential constituents in bleach formulation. The operational practices implemented at the facility led to the discharge of elemental mercury into both soil and groundwater. Subsequent investigations identified the presence of mercury in indoor air at levels surpassing the screening thresholds established by the California Environmental Protection Agency (CalEPA) Department of Toxic Substances Control (DTSC) and the United States Environmental Protection Agency (USEPA) for commercial exposure scenarios. Additionally, these concentrations exceeded the California Office of Environmental Health Hazard Assessment (OEHHA) acute 1 h reference exposure level (REL). The origins of mercury in indoor air have been identified as vapor intrusion associated with subsurface sources, along with a potential secondary indoor air source associated with mercury deposition and adsorption in building materials through the years. In the context of interim vapor intrusion mitigation, air purifiers and fans were deployed to enhance air exchange rates, while a comprehensive assessment led to the identification and sealing of 52 preferential pathways. Even with these interim vapor intrusion mitigation systems in place, elevated concentrations of elemental mercury are still present in the building and may represent the presence of a secondary indoor air source from accumulation of elemental mercury in building materials.



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1. Introduction

In 1919, a confidential company began manufacturing sodium hypochlorite, commonly referred to as bleach, at the site using a chlor-alkali process [1]. The chlor-alkali process typically employs a metal conductor during electrolysis to produce chlorine and sodium hydroxide [2,3]. At this facility, elemental mercury served as the conductive

medium. Elemental mercury was conventionally stored in concrete retorts before use in the chlor-alkali process. Once the mercury was depleted, the porous concrete cells were frequently crushed and heated to recover residual mercury for reuse. Following mercury recovery, the crushed concrete material was repurposed as fill throughout the site, including beneath the building foundation [1,4]. By 1957, the company had transitioned from a closed-loop mercury chlor-alkali process to importing chlorine and sodium hydroxide from external manufacturers for liquid bleach formulation. This practice continued until liquid bleach production ceased in 1981 [5]. Dry bleach production began at the site in 1972 and continued until all bleach manufacturing activities ended in 1992 [1,4–9]. The property was subsequently sold in 2000 to a local business that operated the facility as a furniture showroom until 2002. The building was not remediated for mercury contamination before ownership was transferred.

Since 2002, the interior of the building has undergone subdivision into distinct units, varying in size from a modest storage closet to expansive areas exceeding 743 square meters (m^2), designated for a large industrial enterprise [7]. These units are currently leased and used as photography studios, artist workspaces, storage areas, and facilities for various small businesses [10]. The building configuration is frequently modified through the installation of temporary plywood partitions that redefine interior spaces within the former bleach manufacturing facility. The site contains approximately 76 units, of which 34 are designated for full-time occupancy, defined as four to eight hours per day. An additional 18 units are used for storage and may be occupied for up to one hour per day. Another 18 units are classified as dead-storage areas and are not intended for occupancy because indoor elemental mercury concentrations exceed the California Office of Environmental Health Hazard Assessment (OEHHA) 1 h acute reference exposure level (REL) of 0.6 microgram per cubic meter ($\mu\text{g}/\text{m}^3$). The remaining six units are currently unoccupied [10].

The site is located within a highly industrialized area of the western United States. The former manufacturing facility consists of two adjoining structures connected by a shared wall. The southern building, originally constructed in 1919, features a wood-frame structure with a wooden ceiling and roof. This building underwent several expansions between 1919 and 1968. The northern building was constructed in 1924 and consists of a two-story concrete-frame structure with a wooden roof. Together, the buildings occupy approximately 8919 m^2 . Paved parking areas and access driveways are located north and west of the buildings, with sidewalks extending along the property boundaries. Railroad tracks define the western and southwestern property boundaries, whereas a major highway forms the northern boundary. A highway sub-drainage system located northwest of the site is connected to the regional stormwater management network, which ultimately discharges to a nearby estuary.

Site elevation ranges from approximately 4.6 to 5.5 m above mean sea level (amsl). The subsurface beneath the facility consists of approximately 1.0 to 1.5 m of fill material containing concrete debris, yellow-to-orange fine sand, and loose granular deposits. Native soils exhibit a mottled gray to yellow-brown appearance and consist primarily of silty clay interbedded with granular layers ranging from clayey sands to clean sands and gravels with moderate hydraulic conductivity. These granular intervals occur at depths of approximately 2.4 to 5.5 m, 12 to 24.4 m, 36.5 to 42.7 m, and 61 to 70 m below ground surface (bgs). Across the site, the dominant native soil unit consists of highly plastic silty clay. A shallow perched groundwater zone occurs at approximately 2.4 to 5.5 m bgs, with seasonal fluctuations influenced by precipitation. This perched groundwater zone is hydraulically separated from a deeper confined aquifer by an approximately 6.7 m-thick layer of plastic silty clay. The confined aquifer occurs at depths of approximately 12.2 to 24.4 m bgs near the site.

During periods of heavy rainfall, flooding occurs on adjacent streets and sidewalks, with water levels reportedly reaching approximately 1 m above sidewalk grade. Although municipal maintenance crews routinely remove debris from nearby drainage inlets, these efforts have not resolved the recurring flooding problem. The flooding has been attributed to compromised drainage infrastructure and has persisted for more than a decade [1,4,5,7].

Seasonal hydrologic conditions may substantially influence mercury vapor intrusion. Elevated groundwater levels and transient surface ponding can alter the distribution of air-filled and water-filled pore spaces within the vadose zone, thereby affecting gas transport pathways, volatilization rates, and vapor intrusion behavior. Although increased soil moisture may reduce gas-phase permeability, fluctuations in shallow groundwater can also displace mercury-containing soil gas toward building foundations and other preferential migration pathways. Furthermore, repeated wetting and drying cycles may alter mercury partitioning among solid, aqueous, and vapor phases, thereby contributing to temporal variability in indoor air concentrations. Consequently, the interaction among shallow groundwater, soil moisture, vapor transport processes, and building pressure gradients represents a coupled hydrogeologic–contaminant transport system that remains incompletely understood for elemental mercury contamination settings.

Despite extensive literature addressing chlor-alkali contamination, mercury fate and transport, and vapor intrusion processes, several important knowledge gaps remain. First, relatively few studies have examined vapor intrusion associated with elemental mercury contamination beneath large industrial buildings that have subsequently been converted to multi-tenant commercial properties with variable occupancy patterns. Second, most vapor intrusion investigations focus primarily on subsurface contaminant sources and do not evaluate the potential formation of secondary indoor mercury reservoirs resulting from decades of mercury adsorption to concrete, wood, drywall, and other porous building materials. Third, limited field evidence exists regarding the relationships among seasonal groundwater fluctuations, localized flooding, vapor intrusion intensity, and long-term indoor mercury persistence. This case study addresses these knowledge gaps by evaluating a former chlor-alkali bleach manufacturing facility that now operates as a multi-tenant commercial property, where both subsurface mercury contamination and secondary mercury sources associated with building materials may contribute to occupant exposure.

2. Physiochemical Properties and Toxicity

2.1. Physiochemical Properties

Mercury (Hg) is a naturally occurring characterized by complex environmental behavior and significant toxicological effects. Mercury, historically referred to as “quicksilver”, exists in multiple chemical forms, each exhibiting distinct physical and chemical properties. The mobility, reactivity, and persistence of mercury make it a significant environmental contaminant and toxicant [11,12]. Mercury has been widely used in industrial applications, including the chlor-alkali process, to produce chlorine and caustic soda from brine. In aqueous environments, mercury may persist in its elemental form until undergoing redox transformations that result in the formation of inorganic or organic mercury species [13]. Elemental mercury (Hg^0) is the principal contaminant of concern at the study site.

Elemental mercury is a dense, silvery liquid at ambient temperature. It readily volatilizes to form a colorless, odorless vapor, facilitating atmospheric transport because of its relatively high Henry’s Law constant [14–17] and low water solubility [14]. Elemental mercury solidifies only under sub-zero conditions and vaporizes upon heating. These characteristics enhance its environmental mobility and complicate efforts to control emissions and human exposure at contaminated sites. Table 1 summarizes the physicochemical properties of elemental mercury.

Table 1. Physicochemical properties of elemental mercury [11,12,16,18–23].

Property	Elemental Mercury (Hg ⁰)
Molecular Weight	200.59
Density (g/cm ³) at 25 °C	13.53
Solubility in water (mg/L) at 25 °C	0.06
Diffusivity in water (cm ² /s)	3.01×10^{-5}
logKow (unitless)	0.62
Kd (L/kg)	52
Melting Point (°C)	−38.8
Boiling Point (°C)	357
Vapor Pressure (Pa)	0.26
Henry's Law constant (atm·m ³ /mol) at 25 °C	0.008622
Diffusivity in air (cm ² /s)	0.0715

The environmental behavior of elemental mercury is governed not only by its physicochemical properties but also by environmental partitioning processes. In the subsurface, elemental mercury may exist as a discrete non-aqueous liquid phase, dissolved mercury within pore water, mercury adsorbed to mineral or organic surfaces, and mercury vapor within soil gas. Because elemental mercury possesses a relatively high vapor pressure and Henry's Law constant compared with many metals, volatilization from contaminated soil or groundwater can generate persistent soil-gas sources. Adsorption processes may significantly influence mercury distribution and long-term persistence. Mercury can sorb to soil organic matter, clay minerals, concrete, brick, wood, paint, and other porous construction materials [14–16,24]. The extent of sorption is influenced by moisture content, organic carbon content, chloride concentrations, redox conditions, and surface geochemistry. Desorption from these materials may subsequently act as a long-term secondary source of mercury vapor, potentially sustaining indoor air concentrations even after vapor-entry pathways have been partially mitigated. Consequently, mercury behavior at contaminated industrial facilities is controlled by a suite of coupled processes, including volatilization, diffusion, adsorption, desorption, and phase partitioning, rather than by volatilization alone. These interacting processes can substantially influence the magnitude, spatial distribution, and persistence of mercury vapor intrusion and associated human exposure risks over time.

2.2. Toxicity and Exposure Routes

The toxicological behavior of mercury depends strongly on its chemical species, including inorganic forms (Hg⁺ and Hg²⁺) and organic forms such as methylmercury (MeHg) [13]. Inhalation of mercury vapor represents the predominant pathway for exposure to elemental mercury, frequently occurring because of the rupture of containers or devices [25,26]. Less frequently, it may be introduced into the body through ingestion or through contact with the skin.

The efficiency of absorption is contingent upon the specific chemical form of mercury. The absorption of elemental mercury via the respiratory tract is notably high, with rates observed between 69 and 85% in adult humans [11]. However, when elemental mercury is in its amalgam form, absorption during ingestion is approximately 0.04%. Inhaled elemental mercury that does not get absorbed is eliminated through the process of exhalation. Ingested elemental mercury can be excreted via the kidneys in urine; however, that occurrence is limited. In cases of dermal exposure, full body immersion in mercury vapor results in approximately 2% absorption through inhalation [11].

Once absorbed, elemental mercury enters systemic circulation and is disseminated throughout the body [11]. The highest concentrations are typically found in the kidneys.

The inhalation of elemental mercury limits the oxidation rate, and its distribution into the bloodstream creates a first-pass metabolism effect on the uptake of mercury into the brain and heart, attributable to the vascular proximity of these tissues. Furthermore, elemental mercury is capable of traversing the blood–brain barrier following its oxidation [17]. Inhaled elemental mercury may also be passed to the fetus from an exposed mother or to an infant during breastfeeding.

Metabolism of elemental mercury includes a series of intricate arrays of molecular interactions that form the basis of its biological activity and toxicity including the formation of conjugates with low molecular weight thiol-containing ligands such as cysteine and glutathione [11,27]. These conjugates are actively transported across the basolateral membrane via organic anion transporters, facilitating cellular uptake and systemic distribution [11,27]. The toxicological profile of elemental mercury includes various diverse molecular interactions that disrupt cellular homeostasis and contribute to adverse health effects. These include oxidative stress, mitochondrial dysfunction, and interference with enzymatic activity [11,27].

Elemental mercury is classified as a potent neurotoxicant. Its ability to cross the blood–brain barrier and accumulate in neural tissue initiates specific neurotoxic pathways, leading to cognitive and motor impairments [11]. Acute or prolonged exposure may result in a range of effects, including headaches, tremors, insomnia, alterations in nerve response to stimuli, neuromuscular symptoms such as weakness, muscle atrophy, or twitching, as well as mood fluctuations, irritability, nervousness, excessive shyness, and sensory disturbances. Individuals who have been exposed may exhibit suboptimal performance on assessments of cognitive function. Exposure to elevated levels of elemental mercury vapors can lead to significant adverse health effects, including renal impairment, respiratory failure, and mortality [11,17].

3. Data and Methodology

3.1. Data Sources

This case study relies on a retrospective analysis of environmental data collected during historical and ongoing site investigations. The primary data were extracted from environmental reports authored by several environmental consulting firms, including ChemRisk, AECOM, and Geosyntec Consultants. Because the fieldwork was performed by external contractors under regulatory oversight, this section summarizes the methodologies they employed to generate the dataset analyzed in this study. The raw data is also included in Sections S2–S5 of the Supplementary Information.

3.2. Field Sampling and Analytical Procedures

During the installation of soil vapor and sub-slab probes between 2020 and 2022, contractors screened soil using a Thermo Fisher Scientific Niton XL2 Plus portable X-ray fluorescence (XRF) spectrometer (Thermo Fisher Scientific; Tewksbury, MA, USA) and an AMETEK Brookfield, formerly Arizona Instrument, Jerome 505 (J505) mercury vapor analyzer (AMETEK Brookfield; Middleboro, MA, USA). XRF was utilized to quantify mercury concentrations within the soil matrix, while the J505 measured mercury within the pore spaces. Subsequent active indoor- and outdoor-air sampling was conducted in May and June 2021. Air samples were collected using a pump that drew approximately 36 L of air through laboratory-supplied sorbent tubes over a period of approximately 3 to 4 h. The sampling apparatus was positioned at a height of 1.0 to 1.5 m above the floor, corresponding to the typical human breathing zone.

Routine indoor-air monitoring has been conducted weekly since February 2022 using the J505 analyzer. During each monitoring event, two consecutive 30 s measurements

are collected and averaged to determine the mercury vapor concentration for each unit. Samples are typically collected during afternoon hours to represent potentially elevated exposure conditions, as indoor mercury vapor concentrations may increase relative to morning periods. The J505 analyzer was also used in August 2022 to screen 52 identified preferential pathways (vapor entry points) for potential vapor intrusion. In addition, 30 wipe samples were collected in November 2021 to quantify mercury concentrations in surface dust. Each wipe sample was collected from a standardized surface area of 929 cm², primarily on concrete flooring surfaces.

A notable limitation of the indoor-air dataset should be acknowledged. Active indoor-air sampling was conducted during May and June 2021, and routine weekly monitoring was generally performed during afternoon periods selected to represent potential worst-case exposure conditions. Monitoring was not conducted during pre-dawn, nighttime, or early-morning periods, when lower ambient temperatures may reduce mercury volatilization and indoor-air concentrations. Consequently, the available dataset does not characterize diurnal variability in indoor mercury concentrations. The monitoring strategy is appropriate for identifying periods of potentially elevated occupational exposure; however, it does not provide a time-weighted characterization of average daily exposure.

3.3. Derivation of Wipe Screening Levels

Due to the absence of established regulatory screening levels for wipe sampling, a site-specific screening level for a commercial/industrial adult exposure scenario was derived in consultation with the California Office of Environmental Health Hazard Assessment (OE-HHA). Because the major pathway (inhalation of mercury vapor) was already accounted for and monitored, this screening level exclusively accounts for dermal contact and incidental ingestion. The gastrointestinal absorption factor for inorganic mercury was applied, as elemental mercury exhibits low absorption rates when ingested.

The adult mercury wipe screening level (SL_{wipe}) was calculated using the following equation:

$$SL_{wipe} = \frac{RfD \times THQ \times BW \times AT \times UCF \times WR}{CF \times TE \times EF \times ED \times ((SAd \times ABSd / ABSgi) + (SAi \times Fdo))}$$

The exposure parameters and chemical-specific values utilized in this derivation are detailed in Table 2.

Table 2. Exposure parameters and chemical-specific values used to derive the commercial mercury wipe screening levels for an adult.

Symbol	Parameters (Units)	Adult Value [Source]
RfD	Reference dose for mercury (mg/kg-day)	0.00016 [28,29]
THQ	Target hazard quotient (unitless)	1 [28,30,31]
BW	Body weight (kg)	80 [28,32]
AT	Averaging time (days)	9125 [30,32]
UCF	Unit conversion factor (1000 µg/mg)	1000 [28]
WR	Wipe removal efficiency (unitless)	0.5 [28,30]
CF	Contact frequency (per day)	8 [28,30,33,34]
TE	Surface to skin transfer efficiency (unitless)	0.1 [28,30,33,35]
EF	Exposure frequency (days/year)	250 [28,30,32]
ED	Exposure duration (year)	25 [28,30,32]
SAd	Skin surface area, dermal (cm ²)	6032 [28,32]
ABSd	Dermal absorption for mercury (unitless)	0.01 [28]
SAi	Skin surface area ingestion (cm ²) which is the mean surface area of hands for males and females	980 [28,30,36]

Table 2. Cont.

Symbol	Parameters (Units)	Adult Value [Source]
Fdo	Skin-to-mouth transfer efficiency (unitless)	0.04 [28,30]
ABSgi	Fraction gastrointestinal absorption (unitless) for inorganic mercury (mercuric chloride and other mercury salts)	0.07 [22,28]

3.4. Data Processing and Statistical Analysis

All statistical analyses were conducted to quantitatively validate spatial and temporal differences in mercury concentrations. Data distributions were first assessed for normality using the Shapiro–Wilk test and for homogeneity of variance using Levene’s test. Across all datasets, mercury concentrations severely violated the assumption of normality (all Shapiro–Wilk p -values < 0.001); therefore, robust non-parametric hypothesis testing was employed.

For variables with two independent groups (e.g., seasonal comparisons of wet versus dry periods), median concentrations were compared using the Mann–Whitney U test. For variables consisting of three or more independent groups (e.g., soil depths, building occupancy zones, and groundwater well locations), the Kruskal–Wallis H test was utilized to determine overall significance. When a Kruskal–Wallis test indicated a statistically significant omnibus result ($p < 0.05$), Dunn’s post hoc test with a Bonferroni p -value adjustment was applied to identify exactly which pairwise group comparisons were significantly different while controlling for the family-wise error rate. All statistical tests were evaluated at a significance level of $\alpha = 0.05$.

Data compilation, summary statistics, and visualizations were generated using R version 4.4.2 (31 October 2024), specifically utilizing the ggplot2 package. Boxplots overlaid with violin plots were created to visualize central tendencies (medians and interquartile ranges) alongside the full density distribution of the data. Due to the variability of concentration ranges, log-transformations ($\log_{10}$) were applied to specific datasets. To facilitate this transformation, all zero observations were replaced with a small constant (0.001) prior to transformation to allow inclusion of all data points while minimizing distortion of the distributions. For samples with non-detect results (censored data), the specific laboratory reporting limit was utilized for statistical purposes.

4. Conceptual Site Model

The primary source of mercury in soil and groundwater at the site can be attributed to the historical mercury cell area, resulting from spills associated with mercury handling or leaks from the permeable cell architecture. Additional sources of mercury include the concrete cells that were fragmented and potentially utilized as fill material following the extraction of mercury from 1919 to 1945 [4].

The presence of chloride in soil and groundwater is anticipated to result from the discharge of brine from the production of sodium hypochlorite, which is the active component in traditional bleach formulations. Chloride concentrations can impact the adsorption and mobility of mercury in the environment, where theoretically, an increase in chloride concentrations correlates with a decrease in the soil–water partition coefficient (K_d) of mercury, thereby increasing its mobility [1,37–39].

4.1. Contaminated Media

Elemental mercury was most likely released into the environment as a result of spills, leaks, and the crushing of concrete retorts [5]. Once elemental mercury is released into the soil, its total concentration is distributed across three distinct phases: solid, liquid, and vapor. The solid phase corresponds to the fraction of mercury adsorbed onto the soil matrix

and/or the surfaces of the crushed material. The liquid phase represents the concentration of mercury present in the aqueous phase within water-filled soil pore spaces. Finally, the vapor phase comprises the portion of elemental mercury that has volatilized into the air-filled pore spaces between soil particles [40,41]. The distribution of mercury across these phases significantly influences environmental behavior, including degradation pathways, transport mechanisms, and potential bioaccumulation in terrestrial and aquatic biota.

4.1.1. Soil

Various biogeochemical processes in soil may influence the flux of elemental mercury, including factors such as soil composition, moisture content, pH, redox potential, temperature, and porosity. The type and composition of soil can significantly influence adsorption capacity. Clay-rich soils exhibit a significant surface area and an abundance of negatively charged sites, which can facilitate adsorption processes, resulting in diminished plant uptake and a decrease in groundwater leaching. In contrast, sandy soils exhibit a reduced number of adsorption sites, resulting in diminished retention capacity. The moisture content of soil influences adsorption processes, within moist or saturated soil conditions, water has the capacity to displace adsorbed mercury, leading to diminished adsorption and enhanced mobility. High moisture content in soil (i.e., lack of air between soil particles) may limit the transition to the vapor phase. Furthermore, the content of organic carbon in soil plays a crucial role in the process of adsorption. The fraction of organic carbon (f_{oc}) denotes the ratio of organic carbon present within the soil matrix. A higher f_{oc} value typically correlates with increased adsorption, which in turn reduces the mobility and bioavailability while increasing the total concentration of total mercury [40,42]. The on-site soils are predominantly composed of silts and clays, with organic carbon constituting less than one percent of the total soil mass [5].

Soil pH can influence the adsorption of various chemicals, especially when the sorption mechanism is dominated by ion exchange and/or complexation processes. Soils with a lower pH (acidic conditions) are characterized by an increased presence of positive surface charges. This phenomenon facilitates the adsorption of negatively charged (anionic) substances. Conversely, soils with a higher pH (alkaline conditions) are characterized by surfaces that possess a greater negative charge. Such alkaline soil conditions lead to increased adsorption of positively charged chemical species [40–42]. Additionally, variations in temperature within a soil column can significantly influence the adsorption process, particularly for volatile chemicals like mercury. As temperature increases, mercury's vapor pressure increases, resulting in enhanced partitioning within the vapor phase. This process results in increasing transport through the vadose zone where it can accumulate at higher concentrations under building foundations or other structures placed above the source. Conversely, in lower temperatures, volatilization is generally reduced, and adsorption is favored, resulting in increased soil concentrations [40,42].

The estimated mass of mercury present, based on soil concentration data, at less than one meter of soil beneath the building foundation is approximately 907 kg [6] of combined elemental and ionic (Hg^{2+}) forms [1,4]. Given the duration of its presence, it is anticipated that the soil-associated mercury has reached equilibrium and is stable [6]. The mass of mercury decreases with increasing depth [5,7], with approximately 227 kg of mercury present between approximately one to two meters below the building slab, and an estimated amount of less than 23 kg at depths exceeding two meters [1,5,6]. Table 3 summarizes mercury concentrations in soil by depth.

Table 3. Summary of total mercury concentrations in Soil.

Depth	Concentration Range (mg/kg)	Estimated Kilograms in Soil
<1 m	0.114–1800	907
1–2 m	0.113–2090	227
>2 m	0.16–480	23

Boxplots with overlaid violin plots were generated in R using the ggplot2 package to visualize both summary statistics and the underlying data distributions for each environmental media. Boxplots were used to display medians and interquartile range, while violin plots illustrated the full distribution and density of the data. Due to variability of concentration ranges, some plots were created using logarithmic scales. In these instances, values were log-transformed as indicated. Because logarithmic transformation is undefined for zero values, all zero observations were replaced with a small constant (0.001) prior to transformation to allow inclusion of all data points while minimizing distortion of the distributions. For any censored data (non-detect data) the laboratory reporting limit was used.

Mercury concentrations in soils differed markedly between samples collected adjacent to the manufacturing building and those collected near the building addition, with contrasts evident across all depth intervals in both central tendency and variability (Figure 1). Summary statistics and violin–boxplots highlight consistently elevated concentrations and greater heterogeneity near the manufacturing building (Table 4). At the manufacturing building, shallow soils (<1 m; $n = 23$) exhibited the highest mercury concentrations, with a mean of 1.68, median of 1.92, and standard deviation of 0.999 ($\log_{10}$ mg/kg). These values indicate both elevated central tendency and substantial variability in the near-surface interval. At intermediate depths (1–2 m; $n = 26$), concentrations decreased, with a mean of 0.860 and median of 0.753, although variability remained relatively high ($SD = 0.877$). In deeper soils (>2 m; $n = 28$), concentrations were comparable to the 1 to 2 m interval (mean = 0.984; median = 0.893; $SD = 0.933$), suggesting limited attenuation with depth and persistence of mercury in the subsurface. The similarity in central tendency and spread between intermediate and deep intervals, combined with the broad distributions observed in the violin plots, indicates that elevated mercury concentrations extend throughout the profile and are spatially heterogeneous. However, further statistical tests indicate that mercury concentrations varied significantly across soil depths ($H = 9.71$, $p = 0.008$). A Dunn’s post hoc test with Bonferroni correction revealed that the shallowest soils (<1 m) had significantly different concentrations compared to both 1–2 m ($p = 0.009$) and >2 m ($p = 0.045$). There was no significant difference between the 1–2 m and >2 m depths ($p = 1.000$).

Table 4. Summary statistics of total mercury concentrations in soil beneath or adjacent to the foundation of the former manufacturing building.

Depth	Number of Samples (n)	Mean ($\log_{10}$ mg/kg)	Median ($\log_{10}$ mg/kg)	Standard Deviation ($\log_{10}$ mg/kg)
<1 m	23	1.68	1.92	0.999
1–2 m	26	0.860	0.753	0.877
>2 m	28	0.984	0.893	0.933

In contrast, soils collected near the building addition exhibited lower central tendencies and, in some intervals, reduced variability, and no statistically significant difference in mercury concentrations across the three soil depths ($H = 3.68$, $p = 0.159$) (Figure 1b). Shallow soils (<1 m; $n = 11$) had a mean of 0.571 and median of 0.367, both substantially lower than corresponding values at the manufacturing building, although variability was relatively

high ($SD = 1.20$), reflecting the presence of occasional elevated values (Table 5). At 1 to 2 m depth ($n = 10$), the mean (0.544) and median (0.285) were similarly low, with an even greater dispersion ($SD = 1.27$), consistent with a distribution characterized by predominantly low concentrations with a limited number of higher observations. The deepest interval (>2 m; $n = 8$) exhibited the lowest concentrations overall, with a mean of -0.352 and median of -0.549 , and a comparatively small standard deviation (0.498), indicating both low mercury presence and reduced variability at depth.

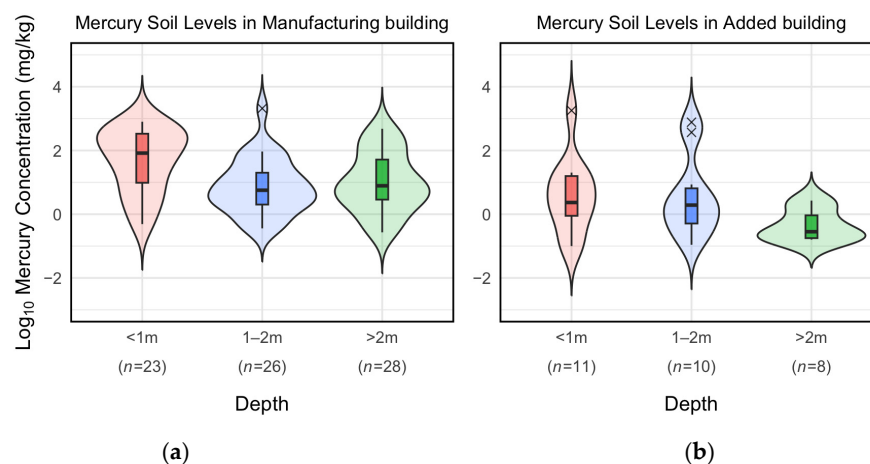


Figure 1. Depth-wise distribution of total mercury concentrations in soil samples collected from the former manufacturing building area and building addition. The distributions are shown for soil samples collected at different depth intervals from (a) locations beneath or adjacent to the foundation of the former manufacturing building; and (b) locations beneath or adjacent to the building addition. Total mercury concentrations are presented as box-and-violin plots, where boxplots show the median, interquartile range, and data spread, and violin plots depict the kernel density distribution of the measurements within each depth interval. All total mercury concentrations were $\log_{10}$ -transformed prior to visualization. Sample sizes (n) corresponding to each depth interval are indicated on the x -axis. The visualization indicates that total mercury concentrations were generally highest in shallow soils (<1 m) and decreased with depth at both locations. Mercury levels and variability were greater beneath the former manufacturing building than beneath the building addition, suggesting a stronger and more heterogeneous contamination source in the former manufacturing area.

Table 5. Summary statistics of total mercury concentrations in soil beneath or adjacent to the building addition.

Depth	Number of Samples (n)	Mean ($\log_{10}$ mg/kg)	Median ($\log_{10}$ mg/kg)	Standard Deviation ($\log_{10}$ mg/kg)
<1 m	11	0.571	0.367	1.20
$1-2$ m	10	0.544	0.285	1.27
>2 m	8	-0.352	-0.549	0.498

Comparatively, median concentrations at the manufacturing building exceeded those at the building addition by approximately one order of magnitude in shallow soils and remained substantially higher at all depths. Furthermore, the manufacturing building exhibited consistently large standard deviations across all intervals ($SD \approx 0.88-1.00$), whereas the building addition showed greater variability in shallow and intermediate soils ($SD > 1.20$) but markedly reduced variability at depth ($SD = 0.498$). These patterns indicate that mercury contamination adjacent to the manufacturing building is both elevated and vertically persistent, whereas contamination near the building addition is generally lower, more sporadic in the upper intervals, and strongly attenuated with depth.

4.1.2. Groundwater

Shallow groundwater is present at the site at a depth ranging from approximately 2.4 to 5.5 m bgs [1]. During the rainy season, groundwater levels may rise by approximately 0.6 to 1.5 m [1,6]. The estimated mass of mercury present in shallow groundwater is approximately 0.5 kg [6]. Based on available site investigations, the predominant dissolved species is believed to be divalent mercury (Hg^{2+}) [1,4,6]. Methylmercury has been detected in groundwater samples collected from the groundwater extraction system that operated between 1988 and 1995 [1,9]; however, available data indicate that methylmercury is not the predominant mercury species in groundwater. Table 6 summarizes the ranges of total mercury and chloride concentrations measured in groundwater between March 1993 and December 2024.

Table 6. Range of total mercury and chloride concentrations in groundwater (March 1993–December 2024).

Monitoring Well Location	Total Mercury Concentration Range ($\mu\text{g/L}$)	Chloride Concentration Range (mg/L)
Source	36–9900	4500–9000
Downgradient (offsite)	<0.50–310	55–2600
Perimeter (offsite near highway subdrain)	<0.17–95	150–670

An important limitation of the groundwater dataset is that routine monitoring primarily measured total mercury concentrations. Spatially resolved measurements distinguishing elemental mercury (Hg^0), divalent mercury (Hg^{2+}), and methylmercury (MeHg) were not consistently available throughout the monitoring period. Consequently, species-specific transport mechanisms, transformation processes, and fate pathways cannot be quantitatively evaluated using the available dataset.

Boxplot analyses demonstrated clear and consistent spatial differences among source, downgradient, and perimeter monitoring wells for both total mercury and chloride concentrations (Figure 2). For total mercury, source wells ($n = 61$) exhibited the highest concentrations, with a mean of 3.74 and median of 3.79 $\log_{10} \mu\text{g/L}$, accompanied by a relatively low standard deviation (0.195 $\log_{10} \mu\text{g/L}$). These results indicate consistently elevated mercury concentrations with limited spatial and temporal variability at the source area (Table 7). Downgradient wells ($n = 86$) exhibited substantially lower concentrations, with a mean of 1.97 and median of 2.04 $\log_{10} \mu\text{g/L}$, representing an approximately two-order-of-magnitude reduction relative to source-area conditions. The larger standard deviation (0.366 $\log_{10} \mu\text{g/L}$) suggests increased variability during downgradient transport, consistent with dilution, dispersion, and heterogeneous subsurface conditions. Perimeter wells ($n = 38$) exhibited the lowest mercury concentrations, with a mean of -0.879 and median of $-1.00 \log_{10} \mu\text{g/L}$, as well as the greatest relative variability (standard deviation = 0.676 $\log_{10} \mu\text{g/L}$). These values correspond to concentrations near or below analytical detection limits and indicate minimal mercury impacts at the site boundary. Mercury concentrations differed significantly based on well location ($H = 155.77$, $p < 0.001$). Post hoc analysis confirmed that Source, Downgradient, and Perimeter wells are all statistically distinct from one another (all pairwise $p < 0.001$).

Chloride concentrations exhibited a similar spatial pattern, although the magnitude and variability differed from those observed for mercury. Source wells ($n = 49$) contained the highest chloride concentrations, with a mean of 7033 mg/L and median of 7300 mg/L . The standard deviation of 906 mg/L indicates moderate variability within the source area (Table 8). Downgradient wells ($n = 69$) exhibited substantially lower chloride concentra-

tions, with a mean of 990 mg/L and median of 1000 mg/L, representing an approximately sevenfold decrease relative to source conditions. The relatively large standard deviation (418 mg/L) is consistent with groundwater mixing and dilution processes during transport. Perimeter wells ($n = 31$) exhibited the lowest chloride concentrations, with a mean of 498 mg/L and median of 500 mg/L, and they displayed comparatively limited variability (standard deviation = 134 mg/L), suggesting relatively stable concentrations near background conditions. Similar to mercury, chloride concentrations differed significantly based on groundwater well location relative to the source area ($H = 112.56$, $p < 0.001$). Post hoc analysis using Dunn's test with a Bonferroni correction confirmed a clear spatial gradient matching the mercury distribution. Source, Downgradient, and Perimeter wells all tested statistically distinct from one another (all pairwise $p < 0.001$).

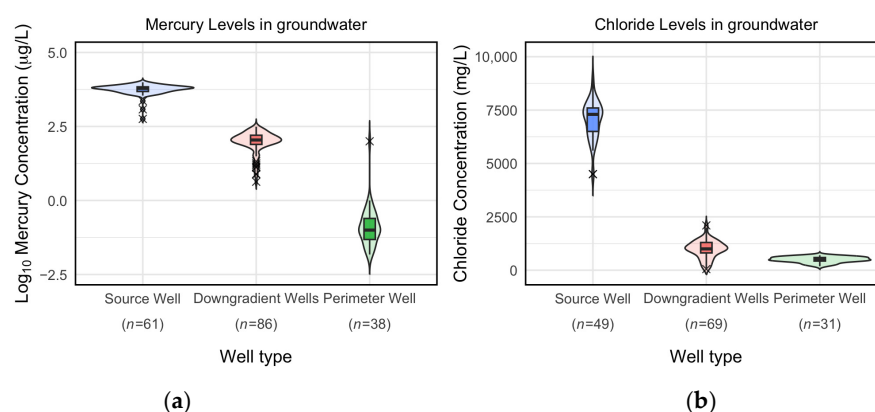


Figure 2. Distribution of total mercury and chloride concentrations in groundwater from source, downgradient, and perimeter monitoring wells. (a) $\log_{10}$ -transformed total mercury concentrations measured in source wells located within the contamination source area (source well), downgradient monitoring wells located along the inferred groundwater flow path, and perimeter monitoring wells situated cross-gradient to the source area. (b) Untransformed chloride concentrations measured in groundwater from the corresponding source, downgradient, and perimeter monitoring wells. Groundwater chemistry data are presented as box-and-violin plots, where boxplots show the median, interquartile range, and data spread, and violin plots depict the kernel density distribution of the measurements at each well type. Sample sizes (n) corresponding to each depth interval are indicated on the x-axis. The visualization shows a clear spatial decline in both total mercury and chloride concentrations from source wells to downgradient and perimeter wells. The highest concentrations occurred in source wells, while perimeter wells exhibited the lowest levels, indicating attenuation of contaminant impacts with increasing distance from the source area and supporting the interpretation that the former source area is the primary contributor to groundwater contamination.

Table 7. Summary statistics of total mercury concentrations in groundwater (March 1993–December 2024).

Monitoring Well Location	Number of Samples (n)	Mean ($\log_{10} \mu\text{g/L}$)	Median ($\log_{10} \mu\text{g/L}$)	Standard Deviation ($\log_{10} \mu\text{g/L}$)
Source	61	3.74	3.79	0.195
Downgradient	86	1.97	2.04	0.366
Perimeter	38	−0.879	−1	0.676

Both mercury and chloride exhibited concentration gradients characterized by elevated source-area concentrations, intermediate downgradient concentrations, and the lowest concentrations at perimeter locations. Total mercury exhibited a substantially steeper gradient, decreasing by several orders of magnitude between source and perimeter wells, as reflected by the $\log_{10}$ -transformed dataset. In contrast, chloride concentrations decreased by approx-

imately one order of magnitude across the same spatial extent. Additionally, variability in total mercury concentrations increased with distance from the source, whereas chloride variability decreased at perimeter locations. These contrasting patterns suggest that mercury and chloride are influenced by different transport, attenuation, and geochemical processes within the groundwater system.

Table 8. Summary statistics of chloride concentrations in groundwater (March 1993–December 2024).

Monitoring Well Location	Number of Samples (<i>n</i>)	Mean (mg/L)	Median (mg/L)	Standard Deviation (mg/L)
Source	49	7033	7300	906
Downgradient	69	990	1000	418
Perimeter	31	498	500	134

4.1.3. Soil Vapor

In 1989, vapor samples were collected from soil borings, which were expected to represent the worst-case scenario, and these samples were utilized to estimate indoor air quality. Mercury concentrations ranged from 1 to 3 $\mu\text{g}/\text{m}^3$ in 16 of 31 samples. The Johnson and Ettinger Model (JEM) was used to estimate the potential concentration of mercury in indoor air. The results of the JEM indicated that the estimated concentration would be below the screening level of 1.3 $\mu\text{g}/\text{m}^3$ (commercial/industrial screening level at the time) and is unlikely to produce a hazard quotient exceeding 1 [4].

From 2020 to 2022, a total of 12 soil vapor probes were installed near the perimeter of the building, 3 soil vapor probes were installed inside the building, and 10 sub-slab probes were also installed inside the building. During the installation process, soil was screened using XRF and a J505 mercury vapor analyzer. Mercury was detected in the soil utilizing XRF, with concentrations ranging from 0 to 1032 mg/kg, and within the pore space using the J505, where concentrations ranged from 0 to greater than 500 $\mu\text{g}/\text{m}^3$. It is important to highlight that the highest J505 measurement ($>500 \mu\text{g}/\text{m}^3$) corresponded to the maximum XRF reading of 1032 mg/kg, which were collected inside the building near the former processing area [7,9].

In February 2021, soil vapor and sub-slab samples were systematically collected for analysis. This sampling event represented the wet season. The concentrations of mercury in soil vapor ranged from 0.0232 $\mu\text{g}/\text{m}^3$ to 0.731 $\mu\text{g}/\text{m}^3$ and 0.0954 $\mu\text{g}/\text{m}^3$ to 101 $\mu\text{g}/\text{m}^3$ in sub-slab samples. In June 2021, the sampling event was repeated to account for seasonal variability with the June event occurring during the dry season. During this sampling event, only four of the eleven soil vapor probes were successfully sampled due to high back pressure. For the samples that were collected, soil vapor results ranged from not detected ($<0.0623 \mu\text{g}/\text{m}^3$) to 1.94 $\mu\text{g}/\text{m}^3$. Sub-slab samples were also collected in June 2021 with results ranging from 0.0448 $\mu\text{g}/\text{m}^3$ to 69.3 $\mu\text{g}/\text{m}^3$ [7,9].

In July 2022 (dry season), five new soil vapor probes were installed, and were subsequently sampled. The results from this sampling event indicated that soil vapor concentrations ranged from 0.03 $\mu\text{g}/\text{m}^3$ to 0.14 $\mu\text{g}/\text{m}^3$. In August 2022, a new soil vapor probe was installed within the interior of the building near the processing area, where the maximum XRF and J505 readings were detected in soil. The concentration of mercury in soil vapor at this location was 360 $\mu\text{g}/\text{m}^3$. Additionally in August 2022, four sub-slab probes were resampled and results ranged from 0.82 $\mu\text{g}/\text{m}^3$ to 4800 $\mu\text{g}/\text{m}^3$ [7]. Table 9 summarizes mercury concentrations in soil vapor and sub-slab by seasons.

Table 9. Summary of mercury in soil vapor and sub-slab.

Monitoring Point Type	Mercury Concentration Range ($\mu\text{g}/\text{m}^3$)	Monitoring Period Season and Years
Soil Vapor	<0.165–0.731	Wet Season (2021–2022)
Soil Vapor	0.03–360	Dry Season (2021–2022)
Sub-Slab	0.0954–101	Wet Season (2021)
Sub-Slab	0.82–4800	Dry Season (2021–2022)

In soil vapor, dry-season concentrations ($n = 10$) were characterized by a higher mean (-0.628) relative to the wet season (-0.929 ; $n = 9$), indicating moderately elevated concentrations under dry conditions (Table 10). However, the median for the dry season (-1.15) was lower than that of the wet season (-0.788), and the standard deviation was substantially greater in the dry season ($\text{SD} = 1.25$ vs. 0.543) (Figure 3a). This divergence between mean and median, combined with the high variance, indicates a strongly skewed distribution in dry-season soil vapor, with the presence of elevated outliers driving the mean upward. In contrast, wet-season soil vapor concentrations were more tightly clustered, with reduced variability and closer agreement between the mean and median, indicating a more stable distribution under wetter conditions. These observations may suggest that dry conditions promote episodic increases in soil vapor mercury, whereas wet conditions may result in more uniform concentrations. However, despite any visually apparent differences in the box plots, Mann–Whitney U tests confirmed there is no statistically significant difference in mercury concentrations between wet and dry seasons for soil vapor ($U = 45.0$, $p = 1.000$).

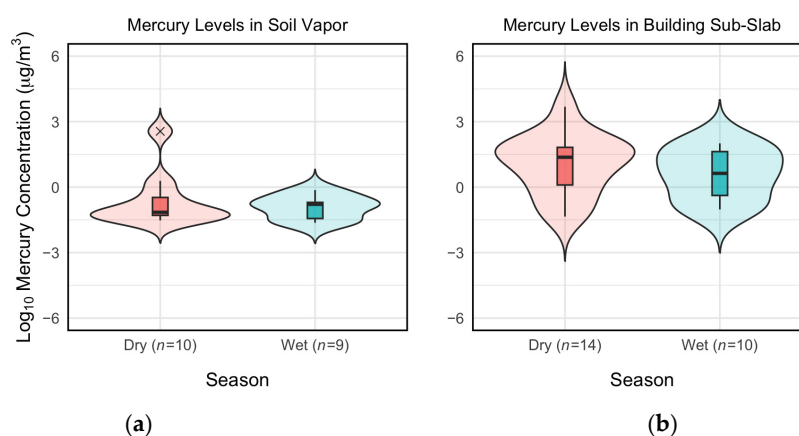


Figure 3. Seasonal distribution of mercury vapor concentrations in soil vapor and sub-slab vapor samples. (a) Mercury vapor concentrations measured in soil vapor samples collected during the dry and wet seasons. (b) Mercury vapor concentrations measured in sub-slab vapor samples collected beneath the building during the dry and wet seasons. Concentrations are presented as box-and-violin plots, where boxplots show the median, interquartile range, and data spread, and violin plots depict the kernel density distribution of the measurements. All concentrations were $\log_{10}$ -transformed prior to visualization. Sample sizes (n) for each sampling season are indicated on the x-axis. The visualization indicates limited seasonal variation in mercury vapor concentrations, with comparable distributions between dry and wet seasons in both soil vapor and sub-slab vapor samples. Mercury vapor concentrations were consistently higher in sub-slab vapor than in soil vapor, indicating greater vapor accumulation beneath the building relative to surrounding soils.

In sub-slab soil gas samples, seasonal differences were more consistent between measures of central tendency (Figure 3b, Table 11). Dry-season sub-slab concentrations ($n = 14$) were higher overall, with a mean of 1.05 and a median of 1.37 , compared to wet-season values ($n = 10$), which exhibited a mean of 0.584 and a median of 0.631 . Variability

remained elevated in both seasons (SD = 1.38 for dry; 1.17 for wet), although slightly greater dispersion was observed during the dry period. The close alignment of mean and median values in both seasons indicates less extreme skewness relative to soil vapor data, suggesting more consistent conditions beneath the building slab. However, Mann–Whitney U tests confirmed there is no statistically significant difference in mercury concentrations between wet and dry seasons for sub-slab measurements ($U = 82.0, p = 0.501$).

Table 10. Summary statistics of mercury in soil vapor.

Season	Number of Samples (<i>n</i>)	Mean ($\log_{10} \mu\text{g}/\text{m}^3$)	Median ($\log_{10} \mu\text{g}/\text{m}^3$)	Standard Deviation ($\log_{10} \mu\text{g}/\text{m}^3$)
Dry	10	−0.628	−1.15	1.25
Wet	9	−0.929	−0.788	0.543

Table 11. Summary statistics of mercury in sub-slab soil gas.

Season	Number of Samples (<i>n</i>)	Mean ($\log_{10} \mu\text{g}/\text{m}^3$)	Median ($\log_{10} \mu\text{g}/\text{m}^3$)	Standard Deviation ($\log_{10} \mu\text{g}/\text{m}^3$)
Dry	14	1.05	1.37	1.38
Wet	10	0.584	0.631	1.17

4.1.4. Indoor Air

Twelve indoor air samples were collected in May 2021 using a pump to draw approximately 36 L of air through laboratory-provided sorbent tubes. This process was conducted over a duration of three to four hours at an elevation of approximately 1 to 1.5 m above the ground, aligning with the typical breathing zone of an individual. The measured concentrations of mercury in indoor air varied between $0.140 \mu\text{g}/\text{m}^3$ and $21.6 \mu\text{g}/\text{m}^3$. The sampling event was conducted again in June 2021, using the same locations and methodology, yielding results that varied from not detected ($<0.0677 \mu\text{g}/\text{m}^3$) to $10.2 \mu\text{g}/\text{m}^3$ [9]. Table 12 summarizes active indoor air mercury concentrations.

Table 12. Summary of mercury concentrations in active air samples.

Media Sampled (Date)	Mercury Concentration Range ($\mu\text{g}/\text{m}^3$)
Indoor Air (May 2021)	0.140–21.6
Outdoor Air (May 2021)	0.0946–0.116
Indoor Air (June 2021)	0.0476–10.2
Outdoor Air (June 2021)	<0.0225

Since the building is currently occupied, systematic indoor air assessments have been performed at the location since February 2022, utilizing the J505 analyzer. On a weekly basis, the J505 is used in the various units to measure mercury vapor concentrations. During these events, typically two consecutive 30 s readings are obtained and subsequently averaged to provide a representation of the mercury vapor exposure concentration within the individual unit (Table 13). Based on the weekly readings, informed decisions are made regarding the continued occupancy status of the individual unit. Considering the chemical characteristics of mercury, samples are typically gathered in the afternoon to account for a worst-case scenario, as mercury levels in indoor air are observed to rise in the afternoon relative to the morning [7–10]. Based on the concentrations of mercury measured in indoor air, various interim mitigation measures (e.g., increased ventilation, filtration) have been deployed for occupant safety.

Table 13. Summary of weekly mercury concentrations in indoor air (J505) December 2020–July 2025 [7,10,43–45].

Unit Status	Mercury Concentration Range ($\mu\text{g}/\text{m}^3$)
Pre-Interim Mitigation Measures	
Full-Time Occupancy	0.00–3.50
Storage Only (entry up to 1 h per day)	0.00–9.13
No entry/Use	0.03–13.94
Vacant/Unoccupied	0.01–0.18
Post-Interim Mitigation Measures	
Full-Time Occupancy	0.00–1.21
Storage Only (entry up to 1 h per day)	0.00–2.47
No entry/Use	0.00–14.47
Vacant/Unoccupied	0.00–0.31

Indoor air mercury concentrations varied by unit use type both pre- and post-interim mitigation measure deployment. Prior to interim mitigation, indoor air mercury concentrations were broadly similar across full-time and storage only units but were elevated in no entry/use units (Figure 4 and Table 14). Full-time units ($n = 506$) had a mean of -1.09 and median of -1.10 , while storage only units ($n = 340$) showed nearly identical values (mean = -1.12 ; median = -1.10), indicating comparable central tendencies and distributions. In contrast, no entry/use units ($n = 338$) exhibited higher concentrations, with a mean of -0.715 and median of -0.824 , reflecting an approximate threefold increase in central tendency relative to full-time and storage only units on a linear scale. Variability was similar across these categories ($\text{SD} \approx 0.58\text{--}0.60$). Vacant or unoccupied units ($n = 11$) showed the lowest concentrations overall (mean = -1.28 ; median = -1.15 ; $\text{SD} = 0.459$), although the small sample size limits interpretability. There was a significant difference in indoor mercury concentrations across occupancy zones ($H = 99.25$, $p < 0.001$). Post hoc testing showed that the “No Entry/Use” zone was significantly different from all other zones (Full-Time, Storage Only, and Vacant/Unoccupied; all $p < 0.05$), while the other three zones did not significantly differ from one another.

Table 14. Summary statistics of mercury in indoor air before interim mitigation measures.

Unit Status	Number of Samples (n)	Mean ($\log_{10} \mu\text{g}/\text{m}^3$)	Median ($\log_{10} \mu\text{g}/\text{m}^3$)	Standard Deviation ($\log_{10} \mu\text{g}/\text{m}^3$)
Full-time	506	-1.09	-1.10	0.586
Storage only	340	-1.12	-1.10	0.583
No entry/use	338	-0.715	-0.824	0.597
Vacant	11	-1.28	-1.15	0.459

Following interim mitigation measures, divergent responses were observed among unit types (Figure 4b, Table 15). Full-time units ($n = 2758$) exhibited a reduction in mercury concentrations, with the mean decreasing to -1.26 and the median to -1.22 , accompanied by reduced variability ($\text{SD} = 0.422$). Similarly, vacant/unoccupied units ($n = 115$) showed a decrease in central tendency (mean = -1.45 ; median = -1.30 ; $\text{SD} = 0.475$), indicating effective attenuation in these categories. In contrast, storage-only units ($n = 1238$) displayed a modest increase in mean (-0.876) relative to pre-interim mitigation measures, although the median (-0.921) remained similar, suggesting persistence of moderately elevated values and continued variability ($\text{SD} = 0.504$). The most pronounced change was observed in no-entry-use units ($n = 1046$), where both the mean (-0.173) and median (-0.131) increased substantially relative to pre-interim mitigation conditions. Mitigation efforts fundamentally altered the distribution of mercury. Post-mitigation, differences across all building zones

were highly significant ($H = 2136.70$, $p < 0.001$). Dunn's post hoc testing confirmed that every single zone is now statistically distinct from every other zone (all pairwise $p < 0.001$).

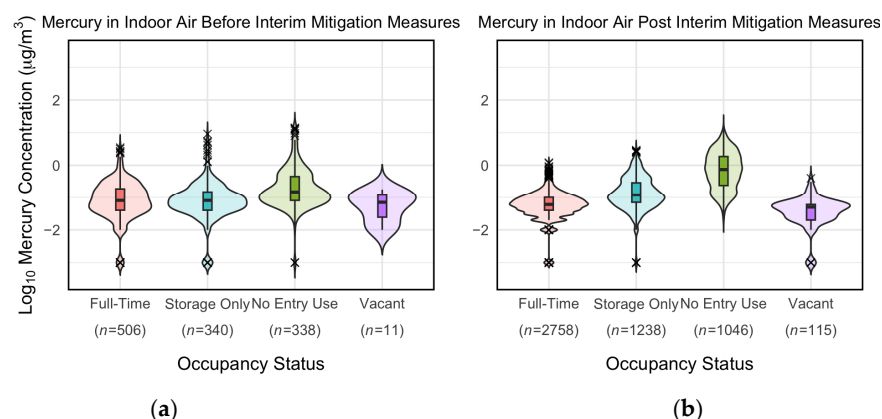


Figure 4. Distribution of indoor air mercury vapor concentration by unit occupancy status before and after implementation of interim mitigation measures. **(a)** Indoor air mercury vapor concentrations measured before installation of interim mitigation measures, grouped by unit occupancy status (full-time occupancy, storage-only use, no-entry use, and vacant). **(b)** Indoor air mercury vapor concentrations measured after implementation of interim mitigation measures, grouped by the same occupancy categories. Indoor air mercury vapor concentrations are presented as box-and-violin plots, where boxplots show the median, interquartile range, and data spread, and violin plots depict the kernel density distribution of the measurements within each unit occupancy category. Sample sizes (n) for each occupancy category are indicated on the x-axis. The visualization indicates indoor air mercury vapor concentrations were broadly similar across occupancy categories both before and after interim mitigation measures. Following implementation of interim mitigation measures, concentrations generally remained low, with only modest differences among occupancy groups, suggesting that occupancy status was not a primary determinant of indoor mercury vapor levels.

Table 15. Summary statistics of mercury in indoor air post interim mitigation measures.

Unit Status	Number of Samples (n)	Mean ($\log_{10} \mu\text{g}/\text{m}^3$)	Median ($\log_{10} \mu\text{g}/\text{m}^3$)	Standard Deviation ($\log_{10} \mu\text{g}/\text{m}^3$)
Full-time	2758	−1.26	−1.22	0.422
Storage only	1238	−0.876	−0.921	0.504
No entry/use	1046	−0.173	−0.131	0.565
Vacant	115	−1.45	−1.30	0.475

4.2. Exposure Pathways and Assessment

4.2.1. Preferential Pathway Analysis

To evaluate the source of elemental mercury in indoor air, a preferential pathway evaluation was conducted to ascertain whether adjacent underground utilities existed at a depth that could serve as a potential entry point for mercury vapor. The investigation in 2016 determined that public utilities were not a preferential pathway, as all communication utilities (internet and phone) were installed approximately one meter outside the building footprint. The waterline was located within the street and right-of-ways, while the petroleum line ran parallel to the railroad tracks (off-property) and increased in depth as it approached the street right-of-way [5].

In light of the absence of identified vapor entry points from major utilities in 2016, an investigation was conducted to assess potential vapor entry points associated with the reconfiguration of the building and its utilization for bleach manufacturing. The preferential pathway analysis for the building was completed in August 2022 and resulted in the discovery of 52 distinct preferential pathways, or vapor entry points. The preferential

pathways were screened with the J505 mercury vapor analyzer to quantify the concentration of mercury vapor. The measured concentrations of mercury vapor from these identified preferential pathways ranged from $0.03 \mu\text{g}/\text{m}^3$ in the storm drain system to $325.30 \mu\text{g}/\text{m}^3$ at a covered opening in the floor. The identified preferential pathways were sealed to the extent feasible (notably, certain pathways like utility conduits could not be sealed) to mitigate the amount of mercury vapor into indoor air [6–10].

4.2.2. Wipe Sampling

An investigation of building materials was conducted to evaluate whether residual mercury contamination was present on the concrete floor of the building. Due to the building being occupied, handheld XRF samples were collected from the concrete floor in November 2021 and February 2022. No other building materials (including walls) were sampled at this time. The findings from the XRF analysis of the concrete floor ranged from not detected to 201 mg/kg. The results varied within individual units, with one unit ranging from 17 mg/kg to 201 mg/kg and another unit ranging from not detected to 128 mg/kg [8]. In light of these findings, the concrete floors across the building underwent decontamination, followed by the application of an epoxy floor coating to further mitigate the risk of mercury vapor intrusion from the subsurface into the indoor air. In November 2021, a total of 30 wipe samples were collected to assess the concentrations of mercury present in dust. Wipe samples were systematically collected from a 929 square centimeter surface area of the concrete floor, with the exception of a single wipe sample obtained from an engineered wood floor. Laboratory analysis revealed that mercury concentrations in the dust collected using wipe samples ranged from 0.001 micrograms per square centimeter ($\mu\text{g}/\text{cm}^2$) to $1.08 \mu\text{g}/\text{cm}^2$. The wipe sample that exhibited the highest concentration of mercury, was found to correlate with the unit that had the maximum concentration of mercury detected with the XRF [8].

As detailed in Section 3.3, due to the absence of established screening levels for wipe sampling, a site-specific adult mercury wipe screening level of $0.013 \mu\text{g}/\text{cm}^2$ was derived in consultation with OEHHA [26]. This paper focuses solely on the adult scenario, as land use restrictions at the site prevent the establishment of daycare centers or schools for individuals under the age of 21 [7,43]. Out of the thirty wipe samples collected, 11 exceeded the adult mercury wipe sample screening level of $0.013 \mu\text{g}/\text{cm}^2$ with concentrations ranging from $0.014 \mu\text{g}/\text{cm}^2$ to $1.08 \mu\text{g}/\text{cm}^2$ [8].

4.2.3. Exposure Assessment

The site is currently a large warehouse that is subdivided into various units that are leased to a diverse array of small business entities. Various units are used for as little as 1 h per day (storage only) to full-time occupancy (various office spaces or art studios). While the former bleach manufacturer is still responsible for the cleanup of the site, the landlord and tenants must manage the inconvenience of weekly indoor air monitoring and moving to different units if the indoor air concentrations exceed the site-specific action levels. The site-specific action levels are presented in Table 16.

Table 16. Summary of health-based indoor air site-specific action levels for mercury.

Purpose	Concentration ($\mu\text{g}/\text{m}^3$)
Full-time occupancy of unit	≤ 0.13
Storage only (entry up to 1 h per day)	$>0.13\text{--}\leq 0.6$
No entry/Use	>0.6

Current confirmed receptors include onsite indoor commercial/industrial workers which consists of the tenants of the units and the building manager. These onsite receptors are primarily exposed to elemental mercury through the soil vapor intrusion pathway (soil vapor to indoor air) and potential secondary source of accumulation of mercury from historical operations into building materials. The typical direct contact pathway from soil is not expected, as the contaminated soil is currently covered by asphalt, concrete, and the building foundation, thereby limiting direct contact with the soil for the commercial/industrial worker. Furthermore, land use controls are present at the site that prohibit the disturbance of soil. Therefore, the primary exposure pathway is soil vapor to indoor air.

Other receptors have been identified at the site, which include outdoor workers and ground intrusive workers. The outdoor workers may be exposed to inhalation of mercury in outdoor air due to off-gassing of the subsurface; however, due to the mixing of outdoor air, this exposure pathway would be minimal as long as the contaminated soil remains covered by the existing concrete or asphalt. Ground intrusive personnel involved in utilities and construction activities, whether adjacent to or onsite, may encounter contaminated soil and groundwater directly. Additionally, the excavation/construction of a utility or construction trench may lead to mercury vapor accumulation in the trench space and direct contact with potentially impacted soil and groundwater.

Individuals engaging in recreational activities might encounter mercury that has been discharged from the site, as a result of the accumulation of affected stormwater, surface water, and groundwater through the stormwater drainage systems situated around the site's boundaries. Collected water in the storm drains (stormwater, surface water, and groundwater) is discharged directly to the estuary which may result in exposure to mercury through direct contact of surface water or the consumption of potentially contaminated fish and shellfish.

The State of California generally mandates the assessment of the hypothetical residential receptor, as the state cannot control off-site property use. This pathway is commonly employed to substantiate the necessity for land use regulation, aimed at restricting particular uses of the site, such as prohibiting the establishment of school or daycare facilities for individuals under 21 years of age, residential housing, hospitals, or medical facilities, among others. In the context of the hypothetical resident, the pathways of soil, groundwater, soil vapor, and indoor air are taken into account.

Given that the delineation of soil, groundwater, and soil vapor plume remains incomplete, the exposure pathways for offsite receptors (commercial/industrial workers, outdoor workers, ground intrusive workers, and the hypothetical residential receptor) are considered to be potentially complete. The site and its immediate vicinity do not utilize groundwater as a source of drinking water.

5. Community and Cultural Demographics

The State of California is characterized by substantial racial and ethnic diversity, with no single racial or ethnic group comprising a majority of the population. Latinos represent the largest demographic group, accounting for approximately 40% of California residents and more than 50% of Californians younger than 24 years of age [46]. The site is located in an underserved community that receives an environmental health screening tool (CalEnviroScreen version 5.0) score of >90–100, indicating one of the highest levels of cumulative environmental and socioeconomic burden in the state. CalEnviroScreen evaluates multiple indicators, including exposure-related factors (lead from housing, traffic-related pollution, toxic releases, particulate matter [PM_{2.5}], diesel particulate matter, ozone, and drinking-water contaminants), environmental effects (solid waste facilities, groundwater threats, hazardous waste facilities, and contaminated site cleanups), sensitive-population indicators

(cardiovascular disease, asthma, and low birth weight), and socioeconomic factors (housing burden, poverty, linguistic isolation, educational attainment, and unemployment) [47]. This community exhibits some of the lowest life expectancy and educational attainment metrics in California, as well as elevated rates of cardiovascular disease and asthma-related emergency department visits [48]. The racial and ethnic composition of the community is approximately 62.79% Latino, 13.65% White, 13.57% Asian American Pacific Islander, 7.00% Black, and 2.90% Other [47].

Residents within the study area may experience cumulative exposures arising from multiple environmental stressors identified through CalEnviroScreen, including air pollution, hazardous waste sites, and socioeconomic vulnerabilities. Within the context of the present investigation, the primary concern is potential occupational exposure to elemental mercury among building tenants and workers, together with possible secondary community exposure resulting from the transport of mercury-contaminated dust or mercury-impacted materials from the site.

Previous studies have reported elevated mercury body burdens among some Mexican-American children relative to non-Hispanic children [49,50]. Both inorganic mercury, which may be associated with certain folk or religious practices, and methylmercury, which is commonly linked to the consumption of contaminated fish, have been documented within these populations [49,50]. Mercury-containing skin-lightening products have also been identified as a source of exposure in some Hispanic and immigrant communities [17,51]. Several investigations have reported mercury concentrations in such products exceeding 1000 ppm, substantially greater than the U.S. Food and Drug Administration (FDA) limit of less than 1 ppm [50–52]. Because imported personal-care products may be readily available through formal and informal markets, mercury-containing products can remain a potential exposure source despite regulatory restrictions. In 1996, a multistate mercury poisoning outbreak affecting Latina communities in Arizona, California, New Mexico, and Texas was linked to an imported skin cream containing mercury concentrations of up to 10% by weight. The incident resulted in adverse health effects among users and secondary exposure among household members due to the release of mercury vapor within residential environments [50]. Intentional use of mercury in certain cultural or religious practices may also contribute to exposure. Within traditions such as Espiritismo, Santería, Palo, and Voodoo, elemental mercury, commonly referred to as “azogue”, may be incorporated into perfumes, candles, oil lamps, amulets, or dispersed within living spaces for ceremonial or spiritual purposes. Mercury used in these applications can volatilize and contribute to indoor air concentrations, potentially resulting in prolonged exposure among household occupants. Previous studies have suggested that residual mercury contamination associated with such practices may persist within indoor environments and remain a source of exposure after the original application has ceased [50].

Although mercury exposure may occur through a variety of environmental, occupational, cultural, and consumer-product pathways, these pathways were not evaluated as part of the present investigation. Rather, the relevance of community demographics to this study is to provide context for evaluating how a persistent occupational mercury source within an environmentally burdened community may contribute to cumulative environmental health risks and existing health disparities.

6. Discussion

Due to the physicochemical properties of elemental mercury, mercury vapor was readily generated within soil pore spaces at the site. Vapor pressure is a key determinant of a chemical’s tendency to volatilize from the solid or liquid phase into the gas phase. Chemicals with higher vapor pressures exhibit greater volatilization potential, resulting

in increased concentrations in the vapor phase [42]. Henry's Law constant (K_H), which describes equilibrium partitioning between aqueous and gaseous phases, further influences this behavior. Because K_H is directly related to vapor pressure, compounds with elevated vapor pressures generally exhibit higher K_H values and a correspondingly greater tendency to volatilize. Elemental mercury is a well-known example of this behavior owing to its relatively high vapor pressure and Henry's Law constant compared with many metals [40,41].

Because mercury is known to adsorb to building materials, including paint, roofing materials, brick, plaster, wooden supports, and structural wood components [24,53], the potential role of building materials as secondary mercury reservoirs was evaluated. Quantitative source apportionment was not possible because direct measurements of mercury flux from individual source compartments were unavailable. Consequently, relative source contributions can be assessed only qualitatively. Multiple lines of evidence support subsurface contamination as the primary historical source of mercury entering the building. These include elevated mercury concentrations in underlying soils, elevated soil-gas and sub-slab mercury concentrations, identification of 52 preferential vapor-entry pathways, and concentration gradients associated with former process areas. However, several observations suggest that subsurface vapor intrusion may not be the sole contributor to indoor-air mercury concentrations. Indoor mercury concentrations remained elevated in portions of the building following preferential-pathway sealing, floor decontamination, and the implementation of ventilation controls. In addition, elevated floor XRF measurements and wipe-sample exceedances suggest the potential accumulation of mercury within building materials and surface reservoirs. Definitive confirmation of this hypothesis would require direct analysis of construction materials, including concrete cores, wallboard, wood structural components, ceiling materials, and other porous surfaces that may retain mercury.

Although building occupants are informed of the presence of mercury contamination and therefore knowingly accept potential occupational exposure, there remains a potential for involuntary exposure within the surrounding community potential pathways for inadvertent exposure outside the facility may also exist. Such exposure could occur through transport of mercury-contaminated dust on workers' clothing [24,53], transfer of contaminated materials from storage units, redistribution of donated items stored on-site, or movement of reupholstered furniture and other materials originating from the property. Given the physicochemical properties of mercury, adsorption and subsequent desorption of mercury vapor may occur in porous materials such as furniture, fabrics, cardboard, and other storage materials, as well as within mechanical equipment and appliances [54]. For example, restaurant equipment or other reusable commercial assets stored within contaminated areas may potentially retain mercury residues. Although routine screening of items before they leave the site is not commonly performed, mercury-vapor testing is relatively rapid and non-destructive. Small items may be placed in a sealed container or vapor-tight bag and allowed to equilibrate under elevated temperatures to enhance possible mercury volatilization. The container headspace can then be screened using a J505 mercury vapor analyzer to evaluate the potential presence of mercury vapor. For appliances and electronic equipment, testing of exhaust air or ventilation pathways may help identify residual mercury dust or vapor contamination [54].

Continued indoor-air monitoring indicates that mercury vapor concentrations remain elevated in portions of the building despite floor decontamination and sealing of preferential vapor-entry pathways. These observations suggest that vapor intrusion alone may not fully explain current indoor-air concentrations. The presence of secondary indoor mercury reservoirs associated with contaminated building materials represents a plausible

explanatory mechanism; however, the currently available evidence remains circumstantial and does not conclusively demonstrate this process. Future investigations should include direct characterization of building materials, including concrete, wallboard, wood, and other porous construction materials, to quantify potential mercury inventories within the structure and evaluate their contribution to indoor-air concentrations.

Since the early 2000s, a growing body of research has documented that underserved communities, particularly low-income populations and communities of color, disproportionately experience environmental pollution burdens and associated adverse health outcomes [48,55,56]. Many older neighborhoods were historically developed adjacent to industrial facilities, including ports, rail corridors, refineries, and manufacturing districts. As buildings age and structural degradation occurs, including foundation cracking, potential pathways for vapor intrusion from subsurface contamination may develop. In addition, deterioration of building envelopes can increase the infiltration of contaminated outdoor air [57]. Housing shortages in metropolitan regions may further complicate these issues. Affordable or subsidized housing developments are often constructed on comparatively inexpensive land, which may be located adjacent to active industrial areas or on properties affected by historical contamination. As a result, historical land-use practices, zoning decisions, and industrial development patterns may contribute to disproportionate environmental exposures among already vulnerable communities.

In response to documented health impacts associated with air pollution, California Assembly Bill (AB) 617 was enacted in 2017 to establish a community air-monitoring and emission-reduction framework [48,55]. As of March 2025, community air-monitoring plans had been developed or were under development in nineteen California communities identified as experiencing elevated environmental burdens. These communities are characterized by high rates of cardiovascular and respiratory disease, elevated unemployment, lower educational attainment, and substantial exposure to emissions from sources such as ports, former military facilities, agriculture, and petroleum-refining operations [48,55].

The conventional management approach for hazardous waste and environmental release sites often evaluates each site independently, with limited consideration of cumulative environmental burdens within the surrounding community. Although such an approach may be adequate in areas with relatively few environmental stressors, it may inadequately capture cumulative risks in communities already affected by multiple pollution sources, socioeconomic challenges, and limited public-health resources. More broadly, changes in air-quality regulations affecting mercury emissions, such as the recent announcement regarding the reversal of the Mercury and Air Toxic Standards (MATS) by the USEPA for power plants in June 2025 [58], have the potential to influence cumulative mercury exposure burdens at the regional scale; however, evaluation of specific regulatory impacts is beyond the scope of the present study.

7. Conclusions and Recommendations

This investigation demonstrates that mercury vapor intrusion at repurposed industrial properties may involve more complex mechanisms than those typically represented in conventional vapor-intrusion conceptual site models. At the study site, historical chlor-alkali operations resulted in persistent mercury contamination in soil, groundwater, soil gas, and indoor air. Elevated sub-slab and indoor-air mercury concentrations indicate the presence of an active vapor-intrusion pathway. However, the persistence of elevated indoor-air concentrations following the implementation of interim mitigation measures suggests that secondary mercury reservoirs within building materials may also contribute to occupant exposure. More broadly, this case study suggests that redevelopment of former industrial facilities into multi-tenant commercial properties can create exposure

scenarios that may not be fully addressed by traditional vapor-intrusion investigations. Occupancy patterns, interior subdivision, variable ventilation conditions, and long-term mercury adsorption to porous building materials may all influence exposure potential and the effectiveness of risk-management measures.

The authors assert that additional outreach and educational efforts may be warranted to improve awareness of mercury exposure pathways among site occupants, workers, and potentially affected household members. This case study highlights several unconventional pathways through which contamination associated with a former industrial facility may affect both occupants and the surrounding community. Potential pathways include transport of mercury-contaminated dust, movement of contaminated materials off-site, and possible redistribution of mercury retained within porous materials.

Conventional environmental risk-management frameworks generally evaluate contaminated sites individually and may not fully account for cumulative environmental burdens within surrounding communities. Although site-specific evaluations remain essential, consideration of broader socioeconomic, occupational, and community factors may provide additional context for risk-management decision-making. The incorporation of cumulative risk-assessment approaches that consider occupational exposures alongside residential, environmental, and community-level stressors may provide a more comprehensive understanding of potential health risks. Such approaches could help identify populations experiencing disproportionate environmental burdens and support more equitable risk-management strategies. For regulators and other decision-makers, cumulative risk assessments may provide a useful framework for evaluating environmental health impacts beyond the boundaries of an individual site. By considering multiple sources of exposure and community vulnerabilities, such assessments may better support strategies aimed at reducing overall environmental health burdens in underserved communities while improving the effectiveness of long-term risk-management decisions.

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Abbreviations

The following abbreviations are used in this manuscript:

CalEPA	California Environmental Protection Agency
DTSC	Department of Toxic Substances Control
USEPA	United States Environmental Protection Agency

OEHHA	Office of Environmental Health Hazard Assessment
REL	Reference exposure level
Hg	Mercury
AMSL	Above mean sea level
bgs	Below ground surface
HgS	Mercury sulfide
MeHg	Methylmercury
H ⁰	Elemental mercury
Hg ²⁺	Mercury (II) ion
K _d	Soil-water partition coefficient
f _{oc}	Fraction of organic carbon
RfD	Reference dose
THQ	Target hazard quotient
BW	Body weight
AT	Averaging time
UCF	Unit conversion factor
WR	Wipe removal efficiency
CF	Contact frequency
TE	Surface to skin transfer efficiency
EF	Exposure frequency
ED	Exposure duration
SAd	Skin surface area, dermal
ABSd	Dermal absorption
SAi	Skin surface area ingestion
Fdo	Skin-to-mouth transfer efficiency
ABSgi	Fraction gastrointestinal absorption
K _H	Henry's Law constant
µg/m ³	Micrograms per cubic meter
XRF	X-Ray fluorescence
µg/cm ²	Micrograms per square centimeter
PM 2.5	Particulate matter with a diameter of 2.5 micrometers or less
CDC	Centers for Disease Control
FDA	United States Food and Drug Administration
AB	California Assembly Bill
MATS	Mercury and Air Toxic Standards

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