

Mercury in the terrestrial environment: a review

Barbara Gworek

Institut Ochrony Srodowiska

Wojciech Dmochowski

Institut Ochrony Srodowiska

Aneta Baczewska-Dąbrowska (✉ a.baczewska-dabrowska@obpan.pl)

Polska Akademia Nauk Ogród Botaniczny Centrum Zachowania Roznorodności Biologicznej <https://orcid.org/0000-0002-6606-6112>

Review

Keywords: mercury, deposition and emissions from land, soil pollution, content in plants

Posted Date: September 9th, 2020

DOI: <https://doi.org/10.21203/rs.3.rs-17336/v3>

License: © ⓘ This work is licensed under a Creative Commons Attribution 4.0 International License. [Read Full License](#)

Version of Record: A version of this preprint was published on October 2nd, 2020. See the published version at <https://doi.org/10.1186/s12302-020-00401-x>.

Abstract

Environmental contamination by mercury is and will continue to be a serious risk for human health. Pollution of the terrestrial environment is particularly important as it is a place of human life and food production. Publication presents a review of the literature on issues related to Hg pollution of the terrestrial environment: soil and plants and their transformations. Different forms of atmospheric Hg may be deposited on surfaces by way of wet and dry processes. These forms may be sequestered within terrestrial compartments or emitted back into the atmosphere, and the relative importance of these processes is dependent on the form of Hg, the surface chemistry, and the environmental conditions. On the land surface, Hg deposition mainly occurs in the oxidized form (Hg²⁺), and its transformations are associated primarily with the oxidation-reduction potential of the environment and the biological and chemical processes of methylation. The deposition of Hg on ground with low vegetation is 3–5 times lower than that in forests. The estimation of Hg emissions from soil and plants, which occur mainly in the Hg⁰ form, is very difficult. Generally, the largest amounts of Hg are emitted from tropical regions, and the lowest levels are from the polar regions.

Background

Mercury is recognized as a toxic, persistent, and mobile contaminant; it does not degrade in the environment and becomes mobile because of the volatility of the element and several of its compounds. Moreover, mercury has the ability to be transported within air masses over very long distances [1].

Over the last few decades, considerable scientific knowledge has been developed on the sources and emissions of mercury, its pathways and cycling through the environment, human exposure, and impacts to the environment and human health [2]. Hg is the only element on the periodic table to have its own environmental convention, i.e., the Minamata Convention on Mercury, thus highlighting the importance of the Hg pollution issue [3].

An improved understanding of the global mercury (Hg) cycle is important for our capacity to predict how regulatory efforts to reduce current emissions to air, water and land will affect Hg concentrations in environmental compartments, biota and humans. Hg is released into the environment through human activities and via natural sources and processes, such as volcanoes and rock weathering. Following its release, Hg is transported and recycled between the major environmental compartments, i.e., air, soil and water, until it is eventually removed from the system through burial in coastal and deep ocean sediments, lake sediments, and subsurface soils [2,4].

Hg is considered to be a peculiar chemical element because it displays particularly strong chemical and biological activity as well as variability in form (liquid and gaseous). Hg compounds with very different chemical and physical properties are included in various cycles of its natural circulation [5-6]. Hg is a globally distributed pollutant due to characteristics such as low melting and boiling points, conversions between chemical forms and participation in biological cycles. As a result of anthropogenic emissions, the global atmospheric Hg deposition rate is approximately three times higher than that in preindustrial times and has increased by a factor of 2-10 in and around the most industrialized regions [7].

Hg-contaminated **land** environments pose a risk to global **public health**, with Hg being listed as one of the ‘ten leading chemicals of concern’ [8]. In 2013, the **United Nations** (UN) introduced the ‘Minamata Convention on Mercury’ [9], which aims for a more global effort for managing the risk presented by Hg to **human health** and the environment. The Minamata Convention has as of today 123 parties and the convention entered into force 16 August 2017. This concerted action, if successful, will have great implications for public health for decades to come; however, there are many hurdles on the way to achieving this goal [10-11].

Our understanding of the critical processes driving global Hg cycling, particularly those that affect the large-scale exchange of Hg among major environmental compartments, has advanced substantially over the past decade. This progress has been driven by major advances in three interconnected areas: new data, new models, and new analytical tools and techniques [12].

The main task for improving our knowledge of mercury sources, fates, impacts, and emission control options was defined at the 14th ICMGP Conference in 2019 [13] as “Bridging knowledge on global mercury with environmental responsibility, human welfare and policy response”. The main four questions that require urgent answers within this subject are defined as follows:

1. How is mercury biogeochemical cycling changing on global, regional, and local scales in response to perturbations caused by major anthropogenic drivers of environmental change?
2. What is the relative risk of mercury exposure to human health and wildlife in the context of human welfare?
3. How can technological development contribute to the reduction in mercury exposure and improvement of environmental responsibility?
4. How can scientific knowledge contribute to the implementation and effectiveness evaluation of the Minamata Convention and other regulatory agreements, and what is the importance of integrating and implementing emerging and future mercury research into the policy making?

Methods

While developing this paper, the method recommended by **Liberati** et al. [14] was used to some extent. However, the strict use of this method was not possible due to the specifics of our publication. The problems of pollution and Hg transformation in the environment caused by significant threats to human health represent the subject of research by numerous research teams around the world. However, many publications do not meet the criterion of quality of research results. Therefore, the following criteria were used to eliminate publications:

1. Accurate descriptions of the research methods were lacking;

2. Methods that did not guarantee the quality of the results were used;
3. The latest publications were selected, but historical works were not omitted; and
4. Publications from various countries and regions were cited.

Hg emission

The advances achieved over the last decade for the assessment of Hg emissions from major man-made and natural sources have contributed to improvements in the assessments of the impacts of atmospheric deposition of Hg on the terrestrial environment [15]. The assessment of Hg emissions poses serious methodological problems. In estimating these impacts, state institutions mainly focus on inventories of their sources, while international organizations apply different models and use emission factors and statistical data of industrial production and consumption of Hg-containing materials. It is particularly difficult to distinguish natural and anthropogenic emissions from re-emissions from the land and oceans [16-17]. Hg emission sources include both natural processes unfolding in the biosphere and anthropogenic sources. In 2008, the following classification was adopted in a UNEP report [18], which distinguished three emission sources:

- Current emissions from natural sources;
- Current emissions from anthropogenic sources; and
- Re-emissions from historical deposits from natural and anthropogenic sources.

Environmental archives offer an opportunity to reconstruct temporal trends in atmospheric Hg deposition at various timescales. Lake sediment, peat, ice cores, tree rings, and Hg stable isotope measurements are offering new insights into historical Hg cycling. Preindustrial Hg deposition has been studied over decadal to millennial timescales extending as far back as the Late Pleistocene. Exploitation of mercury deposits (mainly cinnabar) first began during the Mid- to Late Holocene in South America, Europe, and Asia but increased dramatically during the Colonial era (1532-1900) for silver production [19-20]. Artisanal gold mining is now thought responsible for over half the global stream flux of Hg, followed by the burning of coal [2,21].

Fluxes of Hg to the air occur via the volatilization of Hg⁰ as well as through wind entrainment of Hg bound to dust particles (often referred to as fugitive dust emissions). The relative magnitude of these two types of Hg emission sources varies depending on site-specific conditions. However, due to the long atmospheric lifetime of Hg⁰, these emissions mostly contribute to the global pool of Hg, whereas the fugitive dust emissions impacts are more local [22].

The update global mercury budget shows the impact of human activities on the mercury cycle and the resulting increase in mercury accumulated in soil and oceans. The updated global mercury budget in 2018 [2] is shown in the Table 1. The annual deposits were 480 tons higher than the emissions, indicating considerable enrichment in the environment. In general [23-24], Hg⁰ emissions from undisturbed nongeologically thermally enriched areas are <1 ng m⁻² h⁻¹, whereas fluxes from contaminated sites can be several orders of magnitude higher at >5000 ng m⁻² h⁻¹. For contaminated sites that cover large spatial areas (such as mining operations), the annual emissions from the entire surface area have been shown to range from 19 to 105 kg yr⁻¹ from active industrial gold mines and 51 kg yr⁻¹ from a large abandoned Hg mining area.

Table 1. The update global mercury budget [2] (Mg/yr).

Emission:	
i) Natural (geogenic)	500;
ii) Re emission/re- mobilization (natural and legacy):	
- biomass burning	600
- from soil and vegetation	1000
iii) Anthropogenic	2500 (2000-3000)
iv) Ocean, net evasion (gaseous elemental Hg)	3400 (2900-4000)
total	7000
Deposition:	
i) to land / freshwater	3600
ii) to ocean	3880
total	7480

Various models are applied to estimate Hg emission levels into the atmosphere, However, the emission levels determined by using these models differ substantially. Travnikov et al. [25] compared the global emission levels from natural and anthropogenic sources determined by using four models [26-29]. The models differed significantly in their estimations of global total emissions, with values ranging from 4,000 Mg yr⁻¹ to

9,230 Mg yr⁻¹, of which natural emissions and re-emission ranged from 45% to 66% of the total emissions. A later summary of the four models [30] estimated that global anthropogenic emissions were 1,870 Mg yr⁻¹, although the global natural emissions and re-emissions were already significantly different from each other at 3,995 Mg yr⁻¹ to 8,600 Mg yr⁻¹ [31]. The proportions between the amounts of natural emissions and anthropogenic emissions are not precisely determined. The ratios vary by authors and encompass a relatively wide range from 0.8-1.8 [32-34]. The relatively broad emission ranges presented in various studies are caused by the following factors: volatility

of Hg compounds, distribution of sources, low levels of Hg concentrations in the air, concentrations that are much lower than levels of other basic pollutants and difficulty of determination [35-36].

Annual mercury emissions in selected regions of the world in 2000-2015 are shown in Table 2. Despite the many applicable global, regional and national programs and conventions aimed at reducing Hg emissions, the global THg emissions in the years 2000-2015 increased by 1.8% [11]. Emission reductions only took place in North America and Europe without the former USSR, and the largest increase was in Central America (5.4%) and South Asia (4.0%).

Table 2. Annual emissions of mercury by selected world region [11] (Mg/yr).

	2000	2015	2000- 2015 growth (%/yr)	2010- 2015 growth (%/yr)
USA	127.7	42.7	-66,6	-10.2
OECD Europe	106.2	38.0	-64,2	-5.8
Canada	12.7	8.3	-34,6	-3.2
Eastern Europe	49.4	33.6	-32,0	-1.3
Central America	33.5	34,3	2,4	5.4
South Asia	120.8	191,6	58,6	4.6
Eastern Africa	19.1	72.9	281,7	4.0
East Asia	532.7	1012,3	90,0	2.6
South America	239.0	275.8	15,4	0.7
Southeast Asia	224,6	187,5	-16,5	2,2
Global Total	1964.1	2389.8	21,7	1.8

Emission values [11] for the aggregated source categories and their composition are shown in Table 3. Three categories had the highest absolute value in 2015: gold ASMG (775.1 Mg/yr), coal combustion (558.3 Mg/yr) and cement production (206.3 Mg/yr). From 2010-2015, the largest increase in emissions was demonstrated by cement production (6.3%), gold production at a large scale (5.2%) and industrial metal extraction (4.6%). The reductions [37] were in the following fields: dental (-5.6%) and electrical equipment (-5.2%). This finding is consistent with the continued expansion of the global economy but is less than the growth in world GDP over the same period (5.7%/yr), suggesting continuous improvement in pollutant emissions per unit of production.

Table 3. Annual emissions of mercury by selected source category [11] (Mg/yr).

Source category	2000	2015	2000- 2015 growth (%/yr)	2010- 2015 growth (%/yr)
Gold ASMG	583,7	775.1	32,8	1.3
Coal combustion	359.9	558.3	55,1	0.7
Cement production	74.3	206.3	177,7	6.3
Waste burning	201.9	165,6	-18,0	2.1
Municipal waste	202.6	140.6	-30,6	2.5
Gold, large-scale	82.7	112.1	35,6	5.2
Zinc smelting	81.7	103.6	26,8	1.4
Copper smelting	53.7	70.0	30,4	3.2
Dental	49.4	16.1	-67,4	-5.6
Electrical equipment	118.7	62.0	-94,8	-5.2
Global Total	1964.1	2389.8	21,7	1.8

Mercury emissions from natural sources

The estimate of Hg emissions from natural sources includes contributions from primary natural sources and the re-emission processes of historically deposited Hg over land and sea surfaces. Hg emitted from volcanoes, geothermal sources and topsoil enriched in Hg pertains to primary natural sources, whereas the re-emission of previously deposited Hg on vegetation, land or water surfaces is primarily related to land use changes, biomass burning, meteorological conditions and gaseous Hg exchange mechanisms at the air-water/topsoil/snow-ice pack interfaces [15, 38].

A characteristic feature of natural Hg emissions compared to anthropogenic sources is their distributed nature and wide range of distribution. Hg from natural sources is introduced on a global scale into the atmosphere, and Hg from anthropogenic sources is deposited mainly locally and regionally; therefore, determining emission levels and applying effective control methods are difficult. Natural Hg emission processes also include re-emission of Hg previously deposited from the atmosphere in the process of wet and dry deposition from both natural and anthropogenic sources, which increases the difficulty of estimating Hg emissions from natural sources [34-35, 39].

Annual global Hg emissions from natural sources on land are estimated by various authors and cover a wide value range, e.g., total emissions of 1,600-2,500 Mg yr⁻¹, including re-emissions of 790-2,000 Mg yr⁻¹. According to most authors, re-emissions were higher than

primary emissions [40-42].

Volcanoes

Volcanoes and geothermal activities are important sources of Hg pollution in terrestrial environments. Hg is emitted from volcanoes primarily as gaseous Hg^0 , and the Hg/SO_2 ratio is generally adopted to estimate Hg emissions. The annual average of Hg released to the atmosphere without episodic strong eruptions for volcanoes and geothermal activities is $\sim 75\text{-}112 \text{ Mg yr}^{-1}$ of Hg, accounting for approximately $< 2\%$ of the contribution from natural sources [15, 43-45]. In volcanic plumes, Hg is present both in the gas phase as elemental Hg^0 and reactive Hg^{II} and in the particle phase as Hg_p forms. The proportions of these species are highly variable. Hg^{II} and Hg_p typically amount to $< 5\%$ of THg, with Hg^0 as the most abundant form [46-48]. Hg levels in volcanic ash nanoparticles ($36 \pm 4 \text{ mg kg}^{-1}$) are dramatically higher than their bulk concentrations (0.08 mg kg^{-1}) [49]. Many areas of geothermal activity have long been associated with elevated levels of Hg in the soil and air in places such as Hawaii, Iceland, western parts of the United States and New Zealand [43, 50].

The average annual global Hg emissions estimate from biomass burning (**emissions from wildfires:** forests, savannas and grasslands) for 1997-2006 was $675 (\pm 240) \text{ Mg yr}^{-1}$, which accounts for 8% of all current anthropogenic and natural emissions. The largest Hg emissions are from tropical and boreal Asia, followed by Africa and South America [51-52]. The important factor for Hg concentrations in forest soils is the time since stand-replacing fires have occurred, and high soil burn severity has the potential to reduce the concentrations of Hg in burned soils for tens to hundreds of years [53-54]. In a specific emission source in Nisyros Island (Greece), Hg concentrations in fumarolic gases in Nisyros Island (Greece) ranged from 10,500 to 46,300 ng/m^3 , while Hg concentrations in the air ranged from high background values in the Lakki Plain caldera ($10\text{-}36 \text{ ng/m}^3$) up to 7100 ng/m^3 in the fumarolic areas [55].

Exchange of mercury between atmospheric and terrestrial ecosystems

Air-, soil- or vegetation-covered exchange fluxes are an important part of global and regional biogeochemical cycles [56]. Much of the Hg^{II} deposited in precipitation or taken up by plants is reduced to Hg^0 and may be released back to the atmosphere. Recent vegetation and soil Hg studies suggested that vegetation Hg^0 uptake dominates (50-80%) Hg net deposition at terrestrial sites [57-59].

The different forms of atmospheric Hg may be deposited on surfaces by way of wet and dry processes. These forms may be sequestered within terrestrial compartments or emitted back to the atmosphere, with the relative importance of these processes being dependent on the form of Hg, surface chemistry, and environmental conditions. Many models assume that the net GEM (gaseous elemental mercury) exchange with soil surfaces is zero; however, as discussed below, some components are assimilated into foliage over the growing season and accumulate in soils [59]. Smith-Downey et al. [60] estimated that evasion of Hg linked to the decomposition of soil organic carbon pools and subsequent liberation of Hg^{II} sorbed to soil organic matter is over 700 t/y, thus reflecting the large pool of Hg stored in terrestrial ecosystems globally (over 240 kgt). In total, this study estimated that 56% of Hg deposited to terrestrial ecosystems is reemitted. Similarly, Graydon et al. [61] found that 45-70% of isotopically labeled Hg^{II} wet-deposited to a forested watershed had been reemitted to the atmosphere after a year.

Litter deposition is the predominant source of Hg in soil. Forest litter horizons show significant increases in mass-dependent fractionation (MDF) during decomposition concurrent with augmented total Hg mass, and this relationship is most significant at high-elevation sites [62-63]. Measurements of mercury in litterfall and throughfall have been increasingly used to provide knowledge of mercury deposition over forest canopies. The majority of mercury in litterfall is considered to be from the stomatal uptake of Hg^0 and can be used as a rough and conservative estimation of atmospheric mercury dry deposition (the portion that is retained in leaves). Mercury in throughfall also includes a portion of previously dry-deposited mercury (the portion that is washed off from the canopy). Concurrent measurements of litterfall, throughfall, and open-space wet deposition measurements can be used to estimate dry deposition on seasonal or longer time scales, whereby dry deposition is approximated as litterfall plus throughfall minus open-space wet deposition [64-65].

On the land surface, Hg deposition is mainly in the oxidized form (Hg^{2+}), and its transformations are associated primarily with the oxidation-reduction potential of the environment and with the biological and chemical processes of methylation. For soils in which oxidizing conditions predominate, the Hg^{2+} and Hg_2^{2+} forms dominate, and in soils with reducing conditions, Hg and sulfur compounds are mainly present. Methyl-Hg compounds are most commonly found in soils with transient conditions [5].

For GEM (Hg^0), the residence time is estimated at 6 to 18 months, while GOM (gaseous mercury in oxidized form) and TPM (total particulate mercury) are quickly removed from the air through wet and dry deposition, and their residence times are estimated to be hours or days at most [66-67]. Given the long time for removal from the air, GEM can be transported over large distances [16]. The particular Hg species are characterized by different dry deposition rates, which also determine their residence times. The dry residence times of the different Hg species form the following series [68]:

$$\text{GEM } 0.19 \text{ cm s}^{-1} < \text{TPM } 2:1 \text{ cm s}^{-1} < \text{GOM } 7:6 \text{ cm s}^{-1}$$

According to Marsik et al. [69], the dry deposition rates of GOM and GEM are much higher during daytime than nighttime. As with Lindberg et al. [68], these authors explain this fact by the closure of plant stomata at night. The deposition rates also depend on the type of surface [70]. Caffrey et al. [71] determined that the deposition rates of particulate air pollutants on the ground with low vegetation were 3-5 times lower than those in forests. Deposition is also affected by the weather conditions, air humidity, insolation and atmospheric precipitation. Research by Converse et al. [72] in an uncontaminated high-elevation wetland meadow in Shenandoah National Park, Virginia (USA) showed the highest Hg deposition occurred in spring ($4.8 \text{ ng m}^{-2} \text{ h}^{-1}$), with a decrease occurring in summer ($2.5 \text{ ng m}^{-2} \text{ h}^{-1}$) to near zero flux in fall ($0.3 \text{ ng m}^{-2} \text{ h}^{-1}$), followed by an increase in winter emissions ($4.1 \text{ ng m}^{-2} \text{ h}^{-1}$). These studies also suggest that stomatal processes are not the dominant mechanism for ecosystem-level GEM exchange. Table 4 shows a summary of biome-level Hg depositions and soil Hg turnover times [60]. The concentration of Hg in soils is therefore a function of the deposition rate and carbon turnover time. High soil concentrations in desert ecosystems are driven by a combination of higher deposition and extremely slow Hg turnover. Tropical and temperate lifetimes are similar despite the faster carbon turnover in tropical systems due to the relative balance between Hg provided by wet deposition and leaf uptake.

Table 4. Summary of Biome Level Hg Deposition and Soil Hg Turnover Times [60].

Biome	Mean Hg deposition (g m^{-2})		Mean soil Hg turnover time* (years)	
	Preindustrial	Present day	Preindustrial	Present day
Tropical forest	0.9	3.7	234	126
Temperate forest	0.8	2.9	250	151
Boreal forest	0.5	1.5	998	560
Grassland	1.0	3.5	522	269
Tundra	0.3	0.8	1108	702
Desert	0.5	1.4	2387	1748

*with respect to respiration.

Emissions from soils have the form of GEM and depend on many factors [73-77]:

- The properties of soils, e.g., Hg content, the contents of organic compounds, and saturation;
- The concentrations of oxidants, mainly ozone, in the air; and
- The weather conditions, e.g., solar radiation, temperature, humidity and winds.

Soil Hg fluxes are significantly lower in dark conditions than light conditions for all sites except grassland [64].

It is most difficult to estimate Hg emissions from plants, and these emissions mainly occur in the form of Hg^0 [79-81]. Ericksen et al. [78] suggested the following hierarchy of environmental parameters that influence Hg flux:

soil moisture > light > air concentration > relative humidity > temperature.

Table 5 shows a summary of total Hg (THg) fluxes from terrestrial regions [82], and Table 6 shows the average fluxes (or, in some cases, the range of fluxes) for various ecosystems measured by a number of investigators.

Table 5. Summary of mercury fluxes from terrestrial regions [82]

Region	Evasion (average) (Mg yr^{-1})	Ratio ¹ (%)
Forest	342	7,5
Tundra/Grassland/Savannah/Prairie/Chaparral	448	9,9
Desert/Metalliferous/Non-vegetated Zones	546	12,0
Agricultural areas	128	2,8
Evasion after mercury Depletion Events ²	200	4,4
Total	1664	-
Volcanoes and geothermal areas	90	-
Biomass burning	675	-

¹calculated over the total evasion from natural sources which sum 4532 Mg yr^{-1}

²Friedli [42] and Mason [68] distinguish as a natural source of mercury emission to the atmospheric air of regions where there are episodes of sudden decreases in mercury concentrations in the air by deposition to the ground and then reemission. These phenomena occur mainly in the arctic regions and Antarctica, and the emission from this source is estimated at 200 Mg yr^{-1}

When analyzing the data in Table 6, the following generalizations can be made [82-83]:

- i) The largest amounts of Hg are emitted from tropical regions (45%), followed by the temperate zones (41%), with the lowest emissions from the polar regions (8%), and emissions from volcanoes and geothermal areas account for 5%;
- ii) Areas with vegetation can be ranked according to the size of their emissions as follows: forests > other areas (tundra, savannas, and chaparral) > agricultural areas > grassland ecosystems; and
- (iii) Land areas devoid of vegetation emit more Hg than do areas with plants.

Deforestation can increase GEM emissions due to higher solar radiation and increased temperature at the soil surface [84-85].

Table 6. Average fluxes, or in some cases the range of fluxes, for various ecosystems measured by a number of investigators.

Species/ecosystem	Flux ¹ nmol m ⁻² mth ⁻¹	References.
Ground Level Forest Floor Sweden	1.4-1.7	Hanson et al. [83] Lindberg et al. [68]
Model Estimates Hard wood forest	Max 4	Bash et al. [87]
Model Estimates Forest Soil	4	Bash et al. [87]
Agricultural crops	Max 11	Bash et al. [87]
Temperate forest	5.0	Rea et al. [88]
Deforested site	50	Magarelli and Fostier [89]
Desert Soils	3.6-9.8	Magarelli and Fostier [89]
High Hg regions	Max 1500	Gustin and Lindberg [90]
Maple	20	Hanson et al. [83]
Oak	16.4	Hanson et al. [83]
Spruce	6.1	Hanson et al. [83]
Prairie grass	12.5	Obrist et al. [53]
<i>Typha</i> sp.	60	Gustin et al. [91]
Average global soil	1.5	Selin et al. [92]

¹Values have all been converted to a common flux unit of nmol m⁻² month⁻¹. Results from the older literature are combined in estimates given in various review papers.

The overall background soil Hg flux in the United States is estimated to be 0.9±0.2 ng/m²/h [79], and in areas with significant Hg pollution, soil emissions are much larger. In the canton of Valais, Switzerland, elemental Hg (Hg⁰) is undetectable in soil, although substantial Hg⁰ emissions were found to occur (20-1,392 ng·m⁻² h⁻¹) [94].

Urban areas are of particular concern with respect to the global Hg cycle due to the following [95]:

- Frequently high terrestrial Hg concentrations and the physically and chemically diverse nature of urban surface covers;
- Highly variable time series concentrations of ambient atmospheric Hg as a result of regional and local emissions; and
- Urban meteorology (i.e., heat island effect).

In the city of Tuscaloosa, Alabama (USA), Hg fluxes on bare undisturbed soil surfaces were as follows (median) [95]:

- Residential site - 4.45 ng·m⁻² h⁻¹;
- Industrial site - 1.40 ng·m⁻² h⁻¹;
- Commercial site - 2.14 ng·m⁻² h⁻¹; and
- Mixed land use site - 0.87 ng·m⁻² h⁻¹.

Areas of land devoid of vegetation emit more Hg than those with plants. The annual averaged fluxes in the subtropical forest zones in China from soil in the forests were

14.2 ng m⁻²h⁻¹, and for open-air sites, they were 20.7 ng m⁻² h⁻¹ [96]. Soil Hg fluxes were significantly lower in dark conditions than in light conditions. In grassland sites, the mean soil Hg flux was 0.6±0.9 ng·m⁻² h⁻¹ in darkness, 1.0±0.7 ng·m⁻² h⁻¹ in light, and 0.9±0.7 ng·m⁻² h⁻¹ overall [64].

Cropland is an important component of terrestrial ecosystems. It is estimated that 33% of natural-source atmospheric Hg comes from the emissions at cropland surfaces [97]. The emission of Hg from cropland soil greatly affects the global Hg cycle. Combinations of different crop cultivars and planting densities will result in different light transmittance under canopies, which directly affects the solar and heat radiation flux received by the soil surface below crops. In turn, this might lead to differences in the soil-air total gaseous mercury (TGM) exchange under different cropping patterns. The light transmittance under the canopy was the key control on soil-air TGM exchange fluxes. High light transmittance can enhance soil TGM emission rates and increase the magnitude of diurnal variations in soil-air TGM exchange fluxes. The estimated annual average soil-air TGM exchange flux was 5.46±21.69 ng m⁻² h⁻¹ in corn-wheat rotation cropland with 30 cm row spacing [98-99].

The bidirectional exchange of Hg between the atmosphere and terrestrial surfaces is better understood because of advancements in research that are primarily associated with the interpretation from Hg isotopes, and the latest estimates place land surface Hg re-emission at values lower than previously thought [100].

Methylmercury

High doses of organic compounds of Hg, particularly methyl-Hg, can be fatal to humans and wildlife, and even relatively low doses can seriously affect the nervous system of organisms. Hg has also been linked to harmful effects on the cardiovascular, immune and

reproductive systems. Methyl-Hg passes through both the placenta and blood-brain barrier; therefore, the exposure of women of child-bearing age and of children to methylmercury is of great concern [1].

Methyl-Hg can be both biotically and abiotically produced in the environment. Methylation of Hg tends to occur in environments with low oxygen levels, low pH, Hg bioavailability, temperature, redox potential and high levels of dissolved organic compounds and in environments favored by sulfate-reducing bacteria, which are largely responsible for methylation. These conditions are found primarily in deep sea environments, coastal marine sediments, and some freshwater lakes as well as soils. These conditions are also characteristic for paddy soil [101-102]. Organic Hg is much more toxic to living organisms than inorganic Hg [4, 103-104]. The content of methyl-Hg in soils and plants is significantly lower than that of THg; however, due to its much higher toxicity, methyl-Hg is particularly dangerous for living organisms. In soils in the coniferous boreal forests of Sweden, the background level of pollution from methyl-Hg accounted for 0.35-0.59% of THg [105]. In the Idrija Hg mining area of Slovenia, a heavily polluted region, methyl-Hg accounted for 0.003% of THg, and its background presence in controls was 0.17% [106]. The content of methyl-Hg relative to THg was 1.9% in the roots of rice under background conditions and 0.55% in the leaves; in areas of Hg mining sites, the concentrations were 0.07% in the roots and 0.01% in the leaves; and in areas with artisanal Hg mining sites, the concentrations were 0.63% in the roots and 0.02% in the leaves [107]. Methyl-Hg in heavily contaminated soil in the Rhône Valley (Switzerland) accounted for < 0.8% of THg [108].

A number of factors that control microbial activity and/or the geochemical speciation of inorganic Hg^{2+} govern MeHg formation in the environment [109]. Microorganisms that live in soil can transform inorganic $\text{Hg}^{\text{(II)}}$ species into Hg^0 by using the enzyme Hg reductase, which is found in various bacteria, such as *Pseudomonas* sp., *Staphylococcus aureus*, *Thiobacillus* and many others [110]. Increases in temperature might lead to increases in biological activity as well as higher Hg^{2+} methylation rates [111].

The direct conversion of insoluble HgS species to MeHg in anaerobic soils is generally believed to be low, although this condition can change when environmental conditions favor HgS complexation [112]. The redox potential also seems to be a key factor, as suboxic and mildly reducing conditions seem to promote high Hg^{2+} methylation rates, while anoxic and strongly reducing conditions might lead to elevated sulfide concentrations that eventually prevent Hg^{2+} from being available for methylation of some methylating bacteria, including SRB (sulphate reducing bacteria, e.g., *Desulfobacter* sp.), and some that control the availability of Hg^{2+} for methylation (e.g., *Deltaproteobacteria* or *Clostridia*) [113-114].

S plays a major role in influencing Hg^{2+} methylation by directly affecting the activity of some methylating bacteria, such as SRB, and controlling the availability of Hg^{2+} for methylation [115-116].

The paddy soils in Hg mining areas have a high methylation ability and may eventually result in heavily biological effects on the local residents through the food chains, such as rice containing high methyl-Hg [117]. In the Wuchuan Hg mining areas (Guizhou, China), soil samples present THg values ranging from 0.33 to 320 mg kg^{-1} and methyl-Hg values ranging from 0.69 to 20 ng g^{-1} . The rice grain samples contain elevated methyl-Hg concentrations ranging from 4.2 to 18 ng g^{-1} , while corn grain contained only 0.5-2.0 ng g^{-1} [118]. Research carried out in areas with coal-fired power plants in Hunan (China) [119] shows that in the soil samples, THg varied from 0.068 to 0.220 mg kg^{-1} (mean value of $0.130 \pm 0.040 \text{ mg kg}^{-1}$), and methyl-Hg ranged from 0.30 to 3.5 $\mu\text{g kg}^{-1}$ (mean $1.6 \pm 1.0 \mu\text{g kg}^{-1}$). In rice samples, the $\text{Hg}^{\text{(II)}}$ concentrations varied from 0.002 to 0.022 mg kg^{-1} (mean 0.057 mg kg^{-1}), and methyl-Hg concentrations varied from 1.7 to 3.8 ng g^{-1} (mean of $2.4 \pm 0.72 \text{ ng g}^{-1}$). Meng et al. [107] showed that rice had high affinity for methyl-Hg and that the concentrations in rice seeds may be 2 to 3 orders of magnitude higher in Hg mining sites than in other local edible crop plants. Freshly deposited Hg is more likely to methylate and be incorporated in rice than stored Hg [100].

Forest fires cause a significant reduction in mercury content in soil. Burned soils in northwestern Ontario (Canada) had 82% less methyl-Hg than fresh soils [120]. Current climate change has had a significant impact on Hg transformation processes, especially in the Arctic. The very large mass of mainly natural Hg found in northern permafrost deposits, which is projected to be released with further climate warming, may profoundly affect biotic Hg levels around the Northern Hemisphere, especially because large amounts of organic carbon, which may stimulate Hg methylation rates, will be simultaneously released [121].

Mercury in soil

Mercury has a relatively long half-life in surface soils because of its recycling between the surface environment and atmosphere. Permanent removal of anthropogenic Hg from the biologically active part of the environment will only occur once it is buried in mineral soils [4]. Soil plays an important role in biogeochemical Hg circulation because it accumulates this element and is a source for other environmental components. Hg occurs naturally in soils from geologic sources [12] or as the result of natural events such as forest fires and volcanic eruptions [49]. The total amount worldwide of Hg accumulated in the soils of terrestrial environments is estimated at 200-300 Gg [122-124]. Smith-Downey et al. [60] suggested that organically bound Hg in preindustrial soils is 200 Gg and that a 20% increase in organically bound soil Hg (to 240 Gg) has occurred from preindustrial steady state conditions to the present day.

In the 2013 Technical Background Report for the Global Mercury Assessment [4], based on a global model and budget developed by Mason et al. [124], human activities were estimated to cumulatively increase atmospheric Hg concentrations by 300-500% over the past century. Because of the naturally high Hg amount present in soil, the average Hg increase was only 20% in surface organic soil and negligible in mineral soils. The revolatilization of "legacy Hg" (i.e., Hg from historical sources of pollution) from soil and ocean and its long residence time in those compartments contribute to maintaining atmospheric Hg concentrations and deposition rates at higher levels than those supported by current primary emissions [125]. Recent estimates of the anthropogenic and natural Hg contents in global soils

(organic layers) (data in kilotons) included 182 natural and 89 anthropogenic sources based on Amos et al. [126] and 130 natural and 20 anthropogenic sources according to the AMAP/UN Environment [4].

All results for Hg soil content, which are presented in the next part of this publication, pertain to the topsoil layer. Generally, the average background concentration of Hg in soil ranges from 0.03 to 0.1 mg kg⁻¹, with an average value of 0.06 mg kg⁻¹, whereas Hg-contaminated sites often have soil concentrations that are 2- to 4-orders of magnitude higher [127-128]. Kabata Pendias and Pendias [5] defined a narrower range of 0.05-0.3 mg kg⁻¹, although some volcanic and organic soils, especially in Canada, may contain higher values, and in the vicinity of industrial emission sources, the values can be extremely high. Obrist et al. [129] showed that a dataset with more than 1,900 randomly selected sampling points across the western USA indicated median Hg concentrations of 0.019 mg kg⁻¹ and an average value of 24 mg kg⁻¹, with only 1% of soil samples exceeding background values (e.g., >0.10 mg kg⁻¹).

The LUCAS Topsoil Survey of the European Union organization collected over 23,000 topsoil samples (upper 20 cm) from land in all European Union countries (28) except for Croatia [130]. The average for European topsoil Hg concentrations was 0.04 mg kg⁻¹, with a range of 0-159 mg kg⁻¹. Studies have identified highly polluted, isolated sites, and the larger historical and recent industrial and Hg mining areas show elevated concentrations of Hg. Historically, mining for gold and Hg led to high Hg concentrations in these mining areas, which may explain the high Hg concentrations in some samples from Central Italy, Northwest England and Eastern Slovakia. Moreover, the natural/background Hg level was 0.08 mg/kg in Brazil [131], 0.05 mg/kg in India [132], 0.23 mg/kg in New Zealand [133], 0.11 mg/kg in the Norwegian Arctic [134] and 0.4 mg/kg in Paris [136]. Most soil Hg was found as soil matrix-bound divalent Hg (Hg^{II}), whereas elemental Hg (Hg⁰) was undetectable in soils [94,108,].

Table 7 presents examples of the concentrations of Hg in the soils in the vicinity of industrial emission sources according to different authors. The highest Hg contents were found in soils near Hg mines: Almaden in Spain, with 2,000 years of mining and ore processing (<8,889 mg kg⁻¹) [142]; Idrija in Slovenia, with 500 years of mining activity (<2,759 mg kg⁻¹) [106, 139]; and in Alaska (5,326 mg kg⁻¹) [136]. Chlor-alkali plants are also an important source of environmental Hg pollution. Bernaus et al. [155] estimated that Hg levels in the soil around a chlor-alkali plant in the Netherlands were as high as 1,150 mg kg⁻¹.

According to Richardson and Moore [170], in the urban environment, the diversity of Hg content in soils is relatively high because of the diversity of land functions in towns. Urban soils were found to accumulate higher concentrations and pools of Hg than their rural montane counterparts across New York and southern New England, which highlights the importance of soils in urban systems for sequestering Hg and preventing its movement towards riparian and aquatic ecosystems, where it can bioaccumulate. Moreover, soil Hg concentrations were poorly correlated with pH, loss-on-ignition, and clay content. Instead, proximity to local industrial and agricultural sources proved a significant influence on Hg accumulation.

The lowest median results were determined for soils in Changchun, China, at 0.018 mg kg⁻¹, with a range of 0.012-0.036 mg kg⁻¹ [171] and in Oslo, Norway (0.06 mg kg⁻¹, with a range of 0.01-2.3 mg kg⁻¹) [172], while the highest concentrations were in Palermo, Italy (median value of 1.85 mg kg⁻¹, with a range of 0.004-2.61 mg kg⁻¹) [173], and Glasgow, Scotland (1.2 mg kg⁻¹, with a range of 0.312-5.2 mg kg⁻¹) [174]. Of note is the reduction of 270% (median from 0.68 mg kg⁻¹ to 0.37) in Hg pollution from 1987-2009 for soils from Beijing, China [175].

In agricultural soils, pollution by Hg was relatively low, as indicated by the low median and average values at usually below 0.1 mg kg⁻¹ (e.g., Scandinavia - Ottesen et al. [176]; Poland - Loska et al. [177]; Iran - Ahmadi et al. [178]). However, the ranges of the results were relatively wide, and the maximum values often exceeded 1 mg kg⁻¹, which may indicate a threat to food production due to the need to protect human health in some areas with higher Hg soil levels. In Europe, pastures were slightly more polluted with Hg than plowed fields [177]. Soils in forest environments contained low levels of Hg. Average and median values did not exceed 1 mg kg⁻¹, although compared with agricultural soils, the maximum values were also lower than 1 mg kg⁻¹. Mineral forest soils contained less Hg than organic ones (USA - Woodruff and Cannon, [54] Czech Republic - Navrátil et al. [179]; Sweden - Åkerblom et al. [180]). According to Obrist et al. [129], soil Hg concentrations significantly differed among land covers following the order:

forested upland > planted/cultivated > herbaceous upland/shrubland > barren soils.

Concentrations in forests were an average of 2.5 times higher than those in barren locations.

The global distributions of soil Hg storage and emissions for both preindustrial and present-day simulations in different biomes are shown in Table 4 [60]. The relatively low soil Hg concentrations in boreal and arctic ecosystems are driven by extremely low deposition. The high soil concentrations in desert ecosystems are driven by a combination of higher deposition and extremely slow Hg turnover. The concentration of Hg in soils is therefore a function of the deposition rate and carbon turnover time.

Physical and chemical properties of the soil affect the Hg cycle in the environment. The soil aggregate size fractions have significant effects on the Hg content in soil. The concentrations of Hg and other heavy metals in soils and sediments generally tend to increase with decreasing grain size, which is due to the propensity of metals to bind with finer particles [181]. Generally, higher values of Hg in soil are found in the fraction at <63 µm [182]. In the Amazonian areas without anthropogenic sources, the fine fraction (< 53 µm) of podzolized soils had higher Hg contents than clayey soils [183]. In a temperate forest podzol, Hg mean values increased as the aggregate sizes decrease, as follows:

clay (170 ng g⁻¹) > fine silt (130 ng g⁻¹) > coarse silt (80 ng g⁻¹) > fine sand (32 ng g⁻¹) > coarse sand (14 ng g⁻¹).

Total Hg enrichment in clay-sized aggregates were 2 to 11 times higher than the values shown by the bulk soil (< 2 mm) [184]. In a heavily polluted area near the Wanshan Hg mine (China), the fine soil aggregate size fractions < 231 µm showed higher total Hg

concentrations and higher soil organic matter content than did the larger aggregate size fractions (231 to 2,000 μm) [185].

Humic acid influences Hg transport and transformation in soil-plant systems, especially for soils having low clay content. Humic acid reduces the amount of available Hg in soil and prevents Hg from being transported into plants or leached from the soil. Leaching can result in Hg leaking into natural water systems under normal environmental conditions. In practice, humic acid can be used to control Hg transportation into food chains from soil heavily polluted by Hg [186].

The chemical and mineralogical properties of soil affect oxidation and retention of atmospheric Hg. Abiotic Hg oxidation occurs because organic matter has -SH groups, which have a high affinity for Hg ions, and Hg oxidation is favored in the presence of compounds with high affinities for the Hg ion [131]. A microbial contribution to Hg oxidation was first proposed by Smith et al. [187], who demonstrated that typical soil bacteria (*Bacillus* and *Streptomyces*) can oxidize elemental Hg to Hg^{2+} through enzymatic paths. Recent studies have shown that Hg can also be oxidized by anaerobic bacteria [188-189]. The soil microbial community is very sensitive to Hg concentrations, and this sensitivity is influenced not only by soil properties but also by the plant species growing in the soil. A level of $0.36\text{-mg}\cdot\text{kg}^{-1}$ of Hg in soils is proposed to be a critical concentration above which plant and soil organisms will be affected [131].

Mercury in plants

Vegetation affects environmental factors at the ground surface by reducing solar radiation, temperature, and wind velocity and serves as a surface for Hg uptake [85]. Many studies have recognized the essential role of terrestrial plants in the biogeochemical cycling of Hg (e.g., Gustin et al. [190]; Fantozzi et al. [191]; Mazur et al. [192]).

Approximately 80% of total Hg accumulated in the aboveground biomass is found in the leaves, and approximately 1% of that Hg is methylated. The concentrations of Hg in aspen tissue grown in high-Hg soil increases in the following order [93]:

stems < branches < petioles < roots < leaves

Research conducted by Leonard et al. [193] in Nevada (USA) in an area with high levels of Hg contamination revealed that for the plant species *Lepidium latifolium*, 70% of the Hg taken up by the roots during the growing season was emitted to the atmosphere.

The main source of Hg in leaves comes from air pollution with Hg^0 and not from soil contamination [190, 194-195]. The studies by Fleck et al. [196] of *Pinus resinosa* have shown that neither woody tissue Hg nor any amount of Hg in the soil or forest floor were closely related to foliar levels, while for some relationships, the opposite was true. The authors interpret these data as indicating that Hg in plant tissues is derived directly from the atmosphere and not from the soil. It is estimated that in highly contaminated soils, generally less than 2% of the Hg present is available for plants [143]. Total leaf concentrations of Hg varied among species and were most closely correlated with the number of stomates per sample, thus supporting the hypothesis that stomatal uptake of atmospheric Hg (most likely Hg^0) is a potential uptake pathway [197]. Research by Arnold et al. [198] also indicated the importance of the nonstomatal pathway for the uptake of total gaseous Hg (TGM).

Plants growing beyond the influence of high Hg emissions contained less than 100 ng g^{-1} THg. Plants growing in the vicinity of factories are large emitters of Hg, such as those around Hg mining sites (e.g., Moreno-Imenez et al. [199]; Qian et al. [200]) and chlor-alkali mining sites [201]. Au mining sites [202-203] may also contain extremely high Hg contents. Mushrooms have been identified as organisms that accumulate more Hg than other plants [204]. A synthesis of published vegetation Hg data from the western United States showed that aboveground biomass concentrations followed the order [129]:

leaves ($26\text{ }\mu\text{g kg}^{-1}$) ~ branches ($26\text{ }\mu\text{g kg}^{-1}$) > bark ($16\text{ }\mu\text{g kg}^{-1}$) > bole wood ($1\text{ }\mu\text{g kg}^{-1}$)

Hg concentrations in leaves were monitored from the emergence to senescence and showed a strong positive correlation with leaf age [195, 197, 205].

Toxic effects on plants

Hg does not have any beneficial effects on organisms and is thus regarded as the "main threat" since it is very harmful to both plants and animals; pollutes the air, water and soil; and is toxic [206]. Mercury has toxic effects on plants, even at low concentrations, and leads to growth retardation [207] and many other adverse effects [208].

Hg in plants is strongly bound to sulfhydryl/thiol groups of proteins and forms SHgS. Hg toxicity in plants occurs via its binding to SH groups of proteins, displacement of essential elements and disruption of the protein structure [209]. This biochemical property probably determines the toxic effects on plants [5, 210-211]. Studies of the toxic effects of Hg on soil organisms and native plants in fields are limited. The effects of Hg are usually examined in sterile and much-simplified laboratory conditions, which may differ from field conditions to varying degrees [212].

The field study of Moreno-Jiménez et al. [199] was conducted in the mining district of Almadén (Spain), which is a cinnabar (HgS) enriched zone, from which one-third of the total Hg produced worldwide is extracted. Mining activity began more than 2,000 years ago, and no other region in the world has been influenced by Hg for such a long period. The region is considered to be one of the regions most polluted by Hg in the world. Hg concentrations in the field plants *Rumex induratus* and *Marrubium vulgare* grown in these soils can be considered phytotoxic, although no symptoms of Hg toxicity have been observed in any of the studied plant species. In most contaminated soils and mine tailings, Hg is not readily available for plant uptake [213].

The absorption of organic and inorganic Hg from soil by plants is low, and there is a barrier to Hg translocation from plant roots to tops. Thus, large increases in soil Hg levels produce only modest increases in plant Hg levels by direct uptake from soil [212]. In

terrestrial vegetation, Hg in the aboveground biomass originates primarily from the atmosphere, whereas Hg in the roots comes from the soil [67,214]. The research conducted by Lomonte et al. [215] suggested the existence of Hg stress-activated defense mechanisms in plants and hypothesized that these mechanisms were likely the reason for the increased production of sulfur compounds in the tested plant species, which stimulated their growth. Hg has very limited solubility in soil, low availability for plant uptake and no known biological function, which may explain why Hg-hyperaccumulating plants have not yet been identified, meaning that a method for Hg phytoremediation in soils contaminated with Hg has not yet been developed [215]. However, studies suggesting the use of transgenic plants for phytoremediation have been published recently [216-217].

The significant toxic effect of Hg on plants is the generation of reactive oxygen species (ROS) [218], e.g., superoxide anion radicals, H_2O_2 , and hydroxyl radicals ($OH\cdot$) [219-220]. Detoxification mechanisms to combat Hg-induced oxidative stress include enzymatic antioxidants and some nonenzymatic antioxidants, such as the following: glutathione [221], phytochelatin [222], salicylic acids [223], ascorbic acid [224], selenium, [225], proline [226] and tocopherols [227]. This process is correlated with the disruption of biomembrane lipids and cellular metabolism, resulting in plant injury [228].

Increasing levels of mercury species in the soil exert a wide range of adverse effects on the growth and metabolism of plants [206, 229-230], such as reduced photosynthesis, transpiration, water uptake, chlorophyll synthesis [228, 231-232] and increased lipid peroxidation (Cho and Park [219]). A high Hg content in plants affects the activity of most enzymes. The total activity of stress indicators such as superoxide dismutase (SOD), peroxidase (POD) and ascorbate peroxidase (APX) increased after Hg treatment, but the vast majority of enzymes were inhibited at higher concentrations (e.g., Manikandan et al. [233]; Mahbub et al. [234]; Zhou et al. [235]).

Conclusions And Commentary

Atmospheric contamination by mercury continues to be one of the most important environmental problems in the modern world. The following general conclusions can be drawn from this review of the literature and are accompanied by the authors' critical commentary:

- Models differ significantly in their estimations of global total Hg emissions—from 4,000 Mg yr⁻¹ to 9,230 Mg yr⁻¹—of which natural emissions and re-emissions ranged from 45% to 66%.
- Many factors contribute to such large differences in the assessments of the level of global emissions: (i) methodological difficulties exist in assessing re-emission from heavily polluted areas under the influence of contemporary and historical emissions and areas with background pollution; (ii) the transformation of various forms of Hg depends on many difficult to evaluate processes, which makes estimating emissions difficult; and (iii) unusual phenomena associated with the transformation of various Hg forms (e.g., mercury depletion events (MDEs), which consist of episodes of sudden drops in total gaseous mercury concentrations in the air in the Antarctic and Arctic, can occur.
- Despite the many applicable global, regional and national programs and conventions aimed at reducing Hg emissions, global total Hg emissions in the years 2000-2015 increased by 1.8%. In many countries, including those with a high national income, there is a lack of understanding by society at large and politicians about the need to reduce emissions.
- The proportions between the amounts of natural emissions and anthropogenic emissions have not been precisely determined. This ratio, which is dependent on the authors, has been estimated over a relatively wide range of 0.8-1.8.
- Annual global Hg emissions from natural sources on land are estimated by various authors over a wide range, with total emissions of 1,600-2,500 Mg yr⁻¹, including re-emissions of 790-2,000 Mg yr⁻¹. The low share of Hg taken up by plants from the soil is beneficial from perspective of protecting food against contamination, although it also limits the possibility of using plants in the phytoremediation of contaminated soils.
- On the land surface, Hg deposition is mainly in the oxidized form (Hg^{2+}), and its transformations are associated primarily with the oxidation-reduction potential of the environment and the biological and chemical processes of methylation.
- The main source of Hg in plant leaves comes from air pollution with Hg^0 and not from soil contamination. It is very difficult to estimate Hg emissions from plants, which mainly occur in the form of Hg^0 .
- Methyl-Hg can be produced both biotically and abiotically in the environment. Methylation of Hg tends to occur in environments with low oxygen levels, low pH, Hg bioavailability, temperature, redox potential and high dissolved organic compound levels, and environments favored by SRB are largely responsible for this methylation.
- Rice growing conditions mean that the Hg methylation process is extremely intensive. Consequently, rice may contain significantly more Hg than other crops, which is particularly dangerous because rice in many regions of the world is the basis for feeding the population.
- Hg has very limited solubility in soil and low availability for plant uptake, and it does not have any known biological function. These factors may explain why Hg-hyperaccumulating plants have not yet been identified, meaning that an effective phytoremediation methods for soil contaminated with Hg has not yet been developed, which may explain why Hg hyperaccumulator plants have not found practical use in phytoremediation of contaminated soils. Such work may be more applicable in wetland environments. The possibility of using contaminated soil in phytoremediation by transgenic plants is promising and a future research direction.
- A large number of scientific publications have been devoted to the problem of Hg environmental pollution. However, these studies face many difficulties: (i) analytical difficulties exist, which are caused by very low Hg contents in all elements of the environment; (ii) the need to determine a specific form of Hg prohibits providing results in total Hg; and (iii) the form in which Hg occurs depends on many environmental factors that must be accurately recognized and described. Unfortunately, many scientific publications do not meet these requirements.

Abbreviations

Hg – mercury

UN - United Nations

UNEP - United Nations Environment Programme

GEM - gaseous elemental mercury Hg⁰

GOM - gaseous mercury in oxidized form

TPM -total particulate mercury

TGM - total gaseous Hg

ROS - reactive oxygen species

OH - hydroxyl radicals

SOD - superoxide dismutase

POD - peroxidase

APX - ascorbate peroxidase

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Availability of data and materials

Not applicable; presented information is based on previously published data only.

Competing interests

The authors declare that they have no competing interests.

Funding

The work was financed from the Own Research Fund Institute of Environment Protection - National Research Institute.

Authors' contributions

WD has been responsible for the concept of the manuscript and drafted the manuscript. BG and AH B-D helped to further elaborate the manuscript. All authors improved the final manuscript.

Acknowledgements

Not applicable.

Author details

¹ Institute of Environmental Protection – National Research Institute, Warsaw, Poland

² Polish Academy of Sciences Botanical Garden – Center for Biological Diversity Conservation in Powsin, Warsaw, Poland

References

1. Pacyna JM (2020) Recent advances in mercury research. *Sci Total Environ* 738:139955-139955. <https://doi.org/10.1016/j.scitotenv.2020.139955>
2. UN Environment (2019) Global Mercury Assessment 2018. UN Environment Programme, Chemical and Health Branch Geneva, Switzerland.
3. Bank MS (2020). The mercury science-policy interface: History, evolution and progress of the Minamata Convention. *Sci Total Environ* 722:137832. <https://doi.org/10.1016/j.scitotenv.2020.137832>
4. AMAP/UN Environment (2019) Technical Background Report for the Global Mercury Assessment 2018. Arctic Monitoring and Assessment Programme, Oslo, Norway/UN Environment Programme, Chemicals and Health Branch, Geneva, Switzerland.
5. Kabata-Pendias A, Pendias H (2001) Trace elements in soils and plants (3rd ed.). CRC Press, Boca Raton. <https://doi.org/10.1201/9781420039900>
6. Newman MC (2014) Fundamentals of ecotoxicology: the science of pollution. 4th Edition, CRC Press, Boca Raton. <https://doi.org/10.1201/b17658>
- 7. Hylander LD, Meili M (2003) 500 years of mercury production: global annual inventory by region until 2000 and associated emissions. *Sci Total Environ* 304:13-27. [https://doi.org/10.1016/S0048-9697\(02\)00553-3](https://doi.org/10.1016/S0048-9697(02)00553-3)**
8. WHO (2017) Ten chemicals of major health concern. Retrieved from www.who.int/ipcs/assessment/public_health/chemicals_phc/en/index.html
9. Minamata Convention on Mercury. (2020) Available via <http://www.mercuryconvention.org/Portals/11/documents/conventionText/Minamata>
10. Selin H, Keane SE, Wang S, Selin NE, Davis K, Bally D (2018) Linking science and policy to support the implementation of the Minamata Convention on Mercury. *Ambio* 47:198-215. <https://doi.org/10.1007/s13280-017-1003-x>
11. Streets DG, Horowitz HM, Lu Z, Levin L, Thackray CP, Sunderland EM (2019) Global and regional trends in mercury emissions and concentrations. 2010–2015. *Atmos Environ* 201:417-427. <https://doi.org/10.1016/j.atmosenv.2018.12.031>
12. Obrist D, Kirk JL, Zhang L, Sunderland EM, Jiskra M, Selin NE (2018) A review of global environmental mercury processes in response to human and natural perturbations: Changes of emissions, climate, and land use. *Ambio* 47:116-140. <https://doi.org/10.1007/s13280-017-1004-9>
13. *ICMGP 2019) 14th International Conference on Mercury as a Global Pollutant*, 8-13 September 2019 Cracow, Poland, Available via <http://www.mercury2019krakow.com>,
14. Liberati A, Altman DG, Tetzlaff J, Mulrow C, Gøtzsche PC, Ioannidis JP, et al. (2009) The PRISMA statement for reporting systematic reviews and meta-analyses of studies that evaluate health care interventions: explanation and elaboration. *J Clin Epidemiol* 62:1-34. <https://doi.org/10.1016/j.jclinepi.2009.06.006>
15. Pirrone N, Cinnirella S, Feng X, Finkelman RB, Friedli HR, Leaner J, et al. (2010) Global mercury emissions to the atmosphere from anthropogenic and natural sources. *Atmos Chem and Phys* 10:5951-5964. <https://doi.org/10.5194/acp-10-5951-2010>
16. Pacyna EG, Pacyna JM, Sundseth K, Munthe C, Kindbom K, Wilson S, et al. (2010) Global emission of mercury to the atmosphere from anthropogenic sources in 2005 and projections to 2020. *Atmos Environ* 44:2487–2499. <https://doi.org/10.1016/j.atmosenv.2009.06.009>
17. Gworek B, Dmochowski W, Baczevska AH, Brągoszewska P, Bemowska-Kalabun O, Wrzosek-Jakubowska J (2017) Air Contamination by Mercury, Emissions and Transformations - a Review. *Water Air Soil Poll* 228:123. <https://doi.org/10.1007/s11270-017-3311-y>
18. UNEP Chemical Branch (2008) The global atmospheric mercury assessment: sources, emissions and transport. Geneva: UNEP-Chemicals.
19. Cooke CA, Martínez-Cortizas A, Bindler R, Gustin MS (2020) Environmental archives of atmospheric Hg deposition—A review. *Sci Total Environ* 709:134800. <https://doi.org/10.1016/j.scitotenv.2019.134800>
20. Miszczak E, Stefaniak S, Michczyński A, Steinnes E, Twardowska I (2020) A novel approach to peatlands as archives of total cumulative spatial pollution loads from atmospheric deposition of airborne elements complementary to EMEP data: priority pollutants (Pb, Cd, Hg). *Sci Total Environ* 705:135776. <https://doi.org/10.1016/j.scitotenv.2019.135776>
21. Zhu C, Tian H, Hao J (2020) Global anthropogenic atmospheric emission inventory of twelve typical hazardous trace elements, 1995–2012. *Atmos Environ* 220:117061. <https://doi.org/10.1016/j.atmosenv.2019.117061>
22. Eckley C, Blanchard P, McLennan D, Mintz R, Sekela M (2015) Soil–air mercury flux near a large industrial emission source before and after closure (Flin Flon, Manitoba, Canada). *Environ Sci Technol*, 49:9750-9757. <https://doi.org/10.1021/acs.est.5b01995>
23. Eckley CS, Gilmour CC, Janssen S, Luxton TP, Randall PM, Whalin L, et al. (2020) The assessment and remediation of mercury contaminated sites: A review of current approaches. *Sci Total Environ* 707:136031. <https://doi.org/10.1016/j.scitotenv.2019.136031>

24. Agnan Y, Le Dantec T, Moore CW, Edwards GC, Obrist D (2016) New constraints on terrestrial surface-atmosphere fluxes of gaseous elemental mercury using a global database. *Environ Sci Technol* 50:507-524. <https://doi.org/10.1021/acs.est.5b04013>
25. Travnikov O, Dastoor A, Bullock R, Christensen JH (2008) Modeling atmospheric transport and deposition. in: AMAP/UNEP, Technical Background Report to the Global Atmospheric Mercury Assessment. Arctic Monitoring and Assessment Programme, UNEP Chemical Branch, pp. 79-107.
26. Dastoor AP, Larocque Y (2004) Global circulation of atmospheric mercury: a modelling study. *Atmos Environ* 38:147-161. <https://doi.org/10.1016/j.atmosenv.2003.08.037>
27. Henze DK, Hakami A, Seinfeld JH (2007) Development of the adjoint of GEOS-Chem. *Atmos Chem Phys* 79:2413-2433. <https://resolver.caltech.edu/CaltechAUTHORS:HENacp07>
28. Seigneur C, Vijayaraghavan K, Lohman K, Levin L (2009) The AER/EPRI global chemical transport model for mercury (CTM-HG). in: Mercury Fate and Transports in the Global Atmosphere, Pirrone N. and Mason R. (Eds.), Springer, New York, pp. 589-602.
29. Travnikov O, Ilyin I (2009) The EMEP/MSC-E mercury modeling system. in: Pirrone N, Mason R (Eds.), Mercury Fate and Transports in the Global Atmosphere, Springer, New York, pp. 571-587. https://doi.org/10.1007/978-0-387-93958-2_20
- 30. Travnikov O, Angot H, Artaxo P, Bencardino M, Bieser J, D'Amore F, et al. (2017). Multi-model study of mercury dispersion in the atmosphere: atmospheric processes and model evaluation. *Atmos Chem Phys* 5271-5295. <https://doi.org/10.5194/acp-17-5271-201>**
31. De Simone F, Gencarelli CN, Hedgecock IM, Pirrone N (2014) Global atmospheric cycle of mercury: a model study on the impact of oxidation mechanisms. *Environ Sci Pollut R* 21:4110-4123. <https://doi.org/10.1007/s11356-013-2451-x>.
32. Nriagu JO, Pacyna JM (1998) Quantitative assessment of worldwide contamination of air, water and soils by trace metals. *Nature* 333:134-139. <https://doi.org/10.1038/333134a0>
33. Shetty SK, Lin CJ, Streets DG, Jang C (2008) Model estimate of mercury emission from natural sources in East Asia. *Atmos Environ* 42:8674-8685. <https://doi.org/10.1016/j.atmosenv.2008.08.026>
34. Liu G, Cai Y, O'Driscoll N, Feng X, Jiang G. (2012) Overview of Mercury in the Environment. in: Liu G, Cai Y, O'Driscoll N (Eds.), Environmental Chemistry and Toxicology of Mercury. J. Wiley & Sons, Inc., Hoboken, USA, pp. 1-12.
35. Gustin MS, Lindberg SE, Austin K, Coolbaugh M, Vette A, Zhang H (2000) Assessing the contribution of natural sources to regional atmospheric mercury budgets. *Sci Total Environ* 259:61-71. [https://doi.org/10.1016/S0048-9697\(00\)00556-8](https://doi.org/10.1016/S0048-9697(00)00556-8)
36. Gustin MS, Lindberg SE, Weisberg PJ (2008) An update on the natural sources and sinks of atmospheric mercury. *Appl Geoch* 23:482-493. <https://doi.org/10.1016/j.apgeochem.2007.12.010>
37. Streets DG, Horowitz HM, Jacob DJ, Lu Z, Levin L, Ter Schure AF, et al. (2017) Total mercury released to the environment by human activities. *Environ Sci Technol* 51:5969-5977. <https://doi.org/10.1021/acs.est.7b00451>
38. Mason RP (2009) Mercury Emissions from Natural Processes and their Importance in the Global Mercury Cycle. in: Pirrone N, Mason R (Eds.), Mercury Fate and Transports in the Global Atmosphere, Springer, New York, pp. 173-191. https://doi.org/10.1007/978-0-387-93958-2_7
39. Wang Y, Greger M (2004) Clonal differences in mercury tolerance, accumulation, and distribution in willow. *J Environ Qual* 33:1779-1785. <https://doi.org/10.2134/jeq2004.1779>
40. Bergan T, Gallardo L, Rohde H (1999) Mercury in the global troposphere - a three-dimensional model study. *Atmos Environ* 33:1575-1585. [https://doi.org/10.1016/S1352-2310\(98\)00370-7](https://doi.org/10.1016/S1352-2310(98)00370-7)
41. Mason RP, Sheu GR (2002) Role of the ocean in the global mercury cycle. *Global Biogeochem Cy* 16:1093. <https://doi.org/10.1029/2001GB001440>.
42. Lamborg CH, Fitzgerald WF, O'Donnell J, Torgersen T (2002) A non-steady-state compartmental model of global-scale mercury biogeochemistry with interhemispheric atmospheric gradients. *Geochim Cosmochim Acta* 66:1105-1118. [https://doi.org/10.1016/S0016-7037\(01\)00841-9](https://doi.org/10.1016/S0016-7037(01)00841-9)
43. Nriagu J, Becker C (2003) Volcanic emissions of mercury to the atmosphere: global and regional inventories *Sci Total Environ* 304:3-12. [https://doi.org/10.1016/S0048-9697\(02\)00552-1](https://doi.org/10.1016/S0048-9697(02)00552-1)
44. Pyle DM, Macher RA (2003) The importance of volcanic emissions for the global atmospheric mercury cycle. *Atmos Environ* 37:5115-5124. <https://doi.org/10.1016/j.atmosenv.2003.07.011>

45. Bagnato E, Aiuppa A, Parello F, Allard P, Shinohara H, Liuzzo M, et al. (2011) New clues on the contribution of Earth's volcanism to the global mercury cycle. *B Volcanol* 73:497-510. <https://doi.org/10.1007/s00445-010-0419->
46. Bagnato E, Aiuppa A, Parello F, Calabrese S, D'Alessandro W, Mather TA, et al. (2007) Degassing of gaseous (elemental and reactive) and particulate mercury from Mount Etna volcano (Southern Italy). *Atmos Environ* 41:377-7388. <https://doi.org/10.1016/j.atmosenv.2007.05.060>
47. Witt MLI, Mather TA, Pyle DM, Aiuppa A, Bagnato E, Tsanev VI (2008) Mercury and halogen emissions from Masaya and Telica volcanoes, Nicaragua. *J Geophys Res Solid Earth* 113(B6). <https://doi.org/10.1029/2007JB005401>
48. Martin RS, Witt MLI, Pyle DM, Mather TA, Watt SFL, Bagnato E, et al. (2011) Rapid oxidation of mercury (Hg) at volcanic vents: Insights from high temperature thermodynamic models of Mt Etna's emissions. *Chem Geol* 283:279-286. <https://doi.org/10.1016/j.chemgeo.2011.01.027>
49. Ermolin MS, Fedotov PS, Malik NA, Karandashev VK (2018) Nanoparticles of volcanic ash as a carrier for toxic elements on the global scale. *Chemosphere* 200:16-22. <https://doi.org/10.1016/j.chemosphere.2018.02.089>
50. Coolbaugh M, Gustin M, Rytuba J (2002) Annual emissions of mercury to the atmosphere from natural sources in Nevada and California. *Environ Geol* 42:338-349. <https://doi.org/10.1007/s00254-002-0557-4>.
51. Cinnirella S, Pirrone N (2006) Spatial and temporal distributions of mercury emissions from forest fires in Mediterranean region and Russian federation. *Atmos Environ* 40:7346-7361. <https://doi.org/10.1016/j.atmosenv.2006.06.051>
52. Friedli HR, Arellano AF, Cinnirella S, Pirrone N (2009) Mercury emissions from global biomass burning: spatial and temporal distribution. in: Mason R., Pirrone N. (Eds), *Mercury Fate and Transport in the Global Atmosphere*. Springer, Boston, MA. https://doi.org/10.1007/978-0-387-93958-2_8
53. Obrist D, Gustin MS, Arnone III JA, Johnson DW, Schorran DE, Verburg PS (2005) Measurements of gaseous elemental mercury fluxes over intact tallgrass prairie monoliths during one full year. *Atmos. Environ.*, 39, 957-965. <https://doi.org/10.1016/j.atmosenv.2004.09.081>
- 54. Woodruff LG, Cannon WF (2010) Immediate and long-term fire effects on total mercury in forests soils of northeastern Minnesota. *Environ Sci Technol* 44:5371-5376. <https://doi.org/10.1021/es100544d>**
55. Gagliano AL, Calabrese S, Daskalopoulou K, Cabassi J, Capecchiacci F, Tassi F, et al. (2019) Degassing and Cycling of Mercury at Nisyros Volcano (Greece). *Geofluids*, 2019, ID4783513. <https://doi.org/10.1155/2019/4783514>
56. Pierce AM, Moore CW, Wohlfahrt G, Hörtnagl L, Kljun N, Obrist D (2015) Eddy covariance flux measurements of gaseous elemental mercury using cavity ring-down spectroscopy. *Environ Sci Technol* 49:1559-1568. <https://doi.org/10.1021/es505080z>
- 57. Obrist D, Johnson DW, Lindberg SE (2009) Mercury concentrations and pools in four Sierra Nevada forest sites, and relationships to organic carbon and nitrogen. *Biogeosciences* 6:765-777. <https://doi.org/10.1016/j.atmosenv.2004.09.081>**
58. Jiskra M, Sonke JE, Obrist D, Bieser J, Ebinghaus R, Myhre CL et al. (2018) A vegetation control on seasonal variations in global atmospheric mercury concentrations. *Nat Geosci* 11:244-250. <https://doi.org/10.1038/s41561-018-0078-8>
59. Gustin MS (2012) Exchange of mercury between the atmosphere and terrestrial ecosystems. in: Liu, G., Cai, Y., O'Driscoll, N. (Eds.), *Environmental chemistry and toxicology of mercury*. John Wiley & Sons, pp. 423-451
60. Smith-Downey NV, Sunderland EM, Jacob DJ (2010) Anthropogenic impacts on global storage and emissions of mercury from terrestrial soils: Insights from a new global model. *J Geophys Res: Biogeosci* 115(G3). <https://doi.org/10.1029/2009JG001124>
61. Graydon JA, St. Louis VL, Lindberg SE, Sandilands KA, Rudd JW, Kelly CA et al. (2012) The role of terrestrial vegetation in atmospheric Hg deposition: Pools and fluxes of spike and ambient Hg from the METAALICUS experiment. *Global Biogeochem Cy* 26:GB1022. <https://doi.org/10.1029/2011GB004031>
62. Zheng W, Obrist D, Weis D, Bergquist BA (2016) Mercury isotope compositions across North American forests. *Global Biogeochem Cy* 30:1475-1492. <https://doi.org/10.1002/2015GB005323>
63. Sommar J, Osterwalder S, Zhu W (2020) Recent advances in understanding and measurement of Hg in the environment: Surface-atmosphere exchange of gaseous elemental mercury (Hg⁰). *Sci Total Environ* 137648. <https://doi.org/10.1016/j.scitotenv.2020.137648>
64. Wright LP, Zhang L, Marsik FJ (2016) Overview of mercury dry deposition, litterfall, and throughfall studies. *Atmos Chem Phys* 16:13399. <https://doi.org/10.5194/acp-16-13399-2016>

65. Lyman SN, Cheng I, Gratz LE, Weiss-Penzias P, Zhang L (2020) An updated review of atmospheric mercury. *Sci Total Environ* 707:135575. <https://doi.org/10.1016/j.scitotenv.2019.135575>
66. Skov H, Bullock R, Christensen JH, Sørensen LL (2008). Atmospheric pathways. in: *AMAP/UNEP Technical Background Report to the Global Atmospheric Mercury Assessment*. Arctic Monitoring and Assessment Programme, UNEP Chemical Branch, pp. 64-72.
67. Selin NE, Jacob DJ, Park RJ., Yantosca RM, Strode S, Jaeglé L, et al. (2007) Chemical cycling and deposition of atmospheric mercury: Global constraints from observations, *J Geophys Res* 112:D02308, 14 PP. <https://doi.org/10.1029/2006JD007450>
68. Lindberg SE, Dong W, Meyers T. (2002) Transpiration of gaseous elemental mercury through vegetation in a subtropical wetland in Florida. *Atmos Environ*, 36:5207-5219. [https://doi.org/10.1016/S1352-2310\(02\)00586-1](https://doi.org/10.1016/S1352-2310(02)00586-1)
69. Marsik FJ, Keeler GJ, Landis MS (2007) The dry-deposition of speciated mercury to the Florida Everglades: Measurements and modeling. *Atmos Environ* 41:136-149. <https://doi.org/10.1016/j.atmosenv.2006.07.032>
70. Hanson PJ, Lindberg SE, Tabberer TA, Owens JA, Kim KH (1995) Foliar exchange of mercury vapor: evidence for a compensation point. *Water Air Soil Poll* 80:373-382. <https://doi.org/10.1007/BF01189687>
71. Caffrey PF, Ondov JM, Zufall MJ, Davidson CI (1998) Determination of size-dependent dry particle deposition velocities with multiple intrinsic elemental tracers. *Environ Sci Technol* 32:1615-1622. <https://doi.org/10.1021/es970644f>
72. Converse AD, Riscassi AL, Scanlon TM (2010) Seasonal variability in gaseous mercury fluxes measured in a high-elevation meadow. *Atmos Environ* 44:2176-2185. <https://doi.org/10.1016/j.atmosenv.2010.03.024>
73. Carpi A, Fostier AH, Orta OR, dos Santos JC, Gittings M (2014.) Gaseous mercury Carpemissions from soil following forest loss and land use changes: Field experiments in the United States and Brazil. *Atmos Environ* 96:423-429. <https://doi.org/10.1016/j.atmosenv.2014.08.004>
74. Zhang H, Lindberg SE, Marsik FJ, Keeler GJ (2001) Mercury air/surface exchange kinetics of background soils of the Tahquamenon River watershed in the Michigan Upper Peninsula. *Water Air Soil Poll* 126:151-169.
75. Ferrari CP, Dommerguea A, Veyseyrea A, Planchona F, Boutrona CF (2002) Mercury speciation in the French seasonal snow cover. *Sci Total Environ* 287, 61-69. [https://doi.org/10.1016/S0048-9697\(01\)00999-8](https://doi.org/10.1016/S0048-9697(01)00999-8)
76. Gustin MS, Biester H, Kim CS (2002) Investigation of the light-enhanced emission of mercury from naturally enriched substrates. *Atmos Environ* 36:3241-3254. [https://doi.org/10.1016/S1352-2310\(02\)00329-1](https://doi.org/10.1016/S1352-2310(02)00329-1)
77. Poissant L, Pilote M, Xu X, Zhang H, Beauvais C (2004) Atmospheric mercury speciation and deposition in the Bay St. Francois wetlands. *J Geophys Res* 109:1-14. <https://doi.org/10.1029/2003JD004364>
78. Ericksen JA, Gustin MS, Xin M, Weisberg PJ, Fernandez GCJ (2006) Air–soil exchange of mercury from background soils in the United States. *Sci Total Environ* 366:851-863. <https://doi.org/10.1016/j.scitotenv.2005.08.019>
79. Gustin MS (2003) Are mercury emissions from geologic sources significant? A status report. *Sci. Total Environ.*, 304, 153-167. [https://doi.org/10.1016/S0048-9697\(02\)00565-X](https://doi.org/10.1016/S0048-9697(02)00565-X)
80. Gustin MS, Lindberg SE (2005) Terrestrial Hg Fluxes: Is the Next Exchange Up, Down, or Neither?. in: Pirrone, N., Mahaffey, K.R. (Eds.), *Dynamics of Mercury Pollution on Regional and Global Scales*, Springer, Boston, MA, pp. 241-259. https://doi.org/10.1007/0-387-24494-8_11
81. Stamenkovic J, Gustin MS (2007) Evaluation of use of EcoCELL technology for quantifying total gaseous mercury fluxes over background substrates. *Atmos Environ* 41:3702-3712. <https://doi.org/10.1016/j.atmosenv.2006.12.037>
82. Mason RP (2009) Mercury Emissions from Natural Processes and their Importance in the Global Mercury Cycle. in: Pirrone N, Mason R (Eds.), *Mercury Fate and Transports in the Global Atmosphere*, Springer, New York, pp.173-191. https://doi.org/10.1007/978-0-387-93958-2_7
83. Mason RP, Pirrone N, Hedgecock I, Suzuki N, Levin L (2010) Conceptual overview. in: Pirrone N., Keating T. (Eds.), *Hemispheric transport of air pollution -part B.*, United Nations Publication, New York, Geneva, pp. 1-19. <https://doi.org/10.18356/38ccc958-en>
84. Carpi A, Lindberg SE (1997) Sunlight-Mediated Emission of Elemental Mercury from Soil Amended with Municipal Sewage Sludge. *Environ Sci Technol* 31:2085-2091. <https://doi.org/10.1021/Es960910+>
85. Zhu W, Lin CJ, Wang X, Sommar J, Fu X, Feng X (2016) Global observations and modeling of atmosphere-surface exchange of elemental mercury: a critical review. *Atmos Chem Phys* 16:4451-4480. <https://doi.org/10.5194/acp-16-4451-2016>
86. Hanson PJ, Lindberg SE, Tabberer TA, Owens JA, Kim KH (1995) Foliar exchange of mercury vapor: evidence for a compensation point. *Water Air Soil Poll* 80:373-382. <https://doi.org/10.1007/BF01189687>

87. Bash JO, Miller DR, Meyer TH, Bresnahan PA (2004) Northeast United States and Southeast Canada natural mercury emissions estimated with a surface emission model. *Atmos Environ* 38:5683-5692. <https://doi.org/10.1016/j.atmosenv.2004.05.058>
88. Rea AW, Lindberg SE, Scherbatskoy T, Keeler GJ (2002). Mercury accumulation in foliage over time in two northern mixed-hardwood forests. *Water Air Soil Pollut* 133:49-67. <https://doi.org/10.1023/A:1012919731598>
89. Magarelli G, Fostier AH (2005) Influence of deforestation on the mercury air/soil exchange in the Negro River Basin, Amazon. *Atmos Environ* 39:7518-7528. <https://doi.org/10.1016/j.atmosenv.2005.07.067>
90. Selin MS, Lindberg SE (2005) Terrestrial Hg Fluxes: Is the Next Exchange Up, Down, or Neither?. in: Pirrone, N., Mahaffey, K.R. (Eds.), *Dynamics of Mercury Pollution on Regional and Global Scales*, Springer, Boston, MA, pp. 241-259. https://doi.org/10.1007/0-387-24494-8_11
91. Gustin MS, Engle M, Ericksen J, Lyman S, Stamenkovic J, Xin M (2006) Mercury exchange between the atmosphere and low mercury containing substrates. *Appl Geoch* 21:1913-1923. <https://doi.org/10.1016/j.apgeochem.2006.08.007>
92. Selin NE, Jacob DJ, Yantosca RM, Strode S, Jaeglé L, Sunderland EM (2008) Global 3D land, ocean, atmosphere model for mercury: Present day versus preindustrial cycles and anthropogenic enrichment factors for deposition. *Global Biogeochem Cy* 22:1-13. <https://doi.org/10.1029/2007GB003040>
93. Ericksen JA, Gustin MS, Schorran DE, Johnson DW, Lindberg SE, Coleman JS (2003) Accumulation of atmospheric mercury in forest foliage. *Atmos Environ* 37:1613-1622. [https://doi.org/10.1016/S1352-2310\(03\)00008-6](https://doi.org/10.1016/S1352-2310(03)00008-6)
94. Osterwalder S, Huang JH, Shetaya WH, Agnan Y, Frossard A, Frey, B et al. (2019) Mercury emission from industrially contaminated soils in relation to chemical, microbial, and meteorological factors. *Environ. Pollut.*, 250, 944-952. <https://doi.org/10.1016/j.envpol.2019.03.093>
95. Gabriel MC, Williamson DG, Brooks S, Zhang H, Lindberg S (2005) Spatial variability of mercury emissions from soils in a southeastern US urban environment. *Environ Geol* 48:955-964. <https://doi.org/10.1007/s00254-005-0043-x>
96. Ma M, Wang D, Sun R, Shen Y, Huang L (2013) Gaseous mercury emissions from subtropical forested and open field soils in a national nature reserve, southwest China *Atmos Environ* 64:116-123. <https://doi.org/10.1016/j.atmosenv.2012.09.038>
97. Wang X, Lin CJ, Yuan W, Sommar J, Zhu W, Feng X (2016) Emission-dominated gas exchange of elemental mercury vapor over natural surfaces in China. *Atmos Chem Phys* 16:11125–11143. doi:10.5194/acp-16-11125-2016
98. Sommar J, Zhu W, Shang L, Lin CJ, Feng XB (2015) Seasonal variations in metallic mercury (Hg⁰) vapor exchange over biannual wheat-corn rotation cropland in the North China Plain. *Biogeosciences Discuss* 12:16105–16158. doi:10.5194/bgd-12-16105-2015
99. Gao Y, Wang Z, Zhang X, Wang C (2020) Observation and estimation of mercury exchange fluxes from soil under different crop cultivars and planting densities in North China Plain. *Environ Pollut* 259:113833. <https://doi.org/10.1016/j.envpol.2019.113833>
100. Bishop K, Shanley JB, Riscassi A, de Wit HA, Eklöf K, Meng, B, et al. (2020) Recent advances in understanding and measurement of mercury in the environment: Terrestrial Hg cycling. *Sci Total Environ* 721:137647. <https://doi.org/10.1016/j.scitotenv.2020.137647>
101. Qiu G, Feng X, Wang S, Shang L (2006) Environmental contamination of mercury from Hg-mining areas in Wuchuan, northeastern Guizhou, China. *Environ Pollut* 142:549-558. <https://doi.org/10.1016/j.envpol.2005.10.015>
102. Li Q, Tang L, Qiu G, Liu C (2020) Total mercury and methylmercury in the soil and vegetation of a riparian zone along a mercury-impacted reservoir. *Sci Total Environ* 738:139794. <https://doi.org/10.1016/j.scitotenv.2020.139794>
103. Goulet RR, Holmes J, Tessier A, Wang F, Siciliano SD, Page B et al. (2007) Mercury methylation in sediments of a riverine marsh: the role of redox conditions sulfur chemistry and microbial communities. *Geochim Cosmochim Ac.*71:3396-3406.
104. Windham-Myers L, Marvin-DiPasquale M, Kakouros E, Agee JL, Kieu LH, Stricker CA, et al. (2014) Mercury cycling in agricultural and managed wetlands of California, USA: Seasonal influences of vegetation on mercury methylation, storage, and transport. *Sci Total Environ* 484:308-318. <https://doi.org/10.1016/j.scitotenv.2013.05.027>
105. Kronberg RM, Jiskra M, Wiederhold JG, Bjorn E, Skjallberg U (2016) Methyl mercury formation in hillslope soils of boreal forests: The role of forest harvest and anaerobic microbes. *Environ Sci Technol* 50:9177-9186. <https://doi.org/10.1021/acs.est.6b00762>
106. Gnamuš A, Byrne AR, Horvat M (2000) Mercury in the soil-plant-deer-predator food chain of a temperate forest in Slovenia. *Environ Sci Technol* 34:3337-3345. <https://doi.org/10.1021/es991419w>
107. Meng B, Feng X, Qiu G, Cai Y, Wang D, Li P, et al. (2010) Distribution patterns of inorganic mercury and methylmercury in tissues of rice (*Oryza sativa* L.) plants and possible bioaccumulation pathways. *J Agr Food Chem* 58:4951-4958. <https://doi.org/10.1021/jf904557x>

108. Gilli R, Karlen C, Weber M, Rüegg J, Barmettler K, Biester H, et al. (2018) Speciation and mobility of mercury in soils contaminated by legacy emissions from a chemical factory in the Rhône valley in canton of Valais, Switzerland. *Soil Systems*, 2:44. <https://doi.org/10.3390/soilsystems2030044>

109. Lima FRD, Martins GC, Silva AO, Vasques ICF, Engelhardt MM, Cândido GS, et al. (2019) Critical mercury concentration in tropical soils: Impact on plants and soil biological attributes. *Sci Total Environ* 666:472-479. <https://doi.org/10.1016/j.scitotenv.2019.02.216>
110. Gonzalez-Raymat H, Liu G, Liriano C, Li Y, Yin Y, Shi J et al. (2017) Elemental mercury: Its unique properties affect its behavior and fate in the environment. *Environ Pollut* 229 :69-86. <https://doi.org/10.1016/j.envpol.2017.04.101>
111. Xu J, Buck M, Eklöf K, Ahmed OO, Schaefer JK, Bishop K, et al. (2019) Mercury methylating microbial communities of boreal forest soils. *Sci Rep* 9:518. <https://doi.org/10.1038/s41598-018-37383-z>
112. Qianqian LING, Yingying GUO, Yong LIANG, Yongguang YIN, Yong CAI (2020) Microbial uptake of HgS nanoparticles and its effect on mercury methylation. *Environ Chem* 2:292-300. <https://doi:10.7524/j.issn.0254-6108.2019112601>
113. Bigham GN, Murray KJ, Masue-Slowey Y, Henry EA (2017) Biogeochemical controls on methylmercury in soils and sediments: Implications for site management. *Integr Environ Assess Manag* 13:249-263. <https://doi.org/10.1002/ieam.1822>
114. Shahid M, Khalid S, Bibi I, Bundschuh J, Niazi NK, Dumat C (2020) A critical review of mercury speciation, bioavailability, toxicity and detoxification in soil-plant environment: Ecotoxicology and health risk assessment. *Sci Total Environ* 711:134749. <https://doi.org/10.1016/j.scitotenv.2019.134749>
115. Zhao L, Qiu G, Anderson CW, Meng B, Wang D, Shang L, et al. (2016) Mercury methylation in rice paddies and its possible controlling factors in the Hg mining area, Guizhou province, Southwest China. *Environ Pollut* 215:1-9. <https://doi.org/10.1016/j.envpol.2016.05.001>
116. Eklöf K, Bishop K, Bertilsson S, Björn E, Buck M, Skjällberg U, et al. (2018) Formation of mercury methylation hotspots as a consequence of forestry operations. *Sci Total Environ* 613:1069-1078. <https://doi.org/10.1016/j.scitotenv.2017.09.151>
117. Qiu G, Feng X, Wang S, Shang L (2005) Mercury and methylmercury in riparian soil, sediments, mine-waste calcines, and moss from abandoned Hg mines in east Guizhou province, southwestern China. *Appl Geoch* 20:627-638. <https://doi.org/10.1016/j.apgeochem.2004.09.006>
118. Qiu G, Feng X, Wang S, Shang L (2006) Environmental contamination of mercury from Hg-mining areas in Wuchuan, northeastern Guizhou, China. *Environ Pollut* 142:549-558. <https://doi.org/10.1016/j.envpol.2005.10.015>
119. Xu X, Meng B, Zhang C, Feng X, Gu C, Guo J, et al. (2017) The local impact of a coal-fired power plant on inorganic mercury and methyl-mercury distribution in rice (*Oryza sativa* L.). *Environ Pollut* 223:11-18. <https://doi.org/10.1016/j.envpol.2016.11.042>
120. Mailman M, Bodaly RA (2005) Total mercury, methyl mercury, and carbon in fresh and burned plants and soil in Northwestern Ontario. *Environ Pollut* 138:161-166. <https://doi.org/10.1016/j.envpol.2005.02.005>
121. Schuster PF, Schaefer KM, Aiken GR Antweiler RC, Dewild JF., Gryziec JD, et al. (2018) Permafrost stores a globally significant amount of mercury. *Geophys Res Lett* 45:1463-1471. <https://doi.org/10.1002/2017GL075571>
122. Hararuk O, Obrist D, Luo Y (2013) Modelling the sensitivity of soil mercury storage to climate-induced changes in soil carbon pools. *Biogeosciences* 10:2393-2407. <https://doi.org/10.5194/bg-10-2393-2013>
123. Amos HM, Sonke JE, Obrist D, Robins N, Hagan N, Horowitz HM, Mason RP, et al. (2015) Observational and modeling constraints on global anthropogenic enrichment of mercury. *Environ Sci Technol* 49:4036-4047. <https://doi.org/10.1021/es5058665>
124. Mason RP, Choi AL, Fitzgerald WF, Hammerschmidt CR, Lamborg CH, Soerensen AL, et al. (2012) Mercury biogeochemical cycling in the ocean and policy implications. *Environ Res* 119:101-117. <https://doi.org/10.1016/j.envres.2012.03.013>
125. Fitzgerald WF, Lamborg CH (2014) Geochemistry of mercury in the environment. In: *Treatise on Geochemistry*, 2nd edition.
126. Amos HM, Jacob DJ, Streets DG, Sunderland EM (2013) Legacy impacts of all-time anthropogenic emissions on the global mercury cycle. *Global Biogeochem Cy* 27: 410-421. <https://doi.org/10.1002/gbc.20040>
127. Wang J, Feng X, Anderson CW, Xing Y, Shang L (2012) Remediation of mercury contaminated sites—a review. *J Hazard Mater* 221:1-18. <https://doi.org/10.1016/j.jhazmat.2012.04.035>
128. Li S, Jia Z (2018) Heavy metals in soils from a representative rapidly developing megacity (SW China): Levels, source identification and apportionment. *Catena* 163:414-423. <https://doi.org/10.1016/j.catena.2017.12.035>

129. Obrist D., Pearson C, Webster J, Kane T, Lin CJ, Aiken GR, et al. (2016) A synthesis of terrestrial mercury in the western United States: Spatial distribution defined by land cover and plant productivity. *Sci Total Environ* 568:522-535. <https://doi.org/10.1016/j.scitotenv.2015.11.104>
130. Tóth G, Hermann T, Szatmári G, Pásztor L (2016) Maps of heavy metals in the soils of the European Union and proposed priority areas for detailed assessment. *Sci Total Environ* 565:1054-1062. <https://doi.org/10.1016/j.scitotenv.2016.05.115>
131. Lima LC, Egreja Filho FB, Linhares LA, Windmoller CC, Yoshida MI (2015) Accumulation and oxidation of elemental mercury in tropical soils. *Chemosphere* 134:181-191. <https://doi.org/10.1016/j.chemosphere.2015.04.020>
132. Sharma R, Ramteke S, Patel KS, Kumar S, Sarangi B, Agrawal SG, et al. (2015) Contamination of Lead and Mercury in Coal Basin of India. *J Environ Prot* 6:1430-1441. <https://doi.org/10.4236/jep.2015.612124>
133. Taylor M (2014) Soil quality monitoring in the Waikato Region 2012. WRC Technical Report 2013/49, Waikato Regional Council, Hamilton, New Zealand. <http://www.waikatoregion.govt.nz/tr201349/>. Accessed 3 Sept, 2014
134. Halbach K, Mikkelsen Ø, Berg T, Steinnes E (2017) The presence of mercury and other trace metals in surface soils in the Norwegian Arctic. *Chemosphere* 188:567-574. <https://doi.org/10.1016/j.chemosphere.2017.09.012>
135. Gasperi J, Ayrault S, Moreau-Guigon E, Alliot F, Labadie P, Budzinski H, et al. (2016) Contamination of soils by metals and organic micropollutants: case study of the Parisian conurbation. *Environ Sci Pollut Res* 25:23559–23573. <https://doi.org/10.1007/s11356-016-8005-2>
136. Bailey EA, Gray JE, Theodorakos PM (2002) Mercury in vegetation and soils at abandoned mercury mines in southwestern Alaska. USA. *Geochem Explor Env A* 2:275-285. <https://doi.org/10.1144/1467-787302-032>
137. Gray JE, Theodorakos PM, Fey DL, Krabbenhoft DP (2015) Mercury concentrations and distribution in soil, water, mine waste leachates, and air in and around mercury mines in the Big Bend region, Texas, *Environ Geochem Hlth*. 37:35-48. <https://doi.org/10.1007/s10653-014-9628-1>
138. Wu G, Kang H, Zhang X, Shao H, Chu L, Ruan C (2010) A critical review on the bio-removal of hazardous heavy metals from contaminated soils: issues, progress, eco-environmental concerns and opportunities. *J Hazard Mater* 174:1-8. <https://doi.org/10.1016/j.jhazmat.2009.09.113>
139. Gosar M, Šajn R, Biester H (2006) Binding of mercury in soils and attic dust in the Idrija mercury mine area (Slovenia). *Sci Total Environ* 369:150-162. <https://doi.org/10.1016/j.scitotenv.2006.05.006>
140. Teršič T, Gosar M, Šajn R (2009) Impact of mining activities on soils and sediments at the historical mining area in Podljubelj, NW Slovenia. *J Geochem Explor* 100:1-10. <https://doi.org/10.1016/j.gexplo.2008.02.005>
141. Banášová V (1999) Vegetation on contaminated sites near an Hg mine and smelter. in: R. Ebinghaus R, Turner RR, de Lacedra LD, Vasiljev O., Salomons W (Eds.), *Mercury Contaminated Sites*. Springer, Berlin, Heidelberg, pp. 321-335. <https://doi.org/10.1007/978-3-662-03754-6>
142. Higuera P, Oyarzun R, Biester H, Lillo J, Lorenzo S (2003) A first insight into mercury distribution and speciation in soils from the Almadén mining district, Spain. *J Geochem Explor* 80(1):95-104. [https://doi.org/10.1016/S0375-6742\(03\)00185-7](https://doi.org/10.1016/S0375-6742(03)00185-7)
143. Dago A, González I, Ariño C, Martínez-Coronado A, Higuera P, Díaz-Cruz JM, et al. (2014) Evaluation of mercury stress in plants from the Almadén mining district by analysis of phytochelators and their Hg complexes. *Environ Sci Technol* 8:6256-6263. <https://doi.org/10.1021/es405619y>
144. Boente C, Albuquerque MTD, Gerassis S, Rodríguez-Valdés E, Gallego JR (2019) A coupled multivariate statistics, geostatistical and machine-learning approach to address soil pollution in a prototypical Hg-mining site in a natural reserve. *Chemosphere* 218:767-777. <https://doi.org/10.1016/j.chemosphere.2018.11.172>
145. Wahsha M, Maleci L, Bini C (2019) The impact of former mining activity on soils and plants in the vicinity of an old mercury mine (Vallalta, Belluno, NE Italy). *Geochem-Explor Envir Anal* 19:171-175. <https://doi.org/10.1144/geochem2018-040>
146. Gemici Ü, Tarcan G, Somay AM, Akar T (2009) Factors controlling the element distribution in farming soils and water around the abandoned Haliköy mercury mine (Beydağ, Turkey). *Appl Geoch* 24:1908-1917. <https://doi.org/10.1016/j.apgeochem.2009.07.004>
147. Xiao R, Wang S, Li R, Wang JJ, Zhang Z (2017) Soil heavy metal contamination and health risks associated with artisanal gold mining in Tongguan, Shaanxi, China. *Ecotox Environ Safe* 141:17-24. <https://doi.org/10.1016/j.ecoenv.2017.03.002>
148. Yin D, He T, Yin R, Zeng L (2018) Effects of soil properties on production and bioaccumulation of methylmercury in rice paddies at a mercury mining area, China. *J Environ Sci* 68:194-205. <https://doi.org/10.1016/j.jes.2018.04.028>
149. Chiarantini L, Rimondi V, Benvenuti M, Beutel MW, Costagliola P, Gonnelli C, et al. (2016) Black pine (*Pinus nigra*) barks as biomonitors of airborne mercury pollution. *Sci Total Environ* 569:105-113. <https://doi.org/10.1016/j.scitotenv.2016.06.029>
150. Hissler C, Probst JL (2006) Impact of mercury atmospheric deposition on soils and streams in a mountainous catchment (Vosges, France) polluted by chlor-alkali industrial activity: the important trapping role of the organic matter. *Sci Total Environ* 361:163-178. <https://doi.org/10.1016/j.scitotenv.2005.05.023>

151. Grangeon S, Guédron S, Asta J, Sarret G, Charlet L (2012) Lichen and soil as indicators of an atmospheric mercury contamination in the vicinity of a chlor-alkali plant (Grenoble, France). *Ecol Indic* 13:178-183. <https://doi.org/10.1016/j.ecolind.2011.05.024>
152. Ullrich SM, Ilyushchenko MA, Kamberov IM, Tanton TW (2007) Mercury contamination in the vicinity of a derelict chlor-alkali plant. Part I: Sediment and water contamination of Lake Ballyldak and the River Irtysh. *Sci Total Environ* 381:1-16. <https://doi.org/10.1016/j.scitotenv.2007.02.033>
153. Esbrí JM, López-Berdonces MA, Fernández-Calderón S, Higuera P, Díez S (2015) Atmospheric mercury pollution around a chlor-alkali plant in Flix (NE Spain): an integrated analysis. *Environ. Sci. Pollut. R.*, 22, 4842-4850. <https://doi.org/10.1007/s11356-014-3305-x>
154. Biester H, Müller G, Schöler HF (2002) Binding and mobility of mercury in soils contaminated by emissions from chlor-alkali plants. *Sci Total Environ* 284:191-203. [https://doi.org/10.1016/S0048-9697\(01\)00885-3](https://doi.org/10.1016/S0048-9697(01)00885-3)
155. Bernaus A, Gaona X, van Ree D, Valiente M (2006) Determination of mercury in polluted soils surrounding a chlor-alkali plant: direct speciation by X-ray absorption spectroscopy techniques and preliminary geochemical characterisation of the area. *Anal Chim Acta* 565:73-80. <https://doi.org/10.1016/j.aca.2006.02.020>
156. Reis AT, Rodrigues SM, Araújo C, Coelho JP, Pereira E, Duarte AC (2009) Mercury contamination in the vicinity of a chlor-alkali plant and potential risks to local population. *Sci Total Environ* 407:2689-2700. <https://doi.org/10.1016/j.scitotenv.2008.10.065>
157. Zheng N, Liu J, Wang Q, Liang Z (2011) Mercury contamination due to zinc smelting and chlor-alkali production in NE China. *Appl Geoch* 26(2):188-193. <https://doi.org/10.1016/j.apgeochem.2010.11.018>
158. Song Z, Li P, Ding L, Li Z, Zhu W, He T, Feng X (2018) Environmental mercury pollution by an abandoned chlor-alkali plant in Southwest China. *J Geochem Explor* 194:81-87. <https://doi.org/10.1016/j.gexplo.2018.07.017>
159. Zhu W, Li Z, Li P, Yu B, Lin CJ, Sommar J, et al. (2018) Re-emission of legacy mercury from soil adjacent to closed point sources of Hg emission. *Environ Pollut* 242:718-727. <https://doi.org/10.1016/j.envpol.2018.07.002>
160. Reis AT, Rodrigues SM, Davidson CM, Pereira E, Duarte AC (2010) Extractability and mobility of mercury from agricultural soils surrounding industrial and mining contaminated areas. *Chemosphere* 81:1369-1377. <https://doi.org/10.1016/j.chemosphere.2010.09.030>
161. Liang J, Feng C, Zeng G, Gao X, Zhong M, Li X, et al. (2017) Spatial distribution and source identification of heavy metals in surface soils in a typical coal mine city, Lianyuan, China. *Environ Pollut* 225:681-690. <https://doi.org/10.1016/j.envpol.2017.03.057>
162. Wu Q, Wang S, Wang L, Liu F, Lin CJ, Zhang L, et al. (2014) Spatial distribution and accumulation of Hg in soil surrounding a Zn/Pb smelter. *Sci Total Environ* 496:668-677. <https://doi.org/10.1016/j.scitotenv.2014.02.067>
163. Wang D, Shi X, Wei S (2003) Accumulation and transformation of atmospheric mercury in soil. *Sci Total Environ* 304:209-214. [https://doi.org/10.1016/S0048-9697\(02\)00569-7](https://doi.org/10.1016/S0048-9697(02)00569-7)
164. Navrátil T, Burns DA, Nováková T, Kaňa, J, Rohovec J, Roll M et al. (2018) Stability of mercury concentration measurements in archived soil and peat samples. *Chemosphere* 208:707-711. <https://doi.org/10.1016/j.chemosphere.2018.06.033>
165. Boszke L, Kowalski A, Siepak J (2004.) Grain size partitioning of mercury in sediments of the middle Odra River (Germany/Poland). *Water Air Soil Poll* 159:125-138. <https://doi.org/10.1023/B:WATE.0000049171.22781.bd>
166. Khan MN, Wasim AA, Sarwar A, Rasheed MF (2011) Assessment of heavy metal toxicants in the roadside soil along the N-5, National Highway, Pakistan. *Environ Monit Assess* 182:587-595. <https://doi.org/10.1007/s10661-011-1899-8>
167. Beckers F, Awad YM, Beiyuan J, Abridata J, Mothes S, Tsang D.C, et al., (2019) Impact of biochar on mobilization, methylation, and ethylation of mercury under dynamic redox conditions in a contaminated floodplain soil. *Environ Inter* 127:276-290. <https://doi.org/10.1016/j.envint.2019.03.040>
168. Bonanno G, Giudice RL, Pavone P (2012) Trace element biomonitoring using mosses in urban areas affected by mud volcanoes around Mt. Etna. The case of the Salinelle, Italy. *Monit Assess* 184:5181-5188. <https://doi.org/10.1007/s10661-011-2332-z>
169. Gosar M, Šajin R, Biester H (2006) Binding of mercury in soils and attic dust in the Idrija mercury mine area (Slovenia). *Sci Total Environ* 369:150-162. <https://doi.org/10.1016/j.scitotenv.2006.05.006>
170. Richardson JB, Moore L (2020) A tale of three cities: Mercury in urban deciduous foliage and soils across land-uses in Poughkeepsie NY, Hartford CT, and Springfield MA USA. *Sci Total Environ* 715:136869. <https://doi.org/10.1016/j.scitotenv.2020.136869>
171. Cheng H, Li M, Zhao C, Li K, Peng M, Qin A, Cheng X (2014) Overview of trace metals in the urban soil of 31 metropolises in China. *J Geochem Explor* 139:31-52. <https://doi.org/10.1016/j.gexplo.2013.08.012>
172. Tjihuis L, Brattli B, Sæther OM (2002) A geochemical survey of topsoil in the city of Oslo, Norway. *Environ Geochem Hlth* 24:67-94. <https://doi.org/10.1023/A:1013979700212>

173. Manta DS, Angelone M, Bellanca A, Neri R, Sprovieri M (2002) Heavy metals in urban soils: a case study from the city of Palermo (Sicily), Italy. *Sci Total Environ* 300:229-243. [https://doi.org/10.1016/S0048-9697\(02\)00273-5](https://doi.org/10.1016/S0048-9697(02)00273-5)
174. Rodrigues S, Pereira ME, Duarte AC, Ajmone-Marsan F, Davidson CM, Grčman H, et al. (2006) Mercury in urban soils: a comparison of local spatial variability in six European cities. *Sci Total Environ* 368:926-936. <https://doi.org/10.1016/j.scitotenv.2006.04.008>
175. Cheng H, Zhao C, Liu F, Yang K, Liu Y, Li M, et al. (2013) Mercury drop trend in urban soils in Beijing, China, since 1987. *J Geochem Explor* 124:195-202. <https://doi.org/10.1016/j.gexplo.2012.09.007>
176. Ottesen RT, Birke M, Finne TE, Gosar M, Locutura J, Reimann C, et al. (2013) Mercury in European agricultural and grazing land soils. *Appl Geoch* 33:1-12. <https://doi.org/10.1016/j.apgeochem.2012.12.013>
177. Loska K, Wiechuła D, Korus I (2004) Metal contamination of farming soils affected by industry. *Environ Int* 30:159-165. [https://doi.org/10.1016/S0160-4120\(03\)00157-0](https://doi.org/10.1016/S0160-4120(03)00157-0)
178. Ahmadi M, Akhbarizadeh R, Haghighifard NJ, Barzegar G, Jorfi S (2019) Geochemical determination and pollution assessment of heavy metals in agricultural soils of south western of Iran *J Environ Health Sci. Eng*, 1-13. <https://doi.org/10.1007/s40201-019-00379-6>
179. Navrátil T, Shanley J, Rohovec J., Hojdová M, Penížek V, Buchtová J (2014) Distribution and pools of mercury in Czech forest soils. *Water Air Soil Poll* 225:1829. <https://doi.org/10.1007/s11270-013-1829-1>
180. Åkerblom S, Meili M, Bringmark L, Johansson K, Kleja DB, Bergkvist B (2008) Partitioning of Hg between solid and dissolved organic matter in the humus layer of boreal forests. *Water Air Soil Poll*, 189: 239-252. <https://doi.org/10.1007/s11270-007-9571-1>
181. Boszke L, Kowalski A, Siepak J (2004.) Grain size partitioning of mercury in sediments of the middle Odra River (Germany/Poland). *Water Air Soil Poll* 159:125-138. <https://doi.org/10.1023/B:WATE.0000049171.22781.bd>
182. Fiorentino JC, Enzweiler J, Angélica RS (2011) Geochemistry of mercury along a soil profile compared to other elements and to the parental rock: evidence of external input. *Water Air Soil Poll* 221:63-75. <https://doi.org/10.1007/s11270-011-0769-x>
183. do Valle CM, Santana GP, Augusti R, Egreja Filho FB, Windmüller CC (2005) Speciation and quantification of mercury in Oxisol, Ultisol, and Spodosol from Amazon (Manaus, Brazil). *Chemosphere* 58:779-792. <https://doi.org/10.1016/j.chemosphere.2004.09.005>
184. Armesto AG, Bibián-Núñez L, Campillo-Cora C, Pontevedra-Pombal X, Arias-Estévez M, Nóvoa-Muñoz JC. (2018) Total mercury distribution among soil aggregate size fractions in a temperate forest podzol. *Span J Soil Sci* 8:190-202. <https://doi.org/10.3232/SJSS.2018.V8.N1.05>
185. Yin R, Gu C, Feng X, Hurley JP, Krabbenhoft DP, Lepak RF, et al. (2016) Distribution and geochemical speciation of soil mercury in Wanshan Hg mine: Effects of cultivation. *Geoderma* 272:32-38. <https://doi.org/10.1016/j.jes.2018.04.028>
186. Wang DY, Qing CL, Guo TY, Guo YJ (1997) Effects of humic acid on transport and transformation of mercury in soil-plant systems. *Water Air Soil Poll* 95:35-43. <https://doi.org/10.1007/BF02406154>
187. Smith T, Pitts K, McGarvey JA, Summers AO (1998) Bacterial oxidation of mercury metal vapor, Hg(0). *Appl Environ Microbiol* 64:1328-1332.
188. Colombo MJ, Ha J, Reinfelder JR, Barkay T, Yee N (2013) Anaerobic oxidation of Hg (0) and methylmercury formation by *Desulfovibrio desulfuricans* ND132. *Geochim Cosmochim. Ac* 112:166-177. <https://doi.org/10.1016/j.gca.2013.03.001>
189. Colombo MJ, Ha J, Reinfelder JR, Barkay T, Yee N (2014) Oxidation of Hg(0) to Hg(II) by diverse anaerobic bacteria. *Chem Geol* 363:334-340. <https://doi.org/10.1016/j.chemgeo.2013.11.020>
190. Gustin MS, Ericksen JA, Schorran DE, Johnson DW, Lindberg SE, Coleman JS (2004) Application of controlled mesocosms for understanding mercury air-soil - plant exchange. *Environ Sci Technol* 38:6044-6050. <https://doi.org/10.1021/es0487933>
191. Fantozzi L, Ferrara R, Dini F, Tamburello L, Pirrone N, Sprovieri F (2013) Study on the reduction of atmospheric mercury emissions from mine waste enriched soils through native grass cover in the Mt. Amiata region of Italy. *Environ Res* 125:69-74. <https://doi.org/10.1016/j.envres.2013.02.004>
192. Mazur M, Mitchell CPJ, Eckley CS, Eggert SL, Kolka RK, Sebestyen SD, et al. (2014) Gaseous mercury fluxes from forest soils in response to forest harvesting intensity: A field manipulation experiment. *Sci Total Environ* 496:678-687. <https://doi.org/10.1016/j.scitotenv.2014.06.058>
193. Leonard TL, Taylor Jr, GE, Gustin MS, Fernandez GC (1998) Mercury and plants in contaminated soils: 1. Uptake, partitioning, and emission to the atmosphere. *Environ Toxicol Chem* 17:2063-2071. <https://doi.org/10.1002/etc.5620171024>
194. Frescholtz TF, Gustin MS, Schorran DE, Fernandez GC (2003) Assessing the source of mercury in foliar tissue of quaking aspen. *Environ Toxicol Chem* 22:2114-2119. <https://doi.org/10.1002/etc.5620220922>
195. Assad M, Parelle J, Cazaux D, Gimbert F, Chalot M, Tatin-Froux F (2016) Mercury uptake into poplar leaves. *Chemosphere* 146:1-7. <https://doi.org/10.1016/j.chemosphere.2015.11.103>

196. Fleck JA, Grigal DF, Nater EA (1999) Mercury uptake by trees: An observational experiment. *Water Air Soil Poll.*, 115, 513-523. <https://doi.org/10.1023/A:1005194608598>
197. Laacouri A, Nater EA, Kolka RK (2013) Distribution and uptake dynamics of mercury in leaves of common deciduous tree species in Minnesota, USA. *Environ Sci Technol* 47:10462-10470. <https://doi.org/10.1021/es401357z>
198. Arnold J, Gustin MS, Weisberg PJ (2018) Evidence for nonstomatal uptake of Hg by aspen and translocation of Hg from foliage to tree rings in Austrian pine. *Environ Sci Technol* 52:1174-1182. <https://doi.org/10.1021/acs.est.7b04468>
199. Moreno-Jiménez E, Gamarra R, Carpena-Ruiz RO, Millán R, Peñalosa JM, Esteban E (2006) Mercury bioaccumulation and phytotoxicity in two wild plant species of Almadén area. *Chemosphere* 63:1969-1973. <https://doi.org/10.1016/j.chemosphere.2005.09.043>
200. Qian X, Wu Y, Zhou H, Xu X, Xu Z, Shang L, et al. (2018) Total mercury and methylmercury accumulation in wild plants grown at wastelands composed of mine tailings: Insights into potential candidates for phytoremediation. *Environ Pollut* 239:757-767. <https://doi.org/10.1016/j.envpol.2018.04.105>
201. De Temmerman L, Waegeneers N, Claeys N, Roekens E (2009) Comparison of concentrations of mercury in ambient air to its accumulation by leafy vegetables: An important step in terrestrial food chain analysis. *Environ Poll* 157:1337-1341. <https://doi.org/10.1016/j.envpol.2008.11.035>
202. Egler SG., Rodrigues-Filho S, Villas-Bôas RC, Beinhoff C (2006) Evaluation of mercury pollution in cultivated and wild plants from two small communities of the Tapajós gold mining reserve, Pará State, Brazil. *Sci Total Environ* 368(1):424-433.
203. Svoboda L, Havlíčková B, Kalač P (2006) Contents of cadmium, mercury and lead in edible mushrooms growing in a historical silver-mining area. *Food Chem* 96:580-585. <https://doi.org/10.1016/j.foodchem.2005.03.012>
204. Falandysz J, Mędyk M, Treu R (2018) Bio-concentration potential and associations of heavy metals in *Amanita muscaria* (L.) Lam. from northern regions of Poland. *Environ Sci Pollut R.*, 25(25):25190-25206. <https://doi.org/10.1007/s11356-018-2603-0>
205. Millhollen AG, Gustin MS, Obrist D (2006) Foliar mercury accumulation and exchange for three tree species. *Environ Sci Technol* 40:6001-6006. <https://doi.org/10.1021/es0609194>
206. Asati A, Pichhode M, Nikhil K (2016) Effect of heavy metals on plants: an overview. *Int J Appl Innov Eng Mgmt* 5:56-66.
207. Ahammad SJ, Sumithra S, Senthilkumar P (2018) Mercury uptake and translocation by indigenous plants. *Rasayan J Chem.* 11: 1-12. <http://dx.doi.org/10.7324/RJC.2018.1111726>
208. Shahid M, Khalid S, Bibi I, Bundschuh J, Niazi NK, Dumat C (2020) A critical review of mercury speciation, bioavailability, toxicity and detoxification in soil-plant environment: ecotoxicology and health risk assessment. *Sci Total Environ* 711:134749. <https://doi.org/10.1016/j.scitotenv.2019.134749>
209. Safari F, Akramian M, Salehi-Arjmand H, Khadivi A (2019) Physiological and molecular mechanisms underlying salicylic acid-mitigated mercury toxicity in lemon balm (*Melissa officinalis* L.). *Ecotoxicol Environ Saf* 183:109542. <https://doi.org/10.1016/j.ecoenv.2019.109542>
210. Zhou ZS, Wang SJ, Yang ZM (2008) Biological detection and analysis of mercury toxicity to alfalfa (*Medicago sativa*) plants. *Chemosphere* 70:1500-1509. <https://doi.org/10.1016/j.chemosphere.2007.08.028>
211. Azevedo R, Rodriguez E (2012) Phytotoxicity of mercury in plants: a review. *J Bot ID* 848614. <http://dx.doi.org/10.1155/2012/848614>
212. Patra M, Sharma A (2000) Mercury toxicity in plants. *Bot Rev* 66:379-422. <https://doi.org/10.1007/BF02868923>
213. Moreno FN, Anderson CW, Stewart RB, Robinson BH (2004) Phytoremediation of mercury-contaminated mine tailings by induced plant-mercury accumulation. *Environ Pra* 6:65-175. <https://doi.org/10.1017/S1466046604000274>
214. Obrist D (2007) Atmospheric mercury pollution due to losses of terrestrial carbon pools? *Biogeochemistry* 85:119-123. <https://doi.org/10.1007/s10533-007-9108-0>
215. Lomonte C, Doronila AI, Gregory D, Baker AJ, Kolev SD (2010) Phytotoxicity of biosolids and screening of selected plant species with potential for mercury phytoextraction. *J Hazard Mater* 173:494-501. <https://doi.org/10.1016/j.jhazmat.2009.08.112>
216. Fasani E, Manara A, Martini F, Furini A, DalCorso G (2018) The potential of genetic engineering of plants for the remediation of soils contaminated with heavy metals. *Plant Cell Environ* 41:1201-1232. <https://doi.org/10.1111/pce.12963>

217. Ahmed I, Sebastain A, Prasad MNV, Kirti PB (2019) Emerging Trends in Transgenic Technology for Phytoremediation of Toxic Metals and Metalloids. In: Prasad MNV (ed) *Transgenic Plant Technology for Remediation of Toxic*

Metals and Metalloids. Academic Press, pp 43-62. doi.org.1016/C2017-0-01241-7

218. Kim Y-O, Bae H-J, Cho E, Kang H (2017) Exogenous Glutathione Enhances Mercury Tolerance by Inhibiting Mercury Entry into Plant Cells. *Front. Plant Sci* 8:683. <https://doi.org/10.3389/fpls.2017.00683>
219. Cho UH, Park JO (2000) Mercury-induced oxidative stress in tomato seedlings. *Plant Sci* 156:1-9. [https://doi.org/10.1016/S0168-9452\(00\)00227-2](https://doi.org/10.1016/S0168-9452(00)00227-2)
220. Israr M, Sahi SV (2006) Antioxidative responses to mercury in the cell cultures of *Sesbania drummondii*. *Plant Physiol Bioch* 44:590-595. <https://doi.org/10.1016/j.plaphy.2006.09.021>
221. Zhang T, Lu Q, Su C, Yang Y, Hu D, Xu Q (2017) Mercury induced oxidative stress, DNA damage, and activation of antioxidative system and Hsp70 induction in duckweed (*Lemna minor*). *Ecotoxicol Environ Saf* 143:46-56. <https://doi.org/10.1016/j.ecoenv.2017.04.058>
222. Gómez-Jacinto V, García-Barrera T, Gómez-Ariza JL, Garbayo-Nores I, Vílchez-Lobato C (2015) Elucidation of the defence mechanism in microalgae *Chlorella sorokiniana* under mercury exposure. Identification of Hg-phytochelatins. *Chem-Biol Interact* 238:82-90. <https://doi.org/10.1016/j.cbi.2015.06.013>
223. Wani AB, Chadar H, Wani AH, Singh S, Upadhyay N (2017) Salicylic acid to decrease plant stress. *Environ Chem Lett* 15:101–123. <https://doi.org/10.1007/s10311-016-0584-0>
224. Kováčik J, Rotková G, Bujdoš M, Babula P, Peterková V, Matuš P (2017) Ascorbic acid protects *Coccomyxa subellipsoidea* against metal toxicity through modulation of ROS/NO balance and metal uptake. *J Hazard Mat* 339:200-207. <https://doi.org/10.1016/j.jhazmat.2017.06.035>
225. Zhou X, Gao A, Zhang C, Xu W, Shi X (2017) Exogenous selenium alleviates mercury toxicity by preventing oxidative stress in rice (*Oryza sativa*) seedlings. *Int J Agric Biol* 19:1593-1600.
226. Seneviratne M, Rajakaruna N, Rizwan M, Madawala HMS, Yong Sik Ok YS, Vithanage M (2019) Heavy metal-induced oxidative stress on seed germination and seedling development: a critical review. *Environ Geochem Health* 41:1813–1831. <https://doi.org/10.1007/s10653-017-0005-8>
227. Hasanuzzaman M, Hossain MA, da Silva JAT, Fujita M (2012) Plant Response and Tolerance to Abiotic Oxidative Stress: Antioxidant Defense Is a Key Factor. In: Venkateswarlu B, Shanker A, Shanker C, Maheswari M (eds) *Crop Stress and its Management: Perspectives and Strategies*. Springer, Dordrecht. pp. 261-315. https://doi.org/10.1007/978-94-007-2220-0_
228. Cargnelutti D, Tabaldi LA, Spanevello RM, de Oliveira Jucoski G, Battisti V, Redin M, et al. (2006) Mercury toxicity induces oxidative stress in growing cucumber seedlings. *Chemosphere* 65:999-1006. <https://doi.org/10.1016/j.chemosphere.2006.03.037>
229. Patra M, Bhowmik N, Bandopadhyay B, Sharma A (2004) Comparison of mercury, lead and arsenic with respect to genotoxic effects on plant systems and the development of genetic tolerance. *Environ Exp Bot* 52:199-223. <https://doi.org/10.1016/j.envexpbot.2004.02.009>
230. Nagajyoti PC, Lee KD, Sreekanth TVM (2010) Heavy metals, occurrence and toxicity for plants: a review. *Environ Chem Lett* 8:199-216. <https://doi.org/10.1007/s10311-010-0297-8>
231. Marrugo-Negrete J, Durango-Hernández J, Pinedo-Hernández J, Enamorado-Montes G, Díez S (2016) Mercury uptake and effects on growth in *Jatropha curcas*. *J Environ Sci* 48:120-125. <https://doi.org/10.1016/j.jes.2015.10.036>
232. Teixeira DC, Lacerda LD, Silva-Filho EV (2018) Foliar mercury content from tropical trees and its correlation with physiological parameters in situ. *Environ Pollut* 242:1050-1057. <https://doi.org/10.1016/j.envpol.2018.07.120>
233. Manikandan R, Sahi SV, Venkatachalam P (2015) Impact assessment of mercury accumulation and biochemical and molecular response of *Mentha arve*
234. Mahbub KR, Krishnan K, Naidu R, Andrews S, Megharaj M (2017) Mercury toxicity to terrestrial biota. *Ecol Indic* 74:451-462. <https://doi.org/10.1016/j.atmosenv.2005.07.067>
235. Zhou ZS, Huang SQ, Guo K, Mehta SK, Zhang PC, Zhi MY et al. (2007) Metabolic adaptations to mercury-induced oxidative stress in roots of *Medicago sativa* L. *J Inorg Biochem* 101:1-9. <https://doi.org/10.1016/j.jinorgbio.2006.05.011>

Table 7

Table 7. Some examples of surface soil layer contamination with mercury in regions of important sources of emissions by various authors.

Country	Location	source of pollution	period	Total Hg mg kg ⁻¹	Mean/ Median	author
USA	Alaska	Hg-mining		0.05-5326		Bailey et al. [136]
USA	Texas	Hg-mining		3.8-11		Gray et al. [137]
China	Wuchuan	Hg-mining	2003	0.33 - 320		Qiu et al. [118]
China	Wanshan	Hg-mining	2002	5.1 - 790		Qiu et al. [117]
China	Xiaoqinling	gold mining		0.04-61.2	mean 2.75	Wu et al. [138]
Slovenia	Idrija	Hg-mining	1991-97	1734-2759	mean 2456	Gnamuš et al. [106]
		Hg-mining	2000-01	24- 1055	median 47	Gosar et al. [139]
Slovenia	Podljubelj	Hg-mining	2003-04	0.35-244	median 3.7	Teršič et al. [140]
Slovakia	Rudnany	Hg-mining		9.1-54.3		Banášová [141]
Spain	Almaden	Hg-mining		6-8889	mean 604	Higuera et al. [142]
Spain	Almaden	Hg-mining		1340-4830		Dago et al. [143]
Spain	Caunedo	old Hg-mining		0.09- 13.1 50.0	mean 13.1	Boente et al. [144]
Italy	Vallalta	old Hg-mining		6-21		Wahsha et al. [145]
Turkey	Halköy	Hg mining	2004	0.10-33	mean 5.5	Gemici et al. [146]
China	Tongguan	Artisanal gold mining		0.69-23.7	mean 2.91	Xiao et al. [147]
China	Wanshan	Hg-mining, artisanal gold	2012	0.5-187	mean 31.0	Yin et al. [148]
China	Guizhou	Acetic acid	2016	1.09-3.71		Li et al. [102]
Italy	Mt. Amiata,	Hg mining, volcano-geothermal		2.4-68		Chiarantini et al. [149]
France	Vosges Mountains	Chlor-alkali	2002	0.16-3.99		Hissler and Probst [150]
France	Grenoble	Chlor-alkali		1.3-10		Grangeon et al. [151]
Kazakhstan	Pavlodar	Chlor-alkali		0.93-22.3		Ullrich et al. [152]
Spain	Flix	Chlor-alkali		0.04-12.9	mean 0.77	Esbrí et al. [153]
Germany		Chlor-alkali		0.5-4.2	mean 1.6	Biester et al. [154]
Netherlands		Chlor-alkali	2004	4.3-1150		Bernaus et al. [155]
Portugal	Estarreja	Chlor-alkali		0.010-91 5.4	mean 5.4	Reis et al. [156]
China	Huludao	Chlor-alkali Zn-smelting	2006-08	0.05-14.6		Zheng et al. [157]
China	An Ning	Chlor-alkali		0.09-1.30	mean 0.40	Song et al. [158]
China	Kunming	Chlor-alkali polyvinyl chloride		0.15-4.79		Zhu et al. [159]
Portugal	Caveira	sulfide mine		1.1-76.5		Reis et al. [160]
China	Lianyuan	coal main, steel industry	2015	1.20-3601	mean 178	Liang et al. [161]
China	Zhuzhou	Zn/Pb smelter	2012	0.62-2.61	mean 1.54	Wu et al. [162]
China	Chongqing	thermometer factory		0.06-0.88		Wang et al. [163]
Czech Rep.	Bohemian	non-ferrous metal	2017	6.49		Navrátil et al. [164]
Switzerland	Canton of Valais	Industrial region	2017	0.2-390		Osterwalder et al. [94]
Poland	Warsaw	thermometer factory	2005	122-393	mean 147	Boszke et al. [165]
Pakistan	Karachi/Lahore	highway		61.5-144	mean 90.7	Khan et al. [166]
Germany	Rhine-Westph.	floodplain soil	2017	31.2		Beckers et al. [167]
Italy	Etna	volcano		0.1-0.4		Bonanno et al. [168]
European Union		23000 samples	2009-12	0-1.59	mean 0.04	Tóth et al. [130]