



Review

Exposure Factors in Health Risk Assessment of Heavy Metal(loid)s in Soil and Sediment

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Abstract: Heavy metal(loid)s (HMs) play an important role in economic development since they are used in various branches of industry. However, all industrial activities emit HMs into the environment, where they are no longer useful but potentially toxic. It has been observed that HMs enter the soil and sediment, and potential human health risk may arise due to their excessive accumulation. Having in mind the importance of assessing the risk due to HMs in these media, we analyzed published works in the last decade and created a summary of exposure factors in health risk models for HMs in soil and sediment. This analysis revealed a remarkable increase in the number of publications about health risk assessment of HMs, especially in the last few years. Since many differences in the values of the exposure factors and their distributions were noted, this study focused on elucidating these differences. Non-carcinogenic and carcinogenic health risk assessment models were evaluated through a deterministic approach that is prevalent in use, and a probabilistic one, which is gaining more and more attention in research. In the end, guided by the studied literature, we propose the values and distributions for the exposure factors.

Keywords: potentially toxic elements; heavy metals; Monte Carlo; hazard index; reference dose; cancer slope factor; pollution; dust



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1. Introduction

Heavy metals (Ba, Cd, Co, Cr, Cu, Fe, Hg, Mn, Mo, Ni, Pb, V, and Zn) and metalloids (As, Sb, Ge, and Te) (altogether referred to as HMs) have played a significant role in human society for centuries [1]. They are used in a variety of industrial and commercial applications, including construction, electronics, and transportation [2]. As the world's population grows and economies continue to develop, the demand for HMs has increased over the years. One of the most important uses of HMs is in the production of steel. Steel is a vital material used in the construction of buildings, bridges, and transportation infrastructure. HMs such as Fe, Ni, and Cr are essential components of steel, making them critical in the modern economy. Furthermore, HMs are used in the manufacturing of electronic devices, including smartphones, laptops, and other gadgets. Additionally, the mining and production of heavy metals provide employment opportunities for millions of people worldwide.

Heavy metals occur naturally in rock in the form of various chemical compounds, most often as sulfides and oxides. The most common ores of heavy metals are sulfides of Fe, As, Pb, Pb–Zn, Co, Au, Ag, and Ni and oxides of Al, Mn, Se, Au, and Sb. The sulfides of different metals are commonly present together, as are oxides. For example, ore pyrite (FeS₂) is mixed naturally with Cu, Pb, Cd, As, and Hg sulfides. An elevated content of HMs in soils can occur due to natural processes such as acid drainage. Specifically, sulfur-containing ores in contact with oxygen and water form sulfuric acid, which results in waters enriched with acids and heavy metals [3]. However, mining and smelting activities represent one of the main anthropogenic sources of HMs that lead to the emission of high

concentrations of these harmful substances. HMs deposited in this way not only are a source of soil pollution but also reach other parts of the environment [4,5]. As a result of mining activities, HMs such as Cd, As, Cu, Cr, Hg, Pb, and Zn are released into the environment [6]. They can be carried by water and wind, thus reaching areas far from the source of pollution or remain as tailings [7]. The HM concentrations in soil can be high at a large distance from the main source [8]. Mining activities are especially problematic in developing countries where waste management is not regulated. Therefore, HMs get into the environment much more easily, where unregulated environmental laws represent a significant problem [9]. Hence, the health risk assessment is widely applied in soil mining and smelting areas [10–13].

In addition to being a significant source of nutrients, the soil is also a place of accumulation of HMs, which cause particular concern [14,15]. HMs are present in soils in different forms that vary in mobility, bioavailability, and chemical reactivity [16]. Thus, HMs can be present in exchangeable forms as free ions, as soluble inorganic and organic complexes, bound to carbonates, bound to iron and manganese oxides, bound in solid-state organic matter, or as residual metals [16,17]. Bioavailable concentrations of HMs in the soil are not necessarily proportional to total concentrations [3]. HMs pose a risk to the entire ecosystem and human health because they are extremely toxic [18–21]. Due to the harmful consequences they can cause to humans, plants, and animals, HMs such as As, Cr, Hg, Pb, and Cd attract much public attention [22]. By accumulating in the soil, HMs affect the physiological processes of plants, and, through the food chain, they reach the human body [7,23]. Furthermore, the fertility and functions of the soil are impaired [24]. Different types of soil have different properties; therefore, there are differences in the content of HMs [25,26]. Understanding soil pollution with HMs is very important for determining the safety of the terrestrial environment.

Analogously, sediment pollution results in the accumulation of HMs in biota, which indirectly affects human health through the food chain [27]. The sediment is a place of accumulation of HMs. After reaching the water system, the polluted substances settle and are absorbed into the sediment [28]. By releasing HMs into aquatic ecosystems, the essential characteristics of water are changed, and the functioning of water systems is disrupted [28,29]. As a result, the quality of water is greatly affected [30]. Contaminated sediment affects the aquatic ecosystem, through which it accumulates in the tissues and reaches the food chain, thus potentially endangering human health [31–33]. The determination of HMs is an important indicator of changes in aquatic environments, as it provides important information about the environment [31,34].

In addition to having a harmful effect on the environment, HMs can seriously damage people's health [35]. HMs can accumulate in the human body due to long-term exposure to their influence. In this way, the development of various types of diseases occurs [36]. Accumulation of HMs in organs leads to various acute and chronic diseases, affecting the entire system: immune, nervous, cardiovascular, endocrine, skeletal, etc. [8]. Humans are exposed to HMs in several ways (Figure 1). The first is when they are in direct contact with the soil: ingestion, inhalation, or dermal contact. Another way is indirectly through food grown on contaminated land. Therefore, it is important to determine to what extent agricultural soil is polluted to ensure food safety [37,38]. Soil pollution caused by industry and agriculture ultimately threatens the quality of crops because HMs pose a risk to human health when ingested through contaminated food [39,40]. As in the case of soil, there are three pathways of exposure to sediment: ingestion, inhalation, and dermal contact [34]. Humans can come into contact with sediment during recreation, during work [33,41,42] or as residents [28].

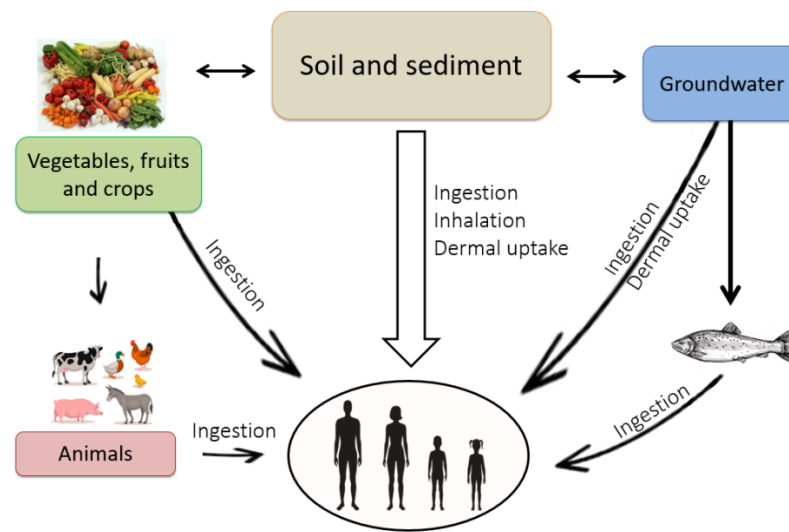


Figure 1. Exposure pathways of HMs from soil and sediment.

Similarly, humans are exposed to HMs present in sediment through the aquatic organisms they consume [43]. HMs include elements necessary for mammals, Cr, Cu, Ni, and Zn, as well as toxic ones at low concentrations, such as As, Cd, Hg, and Pb. On the other hand, even essential HMs can be highly toxic and lead to serious health problems when they enter the human body in huge concentrations [44–46]. HMs can lead to damage of various organs [5], as well as the occurrence of cancer, because the increased risk of developing this disease is correlated with long-term exposure to HMs [47]. When HMs reach the cells, they affect their redox potential and, thus, interfere with the reactions taking place in the cells [45]. Some of the characteristic diseases that occur are as follows: cancer of the respiratory organs due to an increased dose of Cr [48]; As can cause cognitive impairment in children [22] and skin and liver diseases [49]; Cd can lead to osteoporosis, lung cancer, and kidney dysfunction [45]; exposure to high doses of Ni leads to asthma, pulmonary fibrosis, and contact dermatitis [22]; Pb affects brain development in children, the reproductive system [46], as well as damage to the nerves, bones and the immune system [25]; excessive intake of Cu can cause anemia and stomach problems [50].

Figure 2 shows the presence of HMs along with other chemical elements in the Earth's crust. It can be seen that HMs are present in very small quantities. Chemical elements that occur at extremely low concentrations were not included in Figure 2. The list of elements was made according to increasing atomic number in the periodic table. The elements are divided into several groups, depending on their environmental and technological roles.

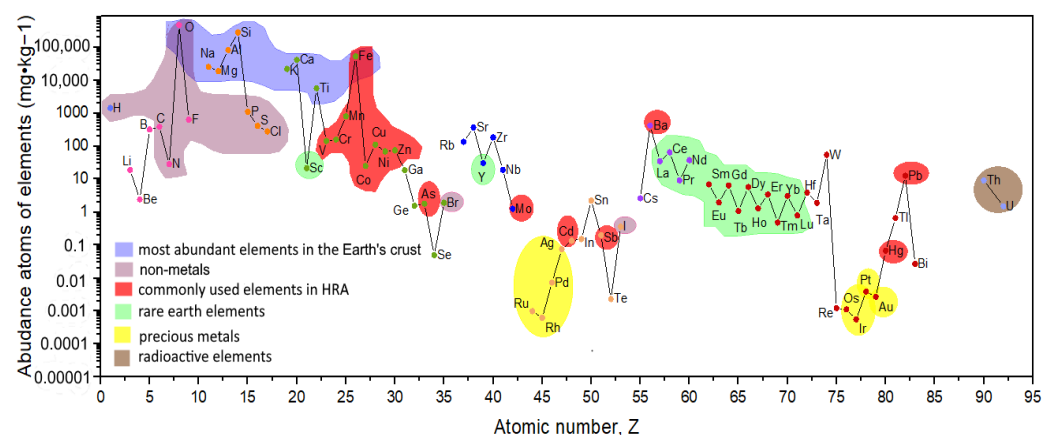


Figure 2. Abundance of elements in the Earth's upper crust.

Certain elements, such as O, Si, Al, Fe, Ca, Na, Mg, K, and Ti, form the basis of soil contexture and are found in large quantities. Other characteristic groups include non-metals (H, C, N, O, F, P, S, Cl, Br, and I), precious metals (Au, Ag, Pt, Pd, Rh, Ru, Os, and Ir), rare earth elements (Sc, Y, La, Ce, Pr, Nd, Pm, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, and Lu), and radioactive elements (Th and U). Moreover, we highlight the elements commonly used in health risk assessment (Cd, Cr, Cu, Ni, Pb, Zn, As, Hg, Co, Mn, V, Fe, Sb, Mo, and Ba) and, thus, their distribution in the earth's crust.

The order in which HMs are represented in the earth's crust is Al > Fe > Ti > Mn > Ba > Sr > V > Cr > Ni > Zn > Cu > Co > Sc > Pb > B > Sn > As > Mo > Sb > Cd > Hg > Se > Bi. Depending on the type of soil and the type of pollution, researchers study different HMs, but the focus is most often on soil pollution by Cd, As, Cr, Cu, Hg, Ni, Pb, and Zn.

Any source of HMs increases their concentration in the environment [48]. Hence, identifying the source of pollution is of great importance because it contributed to the distribution of HMs in the soil [40]. HMs come from different sources and can be present in the environment as a result of natural processes or as a consequence of anthropogenic activities [51–54]. Soil formation, also known as pedogenesis, is the process of soil transformation that prevails in natural soils. On the other hand, in the case of urban areas, anthropological pollution is dominant [35]. This is especially evident in urban areas of the world where the industry is developed [22]. The most important natural source of HMs is the parent material through which they are released [51], as well as volcanic eruptions, river sedimentation [52], erosion, lithogenesis, and weathering [48,55]. As human development progresses, the number of anthropogenic sources of pollution increases. Some of the most common sources are industrial, agricultural (pesticides and fertilizers), and mining activities, construction, traffic emissions (i.e., exhaust gases), fuel and coal burning, and sewage waste [22,39,44,48,56,57] (Figure 3). Moreover, sediment pollution is a major concern because a large number of anthropogenic activities produce large amounts of waste that can potentially enter the sediment [29].

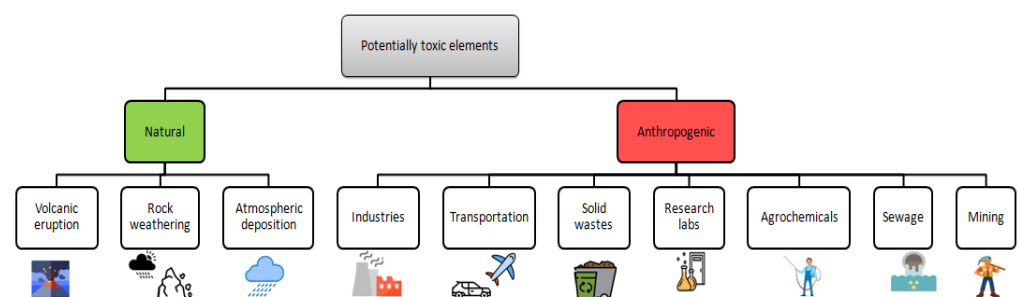


Figure 3. Soil and sediment HM pollution sources.

Determining the sources of HMs is very complex because they may result from different anthropogenic activities and natural processes [52].

To protect human health, it is necessary to assess the health risk of different parts of the environment [37]. Health risk is mostly assessed using the model recommended by the USEPA. The health risk assessment is a very useful and detailed method for determining the carcinogenic and non-carcinogenic risk of various pollutants, including HMs [39,51]. The probability of developing cancer due to contact with HMs is determined on the basis of the total carcinogenic risk (TCR) index, while the hazard index (HI) represents all non-carcinogenic hazards to which people are exposed [58]. This model is adequate for various environmental media, including soil and sediment [36]. In addition, it includes various variables such as gender and age in the calculations; hence, it is possible to make comparisons for different populations.

There are two approaches to health risk assessment: deterministic and probabilistic [5]. The deterministic approach uses the total metal concentration in the calculations, as well as the most likely values of other parameters. Because of this, there is an overestimation

or underestimation of health risk, and a realistic assessment of the risk is uncertain. This approach does not take into account the variation in risk that different people are exposed to. A much more reliable risk assessment result is provided by a probabilistic approach using Monte Carlo simulation (MCS). This approach takes into account uncertainty and variability in parameters, thus giving accurate results [37,59,60]. Furthermore, by using MCS, it can be determined which parameter has the greatest impact on health risk [9].

Due to HMs' harmful properties and their increasing presence in the environment, it is crucial to monitor their presence in soil and sediment. Additionally, proper data processing is necessary to establish the risk they cause. The aim of this study was a detailed and comprehensive presentation of health risk calculations. We selected relevant journals that dealt with sediment or different soils (agricultural, industrial, and urban) and performed a comparative analysis of articles dealing with this topic. The methodology for assessing health risks and parameter values necessary for calculation is evaluated. Deterministic and probabilistic approaches are equally treated and analyzed for both risks (carcinogenic and non-carcinogenic). Since USEPA has issued recommendations and guidelines for health risk assessment, their documents were taken into account. It was necessary to select the right model for health risk assessment of HMs in soil and sediment because of the extent of HM pollution around the world. Particular attention should be paid to soils in the vicinity of mining and smelting locations since these activities represent the main source of HMs.

2. Trends in the Field of Health Risk Assessment of HMs in Soil and Sediment

This study includes an analysis of research trends over time in the field of health risk assessment of toxic metals in soils and sediments. Figure 4 shows the trends in publications and citations of articles from 1996–2022 (a) for health risk assessment in soils and sediments due to toxic elements, and (b) for application of Monte Carlo simulation in risk assessment of HMs. Due to incomplete data for 2023, this year was omitted. Publications and citations were retrieved from the Web of Science (WoS) Core Collection database with the following search strings: ALL = (health risk) AND (ALL = (soil) OR ALL = (sediment)) AND (ALL = (heavy metals) OR ALL = (metalloids) OR ALL = (potentially toxic elements)) for health risk assessment; and ALL = (Monte Carlo) AND (ALL = (soil) OR ALL = (sediment)) AND (ALL = (heavy metals) OR ALL = (metalloids) OR ALL = (potentially toxic elements)) for the application of Monte Carlo simulation in this type of research.

The timeframe from 1996 to 2003 represents the beginning of research on health risk assessment of HMs from soils and sediment (Figure 4a). In this period, the annual number of publications did not exceed 50. This period represents the initial stage of research in this field, when the authors became familiar with the topic. An increase in publications and citations can be observed after 2003, which has continued to this day. Until 2021, there was an almost exponential increase in publications, except for 2011, when a slight decline occurred. The annual number of citations increased stably until 2021 and followed a similar trend to publications. However, the beginning of the year 2023 indicated that interest in this field is growing again, where, at the end of the year, the annual number of publications and readings will most likely be higher compared to 2021 and 2022. In this field, according to Web of Science data, 10,911 articles were published by 25 May 2023.

The literature survey, on the basis of keywords, showed that the authors mostly applied the deterministic approach for the assessment of human health risk associated with HMs in soils [61,62]. Today, the probabilistic approach is also applied; thus, researchers estimate the health risk using both deterministic and probabilistic models [63] or give only a probabilistic model as a better and more advanced approach [37,64,65]. Figure 4b shows the increase in publications and citations of articles that used Monte Carlo simulation (MCS) for a probabilistic approach to health risk assessment from HMs in soil. From 2002 to the present, there was an increase in the use of MCS. The interest of researchers in this topic has not steadily increased for the last 20 years. An occasional decrease in the number of published works can be noticed. A decline in publishing occurred in 2010 and 2011, followed by a sharp spike in 2012, and then another decline in 2013, 2014, and 2015.

However, a sharp increase in the number of published and cited documents occurred in 2021 and 2022, compared to the previous period.

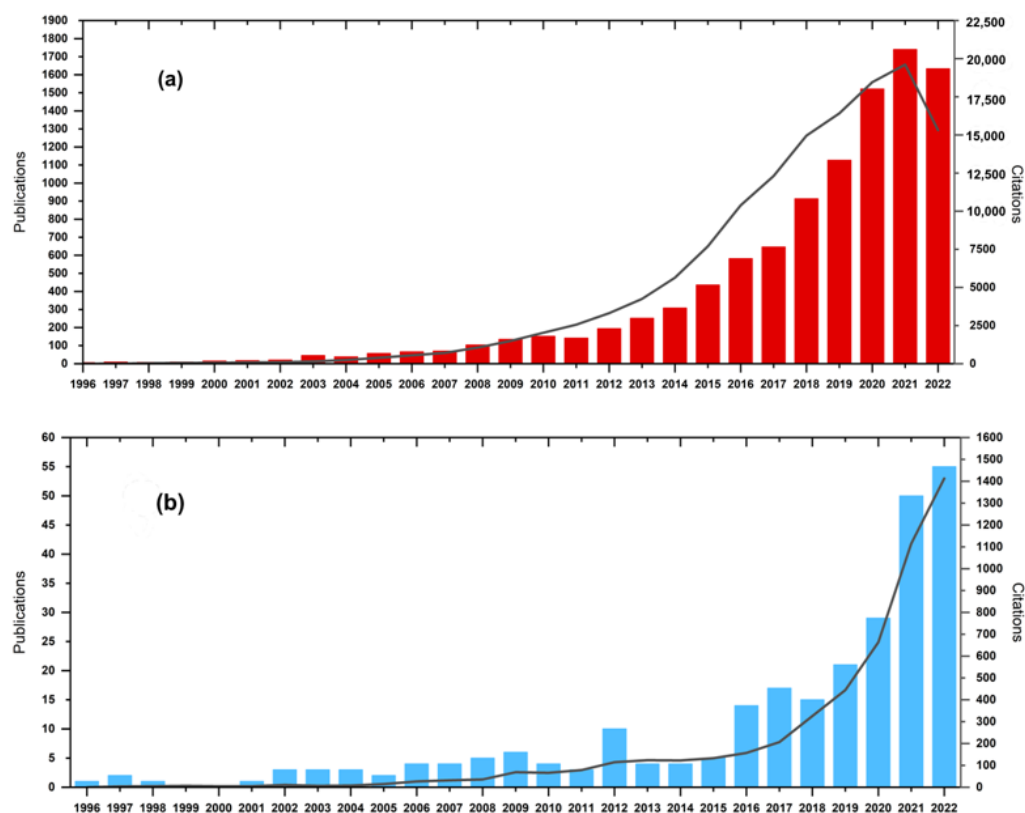


Figure 4. Trends in publication and citation of scientific articles that use health risk assessment (a) and Monte Carlo simulation for probabilistic health risk assessment (b) of HMs in soils and sediment.

Generally, the number of studies and citations has grown in this research field in recent years, using both deterministic and probabilistic approaches. The highest jump in the number of published works was in 2021. The number of citations generally followed the trend of published works. The highest degree of variation was for the number of publications published using MCS in health risk assessment.

3. Health Risk Assessment

Assessment of the health risk of soil or sediment, often referred to as HHRA (human health risk assessment) [66–68], HHR (human health risk) [51,69], or HRA (health risk assessment) [48,70–72], consists of several steps, as shown in Figure 5. The first step is the formulation and identification of hazards that could pose a danger to human health. Because HMs are the primary contaminants in soil/sediment, the health risk assessment is usually estimated for this type of contaminant [73–77]. For each HM, there is a specific intake dose that leads to the appearance of harmful effects. Therefore, it is necessary to define the quantity of HMs that causes health consequences. In addition, the soil/sediment exposure frequency, exposure duration, and exposure pathway should be identified. Three exposure pathways are ingestion, inhalation, and dermal contact [38,78,79]. Ingestion represents swallowing a small amount of soil, i.e., an accidental ingestion. The inhalation exposure pathway addresses the inhalation of small soil/sediment particulates contaminated with HMs, while dermal intake refers to the absorption of HMs through the skin in contact with contaminated soil [80]. The final step of health risk assessment is risk characterization. Humans of every age can be exposed to contaminants in soil/sediment through the exposure routes. In this regard, health risks can be assessed for several exposure age groups, but it is usually generalized for adults and children [81–85]. The term adult in-

cludes both men and women; thus, the same parameter values are used for both. HMs have numerous harmful effects on human health, which can be divided into carcinogenic and non-carcinogenic. Therefore, the health risk assessment is normally based on the estimation of the non-carcinogenic and carcinogenic risks. Risk assessments are not performed exclusively in this way, and some researchers evaluated only non-carcinogenic risk [4,50,86,87]. However, most of the published studies included both non-carcinogenic and carcinogenic risks [10,88–91]. Non-carcinogenic risk is presented through the hazard index (HI), and carcinogenic risk is presented through the total carcinogenic risk (TCR) [8,92]. These indices represent indicators for the development of carcinogenic and non-carcinogenic diseases in humans; thus, their calculation gives better insight into soil/sediment pollution [4,93]. On the basis of the values of these indices, categorization and determination of the degree of danger to human health are carried out [94]. These results can also be used to decide on further treatment for the analyzed soil or sediment if it is shown to be excessively polluted. It should be pointed out that the methods of calculating and assessing health risk differ across articles, which may affect the final conclusions. The parameters that appear in the equations and their values directly affect the assessment of non-carcinogenic and carcinogenic risk, which may lead to different measures that could possibly be implemented to protect human health.

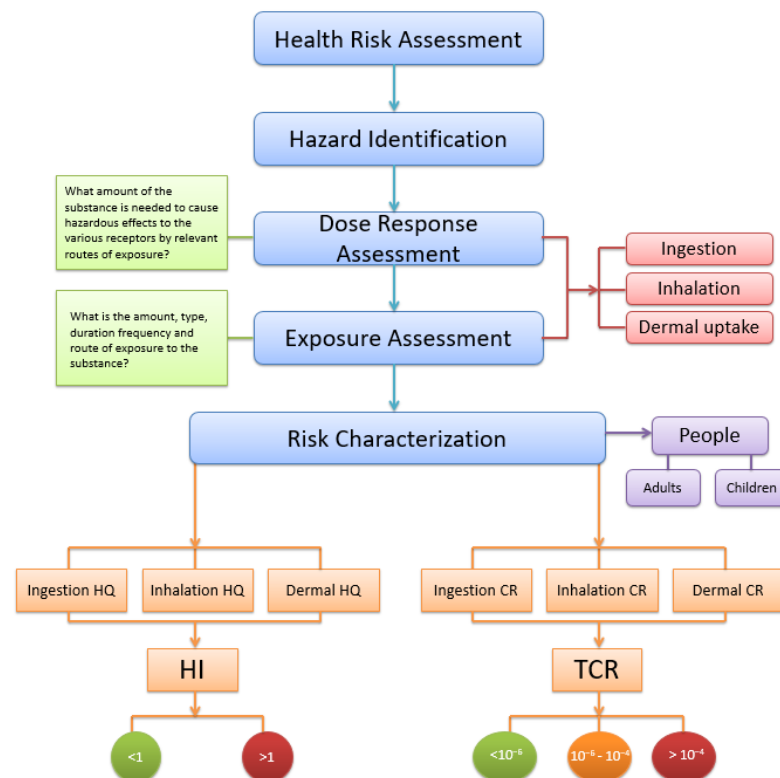


Figure 5. Schematic of HRA procedure of non-carcinogenic and carcinogenic risk. Reference values (<1; >1; <10⁻⁶; 10⁻⁶ – 10⁻⁴; >10⁻⁴) are taken from reference [95].

3.1. Average Daily Dose

In the assessment of carcinogenic and non-carcinogenic risk, the first step is to calculate the daily intake of HMs from the soil/sediment, Equations (1)–(3). It is the basis for further calculations and the final risk assessment. Most researchers take into account all three pathways of exposure (ingestion, inhalation, and dermal contact) [96–100] when assessing the health risk; however, since inhalation has the smallest share of the risk [43,44,59], some researchers based their study only on ingestion [4,59] or ingestion and dermal contact [9,10,44,101–103]. Several terms are used to define the chronic intake of HMs, such as the chronic daily intake (CDI) [47,66,70,76,98,104,105], the average

daily intake (ADI) [36,40,73,93,106], and the most commonly used the average daily dose (ADD) [5,37,52,67,107–109]. Less frequently used indices are D [39,43], daily intake of metals (DIM) [80], and Exp [101]. All the mentioned terms have the same meaning and represent the daily intake of HMs from the soil via the aforementioned pathways. Certain studies separately define the daily intake of metals for carcinogenic and non-carcinogenic, whereby the term ADD is used for the non-carcinogenic risk, while the same formulas with the term LADD (lifetime average daily dose) are used for the carcinogenic risk [18,25,51,108]. There are many variations in the definition and presentation of the average daily intake of HMs, and some researchers completely omit the separate definition of the daily intake equation. In that case, this equation must be integrated with the equations for HI and TCR indices [14,110]. Another term encountered is ADD_{total} , which represents the sum of all three ADD for different routes of exposure [102,107]. The term “ing” has been frequently employed to refer to the ingestion pathway [50,86,88,89,111,112], while other terms include “oral” [9,47,93], “uptake” [49], “ingestion” [58,73,113], “ingest” [19,70,80], and “soil ingestion” [91]. The most typical inhalation term is “inh” [8,46,51,68], but other terms such as “inhale” [70,74,80] and “inhalation” [22,49] are additionally used. “Der” [52,68,69], “skin” [49], and “dermal contact” [73,91] are terms used for skin contact, whereas “derm” [8,35,98,114,115] and “dermal” [76,104,111,116] are the most widespread. The average daily dose is calculated according to Equations (1)–(3):

$$ADD_{ing} = \frac{C \times IngR \times EF \times ED}{BW \times AT} \times CF, \quad (1)$$

$$ADD_{inh} = \frac{C \times InhR \times EF \times ED}{PEF \times BW \times AT}, \quad (2)$$

$$ADD_{derm} = \frac{C \times SA \times AF \times ABS \times EF \times ED}{BW \times AT} \times CF. \quad (3)$$

The unit for the average daily dose for all three exposure pathways is $mg \cdot kg^{-1} \cdot day^{-1}$ [85,117], or abbreviated $mg \cdot kg^{-1} \cdot d^{-1}$ [38]. This was confirmed by dimensional analysis. However, some studies distinguish special units for ADD inhalation, wherein they use $mg \cdot m^{-3}$ [118]. In addition to the abovementioned equations, some researchers give equations with certain variations (with or without additional parameters). For example, a conversion factor in the case of inhalation [5,24,37,45] is sometimes used, while it is not present in the equations for ingestion and/or dermal contact [4,49,119]. Formulas that deviate from the usual working principle and contain additional parameters can also be found [91,109,112]. Generally, the health risk is assessed separately for adults and children, although there are scientific articles on the calculation for an alternative population grouping (e.g., children, adult men, and adult women [52,105,118], children, adults, and seniors [120], or limited only to the child population [4,86] or adults [10,46,74,110]). The differences in exposure factors are summarized in Table S1.

3.2. Non-Carcinogenic Risk

The calculation of non-carcinogenic risk represents a determination of the impact of HMs from soil and sediment on non-carcinogenic effects in humans. The hazard quotient (HQ) and hazard index (HI) are used for the purpose. HQ_i represents the ratio of the average daily dose and reference dose for one HM for all exposure routes, as shown by Equations (4)–(6).

$$HQ_{i,ing} = \frac{ADD_{i,ing}}{RfD_{i,ing}}, \quad (4)$$

$$HQ_{i,inh} = \frac{ADD_{i,inh}}{RfD_{i,inh}}, \quad (5)$$

$$HQ_{i,derm} = \frac{ADD_{i,derm}}{RfD_{i,derm}}, \quad (6)$$

$$HQ_i = HQ_{i,ing} + HQ_{i,inh} + HQ_{i,derm}. \quad (7)$$

After that, all the HQ_i values for each HMs in a given sample are summed to get HI, which represents the sum of the hazard quotient for each determined HM, through all exposure routes in an analyzed sample:

$$HI = \sum_{i=1}^n HQ_i, \quad (8)$$

where n is the number of HMs.

The majority of researchers use the term HI and HQ [48,67,98,112], but HI, HQ, and HI_t [91], THI and HQ [13,80,109], HQ and HQ_t [10], HQ, THQ, and HI [77,89], HQ only [5,66,105], HQ, HI, and THI [69,118], HI only [40], HQ and NCR [102], and HQ, HI, and CHI [22] have also been used. In all cases, the equations are the same but with different notation. However, the abbreviation THI is also encountered, mainly in the source-specific HRA, in order to distinguish HI for individual sources using THI, which represents the sum of the hazard quotient for each metal, for all three exposures, and for all sources of pollution in one sample.

The HI value shows if there could be potential non-carcinogenic effects on health due to exposure to contaminated soil and sediment. The most frequent and widely applied categorization of pollution is to the value one. If HI, or HQ, is greater than one ($HI > 1$), there is a certain doubt that HMs may have a harmful effect on health, whereas, if this value is less than one ($HI < 1$), the impact of heavy metals is insignificant [88,99]. In addition to this method of categorization, there is another that divides HI into several categories [50,66,112]. Some researchers stated only a partial categorization [19,92,105], while others did not mention these non-carcinogenic risk categories [14,69]. HQ and HI are unitless because ADD and RfD have the same unit ($mg \cdot kg^{-1} \cdot day^{-1}$).

3.3. Carcinogenic Risk

Carcinogenic risk measures the impact of HMs from soil and sediment on the carcinogenic effect on humans. For this purpose, two parameters are used: carcinogenic risk (CR) and total carcinogenic risk (TCR). The CR_i is calculated according to Equations (9)–(11) by multiplying the average daily dose and the cancer slope factor for one HM for all exposure routes:

$$CR_{i,ing} = CDI_{i,ing} \times CSF_{i,ing}, \quad (9)$$

$$CR_{i,inh} = CDI_{i,inh} \times CSF_{i,inh}, \quad (10)$$

$$CR_{i,derm} = CDI_{i,derm} \times CSF_{i,derm}, \quad (11)$$

$$CR_i = CR_{i,ing} + CR_{i,inh} + CR_{i,derm}. \quad (12)$$

All the CR_i values for each HM in a given sample are summed to get TCR, which represents the sum of the carcinogenic risk for each analyzed HM, through all three exposure pathways in an analyzed sample:

$$TCR = \sum_{i=1}^n CR_i, \quad (13)$$

where n is the number of HMs.

The TCR value provides insight into the potential carcinogenic effects on human health due to the HMs contamination of soil or sediment. The acceptable or tolerable carcinogenic risks are in the range of 1×10^{-6} to 1×10^{-4} . If TCR or CR is less than 10^{-6} , there is no risk of developing carcinogenic diseases, if this value is greater than 10^{-4} , the risk is unacceptable [121]. Apart from this categorization, some more specific ones also appear [9,92,112], but it also happens that the researchers give only a partial categorization [51,98,119] or do not give it at all [69]. CR and TCR values are unitless, as is non-carcinogenic risk. The vast majority of researchers use the terms CR and TCR [13,24,109,113]. To a lesser extent, only CR is used [18,68,100], although there are also the following terms: risk [66,88,98]; CR and CRt [91]; CR and LCR [8,80]; TCR only [52]; CR and TR [46]; ILCR and ILCRt [10]; RI and ILCR [48]; (RI) risk [96]; RI [107]; R [108]; cancer risk and LCR [70,89]; CR, risk, and total carcinogenic risk [106]; CR, TCR, and CTCR [22].

4. Exposure Factors

4.1. Ingestion Rate (IngR)

The ingestion rate is the amount of soil or sediment an individual ingests during a specific period, usually 1 day, expressed in units $\text{mg}\cdot\text{day}^{-1}$ [22,77,122] or abbreviated $\text{mg}\cdot\text{d}^{-1}$ [24,100]. The ingestion rate for children is always higher than for adults because children are more in contact with the soil or sediment. Therefore, there is a greater possibility of ingesting a certain amount of polluted matter. However, some authors used the same ingestion rates for children and adults [120]. This parameter is used only in the equation for ingestion. The most common abbreviation is IngR [39,71,87], but others are also used: IR_{ing} [37,93], IR [53,88], IR_1 [100], R_{Ing} [8,97], IR_s [9,109,112], R_{ingest} [67], SIR [4], IR_O [59], and IR_{soil} [104]. The most common name of this parameter is the ingestion rate [46,111], but authors also used variations of this name, such as ingestion rate of soil [44,53,68], soil ingestion rate [4,73], ingestion rate in soil [91], frequency of manual–oral intake [123], rate of ingestion [8], soil dust intake rate [49], daily soil intake [10], and ingestion ration [51]. Since most authors accepted that the inhabitants of the investigated land are residential ($\text{EF} = 350$), the ingestion values of 200 and 100 were used for children and adults, respectively [13,44,48,113]. Oral intake was also lower in some studies; ingestion rates in these studies were 100 [105], 60 [69], 50 [25,109], 30 [112], and 20 [49] for children, and 50 [105,120], 30 [69], 20 [25,109], and 10 [112] for adults.

4.2. Inhalation Rate (InhR)

The inhalation rate is the volume of air (in m^3) containing soil/sediment particulates that a person inhales during 1 day ($\text{m}^3\cdot\text{day}^{-1}$ [18,77,92] or $\text{m}^3\cdot\text{d}^{-1}$ [24,80,86]). It is used only in the equation for inhalation, and the most common abbreviation is InhR [18,51,105], but the following are also used: IR_{inh} [35,93], IR_2 [100], IR_i [36,109], R_{inh} [52,97], R_{inhal} [67], IR_a [112], and IR_{air} [88,120]. The values for children ranged between 7.3 and 16.57, where the most often mentioned were 7.5 [47,51], 7.63 [68,113], 7.6 [39,107], and only occasionally 7.3 [38], 8.1 [112], 9.3 [87], 10 [73], 7.65 [100], and 16.57 [118]. For adults, that figure was slightly higher, and values ranged from 12.8 to 20; the most often used was the value 20 [13,18,98]. In a few cases, the following values were also used: 15.7 [46], 15.6 [112], 16.57 [52], 16.1 [67], 16 [25], 14.7 [35], 12.8 [68], 15 [73], and 14.5 [51]. Among the authors, different values for the InhR parameter can be found. A value of 7.5 was used the most for children in combination with value 14.5 or 15 for adults. On the other hand, the most used value for adults was 20, but in combination with 7.6 for children [39,58,98]. Some authors [52,70,118] used the same values of InhR for children and adults, while others separated these two groups. Although the term “inhalation rate” is the most frequently utilized [46,107,111], the name of this parameter is still being considered. Scientific researchers also use phrases such as the inhalation rate of soil [68,113], soil inhalation rate [122], respiratory intake frequency [123], air inhalation rate [50], and respiratory inhalation volume [49].

4.3. Exposure Frequency (EF)

The frequency of exposure to soil or sediment is known as exposure frequency, and it is determined as the number of days per year that a person comes in contact with the polluted matter. Different variations of the same unit are used: $\text{day}\cdot\text{year}^{-1}$ [5,77,103], $\text{days}\cdot\text{year}^{-1}$ [44,48,70], $\text{d}\cdot\text{a}^{-1}$ [8,24], $\text{d}\cdot\text{year}^{-1}$ [43], $\text{days}\cdot\text{y}^{-1}$ [109], $\text{day}\cdot\text{a}^{-1}$ [51], $\text{d}\cdot\text{yr}^{-1}$ [86,107], $\text{day}\cdot\text{yr}^{-1}$ [18], and $\text{d}\cdot\text{years}^{-1}$ [80]. This parameter is marked as EF [37,40,45], but the abbreviations TEF [46,69], $E_{\text{frequency}}$ [48], and F_{exp} [97] are also encountered. The same value is used for children and adults since it is assumed that both groups spend the same amount of time per year in contact with soil/sediment. Accordingly, most researchers used the value 350 [25,58,73], and very few deviated from this value. The majority of authors took a number smaller than 365, taking into account that people are not necessarily in contact with the land every day. However, some authors analyzed more cases, such as residential, industrial, recreational, agricultural, and forest land, and used values of 262.5 [69], 345 [52,118], 312 [53], 180 [87,98], and 365 [113,115]; most classified soils as residential (housing) and, hence, used values of 350.

4.4. Exposure Duration (ED)

Exposure duration represents the period of exposure to soil or sediment exposure expressed in years. All reviewed studies used the term exposure duration, except for Bernardo et al. (2022) [98], where this parameter was referred to as the exposure period. It is abbreviated as ED [9,37,40], although T_{exp} [97] also appears. The unit appears both in the singular and in the plural, such as year [5,35,103], years [9,87], a [24,100], and yr [38]. For children, all authors agreed that the number 6 should be used as the average age of children [18,70,109], while, for adults, different values between 20 and 35 years appeared. The most used number was 24 [5,18,46,87,92], followed by 30 [73,113], while 20 [112], 26 [14], and 35 [53] were used far less.

4.5. Particulate Emission Factor (PEF)

The particulate emission factor is used only in the inhalation formula and represents the number of particulates expressed in m^3 emitted from 1 kg of soil/sediment. It is marked everywhere as PEF [80,99,120], except for one scientific article (Alsafran et al., 2021) as EF_p [58]. All authors used the same value of 1.36×10^9 , which was the same for children and adults [39,52,58,67]. The only exceptions are Xue et al. (2023), Rehman et al. (2018), and Chen et al. (2022) [13,48,123], with a value of 1.32×10^9 . The parameter's name varies, with emission factor [111,115], dust emission factor [114], particulate emission factor [98,112], and inhalation factor for emission particulates [45] in use, but particulate emission factor is the most prevalent [70,73,87,113]. This factor has the least variation in units and used values.

4.6. Skin Surface Area (SA)

This parameter varies the most concerning its name. There are several names for it, such as skin surface area [48,112], skin surface area available for contact [53], surface area of the skin that contacts the PTEs [22], exposed skin surface area [68,115], surface area of skin [36,45], exposed skin area [74,98], surface area of the exposed skin [111], exposed area through dermal contact [114], skin area exposed to soil contact [73], skin surface area contact [91], skin area available for soil contact [25], surface area [118], or just skin area [50]. SA represents the surface area of the skin (in cm^2 [99]) that comes into contact with the soil/sediment. Furthermore, $\text{cm}^2\cdot\text{day}^{-1}$ [105,120] and $\text{cm}^2\cdot\text{d}^{-1}$ [49,118] are found in the literature. This parameter is used only in the formula for dermal contact [18,38,87]. It is abbreviated most often as SA [99,102,108], but ESA [113], ESA_s [58], and A_{skin} [97] also occur. Among parameters in ADD formulas, SA has the most diverse values. Children naturally have a smaller contact surface, and their values range from 899 to 2800. The following values were used in scientific articles: 2373 [53,112], 2800 [24,39,80,100,103], 1600 [68,113], 2448 [51,71], 899 [123], and 2848.01 [50]. For adults, this value was higher and ranged

from 1701 to 6032, where values 5800 [38,80], 5373.99 [50], 1701 [123], 5075 [51,71,122], 4350 [10,25,43], 5700 [18,100,114], and 6032 [91] appear. The most common values were 5700 and 2800 for adults and children, respectively. However, those two values were not always combined together.

4.7. Adherence Factor (AF)

The adherence factor represents the number of HMs that adhere to the skin, and it figures in the formula for dermal contact. In most cases, it is designated as AF [44,45,104], but the following abbreviations are also present: SL [5,13,46,86,87]; SAF [93,98,113]; AF_s [58], and AFsoil [70]. In most cases, researchers used the USEPA recommended values, 0.2 for children and 0.07 for adults [58,69,73]. There were also articles showing other values of 0.65 [52,118], 0.07 [70], 1×10^{-6} [40], and 2×10^{-6} [109] for children, and 0.22 [10], 1 [48], 0.49 [52], 2×10^{-7} [40], 0.2 [35], 0.3745 [105], and 0.7 [45,107] for adults. An analysis of published articles revealed variations in names for the AF parameter. The most used are the adherence factor [103,105,115] and skin adherence factor [43,46,86]. Other names are adherence factors of soil to skin [48], soil-to-skin adherence factor [53], soil adherence factor [112], adhesion coefficient of skin [123], adherence factor to skin [118], skin adhesion [49], and skin adherence factor for soil [44]. This parameter has the highest variability in units: $\text{mg}\cdot\text{cm}^{-2}\cdot\text{day}^{-1}$ [87,92], $\text{mg}\cdot\text{cm}^{-2}$ [13,70,113], $\text{mg}\cdot\text{cm}^{-2}\cdot\text{h}^{-1}$ [39], $\text{mg}\cdot\text{cm}^{-2}\cdot\text{event}^{-1}$ [91], $\text{mg}\cdot\text{cm}^{-1}\cdot\text{d}^{-1}$ [100], $\text{kg}\cdot\text{cm}^{-2}\cdot\text{day}^{-1}$ [109], $\text{mg}\cdot\text{cm}^{-1}\cdot\text{d}^{-1}$ [100], $\text{mg}\cdot\text{cm}^{-1}\cdot\text{day}^{-1}$ [45], $\text{mg}\cdot\text{cm}^2\cdot\text{day}^{-1}$ [49], and $\text{kg}\cdot\text{cm}^{-1}\cdot\text{day}^{-1}$ [73]. According to the USEPA, the unit of the AF factor should be $\text{mg}\cdot\text{cm}^{-2}\cdot\text{event}^{-1}$, i.e., when calculating the soil/sediment risk, $\text{mg}\cdot\text{cm}^{-2}\cdot\text{day}^{-1}$ [95]. Dimensional analysis of the ADD equation indicates that, if AF's unit is $\text{mg}\cdot\text{cm}^{-2}\cdot\text{day}^{-1}$, ADD is in $\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$.

4.8. Dermal Absorption Factor (ABS)

The dermal absorption factor [24,51,98,111], absorption factor [124], contact factor [73], skin absorption factor [109], skin absorbance [80], dermal absorption fraction [8], or skin adsorption coefficient [49] is the factor that only figures in the formula for dermal contact. It is a dimensionless parameter, denoted as unitless [9,48,77,120], nondimensional [107], dimensionless [100], none [73], or simply by leaving a space for a unit [58,71]. The name dermal absorption factor prevails, with the standard acronym ABS [70,102,118]. Abbreviations ABF [89,96,107], DA [98], and DAF [97,113] also appear in the literature. Regarding the value of this factor for children and adults, researchers used it in two ways. In the first case, the value of 0.001 was the same for children and adults, for both carcinogenic and non-carcinogenic calculations, and for all metals [80,87,92,105,109]. In the second case, the value of this parameter was the same for children and adults, for carcinogenic and non-carcinogenic risk, but the value changed depending on the type of metal: (a) for As, this factor had a value of 0.03, but 0.001 for all other metals [8,18,58]; (b) Pb, Cd, Cu, Zn, Hg, and As had different values, but this factor was 0.001 for all other metals [50]; (c) a different value [38,112] was used for each metal. However, despite these variations, case number one is the most common in the literature.

4.9. Body Weight (BW)

As a significant parameter, the average body weight figures in the ADD equation because the impact of metal concentrations on a person's health depends significantly on the individual body weight. Average body weight is expressed in kg and denoted as BW [25,46,107], but acronyms EBW [122], ABW [80,97], and BWA [58] also occur. This value represents the average weight for adults and children, defined by the ED parameter, most often in a certain period of life. There are various names such as body weight [22,53,73], bodyweight of the exposed individual [68,111], average body weight [4,24,98], exposure body weight [122], weight [115], middleweight [123], and recipient weight [49]. The most established weight for children was 15 kg [13,68,92], with 70 kg used for adults [22,80,113]. For this parameter, the greatest variations in numerical values occur, with values of 15.9 [44,58],

16 [24], 22 [35], 16.2 [47], 20 [113], 19.6 [115], 24.7 [109], 29.3 [67], 22.5 [87], 18.6 [105], 19.2 [4], and 29 [49] used for children, and values of 80 [112], 55.9 [13,107], 56.9 [45], 60 [14], 68.4 [18], 61.8 [50], 57 [109], 62.5 [24], 56.8 [44,58], 60.6 [46], 62.57 [67], 63 [36], 62 [74], and 59 [10] for adults. Different authors used different weights, mostly because people do not have the same average body weight in different parts of the world; thus, these numbers were adapted to a specific area.

4.10. Average Time (AT)

The average time [22], averaging time [70,86], Av. Time [53], mean time [50], average time of exposure to contaminated soils [67], average exposure time [109], average exposure time per year [35], mean total exposure time [123], average time non-carcinogenic/carcinogenic [73], averaged time of non-carcinogenic impact/carcinogenic impact [115] or non-carcinogenic average time/carcinogenic average time [122] represents the average time an individual spent in contact with soil/sediment, expressed in days. The unit for AT appears in different forms as day [5,25,59], days [4,58,71], d [24,86,107], or D [38]. As in the case of EF and ED, units occur both in the singular and in the plural. In a few articles, the unit of this parameter was not emphasized [46,69], while the unit year or day was also present [91]. The value of this parameter is different for carcinogenic and non-carcinogenic risks. For non-carcinogenic risk, AT is obtained by multiplying the value of ED with the number of days in a year, i.e., 365. For carcinogenic risk, the average life expectancy (LT) is multiplied by the number of days in a year [5,13,24,58]. Depending on the area of residence, researchers used different LT values, although the most established number of years taken for the average human lifespan was 70. Other values used were 76.6 [36], 77 [5], and 69.5 [74]. For both carcinogenic and non-carcinogenic risks, this parameter is denoted as AT [80,105,118], although some authors separated it into AT_{nc} and AT_c [22,53], AT_{nc} and AT_{ca} [9,115], ET_A and ET_{ca} [58], or AT and LT [112]. Some publications used the same values for both non-carcinogenic and carcinogenic risks [40,51,80].

4.11. Conversion Factor (CF)

A conversion factor is used to standardize the units in the formulas so that ADD is expressed in $mg \cdot kg^{-1} \cdot day^{-1}$. The value of this factor is the same everywhere and is $10^{-6} kg \cdot mg^{-1}$ [13,14,91]. There are cases where the unit is $mg \cdot day^{-1}$ [53], $mg \cdot kg^{-1}$ [48,73], or even unitless [74,120]. Most authors did not mark this factor as a separate parameter; thus, they did not explain it, but it was already found as a numerical value in the formula [107,113,122]. It is often denoted as CF [40,70,108], although $F_{conversion}$ [48] and FC [80] are also used. The name of this parameter varies, e.g., conversion factor [9,13,73], which is the most common, as well as conversion coefficient [115], factor for conversion [48], factor [80], average conversion factor [14], units conversion factor [59], and unit conversion [123] appear. A review of the articles revealed that, among the authors, there was doubt about using conversion factors in the formulas. For example, it occasionally occurred in all three cases [5,24,37]. After dimensional analysis, it was determined that this factor is necessary for the equations for ingestion and dermal contact, while it should not be included in ADD_{inh} .

4.12. Lifetime (LT)

Lifetime represents the lifespan of a person, and the value of 70 years is most often taken [48,105,112]. It is rarely singled out as a separate parameter; hence, many authors did not emphasize it [46,58,100], but designated it as lifetime [44,109]. It is used to define AT carcinogenic to assess the carcinogenic risk, and the same number is taken for both children and adults. In a small number of cases, lifetime was not taken as 70 years [18,44,109].

4.13. Reference Dose (RfD)

The reference dose (RfD) represents an estimate of the daily exposure of HMs that does not have a harmful effect on human health during a lifetime. Each HM has a different

RfD value. RfD is necessary for calculating non-carcinogenic risk and is expressed in the same units as ADD, i.e., in $\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$ [4,25,48], although there are studies with units in $\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$ [5,100,107]. Some researchers differentiated the units for inhalation ADD and, consequently, for inhalation RfD, i.e., $\text{mg}\cdot\text{m}^{-3}$ [52]. As in the ADD formulas, there are separate RfD values for all three exposure pathways: ingestion, inhalation, and dermal contact. In the literature, RfD values have been compared for all three exposure pathways for 15 (Cd, Cr, Cu, Ni, Pb, Zn, Hg, As, Mn, Co, V, Fe, Mo, Ba, and Sb) different HMs. For most HMs, many different RfD values were found. Therefore, the most frequent values, which appear in the largest number of studies, were usually chosen. RfD values for ingestion for most HMs can be found on the USEPA IRIS website [95]. The HMs Cd, As, Cr, Cu, Ni, Pb, and Zn are the most frequently studied in the literature [18,40,43]. A smaller number of studies included Hg, Mn, and Co [46,73,115] in their investigation, while published data related to V, Mo, Ba, Fe and Sb can hardly be found [8,13,48,58]. The RfD data were provided in the main article text or supplementary material (Tables S1–S3), but these values were not given in several published articles. Table S2 contains a summary of the different RfD values for each element.

The largest difference in RfD values for all three exposure pathways occurred in the case of Cd. By far, the most common value was 1×10^{-3} [71,92,109] for ingestion and 1×10^{-5} [5,24,100] for dermal contact, while, for inhalation, the researchers generally took either 1×10^{-3} [48,87] or 1×10^{-5} [52,73]. These two values are used almost equally in the literature, although RfD 1×10^{-5} slightly prevails. In addition to these values, researchers mentioned 1×10^{-4} [24], 3×10^{-1} [73], 3×10^{-3} [52], and 5×10^{-4} [86] for ingestion, 1×10^{-4} [24], 5.71×10^{-5} [109], 5.7×10^{-5} [58], 2.86×10^{-6} [112], and 2.86×10^{-5} [45] for inhalation, and 2.3×10^{-5} [73], 2.5×10^{-5} [109], 5×10^{-4} [77], 5×10^{-5} [48], and 6×10^{-5} [45] for dermal contact.

For Cr, characteristic values that are widely accepted and that coincided among most researchers constitute 3×10^{-3} [18,44,70], 2.86×10^{-5} [5,46,92], and 6×10^{-5} [24,40,51] for ingestion, inhalation, and dermal contact, respectively. Very rarely, researchers did not adhere to these parameters but used different ones, such as 5×10^{-3} [39] for ingestion, 2.9×10^{-5} [114], 1.43×10^{-3} [112], and 3×10^{-5} [86] for inhalation, and 3×10^{-3} [113], 2.5×10^{-4} [50], 1.95×10^{-2} [112], and 5×10^{-5} [92] for dermal contact.

In the case of Cu, almost every source used identical values, i.e., 4×10^{-2} , 4.02×10^{-2} , and 1.2×10^{-2} [24,35,122], respectively, for the three exposure pathways. Specifically for the case of inhalation, in addition to 4.02×10^{-2} , the value 4×10^{-2} [100,113] appears; thus, it can be said that both values are similar, given that the numerical difference is very small. For ingestion, values also included 3.71×10^{-2} [8], 4.2×10^{-2} [10], 1×10^{-2} [112], and 3.7×10^{-2} [39]. For inhalation, 1.43×10^{-2} [49] and 4.02×10^{-3} [38] were used, while, in the case of dermal contact, we can find 4×10^{-2} [91], 1.9×10^{-3} [48], and 5.7×10^{-3} [112].

Ni is often encountered in published studies, and it has characteristic values of RfD parameters that are widely accepted. These include 2×10^{-2} [25,58,109], 2.06×10^{-2} [13,46,113], and 5.4×10^{-3} [44,51]. As in the case of other metals, some atypical values appear sporadically, such as 8×10^{-4} [8], 1.1×10^{-2} [112], and 2×10^{-1} [13] for ingestion, 9×10^{-5} [118], 2.6×10^{-2} [109], 2.1×10^{-2} [50], 2.57×10^{-5} [112], and 2.01×10^{-2} [51] for inhalation, and 8×10^{-4} [109], 1×10^{-3} [48], and 4.4×10^{-4} [112] for dermal contact.

In the case of Pb, the largest variations of RfD values occurred for dermal contact, while only two additional values appear in the case of the other two routes of exposure. Thus, researchers almost completely agreed on the RfD values for Pb, i.e., 3.5×10^{-3} [5,25,87], 3.52×10^{-3} [18,113], and 5.25×10^{-4} [73,100]. For ingestion, 1.4×10^{-3} [58] and 3.6×10^{-3} [86] were also encountered, while, for inhalation, 3.5×10^{-3} [114] and 3.25×10^{-3} [24] were used. Several possible RfD values have been offered for dermal contact, but very few references support their relevance. These include 5.2×10^{-4} [114], 5.25×10^{-3} [115], 5.3×10^{-4} [50], 5.24×10^{-4} [58], 3.6×10^{-4} [112], and 3.52×10^{-2} [13]. Unfortunately, the USEPA IRIS recommends no RfD value for Pb.

There is no dilemma about the RfD value for Zn. It is one of the few HMs for which the RfD values are clearly defined. For ingestion and inhalation, there were single values of 3×10^{-1} [46,109]. On the other hand, for dermal contact, there were a few variations (3×10^{-1} [91], 3×10^{-2} [112], and 6×10^{-1} [38]), although 6×10^{-2} [44,87,100] was by far the most used value.

Along with Cd, As had the largest variations in RfD value for inhalation and dermal contact. However, regarding ingestion, 3×10^{-4} [8,103] was a number present everywhere, except for one study by Battsengel et al. (2020) [73], where 1×10^{-4} was used. It is interesting that, in the case of inhalation, three numbers, 3×10^{-4} [24,77], 1.23×10^{-4} [25,58], and 3.01×10^{-4} [46], appear, which were almost equally used. In addition, some researchers used 4.29×10^{-4} [112] and 1.5×10^{-5} [74]. Concerning dermal RfD, the most used value was 1.23×10^{-4} [18,43,118], while 3×10^{-4} [13] is was to a lesser extent. Far less used values were 1.24×10^{-4} [35], 2.85×10^{-4} [112], and 8.6×10^{-6} [45].

Compared to the metals mentioned above, Hg, Co, and Mn have been analyzed and discussed in the literature to a lesser extent. Although Hg is a highly toxic HM, many researchers did not consider mercury in their risk assessment. On the basis of the available data, it can be concluded that only 3×10^{-4} [5,36,51] should be taken as the RfD value of Hg. The USEPA IRIS also emphasizes this value, but only in the form of mercury chloride [95]. The RfD values for inhalation are the most problematic because they appear almost equally in the literature as 8.57×10^{-5} [5,46], 3×10^{-4} [24,40], and 8.6×10^{-5} [86,91]. Moreover, 2.4×10^{-5} [51] was also recorded in one article. The most variation exists in dermal contact, where the most common value was 2.1×10^{-5} [44,118] and, to a lesser extent, 3×10^{-4} [38,77]. Other numbers that occurred are 4.3×10^{-3} [91], 2.4×10^{-5} [10], and 2×10^{-5} [40].

There were no significant variations regarding Co RfD values for all three exposure pathways. For ingestion, 2×10^{-2} [48,115] was used, except by Chu et al. (2022) [4], where 3×10^{-4} was used in the calculation. This value is also found in EPA reports but was not widely used among researchers. The inhalation reference dose is clearly defined and amounts to 5.71×10^{-6} [18,46,92]. For the dermal RfD, the situation was similar to that of ingestion, and the most common RfD in use was 1.6×10^{-2} [43,48], with 1.6×10^{-5} [115] appearing in only one article.

The reference dose of Mn for ingestion is the most problematic for the reason that there are five possibilities; few researchers mentioned this metal in their research, and almost all offered values different from that recommended by the USEPA IRIS, which is 1.4×10^{-1} or 2.4×10^{-2} [95]. Among the few studies, 4.6×10^{-2} [18,115,122] stands out as the current value, along with 4.7×10^{-2} [92] and 4.66×10^{-2} [48], as well as one article using the USEPA recommended value of 1.4×10^{-1} [112]. The situation is clearer with inhalation and dermal contact, where the characteristic value most widely used can be more clearly observed. For inhalation, the RfD is 1.43×10^{-5} [18,48,92], i.e., 1.4×10^{-5} [112], while the dermal RfD is 1.84×10^{-3} [92,122]; two researchers also mentioned 1.84×10^{-5} [115] and 8.4×10^{-3} [112].

Data on the RfD values of V, Fe, Sb, Mo, and Ba are very scarce in the literature; thus, the correctness of these data cannot be determined with certainty. Regarding V, two ingestion RfD values appear, namely, 7×10^{-3} [50,58] and 9×10^{-3} [4], with the latter also reported in the USEPA IRIS, but in the form of vanadium pentoxide. The USEPA IRIS also gives a value of 5×10^{-3} [95] for vanadium and compounds, but this number has not been used in publications. Inhalation and dermal contact are characterized by 7×10^{-3} [58] and 7×10^{-5} [43].

For Fe, the value of 0.7 [8,48], primarily used for ingestion, matches the USEPA recommendation. However, Taghavi et al. (2022) [92] gave an RfD ingestion value of 8.4. In the case of inhalation and dermal contact, values include 2.2×10^{-4} [92], 7×10^{-3} [48], 7×10^{-2} [92], and 2×10^{-3} [48].

Several studies in which Sb was analyzed used the same RfD ingestion value, i.e., 4×10^{-4} [13,46]. Inhalation and dermal contact had more variation, but references for these

data are very scarce. The researchers stated 5×10^{-4} [13] as a possible value for inhalation, while, for dermal contact, three different values appeared (2.7×10^{-3} [91], 6×10^{-5} [46], and 6×10^{-6} [13]).

The only RfD values mentioned for Mo were 5×10^{-3} [48,73], 2×10^{-3} , and 1.9×10^{-3} [48] for the three exposure pathways, respectively.

The situation is similar with Ba (2×10^{-1} (ingestion) and 2.9 (dermal contact) [91]), except that no information was given about inhalation RfD.

The largest differences reported were found for Pb and Mn in RfD_{ing}, for Cd, Ni, Hg, and As in RfD_{inh}, and for Cr and Ni in RfD_{derm}. Researchers agreed on the RfD values for Zn ingestion, inhalation, and Hg ingestion value, while a different number of data appear for all other RfDs. Due to many different values for the same HMs, it can be assumed that there was an error in writing some articles. Although most researchers gave the final RfD values for all three exposure pathways, some calculated it [91,102,112]. On the one hand, the RfD values for Cu, Ni, Pb, Zn, Hg, and As for all three exposure pathways were of the same orders of magnitude, i.e., a similar degree of danger to human health.

Cadmium has the lowest RfD dermal value, while, for Cr, RfD inhalation and dermal values are lower than for ingestion; Co and Mn have quite low values for inhalation RfD, and the same can be said for V for dermal contact.

Concerning the RfD values, a lower RfD value indicates a greater negative impact on human health. The question arises to what extent the introduction of metals into the body through inhalation and dermal contact can be dangerous compared to ingestion. It can be observed that, among most HMs, the RfD for dermal contact has the lowest value. The EPA gives values for ingestion only, which are taken in the literature for Cd, Cr, Cu, Ni, Zn, Hg, As, Fe, Ba, Sb, and Mo [95]. It is noteworthy that the USEPA IRIS reports RfD values for Co, V, and Mn, which have not been used in published studies.

4.14. Cancer Slope Factor (CSF)

The cancer slope factor (CSF) is the counterpart of the RfD and is used in carcinogenic risk calculations. The units in which it should be expressed are $\text{kg}\cdot\text{day}\cdot\text{mg}^{-1}$ [18,38,46]. However, there were ambiguities among the researchers when defining CSF units. The units $\text{kg}\cdot\text{d}\cdot\text{mg}^{-1}$ [5,104,107] or $\text{kg}\cdot\text{day}\cdot\text{mg}^{-1}$ were most often found in the literature; $\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$ [25,109,113] or $\text{mg}\cdot\text{kg}^{-1}\cdot\text{d}^{-1}$ [51,100] are faulty units, although some researchers left the value of this factor without unit, i.e., unitless [43], or did not even specify the unit [44,58]. To maintain the dimensionlessness of CR/TCR, the CSF unit must be equal to the reciprocal value of the RfD unit, i.e., $\text{kg}\cdot\text{day}\cdot\text{mg}^{-1}$. Since the CSF unit should correspond to the ADD unit, some researchers used $\text{m}^3\cdot\text{mg}^{-1}$ for CSF to correspond to the corresponding unit of ADD inhalation [52,118].

CSF values differ depending on individual HM and exist for all three exposure pathways. In addition to CSF [89,101], the SF [39,93,108] acronym is also found in the literature, although the USEPA documents this factor as CSF [95]. Unlike the RfD, not all HMs have CSF. That is, only certain elements are considered carcinogenic. In the literature, Cd, Cr, Ni, Pb, and As are reported as carcinogenic metals, and researchers considered the carcinogenic effects of all listed metals or only a few [25,35,73]. Rarely, Cu and Co were classified as carcinogenic [67,91,115]. There were variations among researchers regarding the values of this factor, but the most often used values were selected. Sometimes the CSF values were calculated, mostly for dermal contact [74,102,112]. For As, the same CSF values appeared almost always for all three exposure pathways, while there were three or more reported values for the other elements. Similar to RfD values, the CSF data were reported in the main text of an article or supplement, or sometimes omitted. The summary of the different CSF values for each element is presented in Table S3.

Cd is an HM with the largest values appearing for ingestion exposure. There were disagreements among researchers about the CSF values in all three pathways. For ingestion, out of all available values (0.38 [80], 6.3 [48,100], 0.51 [67], 3.8×10^{-3} [92], and 15 [44]), 6.1 [5,24,109] was most often used. It has been reported that Cd affects humans when

introduced into the body via inhalation equally as if ingested because the numerical value 6.3 [71,73,112] is quite close. Rarely, researchers used the following values for inhalation: 6.1 [114], 0.5 [52], 14.7 [74], 0.63 [18], and 1.8×10^{-3} [24]. The most used value for dermal contact was 6.1 [5,24,109]. Furthermore, 0.38 [52,114] and 20 [67,71] were values used for CSF dermal.

The USEPA IRIS recommends a CSF ingestion value of 0.5 for Cr(VI) [95]. Almost all researchers adopted this value; however, in a few cases, 8.5×10^{-3} [5], 0.42 [48], and 0.501 [113] were used. The value of CSF for ingestion is 84 times lower than CSF inhalation, and 40 times lower than CSF dermal. Therefore, it is considered that there is a greater danger of Cr entering through inhalation and dermal contact. The researchers almost unanimously agreed on a value of 42 [18,25,51] for inhalation and 20 [92,109] for dermal contact. Only a couple of researchers mentioned 4.2 [58] for CSF inhalation and 1.5 [52] and 2 [58] as options for CSF dermal.

The values of CSF for Ni were also different in the literature for all three exposure pathways. The most commonly used value for ingestion was 1.7 [13,36,122], and that for inhalation was 0.84 [25,46,112]. In addition, CSF ingestion values of 43 [49], 0.91 [44], and 0.84 [48] were present in some studies, as well as values of 0.9 [58] and 0.91 [39] for inhalation. A considerable doubt arises with CSF dermal because the most used value was 42.5 [36,113,122]. This suggests that the danger of ingesting Ni through the dermal route is more than 20 times greater than ingestion and more than 40 times greater than inhalation. Moreover, this value is twice as high as CSF dermal for Cr(VI) and Cd (assuming the value as 20). In addition to 42.5, some researchers used 1.7 [49], 4.25 [13], and 0.84 [10], but in very few cases. Because of the high uncertainty in CSF values for Ni (primarily due to questionable traceability to reliable sources), these CSF values will not be evaluated in the Final table section.

Minor variations among the data were found for Pb, where the researchers dealt with ingestion and/or inhalation, whereas CSF values for dermal contact were almost not mentioned. Many authors adopted the ingestion value of 8.5×10^{-3} [5,44,70], with no other information on that parameter in the literature. The researchers reported a Pb inhalation CSF value of 4.2×10^{-2} [18,73,112] and an ingestion value of 8.5×10^{-3} [5,44,70]. The influence of Pb on humans through dermal contact is under question because most researchers have not specified the value of this pathway. The values 8.5×10^{-3} [13,100] and 8.5×10^{-2} [112] were mentioned in several places, but the reliability of these data is questionable.

In addition to Cr(VI), the USEPA IRIS recommends an As ingestion CSF value of 1.5; in all the reviewed publications, only this number appears [13,91,113]. Although larger values appear for inhalation and dermal exposure, researchers agreed on the CSF severity for these two routes of exposure. It is accepted that the greatest risk of exposure to As is via inhalation, as this value was often stated 10 times higher than ingestion, amounting to 15.1 [25,38,46]. In addition, 4.3×10^{-3} [24,73], 1.51 [58], 15 [91], and 15.05 [112] can also be found in the literature. For dermal contact, the leading value was 3.66 [5,44], although there were also cases where it was replaced by 3.7 [114], 1.5 [13], 7.5 [10], or 1.58 [112].

Researchers uniquely agreed on ingestion CSF values of 8.5×10^{-3} for Pb and 1.5 for As, while several options emerged for other CSF values. The USEPA has only approved values for As (inorganic) = 1.5 and Cr(VI) = 0.5 for ingestion, while data for other elements are not available [95]. Cadmium and As have many different values for inhalation CSF in the literature. Of the carcinogenic elements, it can be concluded that Pb has the lowest CSF values, i.e., the lowest carcinogenic effect on humans. Although the mentioned data are widely used in the literature, certain illogicalities and discrepancies were noticed. For example, in the case of Cd, Cr, and Ni, it can be observed that the dermal CSF is enormously higher compared to the ingestion values. The inhalation CSF for Cr is also strange, being more than 80 times higher than that for ingestion. CSF values for As during inhalation and dermal contact are also higher than for ingestion, but not to such an extent. According to

these data, inhalation and dermal contact are more dangerous exposure routes than direct ingestion of HMs into the human body.

5. Monte Carlo Simulation

The USEPA-developed human health risk assessment model is still prevalent in evaluating environmental pollutant risks. In the deterministic approach, this model can lead to underestimating or overestimating the actual risk due to variability in metal concentrations, age, gender, body weight, and physiological and metabolic parameters. For the evaluation of probabilistic risk, which reduces the variability and uncertainty associated with health risk assessment, the Monte Carlo simulation (MCS) is widely used. This method gives more accurate health risk estimates since a set of numbers is used instead of single-point values [65,84,125].

MCS calculates risk for pollutants such as HMs on the basis of different values of input parameters, which are randomly selected from a probability distribution function for that parameter. The number of simulations is usually in the range of 1000–100,000. The MCS procedure involves three steps: 1—determination of the probability distribution for each input parameter, 2—random selection of input parameter values at each calculation, and 3—calculation of the health risk based on the selected input parameters using health risk assessment equations. The results of the calculation and their distribution are used to find the health risk probability associated with HMs in soil. As part of the Monte Carlo simulation, a sensitivity test determines the effect of observed parameters on the health risk arising from HMs in the soil [126,127].

For MCS, it is essential to determine the type of probability distribution of each parameter that figures in the calculation for carcinogenic and non-carcinogenic risk. Probability distributions of the parameters used in the latest studies dealing with HMs in soils will be summarized in the Final table section. This review included the analysis of 22 publications. Well-established types of distributions were used for some factors, while distributions were not determined for others, and point values were applied. However, there are parameters where some researchers applied a distribution while others used point values. Probability distributions were usually defined for the following parameters: element concentration, ingestion rate (IngR), inhalation rate (InhR), exposure frequency (EF), adherence factor (AF), and body weight (BW). For the following parameters, point values were usually used: exposure duration (ED), particulate emission factor (PEF), skin surface area (SA), dermal absorption factor (ABF), average time (AT), the conversion factor (CF), lifetime (LT), reference dose (RfD), and cancer slope factor (CSF). The concentrations for most analyzed HMs had a lognormal distribution [5,59,65,78,84,117,121,128–131], while the normal distribution for some elements was used in several articles [84,121,131]. Most researchers used a triangular distribution for the ingestion rate (IngR) of adults and children [5,37,59,65,117,121,128,130–134], but a lognormal distribution was also used in a significant number of articles [12,63,64,78,84,135,136], especially for the ingestion rate of children [12,63,64,78,84,134–137]. In some works, a normal distribution was also used as the type of probability distribution [129], or point values were applied [18]. For the InhR, a lognormal distribution [37,61,64,84,117,128,136] or point values [18,121,130,131,134] were used in the calculation of probabilistic health risk assessment, where the use of the lognormal distribution was more prevalent both for adults and children. A triangular distribution was mainly used for EF [12,18,37,63,65,84,117,128–131,133–137]; however, in some articles, point values [59,78,121,132] or a uniform distribution [64] was also applied. Point values were usually used for ED, at 24 years for adults and 6 years for children [18,37,59,117,130–132,134]. Several researchers stated other values, such as 70 years for adults [78,133,137] and 18 years [133,137] or 10 years [78] for children, as well as a different number of years depending on the element [129]. For ED, a uniform distribution was also applied in several of the analyzed articles [12,63,65,84,121,128,135], as well as triangular [64] and lognormal [136] distributions. Although researchers usually used point values for the SA parameter, various types of distribution (triangular,

lognormal, and normal) were also applied. For AF [12,37,64,117,121,130,131,133,134,137] and BW [5,12,18,37,59,64,65,78,117,121,133,135,137], the most used were lognormal distributions, while, for PEF [18,37,63,84,117,121,128,130,131,134], ABF [12,18,37,63,64,84,117,121,128,133,135,137], and AT [12,37,59,63,65,78,84,117,121,128–132,134–136], point values were usually used. Several researchers used other types of distributions for AF, such as Beta [63,84], Beta-PERT [128,135], triangular [136], and point values [18]. According to most researchers, BW tends to follow a lognormal distribution for adults and children. Nevertheless, the greatest variations in the type of distribution among different researchers could be found for parameter BW. Although a lognormal distribution was prevalent, some researchers used normal [63,84,128,129,136] and uniform [5,37,59,117,132] distributions for adults' body weight, normal [63,84,128,129,136] and triangular [18,133] distributions for children's body weight, and point values for adults and children [130,131,134]. In most analyzed articles, the type of distribution for parameters such as CF, LT, RfD, and CSF was not specified. Several researchers used point values or a triangular distribution for RfD and CSF [18,37,59,117,128,129,132]. Researchers used different terms for point values, such as single, uniform, fixed value, constant, and unitless.

6. Final Tables

On the basis of the reviewed studies on deterministic and probabilistic health risks of HMs in soil and sediment, we created three tables with the exposure factors most commonly used in the health risk assessment. These tables, therefore, can be considered a summary and our proposal for the optimal values of health risk variables. These values were considered as a function of the frequency of appearance in published articles dealing with this topic. Table 1 shows the units and values of the factors used to calculate the average daily dose (ADD) and the type of distribution used in MCS. In addition to the literature data, the units were checked by performing a dimensional analysis, which resolved doubts regarding the disagreement with units among some researchers. The type of distribution was the same for both children and adults. The reference doses and cancer slope factors that appear in the equations of non-carcinogenic and carcinogenic risk are shown in Table 3. The defined values of these two parameters were used equally in the calculations for both children and adults, since this is the most common classification of the human population.

The concentration of HMs is the critical parameter because it determines whether the examined area is characterized as dangerous for human health or not. It is the only parameter that depends on the study being conducted; thus, data on HM concentrations were specific to each case study. Although, unlike the other parameters in the equation, there is no established name for concentration, researchers used different expressions such as heavy metal content [51], concentration of elements in the samples [52], metal concentration in the test sample [107], heavy metal concentration in soil [9,46], and concentration of potentially toxic element [69]. Furthermore, all elements, including Cd, Cr, Cu, Ni, Pb, Zn, As, Hg, Co, Mn, V, Fe, Mo, Ba, and Sb, were equally classified under metals [58], heavy metals [53], potentially toxic elements [77,138], contaminants [13], and trace metals [35]. The adopted names and abbreviations of other parameters are aligned with the available literature and USEPA documents.

Table 3 shows the reference dose values for all three exposure pathways for Cd, Cr(VI), Cu, Ni, Pb, Zn, As, Hg(II), and V. For the other elements, Fe, Mo, MeHg, Cr(III), Ba, and Sb, only RfD ingestion doses were approved by the USEPA due to the scarcity of literature data regarding RfD_{ing} , RfD_{inh} , and RfD_{derm} for these HMs. In the case of Co and Mn RfD inhalation values are omitted. Table 2 contains the cancer slope factor data for four carcinogenic elements: Cd, Cr(VI), As, and Pb.

The RfD and CSF values for some elements were reported in the literature but are omitted in Tables 2 and 3, either due to disagreement among the published data or because traceability to the original study for the reference value was unreliable.

Table 1. Parameters, values, and distributions used for health risk assessment of HMs in soil and sediment.

Parameter	Abbrev.	Units	Children	Adults	Distribution
HMs concentration *	C	mg·kg ⁻¹	-	-	Lognormal
Ingestion rate	IngR	mg·day ⁻¹	200	100	Triangular
Inhalation rate	InhR	m ³ ·day ⁻¹	7.6	20	Lognormal
Exposure frequency	EF	day(s)·year ⁻¹	350	350	Triangular
Exposure duration	ED	year(s)	6	24	Point
Particulate emission factor	PEF	m ³ ·kg ⁻¹	1.36 × 10 ⁹	1.36 × 10 ⁹	Point
Skin surface area	SA	cm ²	2800	5700	Point
Adherence factor	AF	mg·cm ⁻² ·day ⁻¹	0.2	0.07	Lognormal
Dermal absorption factor	ABS	unitless	0.001	0.001	Point
Body weight	BW	kg	15	70	Lognormal
Average non-carcinogenic time	AT _{nc}	day(s)	2190	8760	Point
Average carcinogenic time	AT _c	day(s)	25,550	25,550	Point
Conversion factor	CF	kg·mg ⁻¹	10 ⁻⁶	10 ⁻⁶	Point
Lifetime	LT	year(s)	70	70	Point
Reference dose	RfD	mg·kg ⁻¹ ·day ⁻¹	**	**	Point
Cancer slope factor	CSF	kg·day·mg ⁻¹	**	**	Point

* Concentration of HMs in soil/sediment; ** RfD and CSF factor values are given in Tables 2 and 3.

Table 2. The cancer slope factors (kg·day·mg⁻¹) for HMs.

Elements	CSF _{ing}	CSF _{inh}	CSF _{derm}
As (inorganic)	1.5	15.1	3.66
Cd	6.1	6.3	6.1
Cr(VI)	0.5	42	20
Pb	0.0085	0.042	-

Table 3. The reference doses (mg·kg⁻¹·day⁻¹) for HMs.

Elements	RfD _{ing}	RfD _{inh}	RfD _{derm}
As (inorganic)	3 × 10 ⁻⁴	3 × 10 ⁻⁴	1.23 × 10 ⁻⁴
Ba	2 × 10 ⁻¹	-	-
Cd	1 × 10 ⁻³	1 × 10 ⁻⁵	1 × 10 ⁻⁵
Co	2 × 10 ⁻²	-	1.6 × 10 ⁻²
Cr(III)	1.5	-	-
Cr(VI)	3 × 10 ⁻³	2.86 × 10 ⁻⁵	6 × 10 ⁻⁵
Cu	4 × 10 ⁻²	4.02 × 10 ⁻²	1.2 × 10 ⁻²
Fe	7 × 10 ⁻¹	-	-
Hg(II)	3 × 10 ⁻⁴	8.6 × 10 ⁻⁵	2.1 × 10 ⁻⁵
MeHg	1 × 10 ⁻⁴	-	-
Mn	1.4 × 10 ⁻¹	-	1.84 × 10 ⁻³
Mo	5 × 10 ⁻³	-	-
Ni	2 × 10 ⁻²	2.06 × 10 ⁻²	5.4 × 10 ⁻³
Pb	3.5 × 10 ⁻³	3.52 × 10 ⁻³	5.25 × 10 ⁻⁴
Sb	4 × 10 ⁻⁴	-	-
V	7 × 10 ⁻³	7 × 10 ⁻³	7 × 10 ⁻⁵
Zn	3 × 10 ⁻¹	3 × 10 ⁻¹	6 × 10 ⁻²

It is crucial that the speciation of HMs be carried out in the analytical procedure. The considerable difference in toxicity between individual forms of some HMs can cause a huge error in risk estimates. For example, the inorganic form of As is highly toxic compared to its organic form [139]. Another example is Hg, which is more toxic in its organic form. In the case of Cr, Cr(VI), as a significantly more toxic species than Cr(III), should be highlighted [140].

One should be very careful when the “worst case scenario” is taken in a study, i.e., when the total concentration of individual HM is equal to the concentration of its more toxic form, e.g., MeHg = Hg(total), As(inorganic) = As(total), and Cr(VI) = Cr(total). According to our knowledge, the most common relationship in the environment is not well established for organic/inorganic arsenic. The MeHg/total Hg ratio is about <1% [141], while the most common Cr(III)/Cr(VI) ratio is approximately 6 [95]. However, in the case of industrial pollution, these ratios can be drastically different, which means that deep insight into the origin of pollution should be obtained, and analytical speciation should be performed before assessing the health risk.

7. Summary and Outlook

The health risk assessment methodology for processing the results of soil and sediment analysis has become increasingly attractive over the last few years. Researchers around the world have used this method to answer the question of whether HMs pose a certain danger to human health. Even though everyone deals with the same problem, researchers have large disagreements regarding the labeling, type, factor numerical values, and method of calculating the risk indices. These differences were thoroughly discussed in this review. The Web of Science statistics show increased interest in health risk assessment of HMs in soil and sediment. This is evidenced by the many recent publications and cited works dealing with this topic.

Before assessing the non-carcinogenic and carcinogenic risk, the average daily dose of HMs introduced into the human body is estimated, for which the most frequently used abbreviation is ADD. In most cases, three routes of exposure are included: ingestion, inhalation, and dermal contact; however, it can be emphasized that humans are least exposed to HMs through inhalation. EF, ED, BW, and AT are used in all three ADD equations, while IngR is used only for ingestion, InhR and PEF are used only for inhalation, and SA, AF, and ABS are used only for dermal contact. Due to the discrepancy in the units of some parameters (the most problematic being AF) and the ambiguity regarding the conversion factor, a detailed dimensional analysis was performed, which indicated that the factor CF is necessary for the equations for ingestion and dermal contact. The units of the factors are defined such that the ADD for all three exposure routes is expressed in $\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$.

The health risk is assessed, especially for children, and the adult population is considered residential. The most frequently used indices for non-carcinogenic and carcinogenic assessment are HQ/HI and CR/TCR, respectively. The categorization of the hazard index is based on the limit of one; for carcinogenic risk, the degree of danger depends on where the value of the TCR index is related to the numbers 10^{-6} and 10^{-4} . The HMs under investigation can be divided into three categories. The first group includes Cd, Cr, Cu, Ni, Pb, Zn, and As, which dominate in the literature, followed by the second group, Hg, Mo, and Co, which have been the subject of researchers' interest to a much lesser extent. Lastly, there are V, Fe, Mo, Ba, and Sb, which have hardly been mentioned in the literature. The reason why Hg is omitted in the first HM group is related to difficulties in the analytical determination of low levels of Hg. Specifically, the two common analytical techniques used for some HMs analysis, flame atomic absorption spectrometry (AAS) and inductively coupled plasma optical emission spectrometry (ICP-OES), are not capable of measuring Hg at trace levels, usually found in soil.

For almost all parameters, there are more than five names in publications, except for EF and ED, for which researchers almost unanimously agree on their naming. The

units differ both in labeling and in whether they are written in the singular or the plural, except for the AF factor; the units also differ among different researchers. There is also no consensus regarding CF units. The largest number of variations appears for the factors InhR, SA, and BW, while it can be noticed that most researchers use the same values for IngR, EF, ED for children, PEF, AF, AT, and CF. Determining HQ/HI requires the reference dose and CR/TCR cancer slope factors. The RfD and CSF data in the USEPA IRIS database can be found for ingestion, whereas different RfD values for all three exposure pathways can be found in published studies. Only Zn stands out as an element whose RfD values the researchers agree with.

Furthermore, only one RfD value for ingestion is available for Mo, Sb, Ba and Hg. The largest number of RfD values offered for all three exposure pathways can be found for Cd and Cr. CSF is found almost everywhere, but only for Cd, Cr, Pb, and As, which are also characterized as carcinogenic elements. Cr, as in the case of the RfD, has the largest variations for all three exposure pathways, while for As, the highest variation occurs with inhalation and dermal contact. Researchers fully agree on the value of CSF ingestion for As and Pb. It was found that the same distributions for children and adults are used in probabilistic risk assessment. The most different distributions can be found for the IngR and AF parameters, while the distributions for C, EF, ED, ABS, AT, and CF are consistent almost everywhere. Most researchers do not specify distributions for RfD and CSF.

By comparing all the details and steps of the health risk assessment, we conducted a review that might serve as a guide for determining the degree of danger to human health due to HMs in soil and sediment. It was necessary to comprehensively present the calculation method and highlight the differences that appear in the publications. On the basis of studied literature, summary tables are provided in which we recommend the relevant data for exposure factors.

Supplementary Materials: The following supporting information can be downloaded at <https://www.mdpi.com/article/10.3390/met13071266/s1>: Table S1. Variations in parameters in ADD formula in different studies; Table S2. Variations in reference dose RfD ($\text{mg}\cdot\text{kg}^{-1}\cdot\text{day}^{-1}$) values in different studies; Table S3. Variations in cancer slope factor CSF ($\text{kg}\cdot\text{day}\cdot\text{mg}^{-1}$) values in different studies. All references found in the Supplementary Materials are also present in the manuscript.

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