

Article

Evaluation of Agricultural Soil Quality and Associated Human Health Risks Following Plastic Fire Incidents: Insights from a Case Study

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Abstract

This study examines the impact of an unintended fire at the Drava International plastic processing facility near Osijek, Croatia, on soil quality and the potential human health risks associated with agricultural soils within a 10 and 20 km radius. Surface soil samples (0–5 cm) were collected from ten locations within 10 km three days after the incident, and eight composite samples were taken from sites 10–20 km away 17 days' post-event. Additionally, 18 control samples previously collected for soil fertility or quality monitoring were included for comparative analysis. In total, 36 agricultural soil samples were analyzed for pH, organic matter, total phosphorus, potassium, calcium, magnesium, and trace elements (Cr, Co, Ni, Cu, Zn, As, Pb). Eighteen post-fire samples were also analyzed for polycyclic aromatic hydrocarbons (PAHs), dioxins, and perfluoroalkyl substances (PFAS). Ecological risk was assessed using the pollution load index (PLI) and enrichment factor (EF), while human health risk was evaluated through the estimation of incremental lifetime cancer risk (ILCR) and individual carcinogenic risks (CRi) for As, Cr, Ni, and Pb. Results showed that concentrations of dioxins (TEQ LB and UB), dioxin-like PCBs, and non-dioxin-like PCBs in samples within 10 km were either below detection limits or present in trace amounts (4.0×10^{-6} mg/kg). PFAS compounds were not detected (<0.0005 mg/kg). The total concentration of non-dioxin-like PCBs ranged from 0.0023 to 0.0047 mg/kg, well below the maximum permissible levels. Post-fire contamination profiles revealed consistently higher PAH concentrations in the 0–10 km zone (mean 0.100 mg/kg) compared to the 10–20 km zone (mean 0.062 mg/kg). Twenty PLI values exceeded the threshold of 1 (range: 1.00–1.26), indicating moderate pollution, while the remaining values (PLI 0.82–0.99) suggested no pollution. EF values indicated minimal to moderate enrichment ($EF < 2$), supporting the conclusion that metal presence was predominantly geological with limited anthropogenic influence. All ILCR values for adults and children remained below the acceptable threshold of 1×10^{-4} , indicating low carcinogenic risk under both pre- and post-fire conditions. CRi values followed a consistent decreasing trend across exposure pathways: ingestion > dermal absorption > inhalation.

Keywords: plastic waste fire; open-space combustion; soil contamination; pollution load index (PLI); enrichment factor (EF); incremental lifetime cancer risk (ILCR)



Academic Editor: Claude Hammecker

Received: 25 September 2025

Revised: 21 October 2025

Accepted: 23 October 2025

Published: 27 October 2025

Citation: Perčin, A.; Hefer, H.; Zgorelec, Ž.; Galić, M.; Rašić, D.; Kisić, I. Evaluation of Agricultural Soil Quality and Associated Human Health Risks Following Plastic Fire Incidents: Insights from a Case Study. *Land* **2025**, *14*, 2137. <https://doi.org/10.3390/land14112137>

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1. Introduction

Soil is a natural formation and a very dynamic system governed by certain laws that, even today, at the beginning of the 21st century, are not fully clarified or scientifically determined [1]. It plays a fundamental role in ecosystem functioning, food production, and environmental health [2]. However, despite its vital ecological functions, soil is increasingly threatened by various forms of pollution, among which chemical contamination represents one of the most severe types of degradation, with consequences that affect the entire biosphere, especially heterotrophic organisms and humans as consumers within the food chain [3,4]. This growing concern highlights a broader issue within soil science: the productive role of soil is often overemphasized, while its ecological, regulatory, and protective functions tend to be marginalized [5]. These overlooked functions are crucial, especially considering the rapid and direct pathways through which soil pollutants can enter the food chain. The path from contaminated soil to human consumption is alarmingly short [6], making food safety an increasingly complex and critical challenge [7]. Soil pollution can originate from both natural processes (earthquakes, strong wind, floods, landslides, volcanic eruptions, forest fires, etc.) and anthropogenic activities, including industrial production, agriculture, traffic, and the burning of landfills and waste dumps [8].

Among various waste streams, plastic waste is one of the major contributors to environmental pollution [9]. In Europe, only one-third of discarded plastic is recycled, while a significant portion is incinerated or exported for disposal. Between 1950 and 2015, over 8.3 billion tons of plastic were produced, with more than 6.3 billion tons becoming waste. By the end of 2015, only 9% had been recycled, while the majority was discarded into the environment. By 2019, global production reached 460 million tons annually, resulting in a cumulative total of 9.5 billion tons, exceeding one ton per person worldwide. Such rapid growth in production, combined with improper disposal practices, poses serious environmental challenges [10–12]. In addition to pollution, plastic production significantly contributes to greenhouse gas emissions, particularly CO₂, exacerbating climate change [13]. Polyethylene (PE) is widely recognized as one of the most used types of plastic, with European demand for low-density polyethylene (LDPE) and high-density polyethylene (HDPE) at 17.5% and 12.3%, respectively [14]. Some authors report that 70.2 million tons of plastic were incinerated globally, emitting 0.92 ± 0.53 million tons of toxic aerosols [15], while others emphasize the negative impact of plastic waste storage on soil and humans [16–19].

In its solid form, plastic contributes to soil degradation primarily via the accumulation of microplastics, which alter soil structure, water retention, and chemical properties (pH, cadmium accumulation, cation exchange capacity, microbial activity, salinity, soil organic carbon) [20]. The environmental impact is significantly amplified when plastic waste is subjected to combustion. The burning of plastic, whether in open fires, landfill incidents, or incineration, releases a complex mixture of pollutants. Ash from open burning introduces polycyclic aromatic hydrocarbons (PAHs) and heavy metals into the soil, as plastic packaging often contains additives such as foaming agents, colorants, plasticizers, and flame retardants, including cadmium, chromium, lead, mercury, cobalt, tin, and zinc [18]. Similarly, natural forest fires have also been shown to elevate concentrations of heavy metals (Cr, Ni, Zn, Cu, Pb, Co, As and Hg) in soil and ash [21–23].

Once in the soil, these contaminants may persist for extended periods and pose risks to human health through direct exposure, food chain transfer, and groundwater contamination. Indirectly, plastic impacts health by affecting the safety and quality of food and water during production and storage [24,25]. Directly, post-fire soil contamination with heavy metals, especially with Pb, Cu and Cr, can pose serious risks to human health, particularly through inhalation and dermal exposure pathways [26]. Also, exposure to pollutants from fire smoke can increase risk of diseases related to the respiratory system

(hypertension, asthma, respiratory infections, and cardiac arrest) [19] and even increase risk of cancer later in life [27,28]. Earlier studies indicate that even long-term exposure to emissions from the controlled combustion of plastic waste may increase cancer risk in populations living near incineration facilities [29].

Although plastic pollution and soil contamination have been widely studied as separate issues, the specific mechanisms and impacts of plastic combustion residues on soil quality and human health remain quite underexplored. This manuscript aims to evaluate the impact of an anthropogenic event, specifically the unintended open fire at the Drava International plastic processing factory near Osijek, Croatia, on soil quality and assess the associated potential health risks to humans from the surrounding agricultural soil within a 20 km radius.

2. Materials and Methods

2.1. Fire Location and Climatic Condition

The incident occurred at the industrial complex (Drava International Ltd.) located at Brijest, a southern Osijek suburban settlement in Croatia (N 45°31'10.37", E 18°40'11.76"). Along with the production of plastic products, the firm specializes in the recycling of non-metallic residues and waste, as well as PET packaging recycling. The industrial complex of Drava International factory comprises multiple warehouses and manufacturing units and all operational activities are managed and executed to contribute to environmental preservation. On 4 October 2023, at approximately 00:40 h, an open-space fire broke out at the plastic waste landfill within the factory premises. The blaze, fueled by flammable materials stored within the landfill, quickly escalated, necessitating a multi-unit response. Despite the prompt intervention of all available firefighting forces from the Osijek-Baranja County (300 firefighters and 60 vehicles), the fire intensified for hours, engulfing several thousand square meters of deposited plastic and even part of the factory building. By the afternoon of October 5th, the fire was brought under control by the firefighters. The biggest concern throughout was the extremely black smoke (Figure 1), which, depending on the strength and direction of the wind, threatened areas of the city of Osijek and the municipalities of Antunovac and Ernestinovo, as well as the wider area of Osijek-Baranja County. The hypothesis posited that the fire potentially compromised the air and soil quality in the broader Osijek-Baranja County and eastern Slavonia region. This concern is particularly acute for the soil in Brijest, which is increasingly being allocated to young families with children. Given the area's well-developed sectors in crop production, apiculture, viticulture, and livestock farming, it was imperative to assess the ecological and health risk to preserve human health and high-quality, clean, and healthy agricultural soil.



Figure 1. Cont.



Figure 1. Fire incident at Drava International, Osijek, Croatia: 4 October 2023.

The area has a continental climate with cold and relatively prolonged winters, and warm-to-hot summers. Winter temperatures range from below freezing to 5 °C, with 40–55 mm of monthly precipitation. June records the highest precipitation, with a long-term average of approximately 80 mm, while the period of July–September is drier. Spring and autumn are mild, with moderate rainfall. Given the significance of current weather conditions at the time of the open-space fire at the Drava International factory as indicators for potential future soil contamination it is important to add that during the fire (4–7 October 2023), no precipitation was recorded. Temperatures ranged from 13.2 °C to 15.5 °C, and southwest wind remained dominant at 1–3 Beaufort.

2.2. Soil Classification and Agricultural Practices in the Investigated Area

The soil in the studied area is uniquely classified as Mollic Calcic Gleysols Loamic [30], with a pedosystematic profile structure of P-Gso-Gso/r-GrC-C. This soil type exhibits all hydrogenation processes and is characterized by a mollic aquatic surface black horizon. Due to significant water oscillation, there is pronounced internal migration within the soil profile.

2.3. Soil Sampling

Soil sampling after the fire was conducted on two occasions: 7 October and 21 October 2023 in Osijek, Croatia (Figure 2a,b). During the first sampling (7 October), 10 composite soil samples were collected within a 10 km radius from the incident/fire site at Antunovac (Ant I, Ant II, Ant III, Ant IV, Ant V, Ant VI, Ant VII, Ant VIII, Ant IX, Ant X). These locations are marked yellow on Figure 2c. In the second sampling (21 October), an additional eight composite soil samples were collected within a 10 to 20 km radius at locations in Ernestinovo (Ern I, Ern II, Ern III, Ern IV), Ivanovac (Ivan I, Ivan II), and Tenja (Tenja I, Tenja II). These locations are marked in pink on Figure 2c. Soil sampling was conducted south of the incident site, given the predominance of agricultural land and crops in that area, as well as the prevailing wind direction during the relevant period. It is important to emphasize that the northern side of the incident site borders the urban area of the city of Osijek.

The composite soil samples from all 18 locations were taken from a very shallow surface arable layer at a depth of up to 5 cm. Sampling was conducted using a laboratory shovel at locations without vegetation cover. This method aimed to capture dry atmospheric deposition that settled on the soil surface, as the presence of vegetation could indirectly affect the potential deposition of airborne contaminants and thus influence the final research results.

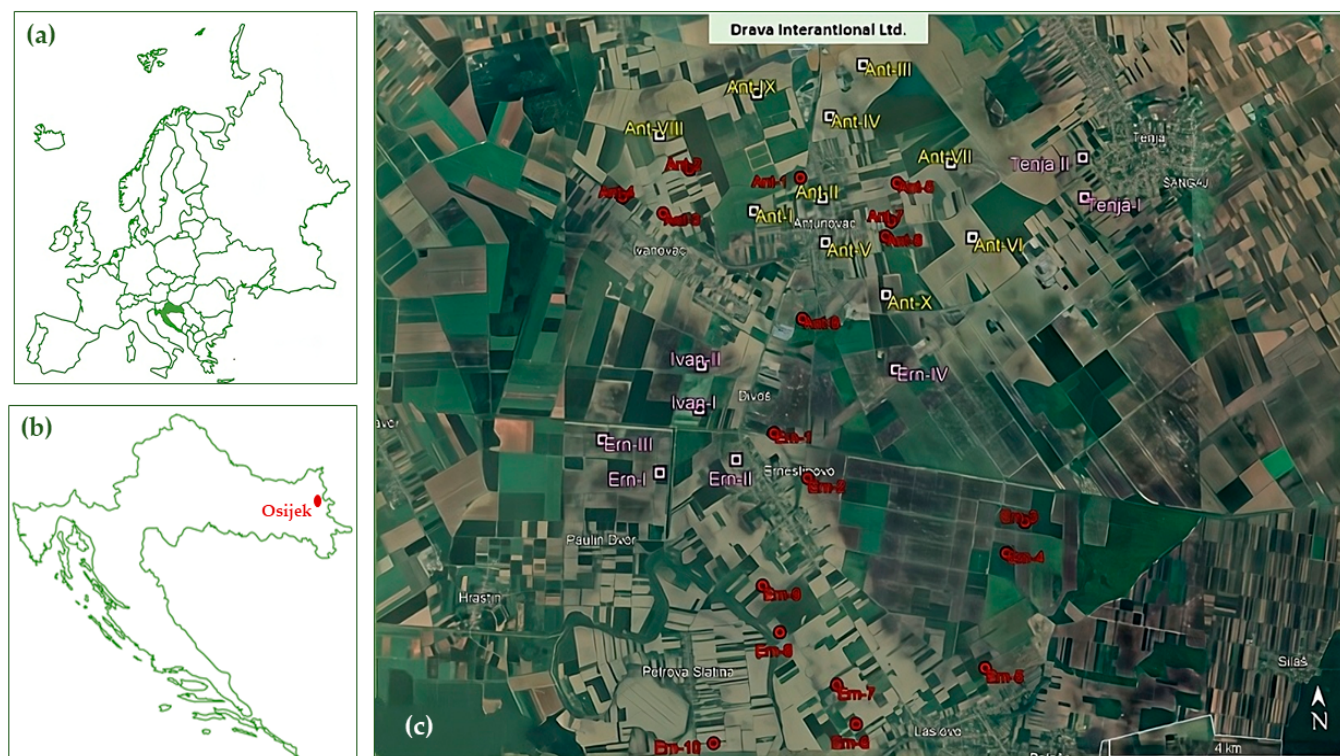


Figure 2. Study area: location of Croatia in European comparison (a). The city of Osijek in Croatia (b); soil sampling locations before (red letters) and after the fire incident (yellow and pink letters) and location of industrial complex Drava International Ltd. (c).

To further confirm the potential impact of the fire on the chemical properties of the soil (pH, organic matter content, total phosphorus, potassium, calcium, magnesium, chromium, cobalt, nickel, copper, zinc, arsenic, and lead), the aforementioned analyses were also conducted on archived control soil samples collected long before the fire, during the period of 2020–2022, by the Faculty of Agriculture, University of Zagreb, and the Croatian Agency for Agriculture and Food from Osijek. These control soil samples were also collected within a 20 km radius from the fire locations in Antunovac and Ernestinovo. A total of 8 composite samples were collected at Antunovac (Ant 1–8), and 10 composite samples were collected at Ernestinovo (Ern 1–10) during this three-year period before the fire. These 18 soil samples were previously collected for soil fertility control or for various studies on soil quality assessment. The locations of these samples are marked in red in Figure 2c. All 36 soil samples were collected from arable agricultural land, mostly used for crop production (maize, wheat, soybean, sunflower, etc.).

2.4. Laboratory Analysis

The soil samples were air-dried, ground, and sieved (<2 mm) in accordance with the ISO standard [31]. In the analytical laboratory of the Department of general agronomy at the Faculty of Agriculture, University of Zagreb, and the laboratory of the Croatian Agency for Agriculture and Food in Osijek, the following soil chemical parameters were determined in all 36 soil samples, with 18 collected after the fire and 18 control samples collected before the fire: soil pH, soil organic matter, total concentrations of calcium, magnesium, phosphorus, potassium, chromium, cobalt, nickel, copper, zinc, arsenic and lead. In 18 samples collected after the fire, in addition to inorganic soil parameters, organic components were also determined (sum of polycyclic aromatic hydrocarbons (PAHs) and content of dioxins and perfluoro-alkyl (PFAS) substances).

Soil pH was determined in a potassium chloride solution (*w/v* 1:5; 1 mol/L KCl) according to the ISO method [32], and soil organic matter (SOM) was determined using the Tjurin method (bichromate volumetric method, modified [33]). The contents of total phosphorus, potassium, calcium, and magnesium were determined by the portable X-ray fluorescence method according to the loose powder method [34] and ISO standard [35]. Also, using the same technique, the content of aluminum in all soil samples was determined. The limits of detection (LOD) for P, K, Ca, Mg and Al were 30, 30, 30, 3500, and 325 mg/kg, respectively. All analyses were performed using a Vanta VCR G2 handheld XRF analyzer (Olympus, Waltham, MA, USA). The accuracy and precision of the analyses were controlled using certified soil material (SRM 2711). Concentrations of chromium, cobalt, nickel, copper, zinc, arsenic, and lead were detected and quantified using atomic absorption spectrometry [36] after the prior digestion of elements in aqua regia according to ISO standard [37]. LOD for Cr, Co, Ni, Cu, Zn, As, and Pb were 0.0060, 0.0050, 0.0050, 0.0030, 0.2000, 0.0030 and 0.0050 ppm, respectively, while the limits of quantification (LOQ) were 0.2500, 0.5000, 0.5000, 0.5000, 0.5000, and 0.5000 ppm, respectively. Analyses were performed using a Varian AA240FS atomic absorption spectrometer (Palo Alto, CA, USA). Soil samples for the detection of organic contaminants were subjected to targeted extraction (Quechers method with acetonitrile for the sum of PAHs; solid-phase extraction for PFAS; and automated extraction for PCDD/Fs and dioxin-like PCBs). PAHs were analyzed using high-performance liquid chromatography (HPLC) equipped with a fluorescence detector (FLD) (Agilent Series 1100, Santa Clara, CA, USA), where the LOD was 0.01 mg/kg. Analytical quantification PCDD/Fs, PFAS, and PCBs was performed using a high-resolution gas chromatography–mass spectrometry system (Agilent 7250 GC/Q-TOF, Santa Clara, CA, USA). The LOD for individual PCB congeners ranged from 2.1 to 25 ng/kg for PFAS 0.0005 mg/kg, and from 0.69 to 22 ng/kg for individual dioxins, with toxic equivalency (TEQ) values for dioxins and dioxin-like PCBs calculated according to WHO-TEF guidelines [38] using the lower-bound (LB) approach.

2.5. Soil Pollution and Health Risk Assessment

The ecological risk assessment of metal exposure in soil before and after the fire was based on the calculation of the pollution load index (PLI) and the enrichment factor [39]. The PLI was calculated according to Equation (1), using previously determined contamination factors (CF). The contamination factors ($CF = C_s/C_b$) at each of the 36 sampled locations, before and after the fire, represented the ratio between the determined content of As, Cr, Cu, Ni, Pb, Zn, and Co in the soil (C_s) and the background values of the content of these accumulated metals at the study area (C_b). The background values (C_b) referred to the metal content in the sampling area as shown in the Geochemical Atlas of the Republic of Croatia (Table 1) [40].

$$PLI = \sqrt[n]{CF_1 \times CF_2, \dots \times CF_n} \quad (1)$$

Table 1. Background metal content (mg/kg) in soil at studied area.

Al	As	Cr	Cu	Ni	Pb	Zn	Co
60,000	12.1	54.8	13.0	23.1	24.1	88.0	8.0

Source: [40].

For additional confirmation, the enrichment factor (EF) was calculated according to Equation (2), indicating possible sources of anthropogenic influence on soil metal contamination. The EF is calculated by comparing the metal content in the observed soil samples with the content of a reference metal in the soil. This calculation also includes knowledge of the reference metal content in the study area, contributing to the geochemical normalization

of the calculation. Aluminum was chosen as the reference metal, and its background value at the study location is shown in Table 1.

$$EF = \frac{\left(\frac{C_i}{C_{ref}}\right)_{\text{sample}}}{\left(\frac{B_i}{B_{ref}}\right)_{\text{background}}} \quad (2)$$

In addition to the ecological risk calculation, the potential carcinogenic risk of the accumulated As, Cr, Ni, and Pb in the soil to human health was also assessed. The calculation included the estimation of the incremental lifetime cancer risk (ILCR) according to Equation (3), which relates to the sum of the carcinogenic risks (CR_i) contributed by the four metals mentioned through three pathways of human exposure [ingestion (ing-soil), inhalation (inh), and dermal absorption (der)]. Carcinogenic risks were calculated according to Equation (4), and the cancer slope factor (SF) values are shown in Table 2. For all the calculations, the average daily doses (ADD) were initially determined according to the relationships shown in the text below (Equation (5)) [41]. The values included in the calculation of the average daily dose (ADD) are shown in Table 3. The calculation was performed for both adults and children.

$$ILCR = \sum_1^i CR_i \quad (3)$$

$$CR_i = ADD_i \times SF \quad (4)$$

$$\begin{aligned} ADD_{\text{ing-soil}} &= \frac{(C_{\text{soil}} \times \text{IngR}_{\text{soil}} \times EF \times ED \times CF)}{(BW \times AT)} \\ ADD_{\text{inh}} &= \frac{(C_{\text{soil}} \times \text{IngR}_{\text{soil}} \times EF \times ED)}{(BW \times AT \times \text{PEF})} \\ ADD_{\text{der}} &= \frac{(C_{\text{soil}} \times \text{AF}_{\text{soil}} \times \text{SA} \times \text{ABS} \times EF \times ED \times CF)}{(BW \times AT)} \end{aligned} \quad (5)$$

Table 2. The cancer slope factors for heavy metals.

Heavy Metal	SF (kg day/mg)		
	Ingestion (SF _{ing})	Inhalation (SF _{inh})	Dermal (SF _{der})
As	1.5	15.1	3.66
Cr	0.5	42	20
Ni	-	0.84	-
Pb	0.0085	0.042	-

Source: [41,42].

Table 3. Parameters used for cancerogenic health risk assessment of metals in soil.

Parameter	Abbrev.	Units	Children	Adults
Metals concentration in soil	C _{soil}	mg/kg	In the soil after the fire	
Ingestion rate	IngR _{soil}	mg/day	200	100
Inhalation rate	InhR	m ³ /day	7.6	20
Exposure frequency	EF	days/year	350	350
Exposure duration	ED	years	6	24
Particulate emission factor	PEF	m ³ /kg	1.36 × 10 ⁹	1.36 × 10 ⁹
Skin surface area	SA	cm ²	2800	5700
Adherence factor	AF _{soil}	mg/cm ² /day	0.2	0.07
Dermal absorption factor	ABS	-	0.001	0.001
Body weight	BW	kg	15	70
Average carcinogenic time	AT	day	25,500	25,500
Conversion factor	CF	kg/mg	10 ⁻⁶	10 ⁻⁶

Source: [41,42].

All results of soil pollution and health risk assessment were interpreted according to the values presented in Table 4.

Table 4. Interpretation values.

Contamination Factor (CF)	
CF < 1	Low contamination
$1 \leq CF < 3$	Moderate contamination
$3 \leq CF < 6$	Considerable contamination
CF ≥ 6	Very high contamination
Pollution load index (PLI)	
PLI < 1	Unpolluted
PLI > 1	Polluted
Enrichment factor (EF)	
EF ≤ 1	No enrichment
$1 \leq EF \leq 2$	Slight enrichment
$2 \leq EF \leq 5$	Moderate enrichment
$5 \leq EF \leq 20$	Significant enrichment
The incremental lifetime cancer risk (ILCR) and carcinogenic risks (CRi)	
CRi/ILCR $\leq 1 \times 10^{-6}$	Negligible cancerogenic risk
$1 \times 10^{-6} \leq \text{CRi/ILCR} \leq 1 \times 10^{-4}$	Acceptable cancerogenic risk
CRi/ILCR $> 1 \times 10^{-4}$	Harmful cancerogenic risk

Source: [39,41–43].

2.6. Statistical Analysis

The statistical analysis encompassed the calculation of descriptive statistical parameters, including the minimum, maximum, mean, coefficient of variation, and standard deviation for pH, total phosphorus, potassium, calcium, magnesium, chromium, cobalt, nickel, copper, zinc, arsenic, lead, and organic matter content in the soil. All parameters were calculated with respect to the sampling time (before and after the fire). Differences in the mean values of these soil properties, contingent on the sampling time, were assessed using analysis of variance (ANOVA) and tested with Fisher's LSD test at a 5% significance level ($p = 0.05$).

3. Results

3.1. Post-Fire Soil Contamination Profile: Status of Organic and Inorganic Pollutants in Soil

To evaluate the environmental impact of the fire incident, soil samples collected after the fire within a 0–20 km radius were analyzed for organic and inorganic pollutants. Table 5 presents the results of the analysis of the presence of toxic organic compounds, including total dioxin toxicity expressed as toxic equivalents (TEQ) for both lower-bound (LB) and upper-bound (UB) estimates, perfluoro-alkyl substances (PFAS), the total TEQ of dioxins and dioxin-like PCBs (LB and UB), and the sum of non-dioxin-like PCBs (LB and UB). All concentrations are reported in mg/kg of dry soil and compared against the Maximum Permissible Concentration (MPC), set at 0.5 mg/kg [44]. In all analyzed samples (Ant I–X) within 0–10 km of fire incident, concentrations of dioxins (TEQ LB and UB), dioxin-like PCBs, and non-dioxin-like PCBs were either below detection limits or present in trace amounts (4.0×10^{-6} mg/kg). PFAS compounds were not detected (values < 0.0005 mg/kg). The total concentration of non-dioxin-like PCBs ranged from 0.0023 to 0.0047 mg/kg, which was significantly below the MPC threshold. Soil samples from more distant locations (Ern I–IV, Ivan I–II, Tenja I–II) exhibited a similar pattern. Dioxins and PCBs were either undetectable or present in minimal concentra-

tions (up to 5.0×10^{-6} mg/kg). PFAS compounds were again below detection limits. The concentration of non-dioxin-like PCBs was slightly lower than in the closer area, ranging from 0.0017 to 0.0033 mg/kg. Slight variations among locations (e.g., Ant II: 0.0047 mg/kg vs. Ant IV: 0.0023 mg/kg) likely reflect differences in soil organic matter and texture, which influence pollutant adsorption. Sites closer to the fire (0–10 km) generally exhibited marginally higher PCB values compared to those farther away (10–20 km), suggesting limited atmospheric deposition as the primary source. Wind direction during the incident (predominantly southwest) and the absence of precipitation minimized pollutant dispersion via runoff, reinforcing air-driven deposition as the dominant mechanism. Soil mineral composition and historical agricultural practices may also contribute to minor variability in trace concentrations.

Table 5. Dioxins and perfluoro-alkyl substances in the soil after the fire incident.

Location	Sum of Dioxins TEQ LB	Sum of Dioxins TEQ UB	Perfluoro-Alkyl Substances PFAS-s	Sum of Dioxins and Dioxin-Like PCB-a TEQ LB	Sum of Dioxins and Dioxin-Like PCB-a TEQ UB	Sum of Non-Dioxin-Like LB	Sum of Non-Dioxin-Like UB	MPC
mg/kg dry soil								
0–10 km radius from the fire site								
Ant I	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0032	0.5
Ant II	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0047	
Ant III	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0044	
Ant IV	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0023	
Ant V	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0034	
Ant VI	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0031	
Ant VII	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0037	
Ant VIII	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0032	
Ant IX	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0038	
Ant X	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0039	
10–20 km radius from the fire site								
Ern I	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0028	
Ern II	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0022	
Ern III	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0018	
Ern IV	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0024	
Ivan I	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0033	
Ivan II	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0025	
Tenja I	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0024	
Tenja II	0	4.0×10^{-6}	<0.0005	0	5.0×10^{-6}	0	0.0017	

MPC—Maximum Permissible Concentrations according to [44,45].

Considering that PAHs are persistent organic pollutants known for their carcinogenic and mutagenic properties, commonly released during the incomplete combustion of organic materials, including plastics, Figure 3 represents the sum of PAHs (expressed in mg/kg of dry soil) across individual sampling locations. Samples from the 0–10 km zone (Ant I–X) consistently exhibited higher PAH concentrations (0.100 mg/kg). In contrast, samples from the 10–20 km zone (Ern I–IV, Ivan I–II, Tenja I–II) showed notably lower concentrations, average 0.062 mg/kg, with the lowest value recorded in the Ivan I location (0.004 mg/kg) and the highest value detected at the Ern IV location (0.169 mg/kg). These differences again suggest that atmospheric deposition, wind-driven dispersion, was the dominant transport mechanism, with higher accumulation closer to the source due to proximity and prevailing southwest winds during the incident. The outlier at Ern IV may reflect localized factors such as microtopography, or industrial activity.

Table 6 summarizes the descriptive statistics (minimum, maximum, mean, standard deviation, and coefficient of variation) for selected parameters, including pH, soil organic matter (SOM), heavy metals (Cr, Co, Ni, Cu, Zn, As, Pb), and macronutrients (K, Ca, P, Mg), measured before and after the incident. Additionally, the results of one-way ANOVA and least significant difference (LSD) tests are presented to evaluate statistical significance. Soil pH ranged from acidic to alkaline at locations before the fire, and from weakly acidic to

alkaline at sampled locations after the fire incident. A slight but not significant increase in pH was recorded after the fire (from 6.82 to 6.93). SOM content remained relatively stable (mean 2.7% before and 2.6% after fire), with no statistically significant differences ($p > 0.05$). Chromium (Cr), zinc (Zn), and lead (Pb) concentrations showed no significant changes ($p > 0.05$), while cobalt (Co), nickel (Ni), copper (Cu), and arsenic (As) exhibited statistically significant differences ($p \leq 0.05$). Potassium (K) showed a highly significant increase ($p < 0.0001$), likely due to ash deposition. Phosphorus (P) also showed significant differences ($p = 0.004$), indicating altered nutrient dynamics, possibly from combustion by-products or changes in soil microbial activity. The coefficient of variation (CV) was highest for calcium and phosphorus, indicating substantial spatial heterogeneity.

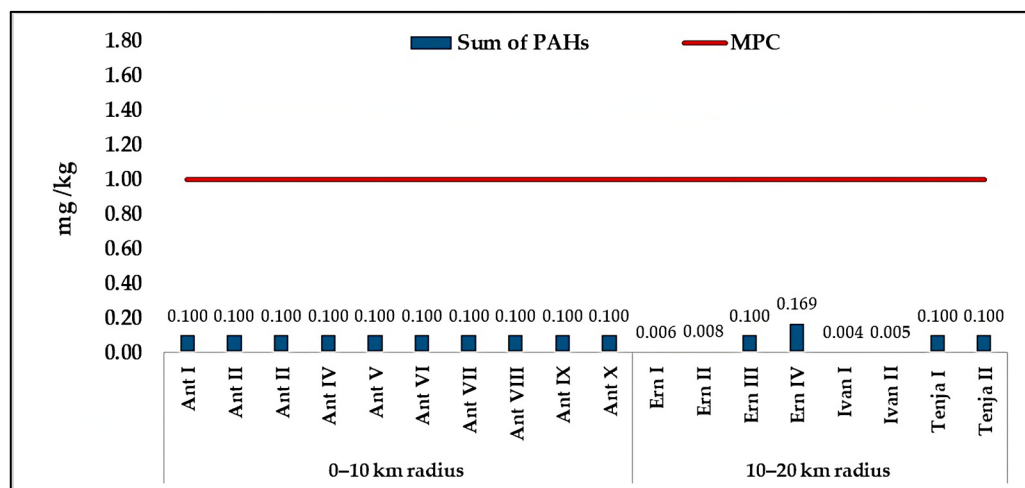


Figure 3. Spatial distribution of PAHs in soil after the fire incident. MPC—Maximum Permissible Concentrations according to [44].

Table 6. Descriptive statistics and ANOVA results for selected soil chemical properties before and after a fire incident.

	pH (1:5, KCl)	SOM	Cr	Co	Ni	Cu	Zn	As	Pb	K	Ca	P	Mg
		%	mg/kg										
Before the fire incident (<i>n</i> = 18)													
Min.	4.92	1.10	29.0	9.9	29.7	20.2	52.4	8.8	16.3	14.0	4.3	22.1	9.1
Max.	7.50	4.50	37.9	17.0	41.5	38.8	83.1	12.3	26.7	17.7	344	1097	28.4
SD	0.64	0.78	2.6	2.0	3.7	4.6	8.1	0.9	2.2	0.9	46.3	283.4	3.1
Mean	6.82	2.73	33.8	13.3	35.9	27.9	65.6	10.1	19.6	15.7	16.7	451.4	11.2
CV, %	9.44	28.7	7.7	15.3	10.2	16.5	12.3	9.3	11.3	5.8	277.3	62.8	27.2
After the fire incident (<i>n</i> = 18)													
Min.	6.08	1.10	17.0	9.0	26.0	19.2	42.0	6.3	11.0	14.4	5.6	274	5.6
Max.	7.41	4.20	52.5	15.0	38.1	32.0	86.0	13.2	26.4	19.7	38.8	1115	23.1
SD	0.34	0.96	12.8	2.0	3.6	3.9	13.5	1.7	3.7	1.38	10.4	252.0	2.96
Mean	6.93	2.58	31.9	11.7	32.6	25.1	63.4	9.1	20.9	16.9	14.0	602.1	9.04
CV, %	4.91	37.1	40.3	17.0	11.1	15.5	21.2	19.2	17.9	8.19	74.4	41.8	32.7
ANOVA results													
<i>p</i> -value	0.528	0.607	0.533	0.023	0.009	0.048	0.570	0.035	0.082	<0001	0.686	0.004	0.0002
LSD	0.339	0.577	6.28	1.36	2.46	2.79	7.52	0.95	2.08	0.448	12.8	102.3	1.14

Min.—minimum; Max.—maximum, SD—standard deviation; CV—coefficient of variation; LSD—least significant difference.

3.2. Ecological Risk Assessment of Heavy Metal Content in Soil Before and After the Fire Incident

The pollution load index (PLI) was employed to assess the extent of heavy metal contamination in soil samples collected both prior to and following a fire incident involving plastic waste. Figure 4a illustrates the spatial distribution of PLI values before the fire. Prior to the fire, most locations within the 0–10 km zone exhibited PLI values above the threshold of 1 (range: 1.09–1.21), indicating moderate pollution, while several sites in the 10–20 km zone also exceeded this limit (up to 1.17), which was likely impacted by anthropogenic activities. In contrast, Figure 4b presents the PLI values calculated based on metal content in soil after the fire. Post-fire measurements revealed a general decrease in PLI values, with most sites falling below 1 (range: 0.82–0.98), except for isolated increases at Ern I–IV (up to 1.26). Acknowledging that the four locations with the highest PLI values are situated furthest from the fire site (Figure 4b), it is important to note that these areas are subject to long-term intensive crop production. Such practices often involve the repeated application of mineral fertilizers and agrochemicals, which can lead to gradual accumulation of trace metals and other pollutants in the soil. This historical anthropogenic input likely contributed to the elevated PLI values observed at these sites, rather than direct deposition from the fire incident. These findings highlight the need to consider both natural soil characteristics and land-use history when interpreting spatial variability in contamination indices.

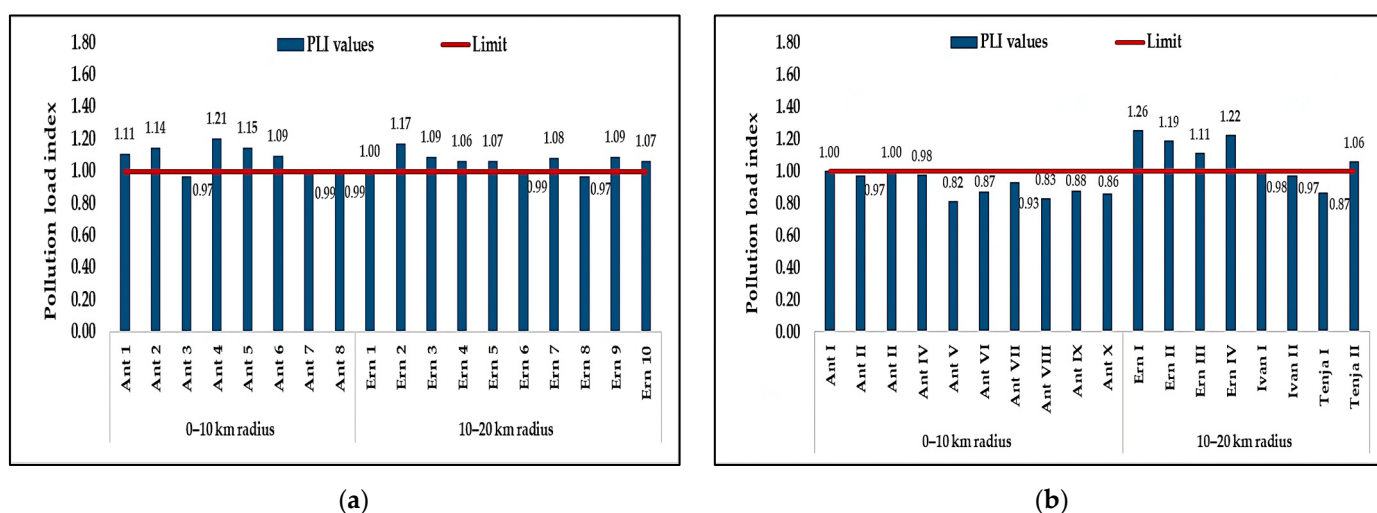


Figure 4. Spatial distribution of PLI values in soil before (a) and after (b) the fire incident.

Figure 5 illustrates the spatial distribution of enrichment factor (EF) values in soil samples collected before (a) and after (b) the fire incident. The EF was calculated to assess the degree of anthropogenic influence on metal concentrations, using a conservative reference element to normalize natural background variability. The mean EF values for Cr, Cu, Zn, Ni, Pb, Co and As before the fire were 0.99, 1.13, 1.01, 1.05, 1.01, 1.07 and 1.00, respectively, and after the fire, they were 1.04, 1.10, 1.02, 1.05, 1.02, 1.06 and 1.02, respectively. Based on the classification of EF proposed by [43], most of the presented factors indicate slight or minor enrichment ($1 < EF < 3$).

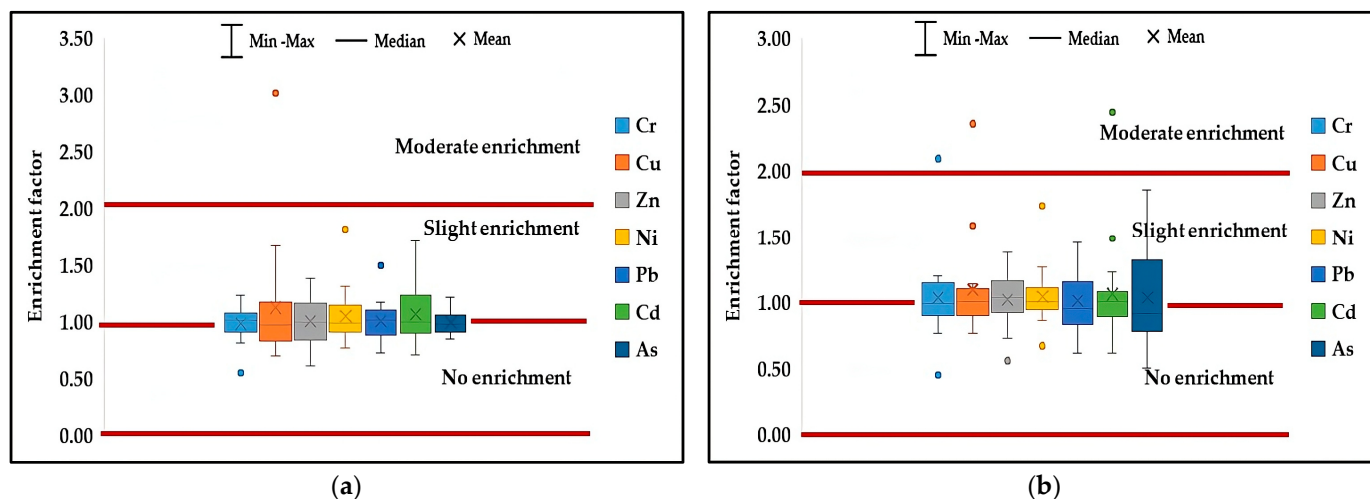


Figure 5. Spatial distribution of EF values in soil before (a) and after (b) the fire incident.

3.3. Health Risk Assessment of Heavy Metal Content in Soil Before and After the Fire Incident

The incremental lifetime cancer risk (ILCR) was calculated to evaluate the potential carcinogenic risk posed by a fire incident. The ILCR values were estimated separately for adults and children across multiple sampling locations. Figure 6a,b illustrate the spatial distribution of ILCR values for both population groups. The results indicate that all calculated ILCR values for adults and children remained below the upper acceptable limit of 1×10^{-4} , suggesting a low carcinogenic risk or acceptable cancer risk under pre-fire and post-fire conditions. However, slight spatial variability was observed. For example, post-fire ILCR values at Ern I–IV (10–20 km zone) were marginally higher compared to other sites, although still within safe limits. These differences may be attributed to localized factors, especially historical agricultural practices, which can influence the retention of trace contaminants. Children consistently exhibited higher ILCR values than adults across all locations, reflecting greater exposure sensitivity due to body weight and ingestion rates. The absence of precipitation during the post-fire period minimized pollutant redistribution, suggesting that observed variations are primarily linked to site-specific characteristics rather than widespread contamination from the fire. These findings align with the PAH and PLI patterns, where Ern I–IV also showed slightly elevated values, reinforcing the role of land-use history in shaping risk profiles.

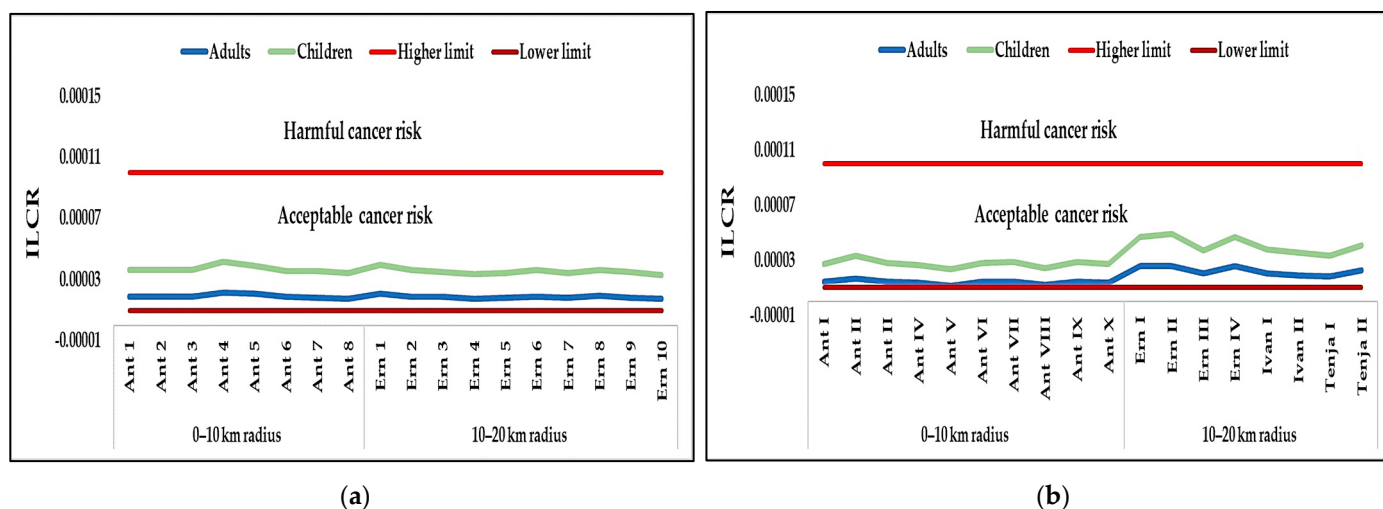


Figure 6. Spatial distribution of ILCR values in soil before (a) and after (b) the fire incident.

4. Discussion

4.1. Post-Fire Soil Contamination Profile: Status of Organic and Inorganic Pollutants in Soil

The results indicate that the plastic waste fire did not lead to a significant increase in the concentrations of the analyzed organic pollutants in the soil (Table 5), even near the fire site (0–10 km radius). All measured values were substantially below the established MPC, suggesting that there was no immediate environmental risk associated with the presence of these compounds in the soil. These findings are consistent with several recent studies that have investigated the environmental impact of plastic combustion. For instance, [46] explained that the open burning of plastic waste can release dioxins and furans, particularly under incomplete combustion conditions. Some authors emphasized that open waste fires are a major source of environmental pollution [19]. The extent of soil contamination depends heavily on the type of plastic burned, combustion conditions, and soil characteristics, especially due to combustion temperature and dispersion [47]. In this case, the low concentrations of dioxins (TEQ LB and UB) and dioxin-like PCBs suggest that either the combustion was relatively complete or that deposition onto soil was minimal. The absence of PFAS compounds (<0.0005 mg/kg) is particularly notable, given their persistence and potential for long-range transport. PFAS are known to adsorb onto soil particles and resist degradation, but their presence is often linked to specific industrial sources or firefighting foams, which may not have been involved in this incident [48]. The total concentration of non-dioxin-like PCBs, ranging from 0.0023 to 0.0047 mg/kg in the near fire zone and 0.0017 to 0.0033 mg/kg in the distant fire zone, remains well below risk thresholds. These values are far below levels (0.0057–103.5 mg/kg) reported in urban industrial soils in Mexico [49]. Nonetheless, even trace amounts of PCBs and dioxins are of concern due to their bioaccumulative nature, endocrine-disrupting potential, and long-term ecological effects. According to environmental risk limits derived by the Dutch RIVM and used in EU soil quality assessments, the MPC for the sum of 16 priority PAHs in soil is approximately 1.0 mg/kg for residential and agricultural land use [45]. This threshold is based on ecotoxicological data and human health risk assessments and is widely referenced in EU soil contamination studies. All measured PAH concentrations in this study were well below the 1.0 mg/kg threshold, indicating that the fire did not result in soil contamination levels that would pose a significant risk to the environment under current EU guidelines. Nevertheless, these results suggest a clear spatial gradient in PAH content, with elevated levels in closer proximity to the fire site. The observed pattern supports the hypothesis that the fire event partly contributed to localized PAH deposition in the surrounding soil. Despite the absence of higher rates of contamination, it is important to note that PAHs are known for their persistence, bioaccumulation potential, and carcinogenicity. Even low-level chronic exposure can pose long-term ecological and health risks, particularly in sensitive environments, like agricultural land. Studies have shown that PAHs can alter soil microbial communities, reduce enzymatic activity, and impair nutrient cycling [50–52]. Relatively low concentrations observed organic compounds in this research may be attributed to several mitigating factors: the rapid dispersion of combustion byproducts due to meteorological conditions; the limited deposition of particulate-bound pollutants in the immediate vicinity; and soil organic matter content, which can influence the sorption and immobilization of PAHs and other hydrophobic compounds [53].

Although soils exposed to plastic combustion residues are generally expected to exhibit elevated concentrations of certain metals, since fires at plastic recycling facilities can alter the chemical composition of surrounding soils through the release and redistribution of heavy metals, the results of this investigation (Table 6) reveal a different pattern. The content of Cr, Co, Ni, Cu, Zn, As and Pb was higher in soil samples before the fire incident; the content of Co, Ni, Cu and As was significantly higher than the content of those metals

in soil samples collected after the fire. The observed decrease in mean and maximum concentrations of heavy metals in post-fire soil samples, compared to pre-fire samples, is primarily attributed to differences in sampling locations rather than the fire event itself. Pre- and post-fire soils were collected from distinct sites with varying land-use histories and natural mineral compositions. Long-term agricultural practices, such as repeated application of phosphate-based fertilizers, likely contributed to elevated metal levels in some pre-fire areas, while post-fire sites were less intensively managed, resulting in lower baseline concentrations. Although most studies report stable or increased metal levels after fire events due to ash deposition [21–23,54], the lower values observed here are unlikely to result from volatilization or removal, as the fire was short-lived, involved primarily plastic waste with limited metal content, and occurred during dry conditions that minimized ash incorporation into the soil. These findings, supported by ANOVA results, suggest that the heavy metals present are predominantly of geogenic origin, with minimal contribution from the fire itself. This observation aligns with results from a study conducted in Greece [55], where the results of a controlled laboratory experiment in which open fire conditions were simulated at temperatures of 650–750 °C and commercially available polymer materials (PVC, PET, PP, PS, LDPE and HDPE) were burned indicate that residue solid ash produced contained low concentrations of Pb (3.05–64.2 µg/kg), Zn (0.52–13.1 µg/kg), Cr (0.28–9.05 µg/kg), and Ni (0.09–17.2 µg/kg) [55], indicating that ash resulting from the combustion of plastic materials can slightly affect the soil onto which it is deposited. Additionally, significantly lower content of Cu and As in soils after the fire incident can be attributed to the fact that those two metals can volatilize at high temperatures and be carried away from the site [56,57]. But still, it must be highlighted that, when plastics burn, especially those containing additives or pigments, it can release As, Cu, Ni and Co which may originate from flame retardants, stabilizers, or colorants embedded in the plastic matrix [58]. While the concentrations of measured pollutants were generally low, this outcome reflects the specific conditions of the incident (short fire duration, absence of precipitation, and limited dispersion) rather than the irrelevance of the selected parameters. These compounds (PAHs, dioxins, PFAS, PCBs, Cr, Cu, Zn, Ni, Pb, Co and As) remain critical indicators of plastic combustion impacts on soil. Nevertheless, future research should incorporate additional environmental aspects, such as air quality, vegetation response, and soil biological activity, to provide a more comprehensive assessment of the ecological consequences of similar events.

In contrast to metals, the average phosphorus content in soils after the fire (602.1 mg/kg) was 35% higher than the average phosphorus content in soils before the fire (451.4 mg/kg), which partly indicates the influence of ash residues resulting from the combustion event. For example, the ash after the poly(vinyl chloride) combustion contained 77.5 ± 6.1 µg/kg of total phosphorus [55].

4.2. Ecological Risk Assessment of Heavy Metal Content in Soil Before and After the Fire Incident

The PLI provides a single, composite value that reflects the cumulative impact of multiple metals, making it a useful tool for summarizing complex contamination data. The PLI values presented in Figure 4a,b represent the seventh root of the product of seven contamination factor (CF) values calculated for each sampling location before and after the fire incident (Appendix A: Tables A1 and A2). In total, 36 PLI values were calculated. Regardless of the sampling period (pre- or post-fire incident), 20 PLI values exceeded the threshold value of 1 (PLI 1.00–1.26) and indicate moderate pollution ($1 < \text{PLI} < 2$) [59], while the rest calculate PLI values indicate no pollution (PLI 0.82–0.99). Thirteen out of twenty PLI values exceeding the threshold of 1 were recorded in soil samples collected prior to the fire, which further suggests that the fire did not directly influence the metal content in

the soil, nor did it significantly contribute to its contamination. This observation is further supported by the comparison of individual contamination factors calculated for each metal and sampling location, depending on the time of soil sampling (Tables A1 and A2). For comparison of PLI values, although selected studies do not pertain to studies specifically investigating the impact of plastic combustion on soil contamination, a study was conducted on the tributaries of the Tiber River in Rome, where a similar range of PLI values (0.57–1.49) was recorded in sediments, and the influence of various anthropogenic activities on river and sediment pollution was confirmed [60]. Conversely, a study carried out in six Mediterranean lagoons in northern Greece demonstrated that human activities also contributed to the accumulation of metals (As, Cd, Cr, Cu, Mn, Pb, Zn, Ni) in sediments. At 1 of the 21 sampling sites, the PLI value reached 12.35, primarily due to a high contamination factor (CF) for arsenic, which was calculated to be 35.28 [61]. Ecological risk assessment based on PLI values also has some limitations because it does not account for toxicity differences between metals. It treats all metals equally, regardless of their ecological or health impact, and it does not reflect the bioavailability or mobility of metals in the environment, but it is a valuable first-step indicator in environmental monitoring. In contrast, the enrichment factor (EF) is specifically designed to assess the degree of anthropogenic influence on metal concentrations in soil or sediment. EF provides a more nuanced and source-sensitive evaluation, making it particularly valuable in post-fire assessments, urban–industrial environments, and ecotoxicological studies. EF values presented in Figure 5a,b, specifically mean EF values, which ranged from 0.99 to 1.13 before the fire incident and 1.02 to 1.10 after the fire, suggested that the content of these metals was predominantly of geogenic origin, with minimal anthropogenic enrichment, even following the combustion of plastic waste. Detailed values regarding the variability of EF, referring to the sampling locations and temporal distribution (pre- and post-fire incident), reveal higher ranges of EF values. EF values across most sampling locations were within the range of minimal to moderate enrichment ($EF < 2$) (Tables A3 and A4), indicating that the presence of metals was largely of geogenic origin with limited anthropogenic input. Only a few isolated sites exhibited slightly elevated EF values, suggesting localized human influence. For example, EF for Cu content before the fire incident ranged from 0.70 to 3.02, while the post-fire result indicated moderate enrichment in two locations (Ern I and Ivan I) where EF values ranged from 0.45 to 2.09 for Cr content, from 0.62 to 2.45 for Co content, and from 0.77 to 2.36 for Cu content (Tables A3 and A4). This can be partially explained by the fact that all sampled locations are agricultural fields managed under conventional arable farming practices, where mineral nitrogen and phosphorus fertilizers are regularly applied. These fertilizers often contain trace amounts of the metals [62,63], which may contribute to the observed concentrations in the soil. For comparison, in an ecological risk assessment study conducted in the vicinity of a landfill site in Tehran, EF values were reported in the following ranges: 0.6–1.9 for arsenic; 1.1–5.4 for chromium; 2.9–13.0 for copper; 1.9–4.1 for nickel; 2.3–10.8 for lead; 1.5–2.1 for cobalt; 1.9–7.2 for zinc [43].

4.3. Health Risk Assessment of Heavy Metal Content in Soil Before and After the Fire Incident

The deterministic approach to the health risk assessment incorporates the total concentration of metals along with the most probable values of other relevant parameters. However, this method does not account for the variability in risk exposure among different individuals, which can lead to either the overestimation or underestimation of health risks. As a result, the actual risk assessment remains uncertain [64]. Despite these limitations, this model has been widely used in scientific studies to assess human health risks associated with exposure to toxic metals from soil or sediment since 1996. Between 1996 and 2003, an average of approximately 50 studies per year were published using this assessment model. Notably, in the first five months of

2023 alone, 10,911 articles related to this topic were indexed on the Web of Science platform [42]. Results presented in Figure 6a,b reveal that all estimated ILCR values for adults and children remained below the threshold of 1×10^{-4} , indicating the acceptable level of carcinogenic risk under both pre-fire and post-fire environmental conditions. In contrast to these results, studies conducted on soils in an industrial area in China and near a landfill in Iran indicate the much more significant impact of metal content in soils on human health. A carcinogenic risk assessment of exposure to arsenic, cadmium, and zinc in soil near a galvanization facility in China revealed that ingestion posed a significantly higher risk to human health than inhalation or dermal contact [65]. The authors reported that the ILCR value for children due to arsenic exposure in soil was 2.42×10^{-3} , while for adults it was 1.74×10^{-3} , indicating the potential for carcinogenic effects in both groups under long-term exposure. Also, authors reported that zinc was associated with a potential carcinogenic risk in children ($ILCR = 1.11 \times 10^{-4}$), but not in adults ($ILCR = 7.94 \times 10^{-5}$) [65]. Similarly, a carcinogenic risk assessment of soil near a municipal waste landfill in Tehran indicated a potential cancer risk from ingestion for both children and adults ($ILCR_{\text{children}} = 1.1 \times 10^{-4}$; $ILCR_{\text{adults}} = 1.1 \times 10^{-4}$). However, no significant risk was observed from the inhalation of soil particles ($ILCR_{\text{children}} = 1.1 \times 10^{-7}$; $ILCR_{\text{adults}} = 2.1 \times 10^{-7}$) or from the dermal absorption of metals from the soil ($ILCR_{\text{children}} = 4.5 \times 10^{-6}$; $ILCR_{\text{adults}} = 9.3 \times 10^{-6}$) [43]. Additionally, in Tables A5 and A6, carcinogenic risk values (CRi) are presented for each exposure pathway, ingestion, inhalation, and dermal absorption, for both adults and children across all 36 investigated locations for pre- and post-fire incident conditions. The CRi values for ingestion in both adults and children represent the sum of the individual CRi contributions from arsenic, chromium, and lead, as cancer slope factors have been defined for these elements via this exposure route. For dermal absorption, the CRi values include only the contributions from arsenic and chromium, while for inhalation, the CRi values reflect the combined contributions of all four carcinogenic metals (As, Cr, Ni, and Pb). Accordingly, the CRi values for both adults and children regardless of the sampling location and temporal soil sampling followed a decreasing trend by exposure route as follows: CRi (ingestion) > CRi (dermal absorption) > CRi (inhalation) (Tables A5 and A6). For future health risk assessments, it is advisable to adopt a probabilistic approach based on Monte Carlo simulation (MCS). This method accounts for both uncertainty and variability in input parameters, thereby providing more accurate and robust results. Moreover, it enables the identification of the parameters that exert the greatest influence on overall health risk [66]. All together indicates the fire incident in this research was not as negative for the environment and human health as other similar events. In contrast to the findings of this study, previous research has demonstrated notable concerns. For instance, one study reported that wildfire-affected soils exhibited elevated carcinogenic risks due to the increased mobility of toxic metals such as arsenic, cadmium, and nickel, particularly in burned soils where the interaction with PAHs intensified the overall health hazard [22]. Similarly, another investigation conducted in the Mediterranean region identified that wildfire-impacted soils contained high concentrations of lead, copper, and chromium, with inhalation, especially among children, emerging as the primary exposure pathway [26].

4.4. Study Limitations

Several limitations of this study should be acknowledged. First, the control samples and post-fire samples were collected from different locations, which restricted direct site-to-site comparison and introduced variability related to soil heterogeneity and land-use history. Second, the relatively short timeframe between the fire and sampling (3 to 17 days) may not capture potential long-term pollutant migration, degradation, or transformation processes. Third, the absence of precipitation during the post-fire period minimized runoff and leaching, which could otherwise influence contaminant redistribution and accumulation in

lower soil horizons or adjacent ecosystems. Additionally, the study focused exclusively on soil contamination and did not include other environmental compartments such as air, water, or vegetation, which may also be affected by plastic combustion. These limitations should be considered when interpreting the results and highlight the need for integrated, multi-media monitoring and longitudinal studies to fully assess the environmental and health impacts of similar incidents.

5. Conclusions

The conducted study demonstrates that the open-space fire at the plastic waste landfill located within the industrial premises in Osijek, Croatia, did not result in a substantial increase in the concentrations of the analyzed organic pollutants, including the sum of polycyclic aromatic hydrocarbons (PAHs), dioxins, and perfluoro-alkyl substances (PFAS). Furthermore, compared to soil quality before the fire incident, the quality of the investigated agricultural soil remained unchanged after the fire in terms of heavy metal content (Cr, Co, Ni, Cu, Zn, As, Pb), suggesting a predominantly geogenic origin with minimal anthropogenic influence. Given the increasing residential development in the area, particularly among young families with children, and the region's strong orientation toward agricultural production, apiculture, viticulture, and livestock farming, the findings of the health risk assessment are particularly relevant. The results indicate an acceptable level of cancer risk, supporting the conclusion that the fire did not pose a significant long-term threat to human health or soil quality. The broader implications of this research extend beyond the local context. First, the results highlight the resilience of agricultural soils under short-term exposure to plastic combustion by-products, suggesting that immediate large-scale remediation may not be necessary when dispersion conditions are limited. Second, the study underscores the importance of integrating rapid soil monitoring protocols into emergency response plans, as even low-level contamination can accumulate over time with repeated incidents. Third, given the global increase in plastic waste and the frequency of uncontrolled fires, these findings provide a reference framework for assessing soil quality and health risks in similar scenarios. Finally, preventive strategies, such as stricter waste management regulations, community awareness programs, and contingency plans for rapid remediation (including phytoremediation and biochar application) are essential to safeguard agricultural sustainability and public health.

Author Contributions: Conceptualization, I.K.; methodology, I.K. and A.P.; formal analysis, D.R. and M.G.; investigation, I.K. and D.R.; resources I.K.; data curation, A.P.; writing—original draft preparation, A.P. and I.K.; writing—review and editing, I.K., Ž.Z., M.G., D.R. and H.H.; visualization, A.P.; supervision, I.K.; funding acquisition, I.K. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding.

Data Availability Statement: Data are contained within the article.

Acknowledgments: The authors would especially like to thank the Croatian Meteorological and Hydrological Service for providing the climate data disrobed in paragraph "Fire location and climatic condition". The authors would also like to thank the Fire Department of the City of Osijek for providing images of fire (Figure 1).

Conflicts of Interest: The authors declare no conflicts of interest.

Appendix A

Table A1. Contamination factor according to sampling locations and individual metal before the fire incident.

CF	0–10 km Radius								10–20 km Radius									
	Ant 1	Ant 2	Ant 3	Ant 4	Ant 5	Ant 6	Ant 7	Ant 8	Ern 1	Ern 2	Ern 3	Ern 4	Ern 5	Ern 6	Ern 7	Ern 8	Ern 9	Ern 10
Cr	0.54	0.54	0.59	0.65	0.65	0.63	0.52	0.50	0.64	0.57	0.61	0.56	0.56	0.58	0.54	0.63	0.58	0.54
Cu	2.98	2.52	1.84	2.40	2.14	2.47	1.93	2.12	1.54	2.45	2.38	2.19	2.03	1.68	2.10	1.66	2.07	2.17
Zn	0.85	0.94	0.60	0.74	0.81	0.77	0.64	0.69	0.72	0.79	0.75	0.79	0.81	0.60	0.84	0.65	0.67	0.73
Ni	1.80	1.71	1.38	1.65	1.68	1.56	1.53	1.56	1.32	1.65	1.63	1.68	1.36	1.34	1.53	1.29	1.78	1.52
Pb	0.72	0.77	0.86	0.90	0.87	0.77	0.73	0.68	0.77	1.11	0.82	0.77	0.84	0.75	0.83	0.83	0.80	0.83
Cd	1.27	1.67	1.24	2.13	1.81	1.72	1.41	1.32	1.50	1.75	1.70	1.65	1.84	1.88	1.74	1.43	2.00	1.91
As	0.91	0.91	0.84	1.02	0.89	0.76	0.91	0.89	0.93	0.87	0.76	0.73	0.80	0.84	0.82	0.78	0.78	0.76

Table A2. Contamination factor according to sampling locations and individual metal after the fire incident.

CF	0–10 km Radius								10–20 km Radius									
	Ant I	Ant II	Ant II	Ant IV	Ant V	Ant VI	Ant VII	Ant VIII	Ant IX	Ant X	Ern I	Ern II	Ern III	Ern IV	Ivan I	Ivan II	Tenja I	Tenja II
Cr	0.41	0.45	0.43	0.34	0.29	0.34	0.36	0.33	0.36	0.36	0.90	0.82	0.81	0.86	0.63	0.62	0.70	0.82
Cu	2.08	1.92	1.92	2.00	1.69	1.46	1.69	1.54	1.62	1.62	2.23	2.08	2.08	2.00	2.15	2.00	2.23	2.46
Zn	0.76	0.67	0.70	0.77	0.48	0.55	0.68	0.63	0.55	0.65	0.93	0.88	0.98	0.98	0.59	0.67	0.64	0.89
Ni	1.52	1.39	1.52	1.52	1.13	1.13	1.43	1.30	1.26	1.65	1.60	1.47	1.65	1.34	1.43	1.43	1.39	1.39
Pb	0.95	0.91	1.00	0.95	1.00	1.08	0.83	0.95	0.79	1.04	0.75	1.00	0.87	0.87	0.75	0.75	0.66	0.66
Cd	1.63	1.25	1.75	1.50	1.38	1.38	1.50	1.38	1.25	1.25	1.88	1.75	1.75	1.88	1.13	1.25	1.13	1.38
As	0.67	0.92	0.67	0.75	0.67	0.83	0.83	0.67	0.83	0.75	0.83	1.08	0.50	0.92	0.83	0.75	0.50	0.67

Table A3. Enrichment factor according to sampling locations and individual metal before the fire incident.

EF	0–10 km Radius								10–20 km Radius									
	Ant 1	Ant 2	Ant 3	Ant 4	Ant 5	Ant 6	Ant 7	Ant 8	Ern 1	Ern 2	Ern 3	Ern 4	Ern 5	Ern 6	Ern 7	Ern 8	Ern 9	Ern 10
Cr	0.55	1.09	1.04	1.11	1.02	0.97	0.82	0.97	1.24	0.94	1.08	0.89	1.04	1.02	0.92	1.22	0.85	1.01
Cu	3.02	0.92	0.70	1.31	0.91	1.15	0.77	1.11	0.70	1.67	0.98	0.89	0.96	0.82	1.23	0.84	1.15	1.13
Zn	0.85	1.22	0.61	1.24	1.11	0.95	0.82	1.10	1.00	1.16	0.96	1.01	1.06	0.73	1.38	0.82	0.95	1.18
Ni	1.82	1.04	0.77	1.21	1.04	0.93	0.97	1.03	0.82	1.31	0.99	0.99	0.85	0.97	1.13	0.89	1.27	0.92
Pb	0.73	1.18	1.06	1.05	0.99	0.89	0.93	0.94	1.11	1.50	0.75	0.91	1.13	0.87	1.09	1.06	0.89	1.12
Cd	1.29	1.43	0.71	1.72	0.86	0.95	0.81	0.96	1.10	1.22	0.98	0.93	1.16	1.01	0.91	0.87	1.29	1.03
As	0.92	1.09	0.89	1.22	0.89	0.85	1.19	0.99	1.02	0.98	0.88	0.93	1.13	1.04	0.96	1.00	0.92	1.05

Table A4. Enrichment factor according to sampling locations and individual metal after the fire incident.

EF	0–10 km Radius								10–20 km Radius									
	Ant I	Ant II	Ant II	Ant IV	Ant V	Ant VI	Ant VII	Ant VIII	Ant IX	Ant X	Ern I	Ern II	Ern III	Ern IV	Ivan I	Ivan II	Tenja I	Tenja II
Cr	0.45	1.06	0.97	0.80	0.77	1.20	0.98	0.90	1.20	1.01	0.95	0.90	1.08	1.08	2.09	0.95	1.15	1.16
Cu	0.97	0.90	1.01	1.04	0.77	0.88	1.08	0.90	1.14	1.01	2.36	0.92	1.09	0.98	1.58	0.91	1.13	1.10
Zn	0.73	0.86	1.06	1.10	0.56	1.16	1.17	0.91	0.95	1.20	0.98	0.93	1.22	1.01	1.08	1.11	0.96	1.38
Ni	0.86	0.89	1.11	1.00	0.67	1.02	1.18	0.90	1.05	1.01	1.74	0.96	1.00	1.13	1.27	1.04	1.01	0.96
Pb	1.03	0.93	1.10	0.96	0.95	1.10	0.72	0.89	1.39	0.83	1.10	0.71	1.46	0.89	1.31	0.84	0.62	1.45
Cd	0.62	0.65	1.01	1.00	0.91	1.02	0.93	1.49	0.73	1.01	2.45	1.10	0.99	0.98	0.87	1.23	1.08	1.07
As	0.68	1.34	0.74	1.13	0.81	1.27	0.93	0.80	1.36	0.91	0.88	1.29	0.50	1.86	1.32	0.88	0.67	1.33

Table A5. Carcinogenic risk values (CRi) for each exposure pathway for adults and children before the fire incident.

Location	Adults			Children		
	Ingestion	Inhalation	Dermal	Ingestion	Inhalation	Dermal
0–10 km radius from the fire site						
Ant 1	1.53×10^{-5}	1.06×10^{-7}	3.62×10^{-6}	3.56×10^{-5}	4.71×10^{-8}	7.27×10^{-7}
Ant 2	1.52×10^{-5}	1.06×10^{-7}	3.60×10^{-6}	3.55×10^{-5}	4.68×10^{-8}	7.22×10^{-7}
Ant 3	1.53×10^{-5}	1.12×10^{-7}	3.87×10^{-6}	3.57×10^{-5}	4.96×10^{-8}	7.76×10^{-7}
Ant 4	1.76×10^{-5}	1.25×10^{-7}	4.29×10^{-6}	4.11×10^{-5}	5.53×10^{-8}	8.61×10^{-7}
Ant 5	1.65×10^{-5}	1.24×10^{-7}	4.28×10^{-6}	3.86×10^{-5}	5.49×10^{-8}	8.59×10^{-7}
Ant 6	1.51×10^{-5}	1.18×10^{-7}	4.10×10^{-6}	3.52×10^{-5}	5.23×10^{-8}	8.23×10^{-7}
Ant 7	1.49×10^{-5}	1.02×10^{-7}	3.48×10^{-6}	3.49×10^{-5}	4.52×10^{-8}	6.97×10^{-7}
Ant 8	1.44×10^{-5}	9.76×10^{-8}	3.32×10^{-6}	3.37×10^{-5}	4.33×10^{-8}	6.67×10^{-7}

Table A5. Cont.

Location	Adults			Children		
	Ingestion	Inhalation	Dermal	Ingestion	Inhalation	Dermal
10–20 km radius from the fire site						
Ern 1	1.67×10^{-5}	1.22×10^{-7}	4.21×10^{-6}	3.90×10^{-5}	5.39×10^{-8}	8.44×10^{-7}
Ern 2	1.52×10^{-5}	1.09×10^{-7}	3.76×10^{-6}	3.56×10^{-5}	4.85×10^{-8}	7.54×10^{-7}
Ern 3	1.48×10^{-5}	1.15×10^{-7}	3.98×10^{-6}	3.45×10^{-5}	5.08×10^{-8}	7.97×10^{-7}
Ern 4	1.40×10^{-5}	1.07×10^{-7}	3.71×10^{-6}	3.28×10^{-5}	4.75×10^{-8}	7.44×10^{-7}
Ern 5	1.46×10^{-5}	1.07×10^{-7}	3.71×10^{-6}	3.40×10^{-5}	4.76×10^{-8}	7.45×10^{-7}
Ern 6	1.53×10^{-5}	1.12×10^{-7}	3.86×10^{-6}	3.56×10^{-5}	4.95×10^{-8}	7.74×10^{-7}
Ern 7	1.45×10^{-5}	1.05×10^{-7}	3.60×10^{-6}	3.39×10^{-5}	4.64×10^{-8}	7.22×10^{-7}
Ern 8	1.53×10^{-5}	1.18×10^{-7}	4.11×10^{-6}	3.56×10^{-5}	5.22×10^{-8}	8.24×10^{-7}
Ern 9	1.47×10^{-5}	1.11×10^{-7}	3.83×10^{-6}	3.42×10^{-5}	4.92×10^{-8}	7.69×10^{-7}
Ern 10	1.39×10^{-5}	1.04×10^{-7}	3.58×10^{-6}	3.25×10^{-5}	4.59×10^{-8}	7.17×10^{-7}

Table A6. Carcinogenic risk values (CRI) for each exposure pathway for adults and children after the fire incident.

Location	Adults			Children		
	Ingestion	Inhalation	Dermal	Ingestion	Inhalation	Dermal
0–10 km radius from the fire site						
Ant I	1.14×10^{-5}	8.02×10^{-8}	2.73×10^{-6}	2.66×10^{-5}	3.56×10^{-8}	5.48×10^{-7}
Ant II	1.40×10^{-5}	8.90×10^{-8}	3.01×10^{-6}	3.26×10^{-5}	3.95×10^{-8}	6.03×10^{-7}
Ant II	1.16×10^{-5}	8.31×10^{-8}	2.84×10^{-6}	2.71×10^{-5}	3.69×10^{-8}	5.70×10^{-7}
Ant IV	1.12×10^{-5}	6.96×10^{-8}	2.32×10^{-6}	2.60×10^{-5}	3.09×10^{-8}	4.66×10^{-7}
Ant V	9.74×10^{-6}	5.94×10^{-8}	1.98×10^{-6}	2.27×10^{-5}	2.63×10^{-8}	3.97×10^{-7}
Ant VI	1.19×10^{-5}	7.02×10^{-8}	2.34×10^{-6}	2.77×10^{-5}	3.11×10^{-8}	4.70×10^{-7}
Ant VII	1.21×10^{-5}	7.35×10^{-8}	2.45×10^{-6}	2.82×10^{-5}	3.26×10^{-8}	4.91×10^{-7}
Ant VIII	1.02×10^{-5}	6.54×10^{-8}	2.20×10^{-6}	2.38×10^{-5}	2.90×10^{-8}	4.40×10^{-7}
Ant IX	1.21×10^{-5}	7.32×10^{-8}	2.45×10^{-6}	2.82×10^{-5}	3.25×10^{-8}	4.91×10^{-7}
Ant X	1.14×10^{-5}	7.22×10^{-8}	2.43×10^{-6}	2.65×10^{-5}	3.20×10^{-8}	4.87×10^{-7}
10–20 km radius from the fire site						
Ern I	1.95×10^{-5}	1.65×10^{-7}	5.83×10^{-6}	4.55×10^{-5}	7.33×10^{-8}	1.17×10^{-6}
Ern II	2.05×10^{-5}	1.55×10^{-7}	5.38×10^{-6}	4.78×10^{-5}	6.86×10^{-8}	1.08×10^{-6}
Ern III	1.55×10^{-5}	1.46×10^{-7}	5.19×10^{-6}	3.61×10^{-5}	6.46×10^{-8}	1.04×10^{-6}
Ern IV	1.97×10^{-5}	1.60×10^{-7}	5.60×10^{-6}	4.59×10^{-5}	7.08×10^{-8}	1.12×10^{-6}
Ivan I	1.58×10^{-5}	1.20×10^{-7}	4.17×10^{-6}	3.70×10^{-5}	5.31×10^{-8}	8.36×10^{-7}
Ivan II	1.49×10^{-5}	1.16×10^{-7}	4.04×10^{-6}	3.48×10^{-5}	5.14×10^{-8}	8.10×10^{-7}
Tenja I	1.39×10^{-5}	1.27×10^{-7}	4.52×10^{-6}	3.25×10^{-5}	5.65×10^{-8}	9.06×10^{-7}
Tenja II	1.70×10^{-5}	1.50×10^{-7}	5.31×10^{-6}	3.97×10^{-5}	6.64×10^{-8}	1.06×10^{-6}

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