The critical factors of landfill fire impact on air quality

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Abstract

Simulations show that landfill fires are a serious regional concern. This study presents a three-parameter method for assessing the impact of landfill fires on the atmosphere based on landfill fire data in Poland from 2018. HYSPLIT simulations were performed to assess the spatial impact of the particulate matter (PM) <10 \(\mu\)m (PM\(_{10}\)) emitted during each fire in regions with elevated concentrations of PM\(_{10}\). Additionally, for each fire, the population exposed to an increased PM\(_{10}\) concentration due to the landfill fire was determined. Three parameters, namely the area and range of the PM concentrations and the population affected, were calculated for each additional concentration level and analyzed using geodetic information software. The analysis of these three parameters shows that despite being correlated for concentrations of 10 and 1 \(\mu\)g m\(^{-3}\), there are no equations that can eliminate any of the three parameters, and all of them are required for the proper assessment of landfill fire impacts on air quality. Choosing the arbitrary reference concentration levels in the assessment shows that our method is adaptable for any law, health regulation, or recommendation.

1. Introduction

Landfill fires are a public health concern linked to waste storage and disposal. They can be caused either by spontaneous processes that lead to ignition or intentional actions [1–5]. In Poland, the annual incidence of largest landfills and waste storage yard fires more than tripled from 23 in 2010 to 79 in 2018 [6]. This increase also coincides with the introduction of new regulations on waste and waste packing management given by the European Parliament and Council Directive 94/62/WE [7].

The direct impact of landfill fires is easy to observe—the life of workers of facilities is endangered and the facility objects are burned, causing measurable financial losses. Moreover, the decline in environment quality at the fire site can be estimated [8–10]. However, many indirect environmental impacts [11] caused by these fires over wider temporal and a spatial scales, are more difficult to evaluate than the direct impacts [12]. One of the most significant indirect impacts is the change in air quality due to the emission of various air pollutants during a fire. Fires involving plastic should be carefully analyzed because, during their combustion and pyrolysis, many substances with negative health impacts, such as dioxins, furans, and polychlorinated biphenyls, are emitted [13].

The qualitative and quantitative assessment of the impact of waste fires on the environment, especially air quality, is crucial for the introduction of bills and regulations that will curtail these fires in the future. One of the challenges involved in environmental impact assessment and writing legislation is defining the crucial indicators that will be used in the assessments of landfill fires. The most important indicators are mass and toxicity of the substances emitted during the fire; however, it is challenging to determine which substances should be selected for analysis. We contend that one of the measurable impacts of waste fires on air quality may be the concentration of particulate matter (PM). In case of PM, the emitted mass and related concentration characterize the environmental and health impacts of
emission since any increase in concentration has been shown to have negative impact on plant [14, 15] and human health [16, 17]. The emission factors (EFs) for open burning have been measured for many chemical substances and for multiple materials in different experiments, and hence, the obtained values have to be merged and curated as in [18]. In general, EFs of over 160 hazardous air pollutants from the open burning of different materials are known [19]. The effective assessment of fire impact requires a factor that is less material specific, and is emitted during all fires. PM is emitted during the burning of any type of material. Hence, we considered all the critical factors assessing landfill fires in the context of the concentration of PM$_{10}$. The air quality in Poland is one of the worst in the European Union [20], which makes the impact of landfill fires on the concentration of PM$_{10}$ even more significant in the context of differences in air quality across Europe.

The second critical factor that we propose is the size of the fire, however, not as the dimensions of the burning object, which is how raw reports classify fires [21], nor the area covered by heat spots which is how fires are classified by satellite observations [22], since these both are strongly affected by the spatial distribution of fuel across a burning site [23]. The proposed factor is to determine size of the fire by the dimensions of the plume and the distance the plume can travel [18, 24]. Thus, such size can be treated as a quantitative measure of the indirect impact of the fire. The dispersion of particles emitted into the atmosphere are usually calculated using meteorological data by employing a software package designed to calculate the dispersion [25, 26]. The analysis of the dispersion requires defining areas—polygons—where landfill fires caused an increase in the average concentration of PM$_{10}$. Waste fires cause a high short-term increase in concentration. The shortest averaging period, which is mentioned in the European directive on ambient air quality and cleaner air for Europe [27], is 1 h and in our study, we used this period. The total impact range can be evaluated as the sum of polygons with an increase in the 1 h average concentration of PM$_{10}$. Since the shape of a union of polygons can be complex, for example, due to meteorological conditions [28], the spatial impact cannot be simply defined by the area of the union set. Spatial impact should also include the distance between the farthest point of these polygon and fire. These two parameters together can better define the longitudinal and transverse diffusion of the plume and describe whether it is oblong or more circular and whether it affects a limited area or points located far away.

The public health aspects of landfill fires are crucial. Since we chose an increase in the concentration of PM$_{10}$ as a measure of air quality change caused by landfill fires, the factor that can assess the public health impact is the number of people who are exposed to the PM$_{10}$ emitted during landfill fires. Based on the census data, we determined the exact number of people living in the regions defined as the spatial impact zone of landfill fires. This number can indicate whether landfill fires should be considered as one of the major health risks at the scale of community, province, state, or continent.

The emission of air pollutants from landfill fires is often discussed as a recognized problem, while there are only a few studies that have evaluated emissions quantitatively [18, 29, 30]. The aim of this study was to present possibilities for the description of the range and scale of the impact of landfill fires in Poland during one year with the three aforementioned critical factors: increase in PM$_{10}$ concentration, size of impact plume, and number of people exposed. Since the critical factors are not limited to only one country, the impact on the entire population was evaluated, instead of the population of a single country. The methodology presented in this study can be adapted, depending on the data availability, computational resources, and assumed uncertainties, and applied to any pollutant in any region of the world.

2. Materials and methods

2.1. Evidence of fires and air pollutant emission

The National Firefighting and Rescue System (polish abbreviation KSRG) units report all rescue incidents in the national database SWD-ST [21]. The report of each incident contains information about the object (e.g. location, dimensions) and fire (e.g. duration, size category, used resources). Since the largest fires have the highest impact on the air quality, we analyzed only fires burning areas larger than 301 m$^2$ or volumes larger than 1501 m$^3$. These values are based on SWD regulations, which categorize fires into large and very large categories [21], and there were a total of 79 large and very large fires in 2018.

The emissions during landfill fires can be estimated based on the dimensions and mass of a burned object. The methodology used in this study for the assessment of total emissions based on the sum of the emissions in the individual fires was previously described in [18].

2.2. Dispersion of PM$_{10}$ in the atmosphere

The critical factors of waste fires, which were described in the current study, were evaluated using the values of the mass of PM$_{10}$ emitted during individual fires. Computational models were designed to evaluate the dispersion of the plume emitted from a fire based on the meteorological conditions [26]. In our work, we decided to use HYSPLIT [31, 32], both the PC-based version and the READY interface [33], and this choice was based on the evaluation of dispersion models [25] while also considering the availability of suitable computing resources.
Based on the data from the evidence of waste fires in Poland, the dispersion of PM$_{10}$ was calculated for each fire. Although the fires took place in Poland, we did not limit our simulation domain to Poland because air pollutants are not restricted by any administrative border. We limited our study to one year, fires that occurred in 2018. For each fire, the reported geographical coordinates of fires were used as source locations. The time when the alarm was sounded at emergency, as indicated SWD-ST and the end of rescue operation time were assumed to be the beginning and end of emissions, respectively. The bottom of the emission was assumed to be zero meters above ground level, and the top was the height of the burned object. The mass of PM$_{10}$ released in a fire was calculated according to [18]. The dispersion of the plume was calculated for 12 h after the end of the fire. The concentration of PM$_{10}$ emitted during fires was averaged every hour after the fire began. Owing to the limitations of HYSPLIT and the complexity of extinguishing landfill fires, two additional assumptions were introduced, which were the averaged vertical concentrations and emission rates.

HYSPLIT limits the minimum concentration layer calculation to 0–100 m above ground level, and we performed all calculations in this layer. Studies of the vertical profile of concentrations [34, 35] have shown that the variability of the concentration in this layer can be assumed to be constant. Due to the potentially high variability in convective and radiative heat transfer rates [36, 37] as well as heat release rate in pyrolysis [38–40], we intentionally did not introduce heat plume equations for the convection of mass released from fires or the heat release rate of entire fires because it would cause large uncertainties in the calculations and results. Since every waste fire is different, the burning processes occur at different moments, and there are no data about fires minute by minute; we assumed a constant emission rate of PM$_{10}$ during the fire. HYSPLIT calculations and the HYSPLIT concplot$^3$ program provided polygons with an increase in 1 h average concentrations of 1, 10 µg m$^{-3}$, 100, 1000, and 10 000 µg m$^{-3}$ for each hour after fire initiation. To present the entire scope of applicability and possible adaptation, we did not limit our calculations to particular concentrations related to local or regional law regulations. They can be analyzed in geodetic information software (GIS)—in our work we used QGIS 3.10 LTR [41]. QGIS allows the calculation of critical factors related to waste fires. The area of the polygons representing the union of polygons of 1 h average concentrations can be calculated for respective concentrations using geoprocessing tools. For each polygon, we measured the distance between the farthest points: the maximum distance between the source and edge points, and used this value as the second critical factor.

The third critical factor—the population—requires adding gridded population data to the analysis. The 1 km grid resolution data on the population of Poland were taken from the last census [42], and for other countries, data for 2018 [43] at the same spatial resolution were used. We calculated the number of people within each polygon, representing the union of a 1 h average increase in concentration. To ensure that we did not overestimate the population living in each grid cell, we only counted whole grid cells inside the polygons. The number we received was not double-counting the population, that is, this population value described the number of people living in an area where there was an increase of 1 h average PM$_{10}$ concentration that lasted at least 1 h. If the increase in concentration was persisting there more than 1 h, the population value was not increased.

The calculation of critical factors cannot be evaluated without estimating uncertainty. The first source of uncertainty was the data about landfill fires. The dimensions of burned objects were not always present or correct [18] which makes it difficult to evaluate the amount of PM$_{10}$ emitted during a specific fire. The methodology for evaluating the uncertainties of emissions and their sensitivity to the most uncertain parameters of landfill fires are discussed in [18] and in the supplementary material of this publication. The uncertainty of the area and distance is related to the settings of the HYSPLIT model. The dispersion in HYSPLIT is strongly dependent on the resolution of meteorological data. For the United States, high-resolution meteorological grids are available (HRRR, High-Resolution Rapid Refresh, with a resolution of approximately 3 km [44]), while the highest resolution data covering Poland and neighboring countries in 2018 was GDAS 0.5° (Global Data Assimilation System) [44]. In our work, we used GDAS 0.5°, and we are aware that it is an important source of uncertainty. Since HYSPLIT uses the geographical coordinates of single-precision, the uncertainty of the horizontal dimensions of the object is of the same order as the coordinate uncertainty. The exact height of the emission is not available, and hence, the uncertainty of the emitter’s position equals the object height. To reduce the impact of uncertainty in the object height, a linear source from 0 m to the height of the object was assumed in the calculations. To reduce the uncertainty of the HYSPLIT grid, the maximum particle limit and the number of particles were tested for each simulation using HYSPLIT internal special run ‘Test Input’ to meet the highest quality of results in a 0.05° grid. The uncertainty in population grid data is related to the movement of the population (commuting or migration) and to fraud during the census.

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$^3$ Concplot program is part of the HYSPLIT package that converts the binary output of HYSPLIT into graphical and GIS formats.
2.3. Dispersion

We tested whether the proposed critical factors are independent variables or values any of these three can be evaluated from the other. Without loss of generality, we assumed that the independent parameter was the area covered by plume \( A \) for the respective concentrations because it was the first parameter determined in the assessment of dispersion in GIS. The dependent parameters were the distance to the furthest point of the plume with a given concentration \( d \) and the number of people living in this region \( n \). We propose that both distance \( d \) in m and number of people \( n \) are power functions of \( A \) expressed in m\(^2\):

\[
\begin{align*}
  d &= \alpha \cdot A^\beta \quad (1) \\
  n &= \gamma \cdot A^n. \quad (2)
\end{align*}
\]

As a measure of the correlation between parameters, we used the coefficient of determination \( R^2 \) \[45\] between the \( A \) and \( d \) or \( n \). We evaluated relationships equations (1) and (2) for every concentration range separately, i.e. we tested ten equations in total for \( d_1, n_1, d_0, n_0, \ldots \).

In equation (1) the exponent \( \beta \) describes the power relationship between area and distance. Depending on the shape \( \beta \) can have different values: for example, for isotropic two dimensional dispersion, when regions with concentration increase are circles the distance to the furthest point is the radius and \( \beta = 0.5 \). In cases when region is elliptical with the emitter at one ellipse vertex the \( \beta = 1 \).

The behavior of exponent \( \delta \) in equation (2) is more complex and dependent on urban and rural settlement locations. However, one special value can be distinguished: \( \delta = 1 \) which corresponds to a constant population density. It is expected that in case of a larger area of dispersion, the population density will be averaged over this area and \( R^2 \) will be higher for bigger areas (those with low increase in concentration).

3. Results

We analyzed the dispersion of PM\(_{10}\) in all 79 large and very large fires that took place in 2018 in Poland. Figure 1 shows the dispersion of PM\(_{10}\) calculated using HYSPLIT for a specific fire. For each hour after the fire began, the regions with an increase in the 1 h average concentration of PM\(_{10}\) were determined. The aggregated regions of increase >1 \( \mu g \) m\(^{-3}\), >10 \( \mu g \) m\(^{-3}\), >100 \( \mu g \) m\(^{-3}\), >1000 \( \mu g \) m\(^{-3}\), and >10000 \( \mu g \) m\(^{-3}\) were aggregated and plotted in this order on a contour map of borders. The described fire was a fire of a scrap in the West Pomeranian Voivodeship (Province) in Poland on 20 September 2018. The wind conditions on that day caused the dispersion of PM\(_{10}\) to the north.

Based on the presented polygons, the three critical parameters of this landfill fire were determined and are summarized in table 1.

The fire presented in table 1 and figure 1 was located in the suburbs of Szczecin (the 7th most populated city in Poland). This is reflected in the high number of people living in close proximity to the landfill fire (concentrations of PM\(_{10}\) >10 000 \( \mu g \) m\(^{-3}\) and >1000 \( \mu g \) m\(^{-3}\)). The number of people affected with lower concentrations of PM\(_{10}\) (>100 \( \mu g \) m\(^{-3}\), >10 \( \mu g \) m\(^{-3}\), and >1 \( \mu g \) m\(^{-3}\)) was high in these regions owing to atmospheric transport of these areas to Scandinavia.

A similar analysis was performed for all 79 landfill fires in Poland in 2019. The results are presented in supplementary material (available online at stacks.iop.org/ERL/16/104026/mmedia). The determined values of the critical parameters for the set of all analyzed fires are presented in figures 2–4. For all fires, there were regions obtained with HYSPLIT, where a 1 h averaged increased PM\(_{10}\) concentration of at least 1 \( \mu g \) m\(^{-3}\) was found. Depending on the fire, the area experiencing an increase of 1 \( \mu g \) m\(^{-3}\) in PM\(_{10}\) concentration varied between approximately 8 km\(^2\) and more than 9 \( \times 10^5 \) km\(^2\). For 75 fires, there were regions that experienced a minimum concentration increase of 10 \( \mu g \) m\(^{-3}\), in areas between 5 and 1.8 \( \times 10^5 \) km\(^2\). For 43 fires, there were regions which experienced a 1 h average concentration of 100 \( \mu g \) m\(^{-3}\) in the area between 0.004 and almost 1.3 \( \times 10^5 \) km\(^2\). The six fires caused a 1000 \( \mu g \) m\(^{-3}\) increase in regions between almost 8 and more than 1.6 \( \times 10^3 \) km\(^2\) in area. For two fires, the calculated concentrations exceeded 10 000 \( \mu g \) m\(^{-3}\), over an area of 6 and 10 km\(^2\). Figure 2 presents the distribution of areas where the 1 h average concentration of PM\(_{10}\) increased by 1, 10, 100, and 1000 \( \mu g \) m\(^{-3}\) due to landfill fire. Since a concentration of 10 000 \( \mu g \) m\(^{-3}\) was found only in two fires, hence a boxplot was not plotted for it. In figures 2–4, the boxplot with whiskers extended from the minimum to the maximum values (none of the points were treated as outliers); the box represents the interquartile range, and the median area is presented as a horizontal line.

Figure 3 presents the distance from the landfill fire to the furthest point, where the 1 h average concentration increased by at least 1, 10, 100, and 1000 \( \mu g \) m\(^{-3}\), that is, at this distance, the concentration was equal to these values. The maximum distance, where the increase of concentration was equal to 10 000 \( \mu g \) m\(^{-3}\) in two fires was equal to 7 and 8.5 km; for 1000 \( \mu g \) m\(^{-3}\), the distance varied from 2.5 to 89 km. The distance with the concentration of 100 \( \mu g \) m\(^{-3}\) varied from 0.3 to 323 km, and for 10 \( \mu g \) m\(^{-3}\) it was between 4.7 km and more than 1560 km and for 1 \( \mu g \) m\(^{-3}\) from 3 km to more than 1800 km.

Figure 4 presents the number of people who were exposed to an additional 1 h average concentration
Figure 1. The dispersion of pollutants from the landfill fire in the West Pomerania on 20 September 2018. The regions with respective increase of 1 h average concentrations of PM10 for every hour were aggregated and presented on contour map of Europe. The regions with lower concentrations are drawn behind regions with higher concentrations.

Table 1. Three critical parameters \((A,d,n)\) of the landfill fire in the West Pomerania on 20 September 2018.

| Area \(A\) (1000 km\(^2\)) | Furthest point \(d\) (km) | Number of people \(n\) (1000) |
|-----------------------------|--------------------------|-------------------------------|
| \(>10 000 \mu g m^{-3}\)   | 0.01                     | 7                            |
| \(>1000 \mu g m^{-3}\)     | 1                        | 89                           |
| \(>100 \mu g m^{-3}\)      | 12                       | 323                          |
| \(>10 \mu g m^{-3}\)       | 49                       | 540                          |
| \(>1 \mu g m^{-3}\)        | 163                      | 1032                         |

Although it is expected that the distributions in figure 4 should resemble the distributions in figure 2 because the population density in the area covered by the plume is averaged, a direct comparison of box-plots shows that the interquartile range is wider (in terms of decades) than in figure 2. The number of people exposed to more than 10 000 \(\mu g m^{-3}\) increase in the 1 h average concentration was equal for one fire of 13 000 and for the other of 23 000. In the case of 1000 \(\mu g m^{-3}\), the number of people varied from 2000 to more than 1.3 million per fire. Between 47 and 4.5 million people were exposed to a 100 \(\mu g m^{-3}\) increase in 1 h average concentration from a single fire. The increase in the concentration of more than 10 \(\mu g m^{-3}\) influenced between 5000 and 36.4 million people per fire. For 1 \(\mu g m^{-3}\), this number varied from 4000 to 105 million people per fire.
The distribution of area $A$ (km$^2$) with the increase of the concentration of PM$_{10}$ by 1000 $\mu$g m$^{-3}$ ($A_{1000}$), 100 $\mu$g m$^{-3}$ ($A_{100}$), 10 $\mu$g m$^{-3}$ ($A_{10}$), and 1 $\mu$g m$^{-3}$ ($A_1$). The orange line represents the median, the box represents the interquartile range, the whiskers are from the minimum to maximum (no data are treated as outliers).

The distribution of distance $d$ (km) to the furthest point with the increase of the concentration of PM$_{10}$ by 1000 $\mu$g m$^{-3}$ ($d_{1000}$), 100 $\mu$g m$^{-3}$ ($d_{100}$), 10 $\mu$g m$^{-3}$ ($d_{10}$), and 1 $\mu$g m$^{-3}$ ($d_1$). The orange line represents the median, the box represents the interquartile range, the whiskers are from the minimum to maximum (no data are treated as outliers).

The distribution of population size $n$ living in area with an increase of the concentration of PM$_{10}$ by 1000 $\mu$g m$^{-3}$ ($n_{1000}$), 100 $\mu$g m$^{-3}$ ($n_{100}$), 10 $\mu$g m$^{-3}$ ($n_{10}$), and 1 $\mu$g m$^{-3}$ ($n_1$). The orange line represents the median, the box represents the interquartile range, the whiskers are from the minimum to maximum (no data are treated as outliers).

4. Discussion

In 2018, 79 landfill fires in Poland caused additional exposure to PM$_{10}$ concentration 10 000 $\mu$g m$^{-3}$ for more than 35 000 people in total, 1000 $\mu$g m$^{-3}$ for more than 1379 000 people, of which 32% were exposed to plumes from more than one landfill fire, 100 $\mu$g m$^{-3}$ to more than 9900 000 people (of whom 25% experienced it more than once), 10 $\mu$g m$^{-3}$ to more than 80 million people (of whom 40% experienced it more than once), and 1 $\mu$g m$^{-3}$ to more than 389 million people (of whom 69% experienced it more than once). The plumes with concentrations of 10 000, 1000, 100, 10, and 1 $\mu$g m$^{-3}$ covered a total of 17, 5666, 59 000, 641 000, and 2.306 million km$^2$, respectively (figure 5).

For two fires, there were regions where a waste fire caused an increase in the concentration of PM$_{10}$ of more than 10 mg m$^{-3}$. These fires had the first and second largest emissions of all fires in 2018. The first fire occurred in a rubber recovery facility on 27 May 2018. This fire continued for almost 90 h and covered 1 ha on which tires, rubbers, plastics, and synthetic fibers were stored. The dispersion of PM$_{10}$ emitted during the fire is shown in figure S31. The fire occurred between two densely populated areas in the Upper Silesia Urban Area (west of the fire location) and the Kraków Metropolitan Area (east of the fire location), with a total residential population of around six million residents [46]. These are regions with poor air quality [47, 48] and there have been many studies documenting air quality and trying to identify the origin of pollutants [49–52]. Typically, at Katowice, the wind is from west-northwest to south-southwest [53] and the plumes from fires in this region would disperse toward the Kraków metropolitan area; however, at the time of the fire, the wind blew in the opposite direction, E to SSE, causing an increase in the concentration of PM$_{10}$ in the Upper Silesia Urban Area. The meteorological conditions were as follows: regions with concentrations higher than 1 $\mu$g m$^{-3}$ were in Great Britain or Norway (figure S31).

The second fire was a fire at a scrapyard, which occurred on 20 September 2018, and continued for more than 18 h (figure 1). The fire was located in a Szczecin metropolitan area [46]. This area is rather sparsely populated, with more than 600 000 people living there. The main material stored in the scrapyard was waste electrical and electronic equipment. The fire also covered the infrastructure of the waste (metal shredders). Similarly, as in the case discussed above, the dispersion of the plume was northwards (figure 1), while the typical wind direction at Szczecin was west-northwest to south-southwest [54]. Our simulation indicated that the PM$_{10}$ plume covered Scandinavian countries, indicating the impact of the
sea and vertical dispersion. There are two air quality monitoring stations in Szczecin; however, one of them is located south of the fireplace, and the second is located north-west, that is, slightly offwind. This station had limited capabilities at that time and measured only PM$_{2.5}$ with a 1 h average resolution. Importantly, according to the Measurement Data Archive of the Chief Inspectorate for Environmental Protection during the fire, the highest concentration of PM$_{2.5}$ since 9 April [55] was registered. We eliminated temperature as a contributing factor to PM$_{2.5}$ concentration because the daily temperature did not differ from neighboring days [56].

We tested whether distance and number of people could be expressed as a function of area according to equations (1) and (2), respectively. We found that only for concentrations of 10 and 1 µg m$^{-3}$, equations (1) and (2) were well fitted. For concentrations of 100 and 1000 µg m$^{-3}$, $R^2$ were low and uncertainties of fitting parameters $\Delta \alpha, \Delta \beta, \Delta \gamma,$ and $\Delta \delta$ were markedly larger than the values of these parameters. Parameters for corresponding equations with the coefficient of determination, $R^2$, are presented in table 2.

The value of exponents $\beta$ in the equation for distance were close to 0.5, and were similar for both concentration thresholds. This indicates that distance is proportional to the square root of the area. The values of exponents $\delta$ in the equation for $n$ were slightly lower than 1.0. As expected, the value of $\delta$ for larger areas, that is, for regions with PM$_{10}$ concentration of 1 µg m$^{-3}$, was 0.92 ± 0.04, which is closer to 1.0, and this was more precisely determined than that for regions with a PM$_{10}$ concentration of 10 µg m$^{-3}$, which always have smaller areas. Similarly, $R^2$ value for $n_1$ is higher than for $n_{10}$. Exponents $\beta$ and $\delta$ were well defined, with small uncertainties, in addition fit was good because the $R^2$ value was

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**Figure 5.** Spatial distribution of regions with the increase of concentration PM$_{10}$ caused by the landfill fires which took place in Poland in 2018. The layers are not transparent and are drawn in order from highest concentration (top layer) to lowest (bottom).
high. However, the values of the multiplicative constant $\alpha$ had large uncertainties and the constant $\gamma$ was, in fact, undefined. One can try to determine the values $d_{10}, n_{10}, d_1, n_1$ with limited GIS analysis, according to the area and fitted equations (1) and (2). However, due to the above-discussed limitations, they can be used successfully only for determining relative values of $d$ and $n$, that is, if the plume of concentrations of $\text{PM}_{10} > 10\, \mu\text{g m}^{-3}$, will cover an area $A_{10}'$ four times bigger than the area of specified fire $A_{10}$ ($A_{10}' = 4 A_{10}$), then the distance to the furthest point is around two times larger than for the specified fire $\left(d_{10}' = \left(\frac{A_{10}'}{A_{10}}\right)^{0.52}\approx 2.06\right)$ and then the number of people is 3.2 times bigger than for specified fire $\left(n_{10}' = \left(\frac{A_{10}'}{A_{10}}\right)^{0.84}\approx 3.20\right)$.

For regions with higher $\text{PM}_{10}$ concentrations (10 000, 1000, and 100 $\mu\text{g m}^{-3}$), the relationship between area $A$ and the distance $d$ or population $n$ was difficult to define because the coefficients of determination were lower than 0.6; hence, we did not consider the results of fitting as applicable. There are two possible explanations for this situation.

(a) The timescale of travel for high $\text{PM}_{10}$ concentrations (10 000, 1000, and 100 $\mu\text{g m}^{-3}$) along the wind direction was comparable with the timescale of transverse and vertical diffusion [57]. The region covered by the plume becomes less oblique along the wind trajectory (figures S43, S49, S70 show what takes place when concentrations are lower.).

(b) The number of fires (data points) to which a function is fitted is way too small. For concentrations of 10 and 1 $\mu\text{g m}^{-3}$, we fitted 75 and 79 points, respectively, while for the 100 $\mu\text{g m}^{-3}$ the dataset shrank to 43 samples and for 1000 $\mu\text{g m}^{-3}$, there were only six samples, which makes fitting unreliable. One might expect that recalculating the dispersion from fires in a finer grid would increase the number of fires, which would cause an increase in the samples with higher concentrations of $\text{PM}_{10}$. However, owing to the greater uncertainties in such a fine grid, it would be better to increase the database of fires. According to the data provided in [6], it can be achieved by evaluating 5 years period, which incudes 2018, the year with the largest number of big landfill fires [18].

| $d_{10}$ | $\beta$ | $R^2$ | $n_{10}$ | $\gamma$ | $\delta$ | $R^2$ |
|--------|--------|--------|--------|--------|--------|--------|
| 1.34 ± 0.51 | 0.52 ± 0.02 | 0.92 | (4.8 ± 4.8) × 10$^{-4}$ | 0.84 ± 0.05 | 0.81 |
| 1.77 ± 0.91 | 0.51 ± 0.02 | 0.87 | (8.2 ± 7.9) × 10$^{-4}$ | 0.92 ± 0.04 | 0.86 |

5. Conclusions

Air pollution caused by emissions from landfill fires is a serious concern. The model with HYSPLIT concentrations showed that although 43 fires (more than half) showed 1 h average concentrations exceeded the level of 100 $\mu\text{g m}^{-3}$ the impacted area and the distance to the furthest point showed that landfill fires cannot be assessed by a single parameter. The three proposed critical parameters described the impact of landfill fires in a concise and reliable manner. The presented methodology can be applied to any gaseous or particulate pollutant, for a range of threshold concentration values for assessing the critical parameters: area, distance, and population. Not only can it be applied to the environmental assessment of historic fire events, it can also be a part of risk evaluation and management in environmental impact assessments evaluating possible risks during the exploitation of waste storage and disposal facilities. The discussed cases of fires that caused an increase in the concentration of $\text{PM}_{10}$ by more than 10 mg m$^{-3}$ showed that the application of modeling methods that involve wind roses and steady-state modeling, as in the reference method in Poland [58], are inefficient and lead to improper results.

Data availability statement

The data generated and/or analyzed during the current study are not publicly available for legal/ethical reasons but are available from the corresponding author on reasonable request.

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