

IMPORTANCE OF COMPREHENSIVE HEALTH RISK ASSESSMENT PROCEDURES FOR MODERN WASTE-TO-ENERGY FACILITIES IN COMPLEX GEOGRAPHICAL CONTEXTS ORIENTED TO CIRCULAR ECONOMY

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ABSTRACT

Although circular economy (CE) principles set material circularity, resource efficiency and waste recycling as priority targets to guarantee the sustainable development of future generations, the thermo-chemical valorisation of municipal solid waste (MSW) still plays a fundamental role in the transition towards the final CE targets. As a matter of fact, the waste-to-energy (WtE) sector allows recovering energy from waste, reducing the pressure on MSW landfills and their related potential environmental impacts; however, recovering material for further uses is not excluded in WtE options. Significant improvements have been achieved in the air pollution control of exhaust gases from direct and indirect MSW combustion during the last decades. The efforts focussed on reducing dioxin emissions especially, and this has let other substances emerge as priority pollutants (e.g., heavy metals). In addition, the location of WtE facilities in certain geographical contexts is still potentially critical from the point of view of human exposure and the related health risk; moreover, the public acceptance of WtE plants is still limited, in spite of their recent role in fighting SARS-CoV-2 risks from waste management. The purpose of the present paper is to underline the importance of implementing correct and complete health risk assessment procedures tailored to the exposed population living in the area of influence of a WtE plant. The paper will present two case studies regarding the projects of two WtE plants in a mountainous region, highlighting the critical issues that arose during the environmental impact assessment procedures. The paper will finally suggest possible options to improve the health risk assessment procedure and alternative measures to reduce the expected impacts of the WtE sector on the environment and human exposure.

Keywords: air pollution, cancer risk, circular economy, emissions, environmental impact assessment, heavy metals, human exposure, incineration, gasification.

1 INTRODUCTION

The atmospheric dispersion of air pollutants emitted from ground-level human activities is particularly limited in mountainous regions. This is due to the complex morphology of mountainous areas, where valley winds (especially down-slope) induce strong local circulations that hinder the vertical air mixing normally occurring in flat areas [1]–[3]. In mountainous regions, this translates into enhanced atmospheric stability conditions, which develop especially at night and during winter, conditions that tend to trap air pollutants near the ground [4], [5]. The main sources of air pollutants are human activities (e.g., road traffic, industries, domestic heating), which are usually located at the valley bottom [6]–[11]. Besides being the most critical part of the year in terms of atmospheric dispersion, winter is the season with the highest emissions of air pollutants, due to the additional contribution of domestic heating. The use of wood-based biomass for domestic heating, especially, is widespread in mountainous areas, where wood biomass is relatively abundant in continental climates, and is a known source of particulate matter, nitrogen oxides, polycyclic aromatic hydrocarbons, dioxin and polychlorinated biphenyls [6], [9], [12]–[14]. Thus, air pollutants



are released in a critical atmospheric layer where the exchange of air with the upper layers is limited.

The presence of population, normally settled at the valley bottom, exacerbates this problem, since the stagnation of air pollutants near the ground may become a threat for human health. In large mountainous contexts, like the Alpine region, the presence of urban areas with relatively high population density requires appropriate waste management plans to deal with the municipal solid waste (MSW) produced. Under the vision of the European Waste Framework Directive (2008/98/EC) [15], priority should be given to politics favouring waste prevention, preparing for re-use and waste recycling in concordance with the circular economy (CE) principles [16]–[19]. Under this scheme, waste-to-energy (WtE) processes are considered as the last viable option before landfilling. In this context, WtE processes are intended as thermo-chemical processes applied to non-biodegradable waste (e.g., refuse-derived fuel, residual fraction of MSW or waste assimilated to MSW) as complementary to biorefineries. Along the pathway that leads toward the full application of CE principles, the WtE sector plays a key role in waste management, as it allows: (1) reducing the pressure on MSW landfills, (2) reducing the environmental impacts of landfills, (3) recycling material from bottom ash and (4) replacing fuels for energy production [20].

From the point of view of environmental impacts, the WtE sector has been significantly improved in the last decades. The best results have been achieved in the emission control of polychlorinated dibenzo-*p*-dioxin and dibenzo-furans (PCDD/Fs) and polychlorinated biphenyls (PCBs), as confirmed also by long-term exposure assessment studies in populations living near waste combustion plants [21], [22]. Following such improvements, other pollutants, which were previously considered of secondary importance, should now receive priority in terms of emission control, environmental monitoring and health risk assessment. The contribution of heavy metals and, especially, hexavalent chromium (Cr VI) to the total cancer risk from WtE plants was recently pointed out [23]. In addition, despite the advances in terms of PCDD/F and PCB emission control, the social acceptance of WtE plants is still limited and influenced by the so-called “not in my backyard” (NIMBY) syndrome [24].

The impacts on human health of the pollutants emitted from civil and industrial activities are estimated through the environmental impact assessment (EIA) process, which was developed and implemented in the United States in 1970 with the National Environmental Policy Act [25]. The assessment of the risk for health, as part of the EIA process, requires caution, because the results may be influenced by the choices and hypotheses made to simplify the evaluation, such as the selection of the target area, the emission sources to consider, the exposure routes, the morphologic and meteorological data, the resolution of the computational domain for meteorological and dispersion simulations [26]. Thus, the development of correct health risk assessment procedures becomes crucial to avoid misleading results, potential underestimations of the impacts of human activities on health and unnecessary precautionary approaches that may lead to socio-economic damages to local communities. Correct EIA procedures have also the power to increase the level of social acceptance of WtE, as recently observed in China [27]. Accurate health risk assessment procedures are even more important in geographical contexts that may amplify the impacts expected by one source, like valleys.

The present paper aims at highlighting the importance of the health risk assessment process in the evaluation of the potential risks involved when authorising WtE plants based on direct or indirect waste combustion. To facilitate the understanding of the critical issues that may emerge when carrying out an EIA process, two case studies will be presented and discussed. Both case studies refer to waste combustion plants that were proposed for



authorisation in an Alpine valley, but that were not realised in the area. The paper will propose possible measures to reduce the environmental impacts from the WtE sector and, meanwhile, improve the risk assessment procedure and the environmental legislation in order to properly account for potential underrated risks for health. The role that WtE are playing during the SARS-CoV-2 does not affect the methodology presented in the following sections.

2 CASE STUDIES

The selected case studies refer to two waste combustion plants that were proposed in different locations of the same valley (Adige Valley), located in northern Italy in the Italian Alps (Fig. 1).

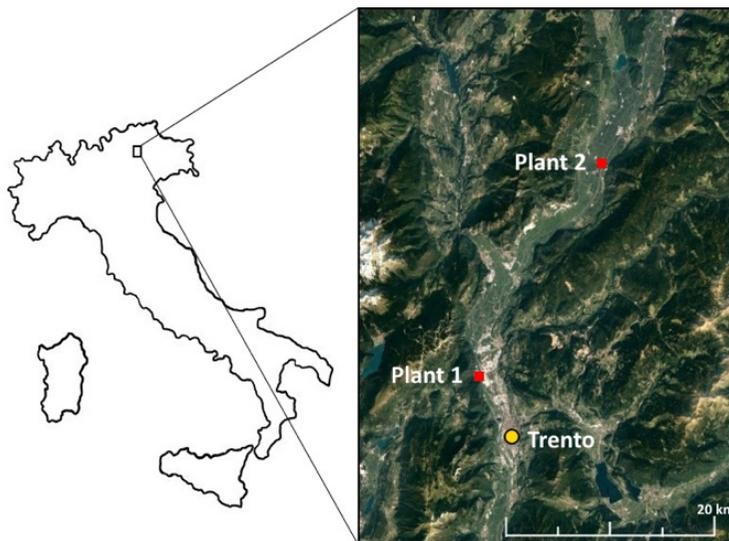


Figure 1: Locations of the Italian WtE plants selected as case studies.

The first WtE plant (Plant 1) was proposed to treat the MSW generated by the population of the province of Trento, accounting for 542,000 inhabitants [28]. The WtE plant, performing direct combustion of waste, would have treated 103,000 t/year of MSW with a nominal thermal power of 60 MW. The resulting airflow rate from the main stack, normalised to the oxygen content (11%), is 109,000 Nm³/h. The exhaust gas would have been released from a 100 m high stack, with an outlet velocity of 20 m/s and a temperature of 140°C [29]. The stack height of 100 m was chosen based on a previous version of the project, which considered an input waste capacity of about 240,000 t/year. The same stack height was kept in the following version of the project, because lower values would have required in-depth meteorological analyses due to uncertainties in predicting the effects of a local wind (Ora del Garda), blowing from a lateral valley [30].

The second WtE plant (Plant 2) was proposed to treat 95,000 t/year of solid recovered fuel (SRF) and non-hazardous waste, with a nominal thermal power of 63 MW. The plant would have implemented the indirect combustion of waste based on waste gasification and syngas combustion. The expected airflow rate from the main stack, normalised to the oxygen content (6.4%), was 156,000 Nm³/h. The exhaust gas would have been released by a 45-m stack, at a temperature of 130°C and with an outlet velocity of 16.5 m/s [31]. Thus, the main

differences between the two plants consist in the airflow rate of the exhaust gas and in the stack height.

In the European Union (EU), the emissions from WtE plants are regulated by the Directive 2010/75/EU [32], which establishes concentration limit values for several pollutants at the emission level. The limit values concern the following air pollutants: total suspended particles (TSP), total organic carbon (TOC), hydrogen chloride (HCl), hydrogen fluoride (HF), sulphur dioxide (SO₂), nitrogen oxides (NO_x), carbon monoxide (CO), cadmium (Cd) and thallium (Tl), mercury (Hg), other heavy metals, polycyclic aromatic hydrocarbons (PAHs), PCDD/Fs and dioxin-like PCBs (dl-PCBs). The other heavy metals regulated by the Directive are: antimony (Sb), arsenic (As), chromium (Cr), cobalt (Co), lead (Pb), manganese (Mn), nickel (Ni) and vanadium (V). The most critical heavy metal is Cr, which is basically composed of trivalent Cr (Cr III) and Cr VI. Differently from Cr III, Cr VI is a carcinogenic compound and its related cancer risk is dominated by the inhalation route [33]. For health risk assessment purposes, the cancer potency of chemicals is numerically quantified by the so-called slope factors (SFs), defined as the cancer risk (dimensionless) per unit of dose. A SF is expressed as the inverse of the ratio between the mass of chemical that enters the human body per body weight and unit of time, i.e. (mg/(kg_{bw} d))⁻¹. Depending on the chemical's nature, SF values can be available for inhalation (SF_{inhal}) and/or ingestion (SF_{ing}). Fig. 2 presents the SF values of the carcinogenic compounds regulated by the Directive 2010/75/EU [32].

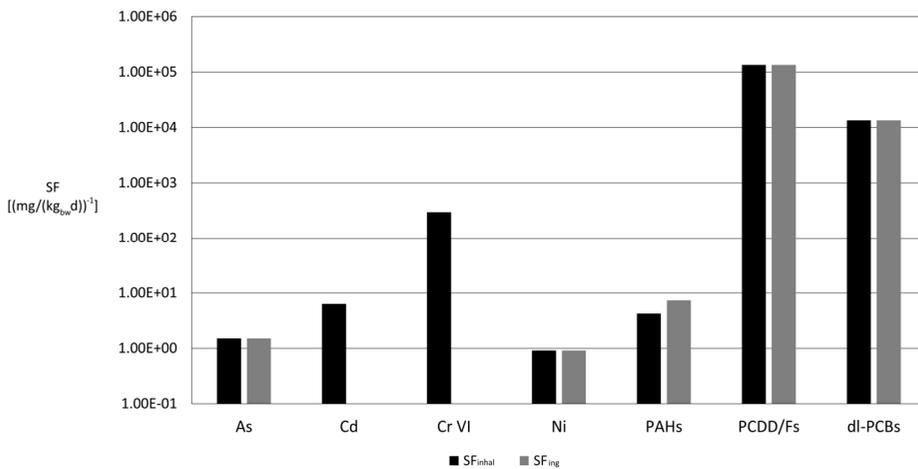


Figure 2: SF values [33] of the carcinogenic air pollutants regulated in EU.

The highest SF values are attributed to PCDD/Fs and dl-PCBs, followed by Cr VI, whose SF_{inhal} is lower than the SFs of PCDD/Fs and dl-PCBs by more than two and one orders of magnitude, respectively. As visible in Fig. 2, no SF_{ing} value has been defined for Cr VI, inhalation being its dominant exposure route. Among carcinogenic heavy metals, Cr VI is the most toxic compound, followed by Cd, As and Ni.

Table 1 presents the maximum design concentration values of the regulated air pollutants expected at the stacks of the WtE plants considered in the present paper. In the case of Cd and Tl and the large group of heavy metals regulated by the EU legislation, compound-specific values were estimated too (Table 2), following a methodology developed in a recent

Table 1: Maximum concentration values of the air pollutants regulated by the EU legislation expected at the main stacks of the two case studies [29], [31].

Air pollutant	Unit	Maximum expected stack concentration values	
		Plant 1	Plant 2
TSP	mg/Nm ³	1.5	1.5
TOC	mg/Nm ³	10	10
HCl	mg/Nm ³	2	2
HF	mg/Nm ³	0.25	0.25
SO ₂	mg/Nm ³	10	10
NO _x	mg/Nm ³	40	40
CO	mg/Nm ³	50	50
NH ₃	mg/Nm ³	10	10
Cd+Tl	mg/Nm ³	0.025	0.025
Hg	mg/Nm ³	0.025	0.025
Other metals	mg/Nm ³	0.25	0.25
PAHs	mg/Nm ³	0.0001*	0.01
PCDD/Fs	ngI-TEQ/Nm ³	0.03	0.025
PCBs	ngI-TEQ/Nm ³	–	0.1

* only for benzo[a]pyrene.

Table 2: Estimation of the specific maximum stack concentrations for the heavy metals grouped according to Directive 2010/75/EU [23], [34].

Heavy metal	Estimated mass fractions in each metal group (–)	Maximum expected stack concentration (mg/Nm ³)
Cd*	0.907	0.02268
Tl	0.093	0.00233
Group total	1.00	
Sb	0.109	0.02725
As*	0.003	0.00075
Cr III	–	0.16600
Cr VI*	–	0.00500
Co	0.003	0.00075
Pb	0.075	0.01875
Mn	0.058	0.01450
Ni*	0.034	0.00850
V	0.034	0.00850
Group total	1.00	

* Carcinogenic.

paper [23]: the maximum design concentration values of the two groups (Cd+Tl and Sb, As, Cr, Co, Pb, Mn, Ni and V) was multiplied by the relative abundance of each compound in its respective group, according to the results of a metal characterisation carried out on Italian MSW incineration plants [34]. The cited study did not perform Cr speciation. However, Cr III and Cr VI concentration values were estimated considering a recent proposal for a

concentration limit value for Cr VI (0.005 mg/Nm^3) [23]. The maximum concentration value expected for Cr III was calculated subtracting this value from the total Cr mass fraction in the heavy metal group (0.684) [34]. Since the maximum stack concentration values guaranteed by the two WtE plants are the same for both groups (Cd+Tl and other metals), the same metal-specific maximum concentrations are expected at both plants.

3 RESULTS AND DISCUSSION

Fig. 3 presents the approximated carcinogenic potentials of the emissions from the main stacks of Plants 1 and 2, calculated as discussed in the previous section. Concerning Plant 1, a maximum concentration value expected at the stack for dl-PCB was not provided by the proposers, thus the EU limit value was considered (0.1 ngI-TEQ/Nm^3). Regarding PAHs, only a benzo[a]pyrene maximum design concentration value was proposed for Plant 1.

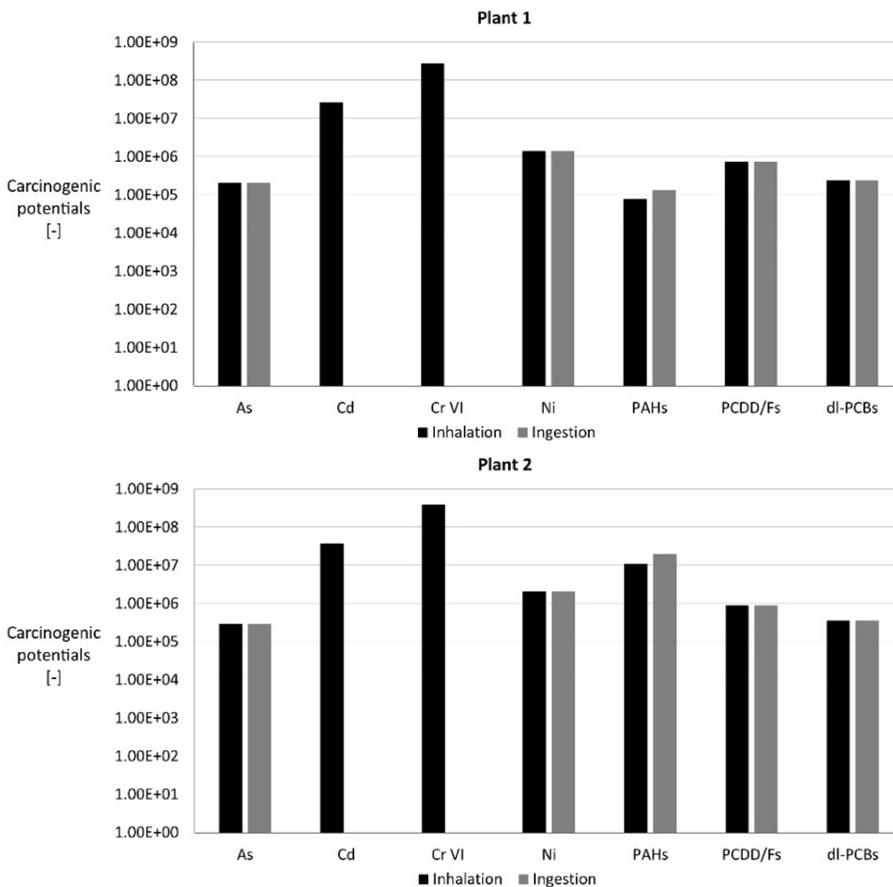


Figure 3: Carcinogenic potentials of the air pollutants released by the two WtE plants [37].

It is of course necessary to consider other key variables that may determine the fate of the air pollutants emitted (e.g., stack height, outlet velocity, exhaust gas temperature, the local meteorology, morphology, land use, the presence of population settled and its typical diet).

However, this approach can give a preliminary idea of the expected burden of carcinogenic compounds released by a plant. In addition, this approach could be adopted for other types of emission sources than WtE plants.

As shown in Fig. 3, despite the higher SFs of PCDD/Fs and dl-PCBs, the greatest contribution to the carcinogenic potentials of the mixture released at the stacks of the two WtE plants is given by two heavy metals: Cr VI and Cd, the first giving a 10-times higher contribution than the second. This is the result of the stack concentration limit values set by the legislation, which defines single total values for Cd+Tl and the sum of the other heavy metals including Cr VI.

In fact, each group is dominated by a single compound: Cd, in the Cd+Tl group, and Cr VI in the other group. The presence of single concentration limit values for groups of heavy metals led the proposers of the plant to define single maximum concentration values guaranteed by the plants for the respective groups. It is worth noting that, according to this legislative scheme, the limit value set for the largest group of heavy metals is 0.5 mg/Nm³. Here, a 100-times lower concentration value was adopted for Cr VI, because of its carcinogenicity. From Table 2, it is possible to calculate the percentage of Cr VI in total Cr (Cr III + Cr VI), which is 2.9%. This percentage is in the range of values available in the few literature studies on Cr speciation in the exhaust gas of WtE plants, which has been found to be 1.15%–6.5% [35], [36]. Thus, Cr VI is likely to be the dominant pollutant for the inhalation exposure route, followed by Cd. On the other hand, Ni and persistent organic pollutants (PCDD/Fs, dl-PCBs and PAHs) are likely to be the dominant pollutants for the exposure via diet. Therefore, in contexts with limited agriculture and/or livestock activities, i.e. where the consumption of local food products is limited, health risk assessment procedures should pay specific attention to Cr VI and Cd. Reducing the release of heavy metals thus becomes a priority.

Besides the higher carcinogenic potentials of Cr VI compared to other pollutants, this compound has proved to be difficult to measure in the environment. According to the feasibility study of Plant 1 [29], the Cr VI detection limit in total suspended particle samplings in ambient air is 2 ng/m³. The continuative inhalation exposure to such Cr VI concentration would induce an excess cancer risk of $2.4 \cdot 10^{-5}$ [37], i.e., 24-times higher than the acceptable value for the exposure to single contaminants defined by the Italian legislation [38]. Thus, the compliance with this limit value could not be verified by field monitoring. Dispersion modelling would be the only applicable methodology. However, an EIA should produce/select local meteorological data with particular care to characterise the impact area accurately. As a matter of fact, the dispersion of air pollutants from a source located in a mountainous region is generally weaker compared to coastal areas, uplands or flatlands. This is visible by a simple comparison between the so-called dilution factors (DFs) of an emission source located in a mountainous region and a similar source located in other geographical contexts. DFs are defined as the ratio between the highest value of the average ambient air concentration (or deposition to soil) of a pollutant on the computational domain chosen for the dispersion simulations required in the EIA process and its respective mass flowrate at the emission level. DFs provide an estimate of the maximum impacts of the different pollutants expected at ground level. The higher the DF, the higher is the impact estimated at ground level. To estimate the impacts of the single heavy metals belonging to a group of metals according to the legislation (e.g., Cr VI in the heavy metal group considered by the EU legislation), it is reasonable to consider the DF of TSP, since metals are mainly released in particle phase [23]. As an example, based on the results of the dispersion simulations carried out during the EIA process, the DF of TSP ambient air concentration referred to Plant 1 is $8.2 \cdot 10^{-6}$ s/m³ [29]. According to a survey on MSW incinerators in Italy [39], the DF of a



WtE plant located near the coast of the Adriatic Sea is $2.7 \cdot 10^{-7}$ s/m³, thus 30 times lower than Plant 1.

On the side of emission prevention, the release of Cr VI can be reduced by adopting improved air pollution control technologies, based on wet or dry processes. Cr VI shows a high solubility in water, thus adopting wet processes (with further wastewater treatment) could be a viable option. However, the tendency of the WtE sector is to prefer dry technologies for ease of management. Hence the importance of proposing a specific limit value for Cr VI and a routine monitoring of its stack concentration. Once the role of Cr VI will have been verified (e.g., via the carcinogenic potential methodology or, more thoroughly, via a complete health risk assessment), it is appropriate to consider setting a dedicated concentration limit value at the stack. Once this in-depth study on Cr VI is performed, it is desirable to implement a similar approach for Cd.

In case the inhalation route of exposure proved to induce an acceptable excess cancer risk, the health risk assessment should focus on the potential dietary intake of pollutants by the local population. It is crucial to both appropriately select the food crops and livestock activities present in the area and retrieve detailed statistics on the local food consumption and dietary habits. The traditional health risk assessment methodology for the ingestion of contaminated food [40] should then be applied, also considering food-chain models available in the literature in case some food products lack specific formulations.

In unfavourable situations like mountainous regions, it is worth considering alternative WtE approaches. One option is to subdivide a large combustion plant into more smaller plants placed in different locations, in order to lower the local impacts. However, this option entails higher investment costs that might not be sustainable for the local government and community. A more economically and environmentally sustainable opportunity is given by waste gasification: instead of burning the syngas generated in the gasification stage, it is possible to convert it into chemicals (e.g., methanol, ethanol, hydrogen, dimethyl ether) that may also partly replace fossil fuels, also preventing odour problems. This way, both local and global environmental impacts would be reduced, as indicated also in recent literature [41], [42].

4 CONCLUSIONS

The present paper is a contribution to solutions to the environmental problems induced by large combustion plants in mountainous regions, like the Alps. To better visualise the potential implications involved, two case studied regarding two waste combustion plants were discussed. A quick method to estimate the carcinogenic potentials of the pollutants released from a source was provided. This method helps define which pollutants should receive priority in a health risk assessment, thus reducing the risk of neglecting key pollutants. The application of a simplified methodology based on the DFs of TSP, which must be calculated from the results of dispersion simulations, may provide indications on the impact expected from single contaminants at ground level and on the benefits achievable when setting specific emission limit values. The present paper also provided insights into a proper application of health risk assessment procedures according to the results of the screening based on the carcinogenic potentials of the pollutants emitted from a plant, on the land use and local diet of the exposed population. Finally, an unconventional WtE scheme was proposed to reduce the local impacts of WtE plants based on combustion. Such scheme, based on waste gasification and ex-situ valorisation of syngas, is preferable in situations with unfavourable morphology, relatively high amount of input waste and where the construction of multiple small-size waste combustion plants is not economically feasible.



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