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Abstract

The present paper describes the assessment of the atmospheric deposition processes in a basin valley through a multidisciplinary approach based on the data collected within an extensive chemical-physical characterization of the soils, combined with the local meteorology. Surface soil cores were collected on a NNW-SSE transect across the Terni basin (Central Italy), between the Monti Martani and the Monti Sabini chains (956 m a.s.l.), featuring the heavily polluted urban and industrial enclave of Terni on its bottom. Airborne radiotracers, namely ^{210}Pb and ^{137}Cs, have been used to highlight atmospheric deposition.

We observed an increased deposition flux of ^{210}Pb and ^{137}Cs at sites located at the highest altitudes, and the associated concentration profiles in soil allowed to evaluate the role of atmospheric deposition.

We also obtained a comprehensive dataset of stable anthropogenic pollutants of atmospheric origin that showed heterogeneity along the transect. The behavior has been explained by the local characteristic of the soil, by seeder-feeder processes promoted by the atmospheric circulation, and was reconciled with the concentration profile of radiotracers by factor analysis. Finally, the substantial impact of the local industrial activities on soil profiles and the role of the planetary boundary layer has been discussed and supported by simulations employing a Lagrangian dispersion model.

1 Introduction

Soil plays a fundamental role in environmental biogeochemical cycling through a wide range of different processes, both naturally and anthropogenically driven. It is characterized by endogenous processes such as soil development, use, and management, as well as by exogenous processes e.g., climatic factors, atmospheric deposition, and runoff, which may add complexity in terms of chemical components transfer, mixing and reworking on not easily predictable space and time scales. As a result, soil behaves as a receptor for atmospheric deposition, reflecting the influence of atmospheric aerosols with the mediation of wet and, to a lesser extent, dry scavenging according to local climatology and pluviometric regime. Atmospheric deposition flux includes numerous chemical species from gas-to-particle reactions derived nutrients such as nitrate, ammonium, and sulfate, to metals, metalloids, and carbonaceous species (Vet et al., 2014a, b). Once deposited, substances may permeate and migrate to depth throughout the soil pores as a function of the relative interaction strength with soil components. Therefore, each atmospherically originated chemical species will produce a vertical concentration profile reflecting the interplay among its own physicochemical properties and soil properties. In particular, the mobility and adsorption processes of a pollutant in soil will depend on the soil mineralogical composition, porosity, water, and organic matter content, pH, and redox conditions. As time goes by, soil itself will evolve leading to temporal accretion and burial of older layers wherein both endogenous and exogenous substances can be stored and redistributed (Vet et al., 2014a; Fowler et al., 2004; Kaste et al., 2003; Lamborg et al., 2000).

Owing to both the morphology and intrinsic complexity of the soil chemical matrix, the assessment of atmospherically-derived components in a soil profile is usually a challenging task because they represent a minor fraction as compared to bulk soil components. In this context, however, environmental radionuclides represent an exception, since both natural and artificial radioisotopes have been historically investigated to trace the fate of atmospheric pollution in depositional

environments including soil matrices, posing the basis for modern biogeochemistry, geophysical radiotracer research, and geochronology (Baskaran, 2011).

The efficacy of natural and artificial radiotracers in environmental investigations is based on two substantial properties. Firstly, radiotracers can be measured very accurately, even at their lowest molar concentration on a routine basis. Secondly, they are emitted by unambiguous sources, which make them easily recognizable and widely used as surrogates of stable pollutants or as geochronometers and radiotracers (Landis et al., 2016; Fowler et al., 1995; Graustein and Turekian, 1986).

Soil contains two main groups of radionuclides: geogenic radionuclides, intrinsically associated to the soil parent material, and atmospherically derived radionuclides, transferred to the soil environment through wet and dry deposition. Geogenic radionuclides are mainly represented by uranium and thorium families together with the primordial ^{40}K. The atmospherically derived fraction includes among others ^{210}Pb ($t_{1/2}$ = 22.3 years) and ^{137}Cs ($t_{1/2}$ = 30.2 years) both widely employed in the study of depositional processes (Baskaran, 2011); ^{7}Be ($t_{1/2}$ = 55 d) is another frequently used atmospheric radiotracer, but owing to the relatively short half-life its use is conditioned by the time elapsed between sampling and measurement.

While ^{137}Cs is an artificial radionuclide deriving from both the global fallout (peak emissions from nuclear weapon testing in 1963) and, in the northern hemisphere, from the Chernobyl accident in 1986 (IAEA, 2006), ^{210}Pb is a natural radionuclide from the ^{238}U radioactive family including ^{222}Rn and ^{210}Pb formed at intermediate stages of the radioactive chain. In particular, ^{210}Pb belongs to ^{222}Rn progeny, but differently from the parent nuclide which is a noble gas, all its progeny including ^{210}Pb, is particle reactive and similarly to ^{137}Cs, gets efficiently associated to submicron aerosol particles, tracing airborne particulate and eventually the aerosol sinks (Graustein and Turekian 1989; Preiss et al., 1996; Persson and Holm, 2011; Mabit et al., 2014; Landis et al., 2014). All these radionuclides

are highly particle-reactive. Therefore, after being produced/emitted to the air, they quickly get associated with ambient aerosol, usually in the accumulation mode (i.e., submicrometric in size), with a residence time of the order of days up to a few weeks. As a result, they are excellent tracers of both atmospheric aerosol (Tositti et al., 2014a; Brattich et al., 2015a, b; 2016), and of atmospheric deposition (Battiston et al., 1987; Bettoli et al., 1995; Tositti et al., 2006).

Airborne radionuclides establish negative concentration gradients in soil with depth. These profiles reflect both their origin and the processes they undergo within the soil, linked to its properties and management. Atmospherically deposited radionuclides are in most cases confined in the upper 15-20 cm of soil which outlines the need for high-resolution sectioning when dealing with soil and the possibility of efficiently detect the influence of atmospheric contribution independently on the nature of the pollutant investigated, as reported in the literature (Suchara et al., 2016, Landis et al., 2014; Graustein and Turekian, 1989).

Radionuclides profiles in soil depend on the extent of atmospheric deposition, which in turn depends on a number of meteorological processes and orographical features. In fact, many studies have observed how the deposition rate of wet-removed chemical species increases with increasing altitude. In areas characterized by complex topographies, this differential behavior has been attributed to the influence of orographic clouds which produce an enrichment in aerosol particles being removed by nucleation through the so-called “seeder-feeder mechanism” (see for example Le Roux et al., 2008 and Likuku, 2006 and references therein). Airborne radiotracers, whose environmental sources and sinks are known in remarkable detail, have been successfully applied to constrain the fate of stable substances and pollutants with similar environmental behavior contributing to enlighten their atmospheric source (Vet et al., 2014a; Fowler et al., 2004; Kaste et al., 2003; Lamborg et al., 2000).

The present paper concerns the assessment of the depositional pollution features in the Terni basin, an area heavily impacted by human activities in Umbria, Central Italy. The present investigation of biogeochemical cycling of pollutants in the Terni area, accounting for its topographical complexity, is part of a comprehensive environmental study where earlier research shed light on the role of the planetary boundary layer in atmospheric dispersion of pollution in the area (Moroni et al., 2012, 2013; Ferrero et al., 2012, 2014). The present work describes the assessment of the atmospheric deposition processes in the Terni basin through a multidisciplinary approach based on the data collected within an extensive chemical-physical characterization of Terni soils, combined with the local meteorology. Specifically, this paper is based on a transect of surface soil cores collected at ten stations along with a mountainous profile across the Terni basin. At each station, radionuclide, inorganic, and organic components concentrations were determined as a function of station height and soil depth. Finally, their connections with atmospheric deposition have been analyzed.

The paper is organized as follows:

- assessment of radionuclidic and chemical vertical profiles at the ten sampling stations;
- identification of atmospheric inputs of radionuclides and station classification by multivariate techniques;
- assessment of differential depositional behavior with elevation;
- qualitative comparison between experimental results and the numerical outputs of a Lagrangian Gaussian puff dispersion model.

2. Materials and methods

2.a. Site description and sample collection

The Terni district is a densely populated and industrialized area located at the margins of the Central Apennines range in Umbria, Central Italy (Figure 1). The town of Terni (170 m a.s.l.) lies in a vast plain area (about 2% of the territory) surrounded by medium-range geographical elevations (average elevation 800-1200 m).

The local climate is classified as Csa (Köppen classification), i.e., as mid-latitude temperate, with warm, humid summers and cold rainy winters. The mean annual temperature is 14.5 °C, while yearly mean rainfall is 854 mm/y (ARPA-Umbria, 2013). Weak winds, due to the local topography, are typically oriented along the course of the Nera river, predominantly along with the N-NE, NE, and S-SW directions (Meloni and Carpine, 2004). Terni began its industrial development as far back as 1884, with the building of the Italian largest forge, for the production of armor and guns for the Navy. Since then the Acciai Speciali Terni (AST) became the strongest industrial asset of the area, focusing on stainless steel production for more than one hundred years, until the most recent owner, Tyssen-Krupp. Besides the steel industry, three waste incinerators are located in the Terni area for industrial and municipal waste management. Previous assessments on the local source profile identified traffic and agricultural activities as significant contributions of both fine and coarse particulate (Moroni et al., 2012). Considerable efforts have been made to mitigate the impact of industrial emissions, wood-burning, and road traffic, but only a modest improvement of air quality standards has been achieved so far. In particular, long-lasting high pollution level events are frequently observed in the cold season, in association with intense thermal inversion episodes (Ferrero et al., 2012, 2014; Moroni et al., 2013).

Soil was sampled in summer 2014 at 10 stations located along a roughly NNW-SSE transect crossing the whole Terni basin roughly along a parabolic section with the bottom coinciding with the Terni urban area in the valley (Figure 1 and Table 1).

The stations are classified into three main groups:

- Stations located close to the main pollution sources within the urban territory (Prisciano-PRI, Pineta Centurini-PCE, and Le Grazie-LGR);
- Stations located north and south from the main emission area, but at a higher elevation and distance from the direct influence of the pollution sources (Torre Maggiore-MTM, S. Erasmo-ERA, Cesi-CES to the north, Miranda-MIR, Larviano-LAR and Piani di Stroncone-STR south from Terni);
- Mt. Martano-MM (1094 m a.s.l.), at about 45 km north of Terni is a background mid-altitude station chosen as reference site distant from pollution sources and influenced by the boundary layer only during summertime (Moroni et al., 2015).

M. Torre Maggiore (IT5220013) and Piani di Stroncone (IT5220021) are Natura 2000 protected areas (https://natura2000.eea.europa.eu/, accessed 21/04/2020).

At each station, 3 shallow cores 20 cm long were drilled using a PTFE liner (diameter = 7 cm) and sectioned every 5 cm. Soil samples were air-dried in a low humidity environment for about 2 weeks in a hood, then crushed and sieved (2 mm certified mesh).

Each core was splitted and one portion of the sample was weighed and oven-dried at 105 °C for 16 hours and then weighted again to determine the residual moisture.

According to the documentation on soil management available through the local administration, all the Terni area is characterized by a high degree of human exploitation. The basic information about

the sampling stations as well as information about the bedrock lithology, recovered and classified using the layer "geology of Italy" downloaded from the "Portale Cartografico Nazionale" (http://www.pcn.minambiente.it/mattm/servizio-di-scaricamento-wfs/, accessed 21/04/2020), is reported in Table 1.

2.b. *Analytical methods*

2.b.1. Elemental analysis by WD-XRF analysis

Soil samples were further homogenized and milled in an agate mortar to obtain the fraction < 10 µm. Powder pellets were prepared for the XRF analysis to determine major and trace element concentrations with a Panalytical Axios4000 spectrometer equipped with a Rh tube. Matrix corrections were applied during data processing (Franzini et al., 1972, 1975). Precision and accuracy for trace element determination were better than 5% except for elements at 10 ppm or lower (10-15%), as estimated from the analysis of international reference materials (Lancianese and Dinelli, 2016). The investigated chemical components include SiO_2, TiO_2, Al_2O_3, Fe_2O_3, MnO, MgO, CaO, Na_2O, K_2O, P_2O_5, As, Ba, Ce, Co, Cr, Cu, Ga, La, Mo, Nb, Ni, Pb, Rb, S, Sc, Sn, Sr, Th, V, Y, Zn, Zr. The minimum and maximum values at each sampling station are reported in Table S1 of the Supplementary Material (hereafter SM).

2.b.2. High-resolution γ-ray spectrometry

Natural and artificial radionuclides including among others ^{210}Pb (E_γ = 46.5 keV), ^{226}Ra (E_γ = 186 keV) and ^{137}Cs (E_γ = 661.7 keV) were determined in soil samples by a HPGe (High Purity Germanium) extended-range detector (PROFILE Hyper Pure Germanium detector by Ortec-Ametek Inc.). The detector (relative efficiency of 20% and resolution (Full Width at Half Maximum, FWHM) of 1.9 keV at 1332.5 keV) was calibrated for energy and efficiency using a multiple radionuclide liquid source

(DKD, Eckert & Ziegler Nuclitec GmbH) in a jar geometry (diameter = 54 mm and thickness = 1 cm). Soil samples were measured in the same jar geometry, weighed and then counted for 24 hours each to optimize peak counting statistics. Spectra were subsequently analysed with Gamma Vision-32 software (version 6.07, Ortec-Ametek) allowing also for self-attenuation at low energy on the basis of the apparent density. ^{226}Ra activity was determined at 186 keV correcting the peak area for ^{235}U emission according to the procedure reported by Gilmore (2008). Atmospheric ^{210}Pb defined as $^{210}Pb_{ex}$, i.e. "in excess" of the fraction supported by in situ production by ^{226}Ra, was calculated by subtracting the activity fraction of ^{210}Pb in equilibrium with ^{226}Ra from the mineral soil component from the total ^{210}Pb detected in each sample (Swarzenski, 2014), after correcting for self-absorption. Further details on γ-spectrometry determinations are available elsewhere (Cinelli et al., 2014; Tositti et al., 2016). Uncertainty (here defined as one standard deviation) on the γ photopeaks was calculated propagating the error resulting from the efficiency calibration fit previously determined over the counting error. Minimum detectable activity (MDA) was determined using the Traditional ORTEC method (ORTEC, 2003) with a peak-cut-off limit of 40%. Analytical quality control has been assessed using certified reference materials DH-1a and UTS-3, both by CANMET. The minimum and maximum radionuclides concentration data are reported in Table S2 of the SM.

2.b.3. PAHs and n-alkanes

The determination of PAHs and n-alkanes were performed by GC-MS on a Varian-Chrompack 3800 gas-chromatograph coupled with a tandem mass spectrometry ion trap detector (ITD-MS) (Varian Saturn 2000) and equipped with a split–splitless inlet and a low bled Factor Four VF-5ms analytical capillary column (Chrompack). The analytical procedure was described in detail in Cartechini et al. (2015). PAHs and n-alkanes concentration data analysed in Terni soils are reported in Table S3 of the SM.

2.b.4. Loss-On-Ignition LOI, TOC and N

Total loss on ignition (LOI) was gravimetrically estimated after overnight heating on 1 gram of sample at 950 °C in a muffle furnace.

A LECO Truspec CN analyzer was employed for the determination of total organic carbon (TOC) and total nitrogen (N). About 20 mg of soil sieved at 0.5 mm were weighed in a tin capsule. TOC was obtained by difference to between total carbon and inorganic carbon made on the same samples kept in the muffle for 5 hours at 550 ° C. The accuracy at which the instrument operates is 0.3 ppm or 0.5% RSD for carbon and 40 ppm 0.5% RSD for nitrogen. LOI, TOC, and N data are reported in Table S1 of the SM.

2.c. Statistical methods and elaboration tools

2.c.1. Principal Component Analysis (PCA) and Gabriel Biplot

Factor Analysis based on Principal Component Analysis (PCA) was applied to the complete transect dataset derived from the application of the analytical techniques detailed above, with the aim of classifying soil samples and assessing similarities/differences among the sampling stations.

Missing data or data below detection limits (LOD) were substituted in the data matrices by LOD/2 in order to optimize the modelling outcome. The dataset was normalized before performing PCA by means of autoscaling, i.e. subtracting the mean value to each observation and dividing by the standard deviation. This process produces new variables with zero mean and unit standard deviation, so that each of them has the same importance regardless their original variances and units of measure as explained by van den Berg et al. (2006) and Jolliffe et al. (2016).

Gabriel biplot (Gabriel, 1971) was also used in order to represent both variables and cases together in two dimensions.

2.c.2. CALPUFF

Calpuff is a Lagrangian Gaussian Puff model and has been one of the preferred models adopted for regulatory purposes to assess pollutants transport in the range from tens to hundreds of kilometers (USEPA, 2005). Similar to other Lagrangian models, it is still recommended as a screening model (USEPA, 2017). Its advantage over gaussian-based models is based on its capability to simulate the transport of pollutants in calm and stagnant condition (Daly, 2007), i.e. eventually those typically affecting the Terni valley.

The Calpuff dispersion model (version 7.2.1) was used in the present work in combination with CALMET (version 6.327), a diagnostic meteorological model that develops wind and temperature fields on a three-dimensional gridded domain. Associated two-dimensional fields such as mixing height, surface characteristics, and dispersion properties are also included in its output file.

The domain of the simulation was a square grid 20 km wide centered on the Terni city. The meteorological data was produced by ARPA Emilia Romagna for the year 2014 on the basis of the LAMA (Limited Area Meteorological Analysis) dataset (Jungen et al., 2006) that covers the Italian territory and the surrounding regions with a horizontal resolution of 0.0625° (~7 km) and a temporal resolution of 1 hour. Boundary conditions were provided by the global scale analysis model ECMWF. The diagnostic model requires both geophysical and meteorological data. Among the former ones, terrain elevations were derived from Umbria high-resolution regional thematic cartography, the 25 m resolution Digital Terrain Models of regional coverage. Domain points falling outside Umbria were obtained from the 3 arc-seconds Shuttle Radar Topography Mission dataset. The land-use categories were computed starting from the Corine Land Cover 2000 (22 Oct 2009 update).

Calpuff model allows to specify the source size and type, as well as to assume the source emission as constant or variable in a known mode according to cycle, time of the day, year, etc. (see the discussion, in the following).

The code resolution enables to account for spatial inhomogeneities deriving from orography localized land use as well as wind circulation and pollutants dispersion in the domain, leading to adequately reproduce calm and breeze regimes. Moreover, the use of a Lagrangian dispersion model allows to evaluate the effects both in proximity and at distance from the simulated sources, including the area of maximum fallout of total suspended particles (TSP). Results of the Calpuff model are expressed as soil concentrations isopleths. In this work, we chose to evaluate mean annual concentrations and maximum daily concentrations. This last value produces the maximum annual value reached by the pollutant in each cell (side 200x200 m).

Most of the Calpuff technical options were left to the default settings. However, for the purposes of this work, we chose to simulate the vertical wind shear, without considering the chemical transformation from source to receptor, and to calculate the dispersion coefficients from the values of micrometeorological variables. As far as the PM10 size parameters for dry deposition are concerned, a geometric mass mean diameter of 0.48 micrometers with a geometric standard deviation of 2.0 micrometers was selected.

In the CALPUFF simulation, we included as a source only the major industrial plant of the region, the AST steel plant. The plant stacks were modelized as constant 47 points sources. The emission rates adopted were proposed by the plant operator in the Application for Site Certification submitted in 2010 or the certified self-monitored concentrations of the emission gases for the year 2010. This source configuration is deemed emitting 96% of NOx and PM10 mass on an annual basis. Since each source was assumed to emit at the highest constant rate for the whole period, the simulation could be considered as a worst-case scenario. However, fugitive emissions such as those associated with the volatilization and vapor emission from open vessels and the releases from materials handling, especially the north-east on-site disposal, were not included.

3. Results and discussion

3.a. Atmospheric radionuclides and depositional patterns

In this work, the radionuclides ^{137}Cs and $^{210}Pb_{ex}$ have been determined in a series of ten surface soil profiles to trace the occurrence and the extent of stable pollution fallout across the study area. These radiotracers have been detected at all the stations investigated, suggesting that the influence of atmospheric deposition is active across the whole Terni district.

In order to outline different depositional patterns among the stations, concentration vertical profiles and inventories of both radiotracers were examined. ^{137}Cs and $^{210}Pb_{ex}$ activity concentration depth profiles are reported in Figure 2.

At best, an undisturbed profile of atmospheric ^{210}Pb shows a monotonic decrease with depth, a situation controlled mainly by the organic fraction to which this radionuclide (as well as the corresponding element) is firmly bound, leading even the possibility of dating when high-resolution sectioning is carried out (Landis et al., 2016). Differently from ^{210}Pb, ^{137}Cs shows ideally two distinct activity peaks corresponding to the 1963 and 1986 horizons, well preserved in the case of undisturbed depositional environments; this is not granted in soil wherein the complex behavior of this radionuclide is hardly predictable, especially when long time from deposition has elapsed. ^{137}Cs can be initially bound to surface organics from vegetable litter and subsequently released through rain permeation leading to a lagged association to the clay and organic components fractions at depth as described in detail in Suchara et al. (2016). However, vertical profiles of radionuclides along a soil profile at every single location reflect their behavior in terms of physicochemical interaction between the mobile phase(s) containing the radionuclides and the solid matrix; also the perturbation (disturbance) caused by local soil management, if any, can play a role.

In most cases, soil concentration for both airborne radionuclides showed maxima in the top layer confirming the influence of active deposition from the atmospheric compartment. In most of the profiles, concentrations monotonically decreased as a function of the depth. In some cases, the decrease was noteworthy for ^{137}Cs, even down to the fourth layer. ^{137}Cs, a monovalent soft cation, has a greater vertical mobility and showed consistently a broader distribution with respect to $^{210}Pb_{ex}$. Thanks to its strong association with the soil organic fraction, $^{210}Pb_{ex}$ is on average less mobile than ^{137}Cs, remaining usually confined in the uppermost first and second layers (Mabit et al., 2014; Suchara et al., 2016).

Altogether, the profiles of the two radiotracers also reflect different post-depositional behavior contributing to a more complete understanding of the atmosphere/soil relationships. The most disturbed (i.e. not decreasing) radiotracer profiles are those collected at the bottom stations (i.e. PRI, LGR, and PCE) all in proximity of residential and/or industrial sides. Specifically, the high degree of perturbation at LGR and PRI largely results from the strong anthropogenic influence from urban and industrial activities. In particular, PRI is in the core of the industrial Terni district while LGR is located in the Terni residential area, connecting the randomized distribution of radionuclides to the local remarkable degree of soil disturbance from human activities (e.g., reworking). On the other side, PCE maintains a slight degree of atmospheric deposition identifiable by the decreasing concentration trend with depth. Atmospheric deposition at PCE is sensibly affected by the airshed of Terni conurbation in terms of chemical composition and is possibly supported by the pine stand, a park area dating back about fifty years ago. Atmospheric deposition is therefore detectable in the area owing both to the absence of soil reworking in the last five decades whereas throughfall and foliar interception may have played a role in soil enrichment of the fallout species (Fowler et al., 2004; Likuku et al., 2006).

These differences in concentration vs. depth profiles arise mainly from the distinct soil characteristics. For this reason, depth profiles of atmospheric radionuclides have been evaluated individually for each sampling station. In order to assess the extent of deposition rate, comparing the different stations, we calculated the radionuclide inventory, a parameter conceptually close to a depositional flux. The inventory, expressed in Bq/m^2, is the integral of the radionuclide activity concentration over depth accounting for soil geometry (unit surface and layer thickness) and soil mass density, and was calculated according to the method proposed by Graustein and Turekian (1986, 1989).

The results are reported in Figure 3 where ^{137}Cs and $^{210}Pb_{ex}$ inventories are represented along a profile cross section. Minimum inventories are observed at the three valley bottom stations while maxima are observed at the three most elevated stations, namely at MTM, and, with slightly lower values, at MM to the north and STR on the opposite branch of the transect (see Figure 1). These results are in agreement with previous studies observing an increased deposition flux with height (Fowler et al., 1988; Le Roux et al., 2008; Stankwitz et al., 2012; Blackwell and Driscoll, 2015).

Atmospheric deposition appears to be higher on the NNW side of the transect with respect to the SSE side, with the exception of ERA station where both ^{137}Cs and $^{210}Pb_{ex}$ inventories are lower possibly due to local disturbances.

This difference in the inventories has been evaluated by the Unpaired Two-Samples Wilcoxon Test at a significance level of 95%. This non-parametric test allows to compare the means in two independent groups of samples without any prior assumption regarding data distribution. The results of the test indicate a statistically significant difference between the NNW (defined by MM, MTM, and ERA stations) and the SSE (defined by PRI, PCE, and LGR stations) sides of the Terni transect (p-value less than 0.05 for both ^{137}Cs and $^{210}Pb_{ex}$) and the exceedance in deposition in the

NNW side might be due to the combination of the meteoclimatic conditions of the district, of the dominant circulation pattern along the transect under the influence of both mesoscale and mountain/valley breeze regime (see further on in this paper), as well as of the distinct origin of the two radionuclides. In fact, while ^{137}Cs has been emitted in a pulsed way by point sources and its transport through the troposphere over the investigated region has occurred well above the atmospheric boundary layer, $^{210}Pb_{ex}$ is generated in the lowest layers of the troposphere through radon exhalation and decay, which means a continuous and extended source area. As a result, it cannot be excluded that $^{210}Pb_{ex}$ deposition flux is contributed by both local and distant sources. As such, the former component may be affected by the seeder-feeder phenomenology through the uplift of radon enriched air masses from the plain, while the latter can be supported also by mesoscale/synoptic processes.

As a whole, we can conclude that the radionuclidic signature traces the atmospheric deposition along the soil transect in a fairly satisfactory way, though with differences that deserve a thorough inspection and evaluation.

3b. Sampling stations and their association with atmospheric radiotracers

We will focus herein on the strategy adopted for detecting associations between atmospheric radiotracers and sampling stations. Owing to the extensive dataset collected, a multivariate approach was selected in order to detect data patterns and significant associations as widely applied in atmospheric science (e.g., Tositti et al., 2014b, 2018a; Núñez-Alonso et al., 2019, Petroselli et al., 2019) and in general in environmental science (e.g., Perez-Bendito and Rubio, 1999).

Factor analysis based on Principal Component Analysis (PCA) was applied to compositional data including the atmospherically derived radionuclides from all stations and in all the sections for

station classification and solved for station affinity; Gabriel biplot showing the two atmospheric radionuclides ($^{210}Pb_{ex}$ and ^{137}Cs) is reported in Figure 4, while factor scores (that are the observations coordinates on the PCA dimensions) are reported in Table 2. While each soil station has its own peculiarities, and will be object of a dedicated paper, the station score distribution shows that all the stations are clustered in groups reflecting substantially their mean height. Similarities are found between MM and MTM, i.e. the high-altitude stations on the NNW branch of the transect, and between STR and MIR on the opposite edge, in connection with minimal human disturbances as compared to bottom stations. A second larger group includes CES, LGR, LAR, PRI, i.e. the stations at low-altitude within the transect, while PCE and ERA appear as isolated with respect to all the others.

Figure 4 shows as PC1 is poorly described by atmospheric radiotracers ($^{210}Pb_{ex}$ and ^{137}Cs), unlike PC2 and PC3. The use of $^{210}Pb_{ex}$ and ^{137}Cs suggests a clear separation among the ERA, MIR, and PRI stations, which conversely does not appear among the PCE, MM, and MTM.

The distinction of the ERA station is in agreement with what previously observed. Also, the PRI station can be justified by an anomaly in the concentration profiles of both ^{210}Pb and ^{137}Cs, which is correlated with disturbances related to local soil use (see Figure 2 and Section 3a).

It is therefore, concluded that the radiotracer approach is efficient in solving atmospheric deposition in soil, even if the extension of the approach to the whole compositional dataset in order to detect unambiguously atmospherically derived components would require an increase of resolution in soil sampling and associated vertical profile, as performed, for instance, by Landis et al. (2016).

3.c. Organic compounds

Differently from elements and radionuclidic components discussed above, PAHs and paraffins, chosen for their likely atmospheric origin similarly to the airborne radiotracers on which this paper

422 is based, were analyzed in the bulk cores, mixing the whole upper soil 20 cm without sectioning. In
423 this case, no vertical profiles of organics were available, therefore data and correlations were
424 studied considering exclusively inventories of organic pollutants in analogy and association with
425 $^{210}Pb_{ex}$ and ^{137}Cs inventories. Total PAH has a high linear coefficient of determination with soil TOC
426 ($R^2 \approx 0.85$), owing to the high affinity with this macro soil component.
427 The comparison of PAH's inventories across the soil transect (Figure 5A) reveals that the bottom
428 stations (LGR, PRI and PCE), all located within the Terni conurbation, present the maximum
429 deposition of these pollutants.
430 In particular, all the three urban stations present PAH concentration values 7 - 45 times higher than
431 the average of all the remaining stations, indicating an extremely high degree of local pollution; in
432 addition PCE presents by far the highest total PAH concentration level together with the highest
433 HMW (high molecular weight) PAH's concentrations, possibly in association again with enhanced
434 aerosol interception by the tree canopy. The highest LMW (Low Molecular Weight) PAH's fraction
435 was detected at LGR, a site more exposed to vehicular traffic. Outside the urban environment, the
436 concentration profile of PAHs along the two sides of the transect, showed a certain degree of
437 increase with height and deposition pattern similar to the radionuclidic inventories with higher
438 depositions on the left branch suggesting the potential for similar depositional behavior with height
439 (see Figure 5B). However, the complex environmental behavior of PAH's ranging from different
440 volatilities as a function of molecular mass to photodegradation and/or nitrification in the
441 troposphere, prevent from conclusive deductions on this class of organic pollutants.
442 Paraffins do not show any clear depositional pattern with height, nor specific trends were found for
443 the carbon preference index - CPI (Lichtfouse, 1995) or low vs. high molecular weight paraffins. This
444 observation is in agreement with the findings of Luo et al. (2012), who attributed a large part of the

variance of paraffins in soil samples vs. elevation to vegetation and bacterial processing rather than to atmospheric transport.

3.d. Pollutant dispersion in the Terni basin valley: CALPUFF modeling

The atmospheric deposition detected along the soil transect is supported by the output of the CALPUFF dispersion model. The model was run for the full 2014 meteorological year and showed higher cumulate precipitations at the higher elevation sites (Figure 6) which are in reasonably satisfactory agreement with measured total precipitations recorded by the Regional Hydrographic Network (https://servizioidrografico.regione.umbria.it, accessed 21/04/2020) reported in Table 3. Measured precipitations show a slight prevalence of the NW stations. We note that the highest depositions of radionuclides and chemical species have been observed on the NW branch of the transect, although relative maxima have also been observed in the SE branch. Since the major mechanism of removal of the accumulation mode (300-700 nm) is wet scavenging, this observation is expected to be in agreement with the pattern of the measured precipitation.

The wind roses obtained with the same model runs (Figure 7) show a consistent northerly-northwesterly pattern compatible with an excess accumulation of atmospheric deposition on the north-western branch of the sampling transect, mainly if a seeder-feeder driven phenomenology with mediation by orographic clouds is invoked.

The total wet and dry deposition pattern obtained as an outcome of the model is shown in Figure 8. Maxima are located in the valley bottom, near the PCE, LGR, and PRI sites, but the TSP extends, consistent with the precipitation pattern, towards the higher elevation areas of the computational grid with a slight prevalence of the Northern sectors.

Based on the results from numerical simulations together with the previous pattern of radionuclides and soil composition, we speculate that the plume developing in the Terni bottom airshed enriched

in secondary inorganic aerosol, heavy metals, organic pollutants, and radon from which $^{210}Pb_{ex}$ is generated, might be uplifted and wet-removed through a seeder-feeder process promoted by the atmospheric circulation. Considering that the CALPUFF modeling is based *de facto* on a single source, the qualitative agreement of the model simulations with the observations along the transect can be considered as a reasonable support of the reasonings herein presented.

4. Conclusions

In this paper, we have presented a systematic approach capable of evaluating and describing the deposition process over the Terni district by intercepting the occurrence of the atmospheric components into the soil and atmospheric modelling. The experimental method is based on the use of airborne radionuclides as tracers of atmospheric deposition in a complex mixture of chemical species ranging from inorganic elements to organic molecular species, directly determined in soil profiles. The similarity between soil sampling stations and their association with atmospheric radionuclides is also achieved by means of multivariate statistical analysis.

A significant anthropic impact at low altitude across the Terni basin stations has been detected, because of their proximity of residential and industrial sides, with respect to higher altitude stations. This piece of information is evidenced both by the presence of higher concentrations of PAHs, linked to more anthropogenic pollution, and in a more considerable disturbance of the depth profiles of atmospheric radionuclides. The significant role of the planetary boundary layer in trapping pollutants within a shallow mixing height has been already pointed out in other studies conducted in the Terni valley (Ferrero et al., 2012, 2014; Massimi et al. 2019). Herein, we confirm that this is the prevalent atmospheric process that regulates the dispersion of locally generated pollutants in basin valleys. A higher deposition of atmospheric radionuclides has also been

assessed in stations at higher altitudes, in particular on the NNW side of the transect. This process is more related to long-range transport of pollutants and enlightens the important role of medium- or high-altitude monitoring sites to study transboundary pollution (Petroselli et al., 2018, 2019; Federici et al., 2018; Moroni et al., 2019).

The comparison of atmospheric deposition data with the output of a CALPUFF application to the Terni district shows a satisfactory agreement, even though part of the phenomenology that we assume as being related to orographic precipitation, is not sufficiently captured due to the limited model resolution.

This work represents a multidisciplinary approach for assessing the atmospheric deposition process into soils, and further studies will be carried out in order to deepen also the soil contamination across the Terni district.

Acknowledgements

The authors thank MIUR and Perugia University for financially supporting the research through the AMIS project ("Dipartimenti di Eccellenza-2018-2022") and Arpa Umbria for collaboration in the CALPUFF dispersion model.

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Figure Legends

Figure 1 – Map of the Terni district depicting topography and name/location of the sampling stations occupied in this work. The color scale depicts the altitude (meters above sea level)

Figure 2 - Vertical profiles of $^{210}Pb_{ex}$ and ^{137}Cs along the soil sampling transect. The experimental errors associated with each radionuclide data are defined as error bars. Layer depth ranges in cm

Figure 3. ^{137}Cs and $^{210}Pb_{ex}$ inventories (expressed in Bq m^{-2}) in the investigated sites as a function of relative distance and altitude from Terni (both expressed in m). The size of the circles is proportional to the flux, with scales reported to the left of the figure

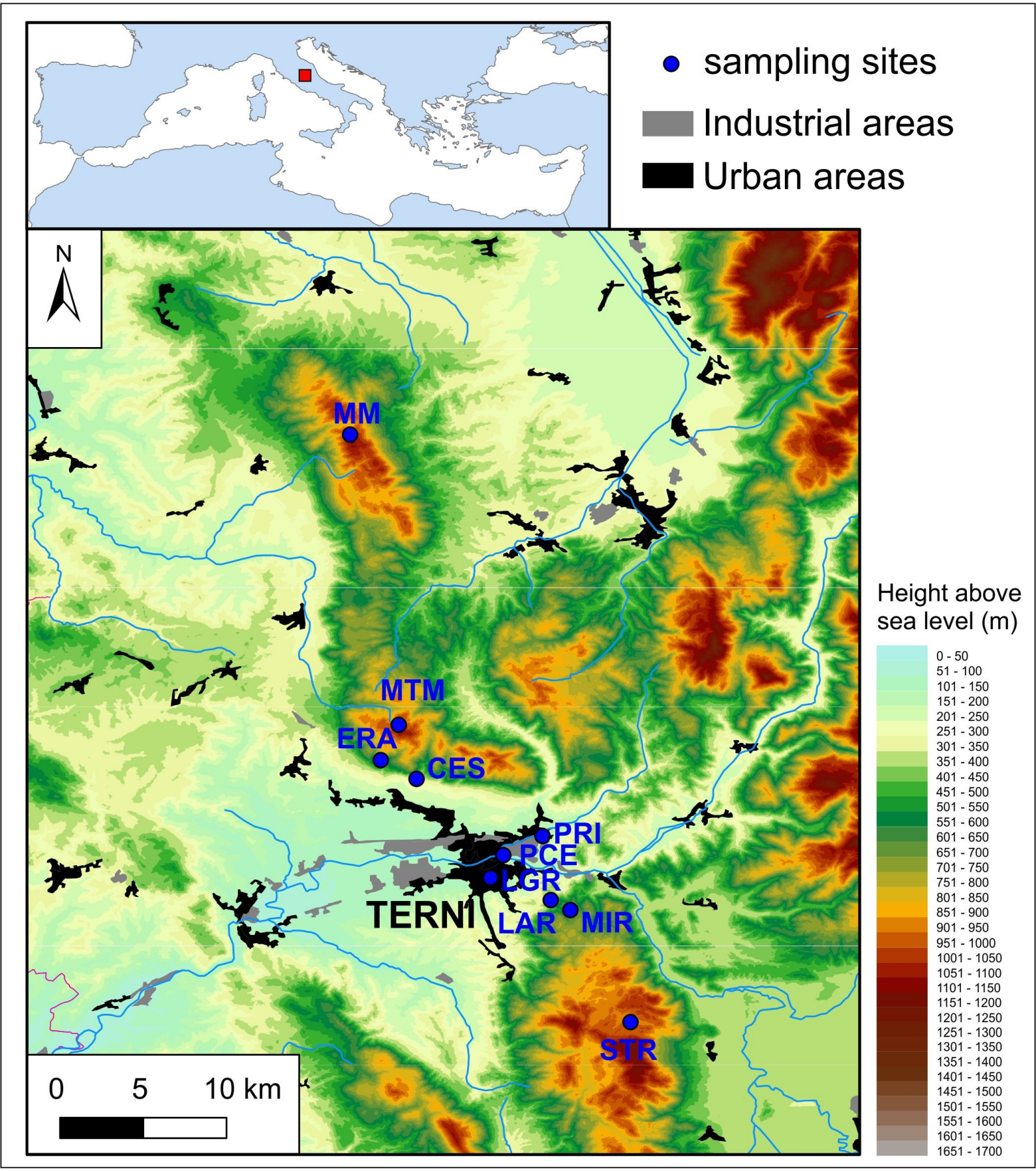
Figure 4. Gabriel biplot of Principal Component Analysis (PCA) with $^{210}Pb_{ex}$ and ^{137}Cs, PC1 vs. PC2 and PC1 vs. PC3

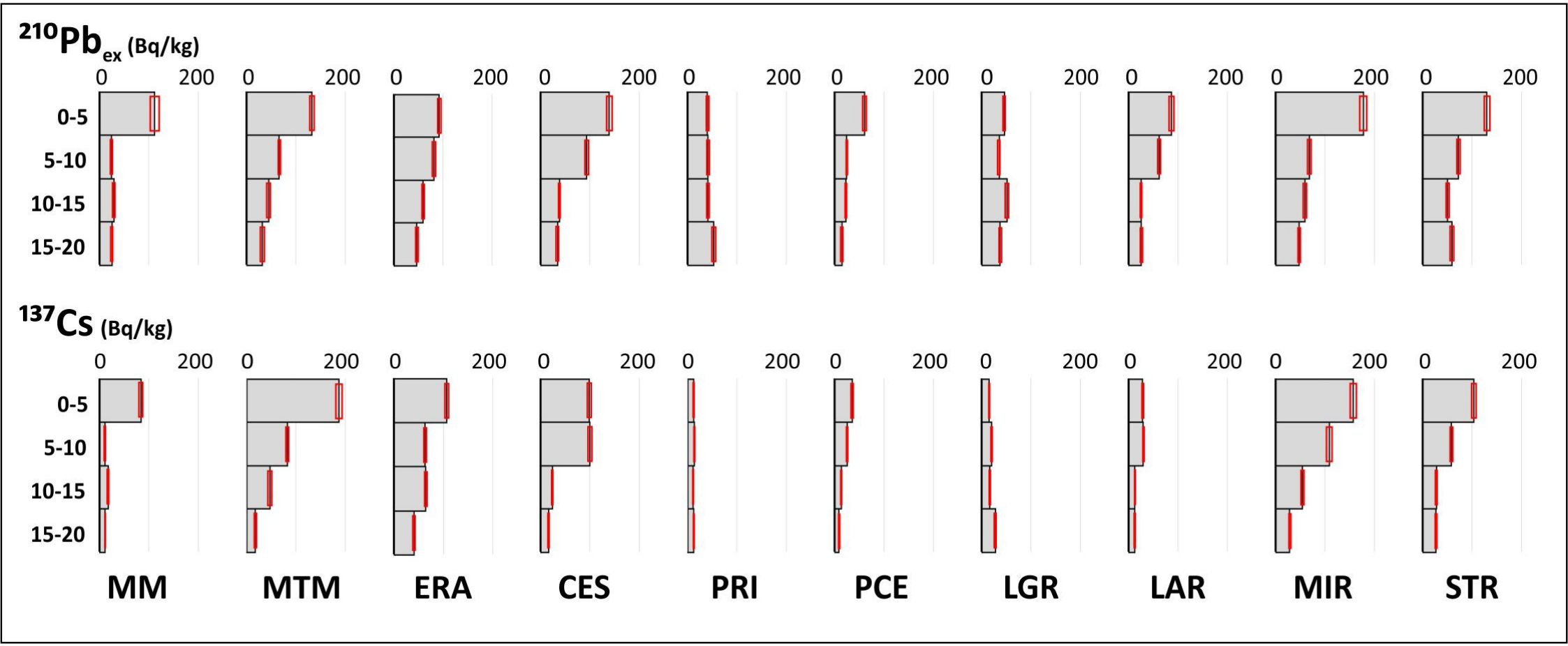
Figure 5. Total PAH inventory (expressed in mg m^{-2}) as a function of altitude from Terni (expressed in m) for all the investigated sites (A) and excluding the three bottom stations in Terni area (B)

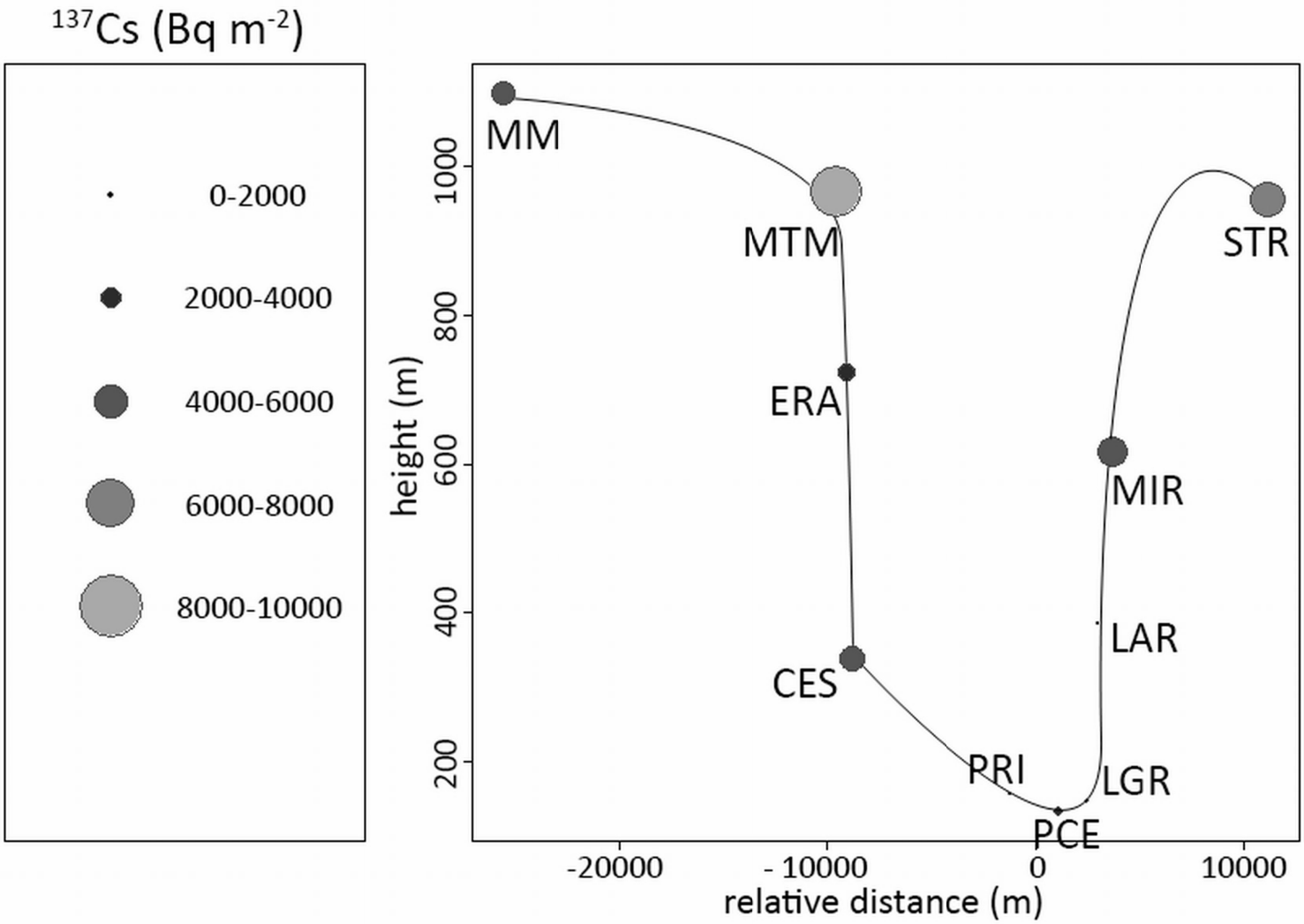
Figure 6. Cumulated precipitations

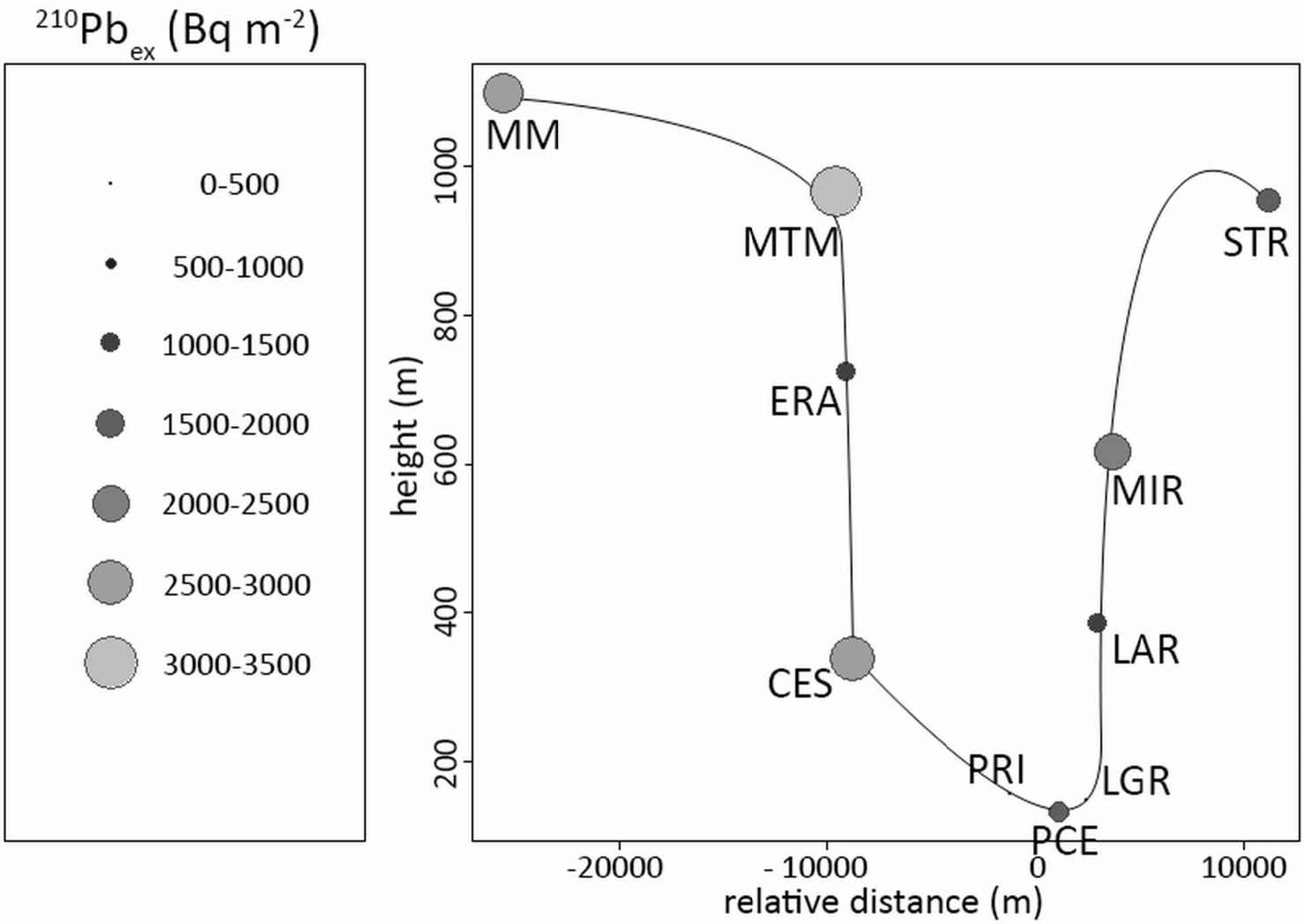
Figure 7. Wind roses for the north-western branch (A) and for the Eastern branch (B)

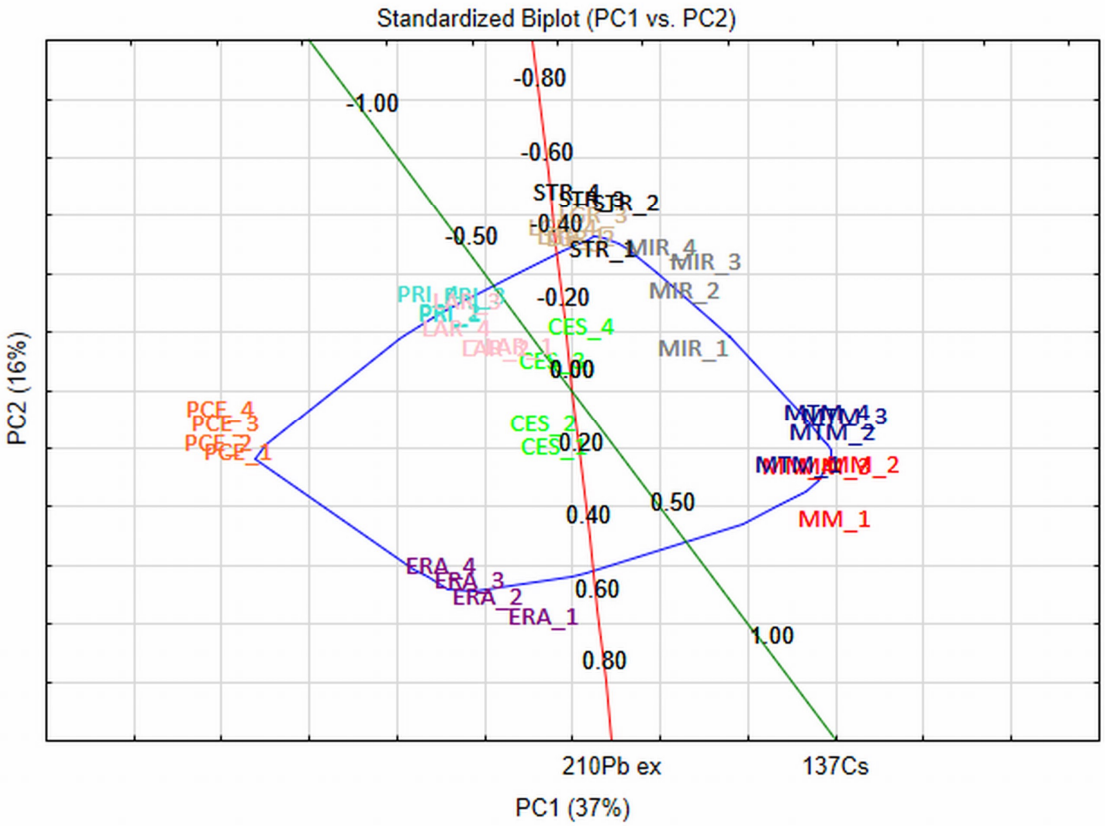
Figure 8. Wet and dry deposition as obtained by the CALPUF model (see text)

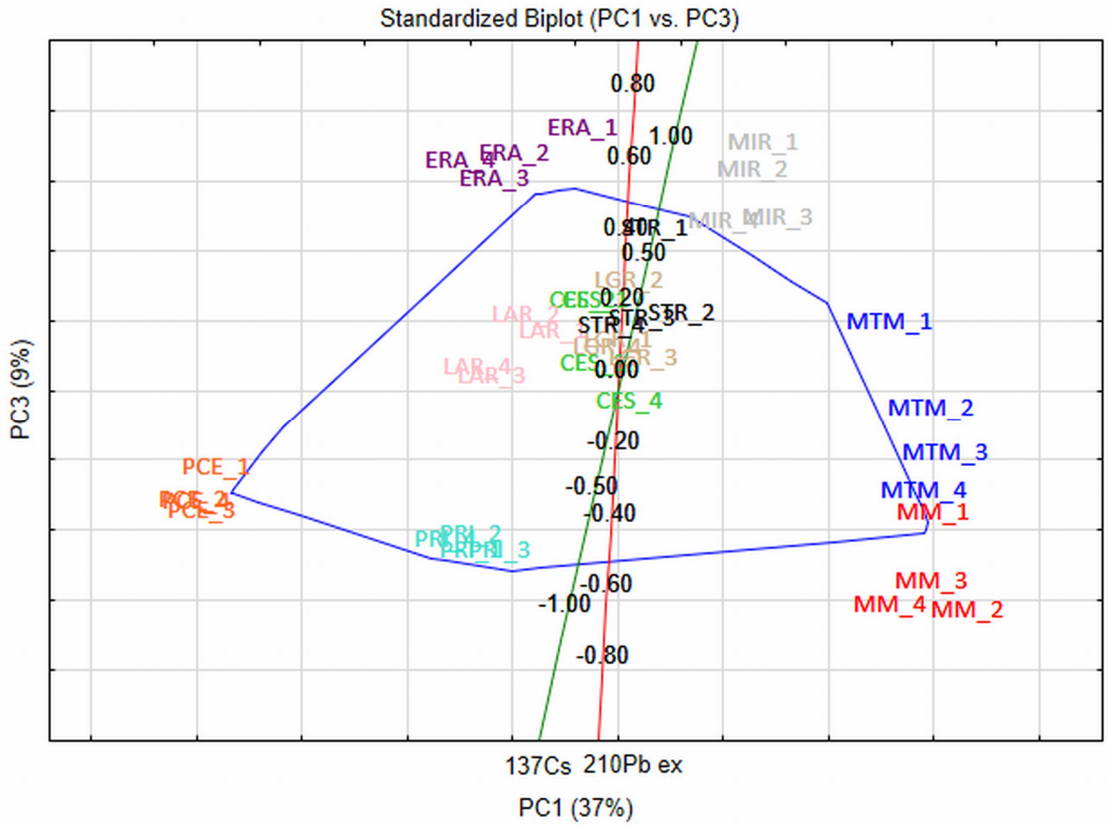

sampling sites
Industrial areas
Urban areas
N
MM
MTM
ERA
CES
PRI
PCE
LGR
TERNI
LAR
MIR
STR
0
5
10 km
Height above sea level (m)
0 - 50
51 - 100
101 - 150
151 - 200
201 - 250
251 - 300
301 - 350
351 - 400
401 - 450
451 - 500
501 - 550
551 - 600
601 - 650
651 - 700
701 - 750
751 - 800
801 - 850
851 - 900
901 - 950
951 - 1000
1001 - 1050
1051 - 1100
1101 - 1150
1151 - 1200
1201 - 1250
1251 - 1300
1301 - 1350
1351 - 1400
1401 - 1450
1451 - 1500
1501 - 1550
1551 - 1600
1601 - 1650
1651 - 1700

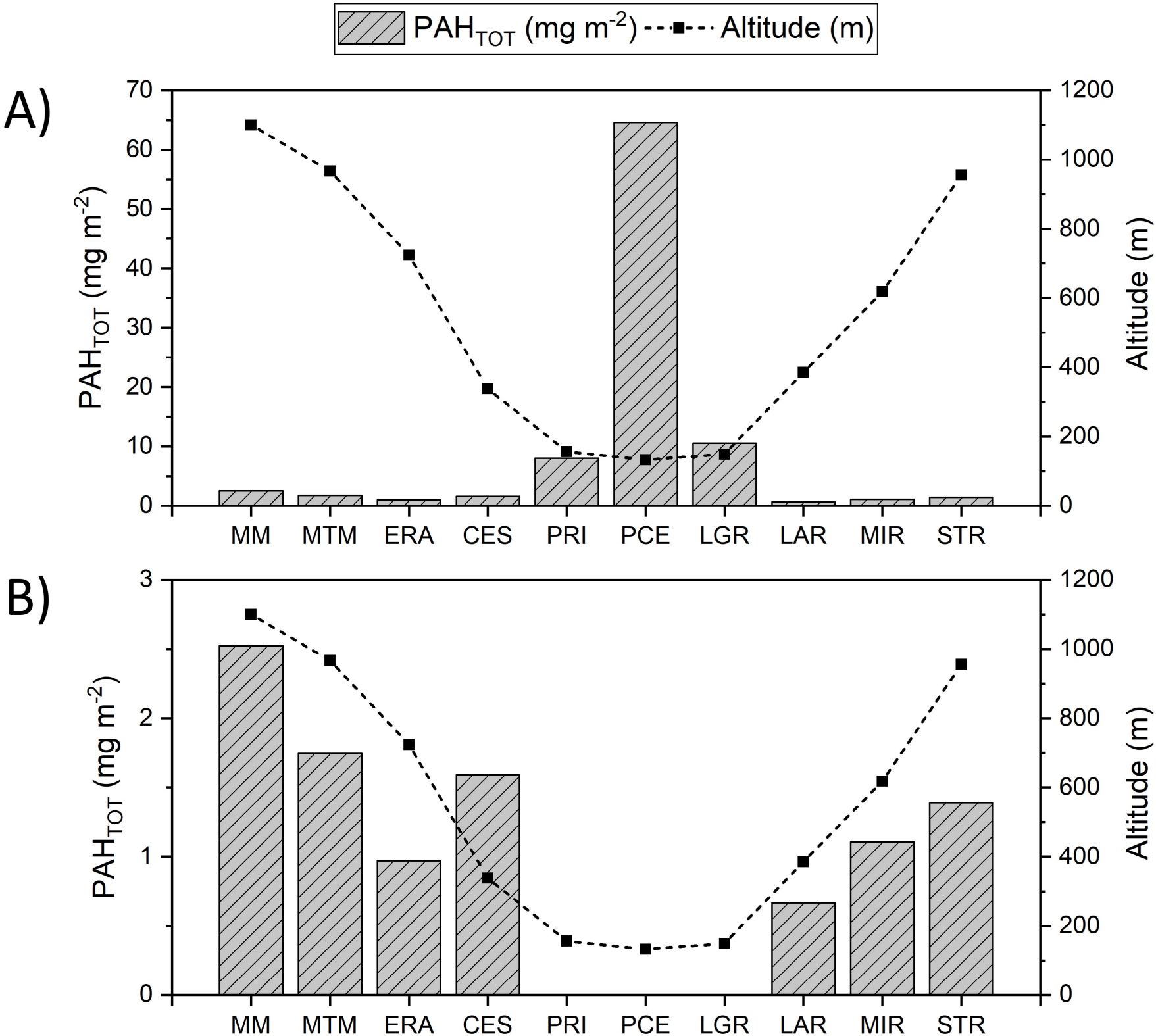

210Pbex (Bq/kg)
137Cs (Bq/kg)
0
200
0-5
5-10
10-15
15-20
MM
MTM
ERA
CES
PRI
PCE
LGR
LAR
MIR
STR

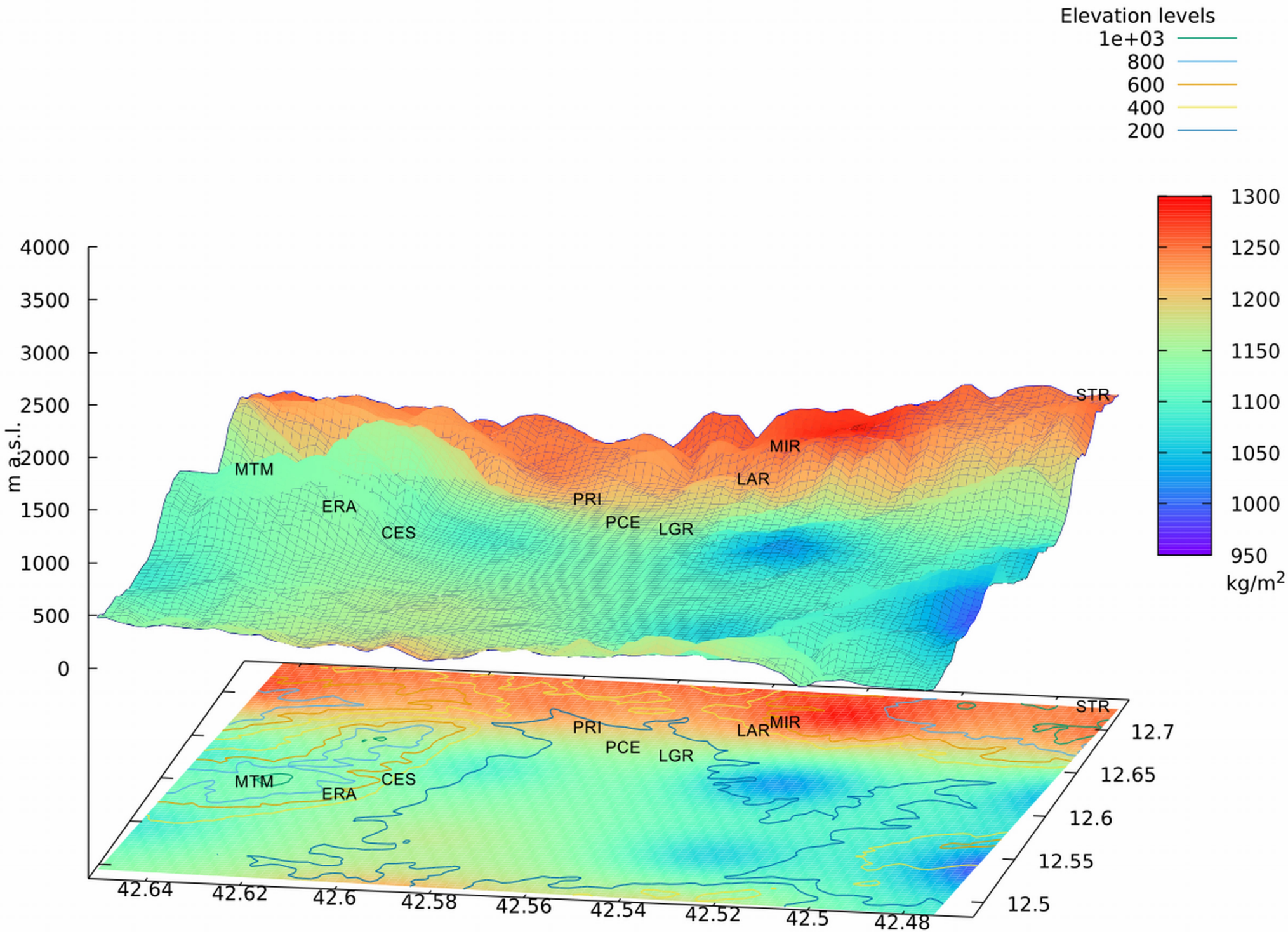

^{137}Cs (Bq m^{-2})
0-2000
2000-4000
4000-6000
6000-8000
8000-10000
MM
MTM
STR
ERA
MIR
LAR
CES
PRI
LGR
PCE
height (m)
200
400
600
800
1000
-20000
- 10000
0
10000
relative distance (m)

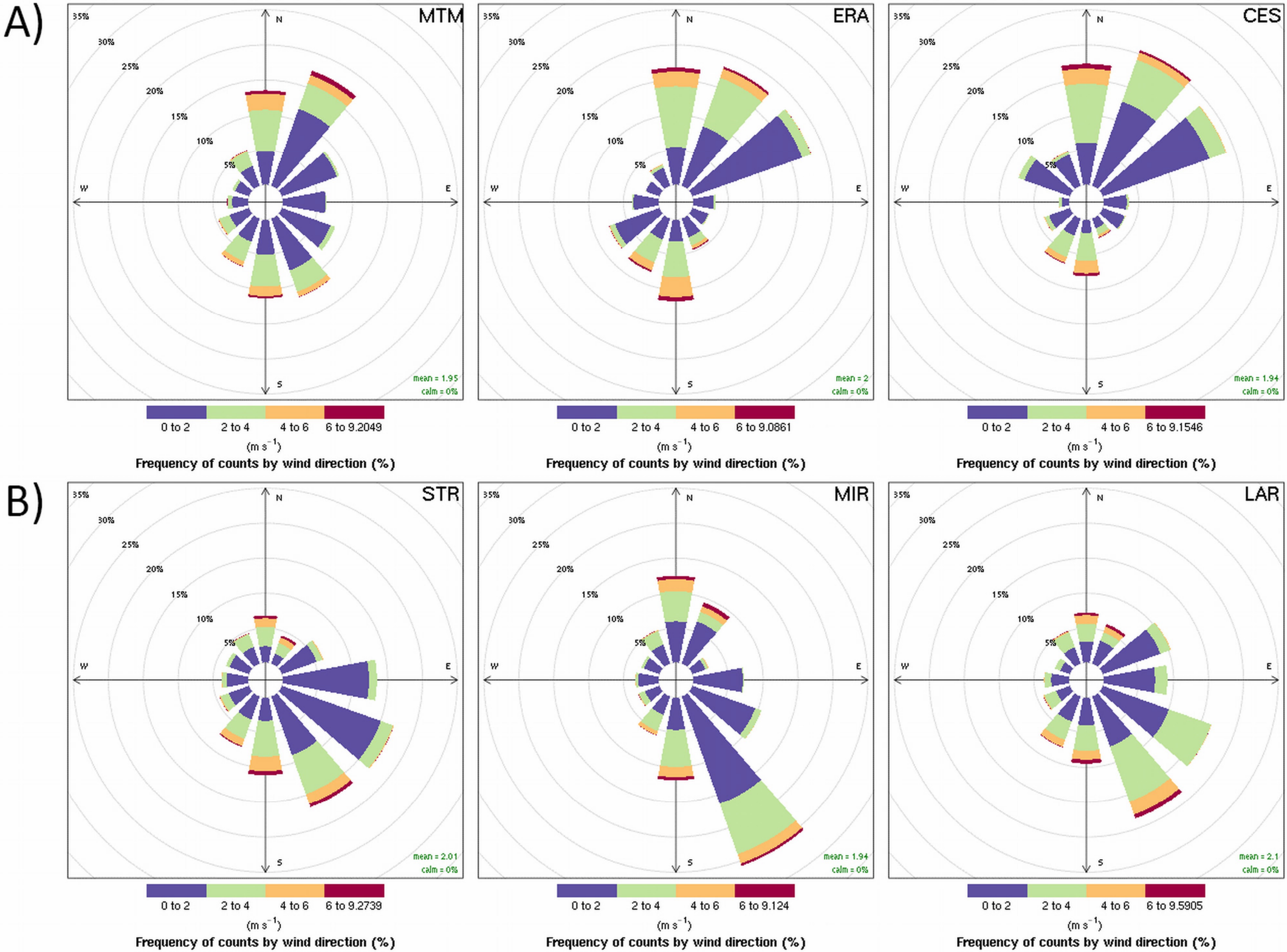

^{210}Pb$_{ex}$ (Bq m^{-2})
0-500
500-1000
1000-1500
1500-2000
2000-2500
2500-3000
3000-3500
MM
MTM
STR
ERA
MIR
LAR
CES
PRI
LGR
PCE
height (m)
200
400
600
800
1000
-20000
- 10000
0
10000
relative distance (m)


Standardized Biplot (PC1 vs. PC2)
PC2 (16%)
PC1 (37%)
210Pb ex
137Cs
-1.00
-0.80
-0.60
-0.50
-0.40
-0.20
0.00
0.20
0.40
0.50
0.60
0.80
1.00
STR_4
STR_3
STR_2
STR_1
MIR_4
MIR_3
MIR_2
MIR_1
PRI_1
PRI_2
PRI_3
PRI_4
LAR_4
CES_4
CES_2
CES_1
PCE_4
PCE_3
PCE_2
PCE_1
MTM_2
MM_1
ERA_4
ERA_3
ERA_2
ERA_1


Standardized Biplot (PC1 vs. PC3)
PC3 (9%)
PC1 (37%)
137Cs
210Pb ex
0.80
0.60
0.50
0.20
0.00
-0.20
-0.50
-0.40
-0.60
-1.00
-0.80
1.00
ERA_1
ERA_2
ERA_3
ERA_4
MIR_1
MIR_2
MIR_3
MIR_4
STR_1
STR_2
STR_3
STR_4
LGR_2
LAR_2
LAR_3
LAR_4
CES_3
CES_4
MTM_1
MTM_2
MTM_3
MTM_4
PCE_1
PCE_2
PCE_3
PRI_2
PRI_3
MM_1
MM_2
MM_3
MM_4


PAH$_{TOT}$ (mg m^{-2})
Altitude (m)
A)
B)
MM
MTM
ERA
CES
PRI
PCE
LGR
LAR
MIR
STR


Elevation levels
1e+03
800
600
400
200
1300
1250
1200
1150
1100
1050
1000
950
kg/m2
4000
3500
3000
2500
2000
1500
1000
500
0
m a.s.l.
MTM
ERA
CES
PRI
PCE
LGR
LAR
MIR
STR
42.64
42.62
42.6
42.58
42.56
42.54
42.52
42.5
42.48
12.7
12.65
12.6
12.55
12.5


A)
MTM
ERA
CES
B)
STR
MIR
LAR
N
S
W
E
35%
30%
25%
20%
15%
10%
5%
mean = 1.95
calm = 0%
mean = 2
calm = 0%
mean = 1.94
calm = 0%
mean = 2.01
calm = 0%
mean = 1.94
calm = 0%
mean = 2.1
calm = 0%
0 to 2
2 to 4
4 to 6
6 to 9.2049
6 to 9.0861
6 to 9.1546
6 to 9.2739
6 to 9.124
6 to 9.5905
(m s⁻¹)
Frequency of counts by wind direction (%)

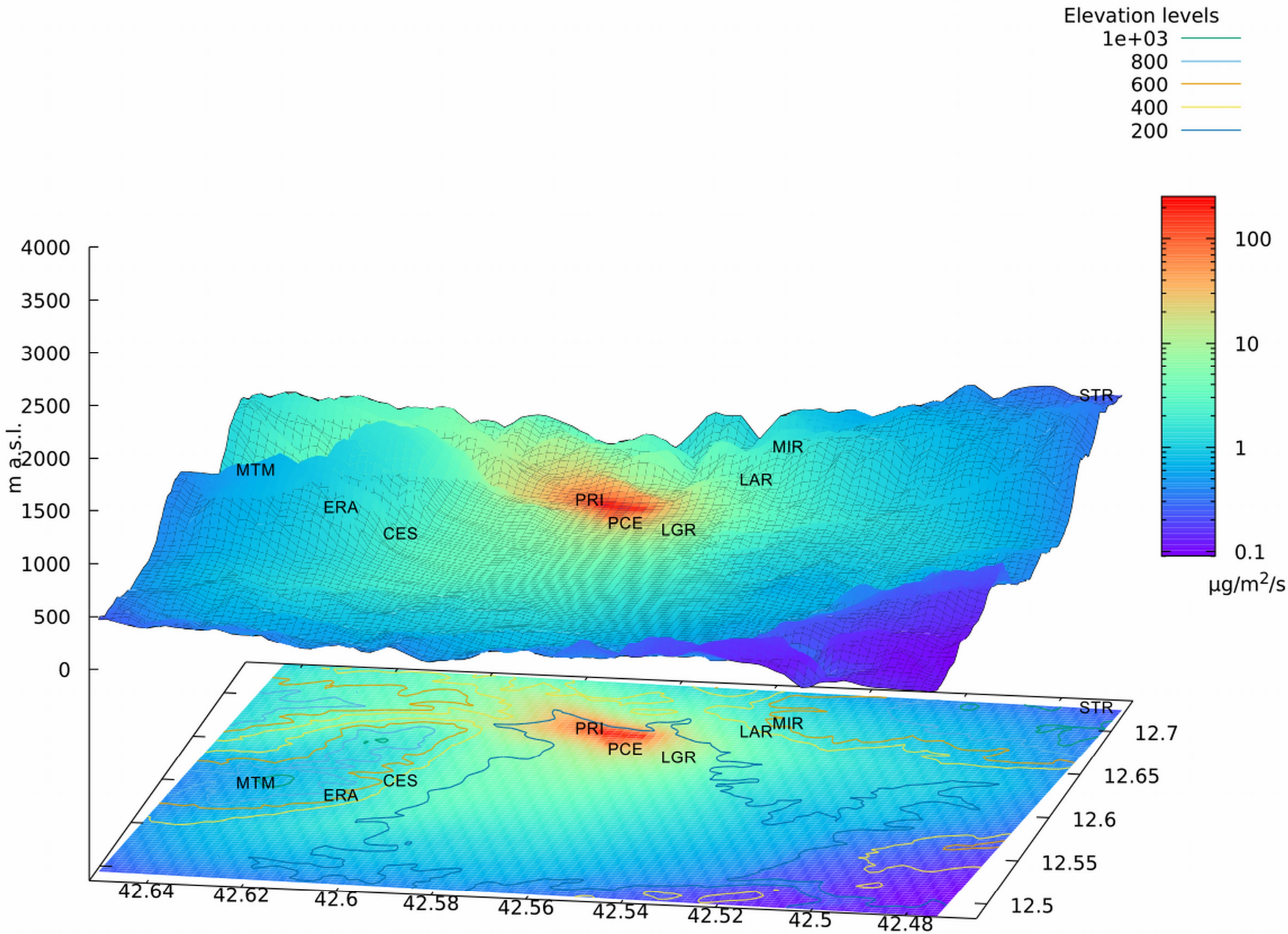

Elevation levels
1e+03
800
600
400
200
4000
3500
3000
2500
2000
1500
1000
500
0
m a.s.l.
MTM
ERA
CES
PRI
PCE
LGR
LAR
MIR
STR
100
10
1
0.1
µg/m2/s
42.64
42.62
42.6
42.58
42.56
42.54
42.52
42.5
42.48
12.7
12.65
12.6
12.55
12.5

Table S1. Minimum-Maximum concentration of major element oxides (SiO2, TiO2, Al2O3, Fe2O3, MnO, MgO, CaO, Na2O, K2O, and P2O5; weight percent, wt %), LOI, T.O.C, N tot (weight percent, wt %), and elements (As, Ba, Br, Ce, Cl, Co, Cr, Cs, Cu, Ga, Hf, La, Mo, Nb, Nd, Ni, Pb, Rb, S, Sc, Sn, Sr, Th, U, V, W, Y, Zn, and Zr; milligrams per kilogram, mg/Kg) for each sampling station

	MM	MTM	ERA	CES	PRI	PCE	LGR	LAR	MIR	STR
SiO_2 (wt%)	28.3 - 34.5	41.3 - 42.2	32.1 - 40.5	32.4 - 36.5	33.5 - 36.1	5.8 - 11.2	53.5 - 54.6	37.6 - 40.8	50.7 - 60.7	65.7 - 70.6
TiO_2	0.73 - 0.85	0.74 - 0.76	0.39 - 0.48	0.55 - 0.63	0.39 - 0.42	0.17 - 0.21	0.64 - 0.66	0.47 - 0.51	0.54 - 0.58	0.5 - 0.56
Al_2O_3	12.7 - 15.5	14.9 - 15.8	6.9 - 8.7	9.5 - 11.3	7.1 - 7.5	1.9 - 2.7	13.3 - 13.7	8.7 - 9.7	11.6 - 13.7	11.1 - 12.4
Fe_2O_3	8.7 - 10.3	9.2 - 9.5	4.1 - 5.3	4.8 - 5.5	5.3 - 5.9	2.1 - 2.4	4.9 - 5.1	4.8 - 5.4	5.3 - 5.7	4.7 – 5.0
MnO	0.5 - 0.5	0.46 - 0.47	0.3 - 0.33	0.13 - 0.15	0.16 - 0.17	0.07 - 0.08	0.13 - 0.13	0.13 - 0.13	0.15 - 0.22	0.12 - 0.13
MgO	3.1 - 3.4	2.34 - 3.37	3.55 - 4.21	2.31 - 2.38	1.87 - 1.98	1.29 - 1.54	2.37 - 2.52	1.23 - 2.34	1.24 - 1.32	0.78 - 1.45
CaO	1.6 - 7.1	0.8 - 1.8	6.6 - 19.1	13.6 - 16.9	21.7 - 23.2	42.3 - 49.2	6.2 - 6.8	14.8 - 20.0	1.2 - 1.6	0.6 - 0.7
Na_2O	0.1 - 0.11	0.09 - 0.14	0.14 - 0.16	0.2 - 0.23	0.2 - 0.24	0.17 - 0.24	0.52 - 0.55	0.06 - 0.15	0.1 - 0.11	0.13 - 0.19
K_2O	1.83 - 1.94	1.76 - 1.86	0.93 - 1.23	1.72 - 1.89	1.24 - 1.33	0.36 - 0.56	1.83 - 1.93	1.89 - 1.98	1.14 - 1.21	1.43 - 1.54
P_2O_5	0.22 - 0.23	0.23 - 0.27	0.85 - 0.9	0.26 - 0.36	0.17 - 0.17	0.19 - 0.24	0.2 - 0.25	0.15 - 0.17	0.12 - 0.15	0.09 - 0.13
LOI	32.9 - 36.1	24.5 – 28.0	31.0 - 31.6	24.3 - 34.4	25.7 - 26.3	37.1 – 40.0	14.5 - 16.1	23.6 - 25.8	16.1 - 27.2	9.5- 14.0
T.O.C.	7.6 – 12.0	6.3- 9.0	6.8 - 11.5	3.2 - 10.1	1.6 - 1.9	0.9 - 2.4	1.4 - 2.4	1.7 - 5.5	4.0 - 9.7	2.1 - 4.5
N tot	0.91 - 1.14	0.75 - 0.96	0.99 - 1.37	0.38 - 1.06	0.17 - 0.24	0.11 - 0.29	0.15 - 0.23	0.2 - 0.49	0.32 - 0.68	0.22 - 0.41
As (mg/kg)	12 - 13	15 - 16	22 - 27	14 - 15	14 - 16	8 - 10	13 - 16	12 - 15	34 - 38	15 - 17
Ba	471 - 570	418 - 447	298 - 443	368 - 396	340 - 369	250 - 261	447 - 486	215 - 263	299 - 346	304 - 343
Br	14 - 24	10 - 17	6 - 7	7 - 8	3 - 4	4 - 4	3 - 4	3 - 4	5 - 6	5 - 6
Ce	156 - 193	177 - 193	58 - 83	78 - 89	50 - 66	19 - 28	84 - 98	57 - 78	157 - 175	109 - 124
Cl	5 - 5	5 - 5	5 - 5	5 - 5	25 - 55	5 - 5	18 - 19	17 - 49	11 - 25	10 - 18
Co	58 - 72	49 - 51	10 - 18	15 - 17	12 - 12	1 - 1	16 - 17	10 - 15	30 - 31	19 - 20
Cr	85 - 90	61 - 62	42 - 48	64 - 68	415 - 444	57 - 203	69 - 73	43 - 58	33 - 44	27 - 32
Cs	6 - 14	10 - 14	4 - 6	8 - 10	2 - 10	2 - 3	8 - 9	2 - 7	9 - 12	8 - 9
Cu	107 - 112	83 - 99	114 - 121	140 - 181	81 - 87	48 - 60	87 - 90	105 - 112	70 - 76	28 - 31
Ga	18 - 20	16 - 17	10 - 11	12 - 14	10 - 10	4 - 6	14 - 15	10 - 12	13 - 14	11 - 13
Hf	3 - 5	2 - 5	2 - 2	2 - 5	2 - 4	2 - 5	3 - 5	2 - 2	2 - 4	3 - 4
La	129 - 147	153 - 175	48 - 68	60 - 63	40 - 44	8 - 16	53 - 57	36 - 50	115 - 140	70 - 84
Mo	0 - 2	1 - 2	1 - 2	1 - 2	8 - 9	1 - 3	1 - 2	0 - 0	2 - 2	2 - 2
Nb	15 - 17	15 - 16	8 - 9	13 - 14	11 - 13	7 - 8	14 - 15	11 - 12	14 - 15	13 - 15
Nd	88 - 110	98 - 108	29 - 41	33 - 43	25 - 39	7 - 12	37 - 46	29 - 42	74 - 85	47 - 54
Ni	139 - 167	133 - 147	38 - 61	48 - 52	88 - 93	12 - 29	58 - 59	35 - 48	69 - 74	47 - 51
Pb	36 - 56	50 - 86	296 - 449	44 - 60	64 - 74	41 - 53	60 - 342	31 - 37	71 - 100	42 - 65
Rb	120 - 158	132 - 142	73 - 116	91 - 96	51 - 52	17 - 28	107 - 111	61 - 76	127 - 172	92 - 98
S	630 - 720	540 - 620	850 - 1040	500 - 1030	400 - 470	1070 - 1330	230 - 280	310 - 390	190 - 360	151 - 290
Sc	9 - 10	6 - 24	7 - 9	7 - 9	6 - 7	2 - 2	4 - 11	6 - 10	2 - 14	2 - 16
Sn	2 - 6	4 - 5	6 - 12	4 - 8	4 - 9	6 - 14	7 - 11	2 - 5	3 - 6	1 - 4
Sr	62 - 84	57 - 62	66 - 83	93 - 95	238 - 255	373 - 422	142 - 153	109 - 127	51 - 53	91 - 108
Th	8 - 13	10 - 14	4 - 9	13 - 23	14 - 16	2 - 14	12 - 26	4 - 10	28 - 36	20 - 26
U	3 - 4	2 - 3	1 - 2	2 - 3	3 - 3	2 - 3	2 - 2	2 - 3	2 - 3	2 - 4
V	111 - 122	93 - 101	40 - 51	69 - 79	64 - 72	29 - 32	74 - 83	62 - 71	74 - 83	70 - 80
W	5 - 6	5 - 6	2 - 3	1 - 4	1 - 2	1 - 1	2 - 3	1 - 2	4 - 6	1 - 3
Y	85 - 99	99 - 124	33 - 42	34 - 39	27 - 28	9 - 12	34 - 38	31 - 34	78 - 87	48 - 56
Zn	128 - 225	142 - 151	180 - 245	83 - 102	158 - 219	59 - 93	103 - 115	65 - 80	82 - 95	84 - 94
Zr	116 - 131	127 - 137	68 - 86	130 - 147	121 - 128	70 - 83	171 - 185	95 - 109	136 - 150	155 - 178

Table S2. Minimum-Maximum radionuclides concentration data analysed in Terni soils (becquerel per kilogram, Bq/Kg)

	MM	MTM	ERA	CES	PRI	PCE	LGR	LAR	MIR	STR
234Th (Bq/kg)	20 - 1112	0 - 451	9 - 455	0 - 30	16 - 3015	0 - 1799	24 - 3169	15 - 1486	26 - 4043	39 - 5266
226Ra	46 - 67	34 - 49	22 - 56	36 - 68	28 - 60	16 - 51	36 - 66	26 - 40	50 - 88	59 - 90
214Pb	0 - 15	7 - 33	11 - 16	1 - 26	9 - 25	0 - 22	11 - 39	3 - 12	1 - 23	26 - 49
214Bi	1 - 15	0 - 27	11 - 25	1 - 27	1 - 22	0 - 26	9 - 35	0 - 12	1 - 20	17 - 44
210Pb	24 - 111	31 - 132	46 - 92	34 - 139	39 - 53	15 - 60	35 - 51	25 - 87	47 - 177	50 - 130
228Ac	52 - 61	51 - 63	27 - 35	51 - 85	33 - 45	16 - 27	44 - 68	34 - 41	90 - 95	80 - 100
212Pb	48 - 59	31 - 56	23 - 35	34 - 54	35 - 41	17 - 24	44 - 59	27 - 36	81 - 91	71 - 83
212Bi	61 - 94	38 - 73	33 - 65	4 - 74	38 - 77	2 - 41	42 - 80	35 - 54	86 - 147	93 - 111
208Tl	15 - 18	14 - 48	7 - 10	13 - 20	11 - 13	5 - 29	13 - 21	8 - 11	24 - 29	22 - 26
40K	445 - 481	389 - 451	295 - 333	409 - 531	304 - 335	151 - 199	427 - 635	399 - 461	273 - 338	381 - 509
137Cs	11 - 83	17 - 186	40 - 107	16 - 100	10 - 13	8 - 35	15 - 27	12 - 29	28 - 157	27 - 104
$210Pb_{ex}$	18 - 65	32 - 95	4 - 81	57 - 92	2 - 15	2 - 41	5 - 13	23 - 54	5 - 128	12 - 103

Table S3. PAHs (Naphthalene, Acenaphthene, Acenaphthene, Fluorene, Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benzo(g,h,i)fluoranthene, Benzo(a)anthracene, Chrysene+Triphenylene, Benzo(b+j)fluoranthene, Benzo(k)fluoranthene, Benzo(a)fluoranthene, Benzo(e)Pyrene, Benzo(a)Pyrene, Perylene, indenofluoranthene, Indeno(1,2,3-cd)pyrene, Dibenzo(a, h)anthracene, Benzo(b)chrysene, Benzo(g,h,i)perylene, dibenzo(x,y)pyrene, and Coronene; nanograms per gram, ng/g) and n-alkanes (from C12 to C39; nanograms per gram, ng/g) concentration data analysed in Terni soils

	MM	MTM	ERA	CES	PRI	PCE	LGR	LAR	MIR	STR
Naphthalene (ng/g)	0.1	5.4	0.2	0.1	0.4	0.3	4.3	0.0	0.3	0.1
Acenaphthene	0.3	4.3	0.6	0.3	0.9	0.7	2.7	0.2	0.2	0.1
Acenaphthene	3.4	4.5	5.2	4.2	0.9	2.0	50.1	0.7	1.5	1.9
Fluorene	3.7	2.1	2.7	2.6	25.4	43.9	9.4	1.3	3.2	2.3
Phenanthrene	0.5	0.1	0.3	0.2	3.3	6.8	1.4	0.1	0.3	0.1
Anthracene	3.3	4.4	3.1	4.3	53.5	262.2	43.4	3.1	6.6	2.2
Fluoranthene	2.9	2.5	2.5	3.8	32.0	152.4	26.7	1.8	3.7	1.7
Pyrene	1.5	0.3	0.5	0.5	2.3	9.3	1.7	0.1	0.3	0.4
Benzo(g,h,i)fluoranthene	3.0	1.0	1.9	2.8	27.2	122.8	14.4	1.1	2.0	1.2
Benzo(a)anthracene	4.4	2.2	2.1	2.5	21.7	100.5	13.6	1.2	3.2	1.4
Chrysene+Triphenylene	3.4	1.2	3.9	6.1	17.2	92.4	13.8	1.5	2.7	1.8
Benzo(b+j)fluoranthene	3.0	0.6	1.8	3.8	8.7	47.6	10.5	0.9	1.9	1.4
Benzo(k)fluoranthene	0.1	0.5	0.4	0.5	3.0	24.7	3.2	0.2	0.4	0.3
Benzo(a)fluoranthene	2.8	2.5	3.9	4.3	17.7	83.8	17.0	1.8	3.5	2.2
Benzo(e)Pyrene	2.9	1.4	2.4	3.6	27.5	135.1	21.9	1.8	2.8	1.4
Benzo(a)Pyrene	3.5	0.1	1.0	0.9	5.1	27.6	4.8	0.5	0.6	0.6
Perylene	1.4	0.5	0.8	0.5	1.3	4.9	1.3	0.3	1.9	0.5
indenofluoranthene	3.7	2.7	4.0	5.5	12.7	90.0	14.4	1.6	2.7	2.2
Indeno(1,2,3-cd)pyrene	2.1	0.7	0.7	0.9	4.3	25.7	3.7	0.6	0.5	0.4
Dibenzo(a,h)anthracene	3.0	0.7	0.5	0.3	4.1	22.3	3.5	0.4	0.2	0.3
Benzo(b)chrysene	3.8	3.9	4.7	5.1	11.6	85.3	20.5	2.7	4.1	2.4
Benzo(g,h,i)perylene	0.8	4.6	1.4	1.6	15.2	29.9	12.8	1.2	2.6	0.9
dibenzo(x,y)pyrene	3.9	5.1	5.6	5.4	3.2	54.4	20.6	2.6	4.4	2.2
TOT	58.1	53.0	53.3	60.3	300.0	1426.5	321.0	26.4	50.2	32.4
C12 (ng/g)	0.43	<LoD	0.45	0.64	0.05	<LoD	<LoD	<LoD	<LoD	0.25
C13	1.32	<LoD	0.74	0.66	0.17	<LoD	<LoD	<LoD	<LoD	0.28
C14	0.31	<LoD	0.42	0.56	0.48	<LoD	<LoD	<LoD	0.20	0.51
C15	0.53	0.12	0.26	0.41	0.29	<LoD	<LoD	<LoD	0.22	0.37
C16	0.46	0.20	0.42	0.42	0.39	<LoD	0.66	<LoD	0.69	0.40
C17	1.20	0.58	0.59	1.14	1.10	0.76	1.42	0.58	1.92	0.82
C18	1.22	1.02	0.81	1.22	0.86	0.48	1.28	0.78	2.24	0.65
C19	3.06	1.45	0.86	2.60	0.79	0.45	1.59	1.63	3.86	1.09
C20	4.87	2.45	1.91	4.33	1.02	0.89	1.93	2.13	5.21	2.44
C21	12.74	5.89	3.34	14.49	4.71	2.53	2.71	5.86	6.96	4.19
C22	10.05	5.36	1.68	4.16	4.55	3.40	3.67	4.39	7.35	2.49
C23	22.31	13.21	4.73	25.07	13.57	4.86	10.14	10.13	16.35	5.01
C24	7.40	8.82	2.61	7.46	8.33	5.93	5.24	9.18	9.38	2.54
C25	46.87	30.83	14.58	47.57	30.37	9.89	19.15	33.09	31.17	12.64
C26	23.35	15.87	6.10	12.66	14.10	5.57	9.18	14.39	14.47	4.60
C27	143.80	132.53	55.71	100.30	83.91	25.23	96.32	114.79	225.49	45.97
C28	49.01	60.21	16.91	26.10	25.07	8.70	24.16	32.30	54.78	12.24

C29	344.58	290.20	232.48	322.16	225.60	75.66	123.87	396.71	947.64	78.70
C30	27.01	21.47	15.21	38.85	22.28	8.25	9.86	24.18	34.16	5.99
C31	353.88	302.09	281.35	619.09	314.12	82.45	103.39	300.93	490.49	68.38
C32	18.64	17.83	9.69	74.35	19.51	8.79	7.95	19.11	11.31	3.34
C33	153.41	142.65	106.42	571.25	95.44	36.39	31.61	110.87	51.97	27.79
C34	12.84	4.92	4.10	116.79	11.50	4.71	2.98	13.94	3.86	2.72
C35	31.94	13.32	25.09	342.83	17.24	8.33	5.78	35.15	13.58	7.73
C36	2.47	0.88	1.25	10.87	7.05	2.67	1.71	6.41	5.63	1.47
C37	1.22	2.87	2.75	7.82	6.80	3.02	2.15	5.07	1.10	1.82
C38	<LoD	<LoD	1.67	<LoD	4.17	<LoD	1.40	<LoD	<LoD	<LoD
C39	<LoD	<LoD	3.20	<LoD	2.17	<LoD	<LoD	<LoD	<LoD	<LoD
TOT	1274.93	1074.76	795.35	2353.81	915.00	298.90	468.15	1141.60	1940.04	1746.15