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Abstract

The city of Torino (45°N, 7°E NW Italy) has a long history of heavy industry. Additional sources of potential pollutants originate from transport such as car emissions. We selected an area potentially at high risk of contamination: it is sandwiched between roads, the circular Turin highway and the motorway which connects to France, and a landfill where special and hazardous solid wastes from industry are disposed of. Our main aim was i) to discriminate between these sources of heavy metals (HM) and ii) to assess a simplified HM transfer scenario. We started with air diffusion models (inputs were meteo and chemistry of the particulate), then we described topsoils (12 samples km^{-2}) and we sampled, reallocated undisturbed, and cultivated an Ap soil horizon, being the soil *ceteris paribus*. The topsoils description discriminated Factor I, related to the vehicular load and soil parent material (mainly Cd, Co, Cr, Ni, V, and Zn), and Factor II, HM univocally dispersible from the landfill (Sb and As). The ecosystem response is resilience: soil tends to buffer loadings of most HM. In the case of mercury, lead, and arsenic our findings indicate that their transfer to the food chain may be massive.

1. Introduction

The number of potentially contaminated sites in the European Union is estimated to be approximately 3.5 millions (CEC, 2006). The stretches of motorway that cross the Alpine arc amounted to less than 100 km in 1963 and reached 4,000 km in 1993, according to the International Union for the Conservation of Nature (IUCN). The Alps are currently the most endangered mountain chain in the world. One example of increasing pressure which must be balanced with socioeconomic results is Torino ("Turin") in Italy. Torino is surrounded by 400 kilometers of mountains as part of the European western Alps, from the Maritime Alps to Monte Rosa. The Torino metropolitan area (approximately 1.7 million inhabitants against a total population of 2.3 million people in the province) is fifty percent mountainous. From a social point of view, the city is losing inhabitants, while the metropolitan area is growing. In other words, Torino is no longer used only by its residents and commuters for living and working but also by those who live and work elsewhere but use its services. The Alpine valleys of the province are the actual destination of metropolitan life, with built-up areas spread out and congested road traffic. Torino hosted the 2006 Olympic Winter Games, which produced positive lasting effects in terms of ecominic growth (tourism) and negative in terms of environmental impacts (pollution). For instance, the quality of the air of Torino has been monitored for the past 25 years and the results demonstrate that a large percentage of the pollution produced is attributable to city traffic. In the last quarter of a century the individual mobility has increased by about eighty per cent over the Torino area; today over three million individual journeys are made daily. A program for the reduction of air pollution, developed between 1993 and 1998, has produced remarkable results for air quality in the city. Over the period of the last five years, carbon monoxide, nitric oxide, and suspended dust have been reduced by one third. Sulphur dioxide has practically disappeared while particulate emissions increased (2.8 Mg PM_{10} km^{-2} y^{-1} in 2003; APAT, 2006). Urban traffic could often be the main source of heavy metals (e.g. from Belgrade, Crnkovi et al., 2006 or Newark, Joselow et al., 1978 to Karak, El-Hasan et al., 2006). In the city of Torino many industrial areas are still active and concerned with the production of heavy metals (HM)-rich dusts which are primarily collected in one important hazardous waste site.

1.1 Waste legislation in the European Union

The only international convention covering waste issues is the Basel Convention. The convention regulates the import and export of hazardous waste to and from the parties of the convention. Every year the parties are required to submit a report (e.g., the quantities of hazardous waste imported and exported) to the Secretariat of the Basel Convention. The Basel Convention is implemented in the EU with the Regulation EEC/259/93 (OOPEC, 2002).

Early phases of Community waste legislation focused on clearly identified problems such as hazardous waste shipments, PCB disposal, and waste from the titanium dioxide industry. Later amendments of the Treaty, particularly the Single European Act in 1987 and the Maastricht Treaty five years later (OOPEC, 1992) introduced a more general objective of protecting and improving the quality of the environment.

EU has introduced its own legislation covering heavy metals in the form of EU Directives. The Framework Directive 96/62/EC set out a common strategy to define and set objectives for ambient air quality. Directive 1999/30/EC was the first of three so called "Daughter Directives" that specified limit values for various substances identified in the Framework Directive. This 1st Daughter Directive addresses NO_x, SO_2, PM10 and lead. An atmospheric limit value for lead, expressed as an average over a calendar year, is set at 0.5 μg m^{-3} to have been achieved by January 2005 (or January 2010 in the immediate vicinity of specific point sources). Following two further Daughter Directives (covering benzene, carbon monoxide and ozone), the Commission has prepared a fourth Daughter Directive on heavy metals. The directive covers the remaining substances identified in the Framework Directive, namely arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons (OOPEC, 2002). At present, a revision of the Integrated Pollution Prevention and Control, IPPC, (Council Directive 96/61/EC, OJ L 257) is envisaged

(CEC, 2006) and the strengthening of soil protection and contamination prevention elements shall be considered. European target HM loadings for soils are still under discussion and the variability between individual partner countries is high (Table 1).

The present study aims to discriminate the sources of heavy metals in a complex urban/rural area in which different HM sources occur and to infer final loadings from a simplified transfer scenario.

2. Materials & Methods

2.1 Site description

On February 10th 2006, the lighted torch inaugurated the XX Olympic Winter Games in Torino. Overall investments for the infrastructure of Torino 2006 amounted to 2,478 million USD, of which 1,415 million USD were earmarked for major works (competition venues, training venues, villages, roads), 272 for associated works (TOROC, 2006).

Before the impact of the winter games on the city, we studied a complex area (Fig. 1) located on the Torino border district (45°N, 7°E NW Italy), a floodplain collecting the ultramafic rock outcrops from the Susa Valley, in between two high-traffic roads: the circular Turin highway and the motorway which connects the city to Lyon (France) through the Frejus mountain tunnel. A landfill is the fourth element of potential HM diffusion.

The soil heavy metal patterns (as parent material sources mainly) have been well described over a regional basis by Facchinelli et al. (2001) while a discussion on the motorway system appears in Campo et al. (1996).

The landfill is currently defined, for Italian legislation, as a ‘controlled II-Class type C landfill’ where special and hazardous solid wastes from industry are disposed of. The landfill is equipped with two HDPE liners and a clay layer to guarantee impermeability of the system. A drainage system consisting of pipes and pumps provides for leachate collection. The main kinds of waste disposed of are dust and asbestos fibers (disposed of in sealed bags), biological washing sludges and physico-chemical inorganic washing sludges. The overall volume of the empty pit is approximately 600,000 m^3. The wastes belong to the following classes: electrical and electronic devices and equipment, electric and electronic wastes, also containing precious metals, contaminated soils and crushed stones from demolition sites, and waste sands. Based on the European Waste Catalogue classification (CEC, 2000), the quantities of waste collected are reported in the Table 2a. The waste contains large amounts of heavy metals and the total inputs, as averaged data and recalculated over the surface, occupied by the landfill are given in Table 2b.

2.2 Soils

The city of Torino was built and spread on an alluvial plain, which has been formed by the rivers Po, Sangone, Stura di Lanzo and Dora Baltea. The watershed of these rivers contains a mixture of very diverse rocks that have contributed to the chemical composition of the deriving soils. In particular, serpentinites are present that might have increased Cr and Ni contents in the alluvial deposits (Biasioli et al., 2006). A 9 km^2 cultivated area situated close to the river Dora Baltea (with the landfill as its centroid) was studied. Soils belong mainly to Fluvisol, Cambisol, Albeluvisol groups (IUSS Working Group WRB, 2006). The quantitative methods developed for this analysis are based on 110 soil samples (11.6 samples km^{-2}). Only Ap horizons (medium thickness 0–10 cm) were collected on the basis of a regular grid pattern.

Within the range of 200-metres from the centroid of the landfill a homogeneous benchmark environment was simulated by five 0.5 m^3–mesocosms (8 replicates each) filled with Ap horizons from a Haplic Eutric Fluvisol, collected as an undisturbed block using a bottomless box. After collection the soil block was placed into a plastic box equipped with a bottom collector for drainage waters.

Each mesocosm was cultivated with a fast growing horticultural crop: rocket, *Eruca sativa* Miller which is most typically used as a flavouring for salads and a garnish and is one of the most diffuse horticultural crops in Italy. Using rocket provided a simple way of testing potential bioavailability and subsequent transfer into the food chain with popular and widely cultivated, fast growing plant (3-4 cycles per year). The mesocosms were cultivated over three years, without removing the roots and adding new seeds and fertilizers, and the leaves were collected three times during each growing season.

2.3 HM air diffusion model

The system of monitoring of air quality was projected coupling and integrating existing models. The integrated system, collecting wet and dry deposition, enables to sort the activity of the landfill from those deriving from neighborhood sources, on a daily basis. Four automatic stations measure daily both meteorological with ultrasonic anemometer, collecting air temperature data from each vectorial wind component, and wet and dry depositions, which were analyzed in laboratory for HMs after digestion. Input data included meteorological data and chemical data of the particulate. In this paper two models have been used: i) the *Minerve* meteorological model and ii) the *Spray* 1.1 diffusion model. Minerve is a mathematical model aimed to pattern locally and tri-dimensionally air temperature and wind direction with specific attention to air turbulence. This model of total emission from was validated by running previous tests over the period 1994-96 (LIFE Project 94/IT/A32/IT00147/PIE, at http://ec.europa.eu/environment/life/project/Projects/index.cfm). Spray 1.1 is a Lagrangian model of particle dispersion; it simulates transport and dispersion of species chemically inert under complex environments (wind calm, and orography) characterized by spatio-temporal anisotropy of the meteorological variables (wind shear and breeze induced by soil roughness). The dispersion and the meteorological models were coupled and implemented with algorithms based on Stokes law.

2.4 Analytical procedures

Soil samples were dried, sieved and then digested. The leaves were washed (leacheates were analyzed and compared to the deposimeter loads) and ground before being analyzed. All samples (three replicates) were dried at 398 K, sieved at 2 mm, and microwave digested using a mixture of HCl-HNO_3 (3:5) acids. Solutions were analyzed spectrophotometrically (ICP-Varian Liberty 100). Standard Reference Material (SRM) used for verification analytical procedures were NBS 1567a and NIST 1573a (Environmental Protection Agency, Test Method for Evaluating Solid Waste Physical/Chemical Method, SW-846 – Method for determination of metals 6020A). SRM was used to verify the accuracy of the plant material analyses. However, even though the soil samples (without replicates) exceeded 2,500 (this number of observations meets the criteria of Italian legislation), we did not verify the accuracy of the soil analyses since there were no commercially available SRM for *aqua regia* similar to the soils of this study. They are, in fact, totally comparable to the concentrations reported in the same area by Biasioli et al. (2006). Twenty percent of analyses were replicated to ensure the consistency of analytical results. Reproducibility of results was verified by establishing method detection limits for each analysis according to procedures outlined by EPA. All quality controls were reproduced with a minimum precision of 10%; the recovery in the SRM ranged between 99 and 102%.

Spatial data have been modeled using univariate and multivariate geostatistics, some parameters adjusted to accommodate the Poisson-distributed nature of data. HM spatial patterns were tested by analysis of variance of fitted semivariogram model parameters such as field observations and laboratory results, and by comparing interpolation maps. In our study, block Kriging was applied to estimate HM at unsampled locations and the accuracy of Kriging was based on cross-validation. The spatial structure of the HM in the soil was determined through fitted variograms in a two-step procedure: (i) computation of experimental variograms, and (ii) fitting them to theoretical models cross-validated. Each variogram lag distance class contained an average of 150 data pairs with a minimum of 50 pairs. Model fitting for the variograms was selected based on sample variograms, and on the statistical results obtained from cross-validation.

3. Results & Discussion

3.1 HM air diffusion model

Metal-rich particles are scattered over the whole area (data not shown) but after sorting the motorway-component from the landfill component as individual sources (Fig. 2), it appears that the spatial distribution of particles concentrated more than 5 µg m^{-3} and could potentially exceed 250-m from the landfill centroid only along the N-S axis.

3.2 HM soil diffusion pattern

Topsoils (0-20 cm) from Torino and the surrounding rural fields were taken from the area delimited by those rivers that might have contributed to the deposition of alluvial materials where the city is located. The measured pH in urban soils was mostly neutral-alkaline with a mean value of 7.2 (Biasioli et al., 2006). The median pH of the studied area is 7.3±0.2.

The descriptive statistics of element distribution in the <2 mm soil fractions is provided in Table 3: Al, Fe and Mn (parent material core fingerprint), are unevenly distributed over the area. The last two patterns clearly indicate the presence of a former "iron oxide landfill". These three elements are from moderately to highly correlated: r^2 0.52, 0.63 and 0.89 ($p<0.05$ Al *vs* Fe, Al *vs* Mn and Fe *vs* Mn, respectively). Despite these correlations, the distributions are different: Al is leptokurtic, Fe is mesokurtic while Mn is platykurtic.

The weathering of ultramafic rocks produces soils abundant in magnesium and iron and rich in Zn, Ni and Cr (e.g. Shallari et al., 1998). In our case, Zn (mean 220 mg kg^{-1}; comparable with the findings of Biasioli et al., 2006) over the whole area is concentrated in two areas 250 m WSW of the landfill, on both the motorway and the northern side of the motorway. The Zn distribution is leptokurtic. Cr, Ni and Zn, as inherited by the parent material, are highly correlated: r^2 .67 and .70 ($p<0.05$ Cr *vs* Ni, and Zn *vs* Ni respectively).

Biasioli et al. (2006) measured mean topsoil Cr and Ni concentrations (mg kg^{-1} *aqua regia* extractable) of Torino urban soils as 191 and 209 mg kg^{-1} respectively. Chromium (mean <150 mg kg^{-1}) is spatially concentrated in areas 350 m NW of the landfill. Its distribution is leptokurtic. Nickel (mean <100 mg kg^{-1}) over the whole area, is unevenly distributed and concentrations between 150 and 200 mg kg^{-1} are found west of the landfill (300 meters) mainly N of the motorway. The distribution is highly leptokurtic. In different environments, Nicholson et al. (1999) calculated that atmospheric deposition of Zn and Ni to be between 32-45% of the total annual inputs of these metals to soils.

Cadmium is strongly associated with Zn geochemistry and the average content in soils lies between 0.06 and 1.1 mg kg^{-1} (Kabata-Pendias and Pendias, 1992; Banat et al., 2007) while in plants it could range between 0.09 and 0.29 mg kg^{-1} (Ingwersen and Streck, 2005). In ambient air, it is mostly found in the fine particle fraction PM2.5 and ambient air levels at rural sites generally do not exceed 0.4 ng m^{-3}. Urban background levels range from 0.2 to 2.5 ng m^{-3} (CEC, 2003) and accumulate at atmospheric deposition rates of around 2 g ha^{-1} y^{-1} (1.9 in the UK, Alloway, 1999). In the present study, Cd is concentrated in soils with mean of 6 mg kg^{-1} and relatively scattered over the surveyed area, without peak concentration clearly related to a single source and with a leptokurtic distribution.

Antimony in the Earth crust is not abundant, and its concentration in soils spans between 0.3 and 10 mg kg^{-1}. For plants it is a nonessential metal and is known to be easily absorbed reaching concentrations of 50 mg kg^{-1} (Kabata-Pendias, 2001). Antimony in our soils seems relatively undispersed (mean 1 mg kg^{-1}) and peak concentrations (up to 15 mg kg^{-1}) occur within 50 meters of the landfill perimeter.

Arsenic is a metalloid that forms a variety of inorganic and organic compounds. It is distributed rather uniformly in major types of rocks and concentrations range from 0.5 to 2.5 mg kg^{-1} (Kabata-Pendias and Pendias, 1992). Arsenic is a constituent of most plants but little is known about its biochemical role. As-phytotoxicity is presumed but critical values vary between species and experiments (20 to 100 mg kg^{-1}, e.g., Macnicol and Beckett, 1985, or Kitagishi and Yamane, 1981). In ambient air it is mostly found in the fine particle fraction PM2.5. Ambient air concentrations of arsenic at rural sites generally do not exceed 1.5 ng m^{-3}, with lowest values of 0.2 ng m^{-3}. Urban background levels show a range of 0.5 to 3 ng m^{-3} (CEC, 2003). Arsenic, in our study area, is concentrated (mean <0.5 mg kg^{-1}) nearby the landfill (<50 meters) and is virtually non-dispersed in the first 350 meters area W but concentrations start to increase unevenly after this threshold distance. Arsenic and Sb distributions are highly correlated r^2 0.83 (p<0.05). Both of them are highly leptokurtic distributed, indicating that there are higher frequencies of values near the means and there are peaked distributions with thick tails.

Berillium exists in relatively small quantities, comprising less that 10 mg kg^{-1} of the major rock types and its abundance in surface soils has not yet been investigated to a large scale, Kabata-Pendias and Pendias (1992) reported a range of 0.27-3.52 mg kg^{-1}. At the studied site, Be (average soil content <0.01 mg kg^{-1}) is virtually non–dispersed and normal distributed.

Boron is not uniformly distributed in the crust of the Earth and its adsorption on Fe and Al oxides is believed to be an important mechanism governing the solubility in soils (Gupta, 1993) where the range of concentrations span between one and hundreds of part per million, and is similar in plants (Kabata-Pendias, 2001). Over the whole study area (mean 73 mg B kg^{-1} soil) B is concentrated in two areas 400 m west of the landfill, both on the southern and northern side of the motorway. The distribution is leptokurtic.

Cobalt is abundant in ultramafic rocks. Its soil concentration reflects the parent material and ranges from 0.1 to hundreds of part per million. In plants, it is more common to observe deficiency than toxicity and mean concentrations range from 0.01 to 200 mg Co kg^{-1} soil (Kabata-Pendias, 2001). Cobalt at the studied site (mean 20 mg kg^{-1}) over the whole area is concentrated in two peak 350 m WSW of the landfill, on both the motorway and the northern side of the motorway. The distribution is platykurtic indicating that most of the values share about the same frequency of occurrence and the distribution curve is plateau-like.

Lead, which is a target element as a linear-diffusive pollutant, follows the general pattern of the motorway (e.g., Massadeh et al., 2004, calculated enrichment factors to extrapolate the degree of urban pollution). Biasioli et al. (2006) measured an average Pb concentration of the Torino surrounding agricultural soils of 149 mg kg^{-1}. In the present study the average concentration is 71 mg kg^{-1} and the distribution is highly leptokurtic.

Copper in the Earth's crust is most abundant in mafic rocks and is of great importance in agronomic practice. At the studied site, Cu the mean concentration was 55 mg kg^{-1}, which is half of the average concentration reported by Biasioli et al., 2006, and two times the concentration reported by Kumar et al., 2005. Copper levels in the studied area was concentrated on the southern side of the motorway and was relatively scarce in the near the landfill. The Cu distribution is platykurtic.

Tin, present in relatively small amounts, follows the As pattern of distribution. While this concentration of Sn may have important effects on human health it is not toxic for rocket plants, Gough et al. (1979) reported the common range in plants to be 20 to 30 mg kg^{-1}. The Sn distribution is highly leptokurtic.

Vanadium is generally concentrated in mafic rocks and, due to weathering passes into the soils. Average worldwide soil ranges from tens to hundreds parts per million: in the United States, a geometric mean concentration of 58 mg kg^{-1} (Shacklette and Boerngen, 1984). In plants, the evidence that it is essential for their growth is not yet conclusive and there are no reports indicating V phytotoxicity under field conditions (Gough et al., 1979). Vanadium follows the Pb general pattern and in our study area it is concentrated on the southern side of the motorway (averaged concentration of 60 mg kg^{-1}) with a mesokurtic distribution.

Mercury concentrations in all types of rocks is fairly low and does not exceed concentrations of parts per billion (Kabata-Pendias and Pendias, 1992). In highly contaminated soils of the Estarreja Channel, Ria de Aveiro, Hg does not exceed 0.2 mg g^{-1} (Pereira et al., 2005). Total gaseous Hg in Europe varies between less than 2 ng m^{-3} (background locations) to 35 ng m^{-3} at heavily impacted locations (CEC, 2003). In the present study, average concentrations of mercury were <0.5 mg kg^{-1}. With ordinary statistics no diffusion pattern was apparent, although geostatistically a strong relationship with wind and soil concentration following the W direction is apparent. The distribution at our site study is mesokurtic.

The principal component analysis, PCA, enables a reduction in data and description of a given multidimensional system by means of a small number of new variables. According to Morrison (1967), principal components should account for approximately three quarters of the total variance and relevant components are those whose eigenvalue is higher than 1 (Kaiser, 1974). In our case, PCA was carried out by an orthogonal rotation and scores for selected factors were computed by regression (Table 4). According to Kaiser, our KMO value permits use of the factor model. Two factors accounted for a large proportion (75.47%) of the overall variance. Respectively Factor I is closely related to B, Cd, Co, Cr, Ni, Cu, V, Zn, Al, Fe and Mn, and Factor II to Sb and As. They were correlated with the original variables and could therefore be assumed as independent grouping factors. The factor loadings on the original variables after Varimax rotation and factor score coefficient matrix are not shown.

Roads introduce a variety of toxic pollutants to the surrounding environments and are a source of chronic, seasonal and accidental pollution. Lead from gasoline was a common contaminant associated with roads, but other HM also included Ni, Cu (part of clutch lining), Zn (additive to tires), Cd (Bellinger et al., 1982) and V, which is present in carbon containing deposits such as crude oil, coal, and oil shale and is usually recovered as a oil by- or co-product (Lide, 2006). The contour maps of the Factors distribution (Figs. 3a and 3b) are plotted overlapping this zone. Factor I (Fig. 3a), formed by metals abundant in the parent material (Cr, Ni, Al, Fe and Mn) and metals related to the vehicular load (Cd and V), describes the landfill-undirect component. Its spatial distribution is according to the soil types distribution and does not directly reflect depositional scenarios. Factor II, formed by metals univocally dispersed from the landfill, is the landfill-direct component. The spatial distribution roughly follows depositional scenarios but is concentrated within 250-m N from the landfill centroid.

3.3 Ecosystem feedback: a simplified scenario

Jennings et al. (2002), in Cleveland, discovered that public areas commonly have heavy metal contamination significantly above background levels suggesting that redevelopment initiatives should control urban exposure to heavy metal contamination.

In the Torino area, our finding confirms that where the metal loading was high, the accumulation in fast growing plant leaves (Table 5) was high as well, both in terms of maxima and median concentrations. Particles deposited during the growing seasons were not removed by rains or wind and large concentrations of Sb, As, Hg and Pb were observed. The observed concentrations of arsenic may have important effects on human health, but are not toxic for rocket plants which concentrate it in the leaves (Kabata-Pendias and Pendias, 1992).

We sampled and analyzed soils prior to the landfill activity (year 2000), cultivated rocket in mesocosm isolated (soils and percolates were analysed after each cropping), and collected leaves monthly (2 years) and depositions daily (3 years). All the sampling was made on a volume basis. For these reasons, we extrapolated results converting all the metal concentrations in grams HM m^{-2} y^{-2}. The ecosystem response, in terms of median (over the whole area) heavy metal concentrations is resilience (Fig. 4). However, the HM transfer to the food chain is massive in the areas of deposition of metals-rich particles and the spatial correlation between HM soil pseudototals and the concentration detected in leaves cultivated in the mesocosms (where the soil is the same at the time zero of cultivation) at the same location is high: r = 0.964, $p<0.005$.

Even if the decrease in HM concentration was noticeable, i.e. lead, the concentrations in leaves of some metal (arsenic, principally) are above the threshold for human health risks and horticultural crops are distributed everywhere, largely at the urban/rural interface areas. Additionally, the concentrations of mercury increase ten times from the potential anthropogenic sources to the leaves.

4. Conclusions

The study revealed a considerable loading of the topsoils with metals and the PCA allows a reduction in data and description of the given multidimensional system through a small number of new variables. In this case, sorting out single impact factors enables us to find that even elevated anthropic contamination is identifiable.

The ecosystem response: for most HM soil tends to buffer loadings but the HM transfer to the food chain is massive and the correlation between heavy metals potentially dispersible and the concentration detected in leaves is high ($r = 0.964$, $p<0.005$). Even if the decrease in HM concentration in soils was noticeable, i.e. lead, the concentrations in leaves of some elements (antimony and arsenic above all) are higher than the attention threshold for human health risks. Furthermore, as the horticultural crops are diffused everywhere, largely in urban/rural interface areas, particular attention must be paid to the concentrations of mercury, which increase ten times from the source to the leaves.

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FIGURES CAPTION

Fig. 1. The study area (45°06’20”N 7°36’29”, centroid). The landfill is the isosceles triangle (base 360 m, legs 460 m) indicated by the shaded area while the motorway system is in black. Details of the landfill in box-picture (b), while in box-picture (c) the area rose-wind describes direction and intensity of winds from 0.25-0.50 m s^{-1} (open bars) to 1.50-2.50 m s^{-1} (filled bars), following a grey-tone scale. Diamonds (boxes a and b) indicate the locations of four automatic stations while circles (box b) indicate the 8 replicates-mesocosm position.

Fig. 2. Distribution of particles PM_{10} as modelled (example of 2001 December 19th-22nd). The contour map shows the sorted landfill component only. Grey-tone scale from <1 µg m^{-3} (white) to 500 µg m^{-3} (black).

Fig. 3a. Distribution of Factor I (B, Cd, Co, Cr, Ni, Cu, V, Zn, Al, Fe and Mn) in grey-tone scale [average 60229, range 41223 (white)-81240 (black), kurtosis 0.4, skewness 0.1], where the landfill is indicated by the grey-shaded area while the motorway system is in black. Contour map from block Kriging with Gaussian variogram (nugget 257.3). The white portion of the figures occurs as the experimental variogram’s nugget over the whole area was too high, probably due to sampling errors which caused short scale variability. We did not compute those values in both contour maps. Data expressed as cumulative mg HM kg^{-1} soil.

Fig. 3b. Distribution of Factor II, Sb and As (cumulative mg kg^{-1}), in grey-tone scale [average 1, range 0 (white) -24 (black), kurtosis 40, skewness 6], where the landfill is indicated by the grey-shaded area while the motorway system is in black. Contour map from block Kriging with spheric variogram (nugget 1.8).

Fig. 4. HM potential loadings extrapolated as median values over three years time (mg kg^{-1} y^{-1}) coming from wet and dry depositions collected (three deposimeters in Fig. 1b, data in mg kg^{-1} day^{-1}), soil surveys of the first 10 centimetres (9 km^2 area in Fig. 1a, a regular grid with the landfill as its centroid, 11.6 samples km^{-2}, n = 110, data in mg kg^{-1} y^{-1}), rocket leaves collected every three months (eight mesocosm in Fig. 1b, data in mg kg^{-1} $month^{-1}$).

TABLES CAPTION

Table 1

Maximum acceptable soil concentration limits for heavy metals in various European countries (modified from Madrid et al., 2006). Data in mg HM kg^{-1}.

Table 2a

The European Union defines waste as an object the holder discards, intends to discard or is required to discard is waste under the Waste Framework Directive (European Directive 75/442/EC as amended). Here are listed the wastes collected in 2000 (in ton y^{-1}), according to the European codification.

Table 2b

HM loadings: averaged data over the landfill surface (g HM m^{-2} y^{-1}). HM loadings were extrapolated as median values over a period of three years (mg kg^{-1} y^{-1}) coming from wet and dry depositions collected (three deposimeters in Fig. 1b, data in mg kg^{-1} day^{-1}).

Table 3

Descriptive statistics of soil HM concentrations in the first 10 cm (mg HM kg^{-1} soil).

Table 4

Statistics of Factor Analysis (percent).

Table 5

Heavy metal concentrations in rocket leaves (mg HM kg^{-1} dry matter). Descriptive statistics based on twelve observations over a three year period. In italics median concentration in rocket leaves ($MEDIAN_{t_0}$) when the landfill was not operative (four observations, year 2000).

Fig. 1

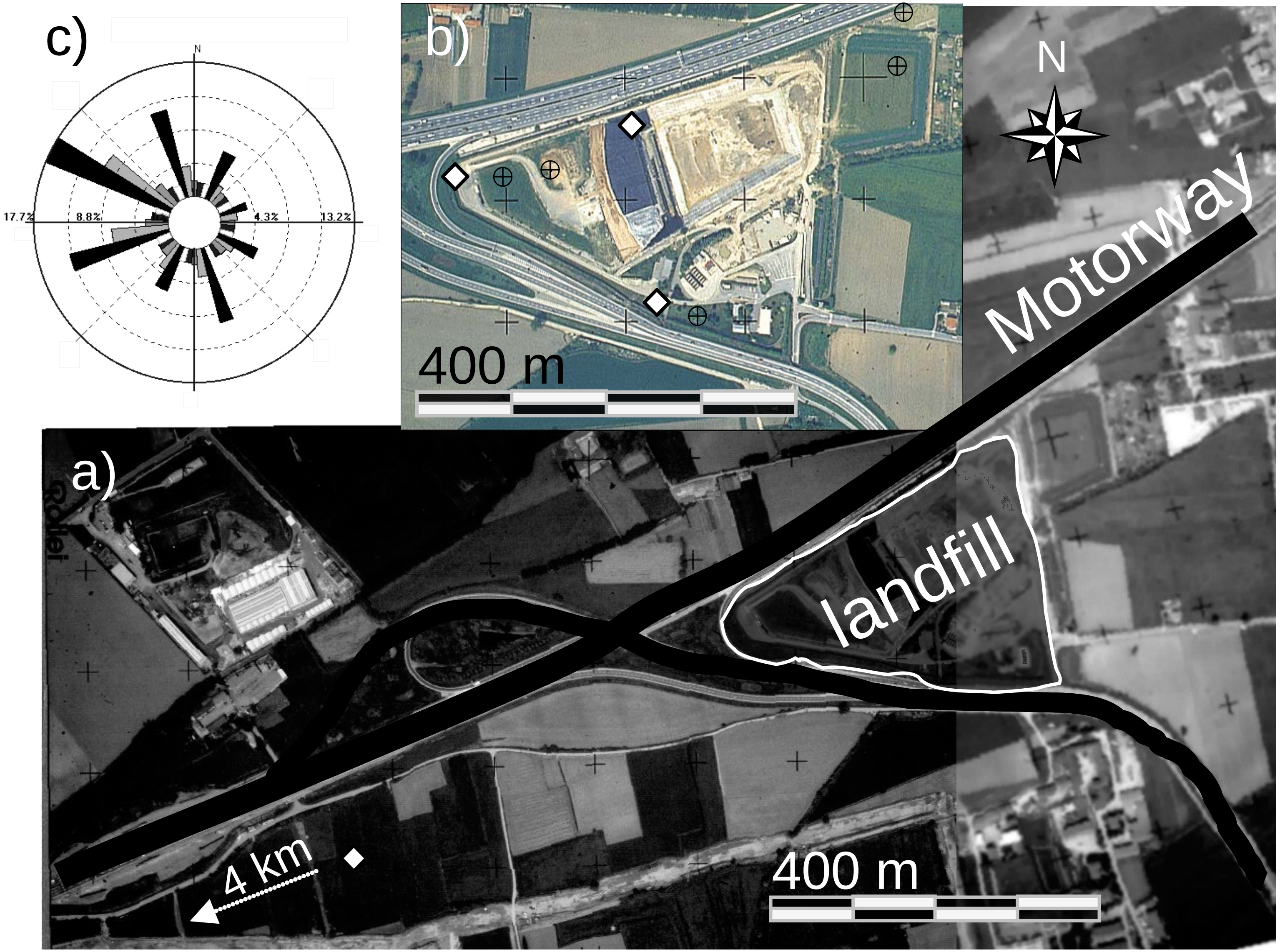


Fig. 2

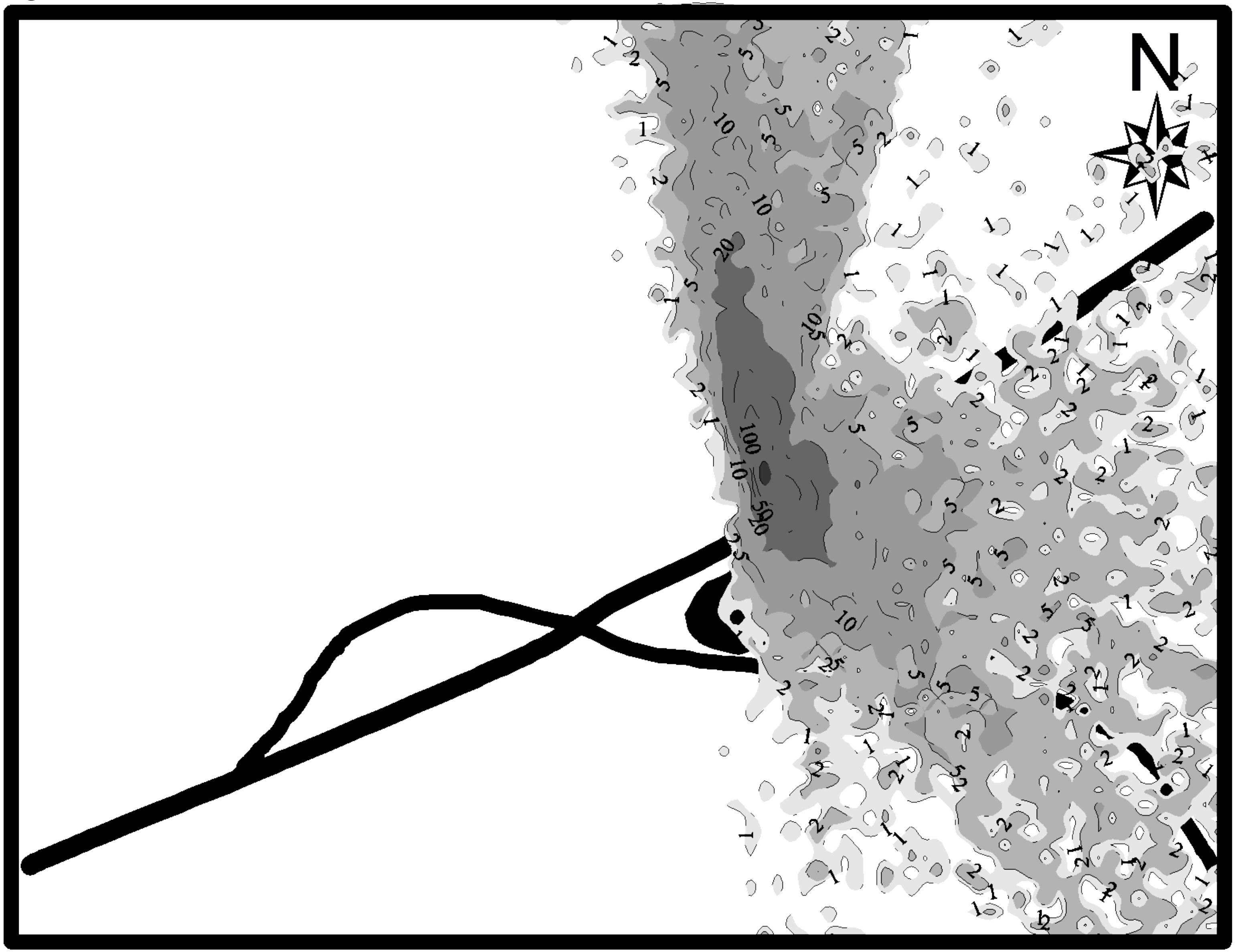
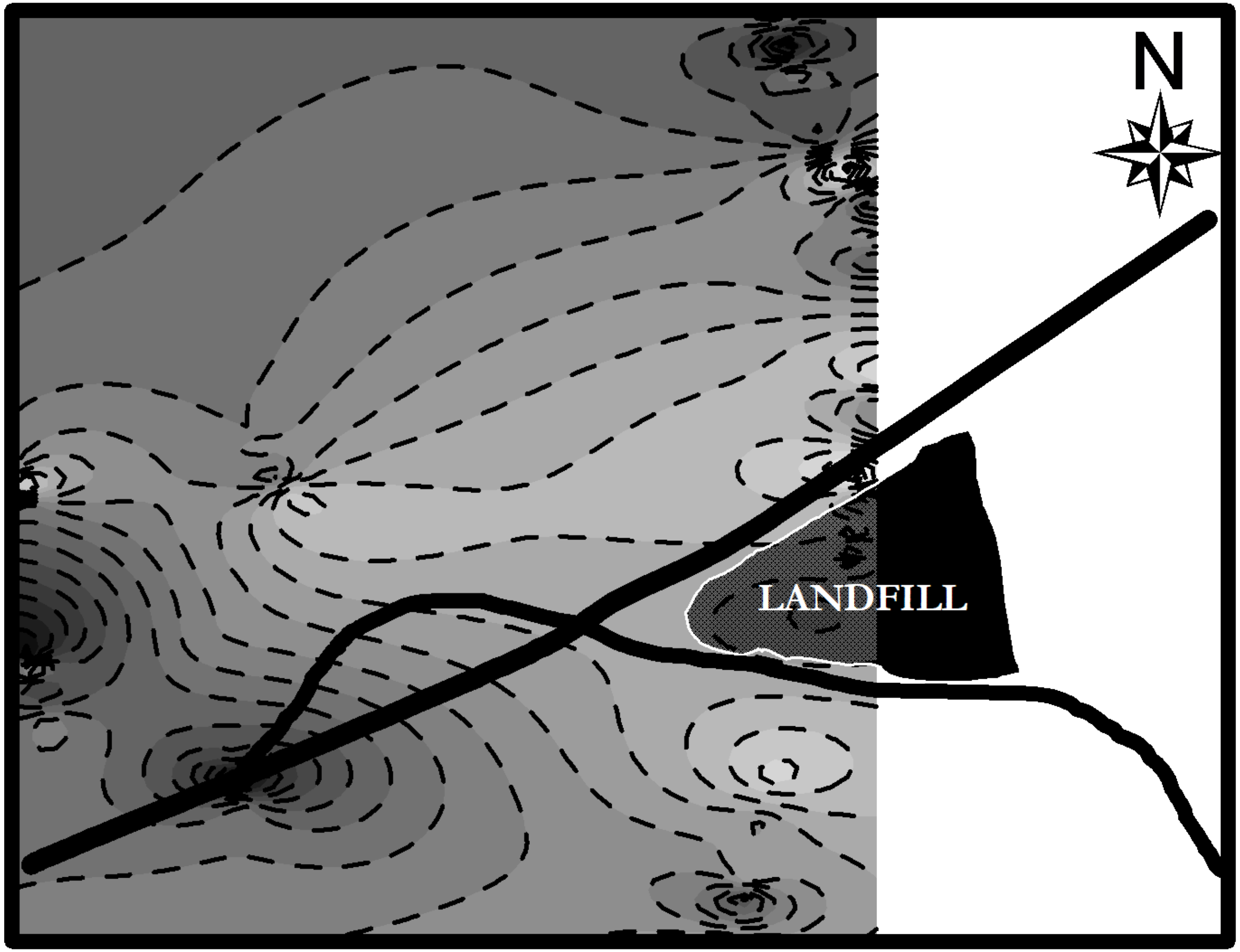
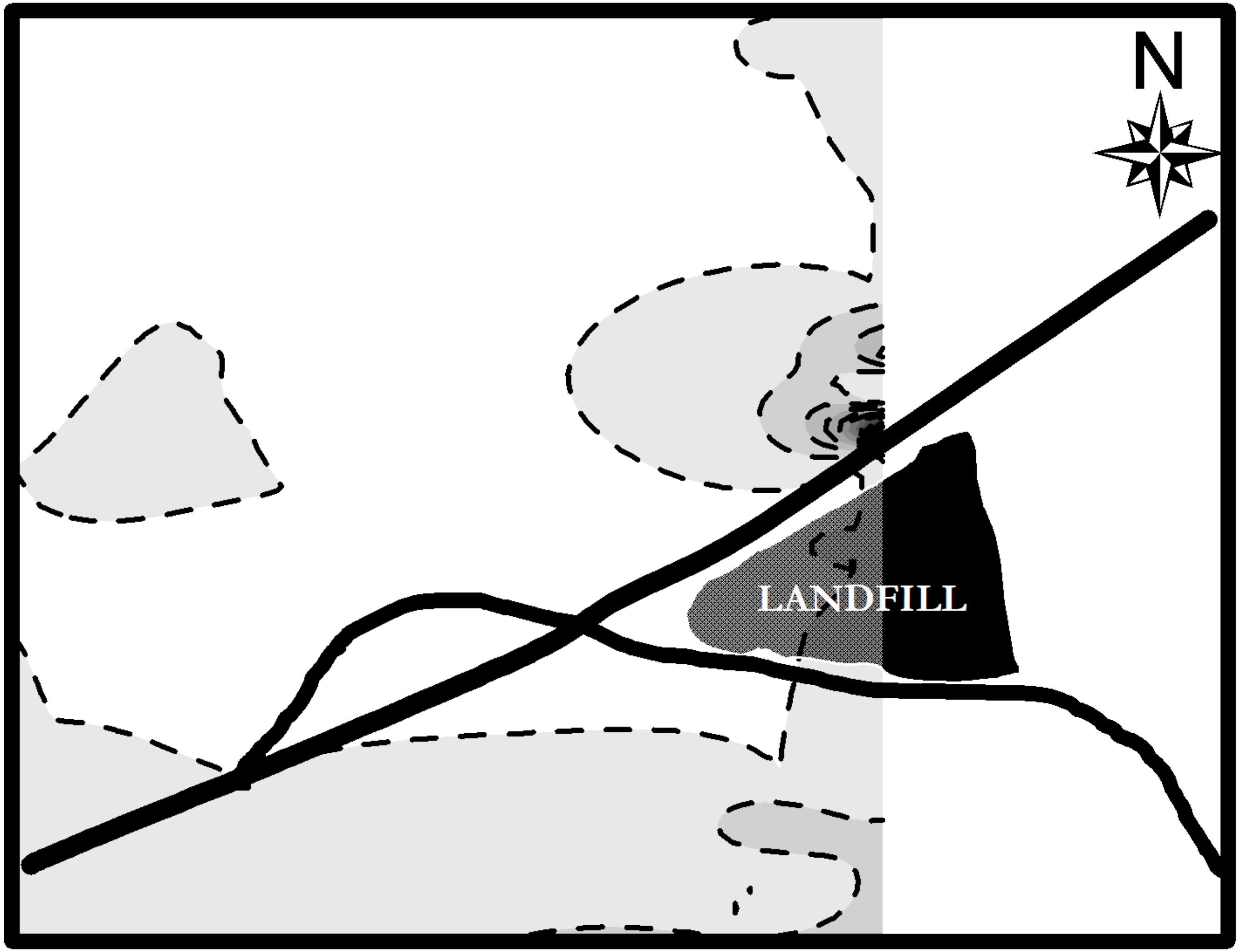


Fig. 3a





525 Fig. 3b

Fig. 4.

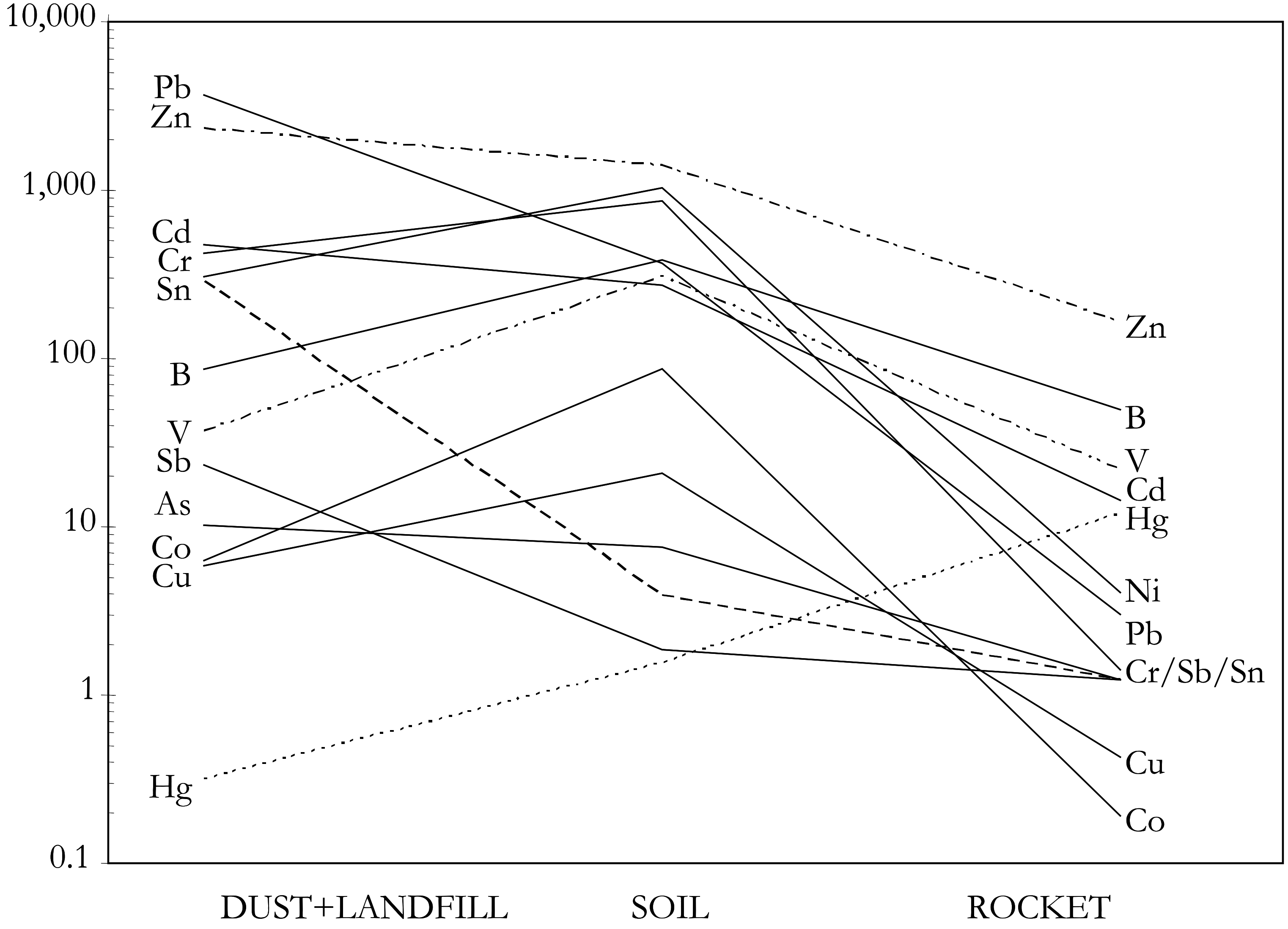

10,000
1,000
100
10
1
0.1
Pb
Zn
Cd
Cr
Sn
B
V
Sb
As
Co
Cu
Hg
Zn
B
V
Cd
Hg
Ni
Pb
Cr/Sb/Sn
Cu
Co
DUST+LANDFILL
SOIL
ROCKET

Table 1

Country	Cr	Ni	Pb	Zn	Cu
Italy	150	120	100	150	120
Sweden	120	35	80	350	100
Netherlands[a]	100/380	35/210	85/530	140/720	36/190
Slovenia[b]	100/150/380	50/70/210	85/100/530	200/300/720	60/100/300
Portugal[c]	200/300	75/110	300/450	300/450	100/200
Spain[d]	250-400 /250-400	80-500 /100-300	250-450 /400-500	300-600 /500-1000	150-300 /300-500
United Kingdom[e]	600-1000	70	500-2000	300	130

[a] “Target” and “intervention” values,
[b] “Limit”, “warning” and “critical” values,
[c] Different values are given for pH below and above 7,
[d] “Research required” values. Ranges instead of single values are given,
[e] “Threshold” values.

Table 2a

Classes of wastes	Quantity	E.U. code
lead metallurgy (1st and 2nd smelting)	16,633	100401
chemical treatment metals surface (non-Fe metallurgy)	21	110104
wastewater treatment (industrial wastes)	18	190804
contaminated soils (with hazardous substances)	1,499	170501
ceramic product with hazardous substances	99	101299
from industrial treatments (2nd, 3rd wastes)	1,594	190301

Table 2b

	Parent material		Road dust		Landfill	
Sb	1	±0.2	1	±1	46	± 5
As	0.8	±0.1	0.2	±0.2	24	± 0.8
Be	0.9	±0.021	n.d.		0.00	± 0.001
B	0.4	±0.02	1.3	±1.10	0.0	± 0.01
Cd	0.1	±0.11	0.03	±0.01	18	± 0.50
Co	51	±17.0	0.3	±0.20	12	± 0.90
Cr	111	±76	5	±3	716	± 34
Ni	863	±93	6	±4	463	± 38
Pb	354	±306	4	±1	4,703	±112
Cu	32	±12	5	±2	715	± 82
Sn	11	±4.5	0.6	±0.1	551	± 47.0
V	54	±9	1	±1	77	± 6
Zn	632	±331	16	±8	2,901	±203
Hg	0.001	±0.005	0.032	±0.045	0.600	± 0.001
Al	633	±881	345	±209	11,870	±654
Fe	994	±1,153	589	±377	24,849	±965
Mn	573	±374	14	±9	2,350	±274

Table 3

	mean	SD	Max	kurtosis
Sb	1	2	15	26
As	<0.5	1	10	54
Be	<0.05	0	0	0
B	73	27	119	1
Cd	6	1	8	3
Co	20	2	24	-1
Cr	146	34	255	2
Ni	175	56	480	17
Pb	71	121	905	45
Cu	55	8	70	-1
Sn	<0.1	3	18	28
V	60	16	90	0
Zn	222	116	690	5
Hg	<0.5	0	2	0
Al	24953	5813	41200	1
Fe	33272	3257	38800	0
Mn	1213	146	1515	-1

Table 4

Factor	Eigenvalue	Total variance	Cumulative variance
1	10.28	64.25	64.25
2	1.79	11.22	75.47

Table 5

	min	Max	median	$MEDIAN_{t_0}$
Sb	0.16	2.82	1.63	*0.10*
As	0.33	11.90	4.60	*0.31*
Be	<0.05	<0.05	<0.05	*<0.05*
B	12	25	18	*12*
Cd	0.41	0.65	0.54	*0.40*
Co	0.15	0.49	0.31	*0.38*
Cr	1.9	4.0	3.2	*0.6*
Ni	0.1	4.9	4.3	*1.4*
Pb	6.5	16.3	10.7	*1.1*
Cu	3.8	6.2	4.9	*4.2*
Sn	0.4	20.9	12.2	*0.4*
V	0.0	90.0	60.0	*6.0*
Zn	2.2	109.0	90.0	*35.0*
Hg	<0.5	4.5	1.1	*<0.5*
Al	108	899	237	*40*
Fe	284	783	476	*100*
Mn	13	35	21	*22*