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Vapor intrusion evaluation for redevelopment of former industrial facilities

G Lonati¹, S Saponaro¹ and E Sezenna^{1,*}

¹ Department of Civil and Environmental Engineering, Politecnico di Milano, P.za L. da Vinci 32, Milano, 20133, Italy

*elena.sezenna@polimi.it

Abstract. Former industrial facilities in our cities are finding new life as they are repurposed for use as nonindustrial commercial or even residential spaces. Often, these buildings are brownfield properties, complicating redevelopment because of past industrial uses that led to soil and/or water contamination at the site. In case site contaminants include volatile chemicals (VCs), concerns about air quality at the site may arise. VCs can vaporize from the subsurface and migrate through soil to outdoor air or into indoor spaces of overlying buildings, where they may accumulate.

Vapour intrusion is a relevant problem especially at sites where building modernization and revitalization is to be carried out, as environmental investigations may be complicated by the structure of the built environment, and the contamination may remain undetected. Moreover, upgrading structures to meet building codes and energy conservation requirements can create tight buildings that may enhance the effects of vapor intrusion.

For accurately predicting whether indoor air quality is being or will be adversely affected by subsurface contaminations a multiple lines of evidence approach should be used. Many issues can in fact introduce uncertainty in predicting indoor air concentrations related to vapor intrusion, including i) sampling and analytical methods ii) modelling of fate & transport from subsurface into building, iii) indoor/ambient background sources.

This work refers about the risk management strategy at a site in Milan (Italy) where soil remediation (excavation and off-site disposal) left a residual volume of soil polluted with petroleum-derived hydrocarbons next to redeveloped buildings. Furthermore, the site was also affected by an extended plume of chlorinated solvents in groundwater from an unknown source, likely external to the site. Indoor air, outdoor air, crawl-space air and soil gas samplings were carried out to collect robust information for evaluating decisions points in the vapor intrusion process. Despite few ambient air and crawl-space measurements resulted in episodic high values, as a general trend, the average outdoor and indoor concentrations did not differ significantly from the background values, suggesting other sources than soil pollution were affecting the quality of the air at the site. Although not strictly necessary in terms of time-averaged health risk, a mitigation system of the residual soil contamination was, however, installed to prevent future uncontrolled exposure.

1. Introduction

Vapor intrusion is the vapor phase migration of volatile contaminants from a subsurface source into overlying buildings or other structures. Vapor intrusion has been recognized as a potential exposure pathway to be considered in risk assessment at contaminated sites to evaluate whether pollution may cause unacceptable risks for human health, thus resulting in the need of remediation activities or pollution confinement.



Potential vapor intrusion may be associated to volatile contaminants (VCs) such as petroleum hydrocarbons, chlorinated solvents, as well as mercury, methane and even radon, an example of natural occurring indoor pollutant.

Until recently, risk of vapor intrusion for VCs was assessed most commonly by means of screening analytical models that allow for the calculation of the pollutant concentrations at the exposure point [1]. The impact of vapor inhalation is directly proportional to the pollutant concentration in the air inhaled indoors or outdoors. However, due to the uncertainty associated with models for the volatilization pathway, field samplings of a vapor phase (e.g., soil gas, ambient air) can be suggested or required to check model predictions [2] [3].

Soil gas measurements allow to exclude the partitioning model in the secondary source, but the results are strongly affected by a number of factors, as soil properties at the monitoring point/depth, soil moisture, temperature, and atmospheric pressure [4] [5] [6]. Ambient air sampling allows to bypass completely the modeling tools, but field data may be affected by background values, local sources unrelated to the subsurface contamination (such as off-gassing from furniture), wind speed (for outdoor measurements), and air conditioning/heating (for indoor measurements) [7] [8] [9]. It is thus important to discriminate air concentrations related to soil and groundwater contamination from those linked to other sources, to avoid unnecessary, inefficient, or unsuccessful remedial actions.

A multiple-line-of-evidence approach is typically recommended [1] [10], with models used in conjunction with site investigation data. In this approach, no single investigation result is considered definitive; rather, all available results are evaluated to determine whether the pathway is likely to be complete or not. Although a wide variety of investigation methods may be included in the multiple-lines-of-evidence approach, VCs concentrations in soil and groundwater, in soil gas, and in both outdoor and indoor air typically constitute the primary lines of evidence considered [1] [2]. When VCs are detected in one or more of these media at concentrations exceeding screening levels, the key challenges for the assessment of the exposure pathway include: temporal and spatial variability in VCs concentrations, and the estimation of the contribution of other sources of VCs (i.e., indoor and ambient sources).

This work reports the approach adopted and the lesson learned in the risk assessment procedure and risk management at a redeveloped former industrial site in Milan (Italy), where soil remediation (excavation and off-site disposal) was partially carried out in 2005, leaving on-site a residual volume of soil polluted with petroleum-derived hydrocarbons. In addition, groundwater at the site was also affected by an extended chlorinated aliphatic compounds (CAHs) plume, especially perchloroethylene (PCE) and trichloroethylene (TCE), probably originated from external areas. CAHs showed complicated contamination patterns and groundwater data alone, as often occurs, were inconclusive, as limited by several on-site and off-site structures and seasonal fluctuations, for the correct delineation of the contaminated plume as well as for source identification.

2. Materials and methods

2.1. Site description

The field study site, approximately 24000 m², is a former metal factory recently redeveloped as a part of a more extended service-sector area of the city (Figure 1). The southern part of the site is used as a parking area, whereas the northern part has a recreational use; a building ("A" in figure 1) is located and already used in the centre of the site. Between the existing building and the parking area, another building ("B" in figure 1) has been recently built. A railway line runs alongside the eastern boundary of the site.

The local geology of the site, assessed with more than thirty boreholes at maximum depths between 4 and 25 m below ground surface (b.g.s.), results in the following stratigraphy: i) pavement, with an average thickness of about 0.5 m; ii) sandy silt with pebbles fill, down to a depth between 1.5 and 7.5 m b.g.s.; iii) gravelly sand or sand with gravel, in the remaining investigated thickness.

Groundwater flows from N/NNW to S/SSE (Figure 1), with hydraulic gradient in the 0.2-0.3% range. The average water table elevation at the site, as a result of reduced groundwater withdrawal in the area,

has raised up from about 20 m b.g.s. in 2004 to about 15 m b.g.s. in 2013 - 2017, with yearly fluctuations lower than 3 m.

Over the period of time that the site operated, the soil became contaminated mostly with mono- (BTEXS: benzene, toluene, ethylbenzene, xylenes, styrene) and poly-aromatic hydrocarbons (PAHs: benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(g,h,i)perylene, chrysene, dibenz(a,e)pyrene, dibenz(a,h)anthracene, indeno(1,2,3-cd)pyrene, pyrene and mostly acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, naphthalene) and total petroleum hydrocarbons, with reference to the Italian regulatory limits for residential areas. Few vadose soil samples, close to SG7 and SG15, exhibited also tetrachloroethylene (PCE) concentrations (up to 1.3 mg/kg) slightly exceeding the Italian regulatory limit for residential areas (1 mg/kg).

Groundwater samples, from few onsite monitoring wells, showed concentrations above the limits for PCE, TCE, hexachlorobutadiene (HCB) and trichloromethane (TCM), with little to no evidence of occurring degradation. Chlorinated aliphatic hydrocarbons (CAHs) were also known to affect groundwater at large scale in the urban setting, with values similar to those observed at the site, even though limited data did not let to discriminate between possible local sources and anthropic background values.

Partial remedial activities for soil (hot-spots excavation and off-site disposal) were carried out in 2005, leaving variously distributed residual soil contamination between 0.8 and 22 m b.g.s., to be addressed with site specific risk assessment.

2.2. Field activities

For the management of residual soil contamination and the CAHs dissolved plume, groundwater, soil gas and air monitoring was initiated in 2009 with the twofold objective of i) collecting data for human health risk assessment, and ii) to investigate, especially by means of groundwater monitoring and soil gas surveys, the possible presence of local unrecognized CAHs residual sources.

Indoor air, outdoor air, crawl-space air, and soil gas samplings had been carried out since January 2009; however, in March 2011 the monitoring network was upgraded to its final configuration (figure 1) consisting in 60 soil gas probes (15 at 1 m b.g.s., 17 at 4 m b.g.s., 14 at 10 m b.g.s. and 14 at 15 m b.g.s.), 18 indoor, 7 outdoor and 4 crawl-space sampling locations including background monitoring points ("Fondo OUTn" for outdoor air, "DIGn" and "ENn" for indoor air).

The dataset resulting from monitoring in the period March 2011–July 2012 was used in risk assessment (4 seasonal campaigns to represent the mean annual situation) and to confirm the procedure results over time; after that, however, soil gas ambient and crawl-space air were monitored three times a year (October 2012; January, May, and October 2013; March, July, and November 2014; February, June, October 2015; March, July, and November 2016, 2017 and 2018).

Although not strictly necessary in terms of time-averaged health risk, in fact, it was deemed convenient to mitigate the residual soil contamination, especially in the central portion of the site; after pilot tests, in December 2013 a full-scale Soil Vapor Extraction (SVE) plant was designed and installed at the site. It consists of two pairs of extraction wells, SV1A and SV2A, screened between 1 and 8 m b.g.s., and SV1B and SV2B, screened between 8 and 15 m b.g.s., as shown in figure 1. Each couple "A" and "B" of extraction wells is connected to a moisture separator and a blower, applying vacuum to the soil matrix to create a negative pressure gradient that causes movement of soil gas with VOC vapors towards the extraction wells, where they are extracted from the subsurface and conveyed to an above ground treatment unit. Until March 2015, the off-gas treatment was performed by catalytic oxidation and activated carbon adsorption; since then, the catalytic burner has been bypassed, due to a significant decrease in the pollutant concentrations in the extracted flow. Except for short maintenance periods, the plant continuously operated until April 2016, when it was turned off for about one year for rebound testing.

SVE technology has been proven effective to reduce soil concentrations of VCs at contaminated sites; additionally, because soil gas is drawn through vacuum towards the extraction wells, the SVE

system may concurrently reduce or prevent vapour intrusion into neighboring buildings through the same mechanisms.

In 2011, with the aim of assessing the CAHs dissolved plume outline at the site, new groundwater monitoring wells were also placed at the eastern boundary of the site (EP1-EP10) and existing piezometers in the neighborhoods were included in the monitoring activities, leading to groundwater network setup of 22 piezometers, screened between 10-20 m b.g.s.. Despite the pattern of groundwater contamination suggested an external source to the CAHs plume, groundwater data alone were still inconclusive as limited by several on-site and off-site structures (e.g. buildings and the railway) and seasonal fluctuations. Measurements of CAHs concentrations in both groundwater and soil gas allowed to optimize the placement of two further groundwater monitoring wells outside the area (EP11-EP12) to draw conclusions about the external origin of the groundwater contamination.

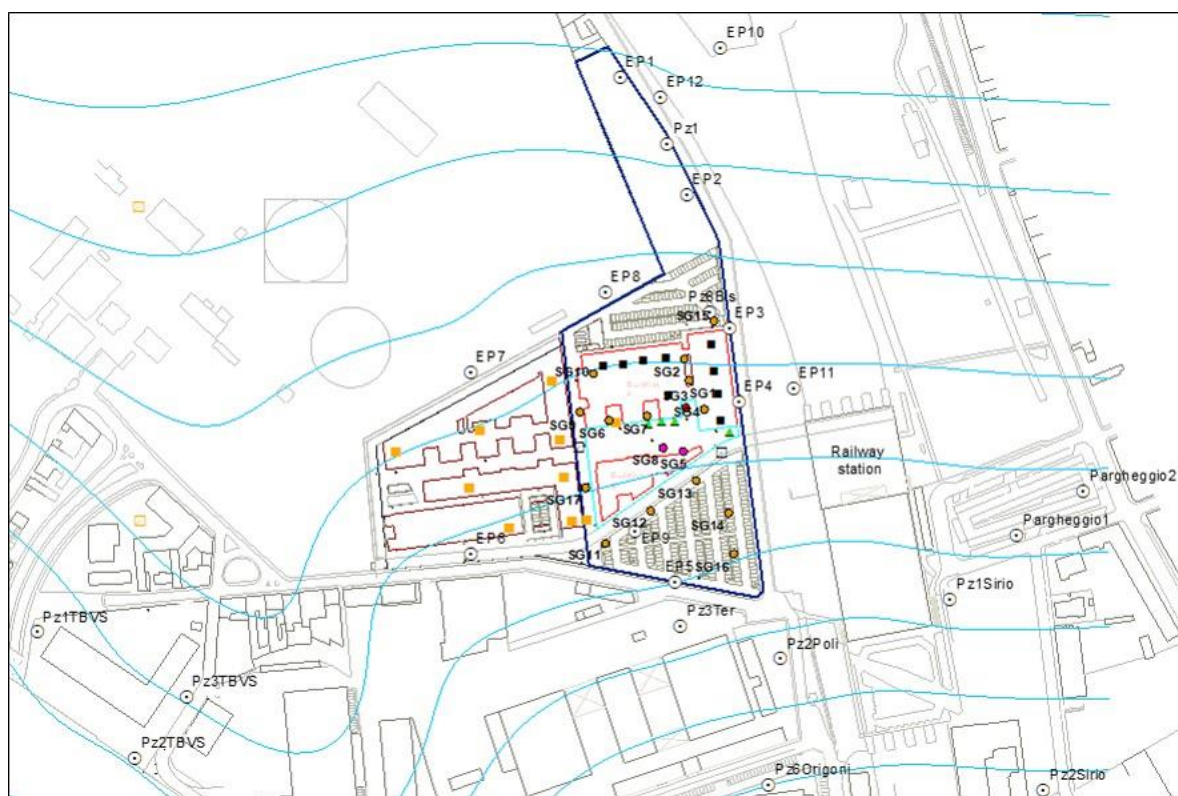


Figure 1. Site map. —: site boundary; —: vapor-tight membrane boundary ●: soil gas probes at 1, 4, 10 and 15 m b.g.s.; ●: soil gas probes at 1 and 4 m b.g.s.; ●: soil gas probes at 4 m b.g.s.; ⊙: groundwater monitoring wells; —: contour mapping of the water table elevation (spacing 0.2 m) ■: indoor monitoring locations at the site; □: outdoor monitoring locations at the site; ▲: crawl-space monitoring locations; ■: indoor background monitoring locations; □: outdoor background monitoring locations.

2.3. Analysis

All the samples (groundwater, air, and soil gas) were analyzed for BTEXS, petroleum hydrocarbons, PAHs, and selected CAHs, principally PCE, TCE, HCB, TCM, and possible metabolites based on detections reported during historical site sampling activities.

Groundwater samples were collected according to low-flow purging and sampling method (USEPA, 1996). Depth to groundwater and indicator field parameters (pH, temperature, dissolved oxygen, conductivity and oxidation-reduction potential) were registered during purging. Groundwater samples were collected in 40 ml glass vials with Teflon-lined septa and no headspace for the analysis of VOCs and in 1 L amber glass bottles for total hydrocarbons and PAHs quantification. Samples were shipped

on ice to the laboratory and analyzed according to the USEPA Method SW8260 for selected CAHs, USEPA 8270 D:07 for PAHs and ISO 9733 for total hydrocarbons.

Active sampling of the vapor phase was performed at the flow rate of 1 l/min for 4 to 7 h. Hydrocarbons were fractionated with a modified MADEP approach. Activated carbon sorbent tubes (ORBO 32, 400/200 mg, Sigma Aldrich) and XAD-2 (ORBO 609, 400/200 mg, Sigma-Aldrich) to capture VOCs (benzene, toluene, ethylbenzene, xylenes, MADEP Aliphatic C6-C8 and C9-C10, MADEP Aromatic C9-C10, PCE, TCE, and HCB) and semi-volatile organic compounds (SVOCs: PAHs, MADEP Aliphatic C10-C18), respectively. A PTFE filter (37 mm diameter, 0.5 μm pore size) was placed in the sampling line to remove water vapor.

VOCs were extracted from the sorbent cartridges with 2 ml carbon disulfide, separated chromatographically with a Petrocol DH column (50 m x 0.2 mm internal diameter (i.d.), 1.5 μm film thickness) and quantified by gas chromatography-mass spectrometry - GC/MS (Trace DSQ, Thermo Corporation) operating in selected ion monitoring mode with the following instrumental conditions: 220 $^{\circ}\text{C}$ split injector temperature, 35 $^{\circ}\text{C}$ initial oven temperature (held for 5 min), 220 $^{\circ}\text{C}$ final oven temperature (held for 15 min), oven temperature ramp 8 $^{\circ}\text{C}/\text{min}$.

SVOCs were separated with an Equity-5 column (30 m x 0.25 mm i.d., 0.25 μm film thickness) and quantified by GC/MS in selected ion monitoring mode with the following instrumental conditions: 280 $^{\circ}\text{C}$ split/splitless injector temperature, 50 $^{\circ}\text{C}$ initial oven temperature (held for 1 min), 50 $^{\circ}\text{C}$ to 150 $^{\circ}\text{C}$ at 12 $^{\circ}\text{C}/\text{min}$, 150 $^{\circ}\text{C}$ to 290 $^{\circ}\text{C}$ at 7 $^{\circ}\text{C}/\text{min}$, 290 $^{\circ}\text{C}$ final oven temperature (held for 15 min).

2.4. Risk assessment

Values for the physical-chemical and toxicological properties were taken from the Italian Health Institute database [11] and [12] or, whenever not reported, from RISC 5.0 database [13].

For each monitoring campaign, carcinogenic risks and hazard indices were estimated according to the RBCA ASTM standard [14] from the representative soil gas concentrations at the different depths, which were calculated with ProUCL 4.00.02 as the Upper Confidence Limit 95% of the pollutant concentrations measured in: i) SG1 to SG10, SG13 and SG15, for indoor exposure in building "A"; ii) SG1 to SG14, for indoor exposure in building "B"; iii) SG1 to SG17, for outdoor exposure. Soil was assumed as homogeneous sandy gravel (0.25 total porosity, 0.10 water-filled porosity). Wind velocity (0.8 m/s) was calculated as the average value over a ten-year period of measurements at the nearest meteorological station. For the building foundation parameters affecting vapor intrusion and exposure parameters the default values reported in [15] were used.

3. Results and discussion

3.1. Field data

Figures 2 and 3 summarize respectively the soil gas concentrations (as a function of depth) and the crawl-space and outdoor indoor air concentrations for some selected pollutants, either associated to the residual contamination in soil (BTEX, naphthalene, and hydrocarbons fractions aliphatics C6-C8, aliphatics C9-C18) or to the groundwater plume (PCE, TCE, DCM and HCB).

3.1.1. Soil gas. The comparison between the mean values in soil gas and air points out that the concentrations in soil gas are at least one order magnitude higher than in air. For the soil pollutants (e.g. BTEX, TPH, naphthalene) the concentrations in the soil gas show minimal differences in 10-15 m and 4 m b.g.s. probes, whereas a net reduction (about two orders of magnitude) occurs between 4 m and 1 m b.g.s. samples; such a behavior is due to the residual contamination that extends from approximately 5 m b.g.s. downwards. Lower concentrations in the 1 m b.g.s. probes reflect attenuation due to upward diffusion into the atmosphere and to biodegradation.

Differently, CAHs concentrations show a clear and gradual decrease with depth, confirming the groundwater plume as the major source in the soil gas. In case of residual soil contamination, a profile with depth similar to that of BTEXS or total petroleum hydrocarbons (TPH) would have been observed.

In the monitoring period, groundwater CAHs concentrations at the site remained fairly constant, with up to one order of magnitude maximum variation in each single monitoring well. No systematic seasonal effect was observed, except for a sporadic rise of PCE levels in groundwater in March 2011 (up to 500 $\mu\text{g/l}$ in the upgradient well Pz1). This pulse was confirmed by high PCE levels (up to about 4 mg/m^3) observed in March and July 2011 in the deep soil gas probes (SG1 – SG15) of the monitoring area downgradient Pz1. At single monitoring point, soil gas concentrations in the deep probes (10-15 m) displayed a variability generally limited to one order of magnitude, whereas higher variations, up to about two orders of magnitude, were registered for the 1 m and 4 m probes.

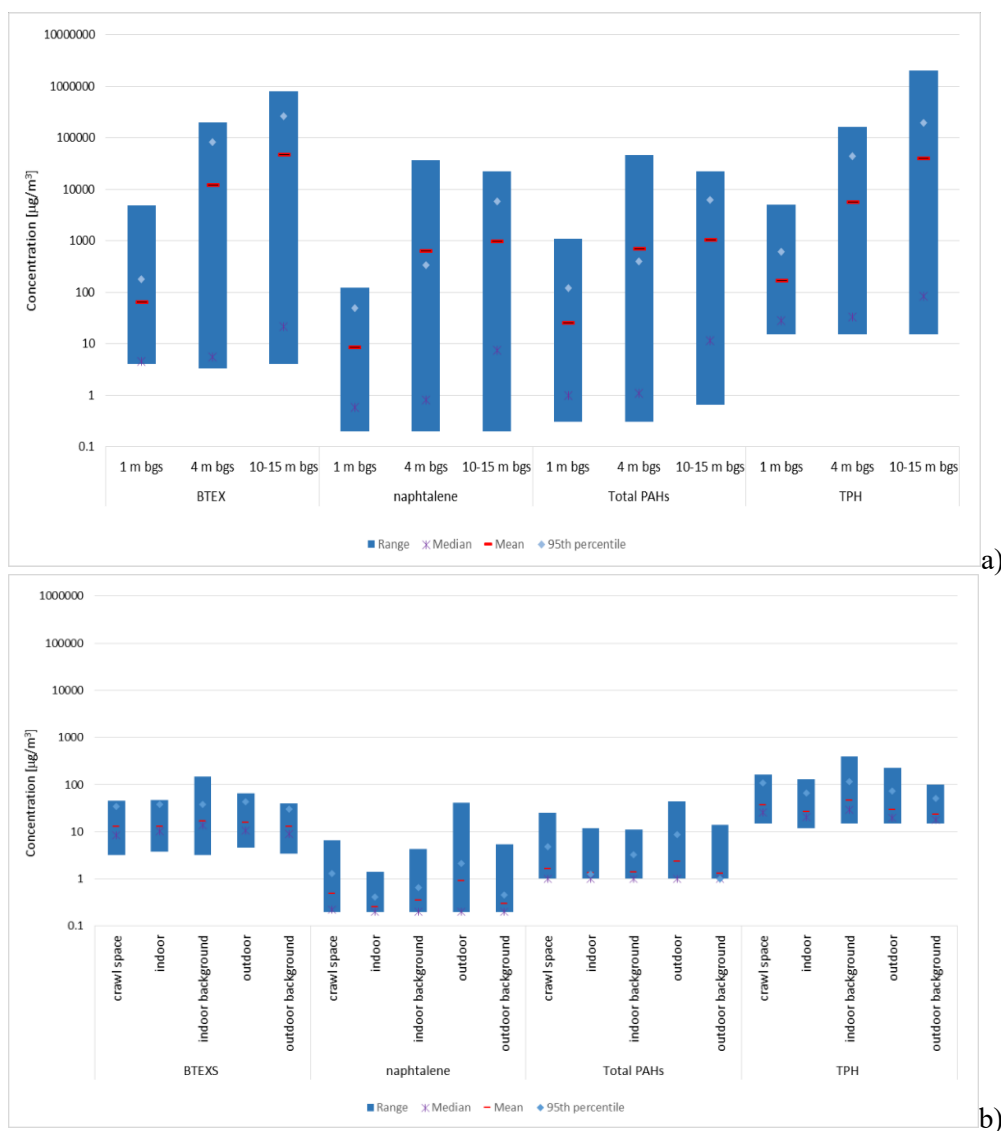


Figure 2. Summary of soil gas, at the different depths (a) and air – indoor and outdoor (b) concentrations for selected compounds associated to the residual soil contamination at the site.

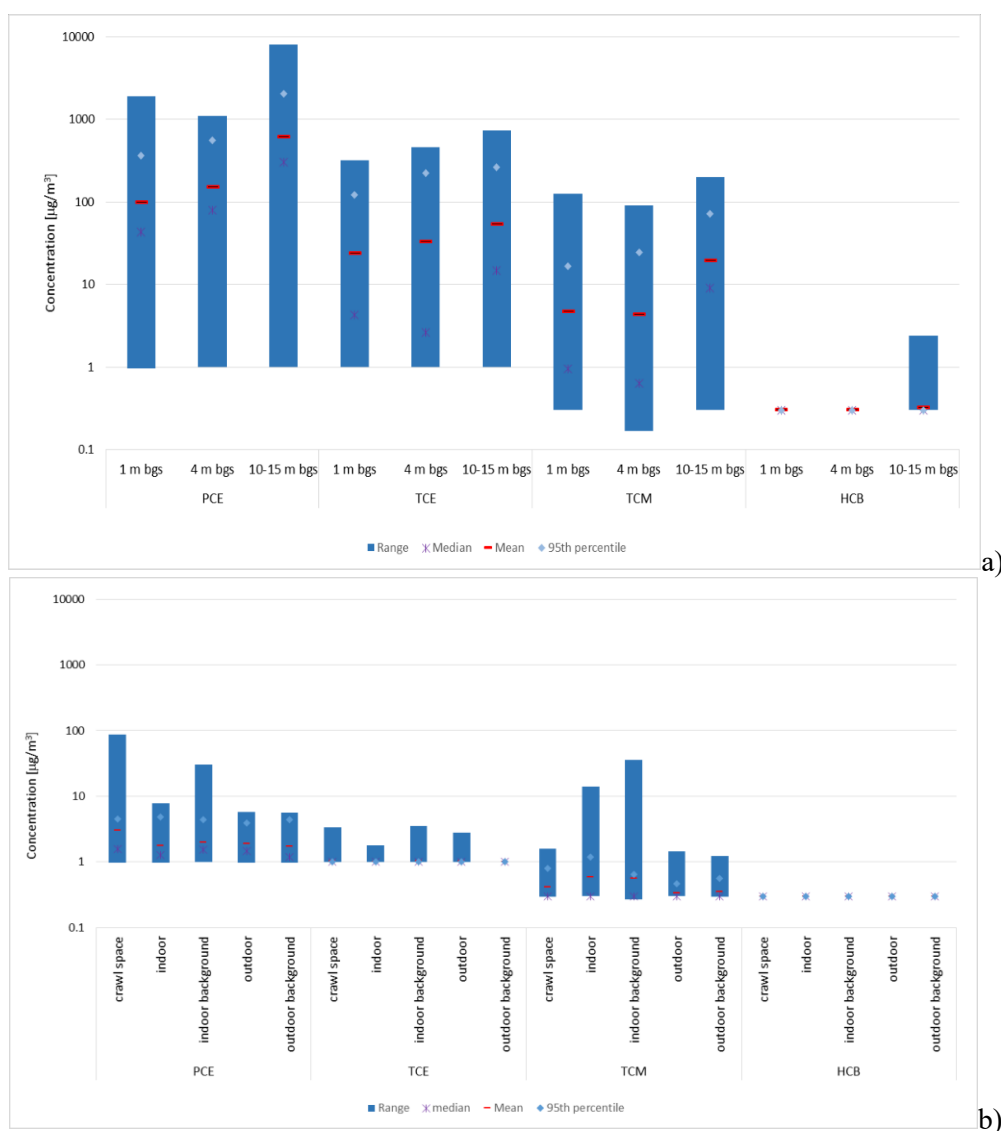


Figure 3. Summary of soil gas, at the different depths (a) and air – indoor and outdoor (b) concentrations for selected chlorinated compounds associated to the groundwater plume.

Spatially, PCE concentrations in soil gas and groundwater showed a similar pattern, with the highest soil gas concentrations recorded in the probes in the eastern part of the site overlying the area where PCE in groundwater was at its highest levels (figure 4, showing, as an example, the situation in October 2012).

Figure 5 shows the average PCE, TCE and HCB groundwater concentrations in the monitoring wells from different areas of the site with respect to N-S, main groundwater flow direction. Despite all dissolved PCE concentrations ranged in 10 – 100 $\mu\text{g}/\text{l}$ on average, it is evident a higher PCE area at the eastern boundary of the site. In case of TCE and TCM (data not shown), the average concentrations were similar in all the groundwater wells, with no high concentration area or significant difference between upgradient and downgradient locations. HCB was correlated to high PCE values, being HCB quantified only in the high PCE wells (with the exception of a single detection in PZ1 TBVS). In figure 6 the average soil gas concentrations for PCE and TCE at the different soil gas probes in 2009-2013 are shown. HCB is not reported as it was mostly below the detection limit (0.3 $\mu\text{g}/\text{m}^3$); actually, it was quantified (up to 2.4 $\mu\text{g}/\text{m}^3$) only in the deep soil gas probes (10-15 m) in SG15 and SG2 in 2011. TCE concentrations in the soil gas were typically below 100 $\mu\text{g}/\text{m}^3$, except for the parking area (SG13, SG14,

and SG16), where traces CAHs were found in soil. Similar to groundwater data, PCE in the deep probes (10-15 m b.g.s.) at the eastern portion of the site was on average above 1000 µg/m³, that is 3-10 times higher than in the remaining portions of the site.

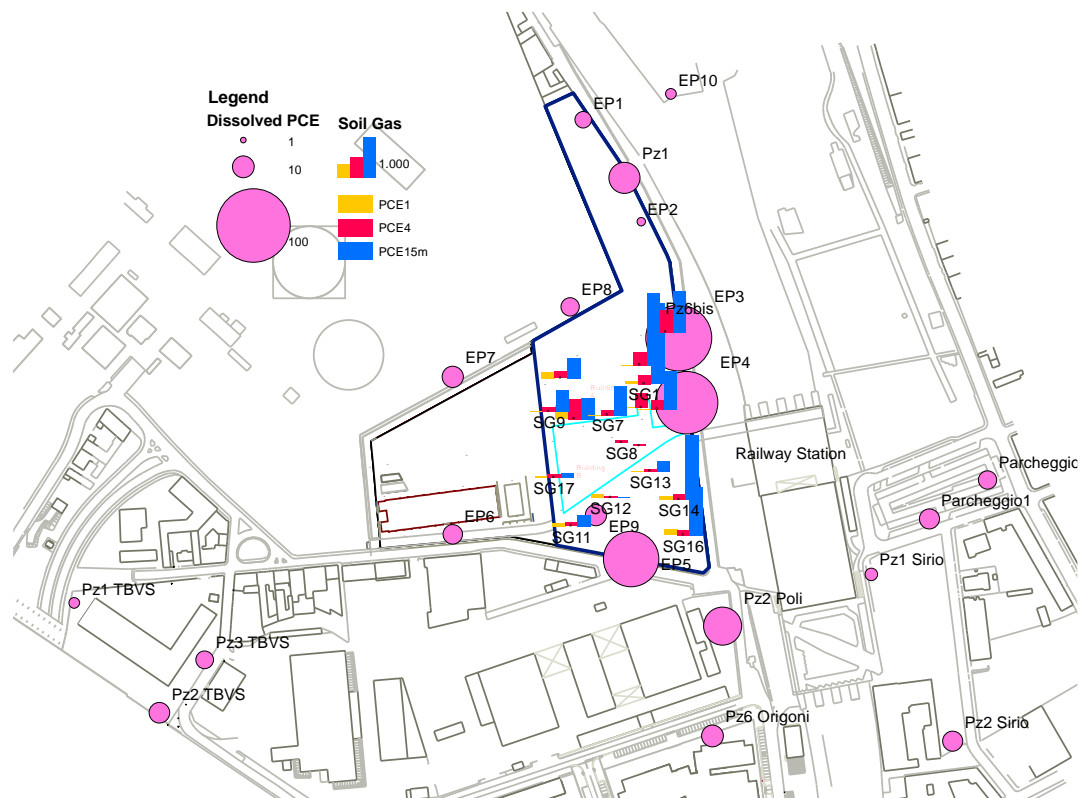


Figure 4. PCE concentrations in groundwater and soil gas at different locations and depths (October 2012).

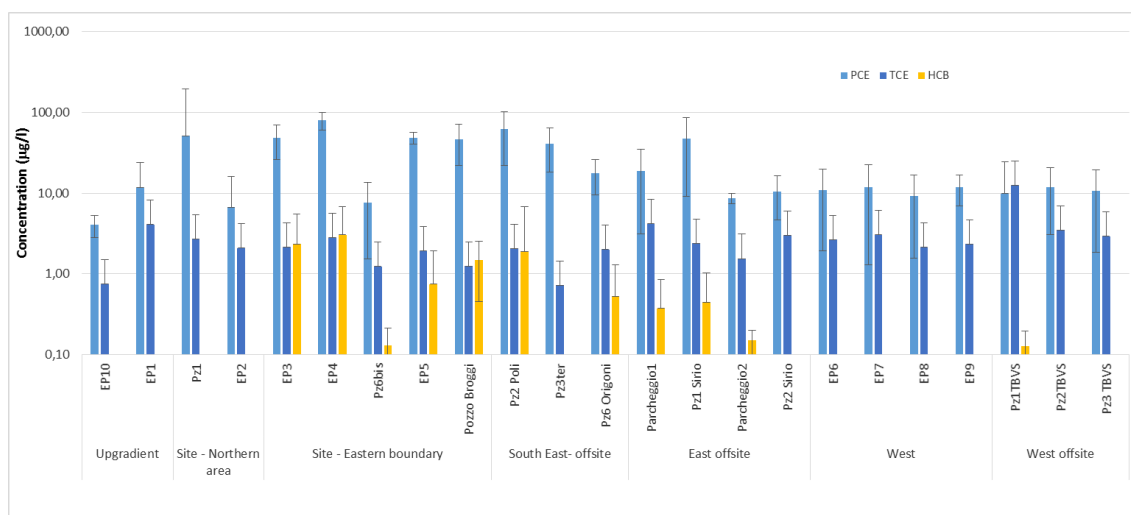


Figure 5. Average groundwater concentrations for PCE, TCE and HCB at monitoring wells in 2009-2013. The error bars show the standard deviation of data.

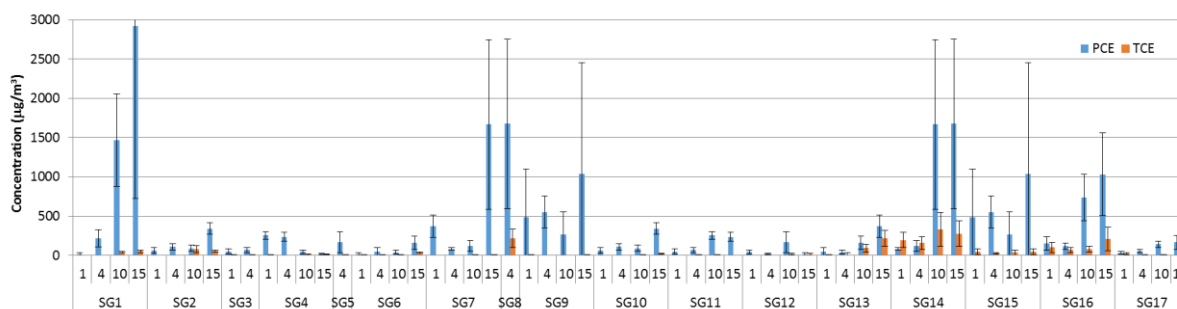


Figure 6. Mean soil gas concentrations of PCE and TCE in the different probes and depths (period 2009-2013). The error bars show the standard deviation of data.

The groundwater and soil gas data pointed out that CAHs contamination was probably caused by an off-site cross gradient source located somewhere in the railway area. Based on such information a new piezometer EP11 was installed next to the railway area; PCE concentrations registered in 2014-2018 in EP11 were constantly above 150 µg/l, about 2-3 fold the concentrations in the piezometers EP4 and EP5, confirming the external origin of the CAHs contamination.

3.1.2. Air. Despite the large variance of the datasets, on average the outdoor and indoor concentrations recorded at the site were in line with the measured local outdoor and indoor background levels, respectively. Moreover, for all the pollutants, the outdoor and indoor levels are comparable. The indoor values, as shown in table.1, are also within the typical ranges of literature indoor background values reported by USEPA [16].

Table 1. Comparison of the indoor air concentrations at the site with USEPA background values

Volatile Chemical	Indoor Air Concentrations at the site (n= 229)			¹ Range of 50 th percentile background indoor air concentrations [16]
	average ± dev. st.	% detect	Range	
Benzene	1.50 ± 1.37	96.9	<0.2-7.2	<RL-4.7
Ethylbenzene	1.26 ± 0.55	39.3	0.96-4.12	1-3.7
Toluene	6.45 ± 5.32	98.3	1-26.8	4.8-24
Xylenes	3.71 ± 2.80	93.4	0.95-13.3	2.6-17.6
Naphthalene	0.25 ± 0.16	36.2	<0.2-1.43	0.18-1.7 ⁽²⁾
Total Petroleum Hydrocarbons	26.7 ± 17.0	80.3	<13-130	116-594 ⁽³⁾
Tetrachloroethylene	1.77 ± 1.23	58.5	<1- 7.8	<RL-2.2
Trichloroethylene	1.0 ± 0.05	4.8	<1- 1.8	<RL-1.1
Trichloromethane	0.68 ± 0.42	37.1	<0.3- 3.1	<RL-2.4
Hexachlorobenzene	<0.3	0	<0.3	-

⁽¹⁾ <RL below reporting limit [16].

⁽²⁾ urban houses [17].

⁽³⁾ range TPH 50th to 90th percentile [18]

Throughout the years, for TPH and BTEX it was possible to identify the summer (July) as the period with the lowest average air concentrations. As a general trend, for all pollutants the highest average values were measured from October to March. These results suggest that, though the temperature in July is higher than in the other monitoring periods and might enhance emission from soil, the worst air quality is found during the cold season, when the atmospheric conditions promote stagnation and sources other than pollution in soil (vehicular traffic, space heating systems) are active.

3.2. Risk assessment

Figure 7 shows the cumulative carcinogenic risk and hazard index values for indoor exposure in building "A" or "B" and outdoor exposure at the site, based on soil gas concentrations at the different depths. Data collected at 1 m b.g.s. resulted in calculated values lower than those based on data from deeper soil gas probes, with the difference up to about two orders of magnitude for indoor exposure.

This result was determined by the assumption of homogeneous features of the soil in the screening diffusion model and by neglecting any pollutant degradation. Actually, both soil lithology and degradation result in lower concentration levels in soil gas samples taken closer to the ground surface. Running biodegradation was confirmed by the low O₂ and the high CO₂ concentrations measured at 10 and 15 m b.g.s. in many monitoring probes below or near the building (data not reported).

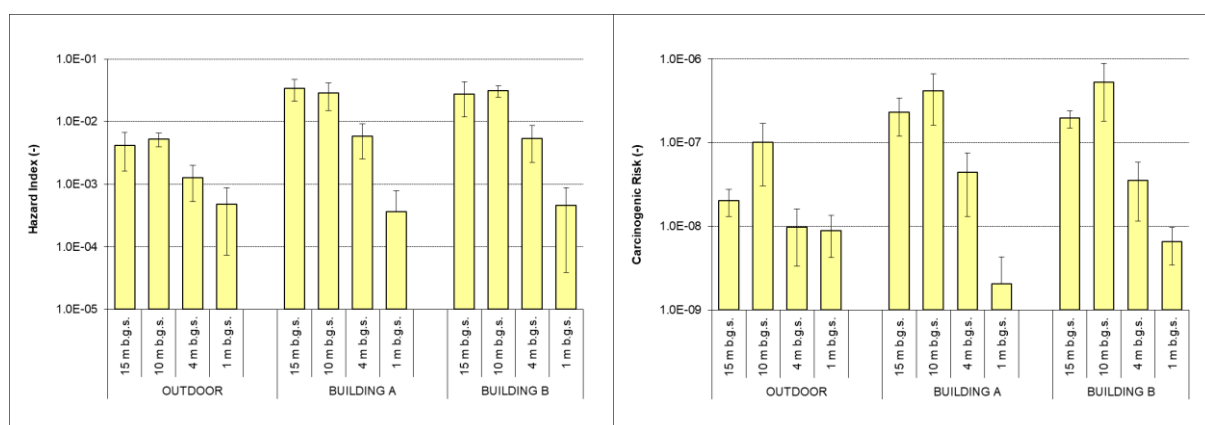


Figure 7. Annual average cumulative carcinogenic risk and cumulative hazard index for outdoor and indoor exposure in building "A" and "B", based on soil gas concentrations at the different depths in 2011-2012.

3.3. SVE operation

Soil gas monitoring during initial SVE operation showed up to two orders of magnitude reduction in soil gas concentration at each monitoring point, with the exception of the samples at 10-15 m b.g.s. in selected vapor monitoring probes (figure 8, naphthalene as an example). Prior to system shutdown for rebound testing at the end of June 2015, the SVE wells appeared to reach a mass removal asymptote for this extraction configuration, which is the primary criterion for initiating the rebound test (Figure 9). After the SVE shutdown VCs levels did partially rebound in several soil gas monitoring points and thus the SVE system was switched on, and nowadays a second shutdown test is ongoing.

Air data, however, showed no relevant variation with the SVE system switch on/shutdown. This confirms that air quality at the site was most affected by other environmental sources than soil and groundwater contamination. Nevertheless, SVE system was effective in addressing residual soil contamination, as evidenced by the lower concentrations prior to SVE system shutdown, along with the slow rebound in soil gas concentrations following the shutdown.

4. Conclusion

Health risk calculations at the site resulted in a cumulative carcinogenic risk up to $1.0 \cdot 10^{-6}$ (indoor exposure based on soil gas data at 10 m b.g.s. in October 2011) and a cumulative hazard index up to about 0.05 (indoor exposure based on soil gas data at 15 m b.g.s. in March 2011). Estimates lower by about two orders of magnitude were obtained when data at 1 m b.g.s. were used for calculations. A few outdoor air and crawl-space measurements resulted in episodic high values. However, as a general trend, the average outdoor and indoor concentrations did not differ significantly from the background values. The worst air quality was found in autumns and winters, when the atmospheric conditions promote stagnation and sources other than pollution in soil (vehicular traffic, heating systems) are active. Compared with background values, indoor and outdoor air concentrations at the site suggested that hydrocarbons were widely spread and sources other than pollution in soil were affecting the quality of

the air inhaled at the site. Although not strictly necessary in terms of time-averaged health risk, the SVE system allowed to extract significant amounts of mono- and light poly-cyclic aromatic hydrocarbons. This has helped manage concerns about future uncontrolled exposure of people working at the site. Repeated soil gas sampling at different locations and depths provided also information about the concentrations gradient, which was useful to distinguish between background and point sources. In all soil gas monitoring wells, CAHs concentration increased with depth, indicating groundwater as the major source of vapors at the site. Moreover, soil gas and groundwater concentrations showed a similar pattern, with the highest soil gas concentrations recorded in the probes overlying highly contaminated groundwater. Soil gas data were thus useful for corroborating groundwater data and addressing site investigations. The data suggested that groundwater contamination at the site was to be traced to the diffuse plume (background PCE values about 15 $\mu\text{g/l}$), even if a local discernible plume of PCE from an off-site cross gradient source, responsible for the higher PCE concentrations (45-80 $\mu\text{g/l}$) in the eastern part of the site, was recognized in the railway area.

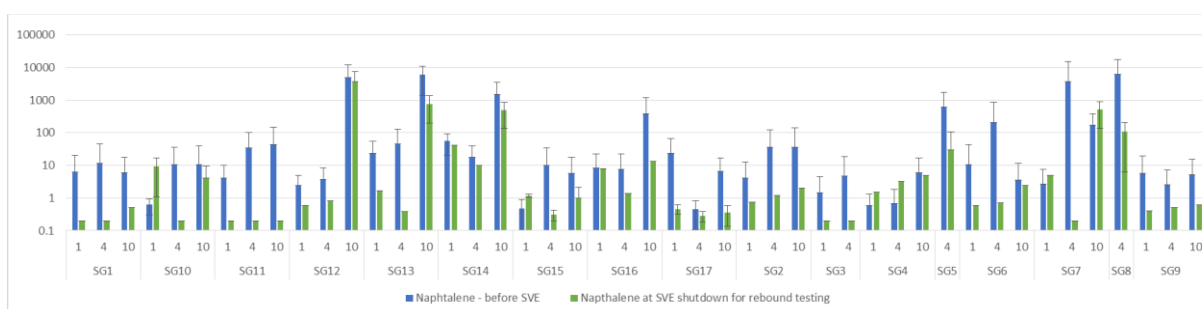


Figure 8. Naphtalene concentrations in soil gas before and during SVE operation at the site.

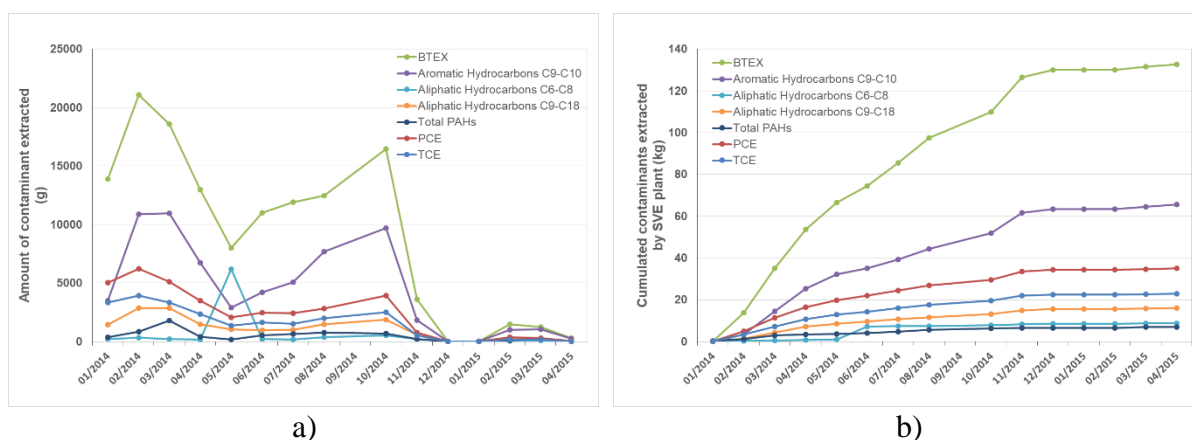


Figure 9. Contaminants extracted with the SVE system, on a monthly basis (a) and as the cumulative amount (b).

Acknowledgments

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