Collembolan Biodiversity in Mediterranean Urban Parks: Impact of History, Urbanization, Management and Soil Characteristics

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Collembolan biodiversity in Mediterranean urban parks: impact of history, urbanization, management and soil characteristics

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ABSTRACT

Urban parks provide esthetic and recreational services and improve the quality of life in cities. Sometimes considered as biodiversity hot-spots in cities, they are subjected to different management practices which may affect soil biological quality. This is the first study – performed in urban parks of Naples (southern Italy) – aiming to evaluate the effects of park history (age, previous land use of the area and soil origin), urbanization (sealed surface density of park neighborhood), current management (land cover type and litter presence/absence) or soil physico-chemical properties on collembolan communities diversity, as indicators of soil biological quality. Our results showed that the maintenance of specific land cover types and the presence of a litter layer were crucial factors in favoring high collembolan richness in urban parks, likely by ensuring adequate trophic resources and spatial niches. In addition, park age, urban density and previous land use of areas may be involved in shaping collembolan communities. Indeed, the most diverse and structured communities inhabit soils of the oldest urban park, with the lowest surrounding urban density and mild land use change.

1. Introduction

Urbanization is a major driver in the disappearance of green areas (Boyko and Cooper, 2011; Dallimer et al., 2011; Tratalos et al., 2007), disrupts ecosystem integrity and contributes to the degradation of environmental quality (Alberti and Marzluff, 2004). In this context, urban parks provide esthetic and recreational services, improve citizens' health and quality of life (Gómez-Baggethun et al., 2013) and maintain species biodiversity (Francis and Chadwick, 2012; McKinney, 2008). Even though urban parks are considered as biodiversity hot-spots within cities, they are usually still far from reaching this status (Brown, 2003). On the one hand, the conversion of natural areas into man-made ecosystems is accompanied by a decrease in biodiversity with slow recovery, if any (Santorufo et al., 2015). On the other hand, the transformation of urban areas into parks, and land use legacies, may affect ecosystem structure and functioning in the long term.

In order to match people needs and maintain biodiversity in cities, local stakeholders have to plan more green spaces and adopt more conservation programs and urban greening initiatives. Nonetheless, there is still little information available about the best management methods or vegetation features to maintain biodiversity in urban parks (Politi Bertoncini et al., 2012), especially in the case of soil fauna. For example, as highlighted by Norton (2011), the sanitization of urban parks provokes a decline of ground-dwelling arthropod communities, through the simplification of the vegetation structure and leaf litter and woody debris removal. In addition, urban parks may be constituted by very heterogeneous areas strongly affected by anthropogenic activities (Sarah et al., 2015). For example, picnicking or simply passing visitors cause changes in soil properties (i.e. soil compaction, moisture, organic matter content) and vegetation characteristics (i.e. vegetation height and species diversity) (Politi Bertoncini et al., 2012; Sarah and Zhevelev, 2007; Sarah et al., 2015). The soils of urban man-made green areas, falling into SUITMAs (Soils in Urban, Industrial, Traffic and Military areas; Morel et al., 2015), are often constituted by Technosols (IUSS, 2014), which can affect soil biodiversity. Indeed, these soils are often considered to be almost devoid of life (Leguédois et al., 2016), mainly because of their potential organic and inorganic contamination or their hostile physical properties (Eijsackers, 2010; Hafeez et al.,

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Table 1
Geographical coordinates, number of sampled plots, park age, urban density of parks, previous land use, park size, as well as number of sampled plots for soil origin and for each land cover type crossed with litter presence/absence of eight public urban parks in Naples (southern Italy).

Park	Total number of sampled plots	Age[‡]	Urban Density[§]	Previous Land Use	Park size (m^2)	Soil Origin		Land Cover[†]			
								open		closed	
								Litter			
						native	Technosols	absent	present	absent	present
Astroni **AST** 40° 50′ 39.6″N 14° 9′ 16.2″E	4	very old	very low	Forest	25,00,000	4	0	0	0	0	4
Capodimonte **CAP** 40° 52′ 25.9″N 14° 14′ 25.6″E	8	old	low	Forest	13,00,000	8	0	4	0	0	4
Virgiliano **VIR** 40° 47′ 55.3″N 14° 10′ 46.5″E	8	old	low	Forest	70,000	2	6	3	1	2	2
Nicolardi **NIC** 40° 52′ 30.0″N 14° 14′ 16.3″E	4	young	low	Tuff quarry	12,000	0	4	4	0	0	0
Poggio **POG** 40° 51′ 59.0″N 14° 14′ 24.8″E	8	young	medium	Tuff quarry	40,000	0	8	2	2	3	1
Scampia **SCA** 40° 52′ 24.2″N 14° 11′ 48.6″E	4	young	mediun	Abandoned area	1,40,000	4	0	4	0	0	0
Letizia **LET** 40° 50′ 20.0″ N 14° 19′ 21.4″E	4	young	high	Abandoned area	20,000	0	4	0	4	0	0
Troisi **TRO** 40° 50′ 0.7″N 14° 18′ 48.6″E	8	young	high	Agricultural area	1,20,000	6	2	2	2	0	4

[†] In each park, 4 or 8 plots were sampled, depending on the presence of only one - open or closed - or both land cover types. "Open vegetation" means habitats mainly composed by lawn or grassland, whereas "closed vegetation" implies woody and bushy habitats.
[‡] Park age was determined using the date when parks were first opened to the public, and classified as "young" (1–50 years), "old" (51–300 years), and "very old" (> 300 years).
[§] The percentage of urban density was classified as "very low" (< 20%), "low" (21–40%), "medium" (41–60%) and "high" (> 60%), since there was no occurrence in the 80–100% range.

2012; Huot et al., 2014; Lucisine et al., 2015). Nevertheless, it has been reported that Technosols can evolve into biologically active soils even when they are polluted (Frouz et al., 2013; Lucisine et al., 2015).

Even if soils host an extremely rich fauna (André et al., 1994), they are still largely unexplored (Coleman et al., 2004). Some soil invertebrates have been used as indicators of soil quality since the early 1960s (Cortet et al., 1999; Volz, 1962). For example, the use of Collembola is recommended in monitoring programs as they have been selected at the European level as indicators of soil biological quality (Bispo et al., 2010). Nevertheless, their use in biomonitoring of urban ecosystems still remains poorly developed (Fiera, 2009; Santorufo et al., 2015). Collembola are a proxy for soil biodiversity because: (1) they are abundant in many ecosystems and differ in microhabitat preferences and vegetation types (Gillet and Ponge, 2003; Heiniger et al., 2015; Vanbergen et al., 2007); (2) species exhibit different specialization levels, ranging from strictly specialist to generalist species (Auclerc et al., 2009); (3) many soil physico-chemical characteristics are crucial for species survival (Berg et al., 1998; Loranger et al., 2001; Ponge, 1993) and (4) collembolan local biodiversity is inversely related to landscape temporal heterogeneity (Ponge et al., 2003), especially if habitat patches are isolated (Heiniger et al., 2014).

In this study, we aimed to investigate whether collembolan communities diversity was driven by park history (age, previous land use of the area and soil origin), urban density (sealed surfaces in the neighborhood), local management (land cover type and presence of litter), or soil physico-chemical properties, using total metal concentrations as proxies for airborne pollutants (Pouyat et al., 2015). To this end, we compared, using community analysis and different biodiversity metrics, the structure of collembolan communities collected in several urban parks in Naples (southern Italy). We expected vegetation management to be the main factor modifying collembolan structure of communities by affecting niche parameters such as trophic resources and microhabitat quality.

2. Materials and methods

2.1. Study area and sampling design

The study was carried out in eight public urban parks in Naples (40°50′N; 14°15′E), the main metropolis of the Campania region (southern Italy), extending for about 119.02 km^2 (http://www.istat.it/storage/urbes2015/napoli) and representing the third largest metropolitan area in Italy after Rome and Milan. Moreover, Naples is nowadays also one of the most densely populated (8310.23 inhabitants km^{-2}) (http://www.istat.it/storage/urbes2015/napoli) European metropolises in the Mediterranean area, concentrating in its urban area more than half of the regional population (Scaramella, 2003). The area is characterized by a Mediterranean climate with warm-dry summers, and mild-wet winters (http://www.dst.unina.it/clima-di-napoli).

Information on the urban parks (i.e. age, previous land use of the area and soil origin) was obtained from municipality park offices (Table 1). The age of the parks was determined using the date when they were first opened to the public, and they were classified as "young" (1–50 years), "old" (51–300 years) and "very old" (> 300 years). Four classes of previous land use of the park area have been defined: forest, tuff quarry, agricultural or abandoned area. The urban context around each park was also characterized. To this end, the percentage of sealed surfaces (i.e. streets and buildings), in an area extending for 200 m from the perimeter of each park, was calculated with a Geographical Information System platform (QGIS Development Team, 2014), using vector maps of streets, buildings and public parks (http://sit.regione.campania.it/portal/portal/default/Download) and aerial Photo 2011 AGEA (Agenzia per le Erogazioni in Agricultura), at the scale of 1:10,000 (data retrieved June 4, 2014). The calculated percentage was then classified into four classes (Table 1): 1 –"very low" (< 20%), 2 – "low" (21–40%), 3 – "medium" (41–60%) 4 – "high"

Table 2
Mean values ± standard deviations of pH, bulk density (BD), water content (WC), total C and N concentrations, dissolved organic carbon (DOC), available P and total concentrations of Cu, Ni, Pb and Zn in the soils collected at the eight public urban parks in Naples (southern Italy).

Park[†]	AST	CAP	LET	NIC	POG	SCA	TRO	VIR
pH	6.023 ± 0.296	6.285 ± 0.814	7.075 ± 0.373	6.530 ± 0.238	6.736 ± 0.193	6.610 ± 0.136	6.769 ± 0.310	6.740 ± 0.258
BD (g cm^{-3})	0.342 ± 0.169	0.593 ± 0.115	0.749 ± 0.092	0.753 ± 0.055	0.644 ± 0.084	0.736 ± 0.195	0.710 ± 0.119	0.494 ± 0.106
WC (g g^{-1} d.w.)	1.032 ± 0.201	0.423 ± 0.165	0.234 ± 0.038	0.190 ± 0.057	0.207 ± 0.087	0.296 ± 0.102	0.340 ± 0.166	0.269 ± 0.073
Ctot (g Kg^{-1} d.w.)	153.359 ± 25.458	95.870 ± 45.698	40.680 ± 3.756	49.297 ± 15.79	44.735 ± 11.931	41.337 ± 7.710	57.819 ± 12.41	39.329 ± 14.259
Ntot (g Kg^{-1} d.w.)	11.303 ± 1.975	7.524 ± 3.001	3.539 ± 0.133	4.252 ± 1.333	3.966 ± 1.474	3.809 ± 0.790	4.840 ± 1.359	2.948 ± 1.158
DOC (mg Kg^{-1} d.w.)	2262.675 ± 604.094	1048.763 ± 373.975	424.410 ± 233.721	741.675 ± 330.928	593.274 ± 249.467	393.593 ± 150.918	1197.563 ± 389.325	725.044 ± 195.043
POlsen (g Kg^{-1} d.w.)	0.024 ± 0.012	0.032 ± 0.035	0.005 ± 0.002	0.019 ± 0.035	0.012 ± 0.014	0.023 ± 0.034	0.015 ± 0.029	0.012 ± 0.015
Cutot (mg Kg^{-1} d.w.)	27.511 ± 7.207	36.710 ± 21.535	57.791 ± 10.559	47.111 ± 55.051	26.343 ± 16.395	45.493 ± 19.457	134.022 ± 66.912	18.185 ± 4.554
Nitot (mg Kg^{-1} d.w.)	5.100 ± 1.915	5.350 ± 1.669	13.100 ± 1.915	4.600 ± 1.155	4.850 ± 2.605	7.600 ± 2.828	12.350 ± 4.132	3.350 ± 1.282
Pbtot (mg Kg^{-1} d.w.)	51.944 ± 13.280	107.945 ± 36.988	61.276 ± 12.517	38.753 ± 14.965	61.792 ± 50.335	56.343 ± 24.953	158.991 ± 51.706	38.453 ± 20.564
Zntot (mg g^{-1} d.w.)	92.473 ± 21.688	110.146 ± 46.060	110.781 ± 21.434	79.175 ± 2.774	101.243 ± 68.154	100.948 ± 51.550	187.631 ± 64.819	63.202 ± 8.700

[†] Park labels are reported in Table 1. Minimum and maximum values of pH, BD, WC, C, N, DOC and P are reported in bold. Total metal concentrations higher than the threshold values defined by Italian law (D.M. 471/99) are also reported in bold.

(> 60%), since there was no occurrence in the 80–100% range. In each park, four or eight plots (4 m^2 each) were identified, depending on the presence of only one – open or closed – or both land cover types. Each plot was situated in a homogeneous (open or closed) parcel of minimum 225 m^2. More specifically, habitats composed by lawn or grassland were classified in the "open vegetation" class, whereas woody and bushy habitats were placed into the "closed vegetation" class. Each plot was characterized for soil origin (native or Technosol) and for the presence or absence of a continuous litter layer on the soil when sampling was performed (Table 1). For closed vegetation plots, the continuous litter layer was mainly constituted by *Quercus ilex* leaves, whereas in open vegetation plots, the litter layer was mainly composed by thatch and turf clippings mixed to leaves from tree species in the neighborhood. From each plot, a core of surface soil (0–5 cm depth, diameter 5 cm) was collected after litter removal (if present), according to ISO 23611-2 (2006) as also reported by Cluzeau et al. (2012) and Ponge et al. (2013). For avoiding a potential "edge effect" on collembolan communities, each soil core was sampled at a distance of at least 2 m to parcel borders. To enable data comparisons, samples in closed vegetation plots were always collected under *Q. ilex* L., the dominant native evergreen species in these parks; similarly, samples for open vegetation plots were always collected under turf.

Soil sampling was carried out on April 20th and 21st 2015, as springtime is considered as the period of highest microarthropod species richness in Mediterranean areas (Cortet and Poinsot-Balaguer, 1998; Renaud et al., 2004; Sousa et al., 2004). During the week before sampling, mean temperature was 16 °C and relative humidity was 70% (http://archivio-meteo.distile.it) (retrieved October 13, 2016). Finally, 48 soil samples have been collected and used for the analysis of collembolan communities, as well as for physical and chemical analyses.

2.2. Collembola identification

Microarthropods were extracted from soil samples, through dry extraction for a week (Macfadyen, 1961). Temperature above the samples was increased during extraction from 35 °C (2 days) to 45 °C (2 days) and then to 60 °C (3 days), while temperature below the samples was kept at 5 °C. Animals were then preserved in 70% ethanol. For each sample, Collembola were counted and identified at morpho-species level under a binocular stereo-microscope (Kyowa optical, SDZ-TR-PL, 7–45× magnification). The specimens have been whitened in a Marc André I solution and mounted in a Marc André II solution (André, 1946). Species identification was verified under an optical phase-contrast microscope (Zeiss, Aaxio Scope A1, 200–630× magnification) according to various dichotomous keys (Bretfeld, 1999; Dunger and Schlitt, 2011; Hopkin, 2007; Jordana, 2012; Potapow, 2001; Thibaud et al., 2004; Zimdars and Dunger, 1994). When more than 10 specimens were counted for each morpho-species, a subsample of 10 specimens was randomly selected for species identification.

For each sample, collembolan density (number of Collembola per m^2) and richness (total number of species, S) were recorded and used to calculate the Shannon diversity index, *H*' (Shannon, 1948):

$$H' = -\sum P_i \log_2 P_i$$

where P_i is the ratio between the abundance of species i and the total number of organisms.

Moreover, evenness was evaluated according to the Pielou index, *J*' (Pielou, 1969):

$$J' = \frac{H'}{\log_2 S}$$

Collembola community structure for each urban park was evaluated through Whittaker's curves (Magurran, 2004). The most widely used models of community structure (i.e. brokenstick, pre-emption, log-Normal, Zipf and Zipf-Mandelbrot) were fitted to the data and the best

Table 3
Mean values ± standard deviations of the Collembola species density (individuals m^{-2}) in the eight public urban parks in Naples (southern Italy).

Species[†]		Frequency	AST		CAP		VIR		NIC		POG		SCA		LET		TRO	
Anurida sp.	1	6	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	3694	± 8668
Brachystomella parvula (Schäffer, 1896)	2	28	764	± 1529	1019	± 2327	0	± 0	2166	± 3040	1783	± 3008	1656	± 3312	0	± 0	446	± 921
Cyphoderus albinus (Nicolet, 1842)	3	2	127	± 255	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Ceratophysella denticulata (Bagnall, 1941)	4	2	4841	± 9681	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Ceratophysella gibbosa (Bagnall, 1941)	5	28	1401	± 1831	1656	± 4482	1747	± 4190	0	± 0	382	± 893	127	± 255	255	± 510	828	± 1963
Cryptopygus thermophilus (Axelson, 1900)	6	70	0	± 0	10,892	± 15,872	7570	± 9952	29,554	± 40,493	26,051	± 47,087	10,191	± 12,008	7261	± 9805	2484	± 3216
Desoria tigrina (Nicolet, 1842)	7	4	0	± 0	64	± 180	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	191	± 540
Entomobrya sp.	8	15	382	± 764	0	± 0	0	± 0	15,29	± 2162	191	± 540	0	± 0	127	± 255	255	± 545
Folsomia fimetaria (Linneaus, 1905)	9	2	764	± 1529	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Folsomia inocultata (Stach, 1946)	10	4	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	1656	± 1924	0	± 0
Folsomides parvulus (Stach, 1922)	11	9	255	± 510	64	± 180	0	± 0	0	± 0	573	± 1621	0	± 0	0	± 0	573	± 1621
Folsomia penicula (Bagnall, 1939)	12	6	3694	± 4311	0	± 0	0	± 0	0	± 0	127	± 360	0	± 0	0	± 0	0	± 0
Folsomia quadrioculata (Tullber, 1871)	13	6	764	± 976	0	± 0	0	± 0	127	± 255	0	± 0	0	± 0	0	± 0	0	± 0
Folsomia sexoculata (Tullber, 1871)	14	4	0	± 0	955	± 2702	0	± 0	0	± 0	191	± 540	0	± 0	0	± 0	0	± 0
Friesea truncata (Cassagnau, 1958)	15	2	0	± 0	764	± 2162	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Heteromurus major (Moniez, 1889)	16	2	0	± 0	0	± 0	73	± 193	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Isotomurus maculatus (Schäffer, 1896)	17	6	0	± 0	0	± 0	0	± 0	0	± 0	318	± 901	0	± 0	0	± 0	637	± 1606
Isotomiella minor (Schaffer, 1896)	18	17	255	± 510	5223	± 6996	437	± 1156	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Isotomodes templetoni (Bagnall, 1939)	19	30	15,032	17377	4076	± 8024	291	± 578	0	± 0	0	± 0	1911	± 2644	11,974	± 15,845	2420	± 6444
Isotomodes trisetosus (Axelson, 1907)	20	2	0	± 0	0	± 0	146	± 385	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Lepidocyrtus curvicollis (Bourlet, 1839)	21	2	2293	± 4586	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Lepidocyrtus lanuginosus (Linneaus in Gmelin, 1788)	22	4	637	± 1274	191	± 540	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Lepidocyrtus lignorum (Fabricius, 1775)	23	4	255	± 294	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Lathriopyga longiseta (Caroli, 1910)	24	4	382	± 488	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Mesaphorura sp.	25	6	0	± 0.00	892	± 2522	1820	± 3139	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Protaphorura armata (Tullber, 1871)	26	6	0	± 0	2420	± 4083	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Protaphorura aurantica (Ridley, 1880)	27	15	0	± 0	0	± 0	0	± 0	0	± 0	5732	± 7483	0	± 0	4076	± 8153	0	± 0
Paratullbergia caroli (Lucianez Ortega and Simón, 1991)	28	68	4076	± 5061	2293	± 2984	23439	± 35,590	15,796	± 15,439	4204	± 8521	1783	134 ± 8	1529	± 1951	9873	± 11,073
Pseudosinella immaculata (Lie-Pettersen, 1896)	29	6	1401	± 1577	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Proisotoma minuta (Tullber, 1871)	30	11	0	± 0	0	± 0	0	± 0	1019	± 2038	0	± 0	1656	± 3312	6624	10,002	0	± 0
Parisotoma notabilis (Schäffer, 1896)	31	13	6242	± 7698	1146	± 3243	510	± 1348	5478	± 10,955	0	± 0	0	± 0	0	± 0	2420	± 6846
Pseudosinella petterseni (Börner, 1901)	32	2	0	± 0	0	± 0	218	± 578	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0
Proisotoma ripicola (Linnaniemi, 1912)	33	2	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	1146	± 2293	0	± 0	0	± 0
Sminthurinus elegans (Fitch, 1862)	34	15	0	± 0	127	± 236	146	± 249	0	± 0	127	± 360	0	± 0	0	± 0	255	± 545
Sminthurus viridis (Linneaus, 1758)	35	9	0	± 0	0	± 0	73	± 193	0	± 0	127	± 360	255	± 510	127	± 255	0	± 0
Tomocerus minor (Lubbock,1862)	36	2	255	± 510	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0	0	± 0

[†] Park labels are reported in Table 1. Species are labeled by Arabic numerals. Species name information are available on http://www.collembola.org/taxa/collembo.htm - Last updated on 2017.03.31 by Frans Janssens.

Table 4
Mean values ± standard deviations of species richness (S), Shannon (H′) and Pielou (J′) diversity indices for each Collembola community in the eight public urban parks in Naples (southern Italy).

Park	S		H′		J′	
AST	7.750	± 1.500	1.421	± 0.165	0.702	± 0.081
CAP	4.250	± 1.488	1.021	± 0.389	0.723	± 0.158
VIR	3.429	± 0.787	0.651	± 0.256	0.579	± 0.278
NIC	4.000	± 0.816	0.812	± 0.174	0.617	± 0.233
POG	4.000	± 2.160	0.704	± 0.452	0.566	± 0.154
SCA	3.500	± 1.291	0.791	± 0.290	0.719	± 0.138
LET	4.750	± 1.500	1.113	± 0.178	0.740	± 0.144
TRO	3.750	± 1.669	0.741	± 0.402	0.632	± 0.180

model in each case was chosen according to Akaike and Bayesian Information Criteria (Oksanen, 2016) using the "vegan" package within the R 3.0.1 programming environment (R Core Team, 2013).

2.3. Soil physico-chemical characterization

Water content (WC, g^{-1} d.w.) was gravimetrically determined after soil drying at 105 °C until constant weight. To evaluate soil porosity, bulk density (BD, g cm^{-3}) was calculated as described in Baize (2000) on 100 cm^3 soil cores, taken from the top 5 cm of soil after litter removal (if present). Specifically, cores were dried at 105 °C for 24 h, and BD was calculated as the ratio between dry weight and core volume.

Soils sieved (at 2 mm) and oven-dried (at 75 °C) were characterized for:

pH (ISO 10390, 2005) with a soil:water ratio of 1:5 v:v;
total nitrogen (N, g kg^{-1} d.w.) concentration (ISO 13878, 1998);
total carbon (C, g kg^{-1} d.w.) concentration (ISO 10694, 1995);
dissolved organic carbon (DOC, mg kg^{-1} d.w.) (Thurman, 1985);
available phosphorous (P, g kg^{-1} d.w.) according to the Olsen method (ISO 11263, 1994);
metal concentrations (United States Environmental Protection Agency, 1996).

In particular, total copper (Cu, mg kg^{-1} d.w.), nickel (Ni, mg kg^{-1} d.w.), lead (Pb, mg kg^{-1} d.w.) and zinc (Zn, mg g^{-1} d.w.) concentrations were measured on an aliquot of 0.5 g pulverized and dried soil sample digested with 10 ml HNO_3 (65%, Sigma-Aldrich, Germany), 5.5 ml H_2O_2 (AnalaR Normapur®, France) and 5 ml HCl (37%, Carlo Erba, Italy) at 95 °C for 4 h. Total metal concentrations were measured using an ICP spectrometer (iCAP duo 6000 Series, ThermoScientific). Accuracy was checked through a concurrent analysis of standard reference material from the EU Community Bureau of Reference (BCR No. 142R: sandy loam soil) and metal recoveries ranged from 86 to 98%.

2.4. Statistical analyses

A multivariate analysis was performed to evaluate the effects of environmental parameters (C, N, DOC, P, BD, pH, WC, Cu, Ni, Pb, Zn, land cover type, soil origin, park age, previous land use, urban density, litter presence) on collembolan communities. In particular, non-metric multidimensional scaling (NMDS) was carried out on the collembolan data set, based on two axes, a Bray-Curtis distance metric and 9999 iterations. NMDS is commonly regarded as the most valuable unconstrained ordination method in community ecology (Minchin, 1987), while Bray-Curtis' algorithm is deemed adequate for the detection of underlying ecological gradients (Faith et al., 1987). To evaluate the effects of these gradients (C, N, DOC, P, BD, pH, WC, Cu, Ni, Pb, Zn), they were fitted, a posteriori, on the NMDS space with generalized additive models (GAMs). To achieve better results in the component selection within GAMs, the restricted maximum likelihood estimation (REML) method was used as the smoothing parameter estimation, which is in line for unknown scales and preferred when the primary purpose of the analysis is to carry out smooth component selection (Marra and Wood, 2011). To evaluate the effects of the categorical variables (land cover type, soil origin, park age, previous land use, urban density, litter presence), the associated confidence ellipses (for $\alpha = 0.05$) were superimposed on the NMDS biplot, and the significance of the differences between collembolan communities were tested using permutational multivariate analyses of variance (PERMANOVAs) (number of permutations = 999,999) (Anderson, 2001). PERMANOVA is a non-parametric test, extremely robust and applicable to any situation where the simultaneous responses of many dependent variables (i.e. usually abundances of species in an assemblage) have been measured in samples from a one-factor or multifactor ANOVA design (Anderson, 2001). The multivariate homogeneity of group variances for each PERMANOVA model was tested through the PERMDISP test (Anderson, 2006). Since the effect of the different environmental parameters was tested separately through GAMs in the NMDS space or through PERMANOVA, the collinearity among predictors did not affect results, but should be considered in their interpretation. Thus, a correlation matrix (i.e. the Spearman's, interclass' and Cramer's V coefficients of correlation) among the environmental parameters was illustrated in the Supplementary Material. All the analyses were performed with the R 3.0.1 programming environment (R Core Team, 2013) using functions from the "vegan" package.

3. Results

3.1. Park characteristics and soil physico-chemical properties

All parks falling into the "very old" (AST) and "old" (CAP and VIR) classes were built on previously forested areas and are located in "very low" and "low" urban density areas (Table 1). In particular, AST park was characterized by the lowest urban density (9.4%). Among these, AST and CAP had native soil, whereas in six out of eight sampling areas at VIR the soil was a Technosol (Table 1). "Young" parks differed either in previous land use or in the urban density of the surrounding area (Table 1). NIC park is 7 years old and was built with Technosols, as POG and LET parks, whereas in TRO and SCA parks, most or all of the sampling areas were on native soils (Table 1). Among these parks, TRO is located in a densely populated district, reaching the highest urban density value (64.0%).

Soil pH values ranged between 6.023 and 7.075, values measured at AST and LET, respectively (Table 2). The lowest soil water content (0.190 g g^{-1} d.w.) was found at NIC and the highest one (1.032 g g^{-1} d.w.) at AST (Table 2). The soils richest in C, N and DOC were found at AST park, whereas VIR (C and N) and SCA (DOC) parks had the soils with the lowest values of these parameters (Table 2). On average, the highest soil metal concentrations were observed in TRO and LET parks, located in high urban density areas (Table 2).

3.2. Collembolan community

Total collembolan density per sample ranged from 0 to 153,374 individuals per m^2 (detected at POG). A total of 33 taxa have been identified at the species level and 3 at the genus level (Table 3). The most frequently observed species were *Cryptopygus thermophilus* and *Paratullbergia caroli*, with frequencies of 70% and 68% of the total observations, respectively (Table 3). Both species were observed in all the parks (with the exception of *C. thermophilus*, absent from AST) with the highest density in NIC and VIR, respectively (Table 3). Many species have been observed at one site only: *Friesea truncata* at CAP, *Heteromurus major* at VIR, and *Ceratophysella denticulata*, *Cyphoderus albinus*, *Folsomia fimetaria*, *Lathriopyga longiseta*, *Lepidocyrtus lignorum*, *Lepidocyrtus curvicollis*, and *Tomocerus minor* at AST (Table 3).

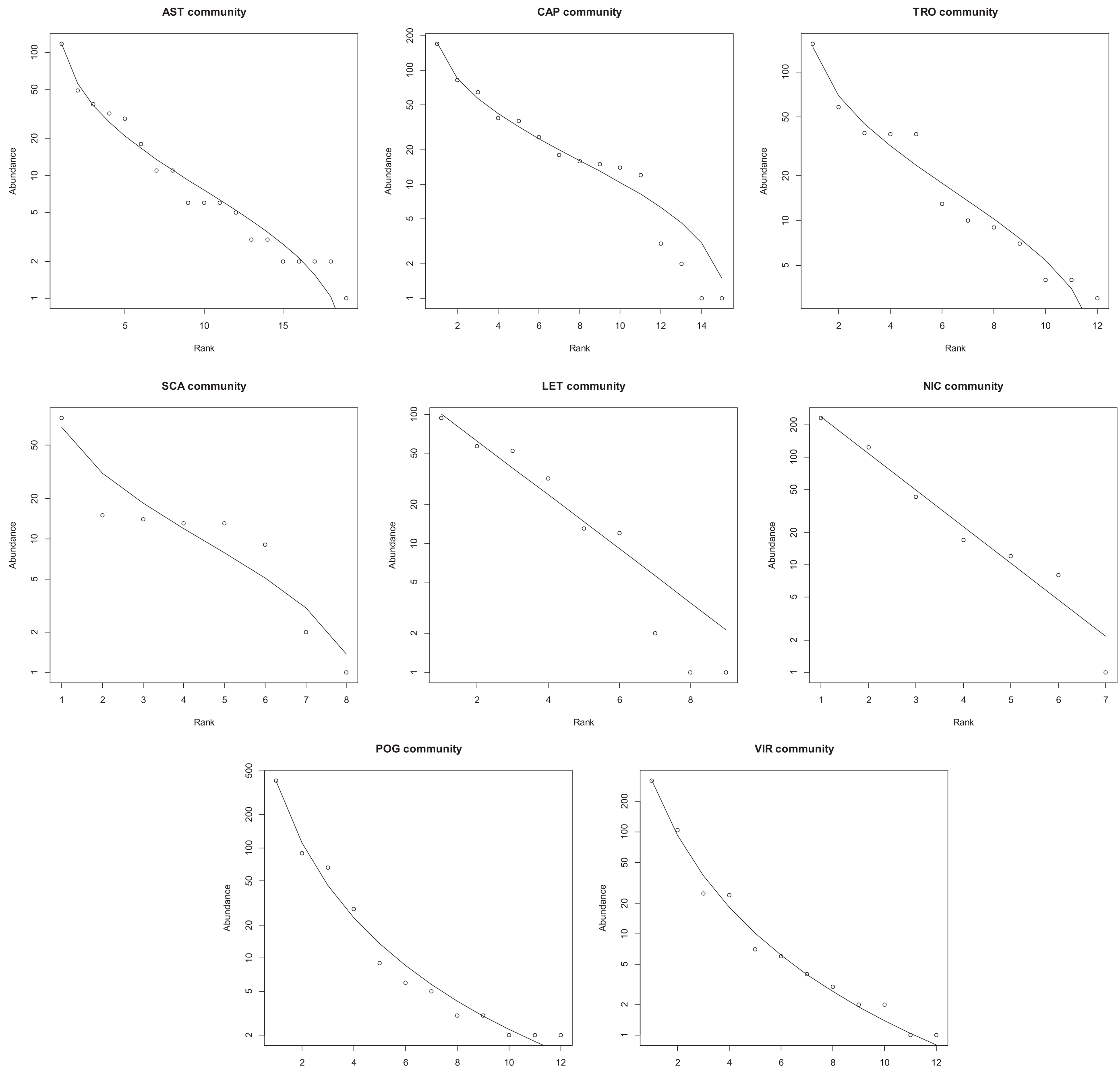

Fig. 1. Whittaker's curves concerning data for collembolans from urban parks in Naples (southern Italy). Species in the rank-abundance space are plotted as circles, whereas predicted abundances based on log-normal (AST, CAP, TRO and SCA), niche preemption (LET and NIC) and Zipf-Mandelbrot (POG and VIR) models are plotted as black lines. Park labels are reported in Table 1.

The least species-rich collembolan communities were observed in VIR soils (3.43), whereas the richest one was found at AST (7.75) (Table 4). The Shannon index per sample ranged between 0.65 (VIR) and 1.42 (AST), whereas the Pielou index ranged between 0.57 (POG) and 0.74 (LET) (Table 4).

Among the abundance-diversity models, the log-normal model was associated to the lowest AIC and BIC (Akaike and Bayesian Information Criteria) values in AST, CAP, SCA and TRO parks (Fig. 1). By contrast, the niche preemption model described better the community structures of LET and NIC, and the Zipf-Mandelbrot model those of POG and VIR parks (Fig. 1).

3.3. Relationships between collembolan communities and physico-chemical variables

Among the physico-chemical variables, GAMs based on soil C, N, DOC, WC and pH were significant ($p < 0.001$; REML scores: 225.75; 110.82; 348.41; −1.64; 29.52, respectively) in modeling the distribution of collembolan species in the NMDS space (Fig. 2). In particular, these GAMs explained 46% (C), 44% (WC), 33% (DOC), 30% (N) and 28% (pH) of the observed deviance. All gradients were parallel to the first NMDS axis, although their direction varied, with pH showing a direction of maximum variation opposite to all the others. Overall, a clear species gradient was recognizable, with *P. aurantiaca*, *P. petterseni*, *F. truncata*, *C. thermophilus*, *P. ripicola*, *I. trisetosus*, *B. parvula* and *S.*

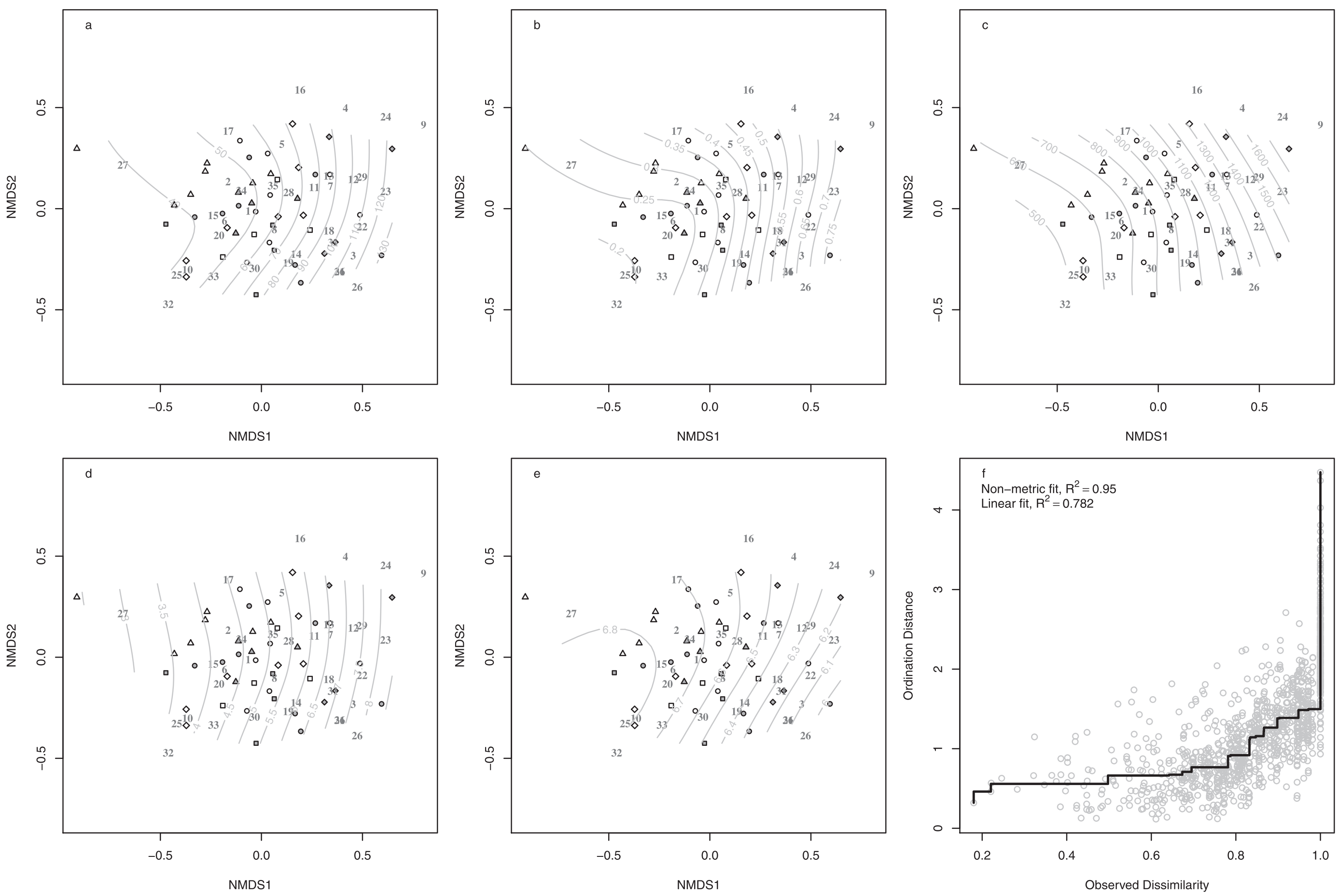

Fig. 2. Gradients, indicated by gray isolines, of soil (a) total C concentration, (b) water content, (c) dissolved organic C, (d) total N concentration and (e) pH on the 2d NMDS space concerning data for collembolans from urban parks in Naples (southern Italy). The stress plot with the linear and non-linear fits is also reported on panel (f). Different parks are coded with different symbols (AST: gray diamonds; CAP: gray circles; LET: gray squares; NIC: gray triangles; POG: white triangles; SCA: white squares; TRO: white circles; VIR: white diamonds). Park labels are reported in Table 1. Species are labeled by Arabic numerals defined in Table 3.

elegans all associated to soils with low C, DOC, N, WC, and with neutral pH values, and *P. immaculata*, *P. armata*, *C. albinus*, *L. curvicollis*, *L. lanuginosus*, *L. longiseta*, *L. lignorum* and *F. fimetaria* on the opposite end of the spectrum for these variables (Fig. 2).

3.4. Relationships between collembolan communities and categorical variables

Collembolan communities significantly differed ($p < 0.001$) between open and closed habitats, although confidence ellipses partially overlapped in the NMDS space (Fig. 3a). Confidence ellipses related to previous land use and the presence or absence of litter overlapped (Fig. 3c, e), but the groups were nevertheless clearly distinguished (both $p < 0.05$) based on PERMANOVA. In particular, parks built on previous tuff quarry sites had different collembolan communities in respect to the others. Finally, urban density and park age showed a similar distribution of confidence ellipses (Fig. 3), with those related to very low urban densities and to the very old park clearly separated from the others and associated to the same species, a differentiation also highlighted by PERMANOVA (both $p < 0.01$). All the categorical variables showed data not overdispersed, except for the litter presence/absence ($p < 0.01$).

4. Discussion

Our study on collembolan communities of Naples urban parks highlighted wide variations in relation to environmental characteristics. Land cover type and litter presence were the main drivers of collembolan community diversity (even if litter results may be linked to a minor overdispersion of data), probably because they largely contribute to soil DOC, C and N inputs and affect water content and pH. It is well known that litter fall in forest ecosystems is the main pathway of carbon transfer from plant to soil (Berg and McClaugherty, 2014; Campos et al., 2017; Koehler and Tranvik, 2015; Prescott, 2010), mediated by detritivore and decomposer activities. This is evident at AST, a Word Wild Fund for Nature reserve established in an ancient volcano, where detrimental management practices such as repeated litter removal and soil poaching are avoided. As a consequence, AST soils were covered by a thick litter layer, partially in an advanced stage of decomposition (personal observation), and exhibited the highest number of rare and forest-specific species. Among these were found: *C. denticulata*, a species commonly living in damp environments and abundant in habitats rich in organic matter (Hopkin, 2007), *F. fimetaria*, a nitrophilous species (Potapow, 2001), and *F. truncata*, *L. curvicollis*, *L. lignorum*, *L. longiseta*, *P. immaculata*, *T. minor* and *H. major*, which are known to live in feces, mosses, roots, under rocks, in dead wood and litter (Fjellberg, 2007; Hopkin, 2007). The maintenance of both canopy cover and litter layer, coupled with an evolution in low anthropogenic pressure conditions, seems thus to positively affect collembolan biodiversity in terms of community composition (Devigne et al., 2016). Conversely, the regular litter removal performed in other parks with closed vegetation or built on previously forested areas may be responsible for the observed substitution of forest-specific species with others, typical of more open habitats (Sattler et al., 2011). The increase in arthropod species richness together with the age of green urban areas was also reported by Sattler et al. (2010); this increase can likely be

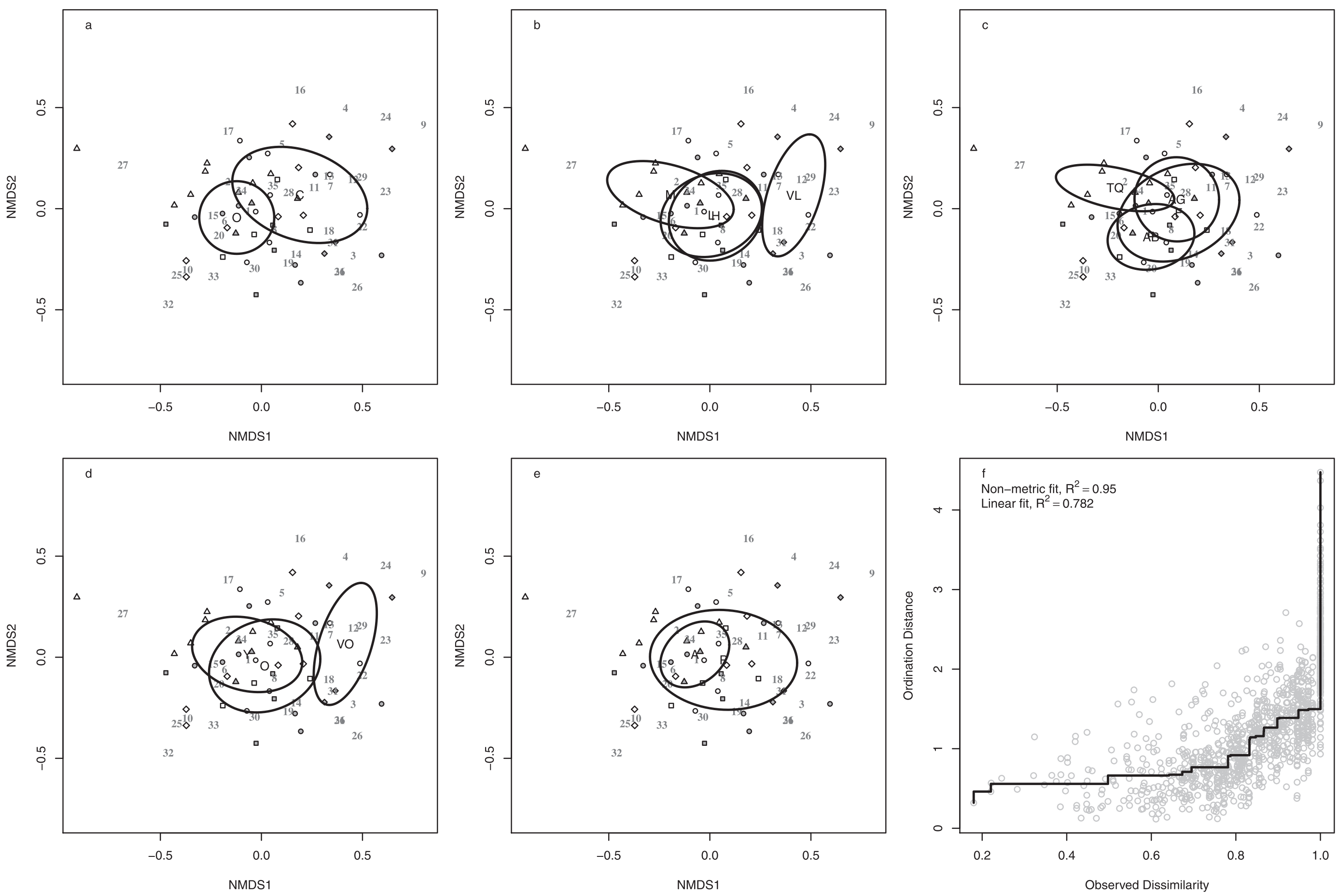

Fig. 3. Confidence ellipses (for α = 0.05), relative to (a) land cover (O: open, C: closed), (b) urban density (VL: very low, L: low, M: medium, H: high), (c) previous land use (AB: abandoned area, TF: tuff quarry, F: forest, AG: agricultural area), (d) park age (Y: young, O: old, VO: very old), (e) litter (P: present, A: absent), superimposed on the 2d NMDS space concerning data for collembolans from urban parks in Naples (southern Italy). The stress plot with the linear and non-linear fits is also reported on panel (f). Different parks are coded with different symbols (AST: gray diamonds; CAP: gray circles; LET: gray squares; NIC: gray triangles; POG: white triangles; SCA: white squares; TRO: white circles; VIR: white diamonds). Park labels are reported in Table 1. Species are labeled by Arabic numerals as defined in Table 3.

attributed to the combination of a progressive occupation of new species-specific ecological niches consequently to advancing succession, and a species accumulation over time caused by successful stochastic immigrations.

In our study, AST represents one extreme in the spectrum of the environmental variables measured, the opposite position being occupied by NIC, the youngest park, characterized by the lowest soil DOC, C and N concentrations, as well as by the highest soil compaction. Accordingly, NIC was characterized by species with low resource requirements, which are usually pioneer species, such as *P. minuta*, tolerant to unfavorable conditions (Potapow, 2001), or *C. thermophilus*, *P. caroli*, and *P. notabilis*, which are typically found in early and medium successional stages of decaying soil improvers (Potapow, 2001). Furthermore, many generalist species have been observed in NIC soils. For example, *C. thermophilus* is a nitrophilous species, typically eurytopic in southern Europe; it reaches high densities in various open and closed habitats, usually disturbed ones, such as sites near highways or industries and in metal-polluted soils (Potapow, 2001). *P. caroli* is a species known to live in either open or closed habitats, preferring dryer substrates in southern Europe. *P. notabilis* also generally reached high abundance, but this could be explained by the fact that this species is in fact a complex of different cryptic species, the populations of which may respond differently to their environment (Porco et al., 2012).

The biodiversity indices employed in the present study showed a low sensitivity in detecting differences among collembolan communities, as already pointed out (Devigne et al., 2016; Van Straalen, 1998). Conversely, the Whittaker's curves approach proved to be especially useful, allowing the analysis of community structure in terms of dominance/diversity (Whittaker, 1965) and its variations among urban parks. The log-normal distribution of species abundance in AST, CAP, SCA and TRO parks suggests complex communities with a dominance structure nearing equilibrium (Wilson, 1991). On the opposite, the communities of LET and NIC, two young parks, described by a niche preemption model of species abundance, appear to be in their early stages of evolution (Motomura, 1932, 1947; Whittaker, 1965). Indeed, the niche preemption model assumes that communities build up sequentially, with incoming species accessing a fraction of the unallocated resources, and has been suggested to be characteristic of resource-poor environments (Whittaker, 1965). Similar to the preemption model, the Zipf-Mandelbrot model, which best described communities in POG and VIR parks, assumes a sequential build-up of the ecological community, although the focus here is on environmental constraints rather than on available resources: early-arriving species are subjected to few constraints and will be more abundant, whereas those arrived later will be rare (Wilson, 1991).

In our study, the absence of any relationship between collembolan biodiversity and soil metal concentrations could be caused by low concentrations in respect to the threshold values defined by Italian law (D.M. 471/99) for urban green areas. Indeed, only CAP and TRO soils exceeded the threshold values (120, 120, 100 and 150 mg kg^{-1}, for Cu, Ni, Pb and Zn, respectively): CAP for Pb and TRO for Cu, Pb and Zn. However, it should be considered that these threshold values have been defined with a view to human health and are probably lower than values really impacting soil fauna. These high Pb and Zn concentrations were likely due to the high urban density and vehicular traffic along the TRO park borders (El Khalil et al., 2008; Wong et al., 2006), whereas the high Cu concentrations could result from the agricultural practices to which the area had been previously subjected (Belon et al., 2012; Saby et al., 2011). An interesting observation in this context was the exclusive presence in these two parks of *D. tigrina*, a nitrophilous and anthropophilous species (Grinbergs, 1958; Ponge, 1993; Potapow, 2001; Solnzeva, 1967), whose potential use as a bioindicator of metal pollution deserves further investigations.

5. Conclusions

To our knowledge, this is the first study aiming to evaluate the effects of park history, urbanization, management and soil properties on soil collembolan communities in Mediterranean urban parks. Our study clearly shows that maintaining a canopy cover and a litter layer are crucial factors favoring high collembolan biodiversity in urban parks, likely by ensuring adequate trophic resources and spatial niches. Conversely, low soil metal concentrations do not affect collembolan communities. Finally, an array of park characteristics, including park age, the urban context surrounding each park and the previous land use, may be in some way involved in shaping collembolan communities in terms of diversity and structure. The related functional characteristics require further investigations, but this study is one step toward the confirmation that Collembola can also be used as relevant bioindicators of soil biological quality in urban contexts.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.apsoil.2017.03.022.

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