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Urban springtail species richness decreases with increasing air pollution

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ABSTRACT

Environmental pollution impacts the structure, species richness and abundance of arthropod communities in urban environments. Detailed quantitative tests with microarthropod communities are nevertheless missing. We assessed species richness and abundances of Collembola within patches of urban green spaces in the city centre of Warsaw, Poland, to ask whether and how soil and air pollution influence springtail communities and whether these arthropods are suited as bioindicators. Springtails did not react significantly to varying environmental soil conditions, including heavy metal concentrations. Species richness was also independent of site area and isolation. However, springtails were highly sensitive to air pollution, particularly to PM_{10} and NO_2 concentrations. In sites having NO_2 air pollution above thresholds set by the European commission, species richness generally decreased. Our results indicate that springtails might not be a candidate taxon for urban soil bioindication. However, even moderate degrees of air pollution might have destructive effects on soil ecosystems.

1. Introduction

Springtails (Collembola) are one the most abundant groups of microarthropods in temperate soils (Greenslade, 1993; Hopkin, 1997). During their long evolutionary history, they have developed specific adaptations to soil and microclimatic conditions (Hopkin, 1997; Heiniger et al., 2015). In natural forest and grassland soils up to 50 species may occur in 1 m^2 area; total abundance often exceeds 10,000 m^{-2} and can be as high as 60,000 m^{-2} (Ford, 1935; Greenslade, 1993). This diversity is associated with a wide spectrum of life forms, and specific habitat adaptations (e.g. Rusek, 1998; Fiera, 2014; Potapov et al., 2016; Malcicka et al., 2017). This ecological diversity has led to the view that springtail communities are sensitive to changes in soil conditions (e.g. Maisto et al., 2017; Winck et al., 2017), disturbance (e.g. Sterzyńska et al., 2014; Turnbull and Lindo, 2015), and pollution (e.g. Austruy et al., 2016). Indeed, in agricultural landscapes and even more in urban areas springtail richness and abundance are much lower than in natural sites (Fiera, 2009; Rota et al., 2015; but see Fountain and Hopkin, 2004a). This decrease has been linked to higher concentrations of pollutants (Santorufo et al., 2012) and lower resource availability (Santorufo et al., 2015; Rzeszowski et al., 2017). Therefore, springtails are often considered as being potential bioindicators of changes in soil characteristics and environmental pollution (Greenslade, 2007; Fiera, 2009; Maisto et al., 2017).

Changes in springtail community composition have most often been linked to changes in land use (Chagnon et al., 2000; Ponge et al., 2003; Chauvat et al., 2007). However, several reviews (Greenslade, 2007; Fontanetti et al., 2011; Andrew et al., 2012) have largely emphasized the lack of knowledge with respect to springtail responses to environmental change. Further, many collembolans are known to be highly tolerant to agricultural and industrial pollution, although individual species differ widely in habitat demands and stress tolerance (Sterzyńska, 1990; Greenslade, 2007; Austruy et al., 2016; Rzeszowski and Sterzyńska, 2016). Therefore, the possible value of springtails as bioindicators might heavily depend on specific habitat condition and location.

Urban areas are characterized by a patchwork of habitat fragments modified by human activities and situated within a matrix of hostile surroundings (Werner, 2011). Varying conditions of atmospheric pollution, soil properties, vegetation, habitat isolation and area might led

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to unstable communities of unpredictable composition. It is generally believed that springtails are sensitive to soils of low pH, reduced nutrient availability, high salinity, and increased concentrations of organic and inorganic pollutants (Van Straalen, 1997; Rusek and Marshall, 2000; Fiera, 2009; Santorufo et al., 2015). However, work corroborating these predictions is surprisingly sparse and mostly confined to simple faunistic comparisons of 'natural' and 'urban' sites (e.g. Sterzyńska, 1990; Kuznetzova, 2009), or to laboratory microcosm experiments (e.g. Chauvat and Ponge, 2002). Fiera (2009) reported overall reduced abundance and species richness of springtails in a heavy metal-polluted urban site and identified species tolerant to soil pollution. Similarly, Kim and An (2014) found springtail behaviour to be affected by soil heavy metal contamination. On the other hand, Fountain and Hopkin (2004a,b) did not find any clear effect of long-term soil pollution on springtail diversity.

Significantly, most of these studies were built on simple comparisons of sites and did not account for additional variables that can influence species richness and abundance. Such variables include site area and isolation, as well as vegetation structure and soil type. According to the theory of island biogeography (MacArthur and Wilson, 1963) species richness increases with patch area and decreases with patch isolation. In urban areas patch isolation within the hostile matrix is of major importance (Fattorini et al., 2017). Isolated patches should be depauperate even if soil conditions and microclimate are favourable. Fertile soils covered by rich vegetation should harbour a rich springtail fauna (Rusek, 1998; Cassagne et al., 2003). Notably, concentrations of pollutants and physiologically important elements such as nitrogen, phosphorus, or iron might be positively correlated. In such cases springtail species richness and abundance might highest at intermediate degrees of soil pollution (Lin et al., 2010; Joimel et al., 2016). In this case positive effects of these elements on plant growth and phytophagous abundances might mask the deleterious effects of pollutants. Previous field tests on the ability of springtails to serve as bioindicators did not account for these covariates (Greenslade, 2007; Fiera, 2009; Santorufo et al., 2012; Maisto et al., 2017).

Here we used a dataset on the distribution and abundance of springtails in the city of Warsaw (Poland) accompanied by detailed environmental data and records on air and soil pollution. Based on the above discussion we asked: (1) whether springtail species richness and abundance responded to major air and soil pollutants; (2) whether urban habitat types and area determine species richness and abundance, and (3) whether we can identify species best suited as indicators of air and soil pollution.

2. Materials and methods

2.1. Study sites and sampling protocol

In 2013 we sampled 50 randomly chosen habitat patches (sites, Fig. 1) in the city center of the Polish Capital Warsaw (52°20′N, 21°00′E, ca. 40 m elev.). The sites consisted of 38 urban street lawns, six mixed urban forest remnants (dominated by *Pinus sylvestris* L.), and six park lawns (raw data are given in the electronic Supplement A). The urban lawns (street and parks) resembled pasture communities (*Cynosurion* alliance) or moist meadows (*Arrhenatherion* alliance).

Each site was geo-referenced using ArcView 9.3. Site area and the degree of site isolation (quantified by the proximity index of Gustafson and Parker, 1994) were assessed using Fragstat 4.2. The environmental data used for the present study are contained in the electronic Supplement A.

Each study site was sampled three times (July, September, October in 2013) during the growing season using three soil cores (5 cm in diameter × 10 cm deep). Collembola from the soil and litter layer were extracted in a modified MacFadyen high gradient apparatus and identified according to the most recent revisions. Identification of Collembola specimens was carried out with the aid of recent comprehensive monographs (Fjellberg, 1998, 2007; Pomorski, 1998; Bretfeld, 1999; Potapov, 2001; Thibaud et al., 2004; Dunger and Schlitt, 2011; Jordana, 2012). Nomenclature follows Bellinger et al. (2018). Material is deposited at the Museum and Institute of Zoology, PAS (Warsaw, Poland).

2.2. Abiotic factors

At each site we assessed moisture (soil volumetric water content, VWC), electrical conductivity (ECe), pH, salinity (total dissolved solids, TDS), as well as total carbon, nitrogen, potassium and phosphorus content (hereafter referred to major elements, quantified in g kg^{-1}). Soil pH was measured using a 1: 2.5 soil:water suspension using the potentiometric method; salinity of soil was measured as the total content of soluble ions ("Total Dissolved Solids", TDS) (see Tan, 2005); concentrations of total carbon (TC) and total nitrogen (TN) were quantified by the total combustion method using an automatic LECO CNS 2000 analyzer. Heavy metal concentrations (Cd, Cr, Cu, Fe, Ni, Mn, Pb, Zn were analyzed after digestion of the samples in a mixture of nitric, perchloric acids and hydrogen peroxide (US-EPA, 1996). The total phosphorus (TP) and potassium (TK) concentrations were analyzed after digestion of the samples in a mixture of nitric and perchloric acids (Ostrowska et al., 1991).

Heavy metal concentrations (hereafter heavy metals), total phosphorus (TP) and total potassium (TK), quantified in mg kg^{-1}, were analyzed by plasma–optical emission spectrometry. From these data we calculated respective C/N and C/P relationships. Average annual air pollutant concentrations (particulate matter concentrations PM_{10} in $\mu g m^{-3}$, SO_2, NO_2, both measured in ppm) for each site were predicted from the Calpuff modelling system provided by the Voivodeship Inspectorate for Environmental Protection in Warsaw for 2012 (see Sterzyńska and Rzeszowski, 2013).

2.3. Statistical analysis

We used ordinary least squares general linear models (GLM) to link species richness and abundance to environmental data (using Statistica 12). Species richness and abundance did not significantly deviate from a normal distribution (Shapiro-Wilk test: P(normal) > 0.40). We selected the most parsimonious model by Akaike information (AICc), using the model selection routine of SAM 4.0 (Rangel et al., 2010). Pearson correlation coefficients between predictor variables were always lower than 0.7, and therefore multicollinearity issues are unlikely in our analyses. To account for possible non-linearity and non-normal error structures, we used natural logarithms of habitat area. To account for the non-random spatial arrangement of the sites, the dominant eigenvector of the Euclidean spatial distances matrix entered the regression as covariate. This eigenvector explained 73% of variance and contained information about the spatial clustering of sites (Diniz-Filho et al., 2012). To separate groups of sites according to habitat characteristics, we used Euclidean distances based one-way Permanova as implemented in Primer 7 (Anderson, 2001).

In a species-specific approach we used ordinary least squares regressions to link the abundances of each species to soil characteristics, concentrations of heavy metals, and the degree of atmospheric pollution. As heavy metals have cumulative effects as pollutants, we calculated the dominant eigenvector (principal component) of the heavy metal variance –co-variance matrix and used this eigenvector and the respective eigenvector of the major elements as predictors in the GLM analysis. These eigenvectors explained 91% and 51% of variance, respectively. We designated species as being significantly positively or adversely related to pollution – and consequently being a candidate species for bioindication – if their significance levels were $P < 0.001$.