



Collembola diversity in the critically endangered Cape Flats Sand Fynbos and adjacent pine plantations

W.P.A. Liu, C. Janion*, S.L. Chown

Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Private Bag X1, Matieland 7602, South Africa

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ABSTRACT

The replacement of indigenous vegetation by commercial alien plantations can strongly affect invertebrate species richness and composition. This has been demonstrated for some invertebrate taxa in the Fynbos vegetation of the Western Cape, but the response of litter-dwelling Collembola has not been documented. Here we compared the richness and abundance of litter-dwelling Collembola assemblages in *Pinus radiata* plantations and in adjacent rehabilitated Cape Flats Sand Fynbos (CFSF) in the Tokai Forest Reserve, Western Cape, South Africa. A total of 48 Collembola morphospecies was identified, of which 14 species were recorded exclusively from the pine plantations, and six exclusively from the CFSF. Abundance and morphospecies richness was higher in the pine plantations than in the CFSF, with 34 morphospecies found in the CFSF, and 42 morphospecies in the pine plantations. Fourteen species were likely invasive taxa, of which eight were found in CFSF, and 13 in the pine plantations. Multivariate analyses indicated significant differences in the assemblages of the two habitat types based on abundance, but using presence–absence data only, no difference was found. Significant relationships were found between springtail richness and abundance and site humidity, which differed between the CFSF (drier) and pine plantation (moister), likely accounting for the differences in richness and abundance among habitat types.

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Introduction

Located at the south-western tip of Africa, the Fynbos Biome is widely known for its high floral diversity and endemism (Goldblatt 1997; Goldblatt and Manning 2002). Fynbos, the predominant vegetation complex of the biome (Rebelo et al. 2006), is threatened by a variety of direct and indirect anthropogenic factors, including urbanization, agriculture, biological invasions by woody species, nutrient loading, and climate change (Richardson et al. 1996; Rouget et al. 2003; Hannah et al. 2005; Rebelo et al. 2006; Gaertner et al. 2009; Wilson et al. 2009). From the landscape change perspective, alien pine plantations and invasions by pines into natural areas not only have a substantial effect on plant species richness and composition, but they also affect other ecosystem characteristics and functioning profoundly, with the exception perhaps of soil pH and N (Richardson et al. 1992, 1996; Richardson and Cowling 1992; Stock and Allsopp 1992; Richardson and Higgins 1998; Scholes and Nowicki 1998; van Wilgen 2009). The impacts on plant community structure and functioning also translate into significant effects on invertebrate assemblages. By comparison with adjacent Fynbos

vegetation, areas dominated by pines have lower invertebrate richness and abundance, at least in the case of the few groups that have been assessed (Donnelly and Giliomee 1985; Pryke and Samways 2009).

Within the Fynbos vegetation complex many of the vegetation types are considered threatened. In particular, Cape Flats Sand Fynbos (CFSF) is listed as critically endangered mostly owing to its localized occurrence (Cape Town Area only), and the substantial transformation of more than 80% of its original area by urbanization and afforestation (Holmes 2008). The few, small, remnant fragments of CFSF have been accorded considerable conservation priority, and large areas of pine plantations are being removed as part of a long-term rehabilitation programme (e.g. c. 150 ha in the Tokai Forest Reserve) (Rebelo et al. 2006). Despite the conservation significance of the CFSF little is known about its invertebrate fauna by comparison with other vegetation types in the Fynbos biome (e.g. Picker and Samways 1996; French and Major 2001; Botes et al. 2006, 2007; Procheş and Cowling 2006; Pryke and Samways 2009, 2010). Moreover, investigations of the effects of landscape transformation on CFSF seem limited to impacts on vascular plants and on the avifauna (Rebelo et al. 2006; Dures and Cumming 2010). More generally, restricted knowledge of invertebrate responses to landscape transformation is not uncommon for the region. Owing partly to the taxonomic impediment (Samways 2007), and partly

* Corresponding author. Tel.: +27 218083396.
E-mail address: cjanion@sun.ac.za (C. Janion).

to the significance of insects as pollinators and seed-dispersers in Fynbos (Johnson 1992), investigations tend to be focussed on pollinating insects, ants, and dragonflies (e.g. French and Major 2001; Donaldson et al. 2002; Braschler et al. 2010; Samways and Sharratt 2010), and only occasionally consider other groups (Witt and Samways 2004; Pryke and Samways 2009). Indeed, the less conspicuous soil mesofauna, such as springtails and mites, is largely neglected, despite its widely acknowledged significance in ecosystem functioning (Brussaard et al. 1997; Bengtsson 1998; Behan-Pelletier and Newton 1999; Wardle et al. 2004).

Such a situation applies particularly to the springtails. Although they are widely distributed in virtually all terrestrial habitats (Hopkin 1997), make a significant contribution to soil system functioning (Rusek 1998), and are likely to show significant diversity and endemism in South Africa (Yosii 1959; Coates 1968a,b; Janion et al. 2011a) with concomitant implications for ecosystem functioning (Bengtsson et al. 2011, 2012), detailed ecological studies are lacking, and the diversity of the group in both disturbed and undisturbed Fynbos areas has not been investigated. Therefore, we provide here an assessment of springtail diversity in a Cape Flats Sand Fynbos system and the adjacent areas dominated by managed *Pinus radiata* plantations.

Material and methods

Study site

The study was conducted in Tokai Forest Reserve (TFR) (S34°3.416' E18°25.659'), part of Table Mountain National Park in the Western Cape Province of South Africa. This particular site was chosen largely because, despite being under rehabilitation, it is one of the most extensive remnant patches of CFSF. Most of the other patches are tiny remnants surrounded by urban areas, 'mismanaged' in a variety of ways (Rebello et al. 2006: 140), and subject to substantial edge effects. The area has a Mediterranean-type climate, with mean daily minimum and maximum monthly temperatures of 7.3 °C and 27.1 °C for July and February, respectively, and a predominantly winter rainfall regime (c. 576 mm year⁻¹, with a peak from May to August, Rebello et al. 2006). Located on acidic and gray regic sands, the predominant indigenous vegetation type of the TFR is Cape Flats Sand Fynbos, which is characterized by four main plant families, viz. Proteaceae, Restionaceae, Asteraceae (in drier areas) and Ericaceae (in wetter areas), and dominated by dense, relatively tall shrubs (Rebello et al. 2006). A total of 108 vascular plant species is known from the CFSF, of which one is extinct and three currently exist in cultivation and are being reintroduced into remnant patches (Rebello et al. 2011). As a consequence of the history of commercial afforestation of the area, remnant patches of CFSF are interspersed among 460 hectares of managed vegetation, predominantly planted *Pinus radiata*, and small patches of cleared areas that are being restored to Sand Fynbos. The reserve has been under the management of South African National Parks (SANParks) since 2005.

Sampling and species identification

Sampling was conducted in mid-winter between 27th July and 26th August 2010. Mid-winter is typically a period of high precipitation and a period of high Collembola activity (Bengtsson et al. 2011). Sampling sites were selected within the lower section of the reserve to minimize elevation differences among sites. A total of eight independent sampling sites was selected: four pine plantation patches (pine habitat) which were planted between 1973 and 1997, and four CFSF patches (CFSF habitat) which have been rehabilitated after pine removal between 1998 and 2007 (Table 1;

Table 1

The year of establishment at each sampling site.

Sampling site	Year of establishment
Pine 1	1996
Pine 2	1973
Pine 3	1995
Pine 4	1997
Cape Flat Sand Fynbos 1	1998
Cape Flat Sand Fynbos 2	2004
Cape Flat Sand Fynbos 3	2007
Cape Flat Sand Fynbos 4	2005

SANParks data). This age variation was unavoidable, as the CFSF sites are critically endangered and are being restored. However, for one of the CFSF sites the age was within a few years of the youngest pine plantations, and we considered age effects much less of a potential confounding factor than the edge effects in smaller, but perhaps older, patches of CFSF elsewhere (see Introduction), as has been found for other taxa (Rebello et al. 2006, 2011; Dures and Cumming 2010). The largest and smallest distances between sites were 1714 m and 171 m, respectively. Sampling took place approximately once every week and sampling was completed for two to three sites per sampling event. Within each sampling site, two parallel transects were laid out approximately 10 m apart and five litter samples were collected at 10 m intervals along both transects, resulting in 10 samples per site. The sampling for both habitat types was undertaken at least 50 m into the vegetation to eliminate edge effects. Thus, a total of 80 samples was collected (i.e. eight sites with 10 samples in each). Collembola were sampled by taking one liter of surface organic litter (approximately 5 cm in depth). A 1 m² grid was placed over each sampling point, and the sample was collected within the grid to fill the one liter container. All of the organic litter samples within CFSF were collected under *Metalasia muricata*, which was one of predominant indigenous shrubs within the Sand Fynbos patches. Within the pine plantation sites, organic litter samples were collected directly under *P. radiata*.

Ambient temperature and relative humidity were taken in the field approximately 3 cm above the soil surface at the shaded side of each sampling point by using a handheld Vaisala HM32 thermohygrometer (Vaisala, Finland). This resulted in 80 measures of temperature and humidity (40 for each habitat type). In addition, two soil samples were collected haphazardly at two sampling points in each site and were mixed together to form a single sample of 450 g, resulting in a sample size of four for each habitat type. These soil samples were analyzed for soil type, pH, P (Bray and Kurtz 1945), C, N, Mg and Ca by a commercial service provider (BemLab (Pty Ltd.), Somerset West, South Africa).

The litter samples collected for invertebrate extractions were kept cool in an insulated container and transported to the laboratory for extraction within 6 h of collection, to minimize death of the sampled Collembola and the risk of them being consumed by predators such as spiders (Rusek 1998; Hopkin 2007). Collembola were extracted from the litter using Berlese–Tullgren funnel extraction (see Macfadyen 1953). Each sample was extracted individually and the Collembola stored in 99.9% ethanol. Individuals were assigned to morphospecies using external morphological characteristics or to species level, where possible, using European keys (Fjellberg 1998, 2007; Bretfield 1999; Potapov 2001; Hopkin 2007; Bellinger et al. 2011) and information currently being collected by a large project within the Centre of Excellence for Invasion Biology in collaboration with specialists in France, Norway and Sweden (Janion et al. 2011a,b; Potapov et al. 2011). The use of morphospecies enables rapid evaluation of the invertebrate assemblages in question by avoiding the need for complete species identifications. Although the approach has several problems, one of us (CJ) has considerable experience with the Fynbos fauna (e.g. Janion et al.

2011a,b; Potapov et al. 2011), and we have confidence that the large majority (>95%) of the species delimitations refer to single species, even where species-level names for these taxa are not yet available. Similarly, invasive species status was assigned based on whether taxa at the given level of identification are considered typical of Africa or of the Southern Hemisphere generally, or are considered are of European origin using keys available (Fjellberg 1998, 2007; Potapov 2001; Hopkin 2007). All sorting was done by one researcher (WPAL) using a Leica MZ 7.5 microscope, and the species delimitations assessed by a second (CJ). Therefore, the data are comparable between the eight sites. The numbers of individuals for each morphospecies in each site were counted.

Statistical analysis

Sampled-based rarefaction curves for Collembola were assembled for CFSF and pine plantation sites to establish the level of sampling efficiency (EstimateS V8.2, Colwell 2009, <http://viceroy.eeb.uconn.edu/estimates>; Gotelli and Colwell 2001). These curves assess the number of expected species from the known number of observed individuals in the samples (Gotelli and Colwell 2001). The non-parametric Incidence Coverage Estimator (ICE), Michaelis–Menten, Chao 2 and first-order Jackknife richness estimators were selected to estimate the sample size adequacy (Palmer 1990; Colwell and Coddington 1994; Chazdon et al. 1998; Brose 2002). These are known as useful estimators because they are most stable even with small sample sizes (Magurran 2004). Sampling may be considered as sufficient when the sample-based rarefaction curves and the four non-parametric estimators converge closely at the highest observed morphospecies richness (Magurran 2004).

Differences in site morphospecies richness and abundance among sites were examined using a generalized linear model assuming a Poisson distribution with a log-link function, corrected for overdispersion where necessary (implemented in Statistica V. 10.0, StatSoft, Tulsa, Oklahoma). The extent to which CFSF and pine-dominated sites differ in assemblage structure was investigated using routines available in PRIMER V 5.0 2001 (Plymouth Routine in Multivariate Ecological Research, see Clarke and Warwick 2001). The data were double square-root transformed prior to analysis to weight common and rare morphospecies equally. The Bray–Curtis similarity measure was used to construct a similarity matrix among all sites (abundances of morphospecies were summed across the ten samples per site). Cluster analysis was then undertaken using group averaging, and a One-way Analysis of Similarity (ANOSIM) was conducted to determine whether the assemblages differ in structure (Clarke and Warwick 2001). The level of difference among habitat types increases as the significant Global R statistic approaches one. Non-metric multi-dimensional scaling (nMDS) (using 500 random restarts) was used to display the relationships among sampling sites. The analyses were repeated after transformation of the data to presence–absence only to determine the extent to which differences in the assemblages among habitat types could be explained by differences in species richness and identity rather than by variation in abundance among species. For further investigation of the numbers of rare species in the two habitat types, common and rare species were defined as the 25% most abundant and 25% least abundant species, respectively, within the assemblages of each habitat type (the proportion of species method – Gaston 1994).

Soil parameters (pH, C, N, P, Ca and Mg) were compared between the CFSF and pine-dominated habitats using Mann–Whitney *U*-tests, while temperature and relative humidity were compared using a one-way analysis of variance, all implemented in Statistica V.10.0. The relationships between temperature and relative humidity, and species richness and abundance across sites were examined

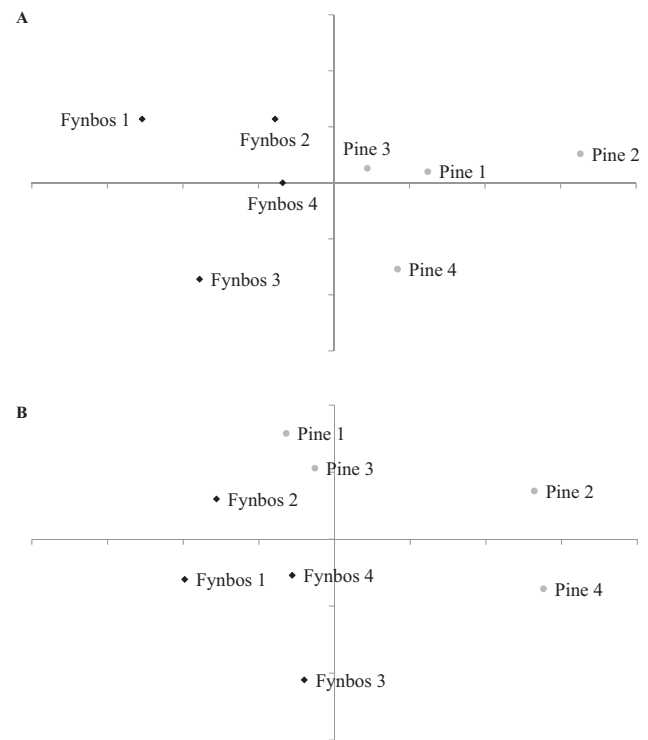


Fig. 1. Non-metric MDS ordination of the sites in each of the sampled habitats based on (A) abundance data (transformed to weight common and rare species equally); among habitat ANOSIM $R=0.531$, $P<0.05$. Stress=0.06. (B) Based on presence–absence data only; among habitat ANOSIM $R=0.385$, $P>0.05$. Stress = 0.03.

using Pearson's product-moment correlation coefficients (Statistica V.10.0).

Results

A total of 21,728 individuals from 48 Collembola morphospecies was collected from the pine plantation and Cape Flats Sand Fynbos sites in Tokai Forest Reserve (Table 2), of which 13,996 individuals and 42 morphospecies were found in the pine plantations, and 7732 individuals from 34 morphospecies in the CFSF. The sample-based species rarefaction curves and the non-parametric estimators converged closely at the highest observed morphospecies richness for both habitats (Supplementary Figs. S1 and S2), and typically well before the full sample total of 40 for each habitat type, indicating that the sampling for morphospecies richness estimates was sufficient. Of the 48 morphospecies collected, 12 were considered rare (i.e. in the lower abundance quartile), and of these species two were shared among the pine and CFSF habitats (*Dicyrtomina cf. saundersi* and *Seira* sp. 3) (Table 2). Six morphospecies, of which four were rare, were unique to the CFSF, while 14 morphospecies, of which six were rare, were unique to the pine plantation. Mean morphospecies richness differed significantly among sites (Pine: 13.5 ± 0.56 , CFSF: 11.3 ± 0.51 , $\chi^2 = 7.6$, $P=0.006$) as did mean abundance (Pine: 350 ± 48 , CFSF: 193 ± 45 , $\chi^2 = 8.5$, $P=0.004$).

An ANOSIM based on the abundance data revealed that the Collembola assemblages of the CFSF and pine plantation habitats differ significantly (Global $R=0.531$, $P=0.029$) (Fig. 1a), although with some variation within these groups. Pine plantation site 1 had the highest morphospecies richness and abundance of all the sampling sites (Fig. 2a and b), while CFSF 3 had the lowest morphospecies richness (Fig. 2a) and CFSF 1 had the lowest morphospecies abundance (Fig. 2b). Thirteen presumed invasive species were identified, with eight found in the CFSF, and 12 found in the pine plantations (Table 2). Suspected invasive species were found only

Table 2
Abundance of Collembola morphospecies from Cape Flat Sand Fynbos (CFSF) and *Pinus radiata* plantations. The abundance values of the 25% least abundance morphospecies in the samples are indicated in bold.

Order	Family	Species	CFSF	Pine	Invasive/endemic		
Poduromorpha	Hypogastruridae	<i>Ceratophysella cf. denticulata</i>	90	780	Invasive		
		<i>Brachystomella</i> sp.	752	801	?		
	Neanuridae	<i>Setanodosa</i> sp.	37	4	Endemic		
		<i>Neanura cf. muscorum</i>	3	23	Invasive		
		<i>Paleonura</i> sp.	0	3	Endemic		
		<i>Micranurida</i> sp.	13	163	?		
		<i>Aethiopella cf. flavoantennata</i>	0	1	Endemic		
	Onychiuridae	<i>Orthonychiurus</i> sp.	0	21	Invasive		
	Tullbergiidae	<i>Mesaphorura cf. macrochaeta</i>	249	69	Invasive		
		<i>Tullbergia</i> sp.	1082	838	?		
		<i>Entomobrya cf. multifasciata</i>	279	803	Invasive		
Entomobryomorpha	Entomobryidae	<i>Entomobrya</i> sp. 1	247	465	?		
		<i>Entomobrya</i> sp. 2	1	0	?		
		<i>Entomobrya</i> sp. 3	2	0	?		
		<i>Entomobrya</i> sp. 4	0	23	?		
		<i>Lepidocyrtus</i> sp. 1	331	1486	?		
		<i>Lepidocyrtus</i> sp. 2	0	78	?		
		<i>Pseudosinella</i> sp.	35	23	?		
		<i>Seira cf. barnardi</i>	0	12	Endemic		
		<i>Seira</i> sp. 1	129	253	Endemic		
		<i>Seira</i> sp. 2	3	11	Endemic		
		<i>Seira</i> sp. 3	2	2	Endemic		
		<i>Seira</i> sp. 4	5	19	Endemic		
		<i>Seira</i> sp. 5	4	28	Endemic		
		Isotomidae	<i>Subisotoma</i> sp.	0	6	?	
	<i>Cryptopygus cf. caecus</i>		237	496	?		
	<i>Cryptopygus</i> sp. 1		516	15	Endemic		
	<i>Cryptopygus</i> sp. 2		714	105	Endemic		
	<i>Isotomurus cf. maculatus</i>		0	332	Invasive		
	<i>Isotomurus cf. palustris</i>		0	192	Invasive		
	<i>Parisotoma</i> sp. 1		1803	721	Endemic		
	<i>Parisotoma</i> sp. 2		900	4683	Endemic		
	Tomoceridae		<i>Tomocerus cf. minor</i>	0	3	Invasive	
			Neelidae	<i>Megalothorax</i> sp.	2	164	Invasive
	<i>Neelus</i> sp.			0	13	Invasive	
	Symphypleona			Mackenziellidae	<i>Mackenziella cf. psocoides</i>	22	0
			<i>Arrhopalites</i> sp.		0	5	?
		Bourletiellidae	<i>Bourletiella</i> sp.	34	0	?	
Dicyrtomidae		<i>Dicyrtomina cf. ornata</i>	30	22	Invasive		
		<i>Dicyrtomina cf. saundersi</i>	4	3	Invasive		
		<i>Dicyrtomina</i> sp.	3	0	?		
		Katiannidae	<i>Sminthurinus cf. elegans</i>	0	906	Invasive	
Sminthurididae		<i>Sminthurinus</i> sp. 1	59	38	?		
		<i>Sminthurinus</i> sp. 2	30	34	?		
		<i>Sminthurinus</i> sp. 3	0	21	?		
	<i>Sminthurides</i> sp. 1	7	0	?			
	<i>Sminthurides</i> sp. 2	37	30	?			
	<i>Sphaeridia</i> sp.	70	301	?			
Total		7732	13,996				

in Pine 2, including *Isotomurus cf. maculatus*, *Isotomurus cf. palustris*, *Orthonychiurus* sp. and *Sminthurinus cf. elegans*. The only invasive species that was only found in CFSF was *Mackenziella cf. psocoides*. The high morphospecies diversity found in CFSF 2 was comparable to that of some of the pine sites (Fig. 2). When the ANOSIM was repeated using presence–absence data only, no significant difference was found among the assemblages of the two habitat types (Global $R=0.385$, $P=0.057$, Fig. 1b). Thus, the differences among the habitat types had more to do with the relative abundances of the species present than a clear distinction between assemblage membership in the two habitats.

No significant differences were found in pH, C, N, P, Ca or Mg among the habitat types (Mann–Whitney $U=4.0$ – 8.0 , $P>0.3$ in all cases). By contrast, the habitat types differed significantly in temperature (ANOVA $F_{(1, 78)}=23.4$, $P=0.0001$; Pines $17.2 \pm 0.26^\circ\text{C}$; CFSF $19.0 \pm 0.26^\circ\text{C}$, mean \pm S.E.) and relative humidity (ANOVA $F_{(1,78)}=35.3$, $P=0.00001$; Pines $68.5 \pm 1.1\%$; CFSF $59.3 \pm 1.1\%$). Moreover, the two variables were strongly, negatively related across the sites ($r=-0.79$, $P<0.001$). Significant relationships were also found between morphospecies richness and both temperature

($r=-0.24$, $P=0.03$) and humidity ($r=0.33$, $P=0.003$), and abundance and both temperature ($r=-0.22$, $P=0.049$) and humidity ($r=0.26$, $P=0.02$, Fig. 3).

Discussion

Despite the conservation significance of Cape Flats Sand Fynbos (CFSF), few assessments of the diversity of groups other than the vascular plants have been made for this vegetation type, and studies of impacts of land transformation are restricted to vascular plants and birds (Rebelo et al. 2006, 2011; Dures and Cumming 2010). Our assessment found a total richness of 48 springtail morphospecies across both the CFSF and pine plantation, with 34 species in the former and 42 in the latter. Owing to the taxonomic complexity of some groups (e.g. *Seira*, Tullbergiidae), this may be an underestimate, although sampling had clearly gone to completion, and we consider our species-level assignments accurate. A richness of 48 species for a total area of 2 km² is in keeping with that found in other areas (see Petersen and Luxton 1982).

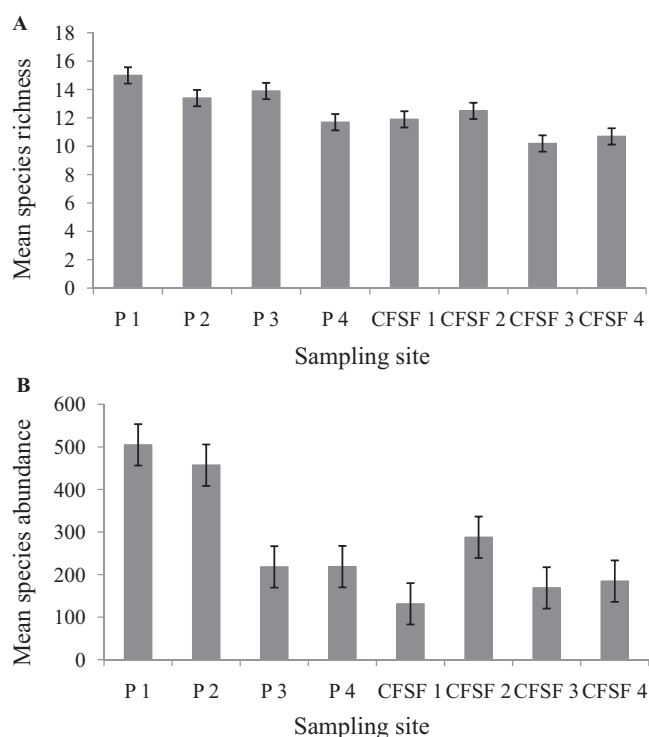


Fig. 2. (A) Mean species richness and (B) abundance per site. The error-bars indicate standard error. P = pine sites, CFSF = Fynbos sites.

The mean site morphospecies richness of 11–14 (CFSF and pine plantation, respectively) is slightly higher than that found for the combined epigaic fauna (i.e. millipedes, scorpions, harvestmen, beetles, ants) examined in the same general area by Pryke and Samways (2010), whilst abundance was very much higher (193–350 individuals in CFSF and pine plantation, respectively) compared with site abundances of less than 100. Springtail site mean richness and abundance is also well within the range found for other groups of epigaic invertebrates investigated in the Fynbos, including spiders, ants, and beetles, and toward the upper end of the values found for the latter taxa (French and Major 2001; Ratsirarson et al. 2002; Witt and Samways 2004; Botes et al. 2006,

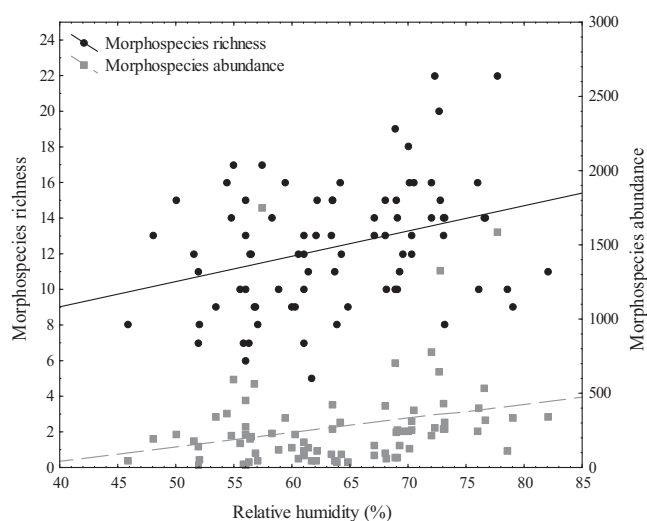


Fig. 3. Variation in morphospecies richness ($r=0.33$; $P<0.05$) and abundance ($r=0.26$; $P<0.05$) were significantly and positively related to relative humidity (%) across the eight sampling sites.

2007). Thus, it is clear that the springtails form an important component of epigaic invertebrate diversity both in the CFSF and pine plantations, as has been recorded elsewhere for this group (Hopkin 1997; Rusek 1998). They are likely also to have much functional significance in these habitats, given that springtails have been shown to have an important role in nutrient cycling elsewhere in the Renosterveld and Fynbos vegetation complexes (Bengtsson et al. 2011, 2012).

By contrast, the extent to which the springtail fauna is restricted to either the location generally, or to the CFSF is not completely clear, mostly because comprehensive investigations of sites elsewhere in the Fynbos biome have yet to be completed (see discussion in Janion et al. 2011a). At least on the local scale, in this study, it is clear that CFSF has six unique morphospecies (*Sminthurides* sp. 1., *Bourletiella* sp., *Dicyrtomina* sp., *Entomobrya* sp. 2, *Entomobrya* sp. 3, *Mackenziella* cf. *psocoides*) (Table 2), of which one is thought to be invasive. However, the significance of this finding is tempered by the fact that the pine plantation included 14 unique morphospecies, of which five are thought to be invasive. Indeed, the invasive species in the pine plantation were typically much more abundant than the invasive species in the CFSF (Table 2).

These results suggested that substantial differences in the assemblages of the two habitat types should be found. Although such a difference was not apparent based only on species identity, it was indeed the case in terms of diversity (i.e. abundance variation among species), and supports other studies showing marked diversity differences among habitat types in the Fynbos, and in the Table Mountain area specifically (French and Major 2001; Ratsirarson et al. 2002; Witt and Samways 2004; Pryke and Samways 2009, 2010). Moreover, the pine plantation springtail assemblage had, on average, both a significantly higher abundance and morphospecies richness than the assemblage in the CFSF. Clearly these differences are not attributable to variation in soil nutrient status because no among-habitat type differences were found in the variables measured (pH, C, N, P, Ca, Mg). Rather, it seems likely that the temperature and relative humidity differences among sites were responsible for the among-habitat type differences in abundance, morphospecies richness and assemblage structure. Springtails typically show substantial sensitivity to desiccation (Choi et al. 2002; Kærsgaard et al. 2004; Chown et al. 2007), and high humidity, low temperature sites would clearly benefit them because both act to reduce the differences in water activity between the organism and its surrounding environment (Wharton 1985; Harrison et al. 1991; Hopkin 1997; Chown et al. 2011). High humidity in the pine plantation is likely associated with the substantial needle litter layer, as is typical of other pine forest sites (Scholes and Nowicki 1998). The favorable conditions in the pine plantation likely also meant a more favorable environment for the suspected invasive species, which reached considerable densities in the case of *Sminthurinus* cf. *elegans*, *Entomobrya* cf. *multifasciata* and *Ceratophysella* sp. (Table 2). Thus, it appears that the non-indigenous vegetation promotes the success of other alien species, an interaction which elsewhere has been known to develop into a substantial synergy known as invasional meltdown (Simberloff 2006). Although this might not turn out to be the case here, the changes in springtail assemblages found, including the increase in the abundance and richness of invasive species, suggest that the invasion of natural Fynbos areas, by pines, as is occurring elsewhere in the region (van Wilgen 2009) may promote invasion by springtail species. Whether this is taking place deserves investigation given the demonstration both in the Fynbos Biome and elsewhere that springtails may substantially affect soil system functioning (Brussaard et al. 1997; Bengtsson 1998; Wardle et al. 2004; Bengtsson et al. 2011, 2012).

In conclusion, the current study has shown that the CFSF houses a diverse assemblage of springtails and that pine plantations are even richer, though they also contain greater richness and

abundance of invasive species. As the particular area (Tokai Forest Reserve) is rehabilitated through the removal of pine plantations, springtail abundance is likely to decline given changing litter levels and abiotic conditions. However, it seems likely that among the springtails the invasive species will be lost, so returning the system perhaps to a state more similar to that of untransformed Fynbos. However, if the rehabilitated areas were invaded by a different plant species which has substantial impacts on both edaphic variables and on humidity and temperature this might not take place. One species that is capable of such transformation is *Acacia saligna* (Yelenik et al. 2004; Le Maitre et al. 2011), and early indications suggest that it might indeed be increasing in abundance in the area (M.A. McGeoch, personal communication). Future work should determine the extent to which this landscape transforming species alters springtail assemblages.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.pedobi.2012.03.002.

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