Spatial patterns of species richness and nestedness in ant assemblages

along an elevational gradient in a Mediterranean mountain range

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Abstract

The study of biodiversity spatial patterns along ecological gradients can serve to

elucidate factors shaping biological community structure and predict ecosystem

responses to global change. Ant assemblages are particularly interesting as study cases,

because ant species play a key role in many ecosystem processes and have frequently

been identified as useful bioindicators. Here we analyzed the response of ant species

richness and assemblage composition to elevational gradients in Mediterranean

grasslands and subsequently tested whether these responses were stable spatially and

temporally. We sampled ant assemblages in two years (2014, 2015) in two mountain

ranges (Guadarrama, Serrota) in Central Spain, along an elevational gradient ranging

from 685 to 2390 m a.s.l.

Jackknife estimates of ant species richness ranged from three to 18.5 species and

exhibited a hump-shaped relationship with elevation that peaked at mid range values

(1100 – 1400 m). This pattern was transferable temporally and spatially. Elevation was

significantly related to ant assemblage composition and facilitated separation of higher

elevation assemblages (> 1700 m) from the remaining lower elevation species groups.

Ant assemblages were nested; therefore species assemblages with a decreased number

of species were a subset of the richer assemblages, although species turnover was more

important than pure nestedness in all surveys. The degree of nestedness changed non-

linearly as a cubic polynomial with elevation. These assembly patterns were observed

over time but not between the two study regions.

We concluded double environmental stressors typical of Mediterranean mountains

explained species richness patterns: drought at low elevations and cold temperatures at

high elevations likely constrained richness at both extremes of elevational gradients.

The fact that species turnover showed a dominant role over pure nestedness suggested

current ant assemblages were context-dependent (spatio-temporal factors) and highly

vulnerable to global change, which threatens the conservation of present day native ant

communities, particularly at high elevations.

Introduction

Predicting the response of biodiversity to the main drivers of Global Change has

become a primary goal of modern ecology [1-3]. Consequently, the analysis of species

richness and assembly patterns along latitudinal and altitudinal gradients provide clues

to project possible effects of climate change on communities [4, 5]. The Mediterranean

mountains are particularly suitable for this purpose, given a characteristic combination

of temperature and water availability gradients in space and time, which determines a

progressively cooler and wetter environment at higher elevations while valleys suffer

more severe summer drought conditions [6, 7]. In addition, the Mediterranean Basin

will probable face particularly marked increases in aridity, temperature and frequency of

extreme climatic events [8-10]; therefore it is critical to advance our knowledge of how

basin communities respond to more severe abiotic factors.

Ants are diverse and ubiquitous, exhibit strong interspecific interactions, and perform

many functional processes in ecosystems [11]; therefore ants are considered a viable

indicator group of ecosystem change by many community ecologists. Ants are also

dominant terrestrial community members [12] and play an important role in plant

community dynamics, acting as seed harvesters [13], dispersal agents [14], and

influencing soil nutrient status and plant growth [15]. As predators, ants control

herbivore abundance, drive relevant top-down effects, and are often used as biological

control agents of insect pests and fungal pathogens [16]. Furthermore, ant distribution

and abundance patterns provide information to address specific environmental

management issues [17, 18]; consequently, ants are particularly useful bioindicators.

Previous research on ant diversity documented variable responses to altitudinal

gradients, which showed a certain dependency on macroclimate. For instance, studies

reporting monotonic decreases in species richness, were frequent in temperate

mountains [19-21], while monotonic increases were observed in arid environments [22,

23]. Longino and Colwell [24] and Nowrouzi *et al*. [25] indicated more complex

patterns in tropical climates, which often combined a monotonic decrease at high

elevations with a plateau at low elevations. Mid-elevation peaks were reported in

tropical areas [26, 27], although also frequent at other latitudes [28, 29]. These patterns

were partially attributed to geometric constraints, such as the mid-domain effect and

area availability [30, 31] or the Rapoport rescue effect [32]. However the role of

climatic factors, including temperature [19-21] and water availability [22, 29, 33, 34] as

direct drivers of ant diversity is also unequivocal. Moreover, several studies

demonstrated these climatic variables also indirectly modulated ant diversity, for

example, by affecting primary productivity [33, 35-37].

Understanding how ant diversity responds to altitudinal gradients is not sufficient,

however, to predict how environmental changes (e.g. rising temperatures) will affect

species geographic ranges, because knowledge on species composition patterns is also

required. Species assemblages and their distributions along ecological and geographical

gradients result from species tolerances to climatic (abiotic) factors and biotic

interactions, which interplay with niche evolution to determine community composition

[38-40]. Kodric-Brown and Brown [41] defined a structured community as one where

the organization is not due to randomness [41], and all situations in which communities

are not random can be explained by spatial turnover, nestedness, or combinations of

both [42]. Nestedness of species assemblages occurs when site biotas with smaller

species numbers are subsets of the biotas at richer sites [43, 44]. If environmental and

habitat filtering and not interspecific competition is responsible for nestedness, most

species can coexist. Therefore, under increasing temperatures, species could migrate or

expand their distribution elsewhere from the species former range, without necessarily

implying other resident species will be displaced by the newcomers. On the contrary,

spatial turnover implies the replacement of some species by others, exhibiting gains and

losses of species from location to location globally [45]. This is a sign that some species

cannot coexist, because of different climatic requirements, competitive exclusion, or

both. In general, global warming will extend insect species ranges to higher altitudes

[46], therefore with spatial turnover patterns, species will migrate up an altitudinal

gradient and those at higher altitudes might suffer a subsequent reduction in geographic

distribution range, and even disappear from the area. However, Maguire *et al.* [47]

acknowledged both processes and community patterns might vary with scale, and

exhibit spatio-temporal variation, which raises concerns on our capacity to generalize

the results based on local studies. Indeed, transferability of results is increasingly

considered in analyzing the effects of global change on species distributions, although

such efforts are rarer in studies of community patterns [48-50].

In the present study, we explored patterns of ant species diversity and community

composition in Mediterranean mountain grasslands, integrating spatio-temporal

transferability. Specifically, we addressed the following questions: (1) what is the

relationship between ant species richness and elevation; (2) how does elevation affect

ant species composition; (3) what are the relative contributions of spatial turnover and

nestedness in ant community composition; and (4) are our findings transferable spatially

and temporally?

Methods

<u>Study areas and sampling design</u>

We studied grassland ant communities in two mountains in the larger Sistema Central

range, Sierra de Guadarrama (reaching 2428 m a.s.l. at its highest peak) and Serrota

(maximum 2294 m a.s.l.). The two mountains are ~ 100 km apart with similar climatic,

geologic, and biotic characteristics. Mean annual temperatures range from ~ 14 ºC at

600 m to ~ 4 ºC at the summits (2300 – 2428 m); and mean annual rainfall ranges from

550 mm to 1500 mm, with severe summer drought [51]. Substrata are primarily

composed of granites, and pasturelands are distributed along the complete elevational

gradient, largely as the result of traditional livestock grazing.

Sample sites were interspersed along the elevational gradient, facing southeast to

southwest with gentle slopes (< 5%), and not affected by anthropogenic disturbance

(i.e., not close to buildings or main paths). We sampled 18 grasslands in 2014 (in

Guadarrama range) and 12 in 2015 (six in Guadarrama range and six in Serrota range).

On each grassland, we randomly placed a 4 x 3 grid of pitfall traps, 2.5 cm diameter x 5

cm deep, with 5 m spacing between traps, which were filled with a 3:1 ethanol/

monoethylene glycol mixture. Ant sampling was conducted in July, during stable

anticyclonic meteorological periods (Guadarrama: 17 - 23 July 2014 and 9 - 16 July

2015; Serrota: 1 - 8 July 2015), taking advantage of the seasonal ant activity peak

during summer in Central Spain [52]. Traps were collected after one week and ant

specimens were sorted to species, excluding winged ones.

Pitfall trapping is considered more objective and unbiased than other methods for

sampling ground ants [53, 11]. We chose pitfall traps previously tested in other studies

in Central Spain, which demonstrated success at capturing the complete species pool

and the entire ant size range variability present in the sampled area [52, 54, 55].

<u>Statistical analyses</u>

We examined the relationship between species richness, taxonomic composition, and

nestedness of grassland ant communities with elevation. Our analytical strategy was to

build descriptive models for the data gathered during the most intensive sampling in

Guadarrama 2014, and to validate the patterns detected with data gathered in a subset of

the Guadarrama grasslands during 2015 (temporal validation) and with data gathered in

the Serrota range during 2015 (spatial validation).

Species richness

First, we assessed the relationship between species richness and elevation. We estimated

the species pool size at each locality with the incidence-based first order jackknife

estimator. The relationship shape between estimates of richness and elevation was

explored applying a generalized additive model (GAM) with Gaussian errors and a thin

plate regression spline for elevation. Models were built with data from Guadarrama

2014 sampling and validated with data from the Guadarrama 2015 and Serrota 2015

samplings. Squared Pearson correlation between jackknife estimates of richness and

jackknife estimates predicted by the model were used to assess species richness pattern

constancy between years and sites.

Taxonomic composition of assemblages

We explored how species assemblages differed based on taxonomic composition by

applying non-metric multidimensional scaling (NMDS) to the hemi-matrices of binary

Bray-Curtis distances among grasslands derived from species occurrences in pitfall

traps. In addition, a model-based analysis was conducted by building separate negative

binomial regressions for each species abundance data (as the multivariate response

variable) by elevation, year, and interaction (as the explanatory variables) [56].

Significance of the interaction (assessed by resampling) would suggest that patterns of

species composition along an elevational gradient changed between years. Likewise, for

Guadarrama 2014 and Serrota 2015, a multivariate model was built by fitting negative

binomial regressions for individual species abundance, elevation, location, and

interaction [56]. Significance of the interaction would suggest how assemblages

changed with elevation were context-dependent.

Nestedness patterns

To assess assemblage nestedness among grasslands surveyed on each mountain and

year we calculated the matrix temperature (T) [57] and the NODF (Nestedness metric

based on Overlap and Decreasing Fill) [58]. While T has traditionally been the most

commonly used metric for assessing overall nestedness [59], the more recent NODF

index is claimed to exhibit more robust statistical properties, and separate contributions

to nestedness of columns (due to species incidence) and rows (due to site composition)

can be quantified [58, 60]. T decreases and NODF increases with nestedness.

The significance of nestedness indices was estimated by comparison with suitable binary null models, which were simulated by randomizing the original matrix transformed to presence/absence binary data. Two null models were built to encompass the range from maximally liberal (equitable) to maximally conservative (constrained) null models [61]. The first was an equiprobable model built by randomizing rows and columns and therefore maintaining only the number of species presences ('r00' algorithm). The second was a proportional resampling model constructed by constraining randomization to maintain row and column totals (site richness and species incidence, respectively), while using marginal column frequency to select species ('quasiswap' algorithm). Thus, the first null model was more liberal and only accounted for matrix fill (the incidence of the total species set), while the second null model was more stringent, because it imposed additional structure on the data, accounting for among-site differences (e.g. different carrying capacities) and among-species differences (e.g. different rarities). Subsequently, significant nestedness was attributed to variation in observed species richness or species incidence, if evaluated with the equiprobable model, or to variation beyond that observed, if evaluated with the proportional resampling model [62]. One thousand randomizations of the original matrix were applied.

The relative degree of nestedness, i.e. how much a grassland was nested within the set of sampled grasslands in the same geographic area and time period, was evaluated by a nestedness rank, according to T. Poorer grasslands, i.e. those having lower species diversity which were a subset of richer grasslands, were assigned higher ranks. These ranks were estimated as the ordinate in nestedness plots built with the T index, which

were calculated as (k - 0.5)/n for k = 1…, n rows (e.g. the bottom row in the graph,

where k = 18 was the more nested site, which for the 2014 sampling in Guadarrama had

a rank of (18 - 0.5)/18 = 0.97).

The shape of the relationship between nestedness and elevation during the 2014

sampling in Guadarrama (n = 18) was explored using GAM with Gaussian errors and a

thin plate regression spline for elevation. This model was validated using Guadarrama

2015 data (temporal validation). Detection of a high correlation between observed (in

2015) and predicted ranks would suggest the nestedness rank of plots did not vary

between years. Similarly, the model built with Guadarrama 2014 sampling data was

validated using the Serrota 2015 data (spatial validation). Detection of a high correlation

between observed and predicted ranks would suggest the nestedness rank of plots did

not vary between regions.

The contributions of spatial turnover and nestedness to the distribution pattern in

Guadarrama 2014 data were calculated using three beta diversity indices: Sørensen-

based multiple-site dissimilarity (β_{SOR}), Simpson-based multiple-site dissimilarity

(β_{SIM}), and nestedness-resultant multiple-site dissimilarity (β_{NES}). β_{SIM} accounts for

spatial turnover and β_{NES} integrates dissimilarity due to nestedness, while β_{SOR}

expresses the total dissimilarity between communities and equals the sum of β_{SIM} and

β_{NES} [42]. *P*-values were estimated through the equiprobable ('r00') null model (the

proportional resampling null model was not useful to assess this partitioning of beta

diversity).

Statistical analyses were performed using R (v 3.3.2; R Core Team 2016) and

specialized packages vegan (v. 2.4-5) [63], mvabund (v. 3.12-3) [56], and betapart

(1.5.0) [64].

Results

We detected 37 species, all recorded in Guadarrama (35 in 2014 and 26 in 2015, 24

were shared between years) and 20 species in Serrota (all species shared with

Guadarrama). A total of 15 species were common among the three surveys (**S1 Table**).

Jackknife estimates of ant species richness in Guadarrama 2014 ranged from three (SE

= 0) species in most of the highest grasslands to 18.5 (SE = 2.6) at intermediate

elevations (**Table 1**). Estimates of richness related to elevation were positive below

1100 m a.s.l. and negative until a lower limit was reached at ca. 2000 m a.s.l. (**Fig 1**).

Overall, richness exhibited a hump-shaped relationship with elevation, which we

described using GAM with a spline for elevation, roughly equivalent to a third-order

degree polynomial (equivalent df = 3.96, F = 12.94, $P < 0.0001$, $D^2 = 83.8\%$, **Fig 1**).

This pattern of species richness was similar between years and between mountain

ranges (correlations for estimates of richness and richness predicted by the model were r

= 0.92, t = 4.68, df = 4, $P = 0.0095$, $R^2 = 85\%$ for Guadarrama 2015 data; and r = 0.86, t

= 3.42, df = 4, $P = 0.0267$, $R^2 = 75\%$ for Serrota 2015 data).

Table 1. **Observed (S) and first-order jackknife estimator (J ± SE) of ant species**

richness in dry grasslands in central Spain.

ID	Sampling	S	J±SE	N	X	Y	Elevation (m)
G1	G2014	7	7±0	13	437876	4494480	685
G1	G2015	7	7.9±0.9	10	437876	4494480	685

G2	G2014	8	8±0	12	434821	4496549	768
S1	S2015	14	18.5±2.4	10	351711	4474792	808
G3	G2014	14	16.7±1.6	11	440394	4503515	865
G4	G2014	14	16.8±1.6	12	435541	4506094	1012
G5	G2014	14	15.8±1.3	12	431692	4513777	1044
G5	G2015	11	11.9±0.9	12	431692	4513777	1044
S2	S2015	11	14.7±2.3	12	357653	4481333	1086
G6	G2014	13	14.8±1.3	9	414145	4507982	1171
G7	G2014	13	18.5±2.6	12	438808	4523233	1285
G8	G2014	15	15.9±0.9	11	416330	4511347	1330
G8	G2015	14	17.5±2.2	8	416330	4511347	1330
S3	S2015	13	15.8±1.6	12	340207	4476196	1374
G9	G2014	9	9±0	12	436806	4524537	1487
G10	G2014	10	12.8±1.6	12	400821	4495703	1625
S4	S2015	4	4±0	11	326465	4482790	1646
G11	G2014	3	3±0	19	400543	4496999	1675
G12	G2014	5	5.9±0.9	12	419833	4519838	1786
G12	G2015	5	5.9±0.9	12	419833	4519838	1786

G13	G2014	5	5±0	12	419076	4519012	1830
G14	G2014	4	4.9±0.9	12	418737	4519177	1920
S5	S2015	2	2±0	12	323719	4482788	2000
G15	G2014	3	3±0	12	419392	4520829	2026
G15	G2015	3	3±0	8	419392	4520829	2026
G16	G2014	3	3.9±0.9	12	417883	4521249	2139
G17	G2014	4	5.8±1.9	12	418676	4521128	2266
S6	S2015	3	3.9±0.9	12	323944	4485467	2286
G18	G2014	3	3±0	12	419420	4522544	2390
G18	G2015	3	3.9±0.9	12	419420	4522544	2390

Grassland plots were ordered following elevation (in m a.s.l.), identified with an ID,

where G stands for Guadarrama, S stands for Serrota, and numbers represent grassland

position in increasing order of elevation. Sampling: data from Guadarrama range in

2014 (G2014) and 2015 (G2015) and Serrota range in 2015 (S2015). N: samples sizes

(number of pitfall traps recovered from each locality). Geographic coordinates (X and

Y) were provided as UTM European Datum 1950.

Fig 1. **Relationship between first-order jackknife estimates of ant species richness**

(see Table 1) and elevation in dry grasslands of a Mediterranean range in central

Spain.

Curve and 95% IC from GAM with Gaussian errors fit to the Guadarrama 2014 survey

(black circles). Richness estimates for the surveys in Guadarrama 2015 (grey circles)

and Serrota 2015 (open circles) were superimposed. Elevation values were slightly

jittered to avoid overlap among some points.

A NMDS solution was built that satisfactorily summarized the Bray-Curtis distances

among grasslands in both years and study regions (stress = 0.10, linear R^2 between

Bray-Curtis and NMDS distances = 0.95). This scaling separated grasslands roughly

according to elevation, but not regions or years and suggested an elevation threshold of

1700 m a.s.l. was a suitable elevation to distinguish the two groups (**Fig 2**). Multivariate

GLMs for species abundances revealed significant elevation effects ($P = 0.001$) and a

significant region effect ($P = 0.047$), but significant effects were not detected for year or

any elevation interactions (**Table 2**). These results suggested taxonomic composition

varied mainly with elevation, but differences between regions cannot be excluded.

Fig 2. Non-metric multidimensional scaling (NMDS) of ant assemblages based on

binary Bray-Curtis distances among grasslands calculated with species

occurrences from pitfall traps in a Mediterranean range of central Spain.

Circles represent data from Guadarrama (black: year 2014, grey: year 2015) and Serrota

(open circles) ranges. Symbol sizes are proportional to the elevation of sampling plots.

As reference, circles are superimposed to a smooth elevation surface (estimated using a

GAM of elevation on a bivariate spline of NMDS scores). The positions of some

illustrative species are given as reference.

Table 2. **Multivariate GLMs (negative binomial regressions) fitted to species abundances.**

	Deviance	P values
a)		
Elevation	192.91	**0.001**
Year	50.27	0.649
Elevation x Year	5.99	0.997
b)		
Elevation	178.14	**0.001**
Region	84.44	**0.047**
Elevation x Region	18.85	0.618

P-values estimated by bootstrap (n = 1000). Significant *P*-values ($P < 0.05$) are highlighted in bold. a) Guadarrama 2014 vs. Guadarrama 2015. b) Guadarrama 2014 vs. Serrota 2015.

Ant species assemblages were nested, showing a pattern suggesting differences in species richness among localities were more important than differences in prevalence among species (nestedness among rows was higher than nestedness among columns, **Table 3**, **S2 Table**). Nestedness ranks for Guadarrama 2014 was described with GAM

and curvilinear term (spline) for elevation, roughly similar to a third-degree cubic

polynomial (df = 3.89, F = 15.38, $P < 0.0001$, D^2 = 85.6%, **Fig 3**). The subset of these

grasslands sampled in 2015 maintained their nestedness ranks and the model transferred

to 2015 data (Pearson's correlation between predicted and observed ranks, r = 0.96, t =

7.28, df = 4, P = 0.0018, R^2 = 93%, **S3 Fig**). The model did not transfer between study

regions (r = 0. 74, t = 2.18, df = 4, P = 0.10, R^2 = 54%, **S4 Fig**). Largely, nestedness

increased with elevation, with the exception of the lowest and highest elevation

grasslands.

Table 3. **Nestedness indices for dry grassland ant assemblages in Guadarrama**

range 2014 surveys (18 grasslands, 35 species).

NODF	NODFr	NODFc	T	fill
38.31: **0.001-0.015**	40.62: **0.001-0.001**	37.72: **0.001**-0.103	25.24: **0.001**-0.912	0.23

Indices were derived from the Nestedness metric based on Overlap and Decreasing Fill

(NODF), NODFr for rows (i.e. localities) and NODFc for columns (i.e. species) and

matrix temperature (T). Numbers under headings for indices provide the index values

followed by P-values estimated by equiprobable ('r00') and proportional resampling

('quasiswap') binary null models. Significant P-values ($P < 0.05$) are in bold face. Fill:

matrix fill (sum of 1s/sum of cells).

Fig 3. **Generalized additive model (GAM) of nestedness ranks (based on nestedness**

temperature (T) index) for elevation in ant species assemblages in dry grasslands

of a Mediterranean range in central Spain.

Curve and 95% IC from GAM with Gaussian errors fit to the Guadarrama 2014 survey

(black circles). Nestedness ranks for the Guadarrama 2015 (grey circles) and Serrota

2015 (open circles) surveys are superimposed. Low ranks indicate poorer species

community richness, where species are a subset of richer species communities.

Elevation values were slightly jittered to avoid overlap among some points.

Species turnover and nestedness were significant beta diversity components, although

species turnover among grasslands was more important than pure nestedness of

assemblages (β_{SOR} = 0.88, β_{SIM} = 0.79, β_{NES} = 0.09, all *P*-values = 0.001). This

relationship among beta diversity components did not vary among surveys (**S5 Table**).

Discussion

Ant species richness and taxonomic composition changed predictably with elevation in

the Mediterranean mountains we investigated, exhibiting largely consistent patterns

between years and among regions. These changes were more notable along mid-

elevations and associated more clearly with differences of species richness among sites

than with prevalence or abundance among species. To our knowledge, this is the first

study specifically designed to describe ant community species richness and taxonomic

composition responses to elevational changes in the Mediterranean Basin. However, we

restricted our study to grasslands, therefore the results might not be a suitable model

system for other habitat types. Notwithstanding, grasslands are among the ecosystems

exhibiting greater economic and ecological relevance in the Mediterranean Basin [65,

66].

Species richness showed a hump-shaped relationship with elevation, peaking at ca. 1100

m a.s.l. This model depiction fit to the primary dataset (Guadarrama 2014), transferred

adequately to the 2015 Guadarrama dataset, and also the spatial validation dataset in the

2015 Serrota range. Despite geometric constraints as mid-domain effects were described

to explain these curve types [30], more recent studies tended to downplay the

importance of these mechanisms [24, 29, 67]. In fact, while mid elevation peaks in

species diversity were reported as relatively common [28, 29], these observations are far

from universal, suggesting regional factors play an important role in driving elevational

patterns of species diversity. Interestingly, research showed monotonic increases in ant

species richness were common in temperate climates [19-21], where elevation was

correlated with temperature. In arid climates, monotonic decreases were observed [22,

23], where hydric stress was ameliorated with altitude. Indeed, hydric stress and

temperature are the two main elevational correlates in Mediterranean climates, with

water availability limiting communities at low elevations, while at high elevations cold

temperature stress is more important [6, 7]. This double constraint might drive a hump-

shaped spatial pattern of ant species richness in Mediterranean mountains.

Most studies on ant communities reported species richness positively responded to

temperature [19-21, 24, 26, 27, 67, 68], but a number of findings also emphasized the

role of water availability [22, 29, 33, 34, 69]. Cold temperatures can reduce local ant

species richness by direct effects that limit ant activity, such as reduced foraging time

[35, 70], and, at larger biogeographical scales, cold temperatures can also limit species

pools by lowering speciation rates [67]. Azcárate *et al*. [70] reported low humidity, in

turn, can limit colony activation by reducing foraging possibilities, and therefore

constitutes a main environmental filter. Alternatively, temperature and water availability

can facilitate ant species richness indirectly, these abiotic factors affecting primary

productivity and therefore the range and abundance of resources ant colonies exploit

[33, 35-37]. Diversity-productivity hypothesis states that as productivity increases, so

does the availability of energy and resources, so density and size of colonies also

increase. These relationships should increase local species richness by abundance,

lowering local extinction risks or activating a sampling mechanism [71, 72]. In

Mediterranean climates, primary productivity can be limited by cold temperatures and

summer drought [73], so maximum primary productivity values likely occur at

intermediate elevations, where summer drought is not too severe and temperatures are

not too cold. Therefore, if species richness increases with primary productivity, a

humped curve is also expected.

As anticipated, elevation also affected the taxonomic composition of ant communities.

We observed a ca. 1700 m threshold separating high mountain ant species communities

from those along the lower elevational gradient. This threshold was consistent among

regions and between years. Furthermore, although nestedness was a significant

component of beta diversity, most elevational shifts among communities were

attributable to species turnover. This result suggested the communities with lowest ant

species diversity (high mountain communities) were also the most unique in a regional

context. The singularity of high elevation ant communities was previously observed in

different biomes [22, 24, 34, 35], which reinforced the conservation value of the

Mediterranean mountains reported for other taxa [74, 75].

The examination of community nestedness suggested current ant assemblages are

context-dependent and likely highly vulnerable to global warming and other

anthropogenic changes. This is because, first, species turnover exhibited a dominant

role over pure nestedness, which indicated different and characteristic groups of ant

species occurred along an elevational gradient, and at the very least, ant species

community composition was distinct at high elevations (> 1700 m) (**Fig 2**). We want to

emphasize that if species distributions followed gradually changing abiotic conditions

(e.g. a decreasing temperature gradient with elevation), then we would expect a

concomitant change in the taxonomic composition of ant assemblages, rather than the

development of new, characteristic, species groups [42, 76]. It is also illuminating to

consider that nestedness arises with contributions from differences among species and

among sites [59]. Nestedness patterns elsewhere were explained as a simple sampling

mechanism, where the smaller species group in less diverse, i.e. poorer communities

were a subset of species in more diverse, i.e. richer communities, reportedly when the

studied localities differed in resource availability (or productivity) and this factor

determined species richness [72, 77]. However, this implied species attributes notably

contributed to nestedness, because rarer species with small distributional ranges (often

specialists or poor-dispersers) were not generally sampled by the least diverse, i.e.

poorest communities, while the more common taxa exhibited a wide distributional

range [61]. In our study, the least diverse communities were more nested than diverse

ant communities, but nestedness among species was less important than among sites.

We conclude the sampling mechanism did not fully explain the nested patterns in the

ant communities we identified and habitat and abiotic filtering provided more plausible

explanations for the patterns. However, we carefully controlled by design the habitat

type (arid grasslands in gentle south facing slopes), which implied those habitat filters ─

if an integral component — should be fine grained. Nestedness patterns along the

elevational gradient were temporally constant, but varied between mountain ranges,

indicating ant community composition was spatially dependent. Finally, an historic

explanation of current ant species distribution patterns in a landscape context is an

important element to include. We studied a mountain range within the Mediterranean

biome where major landscape changes have not occurred over the last thirty years.

Extensive afforestation was conducted during 1950's and a progressive abandonment of

the region took place between the 1960's and 1980's. Therefore, we assume all sites are

equally accessible from the regional species pool. However, we cannot disregard other

time-related mechanisms, which might explain community composition, such as priority

effects favoring one or more species, depending on their relative order of arrival to a site

[78], or metapopulation dynamics involving local extinctions and recolonizations in

periods beyond this two-year study [79].

Overall, species taxonomic composition varied non-linearly with elevation. The

insufficiency of the sampling mechanism emphasizes that ant assemblages are highly

vulnerable to global changes on climate and anthropogenic impacts, which might affect

species at the habitat and/or community level. It is unlikely species' range shifts that

track environmental change will maintain current ant species community composition.

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Supporting information

S1 Table. Species presence in each survey.

S2 Table. Nestedness indices for dry grassland ant assemblages in surveys from all

study areas in central Spain. Guadarrama range (-G2014- year 2014: 18 grasslands,

35 species; -G2015- year 2015: 6 grasslands, 26 species) and Serrota range (-S2015-

year 2015: 6 grasslands, 20 species). Indices are Nestedness metric based on Overlap

and Decreasing Fill (NODF), NODFr for rows (i.e., localities) and NODFc for columns

(i.e., species) and matrix temperature (T). Numbers under headings for indices provide

index values followed by P-values estimated by equiprobable ('r00') binary null models

and proportional resampling ('quasiswap'). Significant P-values ($P < 0.05$) are in bold

face. Fill: matrix fill (sum of 1s/sum of cells).

S3 Fig. Temporal validation of the relationship between nestedness and elevation.

Generalized additive model (GAM) of nestedness ranks (based on nestedness

temperature index) on elevation in ant assemblages from Guadarrama range (central

Spain) fit to Guadarrama 2014 data and validated with Guadarrama 2015 data.

S4 Fig. Spatial validation of the relationship between nestedness and elevation.

Generalized additive model (GAM) of nestedness ranks (based on nestedness

temperature index) on elevation in ant assemblages from Guadarrama range (central

Spain) fit to Guadarrama 2014 data and validated with Serrota 2015 data.

S5 Table. Multiple-site dissimilarities of beta diversity. Multiple-site dissimilarities

accounting for the spatial turnover (β_{SIM}) and the nestedness components (β_{NES}) of beta

diversity, and sum of both values (β_{SOR}) for dry grassland ant assemblages in all study

areas in central Spain. G2014: Guadarrama range 2014; G2015: Guadarrama range

2015; S2015: Serrota range 2015. *P*-values estimated by equiprobable binary null

models ('r00') are given between parentheses. Significant *P*-values ($P < 0.05$) are in

bold face.