

Community Ecology in an Afforested Mediterranean Ecosystem: Consequences for Conservation

Thesis submitted in partial fulfillment
of the requirements for the degree of
“DOCTOR OF PHILOSOPHY”

By

Regev Manor

Submitted to the Senate of Ben-Gurion University
of the Negev

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Beer-Sheva

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Approved by the advisor

Approved by the Dean of the Krietman School of advanced Graduate Studies

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ACKNOWLEDGEMENTS

This study would have suffered significantly without the cooperation of the following people which I am happy to acknowledge. I feel grateful to Iris Bernstein from Jewish National Foundation (JNF) who let me conduct “zoological surveys” although it was against the original goal of JNF, and to Zeev Kuler from Israel Nature and Parks Authority (INPA) who let me use INPA Sherman traps. Thanks are deserved to my adviser David Saltz who let me do things ‘my way’ and was helpful and patient. Thanks are also deserved to Noga Ben-Dor, Ogen Ben-Dor and Ma’ayan Manor for their involvement in field work. Finally, I want to thank Danit for her crucial help and support in the field and outside it.

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COMMUNITY ECOLOGY IN AN AFFORESTED MEDITERRANEAN ECOSYSTEM: CONSEQUENCES FOR CONSERVATION

ABSTRACT

The general motivation behind my Ph.D. research was to show that we can rely on existing ecological theories (mainly in community ecology), when collecting data regarding ecological interactions, in order to assess human impacts and develop practical management tools to reduce the negative effects of these impacts in a given habitat. Specifically, my thesis addresses 3 interconnected subjects:

1. Consequences of homogenization in terms of community. One example of landscape-homogenizing is the establishment of plantations. I studied the effect of planted forests by contrasting plant and small-mammal community composition between planted tree stands and adjacent natural habitat in two different Mediterranean habitats in Israel: (1) inland habitat where I focused on pine (*Pinus halepensis*) and carob (*Ceratonia siliqua*) stands, and (2) coastal sand dune habitat where I focused on planted acacia (*Acacia saligna*) stands. I first wanted to verify whether planted trees modify plant species composition, and second, if and how the small-mammal community is affected by the different habitat conditions created in plantations with different canopy cover. I was especially interested in the abundance of the commensal house mouse (*Mus musculus*). All tree stands underwent biotic homogenization indicated by abundance of house mice coupled with lower diversity of indigenous vegetation and small-mammal abundances and diversities. Habitat structural diversity was positively related with small-mammals diversity and was lower in artificial tree stands in both habitats. My results suggest that using the abundance of a commensal generalist species such as the house mouse relative to other more specialist small-mammals is a good approach to determine ecosystem integrity. Precommercial thinning treatment is a potential management tool to maintain a proportion of native tree species within the canopy of planted tree stands. However, until sufficient data are available for making generalizations, the exact level of thinning necessary to reverse the homogenization processes in man-made plantations and keeping indigenous small-mammal communities diverse and less prone to invasion must be determined empirically.

2. The impact of human activities on density dependent habitat selection. Density dependent habitat selection at the community level is considered a major determinant of biodiversity at the local scale, and knowledge of these processes and how they are impacted by human activities is highly relevant to conservation. By observing the competitive relationships between a specialist and a generalist we can gain useful insights about how different environmental factors determine species abundance and distribution and consequently biodiversity. In this part I describe a study of density dependent processes that determine the community structure of two rodents: a specialist – the broad toothed mouse (*Apodemus mystacinus*), and a generalist - the common spiny mouse (*Acomys cahirinus*) in a Mediterranean maquis habitat, and how this structure is impacted by stands. I carried out two field experiments: The first, based on open field trapping, looking at how rodent communities change with habitat structure. The second experiment was an enclosure study aimed at validating the habitat preferences and competitive relationship between the specialist and the generalist. I identified asymmetric competition relationships in which the specialist was dominant over the generalist. Competition intensity was higher in plots with less than 10% oak cover (mostly within pine stands), where the abundance of both species was low. The specialist was less abundant as the habitat was more homogenized. Competition was found only during the limiting season (summer). Based on these findings I could produce management recommendations to keep indigenous small mammals' biodiversity high. Specifically, anthropogenic planting of pines creating dense pine canopy which reduces habitat structural diversity is predicted to increase competition between the rodents and thus, to decrease their abundances and diversity. Density dependent habitat selection processes play an important role in determining biodiversity, and understanding the mechanisms driving these processes is necessary if changes in biodiversity in response to human disturbance are to be understood.

3. The importance of scale when selecting indicator taxa. Empirical data has repeatedly indicated that species richness within taxa is not indicative of richness in other taxa in a given area. Thus, the search for indicator taxa of biodiversity or human-disturbance has met with little success. In this part I compared human-altered (afforested) habitat vs. Mediterranean maquis and examined the suitability of small mammals and birds as indicator taxa. I studied if and how

different taxa biodiversity and composition can be used as meaningful indicators for human impact by combining information across species groups at different spatial scales. I assumed that the reason for the lack of correlation between bird and small-mammal taxa biodiversity is due to taxa mobility. Specifically, the more mobile taxa can overcome isolation effects more easily resulting in responses on different spatial scales. Within taxa, I expected the more generalist species to be less affected by human-altered habitats than specialist species. Using bird transects, small-mammal trapping, and vegetation transects I found that both vertebrate taxa diversities were correlated with environmental diversity, but on a different spatial scale. Thus, while both groups of taxa can be used as indicators of habitat alteration by humans, they are indicative of alterations at different spatial scales (hundreds vs. thousands of square meters-small mammals vs. birds, respectively). In addition, the generalist house mouse (*Mus musculus*) and Eurasian jay (*Garrulus glandarius*) were much more common within afforested habitat than in natural maquis. Thus, I believe that a new approach should be adopted when trying to estimate ecosystem biodiversity with indicator taxa. Specifically, depending on the scale of conservation measures to be implemented, a set of indicator taxa composed of species with well-known biology should be considered in combination with abundance of the more generalist species, in order to gain a robust, useful, and policy-relevant index.

KEY WORDS: *Acacia saligna*; *Acomys cahirinus*; Afforestation; *Apodemus mystacinus*; *Ceratonia siliqua*; Competition; Conservation implications; Density dependent habitat selection; Ecological indicators; Generalist; Invasive species; Landscape-homogenization; *Mus musculus*; *Pinus halepensis*; Small-mammal community; Specialist.

1. GENERAL INTRODUCTION

‘We are drowning in information while starving for knowledge’

(EO Wilson 1998)

Although ecological theory has developed extensively in the last four decades, vast gaps exist between this theory and its application for conservation (Weiner 1995; Perevolotsky & Pollak 2001; Whittaker et al. 2005). This is not trivial and indeed it is very difficult to translate ecological theories to solve practical real life conservation situations (Hilborn & Ludwig 1993; Weiner 1995; Dimitrakopoulos et al. 2004). Simberloff (1988) was even more extreme by claiming that the only contribution of ecological theory to conservation is the trivial rule that in any habitat more area is better than less area... and no other ecological quantitative rules can be used by conservationists. The limitations of the ecological research in helping to solve practical conservation situations is a result of the unique ecological experimental difficulties: complication of ecosystems, the time needed to conduct ecological experiments, and the limited ability for repetition because of the changing characteristics of the ecosystem in space and time (Hilborn & Ludwig 1993). Generally, it seems that the main stream of ecology, which emphasizes interactions among individuals and species and especially competition, has not been well integrated into conservation (Perevolotsky & Pollak, 2001). The general motivation behind my Ph.D. research was to show that we already can rely on existing ecological theories in community ecology, when collecting data regarding ecological interactions, in order to assess human impacts and develop practical management tools to reduce the negative effects of these impacts in a given habitat.

In this thesis I show that by relying on community ecology we can gain useful conservation insights and practical implementation recommendations in real life conservation situations. This can be done using the following protocol: The first stage would be to identify the human impact(s) that affects our studied ecological entity (population, community or ecosystem). The second stage would be to separate this human impact from other potential predictors. This can be achieved by locating similar habitats with varying levels of the same type of human disturbance and comparing among them. The third stage would be monitoring the desired ecological entities. Of course, we must study the different ecological entities in the appropriate time and spatial scales. By so doing we monitor the habitat and species within it in a way that reflects the structural diversity from the organism(s)

point of view. Finally, if we will study species with well-known biology, we could conclude by the appearing trends of species abundances and interactions how human impacts affect the ecosystem heterogeneity and functioning, and how to curtail this human affect to improve ecosystem performance under the given circumstances.

Habitat alteration and the intentional or accidental introduction of new species are the leading causes of the global biodiversity crisis (Wilcove et al. 1998). The most common way to monitor how humans affect any given ecosystem is by measuring biodiversity. However, although it might be possible to predict changes in the richness of inadequately investigated groups (as opposed to groups for which enough information is available - e.g. biodiversity indicators), the potential of a correlative approach to species richness is strongly diminished by the overall low level of variance explained (Wolters et al. 2006). Here I study how afforestation affects the structure of the small-mammal community and the invasive success of commensal species using insights from community ecology theory. Also, I examined why so often the diversity of one taxocene does not reflect the diversity of other taxocenes, thus putting the notion of biodiversity indicators in question. In the first part I describe a study that checked whether afforestation has the same effect in different habitats. I did it by looking at different stands in two different habitats: (1) inland habitat where I focused on pine (*Pinus halepensis*) and carob (*Ceratonia siliqua*) stands, and (2) coastal sand dune habitat where I focused on planted acacia (*Acacia saligna*) stands. I wanted to see if the same homogenization process results in lower abundance and species composition of the indigenous vegetation and small mammal community and how in turn it affects the invasive success of the commensal house mouse. In the second part I describe a study of competition between the two most common rodents in the study area in order to show that we can gain useful insights from such a study and it can help us produce practical management recommendations in order to maintain high diversity of indigenous species. In the third part I describe a study that focused on the importance of scale in selecting biodiversity indicators. Specifically, I checked how taxa with different movement capabilities (small mammals and birds) respond differently to structural diversity. I also suggest a method to use different taxa as useful biodiversity indicators.

2. PART 1: COMMUNITY HOMOGENIZATION AND THE INVASIVENESS OF COMMENSAL SPECIES IN MEDITERRANEAN AFFORESTED LANDSCAPES¹

2.a INTRODUCTION

Although it subsumes key conservation issues such as invasion and extinction, the ecological consequences of homogenization remain relatively unexplored (McKinney and Lockwood 1999; Olden et al. 2004; Qian and Ricklefs 2006). While habitat alteration and the introduction of new species by humans are leading causes of landscape homogenization (Wilcove et al. 1998; Western 2001), homogenization, in its self, may increase susceptibility to invasion (Elton 1958). One of the more common landscape-homogenizing processes is the establishment of tree plantations primarily designed to produce wood fiber. The world-wide area of plantation forests is increasing (FAO 2001), with uncertain outcomes for biodiversity (Moore and Allen 1999; Palik and Engstrom 1999). There are concerns about the unsuitability of such plantation forests as habitat for indigenous species (Armstrong and van Hensbergen 1996; Hartley 2002), yet others argue that such forests can make a contribution to the conservation of indigenous biodiversity (Allen et al. 1995; Norton 1998; Atauri et al. 2004).

A common generalization is that biodiversity enhances resistance to invasion (e.g. Elton 1958; Naeem et al. 2000). The most common cited mechanism behind this phenomenon is that as species accumulate, competition intensifies and fewer resources remain available for new colonists (Case 1990). However, there is ongoing debate about the relationships between biodiversity and invasibility. The complication is probably because variation in diversity is controlled by, and thus co-varies, with the same factors that influence invasibility (e.g. resource availability, physical stress, competitors etc.) (Naeem et al. 2000). Also, environments where resources undergo marked fluctuations as a result of periodic disturbances are prone to invasion (Davis et al. 2005). In contrast to the growing body of research on the patterns, mechanisms, and

¹ This chapter was published in Biological Invasions as: Manor, R., O. Cohen, and D. Saltz. 2008. Community homogenization and the invasiveness of commensal species in Mediterranean afforested landscapes. *Biological Invasions* 10:507-515.

implications of biotic homogenization in plant communities (Schlapfer and Schmid 1999; Loreau et al. 2001; Qian and Ricklefs 2006) and marine systems (Stachowicz et al. 1999, 2002; Rahel 2002; Olden and Poff 2004), the effect of biodiversity on the resistance to invasion of terrestrial fauna communities has received limited attention. Most of the articles on homogenization have used similarity indexes (such as Jaccard's), which of course omits the crucial dimension of species abundance (Olden 2006).

Man-made plantations usually contain one or very few tree species. Thus, the understory layers contain most of the floristic diversity, which, in turn, plays a key role in overall species biodiversity (Wagner 1993). However, because the density and diversity of planted trees are closely linked to understory plant diversity (MacLean and Wein 1977), animal communities in plantations are expected to change (Lautenschlager 1993; Sullivan 1994). At the microhabitat scale, vegetation (e.g. forest structure, forest composition), rather than spatial characteristics, has a major effect on forest-floor small mammal abundance (Schmid-Holmes and Drickamer 2001). However, the relationship between mammals and surrounding land use varies greatly among habitat types (e.g. Kirkland 1990; Fernandez and Dunstone 1994; Runciman and Sullivan 1996; Findlay and Houlahan 1997; Carey and Wilson 2001; Sullivan and Sullivan 2001), and generalizations cannot be made. However, I hypothesized that because man-made plantations contain very few tree species, compositional diversity in the plant community will be low. This will result in decreased diversity and abundance of small mammals and create opportunities for invasive species.

In this part I describe a study of three different plantations types in the Mediterranean zone of Israel: inland pine (*Pinus halepensis*), inland carob (*Ceratonia siliqua*), and coastal sand dune acacia (*Acacia saligna*) looking at the impacts of these plantations on the understory of the local vegetation, the structure of the native rodent community and its susceptibility to invasion. I first wanted to verify whether planted tree canopy cover modifies plants species composition. Second, I studied whether and how the small mammal community is affected by the different habitat conditions created in plantations with different canopy cover. Finally, I focused on changes in the abundance of the invasive house mice (*Mus musculus*), as a function of the changes in the small mammals community. Specifically, I predicted that resulting changes in the

plant community and reduced native small mammals diversity due to the planting of the trees will enable the house mouse, which has the characteristics of an invader (e.g. a wide niche breadth expressed by the ability to exploit a broad array of food and habitat types, small-size, and high fecundity—McKinney and Lockwood 1999; Swihart et al. 2003) to become the dominant species in the small mammal community.

2.b METHODS

2.b.1 Study sites

Much of the Mediterranean zone in Israel consists of afforested areas planted for human leisure purposes. These are typically mono-cultural tree stands covering in total over 850 km². Of the three types of stands we studied, two (pine and carob), were located in typical Mediterranean inland habitat that was dominated naturally by *Quercus calliprinos* and *Pistacia palaestina*. The third stand (acacia) was located in the sandy coastal plains of the Mediterranean area in which the dominant vegetation community was *Artemisia monosperma* and *Retama raetam* bushes. In the past, pines were rather rare, and constituted only a small percentage of the native arboreal vegetation assemblage (Liphschitz and Biger 2001). Furthermore, the planted pines are quite different genetically from indigenous pines (Grunwald et al. 1986) and invade natural habitats and even threaten the indigenous pines (Lavi et al. 2005). The domestic carobs were planted as orchards along dry creeks within pine plantations to increase the grazing value of the habitat for cattle. *Acacia saligna*, originally planted in the study area to prevent sandy-soil erosion, is considered throughout the world an invasive species that has negative impact on natural ecosystems (Kutiel-Bar et al. 2004).

The Mediterranean inland study area was located in Britania park (31°40' N; 34°50' E), a Jewish National Fund (JNF) project covering 40 km² in which most of the pine and carob trees were about 50-year-old. Carob orchards within the park are ploughed every ca. 5 years to reduce competition from shrubs and maximize annuals productivity. As a result, maquis shrub cover in these plots was very low. The coastal sandy dunes study site was near Nizzanim (31°40' N; 34°30' E), in 60-year-old acacia stands. The climate in both habitats is similar with cool/wet winters and hot/dry summers with mean annual precipitation of 500 mm.

2.b.2 Vegetation and small mammal sampling

In the inland habitat, small mammals were live trapped in spring (May–June) and summer (August–September) of 2003, 2004 to enable tracking changes in population trends during summer. I defined stability of the small-mammals community as the difference in their numbers between spring and summer, which are the high-resource and lower resource periods, respectively. In the coastal sand dunes, small mammals were trapped during July of 2003 and 2004. In the inland habitat 24 plots were selected to represent the three sub-habitats typical to this area. These included: 1. Pine plantations—12 plots with different proportions of pine and maquis cover; 2. Carobs plantations—4 plots; and 3. Natural Mediterranean maquis—8 plots representing a control. In the coastal sandy dunes I sampled 5 plots in acacia plantations and 5 plots in natural sandy habitat serving as control. In all the three types of stands I studied, I selected plots that were adjacent (>1 km) to natural habitats in which I selected our control plots to rule out the possibility that differences among plots were due to some past event and low colonization and to make sure that movement between test and control plots was feasible. The control plots were spread throughout the study sites in a manner maximizing representation. Trapping in each plot was with 40 Sherman live folding traps (8 × 9 × 23 cm) set for three consecutive nights. The traps were laid out as a grid of 20 rows × 2 columns. Rows were 8 m apart and columns were 15 m apart. Traps contained cotton bedding, and were baited with a dry commercial peanut snack that has proven successful in the past. Traps were opened in the afternoon of day 1, checked and closed or reopened and baited in the mornings and afternoons, respectively, of days 2–4, and then removed. All small mammals captured were marked on the belly with a color marker to identify recaptures, sexed, weighed on a Pesola spring balance, and released at the point of capture. Because mark-recapture population models are extremely sensitive to low sample sizes (Otis et al. 1978; Menkens and Anderson 1988), I used the total number of trapped unmarked individuals (minimum number known alive) as an index of small-mammal abundance. For each plot I calculated a relative index of small mammal density as number of unmarked captures divided by the area covered by the grid (15 · 152 = 2280 m²). Biodiversity was estimated using the sample size independent Simpson index (Rosenzweig 1995) recommended by Buckland et al. (2005).

In most habitats, plant communities determine the physical structure of the environment, and therefore, have a considerable influence on the distributions and interactions of animal species (McCoy and Bell 1991; Díaz and Cabido 2001). Thus, I assessed habitat structural diversity using vegetation transects measuring percent of total canopy cover of planted trees, shrub species, rocks, annuals plants, and low (<0.5m) perennials. In each plot six 10m-long transects were conducted, three along each trap column. In plots where traps were set in both years I repeated the vegetation transects and used the two-year average of the different habitat structural layers and small mammal abundances for my calculations. I measured general habitat structural diversity by the same Simpson index mentioned above by using all the variables measured in vegetation transects. To evaluate vegetative structural diversity I used the same variables excluding canopy of planted trees and rock cover. I used a t-test or analysis of variance (ANOVA) to compare habitat characteristics and small mammal abundances and biodiversity among the different sub-habitats within each habitat and regression to relate between habitat characteristics and small mammals abundances. I used a square root transformation on data that deviated from normality based on a Lilliefors test. If transformations did not result in normality, I used a non parametric Mann–Whitney test.

2.c RESULTS

In the inland Mediterranean maquis I found five species of shrubs: *Pistacia lentiscus*, *Quercus calliprinos*, *Rhamnus palaestinus*, *Phillyrea latifolia* and *Pistacia palaestina*. Small mammal species included broad-toothed mice (*Apodemus mystacinus*), Cairo spiny mice (*Acomys cahirinus*), house mice (*Mus musculus*), Wagner's gerbille (*Gerbillus dasyurus*) and lesser white-toothed shrews (*Crocidura suaveolens*). *Pistacia lentiscus* cover was higher within pine plantations, and maquis cover excluding *P. lentiscus* was higher outside pine plantations. There was no difference in total maquis cover in and outside pine plantations (t-tests: $t = 2.395$, $P = 0.027$; $t = -3.652$, $P = 0.002$; $t = -1.390$, $P = 0.181$, respectively). Both general habitat structural diversity and vegetative structural diversity were higher outside pine plantations (Mann-Whitney test: $U = 30$, $P = 0.025$; t-test:

$t = -2.108$, $P = 0.047$, respectively). There was a negative relationship between the cover of *Pistacia lentiscus* and general habitat structural diversity (Spearman correlation $R = -0.593$, $P < 0.01$; Fig 1). Pine cover was not correlated with vegetative structural heterogeneity ($P = 0.297$). Densities of native small mammals were related to maquis cover in spring and summer (Regression: $F = 19.504$, $P < 0.001$, $r^2 = 0.446$, $Y = -0.002 + 0.019X$; $F = 41.978$, $P < 0.001$, $r^2 = 0.641$, $Y = -0.001 + 0.011X$, respectively Fig. 2a, b). The index of small mammal abundance for native species changed less from spring to summer as Mediterranean maquis cover was higher (Regression: $F = 6.86$, $P = 0.016$, $r^2 = 0.238$, $Y = 0.178 + 0.925X$ —Fig. 2c). The indices of abundance for house mice were significantly lower and for native small mammals were higher in the natural maquis in comparison to carobs and pines plantations (Fig. 3a and b, respectively). As a result, the proportion of house mice within the small-mammal community (Fig. 3c) was the lowest in the natural maquis and the highest in the carobs in spring and summer in the Mediterranean habitat ($P \leq 0.05$ after Bonfferoni adjustment).

In the coastal sand dunes indigenous shrubs were *Artemisia monosperma* and *Retama raetam*. Small mammals trapped included Anderson's gerbil (*Gerbillus andersoni allenbyi*), greater Egyptian gerbil (*Gerbillus pyramidum*), lesser Egyptian jerboa (*Jaculus jaculus*) and house mouse. Both vegetative structural diversity and the proportion of shrubs cover were higher outside *Acacia saligna* stands (t-test: $t = -2.966$, $P = 0.025$; $t = -5.214$, $P = 0.002$, respectively). Density of native rodents was related to shrub cover (Regression: $F = 37.778$, $P = 0.001$, $r^2 = 0.84$, $Y = -0.002 + 0.016X$ —Fig. 4). House mice were caught only inside the *A. saligna* stands.

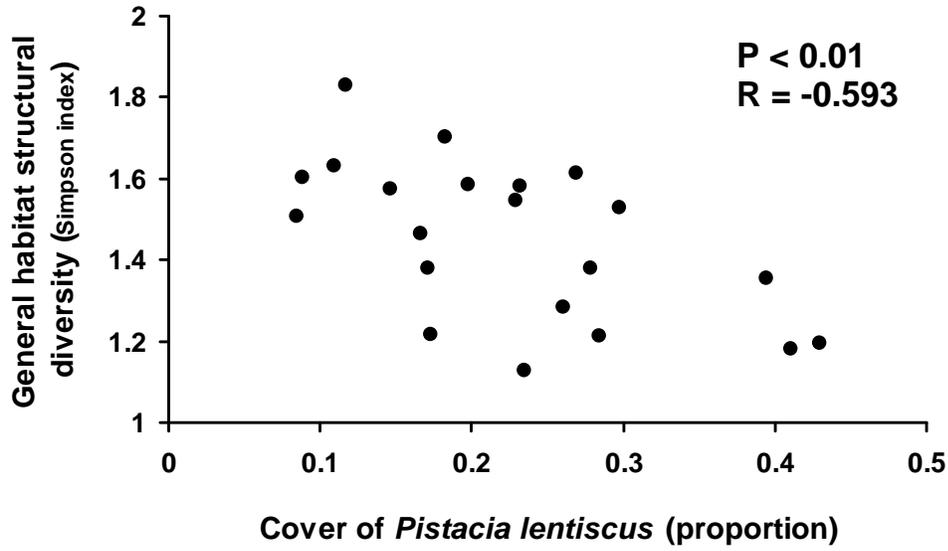


Fig. 1 In the Mediterranean habitat, there was a negative relation between the cover of *Pistacia lentiscus* and general habitat structural diversity.

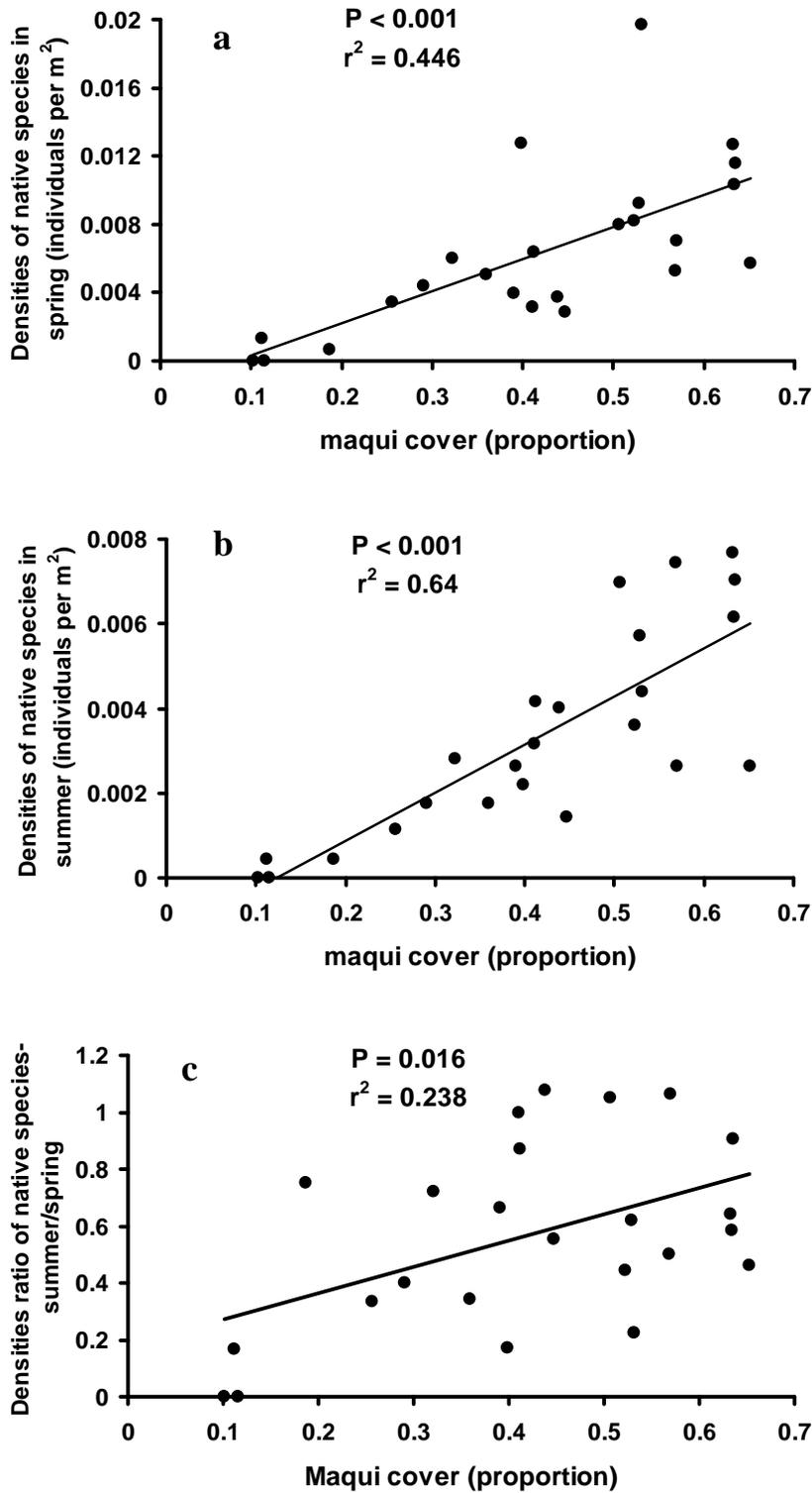


Fig. 2 Densities of native small mammals were related to maquis cover on spring (a) and summer (b). As a result, densities of native small mammals were more stable as Mediterranean maquis cover was higher (c)

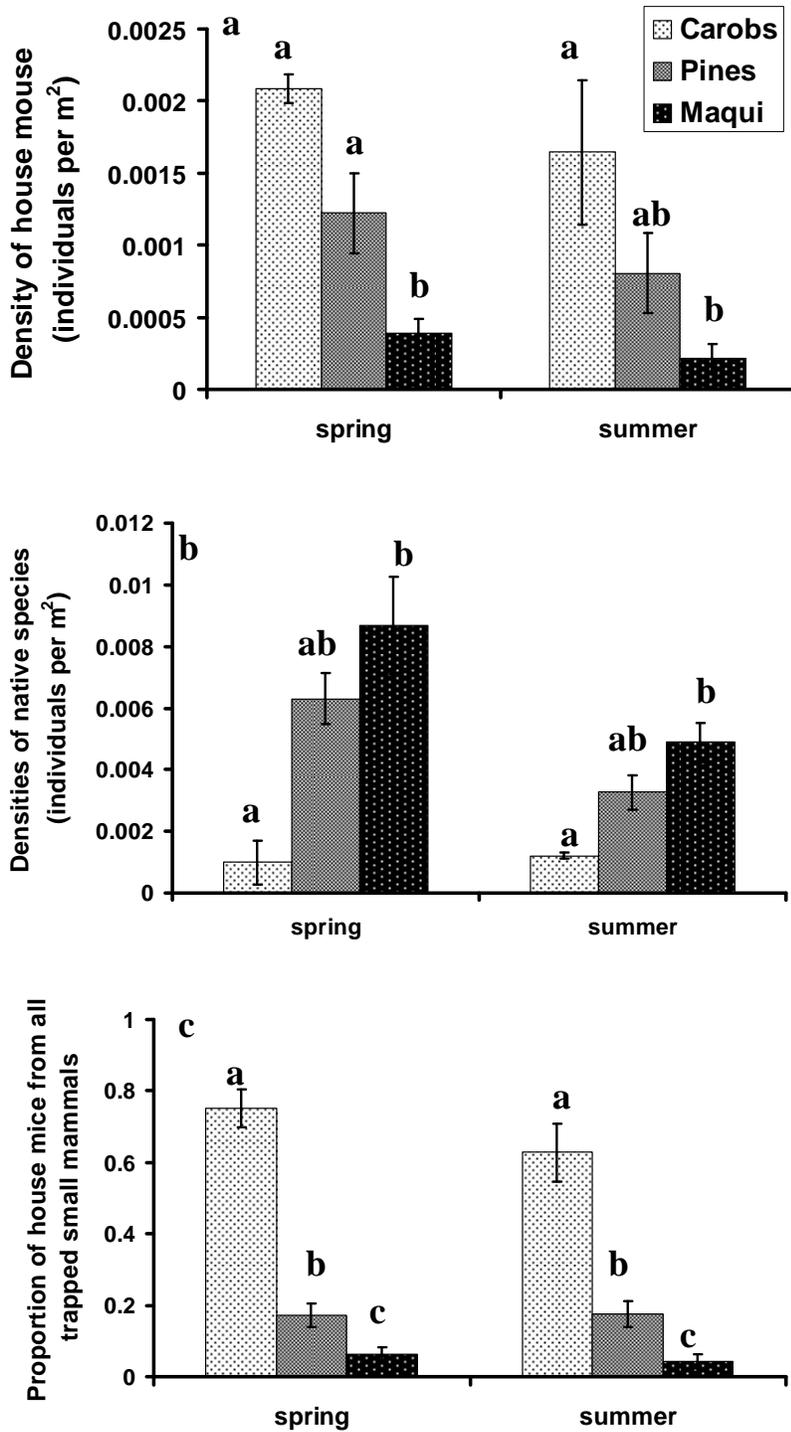


Fig. 3 Densities of house mouse (a), densities of native small mammals (b), and the proportion of house mouse within the small-mammal community (c) among the three different subhabitats on spring and summer in the Mediterranean habitat. Different letters represent a significant difference among subhabitats within each season ($P \leq 0.05$ after Bonferroni adjustment). Bars are 1 SE.

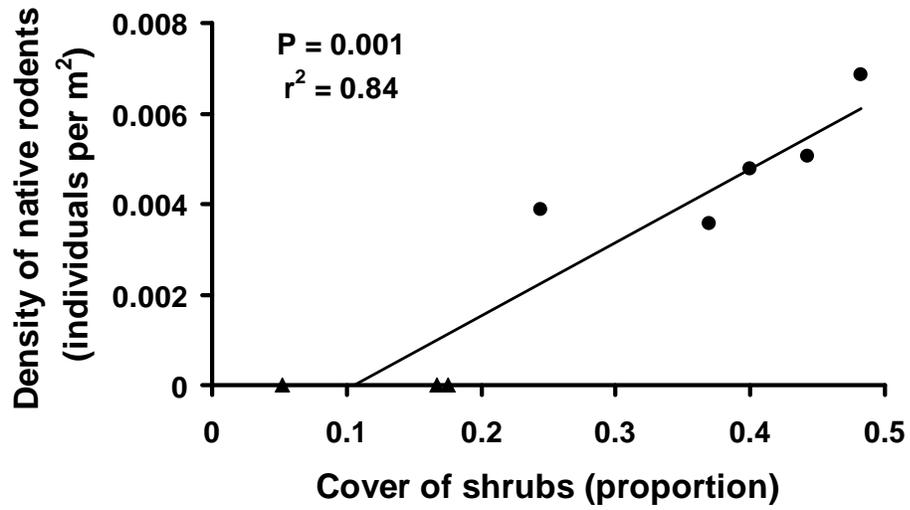


Fig. 4 In coastal sand dunes, density of native rodents is related to shrub cover. Plots represented by triangles (and included in the regression) were inside *Acacia saligna* plantation.

2.d DISCUSSION

According to McKinney and Lockwood (1999) biotic homogenization occurs when a widespread environmental change promotes the geographic expansion of some cosmopolitan, generalist nonnative species and the geographic reduction of regional and endemic native species. Accordingly, all tree stands I studied underwent biotic homogenization indicated by the same process of spreading of house mice coupled with lower diversity of indigenous vegetation and small-mammal abundances and diversity. Most probably, the lack of indigenous small mammals in the planted forests contributed to the house mouse invasion success. The invasive house mouse (originated from northern part of the Indian subcontinent- Boursot et al. 2002; Din et al. 2002), which the indigenous plant and invertebrate species have not evolved with and thus didn't develop means to cope with its presence, is expected to severely impact them, and consequently ecosystem structure and functioning (Smith et al. 2002; Jones et al. 2003).

Niche breadth is a primary determinant of a species' tolerance to loss and change of habitat. Namely, a species' ability to rely upon a broad array of food and habitat types across its geographical range was associated with tolerance to habitat alteration by humans at the population level (Swihart et al. 2003). In an inter-specific context, flexibility in resource use may be an inferior strategy at a local level but provides an advantage at a regional scale by permitting populations of generalists to respond to environmental changes (Ricklefs 1990), including human-induced disturbance. The species-specific relationships among small mammals in my study will be described in the second part. Still, the most tolerant species within my small-mammal community, the house mouse, was scarce in the more natural habitats where indigenous species were abundant (natural sand dunes and natural maquis), and abundant in the disturbed habitats (planted tree stands) where indigenous small mammals were scarce. Furthermore, both indigenous small mammals and the prolific house mouse (which reproduces year round) densities increased through the high resource season (spring) and decreased through the low resource season (summer). These findings suggest that although the immediate causes of the changes in the native rodent communities are not discernable in this study, the absence of the house mouse in the undisturbed habitats (e.g. outside of the plantations) was due to inter-specific competition.

Due to its association with humans and its great adaptability (e.g. it has an omnivore diet which may include carrion, plant material and invertebrates—Berry 1991; Jones et al. 2003), the house mouse is found throughout much of the world and can thrive in environments ranging from equatorial to sub-Antarctic (Berry 1991; Smith et al. 2002). Over much of their range, house mice are predominantly commensal. However, in some situations there are so-called ‘feral’ populations that exist away from human habitation, in crops and in natural and semi-natural habitat, usually where there are few competitors or predators (Berry 1991; Pocock et al. 2005). It is surprising that very few studies, if at all, use human commensal species, such as the house mouse, as indicators of the level of anthropogenic influence and structurally of natural habitats. My results suggest that using the abundance of the house mouse relative to other more specialist small-mammals is a good approach to determine ecosystem integrity (De Leo and Levin 1997). For example, the existence of house mouse in the afforested sandy habitat is far more worrying, from a conservation point of view, than in the afforested inland Mediterranean habitat because the house mouse has overcome the physiological constraints connected to survival in sandy habitats. Thus, the abundance of commensal species may be used as an important monitoring tool for assessing their impact on indigenous species (e.g. predation: Manor and Saltz 2004) or to assess how far a given habitat is from its natural state (this study).

The presence of trees with complex aboveground woody structures and extensive root systems affects soil, water and sediment retention, and climate buffering (Díaz and Cabido 2001). Because the sandy habitat’s indigenous vegetation included only shrubs I expected acacia plantations in the sandy habitat to have a stronger impact on the small mammal community than plantations in the inland Mediterranean habitat where trees are part of the native community. Indeed, the effect of Acacia plantation on indigenous species in the sandy habitat was far more severe in comparison to pine plantations in the inland habitat. Two environmental gradients determine psammophilic rodent distribution on natural sandy habitats: risk of predation (Abramsky et al. 1996) and resource availability (which determine competition intensity) (Abramsky 1988). Although acacia canopy cover increases safety from avian predators, the change in the soil as a result of organic matter dropped from acacia trees enables the invasion of Mediterranean snakes such as *Coluber jugularis*, *Vipera palaestina*, and *Malpolon*

monspessulanus (B. Shacham, personal communication) that may feed on the psammophilic rodents that did not develop (through selection) any protections (behavioral or physiological) against them. Below the acacia trees the cover of annuals is low and ants consume almost all acacia seeds (O. Cohen, unpublished data). Thus, food availability for rodents is very low and they probably experience intense competition. In addition, the change in substrate due to organic matter from the acacias (O. Cohen, unpublished data) makes digging costly both for borrows and for foraging. Consequently, and in accordance with homogenization theory, the specialist species (psammophilic rodents) have retreated from the homogenized habitat (planted sandy habitat). In the absence of competition, the generalist species (commensal house mouse) has proliferated in this habitat (McKinney and Lockwood 1999; Swihart et al. 2003). The carob orchards too, were characterized by low coverage of undergrowth and similarly the house mouse dominated the small mammal community. In the pine stands, densities of indigenous small mammals were related to maquis cover. In addition, densities of native small mammals were more stable through summer when the Mediterranean maquis cover was higher. This stability may explain why these habitats were less prone to house mouse invasion.

In contrast to what could be deduced from other studies (e.g. Lehmann and Perevolotsky 1992) I want to emphasize that understorey plant assemblages in the studied pine stands were highly dynamic and ranged from diverse with many indigenous species to virtually non-existent. Broadly similar patterns of a variable understory have been reported for pine plantations in other countries (Allen et al. 1995; Armstrong and van Hensbergen 1996; Geldenhuys 1997; Ogden et al. 1997; Lavery and Mead 1998). To date, there is no clear understanding of the factors responsible for this variability in the understorey composition. Several authors describe the occurrence of strong successional processes (Allen et al. 1995; Ogden et al. 1997), possibly caused by combined effects of low light availability and intense belowground competition from the dense tree layer. Both are known to be major direct or indirect factors in forest successions (Connell and Slatyer 1977; Mitchell et al. 1999; Thysell and Carey 2001; Roberts 2002). Indeed, the negative relation between pine canopy cover and maquis cover documented in this study supports the notion that competition for light and/or other resources was a problem for indigenous vegetation and promoted biotic homogenization in the pine plantations. Further, one

generalist, shade-tolerant, maquis element—*Pistacia lentiscus*—has become dominant within planted pine stands and masked the decrease in cover of other maquis native species. As a result, habitat structural diversity was reduced, and with it small-mammal diversity making the system susceptible to invasion by the house mouse. As Abramsky (1978) demonstrated, different habitat manipulations can create different small mammal communities including colonization of new species and avoidance by native species. Pre-commercial thinning treatment is a potential management tool to maintain a proportion of native tree species within the canopy of plantations (Hartley 2002). This will cause structural diversity to increase and thus homogenization processes will be reversed in man-made plantations. This in turn will keep indigenous small-mammal and other faunal communities diverse (DeGraaf et al. 1991; Carey and Johnson 1995) and probably less prone to invasion. Until sufficient data accumulates enabling generalizations, the exact thinning protocol should be stand type specific and based on empirical findings.

3. PART 2: CONSERVATION IMPLICATIONS OF COMPETITION BETWEEN GENERALIST AND SPECIALIST RODENTS IN MEDITERRANEAN AFFORESTED LANDSCAPE ²

3.a INTRODUCTION

Intra-community species diversity reflects the spatial and temporal heterogeneity of the environment (Chesson and Rosenzweig 1991; Rosenzweig 1991). Consequently, anthropogenic habitat alteration, its pattern and its scale are expected to have a profound effect on community structure. Questions concerning species' coexistence are commonly related to specialization in resource utilization (e.g. Kotler and Brown 1988; Wilson and Yoshimura 1994; McPeck 1996; Morris 1996) with most natural communities appearing to be composed of competing species, that vary from those with very narrow habitat requirements (specialists) to others with much wider habitat requirements (generalists) (Morris 1996). The importance of competition in determining community structure and species abundance and distribution is well documented (Connell 1983; Schoener 1983). If plasticity mechanisms (either behavioral or physiological) that allow generalists to be tolerant of variable environmental conditions also have increased costs (either increased mortality or decreased mean capture rates of resources), the coexistence of generalist and specialist is possible (Abrams 2006). This assumed trade-off structure has a dramatic impact on the potential for ecological coexistence (Wilson and Yoshimura 1994; Egas et al. 2004). Under optimal conditions specialists tend to outperform generalists, but when the environment is variable or unpredictable, the costs associated with being a generalist may be small in comparison to the benefits of the increased behavioral or physiological plasticity, and the generalist may be favored (Rutherford et al. 1995; Brown 1996).

Because habitat selection by individuals can strongly influence the likelihood of coexistence between competitors (Rosenzweig 1991), density dependent habitat selection at the community level is expected to be a major determinant of biodiversity at the local scale, and knowledge of these processes and how they are impacted by human activities is highly relevant

² This chapter was published in *Biodiversity and Conservation* as: Manor, R. and D. Saltz. 2008. Conservation implications of competition between generalist and specialist rodents in Mediterranean afforested landscape. *Biodiversity and Conservation* 17:2513-2523.

to conservation. Field studies have shown that habitat generalist species are more likely to find human-altered landscape usable (Laurance 1991; Andr n 1994; Swihart et al. 2003). In addition, species that are dietary generalists should be less negatively affected by fragmentation, because they are less susceptible to increased variation in the availability of native food resulting from habitat loss (Swihart et al. 2001). The niche-breadth hypothesis (Lanchier and Neuhauser 2006) thus predicts that a generalist species with the ability to use a broad array of habitats and food types should be less adversely affected by fragmentation and alteration of native habitat than a more specialist species (Swihart et al. 2003). This, in turn, would impact biodiversity by reducing the number of specialist species because they would be out-competed in human altered habitats.

By observing the competitive relationships between specialist and generalist we can gain useful insights about how different environmental factors determine species' abundance and distribution. Here I present a study on how anthropogenic habitat alteration, namely planted pine stands, impacts a small-mammal community in a Mediterranean landscape by means of density-dependent processes. Specifically, I studied the competitive interaction of the broad-toothed mouse (*Apodemus mystacinus*, hereinafter BTM) and the common spiny mouse (*Acomys cahirinus*, hereinafter CSM). The BTM is an arboreal habitat specialist that is omnivorous but prefers oak (*Quercus Calliprinus*) acorns in its diet (Granot 1978; Haim and Izhaki 1994). The CSM is a habitat generalist found in both Mediterranean and desert rocky habitats (Haim and Tchernov 1974; Blaustein et al. 1996). It too, is omnivorous, but prefers arthropods in its diet (Kronfeld-Schor and Dayan 1999). Both species are similar in size – 35g vs. 40g, respectively.

I expected to find that in a more natural habitat, competition between the competitors would be less intense as the habitat structural diversity is higher. Furthermore, in the less altered habitat competition would be asymmetrical, with the specialist impacting the generalist more than the generalist impacting the specialist, because in that habitat the specialist's advantage should be manifest. Finally, the competitive interaction should intensify during the dry season, as food supply declines. I validated habitat preferences and competition relationships by conducting short-term enclosure experiments. Finally, I examined whether studying competitive

species in human dominated habitats can produce a useful management plan to preserve biodiversity of indigenous species.

3.b METHODS

I carried out two field experiments. The first was based on open field trapping, looking at how rodent communities change with habitat structure. Habitat ranged from plots dominated by planted pine stands through mixed maquis with planted pine, to mature natural maquis stands consisting mainly of oak. The second experiment was an enclosure study aimed at validating the habitat preferences and competitive relationship between the BTM and the CSM.

3.b.1 Open field study

The data for the open field study collected from the same trappings described in the first part. To reveal competition interaction I used stepwise GLM using the procedure of normal standardization of species census data before calculating regression parameters (all measured environmental factors mentioned above and standardized numbers known alive of each species) as proposed by Fox and Luo (1996).

3.b.2 Enclosure experiments

I used 40m×90m enclosure built inside a pine grove with 30% canopy cover within the study area. It was constructed of 10-mm wire mesh buried 3 cm into the ground and standing 100 cm high. The top 30 cm of both sides of the fence were covered with aluminum flashing to prevent mice from climbing over. The enclosure contained three different habitats: 1. Mixed maquis (mixture of developed trees and shrubs of maquis elements including *Pistacia lentiscus*, *Quercus calliprinos*, *Rhamnus palaestinus*, *Phillyrea latifolia* and *Pistacia palaestina*) with ca. 70% vegetation cover. 2. Open maquis with the same species but much less developed and much sparser with ca. 30% vegetation cover. 3. Oaks in which oak *Quercus calliprinos* was almost the only maquis element (>90%) with ca. 75% vegetation cover. Forty-one Sherman traps were laid out in an 8×5 grid (the extra one was set because the enclosure was not a perfect rectangle), 15 traps within oaks, 11 traps within open maquis and 15 within oaks according to the proportion of

each habitat type within enclosure area. I cleared the enclosure of resident rodents by continuous trapping. I considered the plot vacant after no individuals were caught for 3 consecutive days. The removed individuals were released in a similar habitat 10km away.

I captured CSM and BTM in the study area outside of the enclosures and kept them in captivity up to 2 weeks before they were introduced into the enclosures. I released 13 CSM inside the enclosure, let them acclimate for 1 night and then monitored their distribution using the Sherman live trap grid for 2 consecutive nights. Then, I added 13 BTM, gave them 1 night to acclimate and again conducted the trapping for 2 consecutive nights. Then, I removed all CSM and gave BTM 1 night to acclimate and again trapped for 2 consecutive nights. Thereafter I trapped out the BTM and cleared the enclosure. I repeated this protocol 3 times during July-September 2005, each time with naïve individuals. To reveal habitat preferences I used one sample t-test. I set the proportion of traps I put in each habitat type as H_0 mean. I used the mean value (of the 2 consecutive trapping night in each stage) of the proportion of trapped rodents (each species separately) in the three consecutive repeats in each habitat type as the values to contrast with the H_0 mean.

3.c RESULTS

3.c.1 Open field study

In the two springs I trapped a total 602 small mammals. Of these 61.8% were BTM, 22.59% were CSM, 11.3% were house mouse (*Mus musculus*), 3.8% were lesser white-toothed shrew (*Crocidura suaveolens*), and 0.5% were Wagner's gerbille (*Gerbillus dasyurus*). In the two summers I trapped 330 small mammals, of which 50.9% were BTM, 28.5% were CSM, 12.4% were house mouse, 6.7% were white-toothed shrew, and 1.5% was Wagner's gerbil. There were more BTM, CSM and house mice in spring in comparison to summer in each plot (paired t-tests: $t=4.241$, $P<0.001$; $t=2.042$, $P=0.05$; $t=2.756$, $P=0.012$, respectively). Recapture rate did not differ between seasons – 0.418 in spring and 0.422 in summer (paired t-test, $t=-0.185$, $P=0.856$)

Oaks cover was more than 3 times higher in natural maquis vs. pines (proportion of 0.2 ± 0.045 vs. 0.065 ± 0.021 ; t -test: $t = -3.005$; $P = 0.007$; fig 5). In plots where oak cover was $>10\%$ (both in maquis and pines) habitat structural diversity was higher in comparison to plots with $<10\%$ oaks cover. (Simpson index: 1.573 ± 0.024 vs. 1.302 ± 0.094 ; t -test: $t = 2.813$, $P = 0.013$)

In the stepwise GLM I found that in spring, after removing one outlier, only total maquis cover had an effect on standardized # of BTM ($t = 3.978$, $P = 0.001$). Similarly, only total maquis cover had an effect on standardized # of CSM ($t = 2.412$, $P = 0.026$), but the effect of the standardized # of BTM was equivocal ($t = -2.03$, $P = 0.057$). Combined # of both CMS and BTM in spring, in plots with more than 10% oaks cover, were not significantly different than the combined # in plots with less than 10% oaks cover, but did differ from plots with less than 10% cover of total maquis (Mean # of CSM+BTM: 19.714 ± 2.343 individuals per plot vs. 12.917 ± 1.823 vs. 0 ± 0 respectively). Overall the difference in mean # of CSM+BTM among the 3 categories (plots with $>10\%$ oaks cover vs. plots with $<10\%$ oaks cover vs. plots with $<10\%$ maquis cover) was significant ($F = 8.035$, $P = 0.003$). A pair-wise comparison revealed that the difference is between the category of less than 10% total maquis cover and the 2 categories with more than 10% maquis cover but not between the categories with more vs. with less than 10% oaks cover ($P \leq 0.045$ vs. $P > 0.1$, respectively after Bonferoni post-hoc adjustment). Thus I concluded that in spring neither interspecific competition nor oak cover had an effect on CSM or BTM abundance.

In summer only oak cover had a significant effect on standardized # of BTM ($t = 3.032$, $P = 0.008$). However, the standardized # of broad-toothed mice had a negative effect, and total maquis cover had a positive effect on the standardized # of CSM: ($t = -2.417$, $P = 0.026$; $t = 2.986$, $P = 0.008$ respectively). Combined # of both CMS and BTM, in plots with more than 10% oaks cover were much higher than plots with less than 10% oaks cover and plots with less than 10% cover of total maquis (Mean # of CSM+BTM: 13.5 ± 1.022 individuals per plot vs. 5.41 ± 0.53 vs. 0 ± 0 respectively). Overall difference in Mean # of CSM+BTM among the 3 categories was significant ($F = 53.611$, $P < 0.001$). A pair-wise comparison revealed a difference between all 3 categories ($P \leq 0.003$ after Bonferoni post-hoc adjustment). To find competition effect (α) of BTM on CSM in summer in plots with more than 10% oaks vs. plots with less than 10% cover

(in the third category- plots with <10% maquis cover no CSM and BTM were trapped at all) I conducted separate regressions for each cover category in which standardized # of CSM was the dependent factor and standardized # of BTM was the independent factor and found that in both categories BTM numbers affected CSM numbers ($F= 8.286, P=0.028$; $F=12.232, P=0.007$ respectively). Competition effect of BTM on CSM (α) was almost double in plots with less than 10% oak cover vs. plots with more than 10% oak cover (regression coefficient was -1.455 vs. -0.861 respectively).

3.c.2 Enclosure experiment

When only CSM were inside the enclosure they were trapped more than expected in mixed maquis, less than expected in open maquis, and as expected in oaks (one sample t-test: $t=9.049, P=0.012$; $t= -15.463, P=0.004$, $t=2.828, P=0.106$ respectively, fig 7). When BTM was added, CSM were trapped in all areas no different than expected ($P>0.3$ for all areas, fig 7). When only BTM were inside the enclosure they were trapped more than expected in oak, less than expected in the open area and as expected in mixed maquis ($t=4.701, P=0.042$; $t=-44.369, P=0.001$; $t=-0.926, P=0.452$, respectively, fig 8). The presence of CSM did not change this pattern ($t=13.971, P=0.005$; $t=-14.618, P=0.005$; $t=-0.223; P=0.845$, respectively, fig 8).

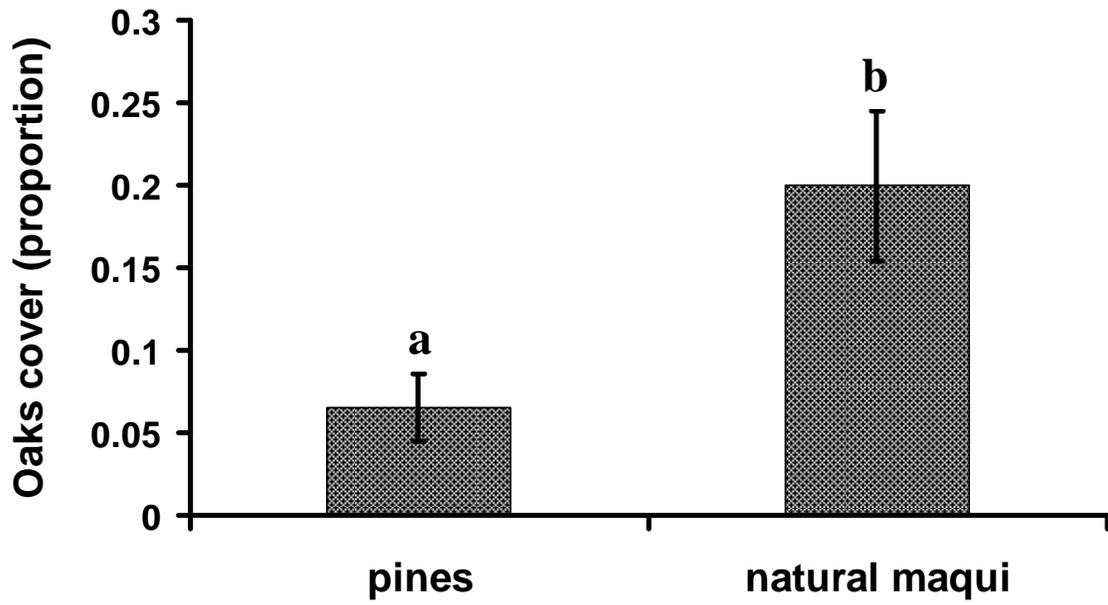


Fig 5 Oak cover was more than 3 times higher in natural maquis vs. pines.

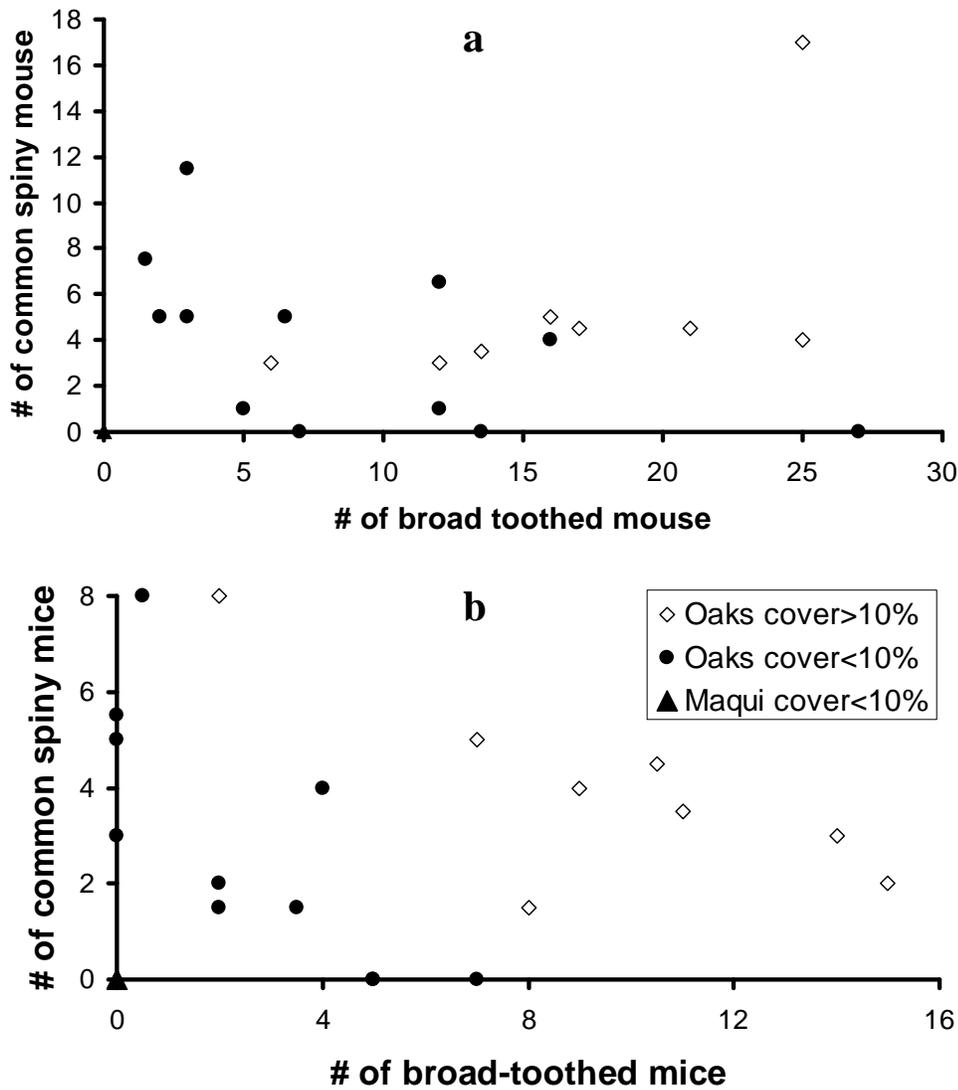


Fig 6 Relationships between broad toothed mouse (BTM) and common spiny mouse (CSM) abundances in spring (a) and summer (b). In spring, only total maquis cover had an effect on both species abundance and no interspecific interaction was revealed. In summer BTM abundance increased with oak cover, and CSM abundance increased with maquis cover and was lower where BTM abundance was higher. In summer, competition effect (α) of BTM on CSM was higher in plots with less than 10% oaks cover, although overall abundance of both species was significantly higher in plots with more than 10% oaks cover (see text for details).

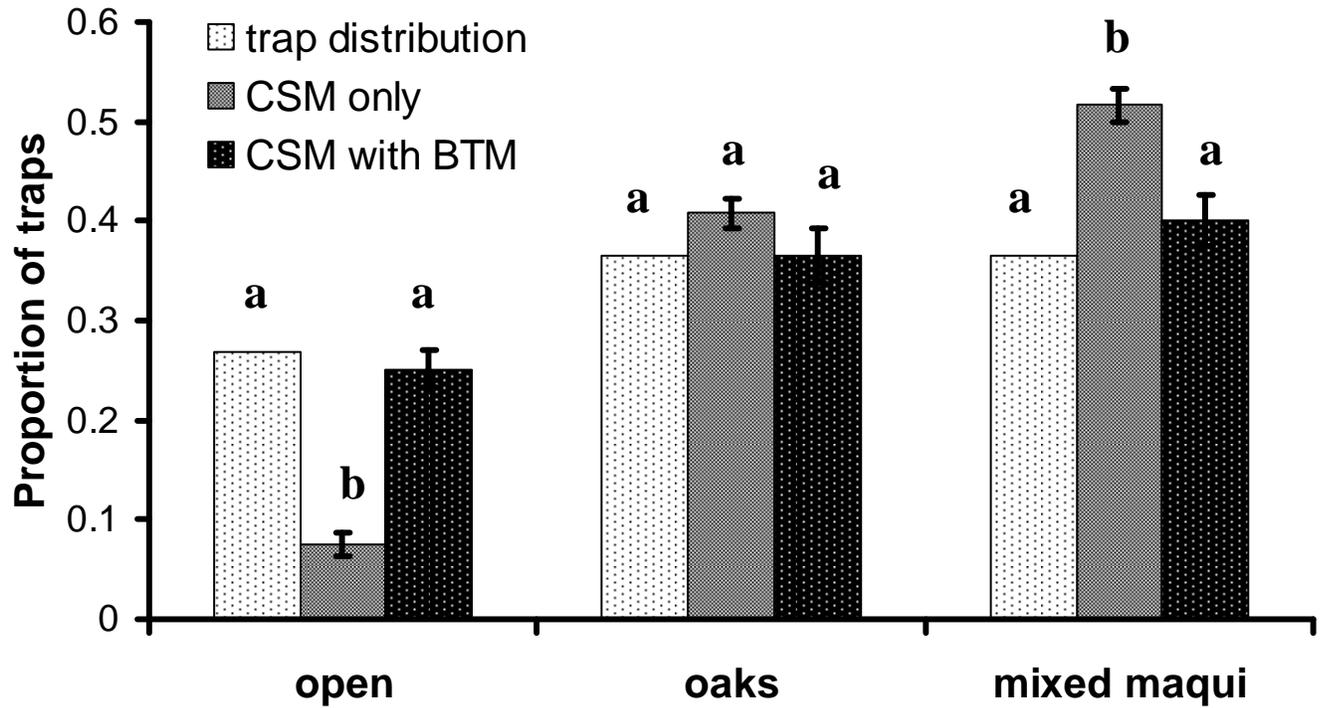


Fig 7 Common spiny mouse habitat selection in an enclosure was affected by broad-toothed mouse presence. It was pushed away from its preferred habitat (mixed-maquis) towards its least preferred habitat (open maquis). Different letters represent a significant difference between the proportion of traps and proportions of trapped common spiny mice within each habitat type (one sample t-test: $P \leq 0.012$). Bars are 1 SE.

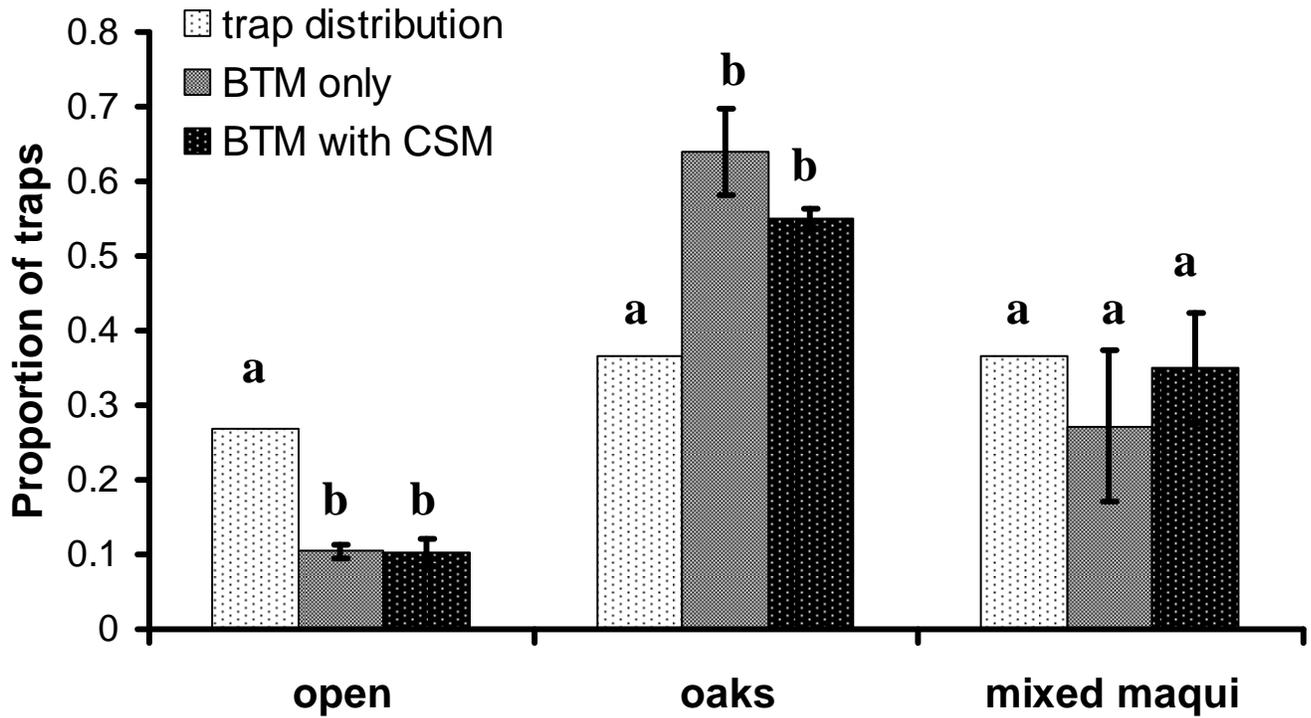


Fig 8 Broad toothed mouse least preferred habitat in an enclosure was open maquis and most preferred habitat was oak. The presence of common spiny mouse did not change this pattern. Different letters represent a significant difference between the proportion of traps and proportions of trapped broad toothed mice within each habitat type (one sample t-test: $P \leq 0.042$). Bars are 1 SE.

3.d DISCUSSION

Knowledge of the way in which organisms select habitat can be useful for understanding and predicting changes in community structure that often follow anthropogenic habitat alterations (Dunning et al. 1995). Indeed, by studying habitat preferences of BTM and CSM and their competitive relationships it became apparent that to sustain high abundances of both species the best habitat is maquis with at least 10% oak cover. In such habitat, despite the fact that both species abundances were the highest, competition intensity during the limiting season remained low.

At the local scale, a combination of competition and environmental fluctuations limit the maximum amount of biodiversity that a trophic level can host (Bastolla et al. 2006), and coexistence is a function of niche dimensions and resource heterogeneity (MacArthur and Levins 1967). Indeed, many kinds of community structure explicitly invoke interspecific density-dependent preferences for habitat (Pimm and Rosenzweig 1981; Rosenzweig 1981; 1991). Individuals do not select habitat independent of the kinds and quality of resources in different places, or the demands on those resources from conspecific and interspecific individuals (Fretwell and Lucas 1970; Rosenzweig 1981; Morris 1988). My study is a good example in which interspecific competition can be mediated via processes of density-dependent habitat selection (Fretwell and Lucas 1970; Rosenzweig 1981; 1991; Morris 1989). Specifically, interspecific competition effect of BTM on CSM was less intense in the more natural (and thus more diverse) habitat (where oak cover was >10%) despite the fact that both species abundances were considerably higher there.

Communities composed of a generalist and specialist species may represent one example of novel mixtures of community organization (Brown 1996; Abrams 2006). The coexistence pattern of the habitat specialist BTM with the habitat generalist CSM can provide useful conservation insights. From conservation point-of-view, the fact the specialist was the most abundant and was dominant over the generalist species (as revealed by the competitive relationships) suggests that the ecosystem in the study area (mostly afforested pines stands) does indeed resemble a native eastern Mediterranean pine forest in which its understory is mainly oak *Quercus calliprinus* (Haim and Izhaki 1994), i.e. the specialist (BTM in this case) exhibited a

specialist advantage and dominated the generalist in a habitat resembling the one that it was selected for (Egas et al. 2004).

This kind of relationships between a specialist and generalist in a human altered habitat can reveal habitat homogenization patterns. According to theory, generalist species are favored when the grain size of the patches is small (costs of specialization are high) and habitat productivities are similar (profits from each habitat are about the same). Specialists have the advantage when habitats are relatively coarse grained, where costs of habitat selection are reduced (Rosenzweig 1981; Brown 1996; Morris 1996). However, the theory continues with the notion that generalists can coexist with specialists as long as at least one habitat is unused or underused by the specialist (Brown 1996; Morris 1996; Egas et al. 2004; Abrams 2006). Indeed, the most generalist species, the commensal house mouse was the most abundant rodent in carob groves within study area that had almost no maquis layer (thus can be regarded as an extreme case of a homogenized and altered habitat), and was least abundant in natural maquis (first part). The second most abundant species within the carob groves was the habitat generalist CSM. The habitat specialist BTM was very rare and was trapped mostly in spring (Manor, unpublished data). Thus, a sound management protocol would be to increase oak cover to at least 10% to curtail the homogenization process and keep indigenous small mammal biodiversity high. Moreover, oak cover was correlated with habitat structural diversity that affects small-mammals community diversity (third part).

The lower competition recorded where oak cover was high is probably due to the fact that the BTM consumes mostly oak acorns and by doing so reduces its diet overlap with CSM that prefers arthropods (Kronfeld-Schor and Dayan 1999). Of course, any other coexistence mechanism (e.g. difference in shelter preferences) shown to account for patterns of coexistence in rodents (e.g. Kotler and Brown 1988) cannot be ruled out. Although the exact coexistence mechanism is not discernible in this study, it is not such a drawback from a conservation point of view, because it does not change the recommended management plan (i.e. increasing oak cover within the understory layer).

Now let me discuss the conceptual difficulty facing anyone who tries to employ density dependent habitat selection theory into real life conservation situations, namely – distinguishing

between habitat preferences and interspecific interactions. The presence of a competitor species can be viewed as a reduction of habitat quality (Abramsky et al. 1991). Thus, the habitat selection behavior can be affected, amongst other things, by the presence of competitors. Sometimes a long strong competition between sympatric similar species may create distinct habitat preferences that eliminate the ability to measure their competitive interaction (e.g. Abramsky 1981). However, it is also possible that competition among such species occurs but just within the limiting season, as I found here. Thus, if temporal change in competition intensity is not taken into account results may be biased. Also, phylogeographic history can give us clues about the possibility of developments of distinct habitat preferences as a result of long, strong competition. e.g. CSM and BTM originated from different continents (Africa vs. Europe; Bates 1994 and Michaux et al. 2005, respectively), and have relatively short histories of coexistence. Thus, habitat preferences for these species probably evolved independent of any current species interaction that, nevertheless maintain the species' coexistence. Of course, if possible, it is recommended (as I did here) to conduct short enclosure experiments in the studied habitats to validate habitat preferences and competitive relationships. Admittedly, without enclosure experiments, it is difficult to separate between the interconnected processes of habitat selection and competition interaction. We must remember, however, that the important question from conservation point of view is how to enable coexistence of indigenous species in human dominated habitats. This usually can be answered if indigenous species with well-known biology and with similar requirements from the environment are studied and their competitive relationship elucidated. Then, it should be possible to link these different processes together, and by so doing, to gain new insights into the pattern and structure of human dominated ecological systems.

Species diversity is perhaps the most desirable and studied entity in conservation biology. Despite the growing concern, no consensus has emerged over the effects of habitat modification on species diversity. Even for comparatively well-studied taxa, disturbance has been reported to increase and decrease diversities with approximately equal frequency (Hill & Hamer 2004). The results of this study indicate that density dependent habitat selection processes play an important

role in determining biodiversity, and that understanding the mechanisms driving these processes is necessary if changes in biodiversity in response to human disturbance are to be understood.

4. PART 3: BIRDS AND SMALL MAMMALS AS ECOLOGICAL INDICATORS: A MATTER OF PERSPECTIVE AND SCALE

4.a INTRODUCTION

Because the standard components of biodiversity (i.e. all species and entire ecosystems) cannot be surveyed adequately within the temporal and budgetary constraints of a planning process (Reyers et al. 2000), systematic conservation planning requires the use of indicators to represent biodiversity in planning protocols (Margules & Pressey, 2000). Although this premise is straightforward in theory, the practice of identifying representative biotic communities is a major challenge, and the search for indicator taxa of biodiversity or human-disturbance has met with little success (Lawton et al., 1998; Ricketts et al. 1999; Perfecto et al. 2003; Thomas 2005; Hess et al. 2006). The limited success is due mainly to a highly variable correlation of species richness between pairs of taxa, the causes of which are not well understood, thus hindering effective conservation-oriented scenario building (Wolters et al. 2006).

An indicator taxon should be a group of organisms whose presence, absence, and wellbeing are readily quantified (in terms of time and budget) relative to other species or environmental conditions of interest. Thus, indicator taxa must have a well-known biology (Landres et al. 1988; Noss 1990) and their distribution must be realistically obtainable (e.g. from limited field surveys). In order to be useful as an anthropogenic impact indicator, the selected species should be typified by high sensitivity to human disturbance. With those characters, species can respond in obvious ways to provide an early warning of human induced ecological and environmental changes (Caro & O'Doherty, 1998). Birds and small mammals are considered some of the best indicator taxa (Wolters et al. 2006), but too, have failed to produce good correlations.

One possible source for the poor correlations is scale of analysis (Ferrier 2002; Hess et al. 2006). A large body of literature reports that maintenance of richness in different organism groups may be determined by the same driving variables (e.g. productivity, structural diversity, climate and extreme environmental conditions – Rosenzweig 1995; Gaston 1996; Waide et al. 1999; Mittelbach et al. 2001; Hawkins & Porter 2003). However, the strength and form of these

relationships may vary due to the influence of disturbance (Kondoh 2001), niche specialization (Kassen et al. 2000), and historical and biogeographic events (Fukami & Morin 2003; Wolters et al. 2006), all of which are scale-dependent (Kotliar & Wiens, 1990; Huston, 1999; Whittaker et al. 2001; Chase & Leibold 2002; Olf & Ritchie 2002).

In this part I examine the suitability of small mammals and birds as indicator taxa in a human-disturbed area. I hypothesized that 1) There will be no correlation between the diversity of these taxa as found in numerous studies (reviewed by Wolters et al. 2006). 2) However, I expected that the diversity of both taxa will respond to the those predictors mostly affected by humans in the study area (i.e. habitat heterogeneity) but at different spatial scales (hundreds vs. thousands of square meters- small mammals vs. birds, respectively; grove scale vs. forest scale in fig 9). Because the decline of regional species pools generally occurs at much higher rates rather than global extinctions (Hughes et al. 2000; Thomas et al. 2004), I addressed this question at the local scales, assessing how different taxa diversity and composition can be used as meaningful indicators for human impact. I assumed that the primary question underlying monitoring is not only whether and at what rate taxa biodiversity changes, but also whether combining information across species groups at different spatial scales makes the index more comprehensible and useful to decision makers. In particular I focused on whether human impact (habitat alteration) can be estimated at different spatial scales by the diversity of the two taxa (birds vs. small mammals) with different mobility. I also checked whether taxa composition in terms of generalist vs. specialist species can help in setting conservation priorities.

METHODS

4.b.1 **Small-mammal surveys**

Small mammals were live-trapped in summer (July-September of 2004 and 2005), which is their limiting season, during which their distribution best reflects their habitat preferences (first part). In 2004, trapping was conducted in 12 plots in the northern part of the park. Which is comprises a mosaic of planted forests most of them pine (*Pinus halepensis*), along with carob (*Ceratonia siliqua*) and other fruit-tree orchards, and small patches of natural maquis. Five plots in maquis and 7 plots in pine stands with varied levels of understory cover. In 2005 I trapped within 6 plots in the northern part of Britania Park (4 in pines and 2 in maquis) and 6 in the southern part which consists mostly of natural maquis and thus were less structurally diverse in comparison to the northern part (southern plots were in maquis with different plant cover). I located plots that were representative of the all-structural diversity of each part. Trapping in each plot was as described in the first part.

4.b.2 **Bird surveys**

To sample nesting bird densities I used 24 line transects (Bibby et al. 1992) of 300m, one near each plot where small mammals were trapped. All transects were surveyed twice during April-May (nesting season) of 2004 and 2005 (i.e. in parallel to the small mammal trappings). The order in which transects were surveyed was chosen randomly each day. All bird species seen or heard were noted. Surveys were conducted under calm weather conditions, lasted 20 min each, and completed before 0830 hrs. All birds within approximately 100m of the transect line were counted. Such strip transects, lacking observer-bird distance estimates, cannot be used to estimate bird densities, but they are efficient at obtaining comparative abundance estimates (Bibby et al. 1992). To determine avian communities composition, I classified each bird species using 2 descriptors: preferred habitat (generalist, grass, shrubs or forest), and regional population trends (expanding, static or shrinking) – based on published reports and personal observation and communication.

4.b.3 Vegetation survey

In most habitats, plant communities determine the physical structure of the environment, and therefore, have a considerable influence on the distributions and interactions of animal species (reviews in Lawton, 1983; McCoy & Bell, 1991). Thus, I assessed habitat structural diversity by measuring percent canopy cover of planted trees, shrub species, rocks, annuals plants, and low (<0.5m) perennials using vegetation transects. In each plot, six 10m-long transects were conducted, three along each trap column. Specifically, I measured local (plot specific) habitat structural diversity by the same Simpson index mentioned above using all the variables measured in vegetation transects.

4.b.4 Data analysis

I used a t-test or analysis of variance (ANOVA) to compare habitat characteristics and small- mammal and bird abundances and diversity among the different sub-habitats (pines vs. maquis, using all 24 plots) or between north and south Britania Park (using 12 plots surveyed in 2005). I applied linear regression to assess the impact of habitat characteristics on small-mammal and bird diversity and abundances. I used a square root transformation on data that deviated from normality based on a Lilliefors test. Means are presented ± 1 SE.

4.c RESULTS

The maquis layer was composed of five shrub species (*Pistacia lentiscus*, *Quercus calliprinos*, *Rhamnus palaestinus*, *Phillyrea latifolia* and *Pistacia palaestina*). The small-mammal community contained five species (broad-toothed mice- *Apodemus mystacinus*, Cairo spiny mice-*Acomys cahirinus*, house mice-*Mus musculus*, Wagner's gerbil-*Gerbillus dasyurus* and lesser white-toothed shrews-*Crocidura suaveolens*). In the nesting-birds community I recorded 30 species. I found that both small-mammal and bird diversity were correlated with habitat structural diversity but on different spatial scales. Specifically, small-mammal diversity and richness were correlated with the immediate habitat structural diversity while bird diversity was not (fig 11 and 12 respectively). Also, for small mammals I found that the minimal number of individuals known alive was correlated with maquis cover, while no such correlation was

found for birds (fig 13). However, bird diversity was higher in the more structurally diverse northern part of Britania Park, while small-mammal diversity was not (fig 14); Consequently, the different response of the two taxa to different spatial scales results in no relation between birds and small-mammal diversities (fig 15).

Maquis cover in maquis plots was almost 50% higher than in the pine plots: (0.528 ± 0.0312 vs. 0.365 ± 0.045 , $t = -2.992$ $P = 0.007$). I found more expanding (as determined from 15-year records) bird species in pine vs. maquis plots (9.18 ± 0.463 vs. 7.6 ± 0.549 , $t = 2.133$ $P = 0.04$). I found more forest bird species in pine plots vs. maquis plots (2.273 ± 0.406 vs. 0.69 ± 0.307 , $t = 3.152$ $P = 0.005$), and bird species associated with shrubs and grass in maquis plots vs. pine plots (4.154 ± 0.464 vs. 2.636 ± 0.309 $t = -2.612$ $P = 0.016$). In both maquis and pines there were on average 8 habitat generalist bird species and there were 3 times more Eurasian jays (*Garrulus glandarius*) in pines than in natural maquis (mean # of jays per plot \pm SE: 3.182 ± 0.892 vs. 0.846 ± 0.004 t -test: $t = 2.657$, $P = 0.014$). Three bird species were recorded only in pine plots (Eagle owl *Bubo bubo*, Eurasian sparrow hawk *Accipiter nisus* and spotted flycatcher *Muscicapa striata*) of which one (the owl) is declining regionally. Two bird species were recorded only in maquis plots (Red backed shrike *Lanius collurio* and Common kestrel *Falco tinnunculus*), of which one (the shrike) is declining regionally. There were more house mice in pine stands vs. maquis (1.429 ± 0.528 vs. 0.167 ± 0.166 $t = 2.279$ $P = 0.056$ see also Manor et al. in press). No difference between pines vs. maquis plots was found in birds, small mammals, or structural diversities: ($t = 1.713$; $P = 0.287$; $t = -1.33$ $P = 0.101$; $t = 0.777$; $P = 0.197$ respectively). Maquis cover was higher in the southern part than the northern part of park Britania (0.556 ± 0.032 vs. 0.388 ± 0.065 $t = -2.336$ $P = 0.042$). In the northern part there were more bird species per plot with expanding populations and more generalist birds species than in the southern part (10.167 ± 0.307 vs. 7.833 ± 0.872 $t = 2.531$ $P = 0.03$; 9.667 ± 0.615 vs. 8.0 ± 0.365 $t = 2.331$ $P = 0.042$ respectively). Four bird species were recorded only in the northern part, of which two are forest species (Eurasian sparrowhawk *Accipiter nisus* and Spotted flycatcher *Muscicapa striata*), one, a shrub specialist (Rufous bush robin *Cecotrichas galactotes*) with a regionally declining population, and one generalist commensal species (House sparrow *Passer domesticus*). Two bush bird species were recorded only in the southern part of the park (Zitting cisticola *Cisticola*

juncidis and Corn bunting *Miliaria calandra*) of which the second one is declining regionally. There were no differences in # of generalist small mammals (spiny mouse and house mouse) or specialists (broad toothed mouse, shrews and Wagner's gerbils) between southern and northern parts of Britania Park (3.167 ± 1.077 vs. 3.0 ± 1.095 respectively, $P \gg 0.05$ for both t-tests). All small-mammal species were in both parts of Britania Park.

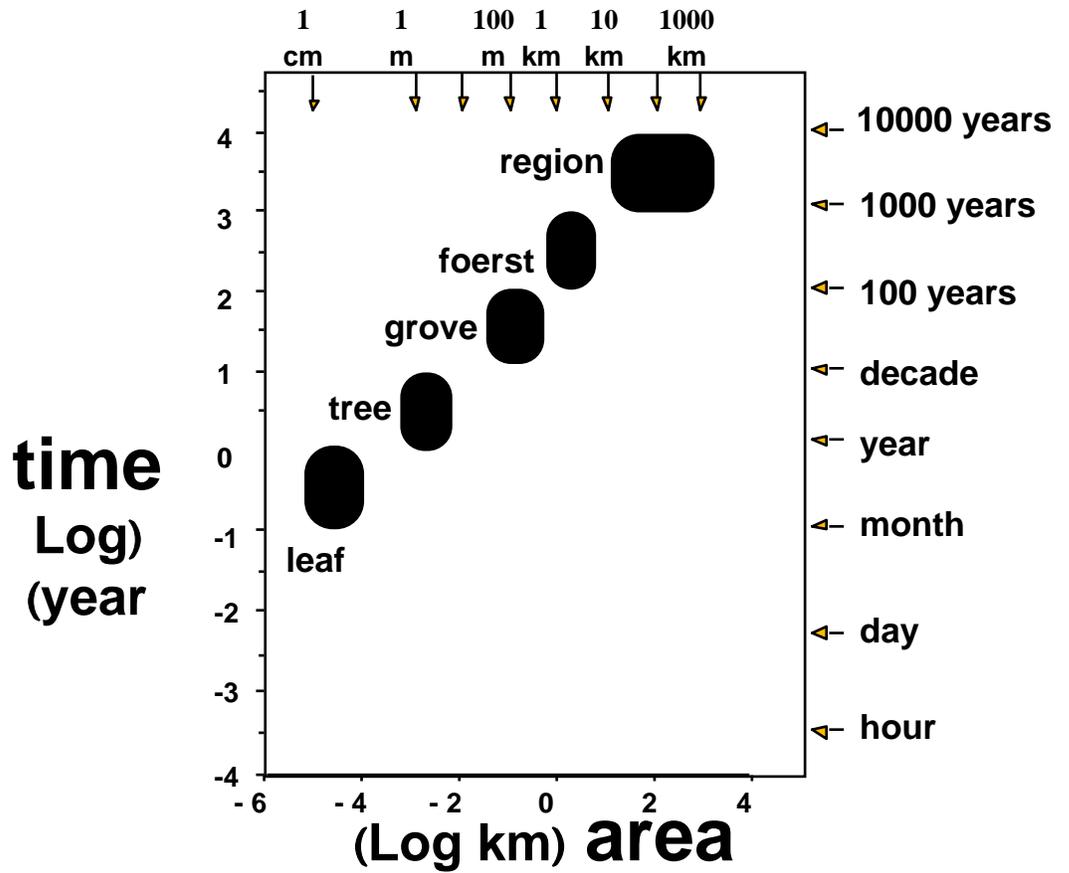


Fig 9 Different time and space scales of a hypothetical forest. According to my approach each scale deserves different indicator taxa depending on mobility and generation time. The decision which spatial and time scales should be monitored should be in accordance to conservation problem in hand.

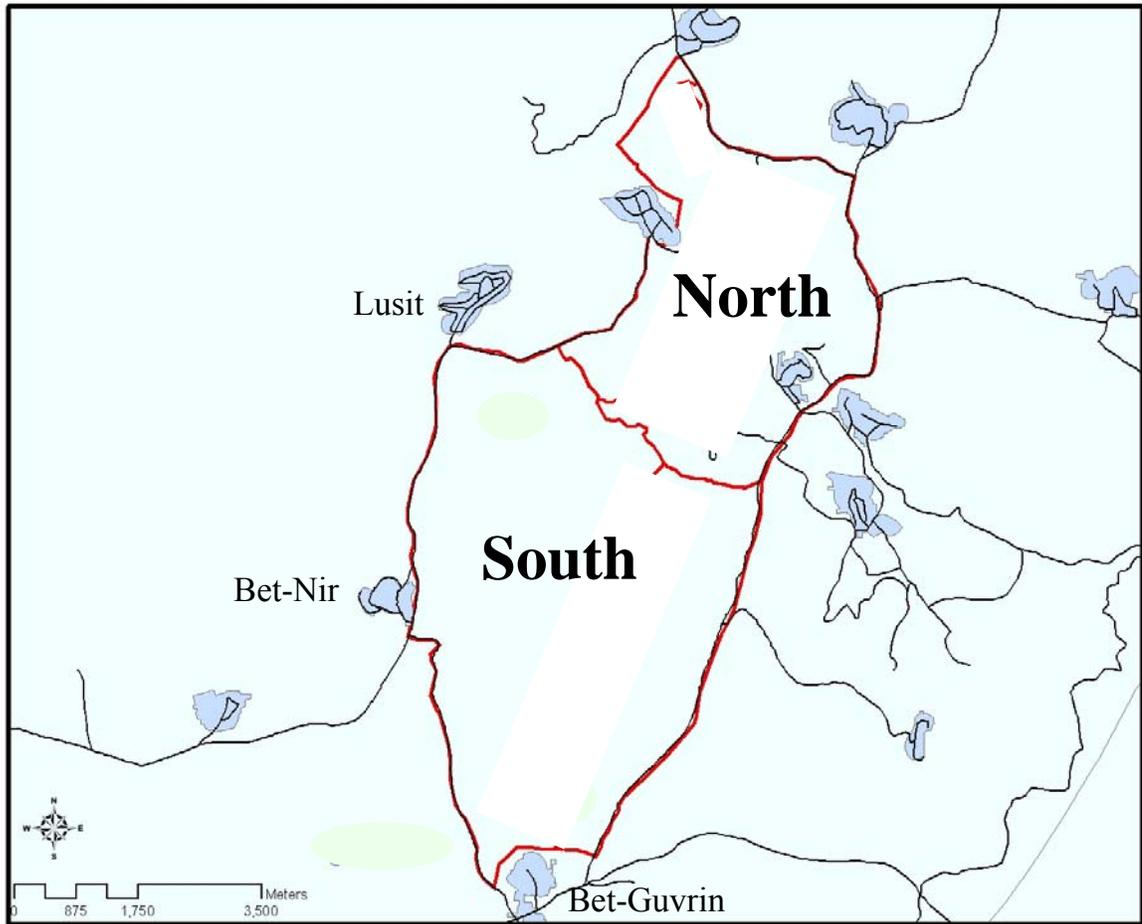


Fig 10 Map of Britania Park. The northern part is mostly a mosaic of planted pine forests along with fruit orchards, and patches of natural maquis, while the southern part is mostly natural maquis.

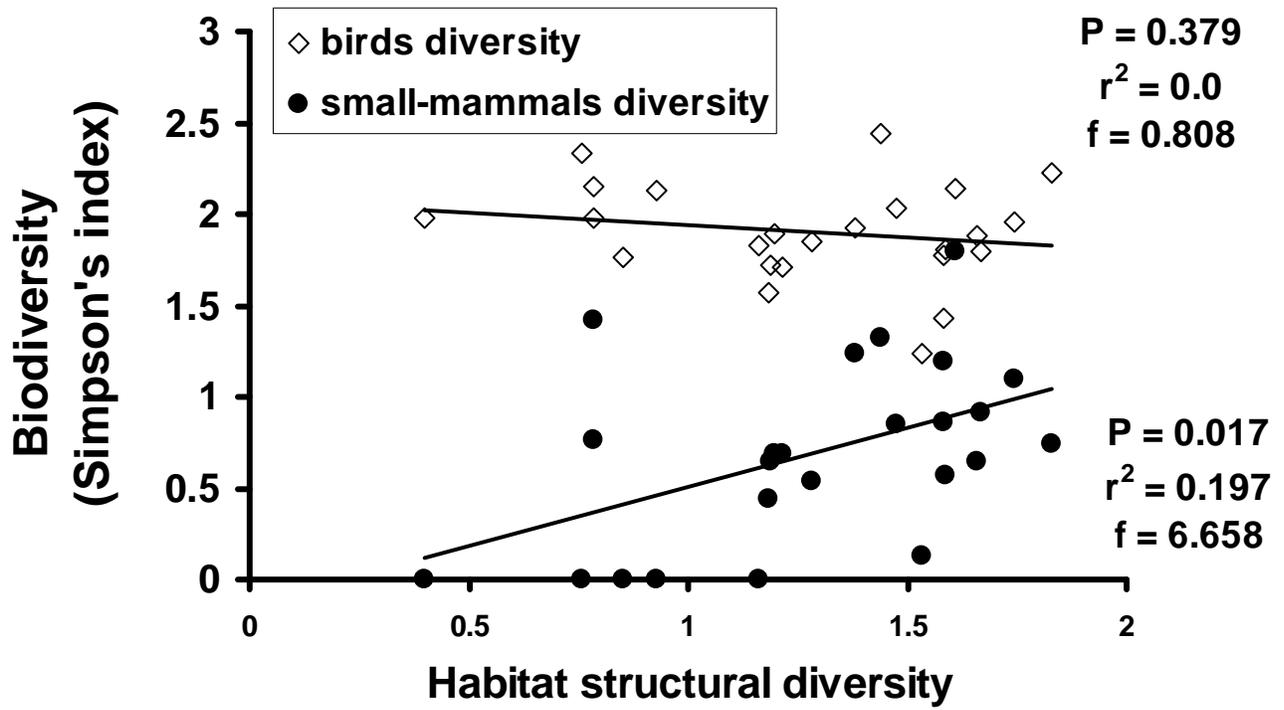


Fig 11 Bird biodiversity was not related to habitat structural diversity while small-mammal diversity was.

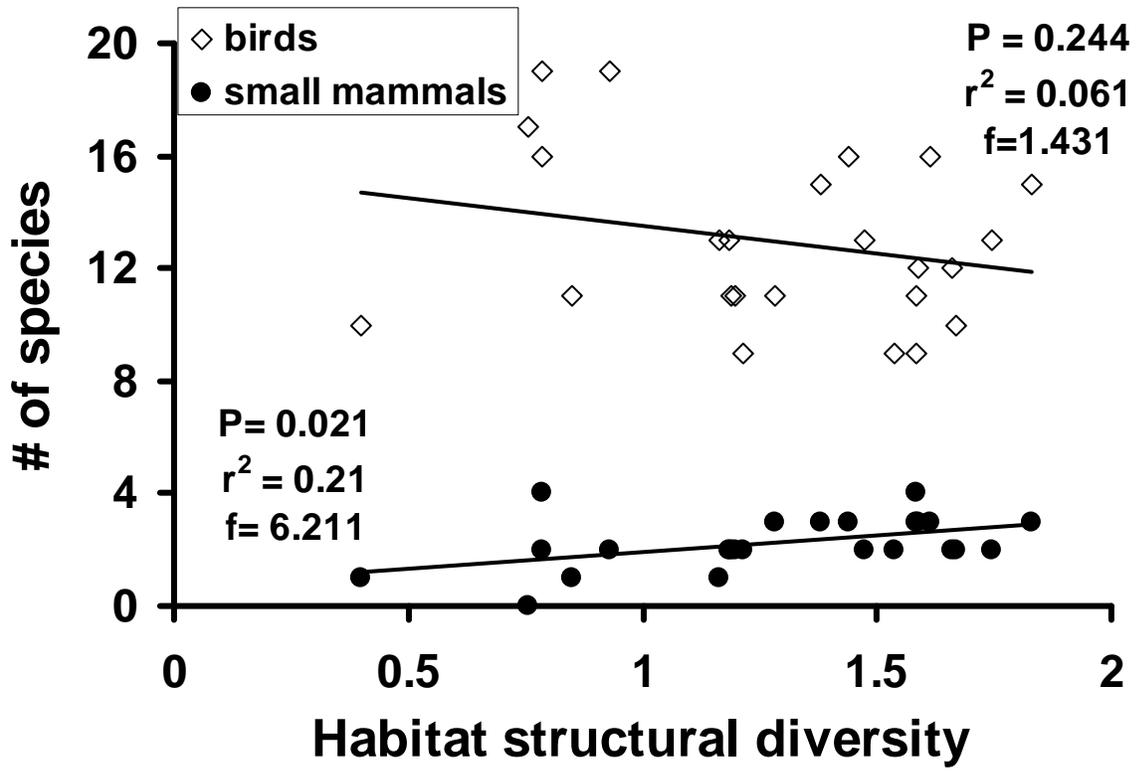


Fig 12 Bird species richness did not depend on habitat structural diversity while small-mammal species richness did.

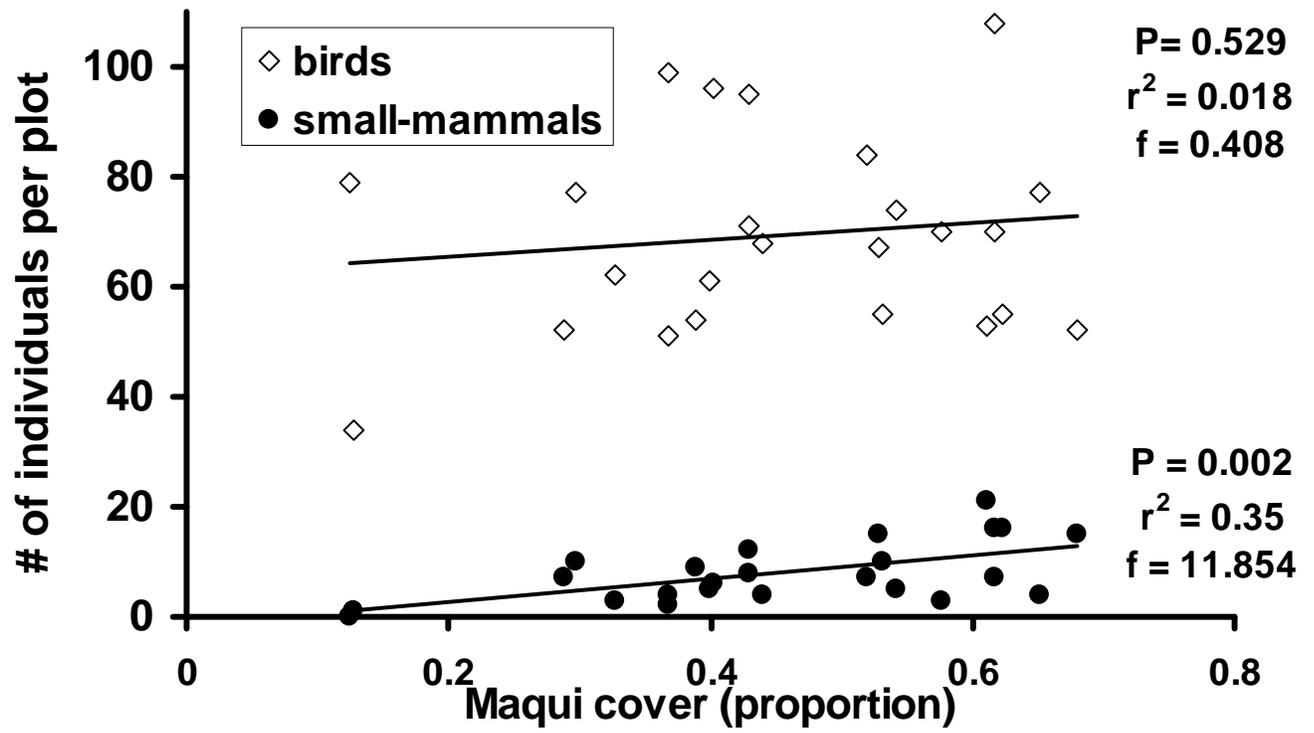


Fig 13 Bird densities were not related to maquis cover while small-mammal densities were.

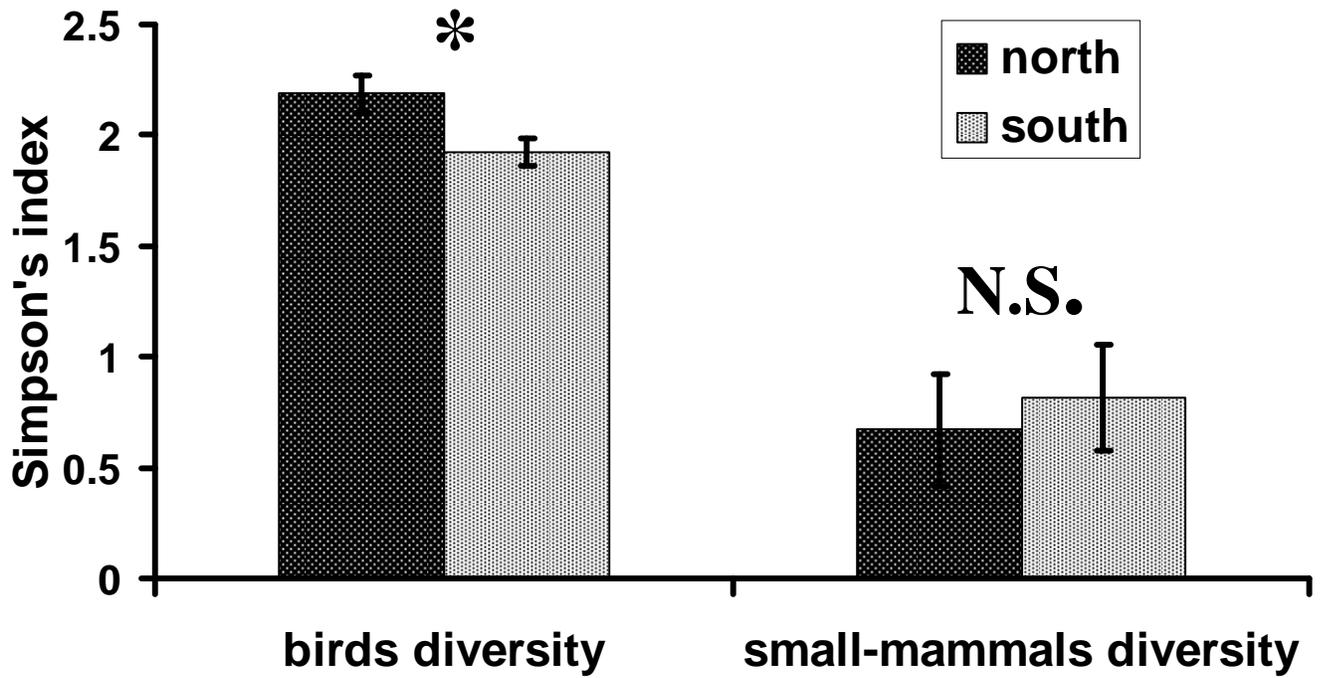


Fig 14 Bird biodiversity was higher in the north (t-test: $t=2.585$ $P=0.027$). There was no difference in small-mammal diversity (t-test: $t=-0.419$ $P=0.684$).

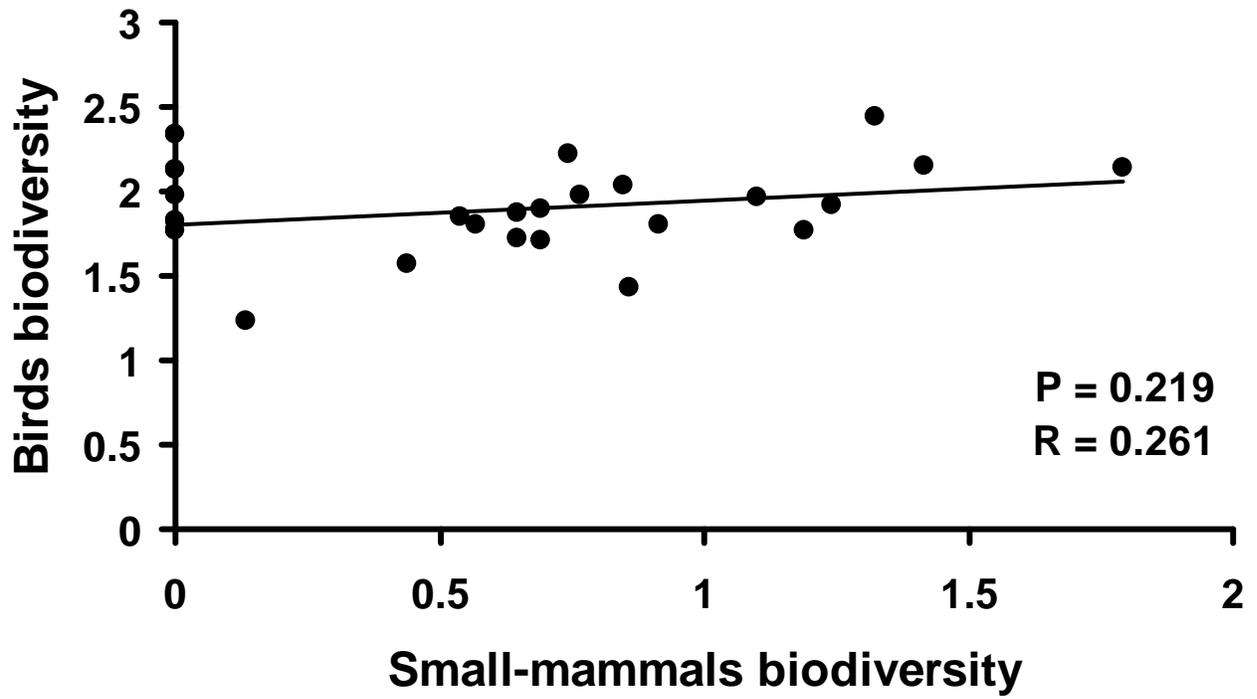


Fig 15 No significant relation between birds and small-mammal diversity was found.

4.d DISCUSSION

The current debates on most aspects of diversity regulation do not explicitly consider the relationship between the spatial scales at which phenomena (e.g. species richness) are evaluated and the spatial scales at which the hypothesized causes of these phenomena operate (Niemi & McDonald 2004). Determining the causes of variation in species diversity requires linking the scales at which variation in diversity is measured to the scales at which the processes hypothesized to affect diversity actually operate (Huston 1999). The search for drivers of local species richness across different spatial scales is of special importance in human-dominated landscapes, where current trends of fragmentation and marginalization affect both habitat quality and landscape structure. Knowing about the processes and the scales at which they operate might help to establish efficient environment schemes for the conservation of biodiversity in human-dominated landscapes (Wolters et al. 2006). Although since the 1980s the concepts of scale and biological diversity independently have received increasing attention in the scientific literature, the rate at which the two concepts have been investigated jointly (i.e. studies that consider multiple spatial scales and species groups within one ecosystem) has grown slower (Tews et al. 2004; Beever et al. 2006). As the optimal indicator of biodiversity is likely to depend on the broad goal and the target to be achieved by the areas selected for preservation, and the scale of analysis (Margules & Pressey 2000; Ferrier 2002; Niemi & McDonald 2004; Hess et al. 2006), the diversity \times scale relationship should receive closer attention in conservation.

Studies looking for correlation among different vertebrate taxa diversities, often didn't find such a correlation (e.g. Croonquist & Brooks, 1991; Oliver et al., 1998; Chase et al., 2000), maybe because the more mobile taxa can overcome isolation effects more easily resulting in responses on different spatial scales. In my study, the correlations among taxa diversity and habitat structural diversity were in accordance with the 'habitat heterogeneity hypothesis' which predicts that structurally complex habitats may provide more niches and diverse ways of exploiting the environmental resources and thus increase species diversity (MacArthur & Wilson, 1967; Bazzaz, 1975). However, these correlations differed between small mammals and birds depending on the spatial scale. Furthermore, I found no relationship between small-

mammal and bird diversity. Thus, while both groups of taxa can be used as indicators of habitat alteration by humans, they are indicative of alterations at different spatial scales; with birds being the better indicators of large scale alterations but poorer indicators of fine grained alterations. In small mammals this pattern was reversed. The fact that both avian diversity and bat species richness are higher in the more structurally diverse northern part of the park (this study and Tzoar 2004) lends support to this hypothesis. Thus, the small mammal and bird taxa compliment each other and should be used in conjunction or depending on what type of disturbance is being evaluated.

I emphasize that I do not claim that birds and rodents are the ultimate indicator taxa in all conservation studies which consider to use indicator taxa. Moreover, effects of habitat heterogeneity may vary considerably depending on what is perceived as a habitat by the species studied. Structural attributes of the vegetation that constitute habitat heterogeneity for one group may be perceived as habitat fragmentation by another taxonomic group (e.g. Okland, 1996). Thus, the selection of indicator taxa should be case specific and according to the array of taxa in the given habitat (aquatic or terrestrial) and the spatial scale that should be investigated. In general, it appears that the 'operational scale' of each taxa depends on its movement capabilities. Therefore taxa with limited movement capabilities (such as small mammals) should be selected in studies focusing on processes at a small spatial scale (i.e. grove scale in fig 9), while taxa with better mobility (such as birds) should be selected when focusing on processes at bigger spatial scales (i.e. forest scale in fig 9). For example, Brose (2003) demonstrated that the effects of habitat heterogeneity for ground beetle assemblages were positive on the micro- and meso-scale (0.25 and 500–1000 m², respectively), while they were non-significant on a macro-scale of 10 km² (Brose, 2003). On the other hand, bigger mammals or big birds of prey with vast home ranges may be more appropriate as indicator taxa of vast areas. However any taxa with the appropriate operational spatial scale can be used with the only limitation that it should be with well known biology.

Small mammals appear to be more sensitive to habitat homogenization and thus may reflect earlier stages of human impact. Indeed, different species in an ecosystem can be differently affected by spatial structures, depending on whether these structures cause

heterogeneity or fragmentation from the organism's perspective (André'n, 1994). Evidently, patches of pine and carob trees increased habitat heterogeneity for birds, but not for small mammals. Furthermore, planted pine and carob forests are often poor habitat for indigenous small mammals (first part), which might suggest that they induce fragmentation in small mammals, but not in birds. This scale-dependence is probably caused by the species operational scales of their home ranges and dispersal and movement capabilities. Generally, the type of correlation between species diversity and habitat heterogeneity will vary according to the species group and the spatial scale where habitat heterogeneity is measured (Tews et al. 2004).

In agreement with theory (Kotliar & Wiens 1990; Whittaker et al. 2001; Olff & Ritchie 2002), I demonstrated that differences in local species diversity may be caused by ecological processes acting at different spatial scales, and species differing in mobility can be regulated by different processes at the same spatial scales. Indeed, at scales smaller than regional species, composition is heavily influenced by dispersal (Rosenzweig & Ziv 1999). Evidently, mobility (dispersal ability) is affecting the scale at which different taxa respond to environmental diversity. Thus, diversity of taxa with significantly different mobility shouldn't be correlated at scales smaller than regional, because mechanisms such as interspecific interactions or dispersal limitations are likely more important at the local level, whereas mechanisms related to speciation and extinction are more important at the regional level (Ricklefs & Schluter 1993). Although richness at the local and the regional scale is interdependent (Ricklefs & Schluter 1993; Cueto & de Casenave 1999), these scales represent distinct levels of biological organization (e.g., reflecting ecological mechanisms that operate within communities versus ecological and historical mechanisms that effect turnover among communities in a region). Accordingly, the importance of particular structuring mechanisms likely differs between such levels of organization (Willig et al. 2003).

Despite growing concern, no consensus has emerged over the effects of habitat modification on species diversity. Even for comparatively well-studied taxa, disturbance has been reported to increase and decrease diversities with approximately equal frequency (e.g. Hill & Hamer 2004). At times, the only clear change in community composition is the replacement of specialist species that were extirpated due to habitat alteration, by generalists. This is, in my

opinion, *the* reason that it is so important to sample taxa composed of species with well known biology (such as small mammals and birds). Field studies have shown that generalist species are more likely to find human-altered portions of a landscape than specialists (Laurance 1991; Andr n 1994; Laurance 1995; Sarre et al. 1995; Gascon et al. 1999; Bentley et al. 2000). The niche-breadth hypothesis thus predicts that a species with the ability to use a broad array of habitats and food types (as determined from studies across its geographical range) should be less adversely affected by fragmentation of native habitat than a more specialist species (Swihart et al. 2003). My study provides a good example: I did not find any difference between diversities of both small mammal and bird communities between maquis vs. pine plots. This is because I chose plots that represented all spectrum of maquis cover within each habitat type, including extremely disturbed plots both in pines and maquis. However, when looking at the composition of both small mammal and bird communities I could identify that generalist species were much more common within pine-dominated habitats. In terms of species richness and diversity (as determined by the Simpson's index) I found no difference between planted plots and the more natural maquis plots, but still I could identify the better habitats for indigenous specialist species. Furthermore, if only local species richness and diversity were considered, than the ecosystem of northern part of the park, that consisted mostly of planted pine forests with patches of fruit orchards and small maquis patches, would appear preferable to the continuous natural maquis because of higher biodiversity. However, when the composition of bird assemblages is accounted for, it is evident that the higher diversity scores in the northern park were due to generalist species and species that expanded their range (the house mouse – first part, and Eurasian jay – this part) rather than specialist or sensitive species. It was best if I could conduct a separate analysis of invasive vs. native species in each taxa. However because of very small number of invasive bird species and individuals I could not conduct an analysis similar to the small mammal study presented in part 1.

Conservation of biodiversity at multiple levels of biological organization and spatial scales is complex and requires both explicit identification and protection of the focal ecosystems and species in a given area, and adequate identification and protection of the associated multiscale ecological processes that support and sustain those ecosystems and species (Pickett et al. 1992,

Meyer 1997). For purposes of biodiversity conservation, functionality or integrity of a conservation area can perhaps best be judged by the extent to which the composition and structure of the focal ecosystems and species are within their natural ranges of variability (Poiani et al. 2000). Thus, scale dependency is a factor causing spatial variability in correlation strength. Such dependence is connected to the biology and mobility of the organisms, but is rarely taken into consideration. I believe that a different approach should be adopted when trying to estimate ecosystem biodiversity with indicator taxa. Specifically, a set of indicator taxa should be considered to measure diversities at different scales from local to regional, depending on the scales of the conservation measure in hand. The appropriate scale for each taxa can be determined in a species-specific manner, preferably based on local studies on space use patterns, or on existing literature on well-studied taxa, and lastly on allometric equations if empirical data and funding are limited.

5. GENERAL DISCUSSION

By addressing a specific human impact, e.g. afforestation, as an environmental experiment I could assess the impact of human landscape modification on the local vegetation, small-mammals and birds communities, and possibly other taxa with similar mobility (or dispersal abilities) if we regard to the above taxa as ecological indicators. There was nothing new in each of the ingredients of my study (identifying human impact and monitoring the ecological entities). This is usually what conservationists do. The innovation here was, in my opinion, the focus on the 'basic details' such as species characteristics (i.e. generalists vs. specialists), and monitoring them in the appropriate spatial scales from the organism point of view. Although the biodiversity of plant communities has been shown to be dependent determined by the nature of interspecific competition (e.g. Tilman 1976; 2004), as far as I know, no one has studied competitive interactions in mobile animals to gain conservation insights. I think this approach is a very useful and effective conservation tool also when investigating mobile taxa even if it requires controlled (enclosure) experiments to validate species interactions. Moreover, such an approach of looking at species characteristics and interactions could be applied to any ecosystem (aquatic or terrestrial) with the only limitation being that the biology of the studied species is well known biology. By using this approach we can get direct indications as to the condition of the ecosystem and, more importantly, we can produce practical management plans to improve ecosystem performance.

Specifically, my study demonstrated how habitat homogenization by afforestation has a considerable impact on the indigenous vegetation, small mammals, and bird communities favoring species with flexible requirements from the environment (i.e. generalists), and hindering the continued existence of the more specialists species. However, this process could be reversed if we manage the man-made forests properly, increasing environmental structural diversity by planting indigenous trees and avoiding the creation of large homogenous patches of exotic species (as carobs and acacia) or exotic sub-species (pines). This in turn, will increase vegetation diversity and will enable the established of diverse communities of all other organisms making the ecosystem more stable and less susceptible to invading species, without opposing of the important rule of those stands as green ecotourism attractions.

Although, Aleppo pine, *Pinus halepensis* are no doubt part of the native components of the Mediterranean maquis, dense pine forests were not part of the landscape of Israel's mountains as they are today. According to an interdisciplinary research project, using botanical, historical and geographical evidence, including Palynological investigations, it can be assumed quite safely that Aleppo pine was rare in the native arboreal vegetation of Israel in the past (Liphschitz and Biger 2001). This is in agreement with my study and thus the optimal Mediterranean pine forests would be, from conservation point of view, with as lower as possible of pines canopy cover that would be replaced (through succession processes) by indigenous plants.

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אקולוגיה של חברות ביערות נטועים: השלכות לשמירת טבע

מחקר לשם מילוי חלקי של הדרישות לקבלת תואר "דוקטור לפילוסופיה"

מאת

רגב מנור

הוגש לסינאט אוניברסיטת בן גוריון בנגב

2009

תשס"ט

באר שבע

באר שבע

העבודה נעשתה בהדרכת

פרופסור דיויד זלץ

במחלקה לאקולוגיה מדברית

ע"ש מיטרני

בפקולטה למדעי הטבע

אקולוגיה של חברות ביערות נטועים: השלכות לשמירת טבע

תקציר

המוטיבציה שעמדה מאחורי המחקר שלי הייתה להראות שאנו כבר יכולים להתבסס על התיאוריות האקולוגיות הקיימות (בעיקר אקולוגיה של חברות), בעת איסוף נתונים הקשורים לאינטראקציות אקולוגיות, במטרה להעריך השפעות אדם ולפתח כלי ממשק יישומיים כדי להפחית את אותן השפעות בבית גידול נתון. ספציפית, התזה כוללת 3 נושאים הקשורים זה לזה:

1. השלכות של הומוגניזציה על החברה האקולוגית. דוגמה אחת להומוגניזציה נופית היא יערות נטועים. חקרתי את השפעת יערות נטע-אדם ע"י השוואה של הרכב חברות צומח ויונקים קטנים ביערות נטועים ובתי גידול טבעיים הסמוכים להם בשני בתי גידול בחבל הים תיכוני בישראל: (1) בית גידול ים תיכוני טיפוסי בו התמקדתי בחלקות של אורן ירושלים (*Pinus halepensis*) וחרוב מצוי (*Ceratonia siliqua*) ו (2) בית גידול חולי של חוף הים בו התמקדתי בחלקות של שיטה כחלחלה (*Acacia saligna*). ראשית, רציתי לבדוק אם וכיצד העצים הנטועים משפיעים על מבנה חברת הצומח, ושנית לראות כיצד חברת היונקים הקטנים מושפעת מתנאים סביבתיים שונים השוררים בחלקות יער עם כיסוי צמרות שונה. התעניינתי במיוחד בתפוצת עכבר הבית (*Mus musculus*) שהוא מין מלווה אדם. בכל סוגי היערות שנבדקו הייתה הומוגניזציה ביוטית שהתבטאה בצפיפות עכבר הבית בנוסף למגוון מינים נמוך יותר וצפיפויות נמוכות יותר של צומח ויונקים טבעיים. מגוון מבני של בית הגידול היה ביחס ישר למגוון מיני היונקים הקטנים והיה נמוך יותר ביערות נטע האדם. תוצאות אלו מראות כי השימוש בצפיפות מינים מלווי אדם כדוגמת עכבר הבית ביחס לשאר המינים הטבעיים הוא גישה המאפשרת לקבוע את מצב המערכת האקולוגית. דילול חלקי של העצים הנטועים הוא כלי ממשקי אפשרי לשמר כיסוי מספיק של עצי חורש טבעי בתוך יערות נטע האדם. אולם, עד שייאסף מספיק מידע, מידת הדילול הנדרשת למנוע את מגמת הומוגניזציה ביערות נטע אדם ושמירה על חברת היונקים הקטנים הטבעיים מגוונת ופחות חשופה לפלישה ביולוגית צריכה להיקבע באופן אמפירי.

2. השפעת האדם על בחירה תלוית צפיפות של בית גידול. בחירה תלוית צפיפות של בית הגידול נחשבת כגורם עיקרי למגוון המינים המקומי, והידע של תהליכים אלו ואיך הם מושפעים מפעילויות אדם הינו רלוונטי במיוחד לשמירת טבע. ע"י מחקר של היחסים התחרותיים בין מין סתגלתן ומין מתמחה ניתן לקבל תובנות כיצד גורמים סביבתיים שונים קובעים את צפיפויות ופיזור של המינים וכתוצאה מכך גם על מגוון המינים. בחלק זה אני מתאר מחקר של תהליכים תלויי צפיפות הקובעים את מבנה החברה של שני מכרסמים: מין מתמחה – יערון גדול (*Apodemus mystacinus*) ומין סתגלתן – קוצן מצוי (*Acomys cahirinus*) בבית גידול של חורש ים תיכוני, וכיצד מבנה החברה משתנה ביערות נטע אדם. ערכתי שני מחקרים: הראשון, המסתמך על לכידות בשטח המחקר, בחן כיצד מבנה החברה ועצמת התחרות משתנה

עם השינוי במבנה בית הגידול. המחקר השני היה במכלאה ומטרתו הייתה לאשר את העדפות בית הגידול והיחסים התחרותיים בין המין המתמחה והמין הסתגלתן. זיהיתי יחסים תחרותיים לא סימטריים בהם המין המתמחה היה דומיננטי למין הסתגלתן. עצמת התחרות הייתה גבוהה יותר בחלקות עם כיסוי אלונים הנמוך מ- 10% (רובן בתוך יער אורנים), למרות שצפיפויות שני המינים בחלקות אלו היו נמוכות. תחרות נמצאה רק בתקופת החוסר (קיץ). בהתבסס על נתונים אלו, ניתן לתת המלצות ממשקיות לשמירה על מגוון היונקים הקטנים גבוה. לתהליכים תלויי צפיפות תפקיד חשוב בקביעת מגוון המינים, והבנת המנגנונים המשפיעים על תהליכים אלו הכרחית אם ברצוננו להבין כיצד הפרעות אדם משפיעות על מגוון המינים.

3. חשיבות הסקלה בבחירת קבוצה טקסונומית כקבוצת אינדיקטור. מידע אמפירי חוזר ומראה שעושר מינים של קבוצה טקסונומית אחת לא משקף את עושר המינים של קבוצה טקסונומית אחרת באזור נתון. לכן, החיפוש אחר קבוצת אינדיקטור למגוון מינים או להפרעות אדם בדרך כלל נכשל. בפרק זה השוויתי בית גידול המושפע ע"י אדם (יער נטע אדם) עם בית גידול טבעי של חורש ים תיכוני ובחנתי את ההתאמה של יונקים קטנים ועופות כקבוצות אינדיקטור. חקרתי אם וכיצד מגוון המינים והרכב המינים של קבוצות השונות יכולים לשמש כמדדים ברי שימוש למידת השפעת האדם ע"י שילוב האינפורמציה של קבוצות שונות בסקלות מרחביות שונות. הנחתי כי הסיבה לחוסר התאמה בין מגוון המינים של יונקים קטנים ועופות הוא בגלל יכולת הניידות השונה. ספציפית, המינים של הקבוצה הטקסונומית הניידת יותר יכולים להתגבר על אפקט הבידוד בקלות יתרה, דבר המביא לתגובות להשפעת האדם בסקלות מרחביות שונות. בתוך הקבוצה הטקסונומית ציפיתי שהמינים הסתגלתנים יושפעו פחות משינוי בית הגידול לעומת מינים מתמחים. ע"י שימוש בחתכי ציפורים, לכידות של יונקים קטנים וחתכי צומח מצאתי כי גם היונקים הקטנים וגם העופות הושפעו מהמגוון המבני של בית הגידול אך בסקלות מרחביות שונות. לכן למרות ששתי הקבוצות הטקסונומיות יכולות לשמש כקבוצות אינדיקטור לשינוי הסביבה בידי אדם, הם מודדות שינויים בסקלות מרחביות שונות (מאות לעומת אלפי מ"ר - יונקים קטנים לעומת עופות, בהתאמה). בנוסף, המינים הסתגלתנים עכבר הבית ועורבני שחור כיפה (*Garrulus glandarius*) היו נפוצים הרבה יותר ביער נטע האדם לעומת החורש הים תיכוני. מכאן, אני מאמין כי צריך לנקוט בגישה חדשה כאשר מנסים להעריך את מגוון המערכת האקולוגית ע"י קבוצות אינדיקטור. ספציפית, כתלות בסקלה המרחבית הרצויה, יש להשתמש בסדרה של קבוצות אינדיקטור של מינים עם יכולת ניידות שונה ועם ביולוגיה ידועה היטב, בשילוב עם צפיפויות המינים הסתגלתנים, בכדי לקבל אינדיקציה שימושית ורלוונטית ליישומי שמירת טבע.