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# Ecological niche modeling under climate change to select shrubs for ecological restoration in Central Mexico

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ABSTRACT

Shrub species were selected for potential use in restoration projects in the semiarid shrublands of Central Mexico. Ecological characteristics of the species were considered, including tolerance to climate change. Inventories of shrubs were carried out in 17 semiarid shrubland fragments of xeric shrubland. The 46 species recorded were ordered using a principal component analysis, considering ecological characteristics such as frequency, land cover, sociability and interaction with mycorrhizal fungi. From these, the 10 species that presented the highest values of the desired characteristics were selected. The response of these species to climate change was evaluated using current potential distribution models and by applying climate change scenario A2, using MaxEnt. The species that presented suitable ecological qualities for restoration and maintained or increased their distribution under the climate change scenario were *Acacia schaffneri*,*Ageratina espinosarum*, *Bursera fagaroides*, *Dalea bicolor*, *Eysenhardtia polystachya* and *Karwinskia humboldtiana*. These species are therefore recommended for use in medium and long-term ecological restoration projects in the semi-arid region in Central Mexico.


## 1. Introduction

The arid and semiarid areas of the world occupy approximately one-third of the land surface of the earth (Aronson et al., 2002). Degradation of these environments is ongoing at an alarming rate, threatening the livelihood of millions of people (Ffolliott et al., 1995). In Mexico, semi-arid zones account for approximately 60% of the national territory (Challenger, 1998). The predominant vegetation types in these environments are xeric shrublands with different plant associations, which have high endemism (Rzedowski, 1998). Semiarid shrublands in Central Mexico are highly fragmented and degraded due to changes in land use and cover, resulting mainly from agricultural activities that have modified the biogeochemical cycles and caused species loss (Challenger, 1998; Montagnini et al., 2008). Furthermore, semiarid ecosystems are of low resilience because the lack of water slows recovery from disturbance (Maestre, 2003; Miranda et al., 2004). These situations justify the implementation of ecological restoration projects for the recovery and conservation of ecosystem services.

Projects focused on the restoration of degraded ecosystems in semiarid areas have used shrub species with mycorrhizal associations (Allen et al., 2005; Carrillo-García et al., 1999; Corkidi and Rincón, 1997; Monroy-Ata et al., 2007) and nurse plants (Carrillo-García et al., 1999; Castro et al., 2002, 2004); however, little consideration has been given to the use of species with ecological characteristics such as sociability, which can facilitate the establishment of species in advanced successional stages (Badano et al., 2006; Gutiérrez and Squeo, 2004; Hillebrand et al., 2008). In other semiarid regions species have been selected to restore degraded ecosystems based on their ample coverage and adaptation to disturbances such as fire and herbivory (Cortina et al., 2004). Furthermore, selecting species for ecological restoration should incorporate the ability of plants to survive alterations in temperature and precipitation regimes brought about by climate change (Choi et al., 2008; Gastón and

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García-Viñas, 2013; Harris et al., 2006; Maschinski and Hanskins, 2012). These criteria can suggest tools land managers can use to deal with the changes to ecosystems caused by climate change.

The effects of climate change on germination, establishment, growth, phenology, reproduction and mortality have been documented in some plant species, and can cause both decreases and increases in areas of distribution (Allen et al., 2010; Gitlin et al., 2006; Lavergne et al., 2010; Michaelian et al., 2011; Walther et al., 2002). Climate change can also cause alterations to the structure and composition of communities (Clewell and Aronson, 2007; Choi et al., 2008; Harris et al., 2006; Rice and Emery, 2003), causing changes in the distribution area of terrestrial biomes (Rehfeldt et al., 2012). Arid climate areas are expected to increase in Mexico at the expense of climates suitable for coniferous forests, semi-deciduous forests and cloud forests. In particular, the Chihuahuan Desert is expected to increase in area by 31% by 2030 (Rehfeldt et al., 2012).

To evaluate the effect of climate change on restoration projects, it is necessary to implement monitoring of plant survival and replacement in the successional trajectory over extended periods of time. However, because of short and medium-term needs, potential distributional models have been used as a tool to indirectly evaluate the effect of climate change. These models provide a habitat and space descriptor for species that can be applied to future climate change scenarios (Lavergne et al., 2010; Pearson and Dawson, 2003). Potential distribution models take into account information regarding habitat requirements and known sites of occurrence, as well as climate and topographic variables (Guisan and Zimmerman, 2000; Phillips et al., 2006). These models have the capacity to predict potential areas of species presence under current and future environmental conditions (Lavergne et al., 2010; Peterson et al., 2011) and have been considered an essential tool for the management and conservation of biodiversity (Cote and Reynolds, 2002; Cote and Reynolds, 2002).

Given the degradation processes semiarid environments are undergoing in Central Mexico, it is necessary to initiate ecological restoration programs. In these programs, land managers (restorers) take into account the future distribution of climate or the probability that future climates could be new. Therefore, a preliminary step for the establishment of an ecological restoration project is the appropriate selection of species, for which it is necessary to take into account ecological attributes and responses to climate change. The present study is a methodological proposal for selecting species for use in ecological restoration under climate change scenarios and is one of the first efforts to evaluate the possible future consequences of climate change on the distribution of potential restoration species. This mainly considers that current global warming is causing accelerated changes in climate (IPCC, 2007). These changes have been observed to have a marked influence on the expansion and contraction of the ranges of biomes and species (Gian-Reto et al., 2002; Hughes, 2000; McCarthy et al., 2001; Rehfeldt et al., 2012).

The objective of our study was to select shrub species with ecological attributes (coverage, density, frequency, association coefficient and presence of mycorrhizal associations) that would enable the rapid recovery of disturbed semi-arid ecosystems. Moreover, the proposed species should maintain or increase their potential distribution under climate change scenarios within the region, so that that they would be present at the site for sufficient time to have a tangible effect on recovery.

## 2. Materials and methods

### 2.1. Study Area

The area of interest is in the semiarid shrubland of Central Mexico (19°50′–20°40′ N and 98°35′–99°25′ W). The area has an annual mean temperature between 15 and 19 °C, and annual precipitation of less of 450–700 mm (Garcia, 1987; Pavón and Meza-Sánchez, 2009). Semiarid shrubland is characterized by species including *Acacia farnesiana*, *Celtis pallida*, *Cordia boissieri*, *Dalea bicolor*, *Forestiera angustifolia*, *Karwinskia humboldtiana, Lantana involucrata* and *Montanoa tomentosa*, among others (Puig, 1991).

Inventories of the shrub species were carried out in 17 fragments of semiarid shrubland in the state of Hidalgo. In each fragment, two transects were established, each 5 m × 50 m divided into 5 × 5 $m^2$, in which abundance and coverage of shrub species were recorded.

### 2.2. Species discrimination

Coverage, density, frequency, and sociability, and the presence of mycorrhizal associations were estimated for each shrub species. These ecological variables were considered to be the ones most important to have high values for indicating that a species was suitable for use in restoration of arid tropical scrub. Sociability of each species was estimated by an association coefficient (Verlaine, 2001). The cover of each species is the horizontal projection of the aerial parts of the individuals on the ground and is expressed as percentage of the total area. The frequency was calculated by recording the presence or absence of each species at each sampling site (Mostacedo and Fredericksen, 2000). The density was expressed as the number of individuals on the total sampled surface. Species were considered to present mycorrhizal association only if a previous study had reported such an association (Bloss and Walker, 1987; Camargo-Ricalde et al., 2003; Gonzalez-Chavez et al., 2008; Montaño-Arias et al., 2008), although this does not necessarily mean that other species did not also have a mycorrhizal association.

Using principal component analysis (PCA), species were ordered according to the magnitude of the desired ecological attributes (Johnson, 1998).

The purpose of using a PCA was to generate only two variables (principal components) that would explain the most variance. The species would be distinguished mainly on the first axis in a gradient from low (negative) to high (positive) values (Gauch, 1982). Since the variance explained by the first axis was high (54%), we consider that this objective was fulfilled, and therefore selected the species that appeared at the positive end of the ordination axis; those which have a combination of high values for the five variables used. Before applying the PCA, we combined the data obtained from samples taken at the 17 sites. This is because not all species were present at all sites, so the complete matrix had too many zeros. Because of this, we did not have replicates for all 46 species of shrubs. The PCA was performed using the PC-ORD 2.0 statistical package.

### 2.3. Ecological niche models

The results of the PCA enabled us to select the group of species that presented most of the desired attributes. These species were then used to produce ecological niche models projected to the year 2050. In order to construct the models, we used a maximum entropy algorithm (MaxEnt). This algorithm calculates the most probable potential geographic distribution of a species, which is produced mainly from the relationship between the geographic data and the known geographic distribution for the species (Elith et al., 2011). A detailed technical explanation of MaxEnt can be found in Phillips et al. (2006). Before the model was applied, a database was generated of all sites in Mexico where specimens have been collected of any species recorded in the sampling conducted in the present study. The database included the geographical position in both Cartesian and Mercator (UTM) coordinates (georeferences) of each of these sampling sites. To generate the models, we used the georeferences, and environmental and topographic variables. Data were obtained from the field samples collected in this study and the following databases; Global Biodiversity Information Facility (GBIF), the World Information Network on Biodiversity (REMIB), "Unidad Informática para la Biodiversidad" (UNIBIO), Missouri Botanical Garden, and the New York Botanical Garden. We only considered data obtained from the GBIF, REMIB and UNIBIO for which georeferences could be corroborated with data present in this locality.

The georeferences were corroborated by comparing them to data from the Mexican National Institute of Statistics, Geography and Informatics (INEGI). This was because it is common to find some erroneous georeferences in the databases that mistakenly point to locations in the ocean, other bodies of water, or urban areas.

A 1 km × 1 km grid was generated, onto which sampling localities were projected. Some sites had more georeferences than others; however, in order to avoid the model being biased towards areas where sampling may have been more intense, only one record was included per 1 $km^2$. Species with fewer than 10 georeferences in all the study zones taken together were not considered in the analysis, because prior studies have demonstrated that using fewer will not be meaningful without extensive habitat requirement data and defining climate envelope constraints (Stockwell and Peterson, 2002).

Climate layers (19) were obtained from Worldclim (www.worldclim.org) to represent current climate, and are the result of interpolating on global temperature and precipitation data from 1950 to 2000 at a spatial resolution of 1 $km^2$ (Hijmans et al., 2005). These variables are derived from monthly temperature (maximum and minimum) and rainfall data. They express seasonality, annual trends and extreme conditions (Mendoza-González et al., 2013). A special report on Emission Scenarios (SRES) was used to evaluate climate change and to create the models. The SRES selected for this study was the A2 scenario, given that our main interest was to establish potential distribution under the most adverse conditions. This scenario foresees a temperature increase of between 3.6 and 5.6 °C, and annual variation in precipitation between +5% and −10% (Arnell et al., 2004; IPCC, 2000). These future climate variables correspond to the HADCM3 (The Hadley Centre for Climate Prediction and Research) and CGCM2 (Canadian Centre for Climate Modeling and Analysis) general circulation models, both at a resolution of 1 $km^2$ (Flato et al., 2000). Also, 3 topographic layers were also included in the model: (1) slope, (2) aspect, and (3) topographic index (Gesch and Larson, 1996).

The resulting models were obtained in a logistic format, since this is robust when occurrence is unknown and biological interpretation is easy, mainly because it assumes that the estimated probability for a species is based on the restrictions imposed by climate variables (Phillips and Dudik, 2008). For each species, 70% of the presence data was used to generate the model, and the remaining 30% to evaluate it. To assess the prediction, MaxEnt was used to calculate the AUC (area under the curve) values, which are used to characterize model performance. The classification values range from 0 to 1, where a value of 1 represents a perfect adjustment (Feria et al., 2010). Binary maps of the presence/absence of species were then generated, with a consensus map combining the two general circulation models for 2050. All models were generated for the whole of Mexico, and subsequently clipped to the area of interest. Within this area, comparisons were made between the percentages of area gained, lost and conserved for potential distributions under climate change scenarios relative to the contemporary potential distribution map.

## 3. Results

### 3.1. Species selection

A total richness of 46 species was recorded, distributed in 41 genera and 21 families. The most represented families were Asteraceae and Fabaceae, with 10 genera each. The PCA enabled us to distinguish between two different groups (A and B) of species (Fig. 1) that had the highest values of the desired ecological attributes.

Only ecological niche models with species in these two groups were applied. However, the group A species were clustered closer to the positive end of the first axis, which was positively correlated with frequency, density and coverage, with correlation coefficients

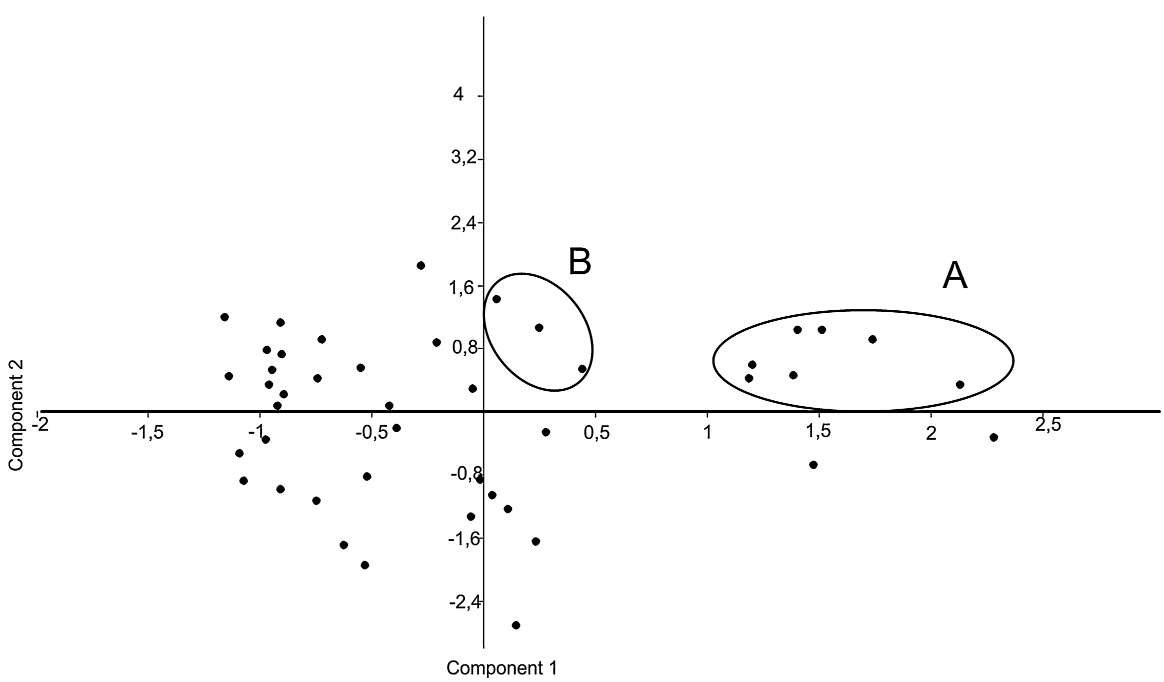


**Fig. 1.** Ordination of 46 shrub species by the first two components in PCA. Two groups of species were selected; group A, formed by 7 species, and group B defined by 3 species.

of 0.95, 0.89, and 0.78, respectively. Group A consisted of *Acacia schaffneri* (S. Watson) F. J. Herm, *Ageratina espinosarum* (A. Gray) R.M. King and H. Rob, *Bursera fagaroides* (Kunth) Engl, *Dalea bicolor* Humb. and Bonpl. Ex Willd, *Eysenhardtia polystachya* (Ortega) Sarg, *Karwinskia humboldtiana* (willd. Ex Roem. and Schult.) Zucc and *Montanoa tomentosa* Cerv., and group B consisted of *Gymnosperma glutinosa* Less, *Flourensia resinosa* (Brandegee) S.F. Blake and *Mimosa pringlei* S. Watson. Together, the first 2 components identified by the PCA explained 77.9% of total variation. The first component explained 54.8% and the second component explained 21.1% of total variation. The association coefficient and the presence of mycorrhizal associations had the highest correlation coefficients, 0.97 and 0.77, respectively. In general, species of both groups A and B had the highest importance values and can be considered the dominant species of the desert scrub community in the study area.

### 3.2. Model results

Current distributional models and those under climate change were produced with species in groups A and B only, excluding *G. glutinosa* and *M. pringlei*, because these two species have fewer than 10 total georeferences. The eight remaining species were modeled, resulting in AUC values above 0.83. For each of these species, between 40 and 200 occurrence points were obtained. Models included in the analysis were reclassified to the binary maps, using 10% of the training set value as the threshold value. This value is suitable for models generated with data taken from a wide variety of sources.

Of the eight species modeled, *B. fagaroides* currently occupies the greatest area, and *F. resinosa* the least (Table 1). However, under the A2 climate change scenario, the latter species would undergo the greatest increase over its current area; 34.3% according to CGCM2 projections. Under the same scenario, *D. bicolor* is the species predicted to undergo the greatest loss (6.7% of its current area of 65,308 $km^2$). *K. humboldtiana* is the species predicted to increase the most (15.5%) according to HADCM3 projections, while *M. tomentosa* would undergo a high reduction of 29.1% (Table 1). Consensus maps from the two General Circulation Models for the A2 scenario under the climate change projections to 2050 show that legume species (*A. schaffneri, E. polystachya*) and the composites (*M. tomentosa* and *A. espinosarum*) would lose more area under the climate change projections than the other species in the model (Fig. 2). These species would not tolerate environmental modifications caused by climate change under scenario A2, and they are therefore unsuitable for use in long-term ecological restoration projects. The remaining species modeled maintained or, in some cases, gained a small amount of potential distributional area under this climate change scenario (Fig. 2).

These results lead us to propose *A. schaffneri, B. fagaroides, D. bicolor, E. polystachya, K. humboldtiana, G. glutinosa, F. resinosa* and *M. pringlei* as having potential for use in ecological restoration projects in the semiarid zone of Central Mexico. This selection was made with two main considerations; the presence of suitable ecological attributes for restoration (sociability, cover, abundance, micorrization) and tolerance to projected climate changes that would occur under the A2 scenario.

## 4. Discussion

For ecological restoration, the effect of climate change on species distribution must be taken into account (Gastón and García-Viñas, 2013; Harris et al., 2006). If the selected species cannot tolerate the environmental modifications predicted under climate change, the restoration strategy will not work in the mid and long term. To date, only historical and current climatic conditions are taken into consideration in establishing the necessary requirements for restoration projects (Ravenscroft et al., 2010). The scenarios derived from global climate change studies predict a significant increase in global temperatures, which could alter cloud and precipitation patterns and trigger the appearance of new arid zones or desertified areas worldwide (IPCC, 2007; Lawlor, 2001; Rehfeldt et al., 2012; Sáenz-Romero et al., 2010). Rehfeldt et al. (2012), using different biome models under climate change scenarios for three periods (2030, 2060 and 2090), found that tropical dry forests and deserts could be expected to expand in Mexico. This could cause significant and unpredictable changes to occur in ecosystems under restoration (Gastón and García-Viñas, 2013; Harris et al., 2006). It has been argued that future restoration efforts should be directed towards the establishment of ecosystems that are capable of persistence under these conditions (Cairns, 2002; Choi, 2004; Gastón and García-Viñas, 2013), because climate change management requires short and long-term strategies that will enable improved resistance and resilience in the ecosystems (Millar et al., 2007).

Ecological niche models have become a widely used tool for generating projections of species distributions. However, the success of the models in predicting the response of species to climate change must also consider other factors that may exercise an influence. An example is the case of *Quercus sartorri*, associated with warm-humid climates, which can probably survive the increased temperature that will result from climate change; however, it has been found that this species is not tolerant of high radiation. Solar radiation is increasing as the frequency of fog and cloud cover (with decreasing precipitation) decreases in areas currently inhabited by this species (Barradas et al., 2011).

While these models provide an approximation of distributions under climate change scenarios, various sources of uncertainty can affect this estimate. Models predict the potential area of the included species, but these results can be affected by external factors such as historic limitations, interactions between species, geographical barriers, or changes in land use patterns (Anderson et al., 2003; Sánchez-Cordero et al., 2005). We found no strong differences between the two GCM used in terms of percentage

**Table 1**
Percentages of pixels for the potential distribution of the species studied under different climate change scenarios (percentage based on the species could potentially occur with reference with records in present).

| Especies | Scenarios | | | | | |
|---|---|---|---|---|---|---|
| | Current (pixeles) | Current (%) | Canadian A2 (pixeles) | Canadian A2 (%) | Handley A2 (pixeles) | Handley A2 (%) |
| *A. espinosarum* | 39896 | 100 | 39316 | 98.5 | 39599 | 99.3 |
| *E. polystachya* | 78414 | 100 | 86176 | 109.9 | 80464 | 102.6 |
| *D. bicolor* | 65308 | 100 | 60906 | 93.3 | 63462 | 97.2 |
| *B. fagaroides* | 87739 | 100 | 87158 | 99.3 | 80522 | 91.8 |
| *K. humboldtiana* | 56738 | 100 | 58510 | 103.1 | 65535 | 115.5 |
| *F. resinosa* | 13308 | 100 | 17872 | 134.3 | 13308 | 100.0 |
| *A. schaffneri* | 78414 | 100 | 86176 | 109.9 | 78946 | 100.7 |
| *M. tomentosa* | 78414 | 100 | 86176 | 109.9 | 55569 | 70.9 |

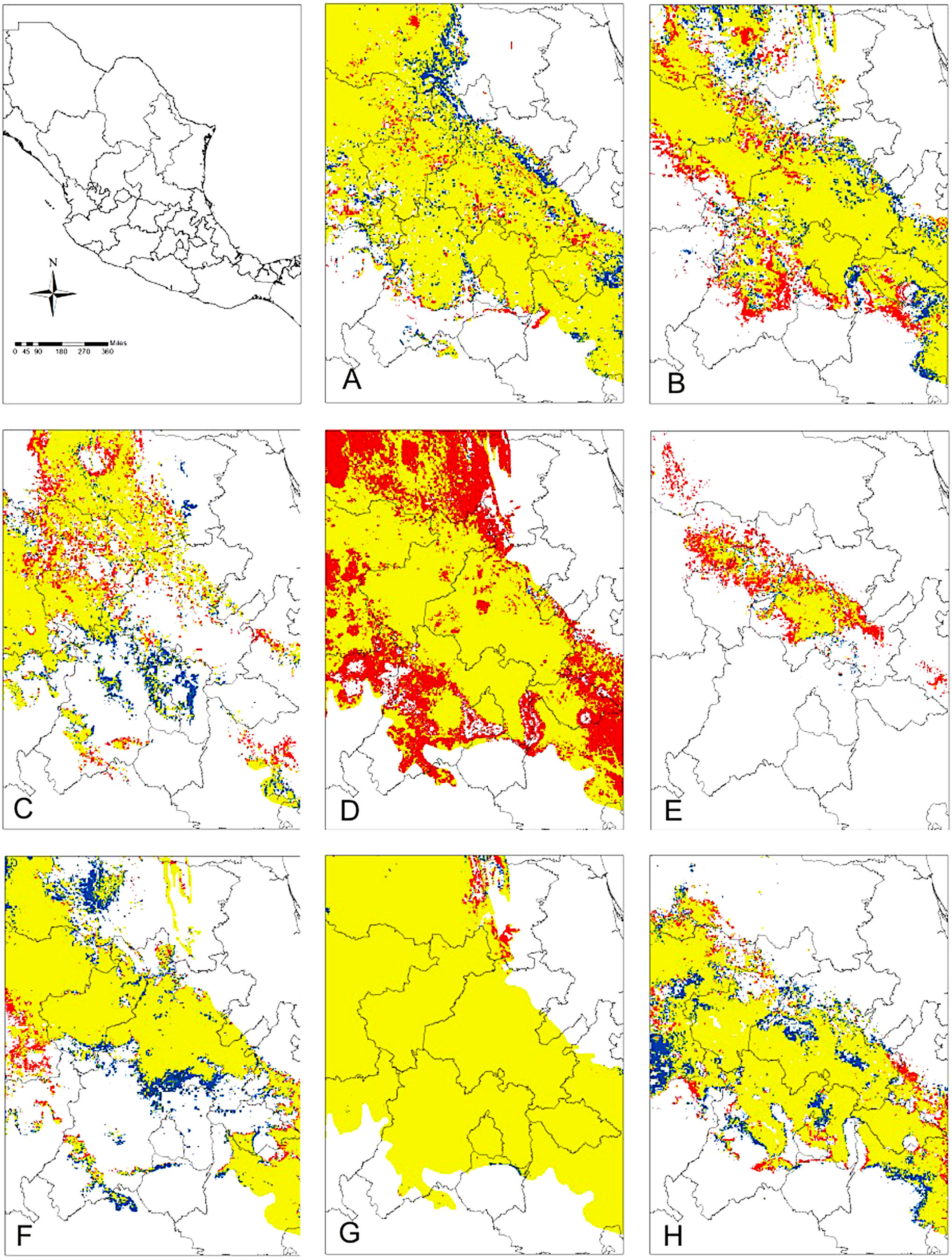


**Fig. 2.** Maps of potential distribution area, under climatic change scenario A2 for 2050, of 8 shrub species of the semiarid shrublands of Central Mexico. (a) *A. espinosarum*, (b) *F. resinosa*, (c) *M. tomentosa*, (d) *A. schaffneri*, (e) *E. polystachya*, (f) *D. bicolor*, (g) *K. humboldtiana*, and (h) *B. fagaroides*. Gray denotes the area conserved by the species, black denotes the area gained by the species and red denotes the area lost by the species by 2050.

changes in distribution range predicted by these models. The change is generally low except for *M. tomentosa* at 39%. This result may reflect the fact that in general, semiarid zones in Mexico are predicted to remain relatively stable under climate change scenarios, and in the northwest of the country these areas could even increase (Rehfeldt et al., 2012).

Given the limitations of potential distribution models, and the complexity of natural systems, it has been suggested that there are fundamental limitations to the precise prediction of future species distributions (Pearson and Dawson, 2003). In spite of these limitations, the models are still useful for the identification of possible magnitudes of future changes to distribution patterns, and to indicate which species, habitats and regions are most threatened by climate change (Berry et al., 2001; Midgley et al., 2002).

The response of species to environmental change can affect the results of restoration mainly because it is a process of long-term impact, and the effects of climate change on other biological groups have already been demonstrated (Harris et al., 2006). Given all these factors, it is important that species for restoration be selected from species native to the respective area of interest, because species that may be appropriate for one area are not necessarily suitable for others. We therefore consider that the approach used in this study would be a useful tool in the evaluation of other sites and environments.

The first ecological attributes we considered in the selection of species for restoration of arid tropical scrub were frequency and density, since it has been observed that the role of abundant species within ecosystem processes is greater than that of rare species (Cardinale et al., 2006). Although dominance could be a contradictory characteristic to sociability, the species selected in the present study had high association values. It is also important to carry out germination and establishment studies of the selected species, since these are clearly relevant attributes in restoration projects. For example, the selected species *B. fagaroides* presents low germination and recruitment percentages (Hernández and Espinosa, 2002; Ortiz-Pulido and Rico-Gray, 2006), which could preclude its consideration for use in the restoration area. Of the selected species, only *F. resinosa* has mycorrhizal associations, which are of great importance in semiarid environments because in some studies they have been shown to increase survival. For example, survival of some plants of *A. farnesiana* and *Prosopis laevigata* in the semiarid area of Hidalgo, Mexico increased by between 18 and 54% when they had been previously inoculated with mycorrhizal fungi (Monroy-Ata et al., 2007). The mycorrhizal association presented by many of these species acts to increase survival (Allen, 1991), photosynthetic efficiency (Augé, 2004), and resistance to drought (Allen and Allen, 1986) in these plants. The fact that these species have wide canopy coverage also means that they could potentially function as nurse plants for the germination and establishment of other species. A nurse plant can maintain a relatively higher content of nutrients and humidity in the soil and reduce the effects of solar radiation beneath the canopy relative to soils without such coverage (Castro et al., 2002; Godinez-Álvarez et al., 2003; Valiente-Banuet et al., 1991).

Furthermore, it is interesting that *F. resinosa* and *K. humboldtina*, the only evergreen species that were modeled, are those with the largest predicted increase in area over their current distribution under climate change, at 34.3% and 15.5% respectively. Evergreen species have traditionally been considered to respond to water stress better than deciduous species due to a combination of characteristics such as their deep roots, stomatal conductance, and low cuticular transpiration (Valladares et al., 2004). This functional group also has greater biomass and a low growth rate and therefore has less need for nutrients (Givnish, 2002). Climate change projections under the A2 scenario predict an increase in temperature and water deficit in the study area. It is generally expected that the actual availability of water for plants will decrease during the twenty-first century, which will lead to increased evapotranspiration as a result of the increase in temperature (IPCC, 2001), so evergreen shrubs can be expected to show a better response to water stress than deciduous shrubs. This could partly explain why the two evergreen species were the species that showed the largest increase in area under climate change. In general, the loss of area was low for all eight species modeled, which implies the coexistence of both functional groups of shrubs; evergreen and deciduous. Survival studies of both these groups of shrubs are required, evaluating both adults and seedlings, to assess whether the abundance of evergreen shrubs may exceed that of deciduous shrubs in future arid tropical scrub zones in Central Mexico.

Land managers require tools to support decisions made in response to ecosystem changes caused by climate change (Rehfeldt et al., 2012). This paper proposes a methodology for selecting species with potential for use in ecological restoration programs carried out in light of predicted future climate changes. Ecological niche models are an efficient tool for planning suitable climates for species (Pearson and Dawson, 2003), and can be used to propose solutions to problems in a variety of disciplines such as conservation biology and ecological restoration.

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