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SPECIAL THEME: ECOLOGICAL RESTORATION IN MEXICO

Creating Refuges for the Axolotl (*Ambystoma mexicanum*)

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Cropping in the naturally fertilized Xochimilco wetlands during seasonal floods helped Aztecs to build one of the most advanced civilizations of the Americas. At the edge of Mexico City, Xochimilco hosts more than 140 migratory birds and an endemic crayfish (*Cambarellus montezumae*) and fish (*Menidia humboldtiana*). It is the last remnant habitat of one of the most iconic animals from central Mexico: the axolotl (*Ambystoma mexicanum*) (Figure 1). This salamander has been associated with Mexican culture as twin of the most important Aztec god Quetzalcoatl and inspiration for writers and philosophers.

Mexico City requirements for ecosystem services provided by Xochimilco wetlands have intensively perturbed this aquatic system in the last 60 years. Its water quality has decreased, with high bacteria, nutrient, and heavy metal concentrations (Zambrano et al. 2009). Consequently, axolotl populations have decreased alarmingly in the last decade, with high probabilities of extinction in the wild by 2019 (Zambrano et al. 2007). The few remaining populations are scattered and isolated, and therefore restoration measures are critical.

Top-down management and restoration attempts in Xochimilco have not worked, leading to abandonment of traditional agriculture for greenhouse production, reducing even more the water quality and axolotl habitat. An aquaculture fish introduction program resulted in an overabundance of tilapia (*Oreochromis niloticus*) and carp (*Cyprinus carpio*), which affect axolotl populations by predation and modifying food webs. A program of axolotl reintroduction conducted by an academic institution and sponsored by local government, using salamanders grown in captivity, is not expected to increase population numbers (see Zambrano et al. 2007). Moreover, if siblings are used for the introduction program, the genetic diversity of wild populations may be reduced.

Long-term success of restoration measures requires involvement of local people in the management program (Gall and Staton 1992). Scientists together with local farmers (chinamperos) are working to create an alternative axolotl restoration program. Traditionally, these local farmers produce in "chinampas," which are land structures forming a labyrinth of canals connecting lakes and wetlands (Contreras et al. 2009). Chinamperos use canal sediments for cropping, reducing chemical fertilizers. Also, canals surrounding the chinampas increase the wetlands' spatial heterogeneity that seems to be necessary for axolotl survival. This indicates that it is necessary to conserve the traditional agriculture in order to restore the wetland.

By sharing information with chinamperos, we have enough ecological and empirical knowledge about the axolotl to create a restoration program focused on improving its habitat. We now understand that population dynamics depend mostly on the survival of eggs and larvae (Zambrano et al. 2007), which is closely related to plants used by females to lay their eggs (Valiente 2006, Marín 2007). The axolotl is at the top of the food web, but shares most of its food sources with the non-native carp and tilapia (Zambrano et al. 2010). Axolotl distribution is limited to a few small canals, surviving away from urban and greenhouses areas (Contreras et al. 2009). City demand

Figure 1. **The Xochimilco wetlands (*top*) next to Mexico City provide habitat for the culturally iconic axolotl (*Ambystoma mexicanum*), an aquatic salamander (*bottom*) that is rapidly disappearing from the increasingly degraded wetlands and has been bred in captivity in early restoration efforts.** Photos by Luis Zambrano (***top***) and Carmen Loyola (***bottom***)

for Xochimilco ecosystem services makes it impossible to restore the entire wetland, and isolating the conservation of the axolotl from human activities will break the already weak link between the salamander and citizens. Therefore chinamperos and scientists decided to generate refuges within chinampas border canals.

Canals used as refuges were isolated from the system with rustic filters to exclude non-native fish and improve the water quality (Figure 2). Many chinamperos water their lands manually by carrying water from the canals, and improved water quality will also yield farm products that test free from bacteria and heavy metals, which allows chinamperos to better market their products. This partnership helps both the axolotls and the chinamperos by improving habitat and preserving traditional agriculture.

We are comparing the first experimental refuge canal, established in January 2009, with control canals and evaluating individual axolotl growth and health. The first evaluation suggests an improvement of water quality after four months. Refuge water was significantly lower in turbidity (15%), ammonium (77%), and nitrates (87%) than water in the control canal. Invertebrate abundance did not vary significantly, except for Odonata larvae, which were denser in the refuge canal. In this first evaluation, only 30% of the 12 previously marked axolotls were recaptured. The rest of them could have been hidden in mud or depredated by snakes. However, recaptured axolotls gained, on average, 16% of their weight, which is more than the growth of individuals in experimental colonies within the same period of time (Zambrano et al. 2007).

We need more information to launch a multicanal refuge program, for example, the effectiveness of different plant types as shelter for axolotl eggs and larvae, or food web changes within the refuges. For this reason, a second phase with four additional canals in three chinampas is starting now. This second phase will yield results within the next year to inform a proposed larger refuge network to cover at least 10% of the chinampas canals. Refuges should function as bridges among areas where axolotls still survive (Contreras et al. 2009).

At every stage, scientists and chinamperos have to work together. Therefore we are holding participatory workshops to develop strategies so that chinamperos can manage refuges themselves in the long term.

Figure 2. Rustic filters in one of the axolotl refuges in Xochimilco. The rustic filters improve water quality and keep out non-native fish. Note the chinampa in production on the left. Photo by Homan González

The refuge program aims to conserve both this salamander and traditional agriculture. We are generating a market for farm products that will include the cost of saving local culture, Xochimilco, and axolotls. Restoration of the axolotl is a complex process that must be discussed among stakeholders (such as local government, fishermen, and tourists) in order to produce multiple actions and a monitoring program. Declining axolotl populations in the wild do not leave much time or room for mistakes. But a good restoration process will bring not only increased axolotl populations but also other benefits from a functioning wetland and better market opportunities for chinamperos.

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Tropical Dry Forest Landscape Restoration in Central Veracruz, Mexico

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Until recently, it was thought that native tropical dry forest (TDF) in central Veracruz was completely gone. Fortunately, recent research using Landsat images and ground-truth verification showed the presence of TDF remnants. There is still 7% of the original forest in the region, although one-third is secondary vegetation (F. López-Barrera, Instituto de Ecología A.C., pers. comm.).

Tropical dry forest landscape restoration has been carried out in only a few places. The most integral restoration experience started in Parque Nacional Santa Rosa, Costa Rica, which was expanded to become Area de Conservación Guanacaste, a region ten times larger (Janzen 2008). Other restoration efforts are being carried out on the Pacific coast of Panama (Griscom et al. 2009), Paraná River Valley in Central Brazil (Sampaio et al. 2007), Hojancha and Cañas in Costa Rica (Fonseca-González and Morera 2008), the Ayuquila River watershed in Jalisco (Ortiz-Arrona et al. 2004), and the Tembembe river in Morelos, Mexico (Bonfil et al. 2004). Most TDF restoration experiences indicate that besides establishing native species plantations, restorationists should rely on the resprouting ability of the trees characteristic of these forests (Griscom et al. 2009) and different management techniques to facilitate this natural regeneration capacity (Sampaio et al. 2007). We recognize that natural tree regeneration is not only the least expensive method possible; it is also a tool for designing restoration efforts (Janzen 2008, Viera and Scariot 2006).

This study is part of Project ReForLan, which focuses on the restoration of dryland forest landscapes for biodiversity conservation and rural development in Latin America (Newton 2008). Our goal was to test restoration techniques and identify the main constraints to forest restoration. The two objectives were to define a reference system through the determination of forest structure and tree species composition, and to define a set of tree species to be used in restoration efforts.

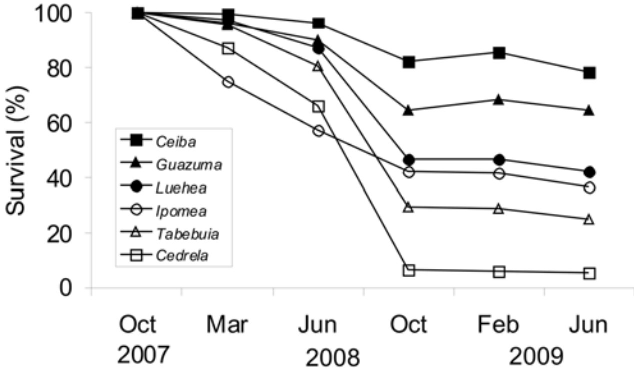

Figure 1. Survival percentages (October 2007–June 2009) for six tree species planted in restoration experiments in central Veracruz, Mexico. Survival of cedro (*Cedrela odorata*) was significantly lower than other species (one-way ANOVA, $F = 3.7$, df = 5,18, $p = 0.02$).

The study area is located in central Veracruz, Mexico (19°17'N, 96°26'W, 100–250 m asl). The climate is hot and dry. Mean minimum and maximum temperatures are 19.8°C and 30.7°C, respectively. Total annual precipitation is 966 mm (range: 502–1,466 mm). The dry season extends from October to May. Soils are mainly Cambisol and Vertisol. In this region, land is mainly used for cattle ranching, generally on a small scale by private landowners; but for common land tenants (ejidatarios), the main activity is growing corn and other crops, such as papaya, bean, green chili, watermelon, sugar cane, and mango. This area is rich in history related to pre-Hispanic settlements (600 to 1500 A.D.) and the Mexican Independence (19th century). In addition, the region includes an important observation point, the River of Raptors, for one of the largest annual migrations of raptors in the world.

The TDF reference system was determined through the selection of ten forest remnants and five early successional sites, 1 to 72 months after the last abandonment, with different land use histories (Williams-Linera and Lorea 2009, Williams-Linera et al. 2010). Vegetation structure was characterized in terms of density, basal area, and height for canopy trees (≥ 5 cm dbh) and understory woody plants (< 5 cm dbh). Mean density was 1,014 ± 104 and 2,532 ± 2,272 individuals/ha, with basal area 30.2 ± 2.11 m^2/ha and 1.96 ± 0.12 m^2/ha, for canopy and understory vegetation, respectively. The forest mean height was 10 m and reached a maximum of 15 m. The early successional sites had a basal area and density ranging from 0.40 to 3.88 m^2/ha and from 900 to 5,450 individuals/ha, respectively.

We recorded 122 woody plant species in the forest fragments (Williams-Linera and Lorea 2009). In the early successional sites, 45 woody species were recorded. Some species were represented only in forests or fallows, but 20 tree species were growing in both early secondary and mature forest sites (Williams-Linera et al. 2010), indicating that mature forest species were entering the successional process at very early stages.

Next, we assessed local knowledge and preferences through workshops, field visits, key informants, informal interviews, and specimen collection to determine tree species that were economically and ecologically valuable resources for restoration. This process identified 75 species as useful, rare, valuable, or important for wildlife (Suárez et al., forthcoming). Using the lists of tree species in forest or fallows as well as those often mentioned locally, we selected six native tree species for the restoration experiments. Two were timber species, one was used as forage, and the other three displayed potential to be used for such nontimber forest products as firewood, ornamentals, handcraft manufacture, and the growth of an edible mushroom that is very popular locally (Table 1).

We established four restoration trials to evaluate the establishment and growth of these six species in fallows with different degrees of disturbance. The early successional sites for the restoration experiments were fenced with barbed wire to exclude livestock. Seeds were collected in the study area and germinated in the local nursery where seedlings stayed four to six months prior to transplant. In September 2007, 960 seedlings were transplanted to four 12 × 20 m plots per site, with individuals 2 m apart. Plant survival and growth in basal diameter and height have been monitored every four months.

Seedling survival was statistically similar among species, except for cedro (*Cedrela odorata*) (Figure 1). After the first dry season, survival of all species was greater than 55%, a result similar to that of restoration studies in pastures in central Brazil, where survival was greater than 60% (Sampaio et al. 2007) or 35%–77% (Vieira et al. 2007). After the second-year dry season, survival was the lowest for cedro (5.1%) and the highest for pochote (*Ceiba aesculifolia*; 78.3%) and guácimo (*Guazuma ulmifolia*; 64.3%) (Figure 1).

The relative growth rate in height (RGR_h) was statistically similar among species, whereas in diameter (RGR_d) it was different among species (Table 1). Cedro had the highest RGR_d. Cedro, despite its excellent growth performance, appeared unable to adapt to drought conditions, as has been previously reported (Ortiz-Arrona et al. 2005). Guácimo maintained a high survival percentage because of its resprouting ability; with the rainy season, some apparently "dead" individuals resprouted, as has been reported in other forests (e.g., Viera and Scariot 2006, Griscom et al. 2009). Pochote grew slowly because its stem top frequently breaks during the dry season, recovering when it rains.

Each tree species may contribute something different to the restoration effort. Results of transplant experiment suggest that all selected species can be used for restoration; however, each species requires different site conditions for transplanting. Timber species (cedro and roble, *Tabebuia rosea*), which are the most important ones for local people, are severely drought intolerant. Therefore, they need to be planted near isolated trees that help to maintain humidity.

Table 1. Tree species used in restoration assays in the TDF of central Veracruz, Mexico, and their mean (± SE) relative growth rate in height (RGR_h), and diameter (RGR_d) during 19 months. The RGR_h was similar across species (one-way ANOVA, $F = 2.27$, df = 5,67, $p = 0.06$), and RGR_d was significantly different ($F = 6.47$, df = 5,67, $p < 0.0001$).

Use	Common name	Species	Family	Relative Growth Rate: Height (cm/cm/y)	Relative Growth Rate: Diameter (mm/mm/y)
Timber	Cedro	*Cedrela odorata*	Meliaceae	0.54 ± 0.21	0.66 ± 0.10
	Roble	*Tabebuia rosea*	Bignoniaceae	0.41 ± 0.07	0.41 ± 0.09
Forage	Guácimo	*Guazuma ulmifolia*	Sterculiaceae	0.29 ± 0.14	0.50 ± 0.06
Nontimber forest products	Pochote	*Ceiba aesculifolia*	Bombacaceae	0.09 ± 0.12	0.19 ± 0.05
	Patancán	*Ipomoea wolcottiana*	Convolvulaceae	0.35 ± 0.12	0.39 ± 0.07
	Algodoncillo	*Luehea candida*	Tiliaceae	–0.01 ± 0.14	0.28 ± 0.02

To avoid seedling mortality, sites with some tree or other canopy cover have been recommended because they offer a milder environment and moister soil than open sites (Viera and Scariot 2006). In contrast, pochote and guácimo are drought tolerant and may be used in disturbed areas with no woody vegetation; their resprouting ability allows establishment and survival in extremely dry conditions. Local people are not enthusiastic about nontimber species, but they are important for restoration by changing the microenvironment, providing suitable conditions for other species of economic importance to establish later.

The procedures used for TDF restoration must be adapted to this particular environment, instead of following techniques developed for temperate or moist forests (Viera and Scariot 2006). Seedling transplant represents one option; however, since primary species are present at early successional sites, it is worthwhile to consider the possibility of restoring TDF through passive restoration via natural colonization. Our next objective will be to determine whether secondary succession is efficient in the recovery of original vegetation. If not, we would enrich successional sites with nonresprouting species or some primary forest species that cannot be dispersed at early successional sites. Additionally, restoration should be implemented to strategically connect remnant forests, successional sites, and historical landmarks that may promote tourism and thus boost the local economy.

Acknowledgments

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Restoring the Vara de Perlilla in La Mesa, Mexico

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The vara de perlilla (*Symphoricarpos microphyllus*), or rejagar, is a shrub found from New Mexico to Guatemala. In Central Mexico, it is found mostly in pine-oak forests and also in the true fir forests, where it blooms from July to September, produces fruits from October to December, and provides forage for wildlife such as deer. Its branches are widely used to make brooms and manufacture Christmas crafts, making this shrub an important nontimber forest product in Mexican temperate areas. Despite its resprouting ability, vara de perlilla populations are starting to decline because of overuse and deforestation or forest degradation. In other cases, the individuals exist, but they are overutilized, too young, and small, so cannot be employed when required.

Propagation and reforestation of this shrub just began a few years ago, mostly for commercial purposes. Yet its seed propagation still is not successful, and the ideal conditions for establishing this species are not well known. Its light requirements have not been studied formally. We considered light to be a key limiting factor because the species does not prosper under dense shade, but it does well under partial shade or full sun. The objective of this work was to determine appropriate light conditions for the establishment of restoration plantings with this species.

This study was carried out in La Mesa ejido, San José del Rincón municipality, in the state of Mexico (19°34'N and 100°10'W). The climate is temperate, with a mean annual precipitation of 904 mm and a mean annual temperature of 12°C, and soil is an Ocric Andosol. Cuttings were collected December 23–26, 2006 (winter, dry season, after fruit production), from local vara de perlilla populations. They were 15–20 cm long and were immediately dipped in Raizone Plus solution and planted in plastic bags filled with local pine-oak forest soil. The plants were grown in a small local forest nursery for six months. A 30% shade cloth protected the plants during five months but was removed one month before planting during the hardening phase. Irrigation was provided every other day, adjusting in case of rain, and was slightly reduced during the last month of hardening.

We planted a total of 900 six-month-old shrubs for this work: 450 small (20–35 cm length) and 450 large (45–60 cm length). Three planting sites (treatments) were chosen:

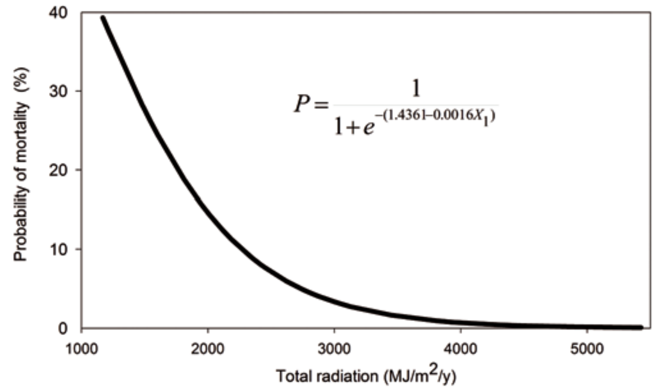

Figure 1. Relationship between total solar radiation (X_2) and probability of mortality (P) for the vara de perlilla (*Symphoricarpos microphyllus*) in an oak forest in Central Mexico.

oak forest (encino laurelillo, *Quercus laurina*, with some individuals of aile, *Alnus jorullensis*); six-year-old pine forest plantation (*Pinus pseudostrobus*, 3.5 m tall on average, 1,111 trees/ha density), on a former oak forest site; and an agricultural field cleared 20 years ago from oak forest. The treatments formed a gradient from less to more light: oak forest (1328.6 to 5224.1 MJ/m²/y), pine plantation (6388.4 to 9140.9 MJ/m²/y), and field (7050.2 to 9324.6 MJ/m²/y). The plots were approximately 150 m apart. In July 2007 during the summer rainy season, we planted 300 shrubs per treatment: 150 small and 150 large. The shrubs were planted one meter apart in six completely randomized blocks per treatment. Each block segregated 50 plants by size. To eliminate competition, grasses and forbs were removed in a circle of 1 m radius around each shrub. To measure light levels at planting time and one year later, we placed, leveled, and oriented to the north a digital camera with a hemispherical lens integrated into a frame on the three shrubs in the center of each plot to take 180° photographs toward zenith. Such hemispherical photographs were analyzed with the HemiView system (Dynamax, Houston TX), with date, latitude, longitude, and altitude to estimate direct, diffuse, and total solar radiation (MJ/m²/y) (averaged between the two observations taken in each site).

After one year of growth in the planting site (July 2008), we measured several morphological variables, including shoot, root, and total biomasses. For this purpose, six plants from each block (3 per size category, 36 per planting site, for a total of 108) were carefully removed, cleaned, and oven-dried. All 900 planted shrubs were observed for survival. The probability of mortality was calculated with logistic regression, using solar radiation as the independent variable. Mortality and biomasses were compared among planting sites with the *t*-test; biomasses among size categories were compared by analysis of variance and least significant difference.

Table 1. Total biomass (± SD) and growth increment of vara de perlilla (*Symphoricarpos microphyllus*) by treatment and plant size (L = large, S = small) in Central Mexico. Total biomass at one year was significantly different from initial biomass for all treatments and sizes ($p < 0.05$).

Treatment	Size	Biomass (g) Initial		Biomass (g) 1 y		Increment (g)	Increment (%)
Pine forest	L	4.5	± 1.3	22.3	± 10.9	17.8	496
	S	2.1	± 0.8	14.3	± 9.9	12.2	681
Oak forest	L	4.5	± 1.3	16.5	± 9.7	12.0	367
	S	2.1	± 0.8	8.4	± 4.2	6.3	400
Agricultural field	L	4.5	± 1.3	13.9	± 4.0	9.4	309
	S	2.1	± 0.8	9.2	± 3.4	7.1	438

The overall survival was 96.4%, with 91% for the oak forest, 98.7% in the pine forest plantation, and 99.7% for the agricultural field, without significant differences among treatments. The oak forest was the only treatment where the logistic model for the relationship between survival and radiation was significant ($p < 0.00001$), with mortality increasing at higher shade levels (Figure 1). However, drought influences survival at different light levels. In dry conditions (for example, derived from late tree planting), shrub survival is higher under shade than in the opening (Hernández-García and Rodríguez-Trejo 2008).

One year after planting, in all cases large plants averaged more shoot biomass than small plants ($p < 0.05$): oak forest (10 g and 5.2 g, respectively), pine forest plantation (13.3 g and 8.7 g), and agricultural field (7.5 g and 4.9 g), with the higher biomass among treatments for the pine plantation ($p < 0.05$). Root biomass behaved similarly: oak forest (6.5 g and 3.2 g, respectively), pine forest plantation (9.0 g and 5.6 g), and agricultural field (6.5 g and 4.2 g), as well as total biomass (Table 1). The growth after one year, measured as percentage of original size, was higher for small plants and higher in the pine forest than in the other treatments ($p < 0.05$) (Table 1).

In our work, total solar radiation intermediate between the oak forest and the pine forest plantation seems to yield higher total biomass, although without differences in shoot/root dry weight ratio among conditions, indicating a semitolerance to shade by the vara de perlilla. In contrast, shade-tolerant species, like salal (*Gaultheria shallon*) and evergreen huckleberry (*Vaccinium ovatum*), increase their shoot/root dry weight ratio under shade in comparison to direct solar radiation (Hawkins and Henry 2004).

In La Mesa, the vara de perlilla may be planted in partial shade or full sun, because it is semitolerant. Even at low light levels (for example, 2000 MJ/m²/y), as in dense oak forests, its one-year mortality probability is only 15%. Midlight levels will maximize shoot biomass and facilitate high survival. In dry years, partial shade may help to prevent high mortalities.

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Germination, Emergence, and Survival of *Buddleja cordata* in an Urban Forest

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Mexico City, one of the world's largest cities, has grown at the expense of different ecosystems. The Parque Ecológico de la Ciudad de México (PECM) is located on Mt. Ajusco south of Mexico City (19°14'N and 99°15'W, elev. 2,600–2,900 m asl), one of the surrounding mountains that are important for conserving biodiversity and recharging aquifers. This park includes 730 ha, 200 of which were deeply disturbed in the early 1980s by an urban shantytown settlement (Cano-Santana et al. 2006).

Vegetation includes oak, pine, and pine-oak forests as well as xerophylous shrubland growing on basaltic substrate. Soils are well developed in the forests but scarce in the shrublands. The basaltic substrate has low water

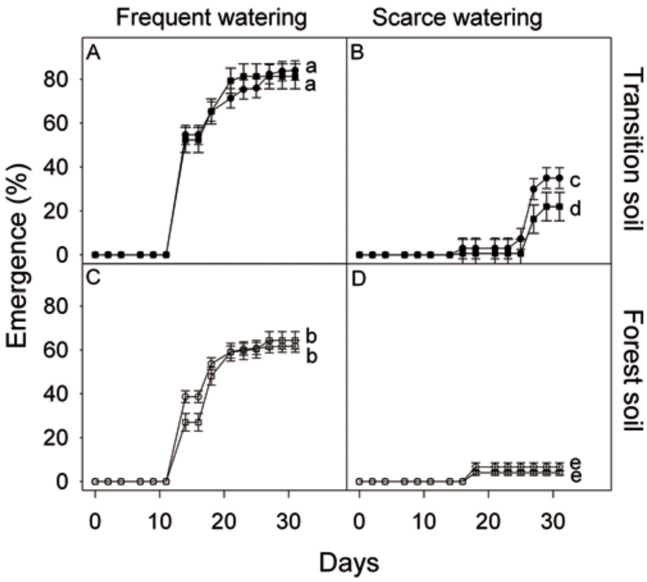

Figure 1. Emergence of tepozán (*Buddleja cordata*) seedlings in a shadehouse in 1995 in Parque Ecológico de la Ciudad de México from seeds collected in 1992 (●, ○) and 1995 (■, □) under different treatment conditions: A) transition soil and watering every three days (frequent); B) transition soil, watering every seven days (scarce); C) forest soil, frequent watering; and D) forest soil, scarce watering. Small letters in the graphs indicate significant differences.

retention and wide temperature variation (0°C–50°C) (Olvera-Carrillo et al. 2009). Average annual temperature is 12°C–14°C, and annual precipitation, concentrated in the summer rainy season, is 700–1,000 mm. In the most disturbed area, volcanic rock was fragmented and removed. The most common pioneer trees in this area are known locally as tepozán (*Buddleja cordata*) and chapulixtle (*Dodonaea viscosa*), while mala mujer (*Wigandia urens*) is the most common shrub. However, the harsh environmental conditions make plant recruitment difficult, and intervention to restore this area is necessary.

Tepozán is common in disturbed habitats from Mexico and Guatemala (Rzedowski and Rzedowski 2001). It flowers between July and December, and seed shedding occurs from October to March. The tiny seeds (0.0278 ± 0.0005 mg) are wind dispersed, need light to germinate, have high viability, and are partially dormant after collection (González-Zertuche et al. 2002). Tepozán has high litter production and growth rates, which make it a suitable species to restore ecosystem structure and functionality in the area (Vázquez-Yanes and Bátis 1996, Mendoza-Hernández 2003)

To make better use of tepozán in the restoration of disturbed shrubland, we tested the effects of seed age and collection time on the germination of this species and measured the effects of substrate and watering on the emergence and survival of seedlings. We seek a better understanding of these early life stages that are crucial to establishment of this important pioneer species.

We collected seeds from ten trees in October 1992 (autumn), November 1993 (autumn), January and February 1994 (winter), and March 1995 (winter). Seeds were cleaned, air dried, and stored at room temperature (23°C–25°C; 20%–50% RH), until use in June 1995. To compare germination among seeds produced during the same season, we used seeds collected in November 1993 and January and February 1994. Seeds from each collection date were divided into three replicates of 50 seeds each and germinated in Petri dishes on a 1% agar plate in a growth chamber (Model 125L, Conviron, Winnipeg, Canada) at 25°C and 12 h of light. We recorded the number of germinated seeds every three days. Final germination percentages were arcsine transformed and compared using an ANOVA test (Statistica, vers. 5, Statsoft, Tulsa OK). Under constant temperature, seeds germinated simultaneously, beginning on the fifth day and ending on the tenth day. Seeds collected in the rainy autumn had higher germination percentages (77.8 ± 2.4) than seeds collected in the dry winter (35.6 ± 1.1 and 28.5 ± 2.5, for the January and February seed lots, respectively) ($F_{2,8} = 22.937$, $p = 0.0001$), suggesting that environmental conditions had a significant effect on seed dormancy and vigor.

Next, we determined the effect of soil type, watering regime, and seed age on seedling emergence. We filled 24 pots with forest soil; half were sown with seeds collected in 1992 and the other 12 with seeds collected in 1995, using 50 seeds per pot. Pots were placed in a shadehouse in the PECM. To emulate precipitation variability, six pots from each cohort were watered every three days (frequent watering), and the other six only once a week (scarce watering). The same procedure was carried out with 24 pots filled with soil from the transition zone (between the forest and the secondary shrubland). Seedling emergence was recorded every three days. Emergence and survival percentages were arcsine transformed and compared using an ANOVA test (Figure 1). There were no significant differences in emergence between seed batches ($F_{1,47} = 0.21$, $p = 0.64$). Watering frequency and soil origin, however, significantly interacted to influence seedling emergence ($F_{1,47} = 8.7$, $p = 0.002$). Seeds sown in soil from the transition zone had higher emergence rates regardless of watering frequency. Seedling emergence began on the 14th day under frequent watering, while under scarce watering it started after 16 and 18 days on soil from the transition zone and forest, respectively.

All seedlings from the scarce-watering regime died one month after emergence (Figure 2). Seedlings from the 1992 collection growing under frequent watering and on forest soil showed the lowest mortality rate. The highest mortality rate occurred in seedlings from the 1995 collection growing on the transition zone soil ($F_{3,19} = 13.68$, $p = 0.0001$). In the shadehouse, a high level of emergence was observed under frequent watering, probably because this species requires a high soil water potential to germinate

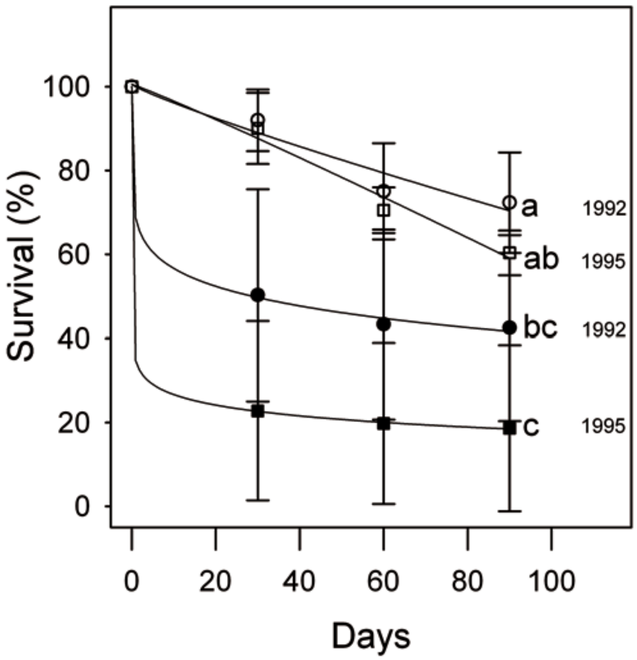

Figure 2. Survival of tepozán (*Buddleja cordata*) seedlings in a shade house in 1995 in Parque Ecológico de la Ciudad de México from seeds collected in 1992 and 1995. Seedlings were grown in pots filled with forest soil (open symbols) and in transition soil (closed symbols). Small letters indicate significant differences.

(González-Zertuche et al. 2002). Soil in the transition zone is porous and contains small pebbles of volcanic rock, thus offering more root emergence opportunities for the seedlings than the more compact forest soil.

Under frequent watering, seedlings coming from seeds collected in autumn had higher survival rates than those coming from seeds produced in winter ($F_{3,19}$ = 47.15, p = 0.00001). Seedling survival was low on soil from the transition zone, where this species typically grows, and higher on forest soil, where it is rarely found, probably owing to low light conditions under the closed canopy. These results indicate that there is a fine balance between the requirements for seed germination and those for seedling establishment.

Since tepozán can be used for restoration in harsh environments with thin, undeveloped soils, such as the PECM shrublands, we highly recommend nursery production using seeds collected in the autumn and forest soil mixed with coarse sand or small pieces of volcanic rock to increase germination and seedling emergence and survival. Considering the small size of seeds and the fragility of seedlings, we recommend producing saplings grown from seeds in pots and transplanting them to the disturbed sites, along with the soil in which they grew. Additionally, seeds can be stored for at least three years without detrimental effects on their viability and vigor, allowing for large seed collections and nursery production over time.

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