CHAPTER 16

Prospects for *Circa Situm* Tree Conservation in Mesoamerican Dry-Forest Agro-ecosystems

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Dry forest once stretched almost continuously along the Mesoamerican (Central American/Mexican) Pacific coast, from Sonora in Mexico to Guanacaste in Costa Rica (area of 550,000 km²). Conditions suitable for dry forest also exist on the Yucatán Peninsula, the coast of the Mexican states of Puebla and Tamaulipas, with significant inland areas in dry valleys (e.g., Motagua Valley, Guatemala) throughout the region (see Graham and Dilcher 1995 and Murphy and Lugo 1995 for the distribution and origins of this forest type). Human preference for the seasonally dry tropical environment (Murphy and Lugo 1995) and the ease of clearing its vegetation have, however, led to the destruction and fragmentation of much of its forest. Janzen (1988: 130) estimated that less than 2 percent of the original forest is in a state "sufficiently intact to attract the attention of the traditional conservationist," with only 0.09 percent having official reserve status. Only on the Pacific Mexican coast are there significant areas of mature, possibly primary, dry forest. Even in these areas it is rare to find a forest with no evidence of occasional timber and firewood extraction, extensive livestock browsing, or hunting. In Central America the dry-forest zone is reduced to a patchwork of various types of agricultural land and small forests of secondary origin. Unregulated selective felling of economically important tree species continues, resulting in the commercial extinction of timber species in some areas (e.g., *Astronium graveolens* Jacq., *Guaiacum sanctum* L.). Reserve selection is often opportunistic, determined more by political, social, and economic constraints than by optimal biological criteria. Consequently the location of reserves may bias floristic composition (at species and gene pool levels), limiting their overall value for biodiversity conservation (Ledig 1988).

Any conservation initiative requires consideration of the extent and value of and threat to the resource to be targeted. Tropical dry forests are species-rich in comparison with most of the

world's forest types (Gentry 1995). Biodiversity can, however, be measured at a variety of levels, and species numbers alone are an insufficient and poor measure of the conservation importance of any ecosystem. Consideration must also be given to the biota's uniqueness, as well as the importance of interactions with other ecosystems. At the species level, Mesoamerican dry forests contain a woody flora that is distinct from that of Mesoamerican wet forests (Gentry 1995). Even for tree species that occur in both wet and dry forests, the dry-forest populations have typically been shown to be genetically distinct from wet forest populations (e.g., *Cordia alliodora* [R. et P.] Oken, Chase et al. 1995; *Cedrela odorata* L., Gillies et al. 1997). Mesoamerican dry forests are also host, depending on the season, to many migratory species, including species from the wet tropics (chapters 7 and 8), such that its conservation is important in protecting more than just its own biota.

Interest in the conservation of these species and their habitat is not confined to biologists. Mesoamerican dry forests are the natural habitat of a disproportionately high number of species commonly used in tropical agriculture and forestry for a variety of goods and services, such as *Gliricidia sepium* (Jacq.) Steudel, *Leucaena* spp., *Prosopis juliflora* (Swartz) D.C., and *Samanea saman* (Jacq.) Merrill. These species are now propagated over much broader climatic and geographical ranges than their natural distributions and have considerable socioeconomic importance (e.g., Stewart et al. 1992). Continued use and improvement of such species depend to some extent on the maintenance of genetic variation within and between natural and semi-natural populations.

OPTIONS FOR CONSERVATION

IN SITU CONSERVATION

Conservation faces a problem. Mesoamerican dry forest is diverse and distinctive, containing many socioeconomically important species; however, existing reserves cover a very small fraction of the original forest, and few intact dry forests

are left to conserve. The degree to which *in situ* reserves are appropriate for conservation depends on a variety of factors, such as forest size, the extent of fragmentation, and the prevailing socioeconomic context. Those forests that do remain are usually small (e.g., 1–500 ha), below the size that might be considered viable, and highly intervened, such that the ideal of maintaining a single large reserve is often irrelevant (as in the "single large or several small" debate; see Soulé 1987). Consideration must therefore be given to the biological and social feasibility of managing networks of small patches within the current land-use mosaic. Conservation initiatives for dry forest must examine approaches that depart from the traditional *in situ* conservation paradigm, involving protected "wilderness areas," and focus as well on ways in which the species of an already highly altered forest type can be conserved.

ECOLOGICAL RESTORATION

Ecological restoration has been proposed as a tool to redress the inadequacies of current reserves (e.g., Janzen 1986a), and extensive areas are currently under restoration management of various types in both the Guanacaste and Tempisque Conservation Areas in Costa Rica. Such restoration is facilitated by the very presence of existing *in situ* reserves, but in practice it is rarely feasible elsewhere given the socioeconomic conditions prevalent in large parts of Mesoamerica. Even, for example, the establishment of Guanacaste National Park required the displacement of a human population living in, or farming, the area, with smaller farmers and squatters faring relatively poorly in the purchase and compensation process (Utting 1993). Nonetheless, in this case, sufficient national and international interest has probably been generated for the long-term maintenance of such management. Guanacaste, along with parts of Pacific lowland Mexico and the Yucatán Peninsula, however, represents the better end of the spectrum in terms of the size and quality of remnant dry-forest patches. Although, at a smaller scale, restoration ecology may have great relevance in the creation of

biological corridors between more or less intact areas of forest, it and traditional *in situ* conservation will be inappropriate for much of the rest of the Mesoamerican dry-forest life zone, given the highly degraded and patchy nature of the forest resource and prevailing socioeconomic conditions.

EX SITU CONSERVATION

Similarly, *ex situ* methods (e.g., seed banks, seed stands, botanic gardens) offer limited long-term potential for most tropical tree species. This is clear from the small number of tree taxa (some 100 worldwide, mainly of economic importance; National Research Council 1991) subject to effective *ex situ* conservation; the scientific, technical, and resource limitations that constrain *ex situ* programs (National Research Council 1991); and inherent deficiencies of *ex situ* populations as conservation gene pools (Brown et al. 1989).

CIRCA SITUM CONSERVATION

The limitations of both *in situ* and *ex situ* approaches, and the recognition that farmers maintain a large array of plant biodiversity in some traditional farming systems, have led to a reexamination of the potential role of "conservation through use" on farms (e.g., Maxted et al. 1997). The term *circa situm* (also "*circa situ*," "farmer-based conservation," "conservation *in hortus*"; Hughes 1998) has been used to distinguish the very different circumstances of conservation within altered agricultural landscapes (e.g., agroforestry systems, home gardens) outside natural habitats but within a species' native geographical range. Tree species have long been maintained, and hence conserved, in Mesoamerican agro-ecosystems. Hughes (1998) attributes the relative abundance of a wide range of *Leucaena* species in Mesoamerica, despite the loss of forest cover in most areas, to their management for several centuries by local communities and farmers in various ways and for various products and services. Indeed, he suggests that *L. salvadorensis* has been saved from extinction only because of its maintenance on farms, and there is also evidence for maintenance of

genetic diversity within these on-farm populations (Chamberlain et al. 1996). Other examples of traditional management systems that maintain tree species are given for the dry-forest zone of Central America (Kass et al. 1993) and the Mixtec region of Guerrero, Mexico (Casas and Caballero 1996). Only recently, however, have tree conservation strategies sought to capitalize on such practices (e.g., Halladay and Gilmour 1995), and consequently there is much to learn if we are to realize the potential of *circa situm* strategies and appreciate their limitations. In essence, *circa situm* conservation requires identification of compatibility between natural resource management systems found in native ranges of priority species (in this case trees) and conservation objectives. Human inhabitants within such areas are thus recognized as agents of conservation. *Circa situm* conservation aims to integrate conservation with prevailing socioeconomic trends, whereas the *in situ* and *ex situ* types of conservation aim to remove areas and species from the influence of those trends.

FOCUS OF THIS CHAPTER

The focus of this chapter is to question the assumption that trees in scattered tropical agro-ecosystems have no long-term viability and therefore no role in conservation. We discuss the role *circa situm* approaches could play in conserving the tree species of the Mesoamerican dry forest. We are concerned here with the prospects for such an approach rather than the precise prescriptions that will need to be formulated independently for each species and situation. We do not propose that *circa situm* conservation can replace *in situ* conservation. Rather, we argue that for highly threatened, disturbed and fragmented vegetation types, it could have an important complementary role. This may be through the conservation of particular species and genes not conserved *in situ* and/or the provision of additional habitat and the facilitation of gene flow between existing reserves. The potential gain is great if the many tree species that exist in such disturbed vegetation can be shown to be conserv-

able there through existing land-use practices, thus freeing scarce resources for the conservation of more critically threatened species that require more conventional, resource-intensive approaches. Provision of effective *circa situm* options requires multidisciplinary research to establish the general potential for the integration of conservation and development and more specifically which species are, or could be, sustainably conserved in such systems, from both biological and human management perspectives. The utility of particular species will not guarantee their long-term conservation if, for example, genetic diversity and regeneration within critical populations are compromised. Conversely, maintenance of genetic diversity and adaptive capacity is irrelevant if current management drastically alters population persistence. We examine results from our ecological, genetic, and socioeconomic research conducted within dry forests of Costa Rica, Honduras, and Mexico, which overlap principally within southern Honduras. In particular, we consider results for two timber species, *Bombacopsis quinata* (Jacq.) Dugand, Bombacaceae, and *Swietenia humilis* Zucc., Meliaceae, both of local socioeconomic importance, with superficially similar ecology and considered of conservation concern (see the section "The Case Study Species" later in this chapter). We consider general principles that may influence under what conditions and for what species *circa situm* conservation will prove effective and how it might be implemented. Finally we highlight research and training needs to explore further the feasibility and implementation of this approach.

RESEARCH IN SOUTHERN HONDURAS

Our research has had a common focus in southern Honduras (taken here as the lowlands and foothills of the departments of Choluteca and Valle), an area within the dry subtropical life zone (Holdridge 1987) with a distinct dry season from October to May. The southern Honduran lowlands are mainly occupied by cattle pasture and export agriculture (cantaloupe, watermelon,

sugarcane). The foothills show a fluctuating mosaic of land uses, with a predominance of basic grain production (*milpas*), alternating with fallows, some low-density cattle raising and, in more humid areas, low-intensity coffee production. The lack of mechanized tillage and the control of livestock movement typically allow natural regeneration of a highly variable range of tree species from seed banks, stumps, and newly arrived seed. From these species many farmers actively protect a subset of those most valued for a range of products (e.g., firewood and timber), reflecting a desire to optimize the use of available resources. Thus some trees reach, and remain in, reproductive maturity in *milpas* or pasture. Another subset of species remains as stumps and resprouts (up to 17,000 per hectare) simply because the removal cost is not perceived to produce adequate benefits (Barrance et al. in review[gb1]). Many of these trees reach reproductive maturity during fallow periods. The area therefore presents suitable conditions in terms of forest patchiness, the surrounding forest-agriculture interface, and socioeconomic circumstances to allow the synthesis of results across research disciplines.

THE CASE STUDY SPECIES

Bombacopsis quinata is a medium to large-size deciduous tree from Central America, Colombia, and Venezuela. It is hermaphroditic, largely self-incompatible (Sandiford 1998), and pollinated principally by the bat *Glossophaga soricina*. Its seeds are loosely attached to kapok, which facilitates their wind dispersal. Its timber is much in demand and has led to investment in large plantations, as well as planting by farmers within Central America. This species can be propagated by stakes and is used for living fences in parts of its range. Because of selective felling, deforestation, and destructive agricultural practices, this once common species is largely limited to isolated forest remnants. It is considered as endangered at the level of some populations (FAO 1986; Oldfield et al. 1998).

Swietenia humilis is a monoecious, medium-sized deciduous tree found along the Pacific

watershed of Central America and Mexico. It is pollinated by small butterflies, bees, and other insects, and its fruits contain large wind-dispersed seeds. Under controlled pollination it is self-incompatible (Boshier unpubl. data). Its sawn timber is highly valued, but commercial reforestation within its native range is hindered by the high incidence of attack by the shoot borer *Hypsipyla grandella*. Where populations of *S. humilis* and *S. macrophylla* are sympatric, hybridization is known, and their distinctness as species has been questioned (Styles 1981). Populations over much of the species' range have been reduced and fragmented, leading to its listing in 1973 in appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and classification as "vulnerable" by the International Union for Conservation of Nature and Natural Resources (IUCN) (Oldfield et al. 1998).

GENETIC DIVERSITY AND ITS MAINTENANCE

Evidence to date shows that a high proportion of tropical tree species are naturally outcrossing (e.g., Bawa et al. 1985), with associated risks from inbreeding such as reduced fertility, growth, and susceptibility to pests or diseases (e.g., Griffin 1990). Maintenance of genetic diversity is therefore vital for the long-term viability and adaptability of populations of many tree species. Reproductive biology, levels of genetic diversity and gene flow, and extent of localized adaptation have been studied within the native range of both *B. quinata* and *S. humilis* (see Sandiford 1998; White 1998; Billingham 1999; White et al. 1999; Boshier and Billingham 2000; and White and Boshier 2000 for methodologies and results). Remnant stands of secondary dry forest, confined principally to hillsides and remnant trees in pastures in the Punta Ratón region of the Honduran Pacific alluvial coastal plains, near Choluteca, were used, along with control plots in more continuous forest in both Honduras and Guanacaste, Costa Rica. The fragments varied in size (7–150 trees) and degree of spatial isolation (1–4.5 km). For both species, genetic markers (allozymes or microsatellites)

showed that at the degree of isolation studied, fragmentation did not impose a genetic barrier between remnants, direct measures of current mating patterns giving altered and proportionately higher levels of interfragment pollen flow over longer distances. Common to both controls and fragments for both species was a predominance of near neighbor mating (<300 m of the maternal tree). A large proportion of pollen donors were, however, from outside each fragment across the sampled area, indicating an extensive network of gene exchange at this spatial scale (16 km^2). So in two fragments (22 and 44 *S. humilis* trees), 62 percent and 53 percent, respectively, of pollen donors were from within the fragment, whereas 24 percent and 34 percent were from distances greater than 1.5 km and 3.6 km, respectively. There was no evidence for increased inbreeding in fragments, with both species continuing to show high levels of outcrossing even in the smallest fragments. Indeed, an *S. humilis* tree, "isolated" by 1.2 km from the nearest flowering trees, showed 100 percent external pollen sources, with more than 70 percent from trees in the main area of forest (>4.5 km), in accord with the species' self-incompatibility and in contrast to predictions that spatially isolated trees are more likely to deviate from random mating and receive pollen from fewer donors (Murawski and Hamrick 1991).

Such enhanced levels of long-distance gene flow into smaller fragments, also seen in *Spondias mombin* L. (Nason and Hamrick 1997), will potentially restore or maintain genetic variation in populations of these species within the modified environment. This contrasts with traditional views of the genetic effects of fragmenting populations whereby increases in spatial isolation and population size reduction have been considered to reduce gene flow between fragments (e.g., Saunders et al. 1991). Although the genetic effects of fragmentation are complex, for some tree species under fragmentation, pollination may occur over much greater distances than previously considered. The distance given here for *S. humilis* is more than ten times greater than that considered as distant by Chase et al.

(1996) and more in accord with distances previously identified by zoologists (e.g., Janzen 1971; Frankie et al. 1976). There will, however, be a distance beyond which genetic isolation will occur, with associated problems for population viability and adaptation (see Young et al. 1996). Although determination is experimentally problematic, thresholds will vary between species depending on pollinator characteristics, availability, the specificity of the tree-pollinator relationship, and the presence and strength of any self-incompatibility mechanism. Self-compatible species that normally show some level of outcrossing (e.g., Murawski and Hamrick 1992) are likely to show increased levels of inbreeding at much shorter distances of separation (lower thresholds) than self-incompatible species.

PATTERNS OF DISTRIBUTION AT THE SPECIES LEVEL IN SOUTHERN HONDURAS

Two rapid botanical surveys (RBS) were undertaken to determine the tree and shrub species composition of the southern Honduras agroecosystem. The RBS technique follows the plotless or "unmeasured" sampling described by Hawthorne and Abu-Juam (1995), with minor modifications. A first, village-level survey consisted of species inventories in four rural communities selected nonrandomly to reflect regional variation across socioeconomic and environmental gradients. Within each community 20 households were selected along a socioeconomic gradient and a random subset of land units selected for inventory from those farmed by these families. Land unit categories followed definitions given by householders. In each land unit species were scored as present or absent, a species being considered present if it could be identified as a woody individual, such that live stumps were included. A second, forest-level survey inventoried tree diversity in a subset of forest patches in southern Honduras. Forests were selected nonrandomly to maximize geographical spread and biased toward the few remaining relatively large forest areas (i.e., >50 ha). All forests appeared to be largely, if not completely, composed of secondary regeneration of varying ages. Pri-

mary dry forest apparently no longer exists in southern Honduras, although it is probable that some older trees are remnants of much older forests (White et al. 1999). The forest cover that does remain is composed of stands of trees of as little as 2 ha in area and rarely greater than 20 ha, areas usually considered too small to be of value to conservation practitioners.

The two species differ in terms of where they occur. *S. humilis,* one of the most commonly encountered species (only 7 of more than 250 woody species identified were found more often), was found as frequently in farmland as in forests (table 16.1). In contrast, *B. quinata* was found less than *S. humilis,* but much more often in forests than on farmland. Both species were more common in fallows, fields, and pastures (table 16.1), which are typified as naturally regenerating management systems, than in home gardens and orchards, where planting is more typical. Both species also showed considerable variation in occurrence between the four communities (table 16.1), probably owing to a combination of biological and anthropogenic factors, although in southern Honduras a lack of undisturbed dry forest where the two species are sympatric makes it difficult to distinguish the principal factors. Two of the larger forests in Choluteca suggest that both species may show similar levels of occurrence in mature forest. In relatively undisturbed semievergreen forest on Cerro Guanacaure *B. quinata* is common, as is *S. humilis* (Gordon pers. obs.). On Cerro Las Tablas, one of the most mature secondary forests in the area, many mature trees of *B. quinata* and *S. humilis* occur (densities up to 17.0 and 9.6 per ha, respectively). In Costa Rica both species occur at similar levels in more undisturbed areas of protected dry forest (e.g., Lomas Barbudal, Playa Nancite mature forest area, table 16.2). Thus, we have grounds to speculate that in southern Honduras the relatively low occurrence of *B. quinata* on farms results, at least in part, from unfavorable management practices (further discussed later in this chapter) rather than a natural tendency toward low densities.

TABLE 16.1

Occurrence of Bombacopsis quinata *and* Swietenia humilis *by Land Use and Community in Southern Honduras,*
1999

LAND USE	TOTAL NUMBER OF SAMPLES	PERCENTAGE CONTAINING *B. QUINATA*	PERCENTAGE CONTAINING *S. HUMILIS*
Forest	48	27	52
Farm	105	11	56
Fallow, field, and pasture	58	16	67
Home garden and orchard	47	4	36
COMMUNITY/DEPARTMENT			
San Juan Arriba, Choluteca	25	12	4
Agua Zarca, Valle	38	5	74
San José de las Conchas, Choluteca	24	17	46
Los Coyotes, Choluteca	25	16	64

The common on-farm occurrence of *S. humilis* cannot be explained simply by farmer preference (see the section "Patterns of Management of *B. quinata* and *S. humilis* in Southern Honduras" later in this chapter), given that natural regeneration has to occur before farmers can aid its recruitment. *S. humilis,* like its close relative *S. macrophylla* King, undoubtedly thrives in regimes of heavy disturbance (Snook 1996) as provided by traditional agriculture in southern Honduras. Once established, it can survive for decades in the closed forests that may form when fallows are abandoned. Gerhardt (1994) showed that germination of *S. macrophylla* in Guanacaste was not affected by differences in light levels and could establish equally well in pasture and young secondary forest. A study of localized adaptation in both Costa Rica and Honduras, with seedlings in pasture and secondary and mature forest, showed zero survival of *B. quinata* at three years of age in any of these land uses, compared with 15–60 percent for *S. humilis* (Billingham and Boshier unpubl. data). This is borne out at Playa Nancite (Santa Rosa National Park), where *S. humilis* regeneration in adjoining areas of abandoned degraded pasture (table 16.2) is abundant and that of *B. quinata* nonexistent. In contrast to other predictions and some studies (e.g., Ghazoul et al. 1998), seed production of *S. humilis* (table 16.3) was also more reliable and higher in disturbed environments than in closed forest, whereas that of *B. quinata* showed no differences by forest type, except at Playa Nancite, where it was low in all four fruiting seasons. During this period 13 percent of the *B. quinata* trees at Playa Nancite died from apparent old age/creepers/wind. Given the lack of any regeneration, the future of *B. quinata* at this site, although it is within a protected area, seems insecure. Indeed, fires and tree felling evident over four years (table 16.2) confirm threats to both species, even within "protected" areas.

We can conclude that species are likely to be scattered nonrandomly in the various land-use types and communities of an agro-ecosystem and that these distribution patterns must be known for *circa situm* conservation options to be assessed and for strategies to be devised and implemented. The fact that 76 percent of the species identified in the surveys were found in

TABLE 16.2

Mortality of Bombacopsis quinata *and* Swietenia humilis *Trees over a Four-Year Period at Sites in Costa Rica and Honduras*

SPECIES	SITE	COUNTRY	SITE TYPE	AREA STUDIED (HA)	NO. OF TREES	MEAN DBH IN CM (S.D.)	PERCENTAGE OF TREES CUT/DEAD
B. quinata	Lomas Barbudal Biological Reserve	Costa Rica	Protected area	25	61	69.2 (25.5)	0/6.6[a]
	Playa Nancite, Santa Rosa National Park	Costa Rica	Protected area/mature	28	62	71.6 (42.8)	0/12.9
	Playa Nancite, Santa Rosa National Park	Costa Rica	Protected area/regeneration	3	0		
	Punta Ratón, Choluteca	Honduras	Private forest/farm	240	172	54.2 (20.9)	8.1/5.8[a]
S. humilis	Lomas Barbudal Biological Reserve[b]	Costa Rica	Protected area	25	57	47.4 (17.0)	0/14.0[a]
	Playa Nancite, Santa Rosa National Park	Costa Rica	Protected area/mature	28	156	46.7 (16.0)	0/5.8
	Playa Nancite, Santa Rosa National Park	Costa Rica	Protected area/regeneration	3	23	20.3 (3.7)	0/0
	Cerro Las Tablas, Choluteca	Honduras	Private forest/farm	68	105	41.9 (14.0)	25.7/1.9
	Punta Ratón, Choluteca	Honduras	Private forest/farm	240	75	37.9 (14.1)	5.3/5.3

Note: Mortality occurred through either natural causes or human intervention. dbh = Diameter at breast height.
[a]Killed by 1994 fire.
[b]Possibly *S. macrophylla* or *S. macrophylla* × *S. humilis*.

TABLE 16.3

Percentage of Bombacopsis quinata *and* Swietenia humilis *Trees per Year with Moderate to Heavy Seed Production at Various Sites*

SPECIES	SITE[a]	SAMPLE SIZE[b]	1994 (%)	1995 (%)	1996 (%)	1997 (%)	1998 (%)
B. quinata	Lomas Barbudal	>45	70.6	66.7		68.9	
	Playa Nancite	>30	61.5	56.4	55.2	31.5	
	Punta Ratón	>105	76.4	61.9	73.3	90.1	
S. humilis	Lomas Barbudal	>46		12.1	0.0	16.3	20.8
	Playa Nancite	>143		32.9	11.0	18.4	17.2
	Cerro Las Tablas	>61		48.7	12.6	29.5	44.6
	Punta Ratón	>45	28.0	36.1	42.2	49.1	33.3
	Comayagua	30	66.7		60.0	83.3	76.7

Note: Moderate to heavy seed production for *B. quinata* is more than 20 seed capsules; for *S. humilis,* more than 10 seed capsules.
[a]Sites are ranked for increasing degree of disturbance: Lomas Barbudal, a relatively undisturbed area within a reserve; Playa Nancite, area of the Santa Rosa National Park with disturbance; Cerro Las Tablas, forest with remnant trees and secondary regeneration; Punta Ratón, remnant trees and secondary regeneration in small fragments and pasture; Comayagua, planted trees by roadside.
[b]Sample size varies from year to year owing to mortality.

agro-ecosystems (i.e., not limited to forest) emphasizes the potential for on-farm trees to make a contribution to conservation strategies. This is borne out by Harvey and Haber (1999), who found 190 tree species in 237 ha of upland pasture near Monteverde Reserve in Costa Rica. Assuming that greater occurrence means greater conservation potential, the potential for different sites to be the foci of successful *circa situm* conservation will vary. For example, taking the distribution evidence in isolation, the Agua Zarca community has a high potential for conservation of *S. humilis* but very little for *B. quinata* (table 16.1). *S. humilis* appears ecologically suited to *circa situm* conservation "through use" on these farms, where tillage is minimal, livestock movement controlled, and fallows sporadically allowed. *B. quinata,* however, appears less suited, being much more susceptible to reductions in forest cover, although even for such species that appear to be predominantly "forest species," the relatively few individuals on farms may be important for gene flow between forests (see the section

"Genetic Diversity and Its Maintenance" earlier in this chapter). Evidently conservation strategies for these two species need to be very different.

PATTERNS OF MANAGEMENT OF *B. QUINATA* AND *S. HUMILIS* IN SOUTHERN HONDURAS

A study in the same small farmer communities where village-level species inventories were taken (see previous section) shows that, in addition to differences in reproductive biology, ecology, and local distribution, *B. quinata* and *S. humilis* differ in management by local people. In general, within these communities, people mentioned that many of the same tree species under active protection are also used for timber, although there was a marked contrast between the frequency of active protection for *S. humilis* (second only to *Cordia alliodora*) and that of *B. quinata* (table 16.4).

A focal group meeting in Los Coyotes, where species were rated for present and past use, also revealed marked differences between the two species in terms of their past and present socio-

TABLE 16.4

Principal Tree Species Mentioned as Used for Timber or Actively Protected within Agricultural Areas as
Sources of Timber or Posts in Four Communities in Southern Honduras in 1998

SPECIES USED	PERCENTAGE OF INTERVIEWEES[a]	SPECIES PROTECTED	PERCENTAGE OF INTERVIEWEES[a]
Cordia alliodora	84.8	*Cordia alliodora*	38.0
Bombacopsis quinata	20.3	*Swietenia humilis*	22.8
Enterolobium cyclocarpum	19.0	*Lysiloma* spp.	20.3
Albizia saman	16.5	*Enterolobium cyclocarpum*	10.1
Lysiloma spp.	16.5	*Albizia saman*	10.1
Swietenia humilis	16.5	42 other tree species, including *Bombacopsis quinata*	1.3–6.3 each
Calycophyllum candidissimum	15.2		
Cedrela odorata	15.2		
Conocarpus/Rhizopora spp. (mangrove)	15.2		
Simarouba glauca	11.4		

Note: The four communities were San Juan Arriba, Agua Zarca, San José de las Conchas, and Los Coyotes.
[a]Figures given as a percentage of 79 interviewees.

economic niches. Previously ("in the time of their grandfathers") *B. quinata* was almost the only species used for timber; however, its exploitation to supply furniture workshops in the nearby town of El Triunfo led to a large decrease in its numbers in the area. Farmers now use a broad range of previously little-used species, including *S. humilis,* others of which (e.g., *Guazuma ulmifolia* Lam) have far inferior timber value. Farmers also contrasted previous management during field clearance, whereby many valuable timber trees were felled and burned on-site or used for firewood, with current practices, whereby valued trees are protected and only timber offcuts used as firewood.

Apparently because of a scarcity of off-field tree resources, resulting from high human population densities and growth rates, many farmers, by favoring certain species during clearance, protect and manage valued trees within their fields to meet their subsistence needs and as a naturally regenerated cash crop. Furthermore, stumps and seeds survive regardless of felling, as stump removal is difficult and mechanical tillage is rarely practiced owing to topographical constraints. Farmers are also adept at controlling grazing, a prerequisite of successful maize cultivation, so that livestock does not necessarily prevent regeneration and indeed may facilitate it (Janzen 1986a). Patches of fallow and forest within the mosaic continue, despite their small size, to be propagule sources of many of the species found in these fields. Up to certain limits the benefits of maintaining trees outweigh negative effects (e.g., shade) that farmers perceive the trees to have on crops (Barrance et al. in review[gb2]). This form of management represents a rational response to resource scarcity rather than a desire to conserve biological diversity per se.

The relatively low frequency of protection of *B. quinata* in fields (tables 16.1, 16.4) does not appear to result from its being less valued than *S. humilis* or from differences in farmers' perceptions of tree/crop interactions (no important negative interactions were reported by farmers between either species and crops). A more probable explanation is the relative scarcity and patchy distribution of *B. quinata* natural regeneration, particularly in fields. Farmers protect it where they find it but find it much less frequently than *S. humilis*. The current scarcity of *B. quinata* appears to stem, at least in part, from its past overexploitation, prior to the self-imposition of tree management controls by farmers as a response to scarcity and its ecological characteristics. *S. humilis* differs in that it appears to have been rather less valued than *B. quinata* and as a result largely escaped overexploitation. It now benefits from the current practice of active protection of the remaining valued species and its ease of regeneration. However, despite farmers' clear preference for both species and their promotion by extension agencies, neither species is commonly planted. This probably reflects the cost of planted trees, compared with "free" natural regeneration and possibly the greater risks to planted trees from cattle that are periodically introduced into *milpas* to eat crop residues.

Both *B. quinata* and *S. humilis* also occur in forest patches and pastures within larger estates (e.g., Cerro Las Tablas), although the management systems and socioeconomic conditions are very different from those found in the small farmer communities. In both pasture and forest remnants on these estates there appears to be a continued decline due to piecemeal felling (table 16.2), possibly as the landowners do not face the same conditions of overall resource scarcity that motivate small farmers to nurture natural regeneration. In addition, conditions in many pastures (they are normally destumped on clearing and then periodically burned to encourage grass development) are less propitious to tree regeneration than those on steeper land under subsistence production systems.

CONCLUSIONS

In situ conservation of the remaining areas of mature Mesoamerican dry forest is likely to be the most effective and efficient means of safeguarding a substantial part of the threatened specific and subspecific tree diversity of this forest type. However, many species and populations of concern are not well or not at all represented in such areas and require a different approach. Habitat restoration may also have a role, but the socioeconomic conditions that make this a possibility are uncommon, and costs may be prohibitive. *Circa situm* conservation has much to offer in these situations but also significant limitations; for it to be effective, its application must be guided by information from multidisciplinary research of the type discussed in this chapter. As the contrasting examples of *B. quinata* and *S. humilis* show, we cannot permit ourselves the luxury of assuming that all species are likely to persist in an agricultural mosaic until this is proved otherwise. However, on the basis of such evidence, neither should we underestimate the capability of many species to persist in large numbers in these agro-ecosystems under current practices, as this could lead to the misdirection of limited conservation resources toward species not under threat. *Circa situm* conservation of tree diversity is already occurring, albeit as a side effect of resource-poor farmer practices. Given apparently similar conditions across large parts of the dry zone foothills of El Salvador, Guatemala, and Nicaragua, *circa situm* populations may represent a considerable conservation resource. If conservation planners were to take this resource into consideration, how many species currently assumed to be threatened by habitat loss might prove to be thriving (Vandermeer and Perfecto 1997)?

An enhanced future role for *circa situm* conservation in the dry tropics of Mesoamerica will depend principally on the existence of species of conservation importance in the agro-ecosystems and forests in question. Species of widespread occurrence whose status does not appear about

to change and species well represented in adequately protected areas should not attract more of the scarce resources available for conservation. Are agro-ecosystems likely to contain the types of species we wish to conserve? To answer this question we offer the following generalizations. Agro-ecosystems are highly disturbed in comparison with the original vegetation, and the dominant species are likely to be either "pre-adapted" to disturbance (i.e., weedy, pioneer, and/or coppicing) or positively selected by farmers (or both). The long-distance dispersal propagules of these typically pioneer type species, coupled with ever increasing disturbance, ensure that they will generally not be conservation priorities. One priority must therefore be the relatively few species with much narrower distributions, endemic to areas now completely converted to agro-ecosystems (e.g., *Leucaena salvadorensis;* Hellin and Hughes 1993). For these species *in situ* is de facto, not a conservation option. A second priority would be those species and species assemblages whose continued *in situ* conservation would be enhanced by their conservation *circa situm.* Here conservation in agro-ecosystems between reserves increases habitat area, connectivity, and population sizes.

Knowledge of which tree species occur where says nothing about the species' viability within an agro-ecosystem. The effects of fragmentation and the regeneration of secondary forest patches are likely to be dynamic, and to ensure that we deal with populations with a long-term future, we must carefully consider the reproductive and regenerative capacities of priority species and the perpetuation of management practices that allow natural or artificial regeneration. Yet knowledge of the effects of human interventions (e.g., logging, fragmentation) on tree gene pools is relatively poor. We do not know at what stage forest fragments become genetically isolated; nor do we understand the consequences of gene flow between managed and remnant natural populations. Selection pressures exerted by farmers, intentionally or unintentionally, at species, population, and within-population levels are also

little understood, and their compatibility with conservation objectives are unclear.

For the species studied here and others elsewhere (e.g., chapter 3), it appears that descriptions of remnant trees and forest patches as "isolated" or "living dead" (Janzen 1986b), with little or no conservation value, may be misleading and more human perception than a true reflection of actual gene flow and any biological reality. It is more realistic to view remnant forests not as islands but rather as existing within a mosaic of land uses that differ in their capacity to provide habitat or permit movement for any organism. In this sense it is important to recognize the complementary habitat role that maintenance of trees on farms is already playing to *in situ* dry-forest conservation. With adequate gene flow and seed production, some remnant forest patches and trees may be important contributors to connectivity and conservation, both *in situ* and *circa situm,* more generally. The capacity of some pollinators to move long distances has been shown, although the potential to move between patches will depend on their behavioral response to such a mosaic. Some bat species move preferentially down forest tracks and pathways (e.g., Estrada et al. 1993). Whereas some bee species are generally restricted in their movements (Powell and Powell 1987), most show some long-distance movements, and some will move more than 4 km across agro-ecosystems between forest patches (e.g., Frankie et al. 1976; Raw 1989). With a range of nonspecialist pollinators, pollen flow dynamics in *S. humilis* are probably far less susceptible to habitat disturbance than tree species with more specialist pollinators, although *Ficus* species, with species-specific wasp pollinators, may form extensive metapopulations in fragmented landscapes (Nason et al. 1998). Changes in pollinator assemblages in fragmented landscapes may strongly affect patterns of gene flow and reproduction in remnant tree populations. Concerns that declines in pollinator populations in such agro-ecosystems may eventually limit tree reproduction require monitoring of numbers, as well as

evidence for pollinator limitation (Allen-Wardell et al. 1998).

Active tree protection by farmers is strongly dependent on their perceptions of the trees' potential value to them and on other community members' recognition of their ownership rights to the trees. However, the *B. quinata* case clearly demonstrates that even if a species is valued its conservation is not guaranteed (Gordon et al. 2003). In fact, *use value* may lead to a species' decline until a threshold is reached whereby farmers are obliged to modify their practices if they wish to ensure a continued supply. It appears that at this point the rate of decline of many species tends to level out as mature individuals are managed on a more rational basis and removals replaced by natural regeneration that receives active protection by farmers. However, if regeneration is poor in agro-ecosystems, as appears to be so for *B. quinata,* it may continue to decline and disappear. Alternatively farmers may switch much of their demand for a particular product to another species.

Hughes (1998) suggests that *circa situm* conservation is probably limited to species that tolerate disturbance and are actively used and preferred by farmers. However, although it indeed favors disturbance-tolerant species, the examples from southern Honduras and those of Kass et al. (1993) suggest its applicability to a broader range of species than just the actively preferred ones. However, survival of nonpreferred species may to a large extent depend on the technology level of land management practices; for example, a larger number of stumps of nonvalued coppicing species is allowed to persist in low-intensity swidden agriculture of southern Honduran hillsides than in ploughed fields of the central valleys of Oaxaca, Mexico. It would be unfair to promote conservation initiatives at a landscape scale across the Mesoamerican dry zone that prevent poor farmers from adopting more profitable practices, although in many parts there are significant biophysical constraints (e.g., topography, resource availability) to those seeking to intensify production. At least in southern Honduras there appears to be little immediate risk of agricultural

intensification leading to massive extinctions (cf. Janzen 1986b). *Circa situm* conservation strategies are more likely to be effective when planned on a species-by-species basis, within the farming systems context of a particular area, with resources focused on target species so as to minimize negative socioeconomic effects. The "area" or "management unit" over which such strategies would operate would be measured in numbers of participating households or land units in which target species conservation is successfully incorporated into existing, or acceptable, farm management practices.

The prospects for *circa situm* conservation for *S. humilis* appear good, but less so for *B. quinata,* and it is ironic that of these two species the former is the one afforded more protection by international convention (appendix II of CITES). However, for it or similar species, some monitoring may be advisable to ensure that its niche in the farming system is maintained in the face of changing socioeconomic trends. Current trends, at least in some areas, involve an iterative process of property subdivision and elimination of forest and fallow areas, with a corresponding increase in numbers of trees in grain fields and pastures. It is possible that farm size will eventually reach a threshold beyond which farmers are unwilling to divide their property further among their heirs, and increased dependence on off-farm income sources will divert pressure from land and tree resources, leading to a stable situation. However, it is unclear at what point a threshold will be reached, to what degree this will be influenced by available off-farm income, and whether farm size reductions will lead farmers to increase tree densities on farms to satisfy tree product needs or reduce them to increase basic grain yields. The challenge offered by *circa situm* conservation is to adapt to these changes over a relatively wide geographical area rather than seek to halt such processes within a confined area such as a reserve. To this end there is a need to raise awareness among development professionals of the value of natural regeneration as both conservation and socioeconomic resources. Efforts to

promote planting and the use of multipurpose or exotic tree species (or both) at the expense of natural regeneration may have deleterious effects not only on conservation but also on farmer welfare. This would be evident through a reduction in the range of forest products normally available to farmers from natural regeneration and an increased cost of tree establishment.

RECOMMENDATIONS FOR RESEARCH AND TRAINING

The data requirements to answer the questions raised in this chapter are considerable, and realization of the potential of *circa situm* conservation requires a range of actions. For *circa situm* conservation to have anything but a minor role in the conservation of tropical forest resources, priorities among species and forest types will need to be set and shortcuts found. We end this chapter by listing recommendations for research and dissemination priorities for the further development of *circa situm* conservation. The list is neither prescriptive nor exhaustive and reflects primarily the situation as we understand it in southern Honduras. It is presented to illustrate the types of information demands that *circa situm* prescriptions will make. However, overriding such specificity is the need for conservation planners, more accustomed to *in situ* methods, to consider the possibility that populations of trees found outside protected areas have a role in the conservation of biodiversity. This in turn will require the direct involvement of development organizations in biodiversity conservation and an effective two-way communication between them and "traditional" conservation organizations to ensure both conservation and development benefits.

· Species distribution patterns need to be established not only at a broad level, to identify species that are rare and therefore a conservation priority, but also at a finer scale, to identify which species are not conserved *in situ* and which land uses within an agro-ecosystem are most appropriate for each species.

· Targeted tree populations need to be described in terms of size class distributions with respect to reproductive maturity. We cannot assume that a *circa situm* population contains the same proportion of reproductively active individuals as an *in situ* population.

· Fragmentation thresholds for gene flow need to be determined and the possible selection pressures exerted by farmers further elucidated.

· Species of conservation concern within agro-ecosystems need to be understood and classified in terms of farmers' perceptions of their value. Species guilds need to be defined, combining biological and socio-economic variables to reduce, by extrapolation, the information demands of *circa situm* conservation.

· Education and training curricula need to be broadened so that more conservation biologists recognize the potential role of on-farm conservation and more rural development practitioners recognize their role in biodiversity conservation.

· The complementary habitat benefits of different agro-ecosystems for conservation need to be evaluated, recognized, and promoted.

· Rapid rural appraisal techniques (see Chambers and Guijt 1995) need adapting and testing for their ability to provide adequate and cost-effective information for *circa situm* management.

· Strategies need to be explored on a species-by-species basis to ensure effective use of resources and minimum disruption to farming practices.

· The costs/benefits to farmers of actively promoting conservation through a *circa situm* approach need to be carefully balanced and, if necessary, forms of compensation considered.

· Effective systems for long-term monitoring of the biological (e.g., pollinator levels, regeneration levels) and socioeconomic

dynamics (e.g., management practice changes) of conservation need to be established.

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