Global Forest Restoration: A Review

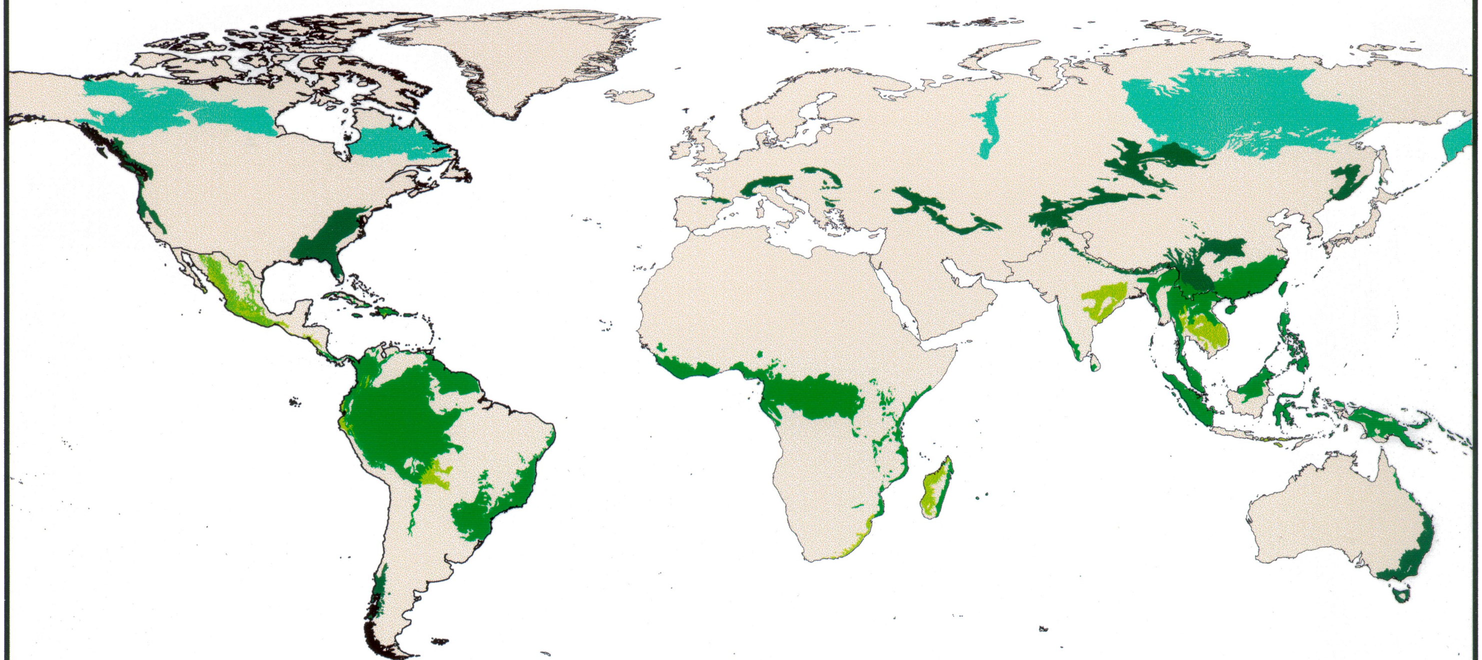

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Table of Contents

List of Tables

List of Figures

List of Plates

GLOBAL FOREST RESTORATION: A REVIEW

Introduction

The Global 200 Strategy ambitiously seeks to conserve the variety of species, ecosystems, and ecological and evolutionary processes that sustain life on earth. Toward this end, the World Wildlife Fund has identified over 200 ecoregions (Global 200) recognized for their high conservation priority, and calls for concentrated conservation planning in these regions. Not surprisingly, of the wide array of ecosystem types included in the Global 200, forest ecosystems constitute the majority. In fact, 87 of the 136 terrestrial ecoregions fall into one of five Major Habitat Types – Tropical and Subtropical Moist Broadleaf Forests, Tropical Dry Forests, Tropical and Subtropical Conifer Forests, Temperate Conifer and Broadleaf Forests, and Boreal Forests and Taiga.

Among terrestrial ecosystems, forests tend to exhibit some of the highest levels of diversity, ecological complexity, biomass, and species endemism (Perry 1994; Jepma 1995). Additionally, humans associate a wide array of values with forests and forest communities; these values range from recreational to economic and from ethical to spiritual (Brown and Pearce 1994; Perry 1994). These values, in turn, influence human behavior in general and the way humans interact with forests. Thus, rooted in these values are both the destruction and conservation of forests. Historically, human interaction with forests has been predominantly destructive (Perry 1994). Among the most important of these include:

(1) conversion due to urbanization, agriculture, ranching, and mining,
(2) commercial exploitation of timber and wood pulp,
(3) local exploitation for firewood,
(4) human-altered disturbance regimes (e.g., fire),
(5) introduction of exotic species, and
(6) construction of infrastructure facilities – particularly roads (see Jepma 1995).

Interest in large-scale conservation is relatively new compared to the long history of resource extraction and natural landscape conversion. Consequently, the world has suffered tremendous losses in biodiversity, and the degradation of large areas of forest in many different ecoregions continues. Effective, long-term forest conservation in the face of the many ongoing threats is critical to the success of the Global 200 Strategy.

The purpose of this report is threefold: (1) discuss the concept of forest restoration from a conservation biology perspective; (2) outline the ecological characteristics, technical constraints, socio-political and economic influences, and overall restoration principles relevant to the Global 200 major habitat types and associated realms; and (3) place forest restoration within the larger context of worldwide forest conservation.

Forest Restoration

Prior to discussing forest restoration (or more accurately reforestation) as it is being implemented throughout Global 200 ecoregions today, it is first important to define what it is we are trying to restore and then briefly review the important terminology pertaining to the topic. It is fundamental to this discussion that forest ecosystems be recognized as not simply groups of trees located in the same general vicinity. Rather, they are complex systems with complex ecologies. Forest ecosystems obviously include trees, but they also include understory vegetation, soil and soil organisms, hydrologic cycles, nutrient cycles, successional and evolutionary processes, and the host of vertebrate and invertebrate species – all of which interact in a rhythmic dance over time and space. In terms of restoration then, reestablishing a forest is much more than just replacing a community of trees – it means reestablishing a functional forest with all its parts and processes. Unfortunately, we almost never have enough knowledge on the original composition and ecology of any ecosystem, including forests, to attempt total replication. Even if we had all the knowledge necessary, it is incredibly difficult technically to restore most forests over large geographic areas. Most restoration projects can only set the stage that allows for natural regeneration to complete the process. Below, we review a collection of restoration terms that are important in laying the groundwork upon which this review is built.

Restoration Terms

Two popular terms found in the literature concerning the human-induced reestablishment of forests are ***reforestation*** and ***afforestation***. Although often used interchangeably, it is generally agreed that ***reforestation*** refers to the managed reestablishment of forests on lands that have contained forests within the last 50 years and ***afforestation*** describes either the managed reestablishment of forests on lands where forests had been removed at least 50 years ago, or the directed establishment of forests on lands where forests would not naturally be found. For the purposes of this report, we are interested only in the establishment of forests on previously forested land or on land that would under natural conditions support forests regardless of how long it has been since the forests were removed from the landscape. For clarification and simplicity, we prefer to discuss the various forms of forest reestablishment under one umbrella term – ***reforestation***. This will apply both to lands where forests have recently been removed as well as to lands where forests have been absent for longer periods, even centuries. Figure 1 shows a simple model of the various alternatives (or pathways) reforestation can take. All have been identified as forest restoration in the popular literature and in some policy documents, but as we will describe in later sections of this review, most forest restoration being actively pursued today throughout most ecoregions in the world are not true restoration but different forms of reforestation which differ substantially in their capacity to contain or further native biodiversity.

Pathway #1 – Natural Regeneration

After primary forests are removed or degraded, the return of forests to those sites can take one of several trajectories. ***Natural regeneration*** is one pathway and is the most common worldwide. As the name implies, natural regeneration entails a measure of restraint from active management in order to allow naturally occurring successional processes to do the work of revegetation (Lamb et al. 1997). In some regions under certain conditions, nature can replicate

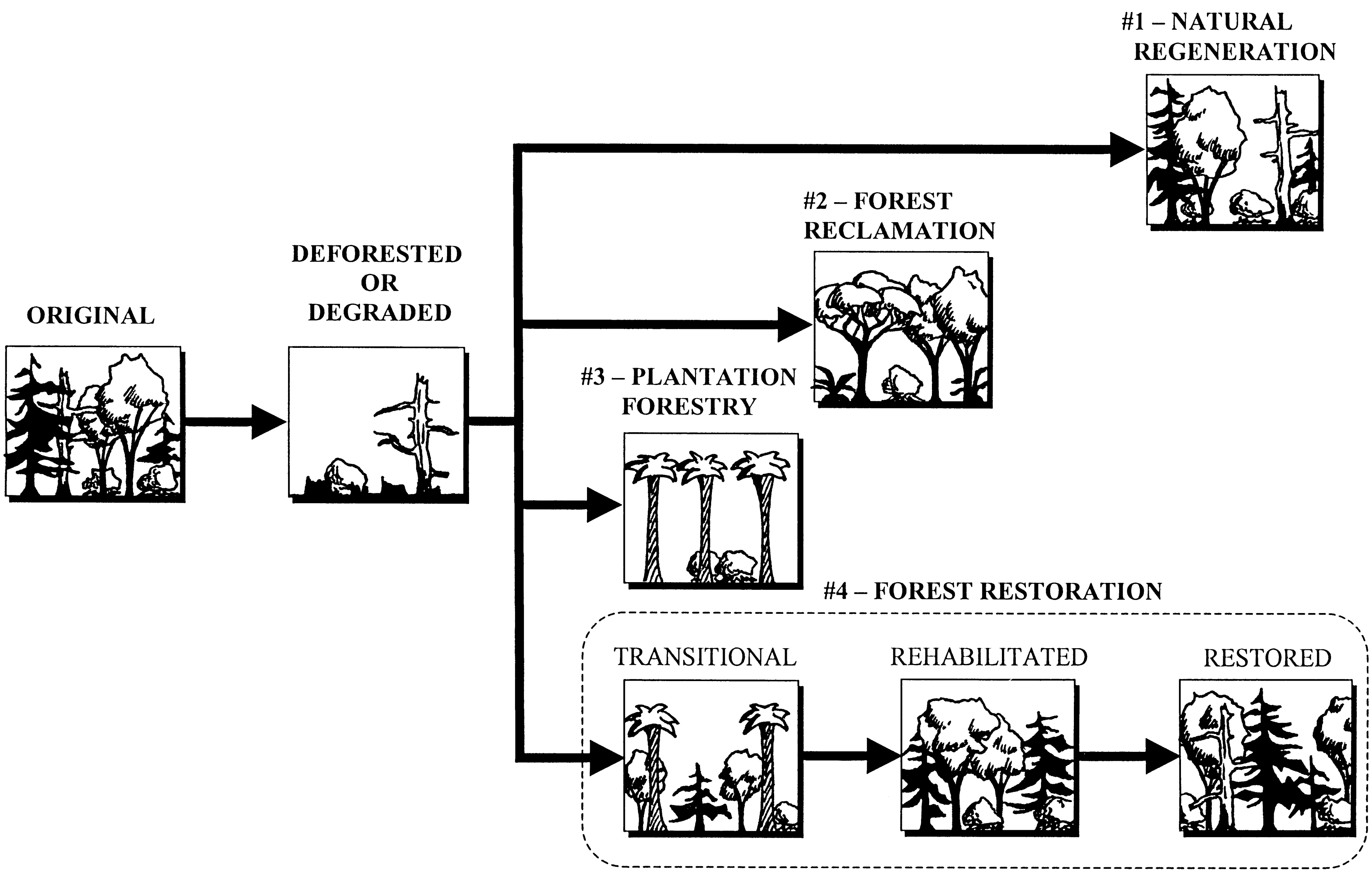

Figure 1. Flow chart of possible reforestation pathways. Starting at the deforested or degraded state, relative horizontal position approximates level of ecological integrity.

composition, structure, and function well. Unfortunately, in other places or under other conditions, natural regeneration is much more difficult or impossible. In most cases, forest ecosystems that recover naturally after human disturbance develop into different systems from the original forest. Some forests never return to what they once were, others require a great deal of time and similar physical conditions under which the original forest evolved to return to the same type of forest. In general, the larger the area and the more severe the deforestation carried out by humans, the more difficult it is for natural regeneration to result in approximating its former state. The issue of time is an important consideration since the frequency of human extraction of forest products in most forest systems never allows the forests to reach their end state. For example, in much of the Pacific Northwest of the U.S., logging rotations often call for return of forest harvest in the same location every 40-50 years. Under this cutting regime, these forests will never approach their late seral characteristics which take hundreds of years to acquire in this region.

For this paper, we elect to focus greater attention on the remaining possible trajectories – those that require significant human energy to initiate and maintain. That is not to say that natural regeneration does not play some role in these other efforts – it does. However, the magnitude of the reforestation workload that nature supplies varies widely and depends on the ecosystem type as well as the nature and geographic extent of the destruction or degradation of the original system. The reality is that all reforestation projects involve natural regeneration to some extent. How much is highly variable. Under some circumstances, natural regeneration can be enhanced by active management (planting, removal of exotics, etc.), but these activities are expensive in time and money. Hopefully, projects that begin with active management can shift emphasis away from these costly activities as soon as possible in order to let natural regeneration do most of the work. However, by definition, "natural regeneration" means no active management.

Although highly desirable for many reasons, natural regeneration is subject to certain ecological factors (including long recovery times, availability of propagules, predation, dispersal agents, soil properties, drought, competition, symbionts, and fire) that make it difficult or virtually impossible to deoend upon (Lamb et al. 1997). Even so, naturally regenerated forests have been documented in many places even after major impacts. For example, in Venezuela and Colombia, forests had suffered huge wildfires about 250 years ago (Saldarriaga and West 1986), while in Puerto Rico, a long history of intensive agriculture characterized the region until about the last 50 years (Birdsey and Weaver 1982). Ideally, all restoration projects should work toward minimal management and maximum natural regeneration, but the details are very ecoregional, and even site, specific.

Pathway #2 – Forest Reclamation

Reclamation is a term often used by ecological engineers, referring to the revegetation of relatively small areas that have been intensely degraded (e.g., abandoned mining operations or industrial sites). Reclamation connotes more than simply replacing vegetation. It also means repairing certain environmental processes. Generally, reclamation actions are taken where: (a) it is legally or contractually mandated; (b) it is socially or politically insisted upon, or (c) it is intended to repair certain economically advantageous environmental services – such as erosion control.

When applied to forest ecosystems, reclamation efforts are seldom applied over large geographic areas because of the high cost and lack of knowledge about most forest ecosystems. One exception has been in trying to reclaim strip-mined areas in North America; however, in many of these cases, the management target is to reclaim what used to be forest land for farm or grazing land. Reclamation pertains to sites where the damage has been so severe that landforms and soils may need to be manipulated before revegetation can even begin (Lugo 1997). Papers on site specific reclamation treatments are quite common in the literature for some forest types but generally rare or non-existent for others. For our purposes, ***forest reclamation*** will be defined as the deliberate manipulation of ecological patterns and processes for the purpose of dampening or forestalling the impacts of land degradation, such as soil erosion or desertification. Reclaimed forests will tend to be functional (particularly regarding those functions relevant to agricultural and water resources) but will not necessarily resemble natural forests in either structure or composition (Brown and Lugo 1994). Thus, environmental services, as opposed to the integrity of ecological systems, form the major management objectives in the case of reclamation.

Pathway #3 – Plantation Forestry

Plantation forestry is the planting of trees for the purpose of providing economic (and to a lesser degree ecological) benefits to human users. Generally, plantation forestry emphasizes productivity at the expense of many aspects of ecological integrity and ecosystem functionality and is therefore the least ecologically sound reforestation alternative from the standpoint of native biodiversity conservation. Tree planting and plantation forestry, no matter how productive, will not by themselves succeed in restoring native forest ecosystems. In fact, many plantations maintained today are composed of non-native species making them even less desirable. However, plantation forestry can be an important forest restoration step by playing a significant role in long-term native forest facilitation. For example, a plantation may provide canopy cover for the establishment of native shade tolerant species while simultaneously generating certain economic benefits some of which can be directed toward restoration costs (Howell 1986). Another important conservation advantage of plantation forestry is that under some circumstances it can be used strategically to reduce the demand for timber and wood fiber from remaining natural forests.

Pathway #4 – Forest Rehabilitation and Restoration

Some authors have chosen to view ***restoration*** of any "natural" system as an unachievable goal of ***rehabilitation*** (Cairns 1995; Harris et al. 1996; Hobbs and Norton 1996). Hobbs and Norton (1996) also argue that use of the term "natural" implies a static and predictable system and that it would be naïve to believe we have the ability to attain it. Aronson et al. (1995), on the other hand, argue that a "natural" reference state is necessary for setting restoration goals. It also has been argued that using the term ***restoration*** to the absolute extreme is absurd and denies the word any practical utility (Harris et al. 1996). Understanding that any restoration project is likely to fall short of its goals is essential for realistic management.

However, to use this fact as a reason for setting lower goals is equally absurd. Marginally missing the target of the completely "natural" state may be preferred to readily attaining some relatively "unnatural," alternate state.

Other authors have offered alternate suites of definitions. Bradshaw (1988), for example, emphasizes the differences between "restoration", "rehabilitation", and "replacement." In this author's view, the terms "restoration" and "rehabilitation" specify goals (full restoration and partial restoration, respectively) that are conceptually identical to our definitions, while "replacement" refers to the deliberate attempt to restore an alternate ecological state in place of the original system.

Forest rehabilitation will be used in this review to describe the deliberate manipulation of ecological patterns and processes for the purpose of creating a functional, partially self-sustaining forest that represents an alternative forest state (Brown and Lugo 1994). Rehabilitation applies when vegetation is present but succession has been arrested (Lugo 1997). While not entirely made up of native species, these forests are preferred to the degraded land conditions resulting from human utilization. Rehabilitated forests will tend to be structurally similar to natural forests but often temporarily (or sometimes permanently) include non-indigenous species for economic or ecological reasons (Lamb et al. 1997). Although functional and partially self-sustaining, rehabilitated lands are not naturally occurring systems and would not exist without human intervention.

Forest restoration is defined as the deliberate alteration of ecological patterns and processes for the purpose of recreating some presumed set of natural, pre-disturbance ecosystem conditions (Brown and Lugo 1994; Lamb et al. 1997). Restored forests would be similar in structure, function, and composition to the historic forests of the region prior to disturbance by humans (Lamb et al. 1997). Some ecological factors relevant to the establishment of original forest conditions at various temporal and spatial scales are: (a) use of indigenous species, (b) incorporation of natural successional dynamics, (c) consideration of ecological relationships, and (d) consideration of the effects of pattern on process. In our view, this is the only true forest restoration, and what is most commonly practiced throughout the world is actually one of the other, less ecologically sound, forms of reforestation.

In reality, the various reforestation practices as reviewed above have different goals, costs, and benefits (see Table 1), and they each result in having some level of ecological integrity over space and time. Although the concept of "ecological integrity" is complex and somewhat difficult to find agreement on in terms of specifics, "ecological integrity" has gained wide acceptance among both the scientific and regulatory communities (Davis 1995). The most influential definition of integrity was proposed by Karr and Dudley (1981), and states that biological integrity describes the:

> "ability of an ecosystem to support and maintain a balanced, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats within a region".

Integrity implies not only continuity of function and productivity but also "an unimpaired condition or the quality or state of being complete or undivided" and "correspondence with some original condition" or naturally evolved state (Karr 1992; Noss et al. *in review*). From this vantage, the goal of forest restoration can be seen as the reestablishment of the integrity of forest ecosystems. Each of the reforestation prescriptions outlined above results in leading to a particular level of ecosystem integrity. Thus, full restoration would imply recovering full integrity; rehabilitation would imply less than full integrity; and so on (see Figure 2). This perspective is not a trivial matter, but rather an effective way of clarifying the degree to which ecological restoration projects contribute to native biodiversity conservation.

Table 1. Summary of management goals and ecological costs and benefits for each reforestation alternative.

Reforestation Approach	**Goals**	**Costs**	**Benefits**
Natural Regeneration	Establish native forests with high ecological integrity	Specific sites or regions may not be able to recover to previous condition without human intervention	The most inexpensive alternative in terms of human time and resources. In some forest ecosystems, forests return to approximate previous undisturbed condition
Plantation Forestry	Maximize forest products	Requires constant human intervention, loss of some ecosystem functions, loss of native biodiversity at all levels, loss of some environmental services	Economic benefits from production, some mitigation benefits , possibly helps divert the need to destroy remaining native forests
Reclamation	Maintain certain environmental services	Financially expensive and requires long-term human intervention, some loss in ecosystem function, some loss of native biodiversity	Provides possible benefits to human enterprise such as agriculture and water quality, maintains some environmental services, provides some habitat for native biota
Restoration	Establish native forest with high ecological integrity	Can be expensive in time and resources, may never be able to replicate original forest	Forests approach original condition in terms of composition, structure, and function

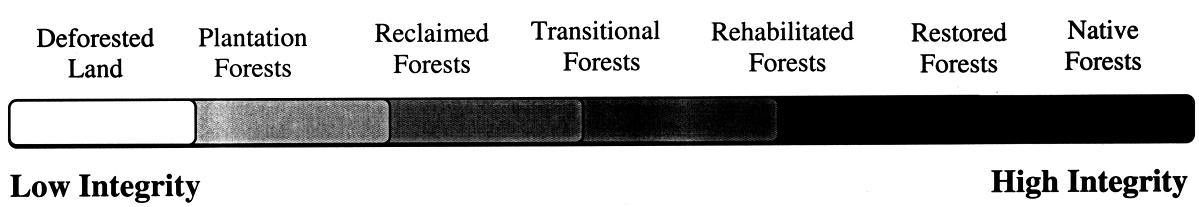

Figure 2. Spectrum of forest ecological integrity as it pertains to forest conditions and reforestation management alternatives.

Global 200 Ecoregions

The following section, which makes up the majority of this review, attempts to outline reforestation for the various Global 200 ecoregions arranged according to their Major Habitat Types and Geographic Realms (Table 2).

Table 2. Major Habitat Types and associated Geographic Realms for the World Wildlife Fund Global 200 ecoregions.

Major Habitat Type	Geographic Realms
1. Tropical and Subtropical Moist Broadleaf Forests	1-1. Neotropical
	1-2. Afrotropical
	1-3. Indo-Malayan
	1-4. Australasia
	1-5. Oceania
2. Tropical Dry Forests	2-1. Neotropical
	2-2. Afrotropical
	2-3. Indo-Malayan
	2-4. Australasia
	2-5. Oceania
3. Tropical and Subtropical Conifer Forests	3-1. Neotropical
4. Temperate Conifer and Broadleaf Forests	4-1. Eastern Nearctic
	4-2. Western Nearctic
	4-3. Neotropical
	4-4. Palearctic
	4-5. Australasia
5. Boreal Forests and Taiga	5-1. Nearctic
	5-2. Palearctic

Each subsection is organized in the same fashion to make this review more useful. Each subsection contains: (1) a description of the ecological setting (including basic ecological constraints where possible), (2) a discussion of restoration and management (including some case studies), and (3) conclusions.

Considering the scope of this paper and the space available, the reviews that follow present only general information for the forests of concern in the designated ecoregions of each realm. In reality, the forests of each ecoregion are comprised of many formations each of which has a multitude of microsite variations due to varying climatic, edaphic and ecological conditions. Small libraries could be filled with the volumes of literature written on the different aspects of forests of certain ecoregions. While large gaps in our knowledge still exist, we know enough about the ecology of many forest types to carry out some form of reforestation.

1-1. Tropical and Subtropical Moist Broadleaf Forests—Neotropical

Ecological Setting

The Neotropical moist broadleaf forests comprise 12 ecoregions, and extend from the Atlantic forests of Argentina, Brazil and Paraquay to the Greater Antilles of the Caribbean, including most of lowland northwestern South America, the isthmus forests of Panama and Costa Rica primarily on the Caribbean side, and a large portion of the Amazon basin (see Plate 1). The incredible diversity and extent of these forest ecosystems is beyond the scope of detailed discussion here. The equatorial climate of most ecoregions provides for year round uniform high sunlight, temperature, humidity and precipitation. A considerable proportion of the soils (82 percent) are acidic of the orders oxisols and ultisols (Sanchez 1989). Though these soils have good physical properties, they are poor in nutrients and susceptible to leaching. The forests are typically evergreen, though deciduous components are common in the Atlantic coast forests. The diversity of species is perhaps the greatest in the world, however a number of common canopy species are widespread throughout the different ecoregions. Regional centers of endemism appear to be derived from ice age refugia (Prance 1989).

The forests of the Amazon basin are located in a relatively stable climatic zone and therefore are not subjected to significant weather disturbance events. However, the forests of the Greater Antilles, Central America, and the northern coast of South America are impacted by wind events from tropical storms. Landslides also produce large natural clearings in forests located on steep terrain. Natural forest openings due to tree fall are typically small in the Amazon basin. Propogule sources are close and regeneration depends in large part on microsite conditions. Regeneration method and dispersal ability become important factors in openings periodically created in those forests subject to larger disturbance events. Natural fire is a rare and extremely limited event in these moist forests. Though, fire has been used by local populations for perhaps a thousand years or more, burning of deforested areas today dwarfs the burning of previous eras (Fearnside 1990). Unnatural fire frequency has the greatest impact of any disturbance on forest recovery.

Restoration and Management

Throughout much of the tropics, extensive areas of moist rainforest have been cleared for pastureland and to a lesser extent shifting agriculture. Remaining rainforest cover and deforestation rates vary widely throughout the neotropics. Deforestation has been greatest in the Caribbean, where at the extreme, only one percent of Haitian rainforest persists. As of 1990, 88 percent of South America's rainforests were still intact (FAO 112 1993), but significant losses have continued. Accompanying both of these agricultural practices is the regular use of anthropogenic set fires to facilitate clearing and control weeds. While tilled fields are typically left fallow after two to four years, pastures may be used for decades before being abandoned. The process of forest restoration depends largely on the degree of soil damage and the availability of seed or regenerative stock.

Recently the neotropics have seen a proliferation of research on natural forest regeneration on damaged lands. At cleared locations, lightly grazed for only a relatively short time span, with minimal fire use and soil damage, native woody vegetation readily sprouts following abandonment (Uhl et al. 1988; Kauffman 1991). Eastern Amazon land, cleared, burned, and lightly grazed with minimal pasture establishment, returned to a 13-m closed canopy forest within eight years of grazing cessation (Uhl et al. 1988). Between 12 to 30 percent of the trees rose from stump sprouts. Cleared and lightly burned land regenerated a dense 4-m canopy from basal sprouts within 20 months in the Para region (Kauffman 1991). Pastures abandoned after more moderate grazing use regenerated only 50 percent of the above ground biomass, with a lower diversity after eight years, as the recovered lightly used pastures (Uhl et al. 1988).

Abandoned pastures subjected to heavy use, including mechanical clearing and repeated burnings, recovered poorly, producing only 6 percent of the above ground biomass and very low diversity compared with the lightly used areas (Uhl et al. 1988). Even without burnings, sprouting and seed bank production was virtually eliminated in abandoned pastures grazed for periods of 10 to 40 years (Aide et al. 1995). Lightly impacted sites have the ability to self-regenerate through sprouting, producing a significant portion of the new growth. However, repeated burnings, lengthy grazing history, and degraded soil can dramatically reduce the regenerative ability through sprouting and the seed bank. Interestingly, one heavily impacted site that received shallow discing produced regeneration through 2.5 years equivalent to that of lightly used sites (Uhl et al. 1988).

Slow initial recovery through the first 20 years of abandoned pastures in Puerto Rico also has been attributed to the suppressive effects of the pasture grass on dispersed seed (Aide et al. 1995). Quick forest regeneration was expected due to small pasture size and the close proximity of an intact forest seed source, but that was not realized in this instance.

In many areas, the presence of remnant trees appears to facilitate woody colonization of abandoned pastures compared with pastures without remnant trees (e.g., the rainforest at Los Tuxtlas, Mexico [Guevara et al. 1986]). Mature primary forest and late secondary species were present in relatively higher quantities beneath remnant trees, and bird dispersal was responsible for 86 percent of the total woody species. The remnant trees either facilitate dispersal or provide a more suitable microclimate for establishment or both.

As much as 90 percent of the tree species forming mature tropical moist rainforests rely on animals for dispersal compared to temperate systems, which depends largely on wind for plant dispersal. Extensive openings are often lacking in many of the native fauna resulting in limited dispersal potential. Combined with degraded soils and drier conditions, many of these areas show extremely slow reforestation rates went left alone. A number of trials have been conducted that have demonstrated some survival and reasonable growth rates of certain indigenous species, even better than exotics in some cases, which are often employed to condition deforested sites that are difficult or slow to recover (Butterfield and Espinoza 1995; Butterfield 1995). Other indigenous species also have been tested on severely degraded soils and have been shown to positively affect soil fertility (González and Fisher 1994; Montagnini et al. 1995).

In general, poor and damaged soils can significantly delay forest regeneration on cleared land. In the upper Rio Negro of Columbia and Venezuela, for example, where small slash and burn sites (between 0.5 and 2 ha in size) were left fallow after two to four years and recovered species richness within 20 to 40 years – above ground biomass and mature structure is expected to take nearly 200 years to develop (Saldarriaga et al. 1988). Topsoil erosion can create a barrier to forest regeneration even when other conditions are conducive for establishment (Aide and Cavelier 1995). Under these circumstances, established grasses actually increased the success of seed germination by improving the microhabitat.

One hundred and sixty indigenous taxa have been tested for early performance for growth on open bauxite mined lands in western Para State, Brazil (Knowles and Parrotta 1995). Through two years, 37 percent were rated good (vigorous growth with at least 75% survival) for open conditions on reclaimed mine land. Forty-four percent of the tested species performed poorly in the open and required shade for establishment and growth. Information on fruiting phenology, dispersal and seed viability was also collected and compiled. The pantropical genus Trema, more specifically *T. micrantha*, has strong potential as an all-purpose nurse crop species for the neotropics (Vásquez-Yanes 1998). The advantages of this native pioneer are easy propogation, tolerance of exposed sites and poor soils, fast growth, short life span, edible fruits for attracting birds, and potential agroforestry use.

The use of tree plantings and even transitional plantations, in addition to establishing immediate tree growth and structure, can ameliorate the soil and positively modify the microclimate. Native nitrogen fixing pasture trees also can significantly increase soil nitrogen as observed in a degraded montane cloud forest sites of Ecuador (Rhoades et al. 1998). However, nitrogen fixing trees do not always produce increased forest regeneration. For example, species richness and diversity was lower beneath a nitrogen fixing native alder (*Alnus acuminata*) plantation after 15 years than in paired secondary forest regenerating in an abandoned pasture (Cavelier 1995). Not everything fits a generality.

The relative catalytic effect of plantations on colonizing undergrowth is greatest in more tropical regions and in the most disturbed sites, especially those with suppressive ground cover (Parrotta et al. 1997). Even plantations of exotic species can be utilized to regenerate native forests. Species richness was greater through 4.5 years beneath an *Alvizia lebbek* plantation in subtropical Puerto Rico than adjacent control plots (Parrotta 1992). After 50 years, the species richness in the understory of a mahogany (*Swietenia macrophylla*) plantation in subtropical Puerto Rico approached that of its paired moist secondary forest (Lugo 1992).

The countries of the neotropics vary widely in their social and political characteristics. In each case, the role of the NGO will differ. However, one common theme occurs in most, if not all, of these countries: participation and support by local people, especially farmers, is critical to the success of forest restoration projects. Moreover, these communities must be involved both from the technical standpoint of contributing knowledge and effort, but also from the legal standpoint of being guaranteed certain rights to the benefits of restoration activities (Davis and Wall 1994). At the same time, farmer participation alone, particularly when motivated by short-term economic and non-economic benefits, does not necessarily ensure long-term management (Thacher et al. 1997). Land tenure and resource rights significantly influence local people's participation. In certain countries, such as Bolivia and Costa Rica, poor land tenure systems provide little incentive to reforest the land or enhance its productivity through long-term investment (FAO 101 1993; Mahar and Schneider 1994 [in Brown and Pearce 1994]; Thacher et al. 1997). NGOs working to influence forest policies also should seek to influence land tenure and resource rights policies in favor of secure, well-defined land tenure arrangements.

In countries like Mexico, where the government has a long-standing (though largely ineffective) tradition of officially supporting forest conservation, local and international NGOs can offer local communities various technical and financial support (Silva 1994; FAO 101 1993). Additionally, these organizations can help legitimize local community perspectives in forest policy debates with relatively sympathetic government bodies (Silva 1994). Similarly, as a result of extensive soil erosion damage and a high standard of living, Puerto Rico is another country that tends to show support for conservation activities (FAO 101 1993). In places like Mexico and Puerto Rico, where conservation efforts do not face significant interference from government bodies, NGOs may be most effective by actively supporting local community efforts (Silva 1994).

By contrast, in countries like Brazil, where the government and dominant social classes show strong opposition to forest conservation, local, national, and international organizations might form broad alliances to politically influence national forest policy (Silva 1994). Most forests in Brazil are state owned (Evans 1992). The forestry department in charge of Brazil's state-owned forests has been described as rigid and bureaucratic, unable to balance economic, social, and environmental values, and lacking in technical capacity and research coordination (Dawkins and Philip 1998). Moreover, Brazil's forestry department does not have a reforestation program (Evans 1992; Dawkins and Philip 1998). Reforestation efforts in Brazil will face tremendous political obstacles from government bodies. External political pressure may be the best means to influence national forest policy and forest restoration in Brazil over the long run (Silva 1994).

However, most neotropical governments seem to fall into a third category of ineffectiveness or indifference. Forestry administrations in Columbia, Ecuador, Venezuela, and Peru tend to be characterized by underfunded, underequipped, and understaffed offices, inactivity, and a lack of authority (Silva 1994; FAO 101 1993). In Peru, for example, forest degradation is due not so much to negative forest policy as to failing agricultural policies that encourage conversion of forestland to farmland as well as a lack of interest from the forestry administration (FAO 101 1993). In these countries, NGOs might be most effective giving long-term community support (Silva 1994). While not facing direct governmental opposition, these communities receive no governmental support. For restoration projects to work on the ground, NGOs must make long-term commitments to both the work and the local communities in these regions.

Conclusions

Studies indicate that lightly disturbed land, if not too extensive in total area, will rapidly recover on its own, largely through the soil seed bank and vegetative sprouting. Colonization by dispersing vertebrate species typically follows, and cover is established if refugia still exists in the surrounding area. More severely degraded sites, in which sprouting capacity and the soil seed bank have been largely eliminated, will require additional measures to facilitate forest restoration. Small group plantings of hardy native species in larger clearings can provide a nexus for bird dispersal and to provide suitable microclimate for undergrowth establishment. Forest regeneration then radiates from these centers and merge with one another over time. Large expanses can be more thoroughly addressed by utilizing plantations to initiate the reforestation process to be later directed toward more ecologically desirable goals - true forest restoration. Research has identified a number of native tree species suitable for use in neotropical plantation overstories. In addition to producing favorable microclimate for colonizing undergrowth, the use of native species contributes to the natural regeneration of the forest. A variety of species should be tested at individual sites to determine suitability for specific conditions including site heterogeneity. Incorporating mixed species and irregular patterning in plantations also can help promote natural diversity and heterogeneity typical of these forest systems. In all cases, targeted reforestation sites must be protected from human introduced disturbances such as grazing and repeated fires for successful restoration.

Most neotropical mainland countries currently retain the majority of their rainforests relatively intact with some notable exceptions (e.g., Atlantic forests). Existing protected areas networks should be upgraded to include important and endangered ecosystems before they become further threatened. The Greater Antillean moist forests of the Caribbean, with most rainforest having been destroyed and increasing pressure on the little that remains, may be the harbinger of what is to become of the mainland forests as well. Conditions are so poor that only drastic protection measures combined with assistance in establish alternative forest product sources on degraded lands will preserve the remaining forests.

In addition to the technical recommendations, forest restoration in the neotropics will greatly benefit by examining what role the greater conservation community could best play given the social and political conditions in particular countries. In countries where the government is generally supportive, technical and financial support may be offered to already active local restoration efforts. Alternately, in countries where the government and ruling class oppose forest conservation, the conservation community might impose external political pressure to try to influence the government's national forest policy. Finally, where the government is inactive or indifferent, the conservation community should make long term commitments of support for local communities to make up for the lack of governmental support while other entities work with the governmental side of the equation. Although the ecological and technical aspects of forest restoration dictate what's possible, the conservation community will be more effective if it also integrates social and political opportunities and constraints into their forest restoration strategies.

1-2. Tropical and Subtropical Moist Broadleaf Forests—Afrotropical

Ecological Setting

Comprising 13 ecoregions, the moist forests of Africa are farflung, including the Atlantic Macaronesians, and Madagascar, the Seychelles and Mascarene Islands of the Indian Ocean, in addition to the Guineo-Congolian block of west-central Africa (see Plate 2). Climate and soil conditions of west-central Africa are similar to those in the Neotropics. The wetter, lowland evergreen forests transition to more deciduous forests in drier zones (Hamilton 1989). Often just one canopy species, frequently a member of the Caesalpinaceae family, will dominate, especially in the evergreen forests. Most primary canopy species have large seeded fruit which rapidly germinate and appear to be suited to gap colonization. These forests are rich in leguminous species and lack dipterocarps as observed in South America – palms are rare. Though separated by an arid zone, floristic similarities tie the East African forests to the Guineo-Congolian block. East African coastal forests are distinctly deciduous and characterized by strong seasonality and water stress (Burgess et al. 1996). The transition to rainforest occurs at elevations above 500 m with the increase in orographic (condensation from rising air masses) rainfall (Chapman and Chapman 1996a). A distinctive feature is the fragmented nature of the East African forests, which may have originated as islands surrounded by savannah. Higher elevation rainforests are often geographically separated on hills from one another. The high endemism of this region is further accentuated by these geographic restrictions which limit many endemics to just one or a few forest remnants. Though not as species rich as the Neotropic or Indo-Malayan moist forests, regional centers of high endemism do exist throughout Africa. Madagascar rainforests, with 90 percent endemism and the third highest primate diversity, have been targeted as a top priority for conservation by the international community.

As with the equatorial moist forests of Amazonia, the Guineo-Congolian forests generally are not subject to natural fires or large-scale climatic disturbances. Warm monsoonal winds originating in the Indian Ocean are the key moderating climatic influence for the East African and Madagascar forests. Under these conditions, forest regeneration largely depends upon gap dynamics – dispersal and local microsite conditions are the driving forces to revegetation. Madagascar forest ecology is unique lacking certain typical animal taxa (Wright 1997). Most birds on Madagascar are insectivores. Frugivores (seed dispersers), which are the major dispersal agents in other tropical forests in this realm, are uncommon. Natural and anthropogenic fire is widespread in Africa, but mostly is restricted to drier ecosystems. However, widespread fire can result from unusual drought conditions, as witnessed in 1983 in central Africa (Whitmore 1991).

Restoration and Management

For millenia, native populations have lived in harmony with the forests on mainland Africa, practicing shifting cultivation and harvesting forest products. The colonization of Madagascar has been a much more recent event. Secondary forests created from long rotation fallows are pervasive in many areas and often indistinguishable from primary forests (Hamilton 1989). However, rapid population growth in the latter half of the twentieth century has dramatically increased the pressure on these forests. Forest clearing for fuelwood is the most notable of the destructive practices, accounting for three-fourths of the wood harvested in Cameroon from 1985 to 1987 for example (Alpert 1993). Meanwhile the amount of timber harvested also doubled from ten years earlier. Furthermore, as in Ghana (Hawthorne 1994), forest fires are becoming more common, amplifying the threat to increasingly fragmented, moist forests. East African forests are unique in that they may have originated under conditions of high insularity. Increasing population pressure and the accompanying shifting cultivation, as with the rest of Africa, have further fragmented the remaining forests. Though originally allocated for timber resources, most of the remaining East African forests are located in reserves with varying degrees of protection (Burgess et al. 1996). Relaxed protection after independence led to the destruction of half the remaining forest between 1960 and 1990 on Madagascar

(Wright 1997). Currently Madagascar boasts over 50 protected areas, mostly in the eastern rainforest region.

Colonization of open areas depends upon dispersal capability, in part. Small seeded species were more widely dispersed by a variety of animals including birds in Gabon (Hladik and Mitja 1996). Large seeds tend to drop near the seed source and were dispersed less by animals, especially birds. On Madagascar, birds are not typically a dispersal mechanism. The coastal forests of East Africa seem to contain few pioneer tree species capable of colonizing large disturbed areas (Burgess et al. 1996). Regeneration by vegetative sprouting also plays a crucial role (Hladik and Mitja 1996). At Ranomafana National Park, Madagascar, over 50 percent of felled trees resprouted vegetatively following logging (Wright 1997). Following successive cycles of cultivation, species with sprouting capacity tend to dominate in fallow fields.

Recent studies also show that different plantation overstory species may have different influences on understory colonization and growth. For example, *Pinus caribeae* plantations developed 50% higher native woody stem density and diversity compared to similar aged *Cupressus lusitanica* plantations in Uganda (Fimbel and Fimbel 1996). After 30 years, plantations of two Pinus species in Uganda contained 47 of the 78 tree species present in a comparable area of nearby intact forest (Chapman and Chapman 1996b).

The two most commonly cited socio-economic influences on reforestation efforts in tropical Africa are local participation and land tenure. Promoting forest restoration in these areas will require consideration of these two factors. However, outsiders, particularly from western countries, must be careful not to make certain assumptions when trying to affect these changes. Western conservation values are not necessarily mirrored by local land use perspectives, no matter how complementary the two sides apparently seem (Fairhead and Leach 1994). Likewise, western ideas about property and land use are generally not compatible with African views on land tenure (Leach 1994; Cox and Elmqvist 1997). Thus, when trying to influence these socio-economic variables, the conservation community must be careful not to impose western perspectives onto the African people, or the restoration efforts will most likely deteriorate over time.

Conservation activities in West Africa are recently initiated and largely informed by western interests and ideas. These perspectives are typified by a sense of urgency to preserve rainforest biodiversity, a paternalistic attitude toward African indigenes, and a suite of pre-selected conservation strategies that include the bureaucratic designation and enforcement of non-use protected areas (Goldsmith 1998). Camaroon provides a good illustration of this western influence. For example, the plans behind Camaroon's Korup National Park still promote forced resettlement of the local population while the Plan de Zonage, covering the entire forested region, takes little or no account of actual village land use patterns (Goldsmith 1998).

Recent work, however, points to the inappropriateness of these attitudes for forest conservation in West Africa. The success of a United Nations Development Programme, Africa 2000 Network, reforestation project in the Sine-Saloum region of southern Senegal, for example, will depend on the skillful balancing of two primary objectives: (1) to restore the forest environment and (2) to ensure local distribution of the restoration benefits (Cisse and Helmore 1992). Boubacar Fall, the Senegalese national coordinator, has stated that, "unless local people take part in ownership and leadership of the project, the project will fail." To gain the trust of the local villagers, OSDIL, a Senegalese NGO involved in reforestation, engaged in a long courtship process where they learned about village economic conditions and asked for the villagers' perspectives, particularly those of the women who grew vegetable gardens. Without directly offering money, OSDIL exchanged what the villagers felt they needed most – fencing, water and training in the various aspects of gardening – for involvement in the reforestation effort (Cisse and Helmore 1992). The perceived goodwill and short-term benefits of their relationship encouraged the villagers to participate in the long-term project. This would not have been possible if the local views and needs were not respectfully integrated with the project plans. This conclusion is supported by research in other parts of Africa, as well, including, East Africa (Ståhl 1993), Tanzania (Mgeni 1992), and Madagascar (Hawkins et al. 1990).

Additionally, participatory forestry does not merely take into account current local needs and perspectives, but also entails understanding the people's historical relationships with the land. Fairhead and Leach (1994) examine the roles of land use and political history in the conservation of forests on the Ziama Reserve in Guinea. Quoting a local elder, Fairhead and Leach (1994) suggest a positive relationship between community responsibility and forest management. The authors conclude that participatory forest management will not succeed unless historical land claims and political authority are given substantial consideration in forest management agreements. Otherwise, local communities will tend to feel alienated from the forest management process and see little reason to offer their support.

From an African perspective, conservation is not the most important goal. Rather, given the long-standing connection between humans and forests, establishing the conditions for future human-forest relations forms the primary local concern (Goldsmith 1998). This will sometimes involve conflicts between conservationists and local stakeholders. Resolving these conflicts can take several forms, including: (1) compensating local communities for losses due to externally imposed resource restrictions, (2) reconciling the opposing interests by focusing on complementary interests, and (3) empowering local communities to manage the resources on their own (Leach 1994). The appropriate combination of these three approaches will vary from project to project.

Defining human-forest relations, however, should not come at the expense of conservation goals. The Okomu Forest Reserve of Nigeria provides an instructive example. In the Okomu Forest Reserve, conservation goals were supplanted by agricultural development objectives with the idea that by improving the human condition within the reserve, the fundamental causes of forest degradation would be eliminated. Instead, Oates (1995) argues that the agricultural assistance program actually promoted destructive activities by making deforestation more profitable. Restoration ecologists must be wary of this kind of goal displacement when seeking to integrate human needs and conservation objectives.

Restoration projects being promoted by the non-governmental organizations (NGOs) may suffer another obstacle – political intolerance. The Zimbabwean Government, for example, is very intolerant of NGO autonomy (Thomas 1996). NGO involvement is still somewhat tolerated, in part, because of the funds these organizations bring into the country and due to the poor regulation of NGO activities by government authorities (Thomas 1996). This does not imply that NGOs have much policy influence at the government level, however, even if NGOs and the government are collaborating. Rather, true policy influence can be achieved only at the implementation level by involving local people in ground-level reforestation activities.

Insecure land tenure is a major hindrance to forest restoration in many African countries (Leach 1994). This is particularly true in East African countries like Ethiopia, Uganda, and Tanzania where land is owned by the government and land use rights are held by chiefs and elders (Ståhl 1993). Farmers in these countries seldom make long-term investments in the land but rather develop short-term planning strategies. As an example, Ethiopian farmers, under the Mengistu Regime of the 1980s, facing the possibility of being involuntarily relocated within two years, believed it irrational to make long-term investments in forest plantations (Ståhl 1993). Conversely, in Kenya, where the land tenure system includes well-defined property rights, farmers tend to employ sustainable management practices on their privately owned land holdings (Holmgren et al. 1994). The significance of land tenure holds true in western Africa as well. Leach (1994), discussing forest communities and forest conservation in Sierra Leone, highlights tenure arrangements among various factors influencing local investments in resource production and sustainability. Restoration efforts attempting to gain local participation should take into account land tenure arrangements and their associated incentives.

For years, a model of destruction, Madagascar has recently become a model of local community orientated conservation. Since the late 1980s, the international community has worked with Madagascar to implement the first country-wide environmental action plan in the world (Wright 1997). Nine protected areas were selected to begin long-term integrated conservation and development programs (ICDPs). Ranomafana National Park, established in 1991, is a prime example. With 93 villages (population of 25,000) surrounding the park, local community participation was critical. The greatest continuing threat to the park was slash and burn cultivation. Management teams focused on park management, conservation education, village farming, health, monitoring, and biodiversity research. Discussion and involvement by the local populace, which has become a day to day and year to year process, is still necessary for long-term success of the park's goals. Training of local research assistants, park rangers, tourist guides, and university students is the key to the future. To date, peripheral zone activities have generally been successful. However, full implementation of the program is at risk due to continued funding and equipment needs and low government priority.

Conclusions

Most of the African moist broadleaf forests are highly fragmented and under increasing human pressure, and they require immediate conservation action. Protected areas have been widely established throughout Africa, and the majority of the remaining forests in this realm are already contained in these parks. However, due to a variety of reasons, including understaffing, economic interests, and increasing populations, actual forest protection is mostly ineffective. The first priority is to upgrade the current reserve system to strengthen protection of the existing forests. Protection will be achieved only if community programs are established to provide alternative sources of forest products, thereby removing pressure from the forests. These programs fall under the category of agroforestry, forest reclamation, and rehabilitation, and should be pursued only on lands previously cleared or severely degraded. The environmental action plan established for Madagascar is a model conservation program incorporating the needs of all involved. Finally, current knowledge of forest ecology pursuant to forest restoration in Africa is limited and should to be further investigated before attempting large scale projects.

In addition to the technical and ecological constraints to reforestation in this realm, two social considerations are very important – local participation and land tenure. Local participation is necessary both to meet the needs of local communities and to ensure the success of restoration efforts. To be effective, participation must involve more than just taking into account local needs and values; participation also must include understanding human-forest relationships, both historic and current. Additionally, meeting local needs should not involve displacing conservation goals with development objectives. Finally, secure land tenure is very important to ensure long-term planning and participation, otherwise there is little incentive to invest in future forest development.

1-3. Tropical and Subtropical Moist Broadleaf Forests—Indo-Malayan

Ecological Setting

The moist forests of the Indo-Malayan realm extend from southern China south and east to the Mollucas of Indonesia, and west to include Sri Lanka and the Western Ghats of India (see Plate 3). The heart of this region, the forests of Borneo, may contain the highest species richness of any forests in the world, particularly of trees (Whitmore 1984). A key feature of these forests is the widespread emergent canopy dominance by members of the family Dipterocarpaceae, which features some 400 species (Whitmore 1984; Ramesh et al. 1994; Yuming and Yuachang 1994). Often a forest canopy will be comprised of just a few species, but with restricted ranges, resulting in regional endemism. In a region of exceptional soil diversity in northwestern Borneo for example, 242 dipterocarp species exist, of which 89 are endemic (Ashton 1989). Dipterocarps become less important forest components to the west in India and to the north in China.

Most tree species are animal dispersed and produce orthodox (seed with low moisture content that will not degrade quickly) fruits, which can remain viable in the soil seed bank for extended periods. Many main canopy species, however, particularly the dominant dipterocarps, are poorly adapted for colonizing large openings. Dipterocarps flower gregariously at irregular intervals (every 2 to 10 years), producing winged, recalcitrant (seed with high moisture content that will degrade quickly) fruit that disperses poorly, usually remaining within 60 m of the source (Appanah and Nor 1991; Whitmore 1984). The fruit, which are susceptible to desiccation, typically germinate very quickly, within a few days to two weeks. Once beyond the sappling stage, capacity for vegetative reproduction is greatly reduced for rainforest dipterocarps (Ashton 1989). The irregular fruiting and recalcitrant seeds are important considerations in restoration projects. Canopy species, including dipterocarps, require varying degrees of shade for seedling establishment, but after 18 months will respond to full light with rapid growth.

As with other tropical forests, the soils are generally characterized by low fertility resulting from thorough weathering combined with leaching due to high rainfall. The forests are subject to cyclones and wind events; however, fire is generally not part of the natural disturbance regime. Occasional severe drought can make these forests more susceptible to widespread fire as was the case in Borneo in 1982-83 (Whitmore 1991) and more recently in 1998, but these catastrophic fires are unusual for forests in this region.

Restoration and Management

Traditional shifting cultivation including clearing and burning, which resulted in patches of secondary forest, has been widely practiced throughout the region up to the present (Whitmore 1984). Timber was selectively cut for fuelwood and local construction. Europeans introduced plantation agriculture, which did not allow forest regeneration. Though early commercial logging was selective, severe understory damage still resulted from felling and mechanical equipment. In many places, forest clearing and conversion to plantation has increased dramatically since the 1950s along with the corresponding growth in the international timber industry.

Natural regeneration is strongly affected by the characteristics of the disturbance, particularly repeated human disturbance involving fire. Fifteen years after plantation clearance in the Western Ghats, India, secondary forest structure and diversity was developing comparable to that of nearby mature forest (George et al. 1993). Regenerating *Eucalyptus tereticornis* from the plantation, despite having the highest density, did not inhibit colonization by native species. However, after burning in East Kalimantan, Indonesia, the resulting secondary forest was far less rich in species compared to primary forest (Riswan and Abdulhadi 1992; Whitmore 1984). In general, regeneration on cleared and burned forest has less species richness than regeneration on adjacent cleared and unburned forest. In a field study by Riswan and Kartawinata (1991), unburned plots contained twice the number of primary tree species than in burned plots. Vegetative sprouting, soil seedbank production and recent dispersal can all be important sources of regeneration depending upon the local ecology. Successive disturbance, especially yearly fires, reduces the soil seed bank and the capacity for sprouting, resulting in diminished species richness, particularly primary tree species (Riswan and Abdulhadi 1992; Whitmore 1984). Eventually, repeated fires in the Malaysia-Indonesia-Philippines region produce suppressive *Imperata cylindrica* grasslands, which persist once established.

The use of nurse plantations has been tested extensively in the Indo-Malayan region for two primary reasons – the control of imperata grasslands and the facilitation of underplanted dipterocarp seedlings. Widespread tracts of imperata grassland have replaced rainforest due to repeated human disturbance. Imperata is shade intolerant and can be suppressed through closed canopy plantation establishment combined with initial burning or tilling. Physical disruption of suppressive ground cover, *Dicranopteris linearis,* in Sri Lanka was only partially successful at initiating colonization by pioneering species (Cohen et al. 1995). Fast growing exotics, such as *Leucaena leucocephala* and Acacia (*Acacia spp.*), which also produce valued wood, have been the plantation species of choice (Kuusipalo et al. 1996; MacDicken et al. 1997). Imperata was eliminated after seven years under *Acacia mangium* at which time options were available for site restoration (MacDicken et al. 1997). Evaluation of 83 species for reforestation of imperata grassland in Indonesia, showed natives performed poorly compared with exotics (Otsamo et al. 1997). Although certain native species may compete with imperata successfully, direct planting of dipterocarps and other mature forest species over widespread open areas is impractical. However, establishment of shade tolerant primary forest species (dipterocarps and non-dipterocarps) was generally successful beneath a two-year-old *Paraserianthes falcataria* plantation on imperata (Otsamo et al. 1996), but only after considerable site preparation. Furthermore, partial removal of an 80-month-old *A. mangium* plantation overstory released the underplanted dipterocarp (*Anisoptera marginata*) to increased growth, while imperata remained suppressed (Otsamo 1998). Partial removal of the *Pinus caribaea* plantation canopy promoted greater dipterocarp seedling growth compared to seedling growth under the closed canopy condition at the Sinharaja Reserve in Sri Lanka (Ashton et al. 1997). In general, the majority of studies on dipterocarps focus on the facilitation of this valuable tree species as opposed to native forest recolonization beneath plantations. The results from these studies show that plantations may be useful here as elsewhere for forest restoration on cleared and degraded lands.

With mixed success, China has combated deforestation and desertification with large scale tree planting campaigns for decades. More progressive research on reforestation and forest restoration has been explored in Quangdong Province, southern China (Guohui and Chun 1995). At Xiaoliang Station, forest rehabilitation began on a heavily eroded, 430 ha site in 1959 (Yue et al. 1994). Initial steps involved establishing tree cover to stabilize and ameliorate soils with a plantation of exotic pioneer species of *Pinus* and *Eucalyptus*. Beginning in 1974, portions of the original plantation were replaced with mixtures of native and exotic trees. Soil improved dramatically under the mixed plantations, but was still well below that of nearby intact forests. Complete soil development was estimated to require at least 150 years. Bird use and colonization by outside plant species increased during the 1980s. By 1989 the new forest had 320 plant species and was beginning to achieve some structural and compositional similarities with nearby forests.

The roots of deforestation, and consequent windows of reforestation opportunity, in many Indo-Malayan countries can be traced back to their colonialist histories (Barraclough and Ghimire 1995). Most of these countries including, Indonesia, Malaysia, Papua New Guinea, India, and the Philippines underwent colonial occupation, especially by western powers (Hurst 1990; Barraclough and Ghimire 1995). The impact of generations of western influence can be felt throughout the social and cultural institutions of a number of Indo-Malayan countries; the same can be said for natural resource policy and human-forest interactions (Barraclough and Ghimire 1995).

Generally speaking, imperial occupation eroded communal ownership of land, which generated little revenue, in favor of more profitable individual and state-ownership (Hurst 1990). Also, European control induced a shift in resource use from "minor forest produce" to timber extraction, as well as conversion of forests for cash cropping (Hurst 1990). Finally, colonial influence affected the development of Asian society by creating a small, westernized elite of individuals who saw potential wealth in land ownership and cooperation with colonial powers (Hurst 1990).

After regaining independence, most Indo-Malayan countries began to seek investments by industrial nations under the guise of "development" (Hurst 1990; Jepma 1995). However, as during the colonialist era, individuals recognized the opportunity for personal gain and by capitalizing on foreign investments became the new elite. Of course, the general population of most countries, the Philippines for instance, did not benefit from the so-called "development", because priorities of individual wealth and power strongly affected the activities of Indo-Malayan governments and supplanted the development goals (Hurst 1990).

The social and economic remnants of this history dictate certain requirements for successful forest restoration. Today, forest use and exploitation are carried out by large, multinational corporations. Thus, the timber industry is characterized by foreign control, capital intensive production, and raw material exports (Barraclough and Ghimire 1995). Examples include Australian and British corporations in Papua New Guinea, Britain in India, Spain followed by the United States in the Philippines, and Japanese corporations in Indonesia (Hurst 1990; Barraclough and Ghimire 1995). Attempts to process the materials into finished goods are generally thwarted by heavy tariffs, leaving these underdeveloped nations at a severe economic disadvantage.

Since the 1970s, their economies have been further characterized by heavy debts to foreign nations, which were undertaken for internal industrial development. The countries borrowed from World Bank and other western merchant banks to gain control and export finished products (Hurst 1990; Jepma 1995). Debts, and the accumulated interest, left Indo-Malayan countries in serious financial trouble. In order to make payments, many countries have attempted to expand their raw material exports by converting forests to agricultural and timber plantations (Jepma 1995). The result is a vastly growing population of displaced and landless farmers who must choose between the city and the as yet unoccupied forest, thus exacerbating the pressures placed on natural forest ecosystems (Hurst 1990; Jepma 1995).

To alleviate this negative trend, several lenders have attached certain development-oriented "strings" to their loans. However, many of these have been criticized as inappropriate and counterproductive (Hurst 1990). For example, agricultural development projects displacing local farmers in the Philippines, agricultural expansion projects converting large areas of forest in Indonesia, rehabilitation projects absolving timber companies from responsibility in Malaysia, and other projects actively promoting expansion of the timber industry or the planting of exotic species, have all had detrimental effects on the forests that they were intended to help maintain (Hurst 1990). Often the goal of conservation is effectively displaced by economic objectives when development is seen as the primary tool.

Alternate approaches might be more effective. These could include: (a) alleviating the debt crisis, (b) reforming trade patterns, (c) halting environmental destruction development projects, and (d) reforming land ownership and tenure (Hurst 1990; Bruenig et al. 1995). Mismatched cycles between resource and market, as well as lopsided trading relations, have effectively curtailed forest management in many Indo-Malayan countries (Jepma 1995). Freedom from debt and more equity in international markets would allow these countries to actively promote ecosystem management and even forest restoration. One possibility would be to exchange an amount of debt for an amount of conservation – essentially a "debt for nature swap" as has been tried in South America (Hurst 1990). Halting destructive development projects, particularly those funded by foreign lenders, requires the understanding that forest conservation takes a definite backseat to self-interest when development policies include provisions for personal gain from forest conversion and exploitation. This would entail a certain amount of self-restraint on the part of foreign lenders, and perhaps political pressure from the conservation community. Finally, patterns of land ownership in Indo-Malaysia currently promote destruction of forests due to the dramatic increase of landless farmers seeking their own livelihoods (Hurst 1990).

There are two economies in Indo-Malaysia—market and subsistence. Displacement of the subsistence economy with the market economy, which typifies those countries seeking to increase exports to alleviate debt, does not benefit the rural poor and is at the heart of the farmers' landlessness (Hurst 1990). Adequate land must be afforded for the subsistence of a growing population. Although such land reform will face severe opposition from the powerful land-owning class, it is necessary to meet the needs of people who would otherwise continue to encroach on both natural and restored forests (Rai 1993; Lanh 1994; Andersen 1995; Rasheed 1995). Thus, in addition to the significant Ecological constraints, conservation and restoration in Indo-Malaysia would seem to depend on significant economic and land use reform.

Conclusions

The restoration of Indo-Malayan forest presents a unique and specific set of ecological problems. The reproductive and colonizing characteristics of dipterocarps, which often form the backbone of these forests, are not favored on cleared land. Irregular flowering, poor dispersal, recalcitrant seeds, and limited coppicing capacity all work against restoration efforts. Due to dipterocarp value in the timber trade, its reproduction and regeneration has been intensively researched by the forestry community. The limitations of irregular flowering and recalcitrant seeds have, so far, not been solved. Poor dispersal and limited coppicing restricts natural forest regeneration to small cleared patches at best. Planting of seed trees or perch sites in the open are largely ineffective for dipterocarp establishment. However, larger cleared tracts could benefit from dipterocarp underplanting in plantations. Dipterocarp seedlings have been shown to successfully establish under plantations and thereafter respond to increased light with rapid growth. Plantations also can provide two additional benefits. Plantation cover can in some areas successfully shade out imperata grass, which dominates severely degraded cleared lands and suppresses forest regeneration. The cleared plantation overstory also can provide an economic resource to be put back into the restoration efforts.

Maybe more than in any other region, social and economic constraints significantly affect forest restoration attempts in Indo-Malayan countries. The remnants of colonial occupation in Indo-Malayan countries have created a timber industry that is foreign controlled, capital intensive and primarily exporting raw materials; government policies whereby debt is accumulated in order to ensure continued "development;" and insecure land tenure arrangements. These factors contribute immensely to deforestation and hinder efforts at reforestation. Important strategies that should be utilized in Indo-Malayan countries include alleviating the debt crisis, reforming trade conditions, halting environmentally damaging development, and securing long-term land tenure arrangements. It is doubtful that restoration without the complement of economic and land use reform will be very successful.

1-4. Tropical and Subtropical Moist Broadleaf Forests—Australasia

Ecological Setting

The largest expanses of tropical moist forest in Australasia are located on Australia and on the island of New Guinea which is divided in half by the countries of Indonesia and Papua-New Guinea. Subtropical rainforests also occur on New Caledonia and New Zealand's north island (see Plate 4). While the dominant forest type of New Guinea is rainforest which still covers greater than 80 percent of the land mass, less than 1 percent of Australia's tropical vegetation was originally rainforest (Stocker and Unwin 1989). New Guinea's climate is equatorial while Australia, being farther from the equator, has a strong seasonality even in the rainforests. New Guineas forests have features in common with Indo-Malayan, Oceania, Australian and African forests. The diverse canopy species include dipterocarps, eucalypts and members of the Nothofagus genus (Collins et al. 1991). Australia's warm rainforests contain elements of New Guinean forests as well as Australian Gondwanaland flora. Broadleafed trees predominate in New Caledonias mixed species canopy, but a diversity of conifers is also present (Mueller-Dombois and Fosberg 1998). Locally, single tree dominant stands are found made up most commonly by Nothofagus. The rainforests of New Caledonia have a very high rate of species endemism, approximately 82 percent (Jaffre et al. 1998).

Australia's tropical rainforests are generally restricted to coastal regions of the north and northeastern part of the country that receive high rainfall. Except for the Wet Tropics in the Cairns region, the rainforest generally forms disjunct patches among drier ecosystems dominated by eucalyptus forests and woodlands (Webb and Tracey 1994). The rainforests here are usually bordered by grasslands or eucalypt communities which are naturally maintained by regular fire. The border between these ecosystems is often unusually sharp with narrow ecotones (or transition zones). Fires in the bordering communities can kill rainforest vegetation along the edges. During drought years or following large-scale disturbance events such as cyclones, large areas of rainforest become susceptible to fire. Though grasslands and eucalypts frequently replace burned over lands, rainforest vegetation eventually regenerates and succeeds these communities over time in the absence of further disturbance. The border between rainforest and adjacent ecosystems therefore is dependent upon site conditions and frequency of disturbance, especially fire. The fire regime has generally become more frequent since European colonization, with a corresponding loss in rainforests areal extent. At border locations protected from fire, rainforest has naturally re-established.

The forests of New Guinea are subject to continual disturbance from cyclones, earthquakes, volcanoes and landslides (Collins et al. 1991). The frequent disturbance regime is reflected in the abundance of certain gregarious tree species. Increased logging by multinational corporations is a very real threat to the remaining rainforests of New Guinea. Only a fraction of New Guinea's forests are under active management or included in protected areas. Traditional shifting agriculture has been practiced on New Guinea for millenia and much of the lowland rainforest is secondary in nature, but the recent pressures on the native forests are unprecedented.

The relatively small, total land areas occupied by rainforest on New Caledonia and New Zealand, coupled with the geographic isolation comprise a significant threat to preservation of these ecosystems. Fragmented and covering 20 to 25% of the land area, less than half of New Caledonias original rainforests remain (Jaffre et al. 1998).

Restoration and Management

Of the 1.2 million hectares of rainforest located in Australia prior to European colonization, 80 percent remains as selectively logged or virtually untouched in National Parks and in inaccessible regions (Stocker and Unwin 1989). Nearly all of the tall forests though, have been cleared or logged. World Heritage designation for the Wet Tropics region in 1988 prompted a complete halt of logging in these rainforests. The aboriginal peoples of Australia did not practice shifting agriculture, though they did use fire. Studies show the ability of rainforest species to regenerate vegetatively in Australia. In a 1 ha clearing in Queensland rainforest, created by felling and burning, 90 percent of the tree species regenerated from coppice shoots while 41 percent grew from seed (Stocker 1981). These results show that rainforest species in these regions, normally thought of as fire sensitive, readily regenerate after a single burn.

Diversity and density of plant recolonization was greater beneath a variety of planted native trees than on unplanted, older control sites, which were still dominated by grasses and vines, at four restoration sites in North Queensland, Australia (Tucker and Murphy 1997). The tree taxa used in this "framework species" method (a method devised by Goosem and Tucker (1995) to test several native tree species with the goal of promoting more natural forest succession) did not produce a significant difference in diversity between the four, young tree planted sites. However in a chronosequence study, significantly greater diversity developed beneath monoculture timber plantations of native species compared to the exotic conifer *Pinus caribaea* at four sites in the Atherton Tablelands of North Queensland, Australia (Keenan et al. 1997). The greatest diversity occurred in the oldest plantations (63 years) and beneath the two native, broadleaf species (*Flindersia brayleyana* and *Toona ciliata*) where 51 percent of the species identified in the surrounding rainforest were present. The broadleaf species had less dense canopies and thinner litter layers than the conifers.

In Papua New Guinea, natural regeneration of small patches (<400 m diameter) of logged over forest can proceed rapidly. After six months most of the area was covered by 1 m tall plant growth and by ten years well-drained sites had a secondary tree canopy approximately 20 m tall. Tree species richness increased with time, equaling that of the unlogged forest after ten years. However, 70 percent of the tree species present in the unlogged forest were primary (late successional) compared to only 40 percent in the logged areas. The majority of the secondary (early successional) seedlings developed from the soil seed bank, while at least 50 percent of all tree species could regenerate vegetatively, especially primary species (Saulei 1985, from Saulei and Lamb 1991). The density of tree regeneration was strongly correlated with the area of soil disturbance in Papua New Guinea. Sites with greater than 80 percent of the area disturbed had 50 percent or less the tree density compared to sites with less than 40 percent of the area disturbed (Saulei and Lamb 1991). Birds and bats have been identified as the primary dispersal agents of 35 percent of the local tree species, while 50 percent of the species apparently have no specialized dispersal mechanism (Saulei and Lamb 1991). These species may be under-represented in larger cleared sites. Broadleafed trees, often *Ficus sp.* in this area, were observed to be focal points for bird activity. These sites, left to regenerate following a single disturbance, appear to be substantially recovering both compositional and structural diversity. Disturbed sites 55 years into natural regeneration had yet to achieve the tree size, canopy height and proportion of primary tree species as forests with no evidence of disturbance (Saulei 1985, in Saulei and Lamb 1991). The small site size (5,000m^2) and proximity to intact forest likely facilitated dispersal into the planted restoration sites in a study conducted by Tucker and Murphy (1997). The one site that was isolated (500 m distant) from surrounding forest developed lower species diversity and depended more on vegetative regeneration. Dispersal ability was important considering that the soil seed bank consisted of 82 percent exotic species with only two native trees. Bird dispersal was predominant here and in the Keenan et al. (1997) timber plantations where 80 to 90 percent of the colonizing tree species were primarily bird dispersed. In the majority of the timber plantations, colonizing diversity dropped off between 100 to 200 m from the edge of the surrounding forest. Mammal dispersed species were not present except minimally in older plantations.

Given the socioeconomic and political constraints, large projects may not be possible throughout many of these forests (Lamb et al. 1997). Plantings do not need to be large, however, if strategically located with the right species compositions (Tucker and Murphy 1997). Also, well-designed plantations may be very effective in alleviating consumptive pressures on valuable natural forest areas (Lamb et al. 1997; McDonald 1993). Large seeded species with poor dispersal need to be included in reforestation plantings. In trials of seven native rainforest species, grow tubes greatly enhanced the survival through one year (Applegate and Robson 1994).

Since the 1980s, control of Australia's forests has been shifting from individual states to the national government (McDonald 1993). The Australian government, however, has no history of forest policy decisions outside of converting forestland to agricultural land, and divides land use responsibility between several distinct governmental sectors – Agriculture, Forestry, Conservation, and Soil and Water (McDonald 1993). This means that land use decisions are still, for the most part, *ad hoc* and often conflicting. In general, however, decisions on all levels – national, state, and local – are made toward a common goal of self-sufficiency (McDonald 1993). Toward this goal, plantations and revegetation schemes are conducted for both timber production and conservation ends.

Recently – since logging in state-owned forests ceased following World Heritage listing of rainforests in 1988 – the Australian government has initiated two programs, the Community Nature Conservation Program (CNCP), and the Community Rainforest Reforestation Program (CRRP), aimed at mobilizing community support to restore forest on cleared or degraded areas in northern Queensland (Lamb et al. 1997). The CNCP promotes nature conservation outside protected areas, and has developed a specialized habitat restoration nursery. For example, the CNCP is using the framework species method to reforest a 100 m wide, 2 km long corridor connecting a small (500 ha) conservation reserve, Lake Barrine National Park, with a nearby continuous forest in the Donaghy's Corridor region. The major limitations of this approach are its expense and the small number of private landowners willing to commit large areas for conservation purposes. The CRRP recruits landowners to plant timber plantations of mixed native forest species. As of 1997, 1,500 ha had been planted across the Wet Tropics region. The question of harvest versus long term conservation in these plantations has not yet been addressed.

As seen in most other forested Major Habitat Types, in order to succeed, these reforestation strategies must involve local communities. However, involving local communities may mean a shift in emphasis from biodiversity conservation and forest integrity to economic productivity (Lamb et al. 1997). Although this compromise may be necessary, particularly in areas where deforestation and degradation are not matched by restoration, care must be taken not to allow the complete displacement of conservation goals by economic objectives.

New Zealand, by contrast, saw a shift in the mid-1980s from state afforestation to multinational corporatist forestry (Roche and Heron 1993). New Zealand forestry, therefore, is heavily laden with capitalist priorities and profit-motive decision-making, while reforestation has taken a backseat (Roche and Heron 1993). The shift has altered both the decision-making landscape and prospects for employment in the forest industry. People do not necessarily recognize benefits from reforestation, and consequently, there is little incentive to plant trees and even less to manage the resultant vegetation (Roche and Heron 1993). In both Australia and New Zealand, the conservation community will find strong and adamant opponents in the timber industry.

Conclusions

The moist forests throughout much of Australasia are currently largely intact, though the threat of deforestation is still eminent, particularly logging in New Guinea. Secondly, cleared lands have been shown to regenerate forest rapidly, provided degradation is not severe and further disturbance is eliminated. The capacity of many rainforest species to regenerate vegetatively is essential in this regard. In Australia, protection from dry season fire is the most critical factor in forest restoration. At locations protected from fire, rainforest has supplanted eucalypt communities which originally established due to anthropogenic fire regimes. Rainforest patches on public lands can be enlarged by natural regeneration through the management of forest borders for fire protection. On more severely degraded lands, where the natural regeneration capacity has been compromised, plantations may be useful in facilitating colonization of open land. Additionally, research has shown colonization is more diverse beneath plantations of native trees than exotics. Restoration should be concentrated in the vicinity of existing forest patches to take full advantage of natural dispersal. Though the efforts of private organizations, such as CRRP and CNCP, have met with limited success to date, initiatives that involve the local populace should be pursued.

Since most of New Guinea's forests remain intact with little protection and under great threat, increased protection should be addressed immediately. New Caledonias rainforests, though limited in total relative area and already heavily disturbed as a forest type, are reasonably well captured in protected areas but not in a representative fashion (Jaffre et al. 1998) – only 24 percent of the rainforest species of concern are represented in existing conservation areas. Rainforest conservation outside of protected areas will be possible only if the logging pressure is reduced and restoration efforts successful. Reclamation of the most severely damaged lands with plantation forests may prove most useful. Jaffre et al. (1998) further suggests that all conservation areas should protect both flora and fauna, as opposed to one or the other as is now the case in parts of this MHT. Control of fire, which constitutes the greatest threat to the forests of New Caledonia, is essential for conservation. Finally, the restoration of cleared lands will require additional research on the ecology and reproductive characteristics of these forests.

1-5. Tropical and Subtropical Moist Broadleaf Forests—Oceania

Ecological Setting

The moist forests of Fiji, Tonga, Samoa, American Samoa and Hawaii are isolated on small islands scattered in the vast Pacific (see Plate 5). The small land area and isolation of each island group contributes to high species endemism. The island archipelago of Hawaii is the most isolated in the world, with approximately 90 percent of its nearly 1,000 native plant species being endemic. Historically, lowland moist forests occurred on the wetter, windward sides of the major islands, often with moist cloud forests of lesser extent at higher elevations. Only a fraction of the lowland forests remain. Fiji still retains a variety of more or less naturally functioning ecosystems (Collins et al. 1991), and moist forest is still widespread on the inaccessible eastern ridges and cliffs of Eua, the second largest island of Tonga (Mueller-Dombois and Fosberg 1998). Tropical storms cause frequent minor and occasional major disturbance on Fiji and Tonga, but have less impact on islands further to the east, such as Samoa and Hawaii (Ash 1992; Mueller-Dombois and Fosberg 1998). Major climate fluctuations such as El Nino and La Nina, causing precipitation extremes, stress the moist forests of all of the islands. Protected areas are generally inadequate in number and extent, especially for cloud forests, and the long-term legal status and management of existing sites is often insecure.

The different oceanic islands support a variety of moist tropical forests due to different climate, edaphic, and ecological conditions specific to each forest ecosystem. The critical features shared by these ecoregions are the relatively small land areas involved and the extreme isolation of each ecoregion. The limited extent of original forest has generally resulted in greater destruction and a relatively lower percentage of remaining forest compared with larger ecoregions. Conservation and restoration of these forests is further exacerbated by the characteristic high endemism which creates such high diversity values for these ecosystems. With each being so unique and so little remaining, sources for plant community recolonization are extremely limited. Many Pacific islands are relatively young and resulting in shallow soils which are especially susceptible to erosion on steep slopes (Mueller-Dombois and Fosberg 1998).

Restoration and Management

Shifting agriculture, practiced for up to several thousand years, initiated forest reduction and alteration on these island ecosystems. Europeans introduced cash crops and plantation forestry, which accelerated deforestation of native forests and limited secondary forest regeneration of fallows. Logging of valuable timbers and associated road building has further opened up previously inaccessible forests to exploitation, settlement and clearance (Mueller-

Dombois and Fosberg 1998; Ash 1992). Along with land conversion, exotic introductions, especially since European contact, have produced the most devastating impact on the original forest communities. The number of naturalized exotic species introduced to Hawaii over the last 200 years nearly equals its number of native plant species. Nearly 30 percent of Hawaii's native plant species are listed on the Endangered Species List. Introduced animals such as cattle, goats and pigs not only destroy native flora but assist in the dispersal of exotic plant species.

Left to themselves, many moist forests seem capable of rapid regeneration to varying degrees. Frugivorous birds and bats are the major seed vectors of the Fiji forests, which do not seem to require disturbance for succession (Ash 1992). Different islands likely evolved with different combinations of dispersers. Native canopy species are still able to reproduce in small remnant patches on Tongatapu, the largest island of Tonga (Palmer 1988 in Mueller-Dombois and Fosberg 1998). Major canopy species 'ohi'a (*Metrosideros polymorpha*) and koa (*Acacia koa*) of Hawaii are also pioneer species with the added benefit of koa being leguminous. These two taxa can play dominant roles in both lowland and montane cloud forests (Mueller-Dombois and Fosberg 1998).

Many management efforts on Hawaii have emphasized the removal and exclosure of exotic species but have generally failed to nullify the problem. Exclosures of cattle and feral animals on Hawaii have produced mixed results of koa regeneration while soil scarification has been shown to greatly enhance seedling production (Stone and Scott 1985). Trials of colonization under monoculture plantation canopies of three different exotic species has also produced mixed results on Hawaii (Harrington and Ewel 1997). In this case, small, 16 ha plantations were surrounded by native moist forest invaded by exotic species. The *Fraxinus uhdei* plantation produced the only significant native undergrowth while all three fostered exotic undergrowth to varying degrees. *F. uhdei* also showed the ability to invade native koa stands at other locations. Reforestation under these conditions will still require control of exotics.

Despite the potential resources at hand, public education on the conservation crisis in Hawaii is woefully lacking (Gagné 1988). A bright spot has been the scientific research and the volumes of literature produced. Originating with the nene or Hawaiian goose (*Nesochen sandvicensis*), conservation has focused on endangered species which has resulted in habitat protection. However the pace has slowed to a crawl for the hundreds of plants and animals up for review for endangered species listing. At the state level, funding has been inadequate for protected areas, while the National Park Service and U. S. Fish and Wildlife Service remain at the mercy of Congressional political whims. Of 180 endangered communities identified throughout Hawaii, only 48 were under any level of protection (Gagné 1988). Private organizations, particularily The Nature Conservancy of Hawaii, are also involved in land aquisition and protection.

Western-style conservation may be inappropriate for much of tropical Oceania. As in many other regions of the world, incorporation of both the knowledge and belief-systems of local villagers is a minimal requirement to the long-term success of conservation efforts. The islands of Samoa offer a good example of the need to respect local views and priorities in island conservation efforts (Cox and Elmqvist 1997). Several village-based preserves in Western Samoa have proven to be effective tools in conservation the island's rainforests. However, establishing the preserves involved conflicts between the local leaders and western conservation organizations over control and management of the preserves. By expressing a certain distrust of local leadership and trying to change, rather than understand and reaffirm local perspectives, the conservation community actually succeeded in offending the village leaders (Cox and Elmqvist 1997). Ultimately, local control was difficult for the westerners to swallow and the resulting dysfunction almost completely severed relations between the organizations and the villagers. After careful discussion (and apologies), decision-making authority was finally vested in the local leadership (Cox and Elmqvist 1997). The experiences in western Samoa provide a classic study and highlight the need for respect of local belief-systems in conservation efforts, particularly when foreign money, no matter how well-intentioned, can undermine cooperation and local participation (Cox and Elmqvist 1997).

Conclusions

Essentially, any conservation-restoration program developed for forests of the Oceania islands must deal with the overriding issue of pervasive exotic species. That aside, each island exists under a particular set of conditions that must be addressed separately. Significant intact tracts of mature, rainforest remain on the island of Fiji; however, the few thousand hectares currently preserved in perpetual reserves is totally inadequate and the remaining protected areas are not secure in the face of future logging pressure (Ash 1992). Establishment of limited plantation and production forests in conjunction with enhancement of the protected reserve system would likely ease the future pressure on native forests. Although native forest is still capable of self-regenerating on Tongatapu, very little remains, and the island faces increasing population pressures. Tongas conservation efforts may be better invested in the more extensive and remote forests on nearby Eau.

Hawaii has widespread canopy tree species in the koa and 'ohi'a that are also pioneers and can be utilized as nurse trees to facilitate forest restoration. However, the problem in restoration experiments so far has been the invasibility of exotics. Initially however, Hawaii's protected areas are in need of upgrading and long-term funding.

2-1. Tropical Dry Forests—Neotropical

Ecological Setting

Once widespread, the tropical dry forest is one of the most endangered habitat types in the neotropics and throughout the world. Extensive tracts once extended along the Pacific side of the Americas from Mexico to northern Argentina. As of 1988, less than 2% of Central America's original 550,000 km remained in relatively undisturbed wildlands and only 0.08% was protected in national parks or other conservation areas (Janzen 1988a). Remnants of these forests are also located in Mexico, the Tumbesian and North Inter-Andean Valleys dry forests region of Peru and Ecuador, and the Bolivian lowland dry forests (see Plate 6).

Dry forests of the tropics and subtropics are typically smaller in structure and simpler in composition than moist forests for a given region (Murphy and Lugo 1995). Differences in climate, which is frost free and strongly seasonal in terms of precipitation, leads to enormous variation among forests. In addition to the wide latitudinal extent, the particular climate of neotropical dry forests is usually a result of the regional topography. The mountainous spine, which runs down Central America to the Andes extends the length of South America on the Pacific side, creates a barrier to Atlantic moisture carried with the prevailing east-northeastly winds, forming dryer regions on the western slopes. Classification schemes emphasizing different features or combinations thereof (not always overlapping), including precipitation levels (water limitation), seasonality and vegetation composition have been used to describe these forests (Murphy and Lugo 1995). Dry forests are often referred to as deciduous forests, but the degree of deciduousness can vary greatly.

Functional traits of these moisture-limited forests are closely tied to climatic seasonality (Murphy and Lugo 1995). Growth occurs during the wet season and abundant litterfall is produced during the dry season. There is little evidence to suggest intact dry forests were historically subject to natural fires. However, fire is of greatest significance in forestlands proximate to human disturbance, especially rangelands.

Once the threat of fire, the most critical threat to dry forests, has been eliminated, regeneration of forest cover progresses rapidly (Janzen 1999; Murphy et al. 1995). However, the natural regeneration potential to historic late successional dry forests is often severely compromised by a lack of propagule sources and dispersal mechanisms. The largest remaining forest remnants are scattered along the west coast of Central America, particularly at Guanacaste, Costa Rica. Elsewhere, with virtually all of the original late successional forests destroyed, little remains as a source or for comparison with currently regenerating forests. For example, secondary forests, 150 years in age on the island of St. John, U.S. Virgin Islands are considered no more than half way to maturity (Ray 1993). Puerto Rico's "mature" forests are merely 50 years in the making (Murphy et al. 1995).

Natural regeneration depends upon dispersal, coppicing or soil seed bank viability. Seeds of mature dry forest species are often short lived, less than nine months (Ray 1993) and poorly represented in the soil seed bank of abandoned lands (Rico-Gray and García-Franco 1992; Ray 1993). Wind dispersal can move seeds in quantity up to 200 m into open sites (Janzen 1988b). However, most mature canopy species rely on animal dispersal of seeds, which progresses more slowly particularly on open sites. Pasture trees can facilitate dispersal by acting as magnets for animals particularly birds (Janzen 1988b).

Vegetative sprouting plays an important role in natural regeneration, accounting for up to 50 percent of woody growth (Murphy et al. 1995). However, long-term land use can have a detrimental effects on sprouting ability. Repeated fires have been suggested to cause a shift to sprouting species that can survive regular disturbance with a loss of late successional species (Rico-Gray and García-Franco 1992). The result is secondary forest, 100 years of age, dominated by early successional species.

Restoration and Management

Though forest cover may be regenerated rapidly, recovery of mature composition and structure is a long-term process. After 26 years, only one shade tolerant, canopy tree species had colonized the secondary forest studied in the Yucatan, Mexico (Mizrahi et al. 1997). While on the island of St. John, U.S. Virgin Islands, a broad range of native species, including shade tolerant trees, colonized abandoned fields after 50 years (Ray 1993). Even at 100 years these forests still have a high percentage of saplings. Forest succession is expected to require from 100 to 1,000 years at Guanacaste, Costa Rica (Janzen 1988a).

Initial studies in dry forest restoration indicate that planting of seedlings provides a distinct advantage over seed introduction and stem cuttings (Ray 1993). The use of seeds was limited by short seed life, usually less than nine months, and vulnerability to drought stress. Cuttings exhibited much lower survivorship than the overall 52 percent survival of seedlings. However, cuttings can still provide a useful alternative where seed sources are limited as in certain wind-dispersed species. Seedlings also suffered from drought stress, and though growth was slowed, survivorship was higher in shaded conditions (Ray 1993; Gerhardt 1996). These results confirm the practicality of underplanting extant vegetation or providing initial canopy cover for forest restoration. Nurse tree plantations are currently proposed for forest restoration programs at Guanacaste (Janzen pers. comm. 1999).

The native peoples of neotropical America have practiced shifting cultivation in the dry forests for thousands of years. Extensive habitat destruction in Central America began in the sixteenth century with the first Spanish cattle ranches. Cleared dry tropical forestland provided prime ranching and agricultural land and use therefore was long-term. By the end of the nineteenth century forest fragmentation was already extensive, as cleared land was easy to maintain through felling, fire and introduction of exotic African grasses (Janzen 1988a).

The Central American tropical dry forest ecosystem extends north into southern Mexico before giving way to the Sonoran Desert. Though Mexico has no comprehensive plan for preserving this critically endangered ecoregion some steps are being taken through the establishment of new reserves. With an approach similar to that of Guanacaste illustrated below, a collaboration of the National University of Mexico, NGOs and several private companies worked together to establish the Chamela-Cuixmala Biosphere Reserve. Land tenure was resolved by using entirely private land. Enhancing the welfare of the local populace is suggested as necessary for the long-term success of the reserve. Additionally, the strategy must be based on scientific knowledge and shaped into a regional socioeconomic perspective. As is typical with other regions, most of the native dry forest has long since been cleared for agriculture and grazing. Well preserved forest remnants including nine different major vegetation types are included within the 13,200 ha reserve. Chamela-Cuixmala is becoming a model for Mexico gaining support through a legal and public campaign that used information about the region's uniqueness and biological value. Unfortunately, knowledge of sustainable use of local natural resources is lacking at this time (Ceballos 1995). To protect the range of habitats found in this ecoregion, several additional reserves must be established in a longitudinal gradient from Sinaloa to Chiapas.

With the establishment of Guanacaste Conservation Area (ACG) in 1987, Costa Rica took ecosystem restoration to a larger scale and new level of comprehensiveness. It provides one of the best examples of widespread forest restoration applied research in the world. The primary purpose is to use existing dry forest fragments as seed nuclei to restore the surrounding degraded lands (Janzen 1988a). Located in western Costa Rica, Guanacaste includes a variety of habitats in 120,000 ha and stretches from Pacific Ocean to continental divide. Guanacaste contains some of the last dry forest remnants including the pre-existing Santa Rosa National Park (108 km). The two basic tenets of technically restoring Guanacaste are restoring the size of the ecosystem and eliminating the regular anthropogenic fires. However, the basis for long-term success has been in gaining the support of the local populace through socio-economic integration at all scales and developing financial sustainability for ACG.

The socio-economic integration of ACG has led to the unending quest for uses of the forest that are non-damaging, with the reality that 5% of the biodiversity and ecosystems will be sacrificed to guarantee the remainder. ACG contributes to the local economy by employing only Costa Ricans, including 82% from the immediate region. Increased regional income is also derived from ecotourism and services associated with biodiversity prospecting. ACG also teaches biology to more than 2,000 students yearly living in the vicinity. The conservation area garden also has its own public reading rooms (Janzen 1999). While arrangements with the biodiversity industry and debt for nature swap help pay the bills, Guanacaste still relies on an endowment from international donations.

The first tenet for restoring the dry forest ecosystem at Guanacaste was to establish a large enough area based on five biological principles: (1) maintain habitat diversity; (2) maintain adequate species population sizes; (3) provide dry season refugia and migration routes; (4) minimize edge effects; and (5) maintain large and replicated habitats for human users (Janzen 1988a). The existing Santa Rosa National Park at 10,600 ha contained only portions of major habitats, was mostly edge, and was far too small to absorb the necessary human uses. Expansion was especially needed to the wetter east. Much of its more mobile dry forest biodiversity, especially its insect and bird fauna, seasonally migrate to the rainforests and cloud forests across the mountains to the east and return for the six-month rainy season (Janzen 1999). The borders of Santa Rosa were not determined by ecology, but rather the social reality of surrounding high quality private agro-lands. ACG incorporated Santa Rosa and several other semi-conserved reserves by purchasing the low quality, private agricultural lands in between. This process in itself intertwined the ACG with the local community.

With 120,000 ha and the ability to absorb small-scale disturbances, the goal of ecosystem restoration could be embraced. ACG contains remnants of all original dry forest habitats scattered in small to large fragments, the most significant occurring in Santa Rosa. Centuries of clearing, burning and grazing have created a patchwork landscape that includes at least 50,000 ha of old fields and pastures dominated by the exotic jaragua grass, which is capable of preventing forest reestablishment. The dry forest did not naturally evolve with fire – seedlings are particularly susceptible to intense dry season fires fueled by the jaragua grass. Elimination of regular anthropogenic fires was a critical first step in restoring this forest. Local staff were sufficiently budgeted and vested with responsibility of a region wide education program on the value of fire elimination. Today, ACG is essentially fire free resulting in 40,000 ha of rapidly regenerating young forest (Janzen 1999). However, forest regeneration in the wetter forest pastures in the eastern section of ACG is proceeding at a slower pace.

It is interesting to note that grazing cattle actually prevent more intense fires by reducing the fuel load of the pasture grass. This was discovered in 1977 at Santa Rosa when 2,000 semi-feral cattle were removed prior to fire control. As a result as many as 7,000 cattle were grazed in ACG during its early years until the fire control program was in place (Janzen 1999). Additionally, livestock at low to moderate density further facilitated woody invasion by reducing the amount of suppressive grasses.

Forest invasion and regeneration of abandoned fields and pastures was the subject of intensive research at Santa Rosa even prior to the creation of ACG. Experimental exclusion on a 4 ha jaragua grass pasture over a period of five growing seasons was sufficient to produce a rapidly closing stand of young trees. It is predicted that with the elimination of all fire and livestock, that small pastures (<10 ha) would largely revert to woody vegetation within 20 years, while the largest pastures would require 50 to 200 years. The entire area will require at least 100 to 1,000 years to reproduce the structure of the original dry forest (Janzen 1988a).

With the elimination of fire, wind-dispersed seeds moved in abundance up to 200 m. However, these plants do not disperse readily into the center of larger pastures (hundreds of hectares). A variety of animals also disperse larger seeds. The guanacaste and cenizero trees often occur as isolated trees in large pastures, and are almost exclusively dispersed by horses and cattle. These isolated trees are magnets for animal dispersers crossing the pasture especially birds and bats. The result is progressively growing patches of animal dispersed vegetation with a guanacaste or cenizero tree at the core. The result is woody succession with a species structure somewhat different from that of wind dispersal or in the absence of grazing.

A native tree planting program was also established to facilitate the dry forest establishment. Between 4,000 and 9,000 seedlings are planted each year on denuded land in groupings to mimic natural groves. These trees are expected to establish a future seed source since only 23 of the 321 forest tree species disperse by wind. Three-year survival of seedlings of four monitored species has ranged from 3 to 55 percent (Tenenbaum 1994).

Forest invasion of open lands has proceeded extremely slowly in the rainforest area at the eastern end of ACG. This region is a critical refugia and migratory link for area organisms during the dry season and drought years. A variety of techniques, including plowing, seeding and forced burning techniques have proven ineffective at speeding up the reforestation process. ACG has recently obtained funding to initiate a rainforest regeneration project utilizing gemelina plantations as a nurse crop. Further, the wood provided by the gemelina will provide a living endowment for future ongoing reforestation (Janzen pers. comm. 1999).

One final example worth noting is the establishment of the 3.44 million hectare Kaa-Iya del Gran Chaco National Park in Bolivia in 1995. The local Indian organization Capitania del Alto y Bajo Izozog (CABI) fought for this park and now has been charged with its management. This will assure local support for the park unlike other reserves in Bolivia.

Conclusions

Fragmented, under threat, and with so little natural habitat remaining, immediate protection of the last sizeable tracts is essential for future conservation of this ecosystem. Additionally, conservation areas should incorporate reforestation to expand the extent of remaining tropical dry forest, thereby increasing the chances for preservation. The establishment and successes of Guanacaste provide immeasurable support to similar programs elsewhere. Additional reserves along the lines of GNP and Chamela-Cuixmala need to be established to further the protection of dry forests. Furthermore, the involvement and support of the local community has been demonstrated to be a necessity.

2-2. Tropical Dry Forests—Afrotropical

Ecological Setting

Dry forest was once the dominant native vegetation of western Madagascar, the fourth largest island in the world (see Plate 7). Madagascar is home to over 12,000 species of flowering plants of which 85 percent are endemic (Guillaumet 1984). The diversity and endemism of Madagascar flora rivals that of South African fynbos.

The critical feature of Madagascar is the north - south oriented central highlands which interupt the prevailing winds and form a rainshadow over the entire western half of the land mass (Paulian 1984). Most of the precipitation derived from the Indian Ocean falls on the rainforests of the eastern part of Madagascar and generally decreases to the west. Remnant deciduous dry forests are characterized by low to moderate rainfall and a pronounced winter dry season that may last more than seven months (Smith 1997).

The structure and composition of the dry forests is extremely variable. In addition to the climate, edaphic differences are largely responsible for variation in forest types (Guillaumet 1984). Sedimentary substrate characterizes most of the deciduous dry forest. Evergreen components become more important in wetter and more sheltered climates, and on more fertile basalt derived soils (Smith 1997). Forests dominated by bottle form trees and the unique baobabs (genus Adansonia), of which Madagascar is home to seven species, are more prevalent on less fertile, sandy calcareous soils. With decreasing precipitation, the dry forests grade into the equally unique and endangered thorny scrub desert in the southwest which has 95 percent endemism.

Restoration and Management

Although deforestation is widespread on Madagascar, the rainforests on the island have generally received the bulk of the conservation attention, but it is the dry forests that are likely the more endangered forest ecosystem type. At the time of arrival of the indigenous population 2,000 years ago, it is thought that Madagascar was almost entirely forested (Paulian 1984; Smith 1997). Shifting agriculture and a frequent anthropogenic fire regime have been the primary causes of deforestation. By 1950, dry forests were reduced to 12.5 percent of its original coverage. Exacerbated by high population growth among the largely rural populace, continuing deforestation reduced the dry forest to 2.8 percent of its original extent by 1990 (Smith 1997). Firewood availability has become a critical threat since the principle domestic energy source is charcoal.

The resulting savanna grasslands and mosaics of savanna and secondary forest are maintained with regular burnings, and disturbed areas are rapidly invaded by exotic plant species. Smith (1997) reports that these dry forests seem unable to regenerate on their own due to the absence of sufficient native pioneer species to facilitate natural succession. However, Paulian (1984) reports that natural regeneration is possible wherever land is protected from fire, and this assertion is substantiated by photographic evidence from the reserve of Namoraka revealing recovery of secondary forest over a period of 50 years. In general, regrowth of primary forest is extremely slow. Little is known of the majority of the forest flora, much less their ecology and regeneration dynamics.

For years a model of destruction, Madagascar has recently become a model of local community orientated conservation.with the implementation of a country-wide environmental action plan (Wright 1997) Long-term integrated conservation and development programs (ICDPs) involve the local communities in comprehensive conservation of selected protected areas.

Conclusions

Conservation efforts in the African dry forests face similar constraints as those in the Afrotropical moist forests. Local people must be involved. For example, in Natal of South Africa, the rural poor are unlikely to comply with rules restricting access to natural resources unless they recognize the benefits of doing so (Wynne and Lyne 1995). This is also true in Madagascar where the dry forests are more degraded than the moist forests (Sussman and Rakotozafy 1994). Long-term conservation can only succeed if the attitudes and technical knowledge of local people are understood and incorporated in the conservation activities (Hawkins et al. 1990). Recently initiated ICDPs provide hope for the future conservation and management of these forests. Restoration for the purpose of enhancing tourism is one possible route, but only if local perspectives are respectfully integrated and the benefits return to the local community (Hawkins et al. 1990) and the ecological challenge in restoring these forests are overcome.

2-3. Tropical Dry Forests—Indo-Malayan

Ecological Setting

The largest tracts of dry forest in the Indo-Malayan realm are found in eastern Indochina as characterized by the forests in Thailand. Additional dry forests of concern are located in eastern India (Eastern Ghats) and the Lesser Sunda Islands of Indonesia (see Plate 8). Dry forest ecosystems comprise approximately two-thirds of the total historic forest area of Thailand and to a lesser extent in neighboring Cambodia, Laos and Vietnam. These forests are comprised of three dry forest types (Rundel and Boonpragob 1995). The first is a mixed deciduous forest type dominated by teak. A second is a dry evergreen forest type which is an extension of lowland moist forest and varies from 100 percent evergreen to an evergreen-deciduous mix. The last is a deciduous dipterocarp community forming a more open forest type adapted to a regular natural fire regime. In all cases, a strong seasonal climate prevails throughout this region, and monsoons create significant natural disturbance.

Soils underlying the dry forests of Thailand are mostly shallow and poor in available nutrients and water holding capacity (Rundel and Boonpragob 1995). The forests in Laos and possibly elsewhere share these characteristics (Collins et al. 1991). Dry forests are all adapted to natural fire to varying degrees (Stott et al. 1990; Rundel and Boonpragob 1995). Fires vary markedly in character depending on a range of variables, mainly timing, fuels and topography. The natural, regular fire regimes keep the fuel load low resulting in low to moderate fire intensity. Under these regimes, seedlings often have the opportunity to reach sapling size (greater than 1 m) at which stage survival of low to moderate intensity fires is much greater. However, most fires now have human origins and burn more frequently making seedling establishment difficult. The more frequent fire regime in Thailand is suggested as a major factor in conversion of dry evergreen and mixed deciduous forests to secondary mixed dipterocarp forests and eventually savanna.

Restoration and Management

Eastern Indochina, in particular, was likely to originally have been almost entirely forested. A long history of shifting agriculture and associated burning regularly cleared land in all of these dry forest regions. The most extensive deforestation though, has occurred in the latter half of the twentieth century (Collins et al. 1991, Rundel and Boonpragob 1995). Wartime destruction and especially post-war reconstruction has been particularly damaging to forests in Vietnam and to a lesser extent in neighboring Cambodia and Laos. Population growth has been greatest in Vietnam and India. India contains 15 percent of the world's population as well as 15 percent of the world's cattle. Though India has a long history of wildlife protection, the natural resources are under increasing demand. Population growth has necessitated expansion of agriculture and firewood collection across the realm. Primary dry forest has been converted into secondary forest, savanna and agriculture fields. India's new national forest policy, implemented in 1988, develops forests primarily for new fodder and fuelwood reserves (Collins et al. 1991). While Cambodia and Laos still retain greater than 50 percent of original dry forest cover, India retains only 15 percent and Thailand even less (FAO 112 1993). Through the 1980s, Thailand's annual deforestation rate was 3.9 percent – highest for this realm. Commercial logging operations, initially selected teak for harvest, but in recent years, have expanded to include many other mixed deciduous forest species. In 1989, Thailand became the first country in the world to ban all forest logging (Collins et al. 1991), and plantations were to become the primary source of timber. Reforestation in Vietnam primarily involves exotic species for timber production.

With fire protection, native pine plantations (*Pinus kesiya*) have succeeded to pine-oak forests in northern Thailand (Stott et al. 1990). In lowland areas, excessive fire suppression would lead to fuel build-up and high intensity fires. Clearly the management of fires has a significant impact on the forests and individual site differences need to be considered.

Utilization of nurse crops, naturally established or planted, may assist native colonization of abandoned lands by facilitating bird dispersal and creating a beneficial microenvironment. In dry evergreen forests of northern Thailand, both large and small seeded, bird dispersed canopy trees were limited by poor dispersal (Hardwick et al. 1997). Germination trials show that the large seeded *Beilschmiedia* sp. was more susceptible to drought and direct sun exposure than the small seeded *Prunus cerasoides*. A national reforestation program since the 1960s has resulted in the establishment of 1,500 km^2 of native *P. kesiya* plantations in northern Thailand. Current forest management objectives promote a more natural mix of species, however, *P. kesiya* still comprises some 50 percent of the plantings. A chronosequence study showed that colonization by animal dispersed and woody species was significantly greater in 12-year *P. kesiya* plantations than comparable abandoned fields (Oberhauser 1997). The forest structure continued to develop in the oldest 28-year plantations. Forest restoration through plantation facilitation using exotic species has also been developed on a small scale at Ma Da, Vietnam (Collins et al. 1991).

Conclusions

High human population pressures and high levels of deforestation are common themes across most of these ecoregions. Preservation and restoration of the remaining forest remnants likely requires management of land outside of existing protected areas with the involvement of local communities. Education and extension programs would be valuable in the retention of natural forests. Plantation forestry can also serve a dual purpose here. First, plantations could be further established on already degraded lands to provide essential products for the local populace and reduce pressure on existing forests. If diversified with sufficient native species, these plantations may progress in part to rehabilitated forest. Since many dominant canopy species, especially dipterocarps, are poor dispersers, nurse plantations can be utilized to facilitate the reforestation of open sites. Colonization of plantations has been observed a number of widespread, though small scale sites. Finally, the fire regime must be addressed in any conservation-restoration program. Typically, increased fire frequency due to the human population has degraded native forests. In certain instances though, fire suppression leads to fuel build-up and intense fires. Fire regime management should take into consideration the specific conditions and human needs of each area.

2-4. Tropical Dry Forests—Australasia

Ecological Setting

The Australasia tropical dry forests are represented by the New Caledonia ecoregion (see Plate 9). At 16,750 km^2, New Caledonia is the largest single island in eastern Melanesia, with a subequatorial climate comparable to the Hawaiian Islands. Dry forests only occur along the strongly seasonal, wind protected west coast at low elevations. The canopy is often dominated by *Acacia spirorbis* with a variety of associated species (Mueller-Dombois and Fosberg 1998). The native vascular flora of the dry forests totals 379 species of which 59 percent (223) are endemic, and 59 of the endemic species are restricted to the dry forest (Bouchet et al. 1995).

Due to fragmentation, many of the endemic plants are now found at only one or two localities, sometimes in a single population.

The dry forests of New Caledonia can be viewed as a microcosm of the dry forests of Central America. Originally, the dry sclerophyll forest likely occupied the entire west coast lowland, totaling 4500 km^2. Today less than 100 km^2 (2 percent) exists in relatively undisturbed patches scattered along the coast (Bouchet et al. 1995). These remnants are typically less than 5 ha and none is larger than 200 ha.

The range of the dry forest is restricted to that portion of the island characterized by a pronounced dry season. Soil origin also controls the forest type. Extensive ultrabasic soils support unique sclerophyllous scrub vegetation (maquis) in place of the dry forest (Mueller-Dombois and Fosberg 1998). As with other tropical dry forests around the world, anthropogenic fires have become the single greatest factor in determining the existence and extent of remaining dry forest.

Restoration and Management

Forest clearing probably began with Melanesian arrival over 3,500 years ago (Bouchet et al. 1995). In the 1800s, French settlers accelerated land clearing for agriculture and introduced cattle and the Indonesian deer (*Cervus timorensis*). Grazing destroyed understory layers and prevented establishment of new seedlings. Uncontrolled fires have had the most devastating impact. Fires, which are set every season to clear agricultural fields and to enhance savanna grass for deer, frequently escape control burning into forested areas. Succession to native secondary forest is usually the initial result of disturbance in primary forest (Mueller-Dombois and Fosberg 1998). However, frequent disturbance, such as regular grazing and seasonal fires, allows colonization by fast growing, exotic woody species such as *Leucaena leucocephala*, and thus converts forest into savanna. Savanna, which now occupies most of the western lowlands, may have replaced many dry forest communities.

Conclusions

Remaining dry forest is essentially all on private land, while protected areas are nearly all in rainforest and high maquis (Jaffre et al. 1998). Even these remnants (typically less than 5 ha) are too small for ecosystem level preservation. Jaffre et al. (1998) believe that it is impossible to save all threatened dry forest species in situ and suggest ex situ reproduction of the most endangered species for possible later reintroduction. To preserve this ecosystem, several protected areas (as large as feasible) would need to be established, each consisting of core areas of a number of remnant patches surrounded by potentially restorable land. Cleared land has shown the ability to revert to forest if not too severely damaged or converted to savanna. In these latter cases, plantations may help facilitate forest restoration depending upon the particular site. Possibly the most important factor for dry forest preservation is enhanced fire control. As in similar situations elsewhere, local and political support, long-term financial commitment, public education, and economic benefits are critical for long-term success.

2-5. Tropical Dry Forests—Oceania

Ecological Setting

The islands of Hawaii contain the only seasonally dry Global 200 forests in the Oceania realm (see Plate 5). Hawaii is the most isolated island archipelago in the world, being over 3,700 km from the nearest continent, North America, and 3,350 km from the nearest high islands, the Marquesas (Mueller-Dombois and Fosberg 1998). Prevailing northeast tradewinds produce the highest precipitation on windward sides and dryer climates on the islands leeward sides. Throughout Hawaii a variety of dry forest types are present in the distinctly seasonal, but not droughty, zones between these extremes. The Hawaiian islands are completely volcanic in origin and soils of the dry forests are typically well developed. The pioneering legume, koa, is often present as a dominant canopy species, particularly in the highland mesophyte forests. Along with 'ohi'a, koa is an important forest structure forming tree, harboring native fauna and flora.

The extreme isolation and endemism inherent to the Hawaiian Island ecosytems and resulting vulnerability has been well documented. Though widespread amoung the islands, the land area originally forested is relatively small when compared with continental landmasses. The corresponding resilience of these degraded forests is low. In conjunction with the high rate of settlement and habitat destruction, widespread introduction of exotics (flora and fauna) is the overriding factor contributing to native forest destruction and inhibiting restoration. Detailed discussion of the extent of exotics in Hawaii is beyond the scope of this paper.

Restoration and Management

Most dry forests, particularly at lower elevations, have been converted to other uses, including ranching, agriculture and settlement. Even upper elevation forests, logged and converted to pasture, have been significantly deforested. By the early 1980s, the koa-'ohi'a mesic forest on Hawaii, for example, had been reduced to less than 15 percent of its original range (Jacobi and Scott 1985). Introduction of exotic species has proceeded at a greatly accelerated pace since European contact in 1778. Ungulates, particularly feral goats (*Capra hircus*) and pigs (*Sus scrofa*) have been the most destructive introduced animals in the dry forests. In addition to destroying native vegetation by browsing, rooting, and trampling, ungulates facilitate the spread of exotic plants especially suppressive grasses. *Leucaena sp., Lantana sp., Schinus sp.* and *Myrica faya* are the most invasive plant species (Mueller-Dombois and Fosberg 1998). Most recent conservation management activities have consisted of fencing protected areas and removal of feral goats and invasive weeds (Gagné 1988). Native vegetation response to exclosure was generally positive in a survey of 50 sites established in a range of ecosystems (Loope and Scowcroft 1985). However, once sufficiently established (typically in more damaged sites), exotic suppressive grasses thrived following exclosure. Exclosures to eliminate goat browsing resulted in rapid establishment of koa and other native vegetation in montane koa woodland (Stone and Scott 1985). Initial forest restoration efforts at Hakalau Forest National Wildlife Refuge required perimeter fencing to prevent browsing damage.

Additional research and management has focused on regeneration of native trees. Koa is one of the fastest growing native Hawaiian trees, reaching 10 m in 10 years and 20 m in 30 years under average conditions. Displaying remarkable resilience, koa will continue to regenerate by seed and vegetatively unless fire or grazing disturbance is too frequent. These characteristics plus its high value as a local specialty hardwood make koa ideal for forest restoration. One year survival was 97 percent for koa seedlings and 88 percent for 'ohi'a rooted cuttings (Conrad et al. 1988). In this study, koa vigor was rated average while 'ohi'a vigor was rated below average. At locations above 2,000 m, morning shade seedlings protected from sun exposure during freezing conditions, increased koa survival from below 40 percent to above 80 percent (Conrad and Gill 1996). Soil scarification resulted in koa seedling germination from the soil seed bank in bare areas (Conrad et al. 1988). Germination was greatest within 25 m of live trees, but even distant areas produced seedlings, indicating effective seed dispersal. Exotic grass competition increased with time and suppressed further seed germination but did not affect established seedlings. In two-year trials, koa growth was slow compared with exotic acacia species (Cole et al. 1996). However, koa growth increased ten-fold with high fertility treatment. Cole et al. (1996) reports that past koa plantations have been susceptible to disease and insect attack, and have produced mixed results. Characteristics vary between different sources and should be considered prior to planting.

Conclusions

Restoration of Hawaii's dry forests must begin with the committment to upgrading existing protected areas and establishing additional areas in the fashion of Volcanoes National Park, Haleakala National Park and The Nature Conservancy Reserves. Public education on critical factors, such as the impact of exotics and the need for their reduction, is lacking. Forests above 1000 m elevation are generally less disturbed and have the greatest potential for restoration (Jacobi and Scott 1985). Research indicates that with sufficient protection, relatively intact forest vegetation is capable of regeneration and koa and 'ohi'a may be particularly useful in facilitation. However, most any restoration project must first address the issue of exotic species, both fauna and flora. Control measures include fencing, hunting, mechanical removal, chemicals, and biocontrol. Any one approach on its own, regardless of the target species, is usually insufficient. In addition to the ecological factors, societal issues, such as public opinion, must be engaged when establishing control programs.

3-1. Tropical and Subtropical Conifer Forests—Neotropical

Ecological Setting

With Caribbean forests largely obliterated, the vast majority of Neotropical conifer forests are presently located in Mexico and just making it into southwestern United States (see Plate 6). Mexico's conifer forests contain approximately half the worlds species in the genera *Pinus* (pines) and *Quercus* (oaks) (Fegler and Wilson 1995). Plant communities of the pine-oak forests are comparable in species richness to tropical rainforests. These forests are centered on two north-south trending mountain ranges in Mexico, the Sierra Madre Occidental, and the Sierra Madre Oriental (Perry 1991). To the north, smaller "islands" of forest surrounded by desert form the Madrean Archipelago with its extension into the southwestern United States. Pine-oak forests also extend south into Central America. The climate is strongly bi-seasonal with a dominant dry season, and precipitation is correlated with elevation. The high diversity is promoted by a myriad of microclimates produced by severely varied topography.

In addition to the topography, a key characteristic of these ecosystems is the adaptation to a regular, low intensity fire regime (Fulé and Covington 1996). Regular fires produce a more open understory, keeping fire intensity low and favoring pine regeneration. In addition to natural fire originating from lightning, traditional uses of fire included clearing agricultural plots and flushing game. European settlers continued the use of regular fire to clear fields and spur herbaceous growth. Although human-forest relationships extend far into the past, changing attitudes towards fire in Mexico has only recently increased fire suppression. In contrast, fire suppression has been the rule since the turn of the century in the United States.

Restoration and Management

Due to the remoteness and ruggedness of terrain, much of the Mexican conifer forests have remained relatively intact (Fegler and Wilson 1995). Recently however, ranching, recognition of timber values and an exploding rural population have increased human pressure on these once neglected lands. At the same time, the Mexican government has officially, though ineffectively supported forest conservation (FAO 101 1993), thus providing an opportunity for forest restoration of the neotropical conifer forests.

In addition to increased deforestation, exclusion of regular fire regimes in remaining tracts dramatically alters the structure and composition of pine-oak forests. The impact of fire regime on forest ecology was studied at two, paired, unlogged 70-ha forest sites with historic fire return intervals of four years (Fulé and Covington 1996). Fire was excluded from one site for nearly 50 years, while the historic fire regime continued uninterrupted at the second site. Density of small diameter trees increased significantly at the fire excluded site while its overall diversity decreased compared to the frequent fire site. Prolific regeneration also occurred at site with frequent fires, however the open forest was maintained throughout the natural thinning caused by the fires. A third site burned intensely, causing substantial overstory mortality, after fire return was interrupted for 29 years. The fire exclusion allowed an unnatural build-up of forest fuels resulting in the intense fire as opposed to the low intensity fires these forests are adapted to. Many areas are increasingly threatened with this situation, and forest restoration in the neotropical conifer region may require reestablishing the natural fire regime. After nearly a century of intensive fire suppression, the United States Forest Service and Bureau of Land Management (BLM) are experimenting with thinning and prescribed burns to help restore structure, composition and function to these natural systems. The 10,000 acre Mt. Trumbull project in northwest Arizona is an example of management direction proposed by the BLM (Taylor 1996). The dense proliferation of younger trees produced since fire suppression will be thinned prior to reestablishing a more natural fire regime.

Natural regeneration can be relatively rapid on lands traditionally cleared and burned for short-term agriculture use (González-Espinosa et al. 1991). A chronosequence study in the Chiapas highlands of southern Mexico revealed a gradual increase in the compositional and structural diversity of naturally regenerated forests. A dense, 10 to 12 m, tree canopy developed after 20 to 25 years, and a significant understory was present by 40 to 45 years. A 35 to 45 m canopy existed after 80 to 100 years, and the understory layers continued to develop and add new species. Pine regeneration was favored in open sites, and observations of primary forest at the nearby Cerro Huitepec Reserve indicate that occasional, but severe, windstorms may play a role in maintaining an open canopy.

Conclusions

Large tracts of highly diverse, conifer forest still exist in northern Mexico. The protection of representative forest types should continue to be pursued. Local communities appear willing to become involved in forest preservation and restoration. Restoration and management of these ecosystems in Mexico and the United States will require management that includes a return of the natural fire regime. Small scale experimental application of fire has been pursued for the last decade in the United States with success. Though changes in the U. S. Forest Service management policy from extraction to ecosystem health should benefit these forests, long term protection is not guaranteed.

Reforestation efforts on Haiti and Cuba would likely be reclamation and rehabilitation oriented and tied to agroforestry (Pellek 1990). The local populace would probably resist projects that remove land from agriculture service. The Foret de Pins should be further investigated, as it may be the largest forest remnant worth pursuing for preservation.

4-1. Temperate Conifer and Broadleaf Forests—Eastern Nearctic

Ecological Setting

Southeastern conifer and broadleaf forests, Appalachian and mixed mesophytic forests comprise most of the forests in the eastern United States south of New England (see Plate 10). The mixed mesophytic forest is the most biologically rich ecosystem in the U. S. and one of the richest in the world (Hinkle et al. 1993). A multitude of species, predominantly deciduous, including oaks, maples, beech, chestnut, basswood, poplar and hemlocks, function as canopy dominants due to the variety of microsite conditions. In addition to high species richness, high evenness is also characteristic of these forests. The upland longleaf pine ecosystem of the southeastern United States, dominated by the single tree species *Pinus palustris,* historically stretched from Virginia to eastern Texas in a broad belt along the Atlantic and Gulf coasts covering 36 million hectares and is thought to have predominated on 24 million hectares (Johnson and Gjerstad 1998). Along floodplains in the southern states, especially the Mississippi River and lower tributaries, bottomland hardwoods, often with a strong oak component, dominated. Eastern deciduous forests are adapted to long, warm humid growing seasons with no water deficiency and cold winters, which moderate to the south.

The pine and hardwood forests of the southeastern United States evolved with regular fires. The fires that originated from lightning and Native Americans, frequently burned large expanses. Several pine species require fire for seed release from serotinous cones. Aspens and oaks are highly successful at regenerating vegetatively following fire. The fire interval of eastern deciduous forests was probably 50 to 100 years prior to European arrival (Yahner 1995). An increased use of fire was used by settlers to clear land for agriculture. Fire suppression on upland sites throughout this century has lengthened the fire return interval and has disrupted some ecological processes. The major human impact on the bottomland hardwood forests has been the alteration of the hydrologic regime to control flooding.

Restoration and Management

Most eastern states have greater than 50 percent forest cover today, though most is secondary forest. Most virgin hardwood forests were logged prior to 1920, often with land conversion to agriculture. A major trend in the Appalachian-mesophytic forest region during the twentieth century has been the retirement of private farmland. Forest cover is typically greater now than at the turn of the century, increasing in upland regions to as high as 79 percent is some areas (Hinkle et al. 1993). Public forests, which were managed exclusively for timber production using clear-cutting from the 1950s through the 1970s, are now moving towards ecosystem management. On private lands, which make up most of the region, widespread conversion to monoculture pine plantations continues to intensify and is a serious problem to protecting native biodiversity. This is particularly true for the conifer forests of the southeastern United States where so few native forests remain.

Since European settlement over 250 years ago, this ecoregion has been inexorably altered with the upland longleaf pine ecosystem most significantly by clearing for agriculture, commercial replanting with native and non-native timber species, and the elimination of fire from the ecosystem. Today, longleaf pines are present on less than 4 percent of the original range, which is considerably less than what represents functioning ecosystems. The remaining fragments are extremely small, the largest being 80 ha (Johnson and Gjerstad 1998).

Unlike most of the agricultural lands of eastern United States, the bottomlands of the southern Mississippi valley have been cleared only since early in the twentieth century. Disastrous floods along the Mississippi River during the 1920s prompted massive government programs beginning in the 1930s to construct levees and other water control works. The frequency and intensity of flooding was sufficiently reduced to make many forested bottomlands economically attractive for conversion to agriculture resulting in extensive clearing and long-term land degradation. Lower crop prices in the 1970s caused widespread agricultural abandonment leaving many of these farms available for land restoration. Restoration of bottomland hardwoods in these abandoned areas would likely result in the creation of a higher portion of forests representing the dryer end of bottomland hardwood community spectrum (Newling 1990). Although slightly out of the Global 200 ecoregions, reforested, naturally regenerated and intact bottomland hardwood sites studied in southwestern Kentucky appeared to have succeeded over a period of 50 years a hydric to a more mesic community composition (Shear et al. 1996). This is due to the change in the hydrology from dams and diversions along the river system resulting in a drier state (less frequent flooding and possibly lower water table). This is important since most restoration efforts along the Mississippi have targeted seedlings of tree species adapted to the more hydric conditions of the original river dynamics and characteristics.

The bottomland hardwood restoration performed in the lower Mississippi valley since the 1980s has concentrated on the establishment of 1 to 3 overstory species in plantation style rows, especially Nuttal oak, willow oak, water oak and cherrybark oak (Newling 1990, Allen 1997). Oaks have traditionally been an important component of the forest and have high wildlife and timber value. The rationale for prioritizing oak establishment is that oaks demonstrate limited dispersal capability due to large seeds and shade intolerance. The other species generally disperse more rapidly forming a natural, more mixed forest. Invasion of heavy seeded species from nearby mature forest (oaks, hickory, and honeylocust), in both naturally regenerated sites and restoration sites where plantings did not include heavy seeded species has been very limited (Shear et al. 1996). Being shade intolerant, these species will likely be minimally present or excluded from the mature overstory. As of 1995, 75,000 acres of Mississippi Valley bottomland has been replanted, with another 75,000 acres expected in the next decade.

The initial establishment of oaks in the Mississippi valley bottomlands has been very successful. Site preparation has not been critical here, but has been used periodically to reduce weed competition. Establishment can be by tree seedling or sowing of seed. Seedlings establish more successfully and provide a head start on growth but are more expensive (Allen 1997). Acorn procurement should be planned since few seed trees may be locally present and oaks in this region seed only about once every five years (heavy mast years). Acorns of the red oak group can be cold stored for up to two years with 90% viability, while acorns of the white oak group can only be stored for two months. Sowing is possible at any time of the year except the driest summer conditions. Long term success is usually excellent with a minimum of 35% survival (Newling 1990). Further restoration recommendations are provided by Allen and Kennedy (1989).

Proximity to intact forest patches was important for woody species invasion studied in 10 young, 4 to 8 year old, restored sites (five seeded and five planted) (Allen 1997). Woody species invasion was significantly greater in narrow, linear clearings and within 60 m of forested edges compared with interiors of large sites or isolated locations. Observations at other sites (e.g., Ouachita Wildlife Management area near Monroe, LA) also indicate limited woody colonization. However, understory development (excluding heavy seeded species) of both planted and naturally regenerated sites after 50 years was approaching that of relatively intact mature forests in Kentucky (Shear et al. 1996). The small size of these sites (1.4 to 3.0 ha) and proximity to seed sources likely facilitated this colonization. Additionally, the well-documented heterogeneity in this region suggests caution in the interpretation of these results.

In addition to justifying oak establishment due to their slow colonizing ability, Allen (1997) provides the following suggestions to create more natural diversity at restored sites. Row planting should be discontinued to establish more natural spacing, and planting a lower density or in groups with gaps may increase woody invasion. Variable germination associated with sowing may also lead to increased colonization, but also greater competition. Based on native inventory, plant only 30 to 50 percent oaks with a variety of heavy seeded species. Consider planting wind dispersed species, including specific shrubs, where they form an important component in the understory. Finally, the origin and genetic variability of the stock should be considered.

Beginning in 1985, The Nature Conservancy (TNC) began a restoration program of 1,200 ha of disturbed longleaf pine uplands at Apalachicola Bluffs and Ravines Preserve (ABRP) in the Florida panhandle about 80 km west of Tallahassee (Seamon 1998). The longleaf pines had been previously logged and planted over with slash pine, which had also been mostly cut over. Only 77 ha of the original longleaf pine-wiregrass community remained when TNC obtained its first parcel in 1982.

Nearly one million seedlings have been planted on 920 ha in a little over 10 years with a containerized seedling survival rate of 70 to 85% much higher than bare rooted survival. The overall management goal to enhance community diversity mandated a frequent low intensity prescribed fire program, which is necessary to maintain the longleaf pine-wiregrass system. Experiments on nearby preserves showed that burning wiregrass during the growing season induced wiregrass flowering and seed production later in the year. Wiregrass grown from collected seed at an on-site nursery proved costly. Mechanical seed collection was initiated using a Woodward Flail-Vac Seed Stripper, while an efficient, inexpensive direct seeding method is sought through trials (Seamon 1998).

Conclusions

Though 90 percent of existing forest is on private lands, most large, remaining tracts of intact forest in the eastern United States are located on lands controlled by federal and state agencies. Since sufficient forest land certainly remains, the question of forest conservation and restoration becomes one of land use and management. Until recently, public forests were managed predominantly for the sustainable production of timber, and conservation focused only on the protection of specific endangered species. Public forest management practices are changing to emphasize multiple use and ecosystem health, but what that means in terms of forest biodiversity conservation remains unknown. Forecasts for future land use indicate increased urban development and conversion to industrial forests on private lands (Boyce and Martin 1993).

The eastern mixed mesophytic and southern broadleaf forests have truly proven resilient. Though little original old growth remains, these forests have naturally regenerated following disturbance, often several times over more than 100 years. The best manner in which to restore these forests may to just leave them to grow without further disturbance. The disruption of certain ecosystem processes, hydrology in bottomland hardwoods and natural fire in southern forests for example, prevents the simple restoration of these ecosystems through natural regeneration. Restoration of the disrupted processes should be pursued wherever feasible. In the event that reversal is not possible, as is most often the case, restored forests will not likely be comparable to the original forests. Due to poor dispersal or other limiting factors, certain forest types as previously illustrated, require more intensive efforts to achieve restoration. Although restoration of bottomland hardwoods has already been pursued on a relatively large scale, many more years are necessary to see the full results of this work. Restoration research and field work need to be continued, especially as they apply to rarer and less resilient forest types. Remnants of these forests will be under increasing pressure and subject to further fragmentation in the future.

Forest restoration at this time needs to be pursued through a greater commitment to management for long-term ecosystem preservation in the public forests. Following the lead of The Nature Conservancy and other private organizations, endangered and important habitats need to be secured in protected areas. Taken to the next level, establishment of landscape scale reserves like the Man and Biosphere Program (MAB) model should be considered. The success of such a program will depend upon long-term cooperation, especially in light of the private land rights debate that exist in the United States.

4-2. Temperate Conifer and Broadleaf Forests—Western Nearctic

Ecological Setting

The largest temperate rainforest in the world is located along the west coast of North America (see Plate 10). Warm temperate rainforest of northern California transitions into cool seasonal rainforest in Oregon then perhumid rainforest in Vancouver Island, British Columbia and finally subpolar rainforest in southeastern Alaska. The temperate rainforests are closely associated with climatic conditions under a strong marine influence, which produce moderated temperatures and high levels of precipitation with accompanying clouds and fog. The Sierra Nevada ecoregion extends south into a warmer less marine influenced climate. The Klamath-Siskiyou ecoregion lies between the two in a slight rain shadow with unique serpentine soils as its defining feature. An incredible array of mixed conifer forests occupies different zones within these regions. The Klamath-Siskiyou forests are characterized by high endemism and at the local scale high diversity of conifer species.

Marine climatic influence and elevation are the two factors most important in the distribution of North America's temperate rainforests. Regional differences in natural fire regime reflect climate differences in part and play an additional role in local ecology, especially in the southern portion, the Klamath-Siskiyou and Sierra Nevada ecoregions. The specific regimes however, vary dramatically between and within ecoregions, ranging from very frequent low-intensity fires to long interval (several hundred years return) high intensity crown fires. Short return intervals for example, in western hemlock (*Tsuga heterophylla*) zone common to the lowlands, promotes canopy dominance by Douglas fir (*Pseudotsuga menziesii*) (Agee 1993). The more northern extent of the rainforest is not susceptible to natural fires. Windthrow from winter storms is another frequent disturbance event and impacts all forests in this realm. A major ecological constraint following disturbance for the canopy conifer species is propagule availability and dispersal ability (Weetman and Vyse 1990). Conifers do not regenerate via vegetative sprouting and annual seed production can be variable between species. Douglas fir for example produces seed at intervals ranging from 2 to 11 years.

Restoration and Management

Since the late 1800s, logging, agriculture, and urban development have significantly altered the forests south of Vancouver Island and southern British Columbia. Though much of the area retains tree cover, these lands (both private and public) have largely been converted to secondary production forests, dominated by Douglas fir and only resemble the original ecosystems. Long-term fire suppression has also allowed the build up of fuels resulting in less frequent but much higher intensity fires. The majority of reforestation efforts to date have revolved around Canada and U.S. Forest Service and timber industry plantation forestry programs. Through the 1950s, restocking of cut over land relied upon natural regeneration before switching to seedling planting. Since the conifer species do not sprout vegetatively nor produce a long-lived soil seed bank, natural regeneration requires colonization and establishment by seed (Weetman and Vyse 1990). Conifer seeds are wind dispersed and rarely travel more than 100 to 200 meters, with western hemlock generally the widest disperser. Additionally, annual seed production varies among different species. Together these factors affect species ability to colonize openings, especially larger man-made clearings. The microsite conditions of clearings, particularly the availability of moisture and shade, can dramatically influence the establishment success of colonizing seedlings. Smaller, more natural gaps with part shade for example, favor western hemlock over both sitka spruce and Douglas fir.

Chronosequence studies from southeast Alaska (Alabäck 1982) and western Washington (Henderson 1982) showed rapid recovery of shrubby species within 15 to 25 years after disturbance. Canopy closure of young trees followed shortly thereafter with a coinciding reduction in ground cover. Essentially, the same sequence was observed in logged, burned and Douglas fir-planted sites in the western Cascades of Oregon (Schoonmaker and McKee 1988). Naturally regenerated forests 150 to 200 years in age are still developing structurally and compositionally in these forests (Alabäck 1982; Henderson 1982) and will never regain old growth characteristics under current timber management, which has return cut intervals before the forest can attain full maturity. Forests here require a minimum of 700 years to effectively eliminate seral competition.

The forest restoration work attempted thus far has generally been small in scale and intensive in terms of input. Due to stream quality issues in the region, forest restoration is often integrated with stream, riparian zone or watershed restoration, such as the Mottole River, California (House 1992), Redwood National Park (Belous 1984), and Coos Bay, Oregon (Rumrill and Cornu 1995). For the Redwoods National Park and Little Applegate, Oregon (Whitall 1995), primary concern has been controlling erosion on steep slopes caused by logging roads. The solution has been to remove the roads, recontour the slopes, install check dams for erosion control, and stabilize the soils through tree planting and forest establishment. At Redwood National Park, the single most useful species in this has been the native been red alder (*Alnus rubra*), which is hardy and encourages succession through nitrogen fixation (Belous 1984). Costs for average road rehabilitation were between $8,000 and $25,000 per mile, reaching $40,000 per mile for worst case roads. A comprehensive land preservation and management project is in the planning and early implementation phase for the 500,000 acre Applegate River watershed (Yaffee et al. 1996). The project is unique in that 35 percent of the land is owned by the U.S.D.A. Forest Service, 35 percent owned by the U.S. Bureau of Land Management, and the remainder owned by private interests. Though not defined as such, the Applegate River watershed project bears similarity to the MAB concept. To date, planning a framework for reestablishing the health of the forest, and educating and maintaining dialogue with the public concerning respective goals have been the primary activities.

Integration of fire into forest management has progressed slowly since the 1970s (Teensma 1996). With variability in spatial and temporal scales of a natural fire regime, the question remains how to best implement prescribed fire. The consequences of uncontrolled fire add a social dimension to fire management. Within the context of fire use itself, is the controversial choice between restoring the natural process to the ecosystem and restoring specific forest structure and composition to the forest (Agee 1993). Many National Parks and wilderness areas have approved plans that allow natural fires to burn when they have met prescription criteria (Mutch and Cook 1996), but most of the forest in this region falls outside these land designations.

Conclusions

Large tracts of relatively undisturbed forest are still located in Alaska and northern Canada, and representative systems on public lands should be designated for long-term conservation. Most forests south of Vancouver Island and southern British Columbia have been logged, replanted in Douglas Fir, and bear little resemblance to the original forests. Sufficient forests exist to pursue multiple use and ecosystem preservation management on public lands. Many of these forests are fragmented from checkerboarding with private landowners – an unusually ownership pattern left over from the railroad building days where the landscape was cut up into regular blocks and assigned alternate public-private ownership pattern. Efforts are underway by federal agencies to consolidate fragmented forest but so far, often at the expense of trading higher quality forest, it has been very difficult. The Applegate River watershed project is a landmark attempt at comprehensive watershed conservation in this region. In general, this region lacks projects of this magnitude. However, results from the Applegate River may not be seen for decades. Historically, fire has been a natural, but variable part of ecosystems south of British Columbia. True restoration of these forests should include fire in some capacity, but this will add ecological and social complexity to the effort. The U. S. Forest Service is positioning itself to make greater use of prescribed burns in future forest management (Mutch and Cook 1996), but much needs to be learned and applied to forest conservation.

4-3. Temperate Conifer and Broadleaf Forests— Neotropical

Ecological Setting

The Valdivian temperate rainforest of South America is primarily restricted to the western slopes of the Andes in Chile and small projections into Argentina east of the Andes where physiography permits the passage of humid air (Veblen et al. 1996; Wilcox 1996) (see Plate 6). Sandwiched between the largely deforested, central sclerophyll forests to the north and the North Patagonia rainforests to the south the Valdivian ecoregion includes seven major forest types. Evergreen conifers and broadleaved trees plus deciduous southern beeches (genus Nothofagus) characterize these forests. Approximately 95% of the region's tree species are endemic. The understory is often dominated by bamboo. The western slopes of the southern Andes generally receive greater than 3,000 mm of precipitation, relatively, evenly distributed throughout the year. The Valdivian is the center for arboreal diversity and biomass in the neotropical temperate rainforests, since temperature and growing season decrease to the south.

The Andes are the major determinant of climate and forest composition patterns in the southern Andes. Prevailing westerlies produce high precipitation on the west slopes, which are effectively isolated from the remainder of the continent by the range itself. Of the major Valdivian tree species, the ecology is probably best understood for the long-lived southern beeches. Seed production and germination success is highly variable from year to year, and dispersal is predominantly through gravity and wind (Veblen et al. 1996). Though differential seed size produces a range of dispersion patterns, even small seeded species, such as *N. dombeyi*, drop 95 percent of seed within 20 m of the source. Insect predation of seed can have significant impact on reproduction. Seedling establishment of all beech species is best in high light conditions on bare mineral soil, where litter and competing bamboo are absent. However an advance seedling bank is rarely present due to normal low light conditions and understory competition. In contrast with evergreen species, all deciduous beeches reproduce vegetatively in varying degrees.

Restoration and Management

All Chilean forests originally covered 30 million hectares, now cover 15 million hectares and are being deforested at a rate of 120,000 to 200,000 ha per year (Wilcox 1996). Deforestation that began at the turn of the century has dramatically increased since the mid-1980s, and there has been substantial conversion of the native forests in the northern region to exotic monoculture plantations of Eucalyptus and *Pinus caribaea* (Wilcox 1996; Veblen et al. 1996). Valdivian forests are poorly represented in the Chilean government protected areas, the vast majority of which are established in the less accessible and lower diversity southern Patagonian rainforests (see Armesto et al. 1998). Of notable interest are the private efforts of Doug Tompkins through Foundation for Deep Ecology and Chile's Foundation for Education, Science and Ecology, in acquiring 270,000 ha of Valdivian forest for the establishment of Parque Pumalin, to be a world class reserve and park. The land being acquired contains 35 percent of the remaining Alerse forest type that is an endangered community type. This community type is dominated by the alerse tree that can live in excess of 3,000 years reaching 4 m in diameter. Parque Pumalin has been designated as a national monument in Chile.

The long-lived nature of the dominant species in the Valdivian forests combined with the high biomass content have led to comparisons with other fairly stable ecosystems (Wilcox 1996). The importance of natural disturbance is examined by Veblen et al. (1980) in allowing the relatively shade intolerant Nothofagus to remain dominant over shade tolerant old growth forest species. The forest therefore continues in a dynamic state of flux. The mid-elevation forest region is characterized as tectonically active causing frequent land clearing events via landslides (from earthquakes) and volcanic eruptions including ash falls. Wind and fire may also be significant disturbance events. Nothofagus may be an early tree colonizer of disturbed sites, signifying its potential importance in future restoration efforts. Though individual seedlings have been found up to 200 m from forest edges, photo comparisons of Nothofagus forests, show that 95 percent of naturally regenerated expansion was restricted to within 20 to 38 m of the edge (Veblen et al. 1996). Valdivian Forest restoration efforts are largely lacking.

During the 1970s and 1980s, control of the Chilean forest industry became concentrated in the hands of a few large corporations. Today, four Chilean companies account for >70 percent of forest exports and two own 46 percent of the existing radiata pine plantations. The smaller landowners, who own the majority of the unprotected native forests, harvest the forest for woodchips but find the markets largely controlled by the major timber companies. With the integration of Chili into the new global economy, international pressures have increased to log as much of the remaining native forests as possible primarily in the form of woodchips. For example, wood chip production rose from 76,000 cubic meters in 1986 (all from pine plantations) to 2.5 million cubic meters in 1995 (62% from native forests). The threat to the remaining native Valdivian forests is dramatic.

Conclusions

Due to low population pressure in southern Chile, the Valdivian forests as a whole currently remain relatively intact and undisturbed. However significant incursions, primarily for timber, threaten the ecological integrity of this ecoregion. In all likellihood, there is a short window of opportunity to save the native biodiversity of the Valdivian forests, and little of the Valdivian is represented in Chile's protected areas system. With a recent history of severe forest exploitation, upgrading the protected areas system by adding significant reserves located in the Valdivians is the top priority. The dominant canopy species may also function as early colonizers, facilitating restoration of cleared lands. Of more importance though is emphasizing sustainable management practices on lands already degraded and less exploitation of prime forest.

4-4. Temperate Conifer and Broadleaf Forests—Palearctic

Only one ecoregion in this realm has been highlighted with Global 200 status – Southern Europe Montane Forests. This ecoregion includes forests of the Pyrenees, Carpathians, and Southern Alps. Numerous papers pertaining to forest restoration were located for this region, but all were written in languages other than English. We were therefore unable to acquire enough information for this region to develop a full discussion.

4-5. Temperate Conifer and Broadleaf Forests—Australasia

Ecological Setting

The temperate forests of concern in the Australasian region extend across the south island of New Zealand, Tasmania, and the perimeter of Victoria and New South Wales in southeastern Australia (Plate 9). These forests can be divided into two main types, the eucalypt forests and the temperate rainforests. Dry and wet sclerophyll forests dominated by eucalypts are the predominant forests of southeastern Australia and for a good portion of Tasmania. The eucalypt forests grade into isolated patches of rainforest, dominated by various canopy species in southeastern Australia. Of Australia's one million hectares of temperate rainforest, 765,000 ha are located on Tasmania (Busby and Brown 1994). The Tasmanian forests are similar, but generally dominated by beech (*Nothofagus cunninghamii*). The New Zealand forests do not contain a eucalypt component, but like Tasmania rainforests, are dominated by beech (several species), and beech-podocarp canopies.

Fire forms a natural part of most Australian landscapes and its characteristics (frequency and intensity) in large part shape the eucalypt and rainforest ecosystems. As a result of variability in the fire regime, most native forests have a patchy nature. Closed canopy rainforest is fairly resistant to fire and even if penetrated can regenerate without further disturbance in 50 to 60 years (Busby and Brown 1994). Land cleared in association with more frequent fires promotes the fast growing, shade intolerant eucalypts at the expense of rainforest species. Eucalypt seed is stored in the canopy for three years or more, and colonization requires this source since seed is not stored in the soil seed bank and surface seed is destroyed by fire (Ashton and Attiwill 1994). In addition to the nutrient flush provided by fire, fast growing, nitrogen fixing acacias which often grow in association with eucalypts help improve poor soils, which are widespread in Australia. Once established, short fire intervals and the fire adaptations of eucalypts, such as flammable litter, can hinder rainforest reemergence. Vegetative sprouting is very important for rainforest regeneration and repeat fires can limit this capacity. In the absence of fire rainforest can reinvade and replace eucalypts within 200 years.

Fire disturbance is rare in the wet temperate rainforests of New Zealand. Unlike the beeches of Tasmania, New Zealand's beeches do not have the capacity for vegetative regeneration and as such are susceptible to human set fires (Ogden et al. 1996). Seed production between years can be extremely variable, and dispersal that relies on wind is typically less than 100 m. Seeds usually remain dormant until the following spring, at which time germination rates can be quite high, carpeting the forest with seedlings. Windstorms and landslides are the natural disturbances most common in New Zealand forests opening the canopy and releasing the seedlings.

Southern temperate forests regularly endure freezing conditions. The critical factor delaminating Australia's temperate rainforests and schlerophyll forests is the minimum rainfall during the driest month. At less than 50 mm, schlerophyll forests tend to dominate (Adam 1992; Ashton and Attiwill 1994).

Restoration and Management

Restricted to steep remote terrain, very little of the Australian temperate rainforest, and only 15 percent of the Tasmanian rainforest has so far been cleared (Busby and Brown 1994). On the other hand, 75 percent of New Zealand's rainforest has been lost, though significant forest remains on the western slopes of the Southern Alps. Forest clearing and burning began with the colonization by the Maori approximatly 1000 years ago. By the time of European arrival the forests which originally covered 80 percent of the land were reduced to half their original extent. Accelerated clearing and burning since has further reduced the forest cover by half again (Lucas and Bassett 1995). The vegetation that has replaced burned forest has generally been more flammable than the original vegetation, leading to further burning. Introduction of grazing species, especially red dear, has had a dramatic impact on the understory composition of New Zealands forests. Since the early 1970s a revolution in forest policy has swept New Zealand culminating in the creation of the Department of Conservation in 1987 (Lucas and Bassett 1995). The Department of Conservation now manages 76 percent of the remaining indigenous forests as a conservation estate. Widespread *Pinus radiata* plantations currently account for 98 percent of New Zealand's timber production.

Most, if not all, Australian mainland, and 96 percent of Tasmanian temperate rainforest are owned by the government. Additionally, most large mainland patches are in reserves while 60 percent of Tasmanian rainforests are included in dedicated or recommended reserves. As with the tropical rainforests of northern Australia, "pure" rainforest is reserved from logging, however, many "mixed" and eucalypt forests are open to logging. Recent emphasis on wood chip products has led to greater exploitation of eucalypt forests that historically were bypassed for being poor saw-log producers.

Losses of natural vegetation in the temperate region of Australia can be largely attributed to agriculture (Saunders and Hobbs 1995, Yates and Hobbs 1997). Agriculture also provides obstacles to revegetation by changing the landscape within which revegetation must occur. Revegetation will require consideration of ecological scale and landscape context and will ultimately require integration with the agricultural system (Saunders and Hobbs 1995).

Strategies for forest restoration following human disturbance must not only consider the disturbance but also sources for colonizing species and dispersal mechanisms. As mentioned earlier, this can be especially critical for eucalypts. In dryer eucalypt woodlands, highly fragmented by agriculture and grazing, removal of disturbance by fencing was not sufficient in promoting eucalypt colonization (Yates and Hobbs 1997). Eucalypt recolonization was rare 50 years after farmland abandonment and generally restricted to the immediate vicinity of mature woods. Direct seeding has been successful in areas receiving more than 400 mm of rain annually. Outside of mine rehabilitation, very little effort has been invested in native forest restoration in New Zealand (Ogden et al. 1996).

Conclusions

The vast majority of temperate rainforest remaining in the Australasia realm has been incorporated into protected areas by the governments of Australia and New Zealand. Even still, the fragmented nature of mainland Australia rainforests warrants close attention. Fire protection is critical for the continued conservation of these forests. Further benefit would be generated by allowing the rainforest to expand where possible through natural regeneration along its edges. Though currently protected, New Zealand's rainforests would also benefit from a management policy that promoted rainforest expansion into previously degraded sites. To date, forest restoration has not been a top priority in New Zealand. Australia's eucalypt forests do not have the level of protection of the rainforests. Exploitation of these forests is expected to increase in the future as other resources remain off limits. Restoration of eucalypt forests can be hampered by limited seed sources.

5-1. Boreal Forests and Taiga—Nearctic
5-2. Boreal Forests and Taiga—Palearctic

Although little to no restoration work is being done in the palearctic boreal and taiga region within the Global 200, a few general observations can be made. In Nepal, the Private Forests Nationalization Act of 1957, ostensibly to manage and conserve Nepal's forests, has been credited with causing the heavy exploitation characteristic of the region. The reasons cited for this assertion include: governmental control of forests, exclusion of local people and perspectives in decision-making, lack of incentives for local community development, using revenues for objectives other than community development, expensive infrastructure maintenance, and overall distrust (Basnet 1992). Although this view is not universal (e.g., Karki 1991), it does point to certain recommendations for forest conservation activities. First, as in other regions, local participation and incorporation of indigenous knowledge, techniques, and values is essential (Basnet 1992). Second, informed local development and carefully maintained communication will enhance local participation (Basnet 1992).

In Siberia, on the other hand, forests are slowly (though significantly) being deteriorated (Shvidenko and Nilsson 1994). Rights and responsibilities associated with forests are unclear, the currency is unstable, and local overexploitation is weakening state management of forests (Shvidenko and Nilsson 1994). The major harvesting pattern involves overcutting near roads, railroads, and manufacturing centers (Shvidenko and Nilsson 1994). This situation calls less for restoration and more for protection measures, although eventually, restoration of already deforested lands will need to be desirable.

Role of Restoration in Forest Conservation

The previous regional summaries demonstrate a wide range of forest ecosystem types that differ in their ecological characteristics, ecological constraints, human histories, and current socio-political conditions. Ideally, forests have the capacity to regenerate themselves after disturbance without the help of humans. This depends partly on the degree of disturbance as well as on the specific ecological conditions under which each forest type exists. Forests have evolved under a variety of natural disturbance regimes, but often do not have the capacity to naturally recover after human deforestation. Given enough time, these areas may return to forests, but in many cases, human intervention is required to promote the return of forests on the landscape.

Ecological patterns and processes exist at various spatial and temporal scales and these must be kept in mind when considering forest restoration. Restoration focussed on one spatio-temporal scale may suffer unforeseen difficulties originating from other scales. For example, concentrating on reintroduced species and successional processes without considering patterns of disturbance may result in failed restoration attempts due to ill-advised spatial arrangement. This is not to say that every conceivable scale needs to be accounted for. Rather, it has been suggested that intermediate scales are the most important determinants of biodiversity, ecological functions, and the interactions between them (Risser 1995). Both Noss (1990) and Allen and

Hoekstra (1992) argue for consideration of three scales at once: the scale in question, the one immediately larger, and the one immediately smaller. Risser (1999) further states that most important are "the scales of species habitat and disturbances such as drought, grazing, fire, insect outbreaks, plant disease, and water flows," and that "they dominate the patterns of spatial scales of hundreds of meters to hundreds of kilometers and time scales of years to decades." No one scale is best, although intermediate scales tend to be most informative with respect to integrity (Noss et al. in review). Successfully restoring integrity, therefore, must involve consideration of multiple spatio-temporal scales.

As we have seen through these ecoregional reviews, true forest restoration is often quite difficult scientifically, requiring many decades before the ecological outcomes can be adequately evaluated. In most regions of the world, reforestation activities are not true restoration as defined in this paper. Rather, the actions most commonly taken (e.g., plantation forestry, reclamation, and rehabilitation) often fall short (significantly short in many cases) in restoring the ecological and biological values undisturbed native forests provide and maintain.

Reforestation can play several conservation roles, including: (a) reestablishment of forests and forest habitat on degraded lands, (b) augmentation of existing protected forest areas, (c) mitigation of the ecological impacts of degradation, and (d) production for local and economic benefits (Hobbs and Norton 1996). However, because of the shortcomings of many reforestation activities, true forest restoration should be promoted wherever possible and promoted by the greater conservation community. Several authors offer a set of guidelines for successful restoration in general. These guidelines tend to include consideration of soil properties, the "target" community, invasive and exotic species, natural vegetation dynamics, hydrology, geology, biology, the causes of degradation, goals and objectives, and long-term monitoring (Howell 1986; Horowitz 1990; Harker et al. 1993; Brown and Lugo 1994; Hobbs and Norton 1996). All of these considerations are very important; however, on a case by case basis, different sets of opportunities and obstacles tend to step forward or bow out of the conservation limelight.

No matter what the forest type, forest restoration requires: suitable physical and biological soil conditions, plant propagules, dispersal and colonization capacity by a wide range of species, suitable plant establishment and growing conditions, and the capacity for the full spectrum of ecosystem processes.

All land plants require a substrate for attachment and most utilize some form of soil. Additionally, soils provide the nutrients, trace elements, water and mycorrhizae fungi plants require. Damage to soils as a result of deforestation ranges from minor to complete. In addition to soil loss, soil compaction is also a common problem, which reduces the rooting ability of plants and reduces the overall water holding capacity of the soil. Through mining and industrial operations, soils also can be contaminated with high levels of toxic metals and other harsh chemicals. Lands under the influence of modern agriculture often exhibit unusually high levels of plant nutrients (nitrogen and phosphorous primarily) that result in different, but equally challenging, problems. Stabilizing and amending soils is the first challenge to forest restoration. Only after the soil requirements for forest organisms have been addressed successfully can the reestablishment of the forest community proceed.

Propagule sources for the forest species must exist in the region of restoration activities or can be brought back into the region. Local sources may reside in the soil seed bank or can be resprouted from roots or stumps capable or reproducing vegetatively. Specific seed trees may also be present on-site. Many species will have to recolonize by dispersal from off-site and this requires the presence of healthy forest remnants. In addition, the propagules must be able to disperse to and colonize restoration sites. Unfortunately, the dispersal capabilities of many of the understory components of forests are still unknown. Forest remnants can provide source propagules, but they must be within reasonable proximity of the deforested land. The two most common dispersal agents are wind and animal. Wind dispersal is effective only with smaller seeds. In temperate forests, wind dispersal is more common, while in tropical settings, animal dispersal (especially by birds) dominates. In areas where bird dispersal is critical, early establishment of individual trees or small groves at restoration sites can provide perching and roosting sites in otherwise open land. In general, most dominant canopy tree species worldwide are poor long distance dispersers making these trees prime candidates for planting programs – otherwise colonization may be slow and restricted to forest edges where remnants exist.

Once propagules are present at a deforested site, they must be able to establish, grow and eventually reproduce. Establishment begins with germination and requires suitable conditions including light, temperature, moisture, nutrients and space. However mature forest species typically establish under some degree of canopy cover, which provides very different microclimate conditions than open land. Therefore, cleared land is typically colonized by early successional species that are adapted to open conditions. Succession to mature forest proceeds over decades and even centuries provided propagule sources are available and able to establish themselves in the proper sequence. Succession can sometimes be shortcut by dispersing or planting late successional species beneath the developing early successional canopy. Competition can be severe at these early stages and successful establishment may require weeding or other measures. Additionally, the use of nurse trees in the form of plantations to alter the microclimate and facilitate the establishment of desired native species has been investigated with considerable success (Parrotta et al. 1997). Plantations are also useful in the elimination of suppressive ground covers (e.g., imperata grass in Indonesia) that can dominate a site following disturbance preventing natural regeneration.

If tree seedling planting is necessary or chosen for specific sites, native species should be used whenever possible. Exotic species may be beneficial in the reclamation of severely degraded land in cases where native plants cannot be sufficiently established, but the use of exotic species in forest restoration should always be viewed as a short term measure. If exotics are used, they should be tree species that have few weedy invader properties, and once no longer needed, should be completely eliminated. Even with the planting of natives, mixtures of species should be incorporated into planting schemes in natural proportions whenever possible. Finally, regions invaded by exotics, Oceania in particular, present significant additional problems in terms of restoration. Introduced species, both floral and faunal, may out compete, predate, or suppress establishment of native species. Severe measures to control exotics will prove to be of paramount importance and may pose the most severe challenge to forest restoration in many parts of the world.

Restoration is not truly complete if ecosystem processes are not fully restored. Hydrologic and nutrient cycles, disturbance regimes, and plant-animal interaction are the processes most often altered through deforestation. Return to natural patterns and process should remain in the forefront of forest restoration efforts throughout the world.

On the socio-political side, two common themes emerge from our research. First, effective restoration and management requires a blend of top-down and bottom-up approaches. Initially, widespread restoration efforts have been dominated by top-down influences indifferent to local interests and participation resulting in numerous political and ecological failures. However, the total abandonment of any top-down participation is not warranted; in fact, in many parts of the world, forest restoration will not be achievable without significant top-down organization and support. The challenge is to balance top-down capacity building and financial backing with local control for achieving meaningful forest restoration. Ensuring the trust necessary for local involvement often entails explicitly planning for the local distribution of short-term economic benefits of restoration. In many political jurisdictions, an effective way to accomplish this will be to help secure land tenure and resource rights for active restoration participants.

Second, in many ecoregions, forest degradation and deforestation are closely linked to narrow short-sighted governmental policies – particularly in the agricultural sector. International forces must be exerted on these policies to slow and ideally arrest the deforestation of most of the remaining undisturbed native forests.

Effective, forest conservation requires the understanding of three basic management strategies – protection, certification, and restoration. Each of these three strategies forms a component piece (or supporting leg) of forest conservation (Figure 3). In any particular situation or ecoregion a specific proportions of these elements will be most appropriate. For example, in ecoregions where large areas of pristine forestland remain, representative protection and forest certification should receive most of the attention. In areas where forestland is already highly degraded, protection of the remaining native forest remnants and restoration should move to the forefront of conservation efforts. Thus, restoration is only one component of forest conservation and is the one most difficult to implement.

This report has focused on restoration, but the other two legs of the forest conservation stool are critically important in providing the context for restoration activities. Protection is particularly important. Without adequate protection of native forests - the majority of the world's forests are poorly represented - these regions run the danger of failing to maintain biodiversity or providing the wide range of ecosystem services native forests provide modern human societies. In addition, without proper representation of native forests, we run the danger of losing our natural forest references upon which we depend to plan our restoration activities and the natural capitol to supply the materials needed (e.g., seeds, cuttings, dispersers, etc.) for restoration activities. Forest restoration is NOT a substitute for protection.

In fact, there are a number of obstacles restoration may never be able to overcome, which is a sober warning against the notion that humans can rebuild natural forests after we destroy them. Cairns (1995) lists three:

1) every ecosystem is the result of a unique, non-replicable history;
2) the original structure and function of most ecosystems is not known; and
3) certain indigenous species may not be available for reintroduction.

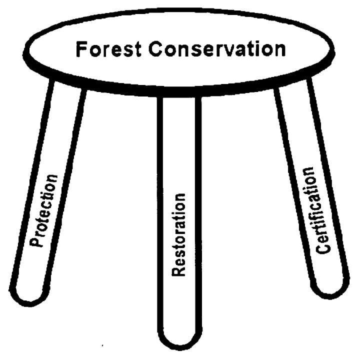

Figure 3. Three-legged stool diagram of forest conservation.

Knowing the limitations of scientific knowledge and capabilities is a necessary precondition before effective forest restoration actions can be taken. Our lack of knowledge about the biology and ecology of most forest ecosystems around the world cannot be overstated. To be effective, forest restoration must be approached in a humble, adaptive manner where we seek to learn as much as we can from the effects of our management decisions. Without adequate protected forests, however, the contribution restoration and certification play in forest conservation is greatly diminished. Only after an ecologically sustainable amount of forestland is protected over its range of physical variability within each ecoregion can we hope to have the role of certification and restoration properly defined and effective action plans developed.

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Plate 1
Tropical and Subtropical
Moist Broadleaf Forests
Neotropical
Forest Ecoregions
1 Andean yungas
2 Atlantic forests
3 Choco Darien moist forests
4 Coastal Venezuela montane forests
5 Greater Antillean moist forests
6 Guayanan forests
7 Napo moist forests
8 Northern Andean montane forests
9 Rio Negro-Jurua moist forests
10 Southwestern Amazonian moist forests
11 Talamancan and Isthmian Pacific forests
12 Varzea flooded forests
N
0 500 1000 1500 2000 km

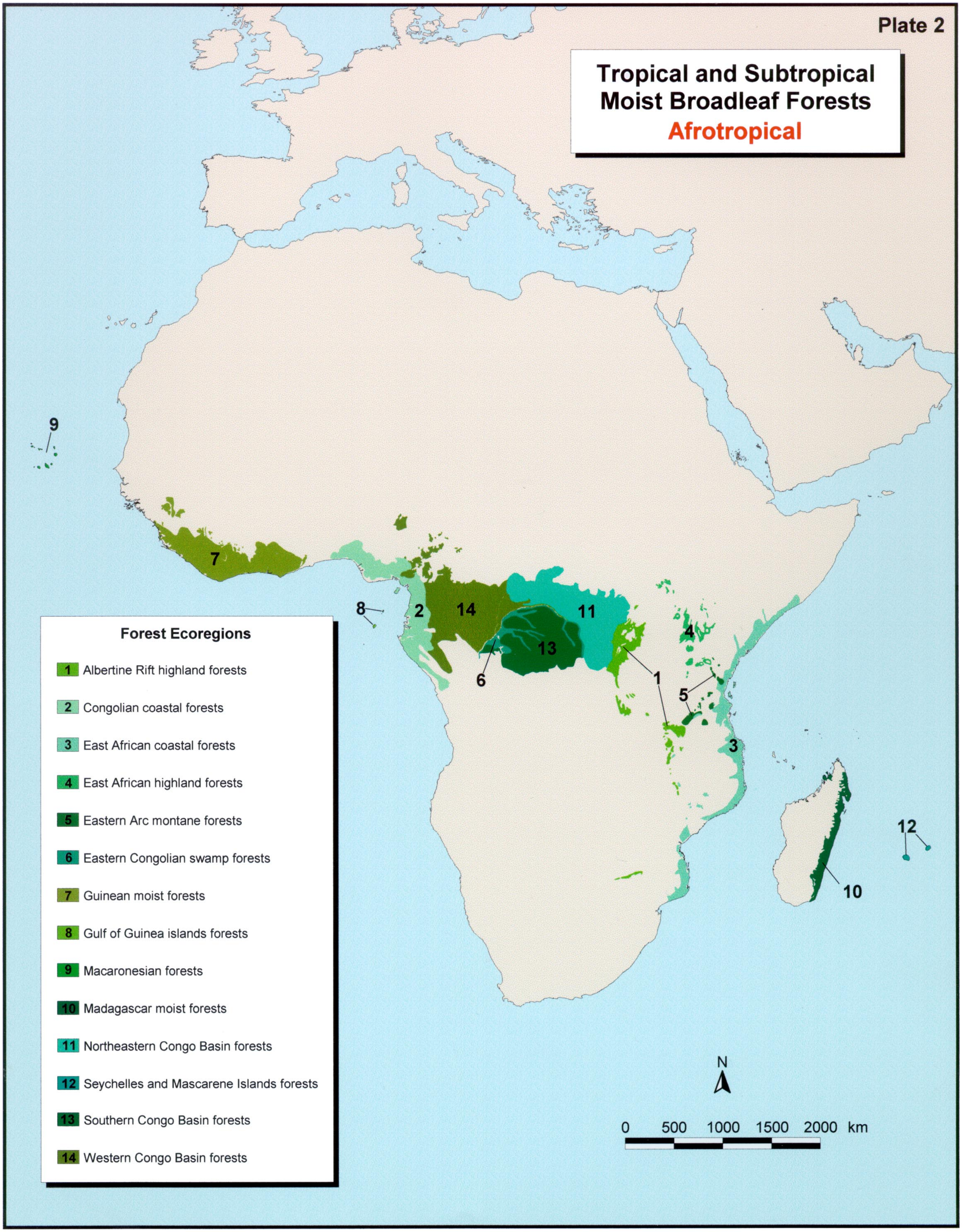


Plate 2
Tropical and Subtropical
Moist Broadleaf Forests
Afrotropical
Forest Ecoregions
1 Albertine Rift highland forests
2 Congolian coastal forests
3 East African coastal forests
4 East African highland forests
5 Eastern Arc montane forests
6 Eastern Congolian swamp forests
7 Guinean moist forests
8 Gulf of Guinea islands forests
9 Macaronesian forests
10 Madagascar moist forests
11 Northeastern Congo Basin forests
12 Seychelles and Mascarene Islands forests
13 Southern Congo Basin forests
14 Western Congo Basin forests
N
0 500 1000 1500 2000 km

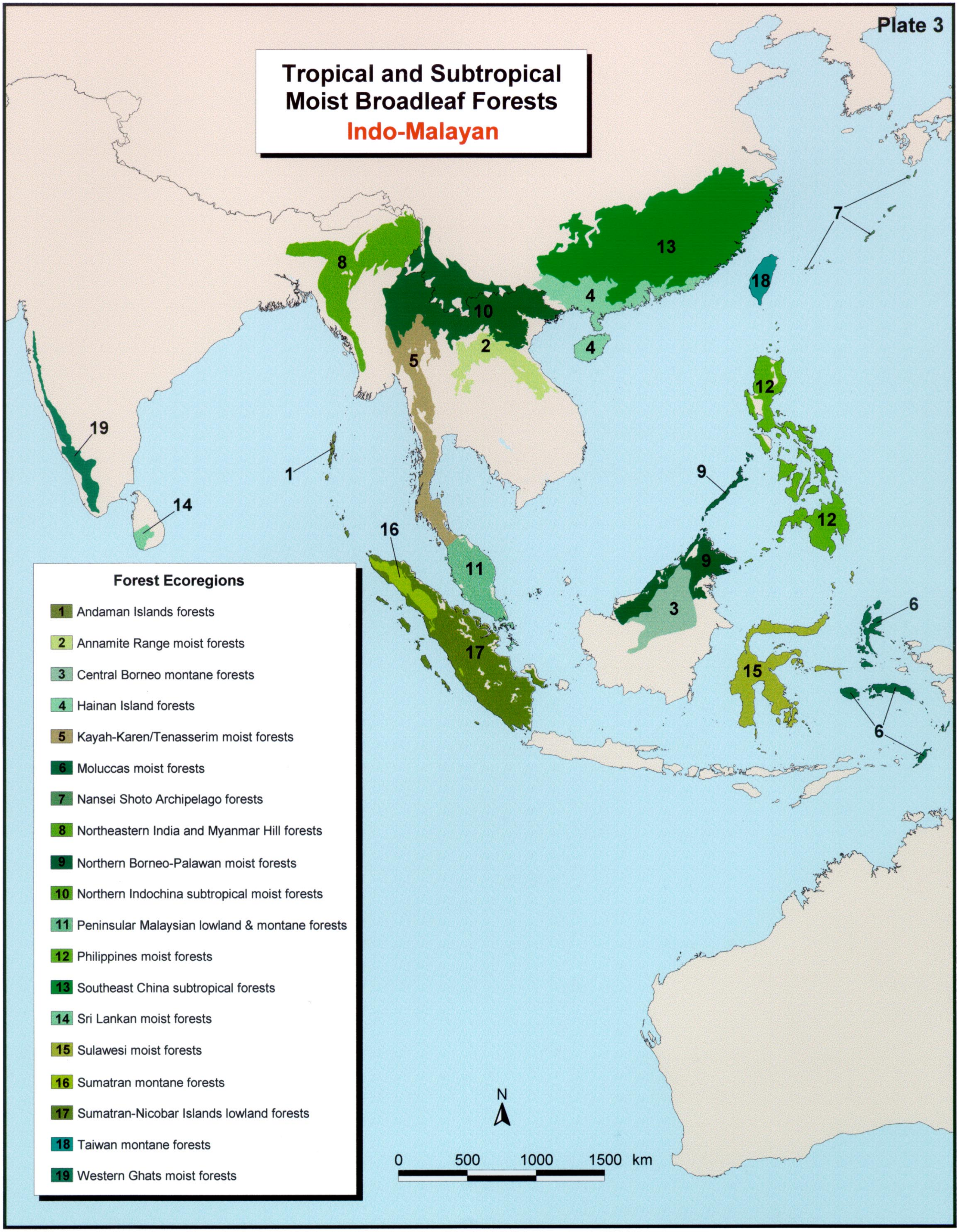


Plate 3
Tropical and Subtropical
Moist Broadleaf Forests
Indo-Malayan
Forest Ecoregions
1 Andaman Islands forests
2 Annamite Range moist forests
3 Central Borneo montane forests
4 Hainan Island forests
5 Kayah-Karen/Tenasserim moist forests
6 Moluccas moist forests
7 Nansei Shoto Archipelago forests
8 Northeastern India and Myanmar Hill forests
9 Northern Borneo-Palawan moist forests
10 Northern Indochina subtropical moist forests
11 Peninsular Malaysian lowland & montane forests
12 Philippines moist forests
13 Southeast China subtropical forests
14 Sri Lankan moist forests
15 Sulawesi moist forests
16 Sumatran montane forests
17 Sumatran-Nicobar Islands lowland forests
18 Taiwan montane forests
19 Western Ghats moist forests
N
0 500 1000 1500 km

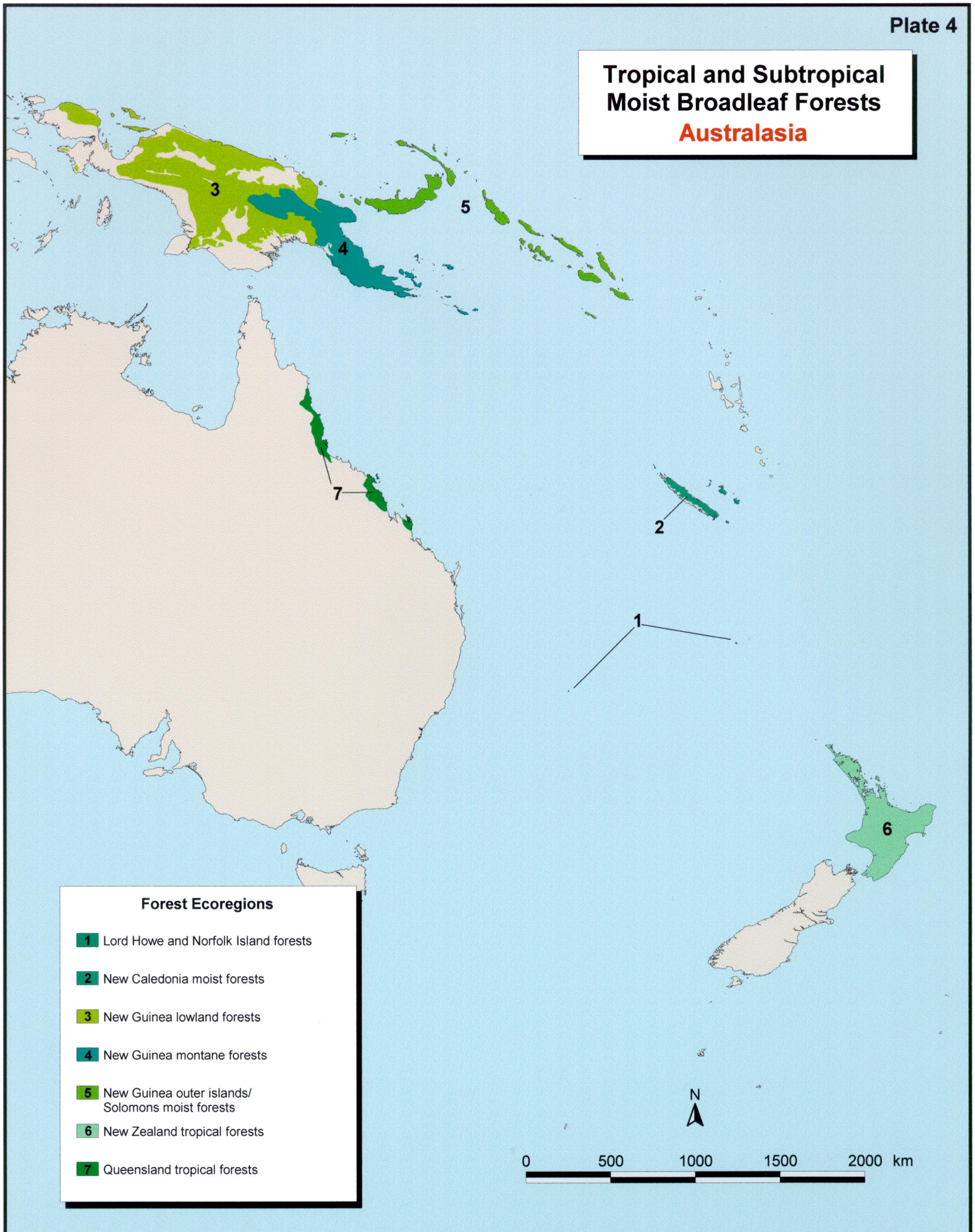
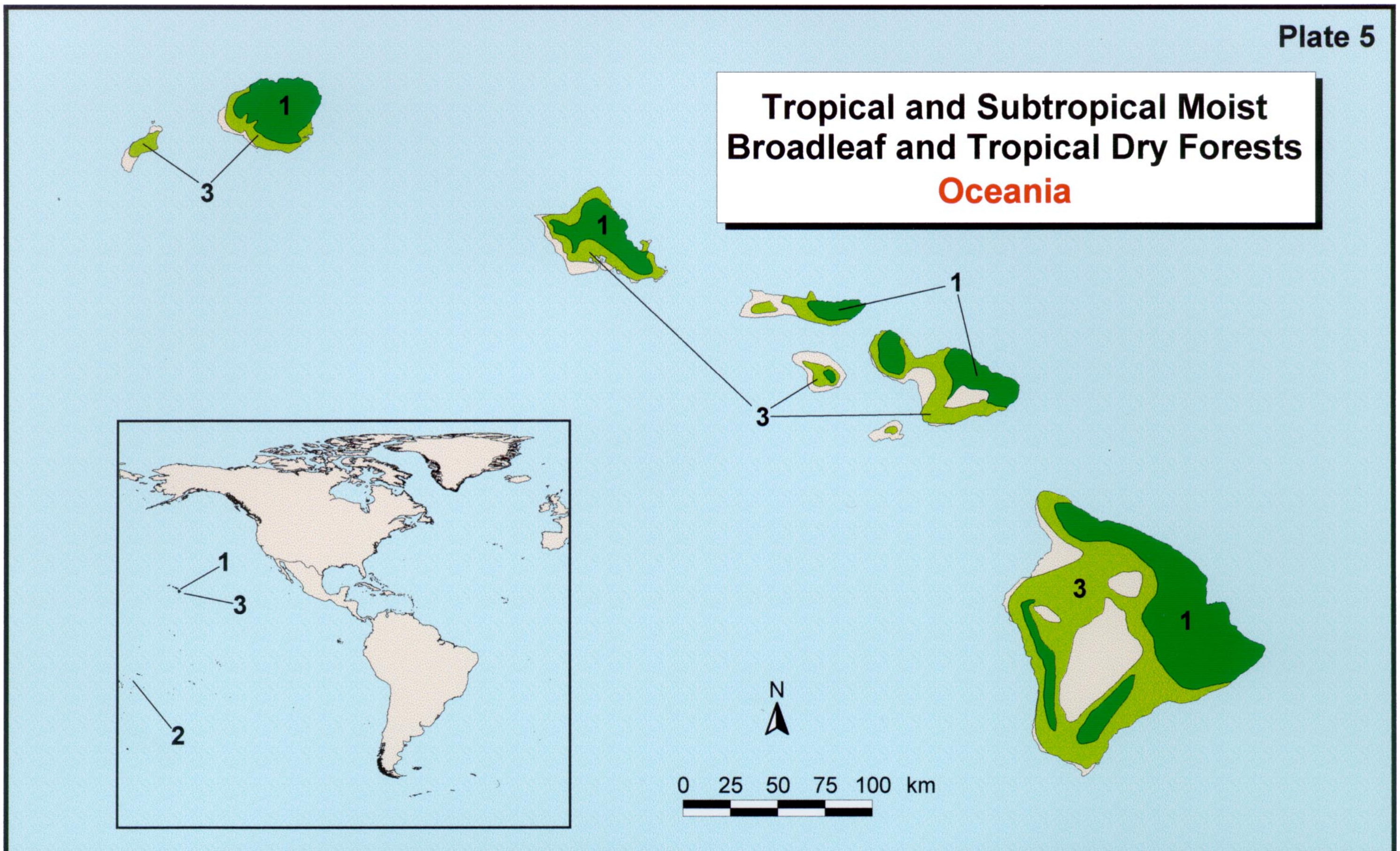
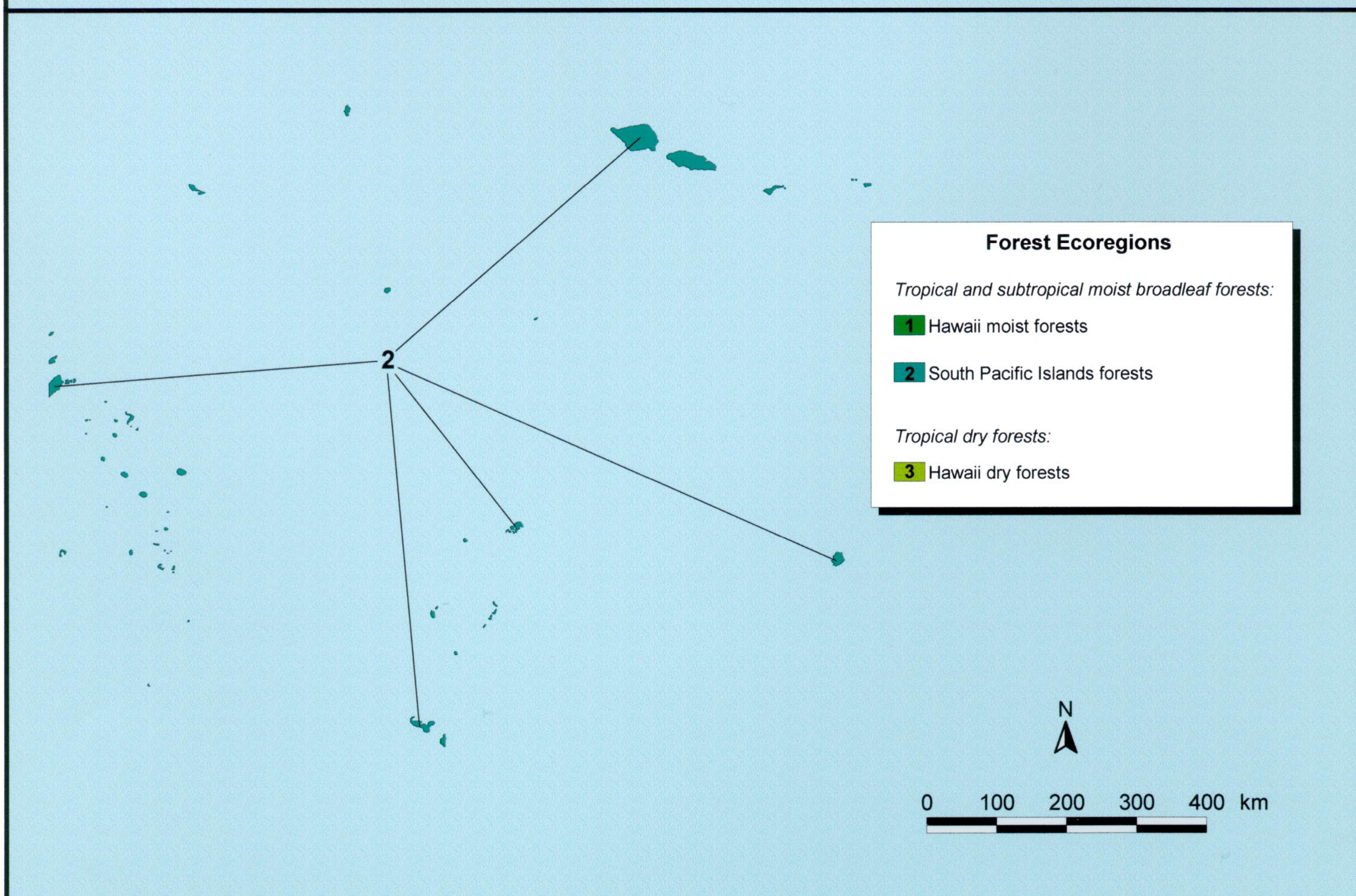


Plate 4
Tropical and Subtropical
Moist Broadleaf Forests
Australasia
1
2
3
4
5
6
7
Forest Ecoregions
1 Lord Howe and Norfolk Island forests
2 New Caledonia moist forests
3 New Guinea lowland forests
4 New Guinea montane forests
5 New Guinea outer islands/
Solomons moist forests
6 New Zealand tropical forests
7 Queensland tropical forests
N
0
500
1000
1500
2000 km

Tropical and Subtropical Moist Broadleaf and Tropical Dry Forests
Oceania





Forest Ecoregions

Tropical and subtropical moist broadleaf forests:

1 Hawaii moist forests

2 South Pacific Islands forests

Tropical dry forests:

3 Hawaii dry forests

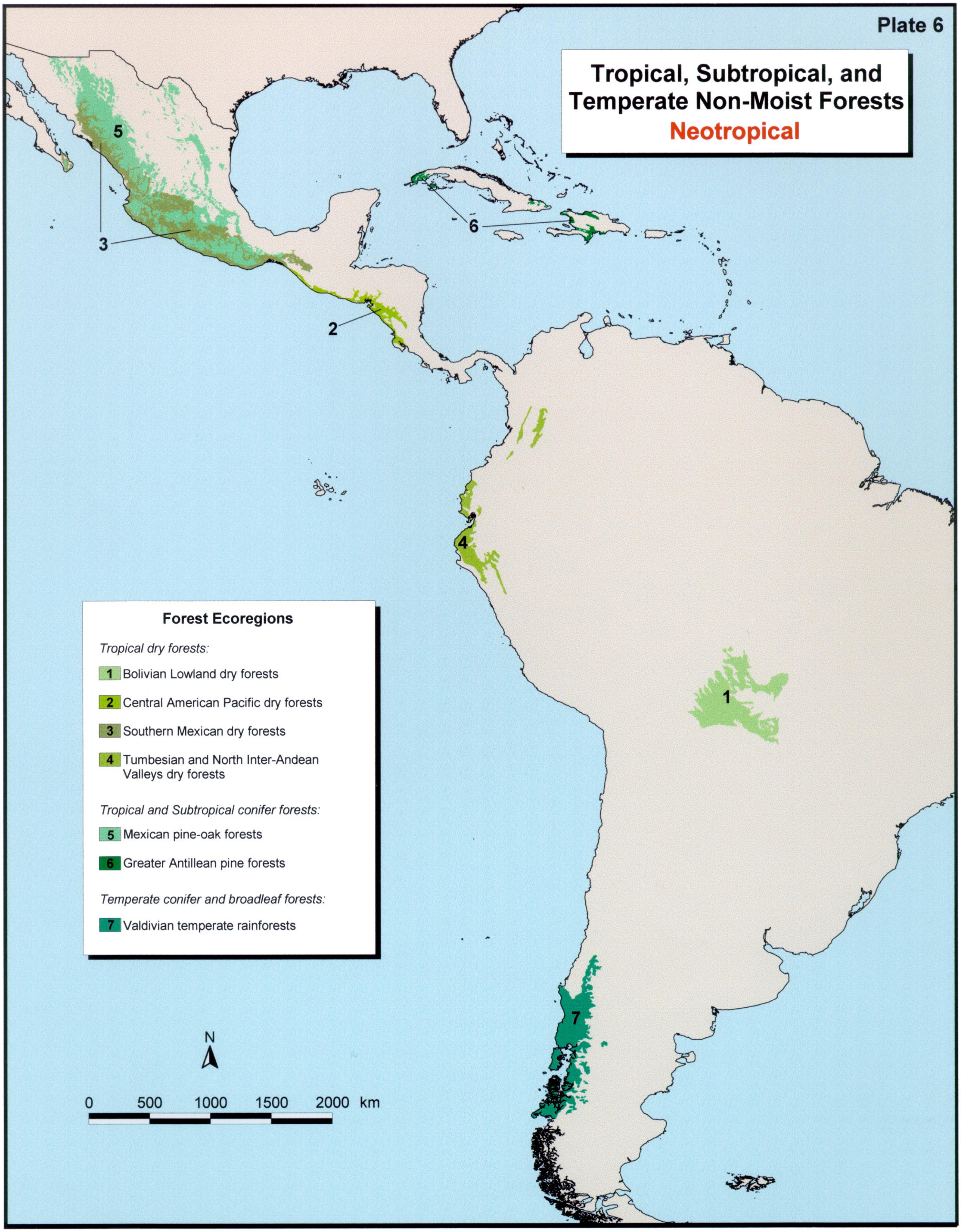


Plate 6
Tropical, Subtropical, and Temperate Non-Moist Forests
Neotropical
Forest Ecoregions
Tropical dry forests:
1 Bolivian Lowland dry forests
2 Central American Pacific dry forests
3 Southern Mexican dry forests
4 Tumbesian and North Inter-Andean Valleys dry forests
Tropical and Subtropical conifer forests:
5 Mexican pine-oak forests
6 Greater Antillean pine forests
Temperate conifer and broadleaf forests:
7 Valdivian temperate rainforests
N
0 500 1000 1500 2000 km
1
2
3
4
5
6
7

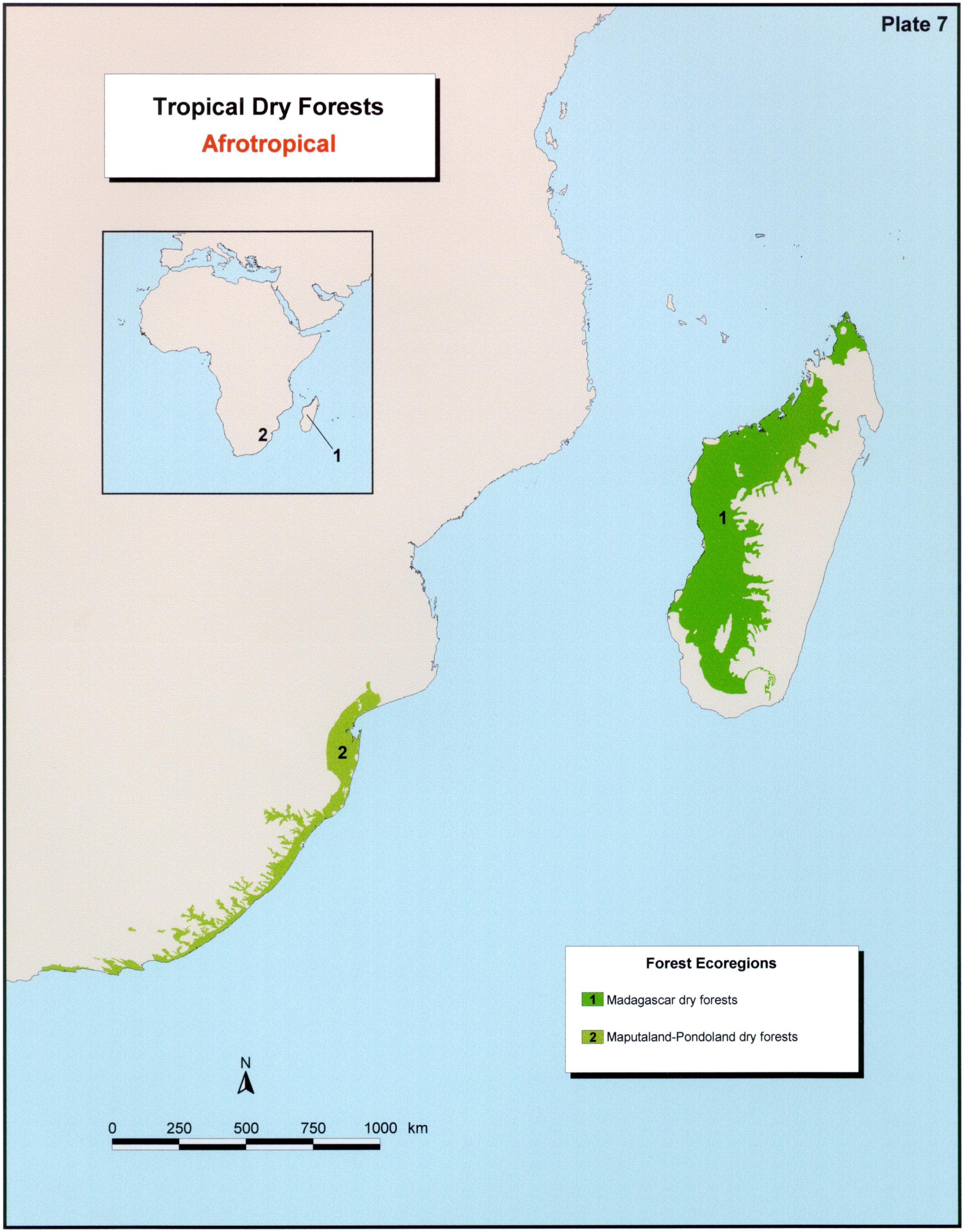


Plate 7
Tropical Dry Forests
Afrotropical
1
2
Forest Ecoregions
1 Madagascar dry forests
2 Maputaland-Pondoland dry forests
N
0 250 500 750 1000 km

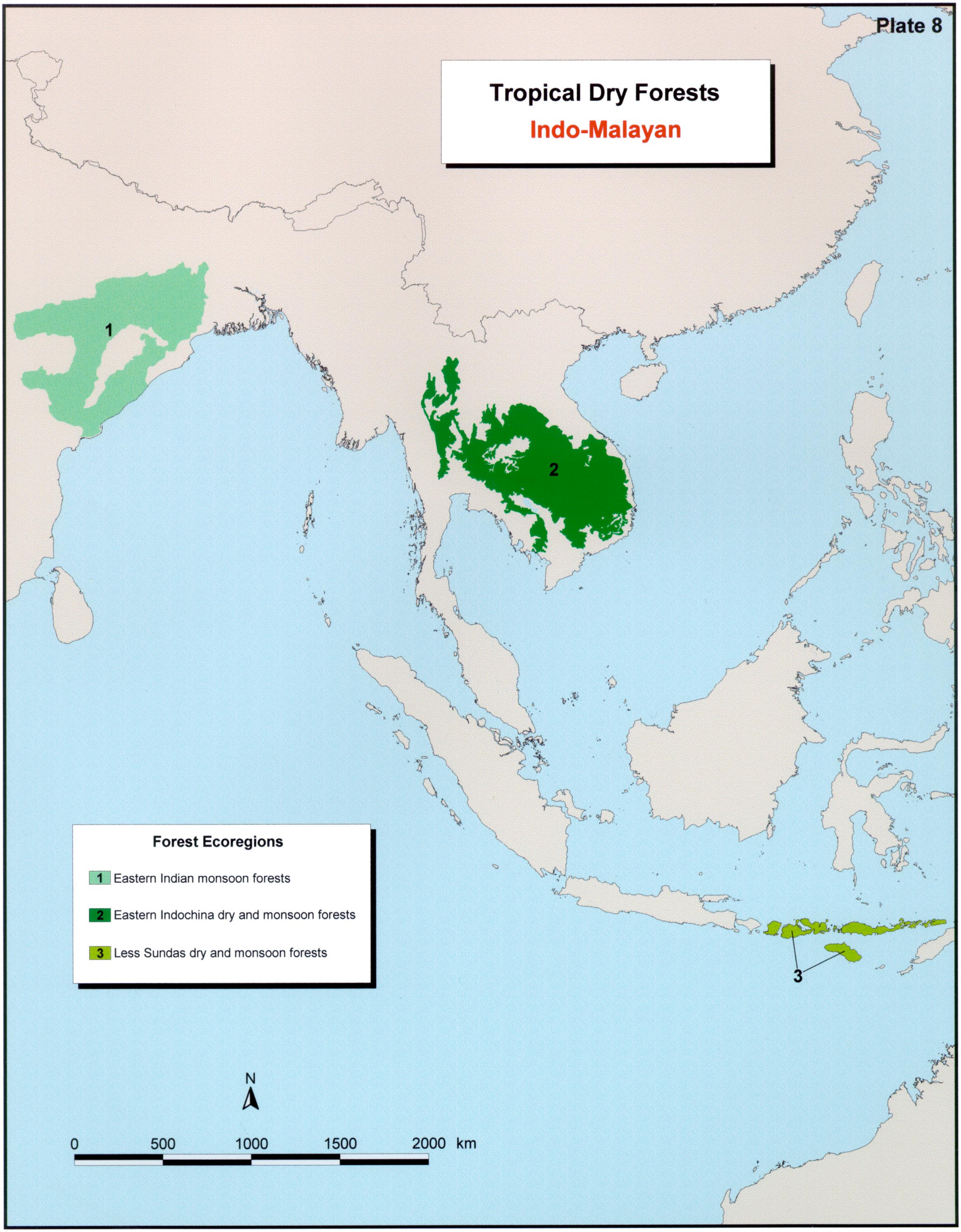


Plate 8
Tropical Dry Forests
Indo-Malayan
1
2
3
Forest Ecoregions
1 Eastern Indian monsoon forests
2 Eastern Indochina dry and monsoon forests
3 Less Sundas dry and monsoon forests
N
0
500
1000
1500
2000 km

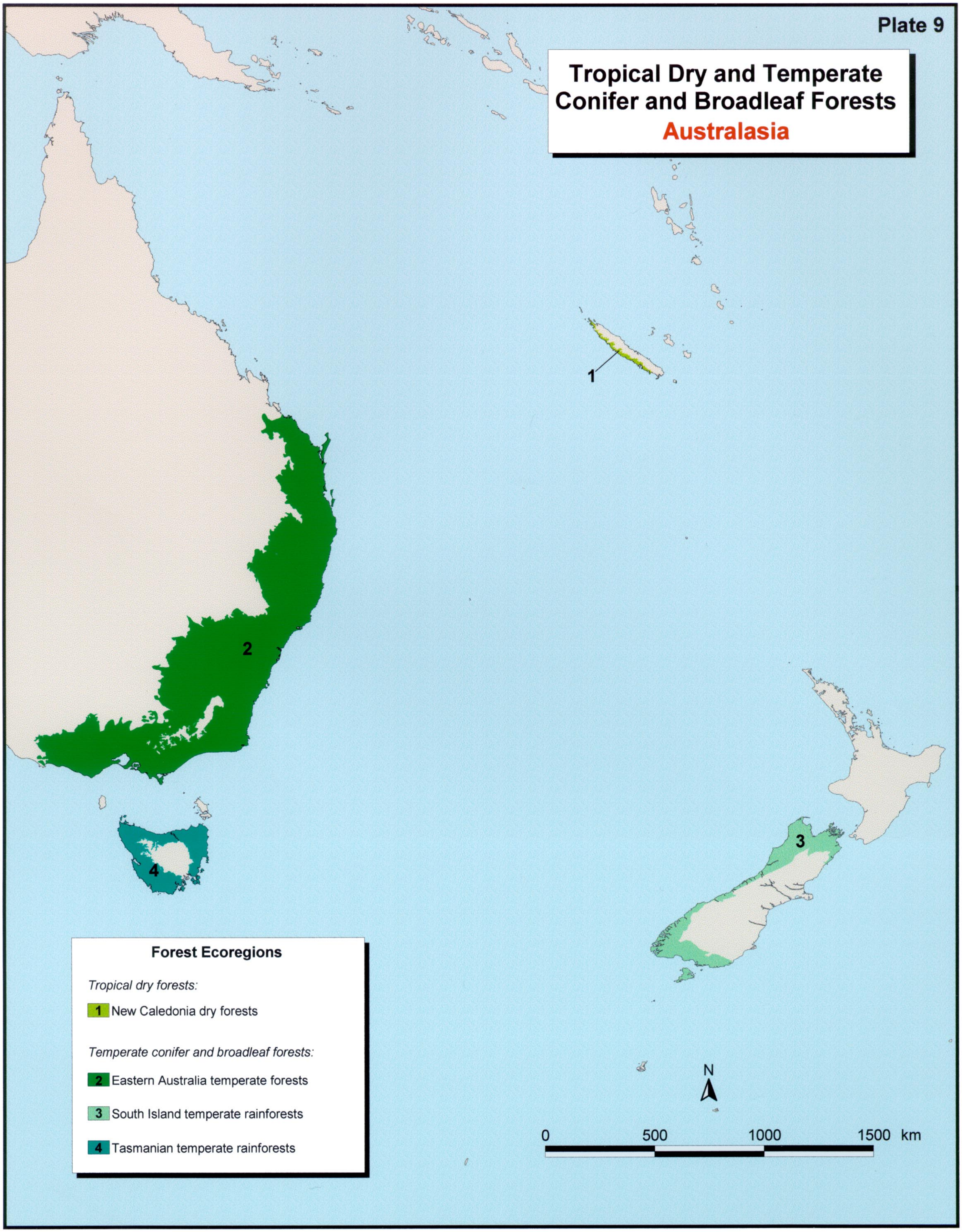


Plate 9
Tropical Dry and Temperate
Conifer and Broadleaf Forests
Australasia
1
2
3
4
Forest Ecoregions
Tropical dry forests:
1 New Caledonia dry forests
Temperate conifer and broadleaf forests:
2 Eastern Australia temperate forests
3 South Island temperate rainforests
4 Tasmanian temperate rainforests
N
0
500
1000
1500 km

Temperate Conifer and Broadleaf and Boreal Forests and Taiga
Nearctic