Biodiversity Conservation in the Context of Tropical Forest Management

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These "Impact Studies" are a subset of the Biodiversity Series of the World Bank's Environment Department Papers. Within this subset the broader question of what the positive and negative impacts of human activities on biodiversity is addressed.

The following studies have been published in this series:

1. Biodiversity Conservation in the Context of Tropical Forest Management
2. Hunting of Wildlife in Tropical Forests—Implications for Biodiversity and Forest Peoples
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Preface

This paper was prepared as part of the World Bank’s Overlay Program and as a contribution to the Bank’s ongoing Forest Policy review. The Overlays Program, launched by the World Bank in partnership with bilateral donors and NGOs, seeks to internalize global externalities into national environmental planning and the Bank’s sector work, operations, and its dialogue with governments and partners. It is an iterative process, combining conceptual studies, reviews of state-of-the-art techniques for measuring and mitigating local, national and global externalities, and testing these concepts and tools in country-level studies as a means of identifying good practices for country planners and Bank task managers. The results will help guide national actions to conserve biodiversity, reduce greenhouse gas emissions, and protect international waters.

The Overlays add a new dimension to traditional sector economic planning by analyzing environmental impacts and opportunities to internalize global externalities. This analysis launches the “Impact Studies” which asks the broader question of what the positive and negative impacts of human activities are on biodiversity. An analytical framework is used which allows for the systematization of these impacts across sectors and areas of particular interest.

One of the most important forest uses is logging or timber harvesting. Logging is the most lucrative forest use and can cause severe direct and indirect environmental impacts. This analysis evaluates the impacts of logging on biodiversity in tropical forests based on a thorough review of the literature and on the best expert opinion.

The study was commissioned from the Wildlife Conservation Society (WCS). A draft paper was prepared by WCS and the University of Florida and was distributed to an expert review panel. Supported by the Forest Policy Implementation Review and Strategy Process, these reviewers convened in Washington with the authors and Bank staff in late February, 1999 for a productive two day workshop after which the paper was revised.

Forests will continue to be harvested and timber harvesting will continue to be an important component of sustainable forest management. This is a reality we cannot escape. Therefore, as this practice continues we need to assure ourselves that it is ecologically and environmentally sustainable. The World Bank has a responsibility to its clients to inform them of best practices and what the potential consequences of natural resources management decisions are. This paper helps fulfill this responsibility.
Executive Summary

Existing parks and protected areas are cornerstones of biodiversity conservation but on their own are inadequate to assure the continued existence of a vast proportion of tropical forest biodiversity. As a result, priority must be given to ensuring that the greatest possible amount of biodiversity is conserved outside protected areas by altering harvest patterns in these landscapes of resource extraction. Of all forest uses within tropical forest regions, logging or timber harvesting, is the most important to influence because not only is it the most lucrative, it also causes the most severe direct and indirect environmental impacts.

Evaluating the impacts of logging on biodiversity in tropical forests is not as simple a task as many would believe. It depends on a thorough review of the literature and on the best expert opinion due to the complexities hidden under the seemingly simple rubrics “logging” and “biodiversity.” As a first step in effectively analyzing the relationship between biodiversity and logging, it is essential to begin by disaggregating the terms “logging” and “biodiversity.”

This paper disaggregates the term “biodiversity” into components (landscapes, ecosystems, communities, species/populations, and genes) and attributes (structure, composition, and function). It then disaggregates “logging” by detailing the vast range of activities subsumed under this term including variation of logging intensities, logging methods, collateral damage, and silvicultural approaches. Using the richness present in both these terms, a framework for considering the impacts of logging and other forest management activities on the various components and attributes of biodiversity is presented. This framework is, in turn, used to evaluate the extensive literature covering different studies of logging in tropical forests.

The analysis produces a number of key findings:

1. All consumptive uses affect some component or attribute of biodiversity, commonly affecting not only the target resource but other elements as well.
2. As a result, only fully protected areas will conserve all components and attributes of biodiversity.
3. Recognizing that all significant interventions in natural forests have biodiversity impacts, all silvicultural decisions necessarily represent compromises. Management for some goods or services necessarily involves management against others.
4. Different intensities and spatial patterns of timber harvesting, along with other silvicultural treatments, result in different effects on the different components of biodiversity.
5. Some components and attributes of biodiversity are more sensitive than others to forest management activities.
6. There are some forested areas and some areas within forests, which should never be logged.
7. Biodiversity objectives can and should be established within production forests.

8. In many cases the logging itself is not the major cause of biodiversity loss, but rather the indirect effects, often promoted by the presence of roads, (for example, hunting, forest fires, and the likelihood of deforestation) represent the major environmental threats.

Unfortunately, due to the complexities hidden under the seemingly simple rubrics "logging" and "biodiversity", the answer to the question "Is logging compatible with biodiversity protection?" can only be the very unsatisfying "It depends." The paper does not conclude with uncritical support for sustainable forest management of timber as a conservation strategy. Such an endorsement is unwarranted given widespread illegal logging in the tropics, widespread frontier logging and logging of areas of high priority for biodiversity protection, the persistence of poor logging practices despite substantial efforts in research and training, and the generally slow rate at which most loggers are transforming themselves from timber exploiters into forest managers.

Nevertheless, even harshly treated forests maintain more biodiversity than oil palm plantations, cattle pastures, or cornfields. From a biodiversity maintenance perspective, natural forest management is preferable to virtually all land-use practices other than complete protection. Forests that are carefully managed for timber will not replace protected areas as storehouses of biodiversity, but they can be an integral component of a conservation strategy that encompasses a larger portion of the landscape than is likely to be set aside for strict protection.
1 Introduction

Existing parks and protected areas are essential for biodiversity conservation but inadequate on their own to assure the continued existence of the majority of natural landscapes, ecosystems, communities, species and genotypes in tropical forests. Even if the oft-quoted goal of 10-12 percent protection were attained and the reserved areas were appropriately located and properly managed, up to 50 percent of tropical species would be expected to go extinct during the next few decades (Soule and Sanjayan 1998). Similar but often less well documented losses of the other components of biodiversity are expected. Another way of considering this dire prediction is that if biodiversity outside of protected areas is neglected, thousands of species are likely to disappear. Faced with this bleak future, the first priority should be to increase the area of forests under strict protection while at the same time improving reserve management. Mechanisms should be developed to halt road building and commercial logging in forest wilderness areas as well as in centers of diversity and endemism. However, promoting more biodiversity-sensitive management of forests outside protected areas is of almost equal priority, given the conservation potential of these still vast areas. Biodiversity conservation within forests managed primarily for timber production, for example, should be fostered by a combination of not logging ecologically important areas and by employing biodiversity-sensitive management methods within production areas.

Decisions about which large forest areas should be designated for strict protection, which areas within managed forests should be protected, and which approaches to silviculture to follow within forests used for timber production should be made on the basis of a wide range of considerations. Forests differ in their biodiversity value, in their capacity to support different intensities of silvicultural use, in pressures for conversion to non-forest use, and in the abilities of the relevant institutions to regulate their management. General conceptual approaches to zoning forests for different uses have been presented recently by various authors including Noble and Dirzo (1997), and Frumhoff and Losos (1998). Methods for integrating conservation functions at different geographical scales were reviewed recently by Poiani and others (2000). This paper focuses on the forests zoned for timber production. The goals are:

1. To contribute to the development of a heuristic construct for considering the range of impacts of forestry activities on tropical forest biodiversity at the levels of landscapes, ecosystems, communities, species, and genes.
2. To indicate the topics on which further research will be particularly useful in evaluating the impacts of different forestry activities on the various components and attributes of biodiversity.
3. To suggest ways to mitigate the deleterious impacts of forestry activities on tropical forest biodiversity.

In the search for land-use practices compatible with biodiversity maintenance, many
environmentalists have focused upon logging [see Grieser Johns (1997) and Haworth (1999) for comprehensive reviews of the environmental impacts of logging in tropical forests]. This emphasis is not surprising given that of all forest uses, logging is often the most financially lucrative and has the most severe environmental impacts. The indirect effects of logging, particularly increased hunting (for example, Robinson and others 1999) and the increased likelihood of deforestation due to improved access, reviewed by Kaimowitz and Angelsen (1998), have also been highlighted recently. What has emerged from the dust, mud, chainsaw noise, and diesel fumes is the idea that conservation of some components of biodiversity could be facilitated by collaboration between loggers and environmentalists. Some of the latter oppose the idea of promoting better forest management as a means for achieving overall conservation goals (or at least oppose financial investment in such efforts, Rice and others 1997, Bawa and Seidler 1998, Bowles and others 1998). This opposition notwithstanding, consideration of the inevitability of logging in much of the tropics, the many constraints on expansion of nature reserves in tropical countries, the challenges of protecting and managing the parks already demarcated on paper, sovereignty issues, development needs, and the implicit adoption of the “use it or lose it” assumption, motivate other conservationists to continue holding sustainable forest management (SFM) as a worthy conservation goal in forests outside of protected areas (for example, Dickinson and others 1996, Chazdon 1998, Poore and others 1999, and Whitmore 1999).

To help elucidate some of the factors upon which the compatibility of tropical forest logging and biodiversity protection depend, this paper first attempts to disaggregate the terms “logging” and “biodiversity.” The wide range of logging intensities, logging methods, collateral damage, and silvicultural approaches appropriate for tropical forests is discussed. A framework for considering the impacts of logging and other forest management activities on the various components (=landscapes, ecosystems, communities, species/populations, and genes) and attributes (=structure, composition, and function) of biodiversity is presented. In the main section of the paper, the components and attributes of biodiversity are used to review the impacts of logging and other silvicultural activities on tropical forests. To promote readability of the text, much of the bibliographical review is presented in appendices organized by the components and attributes of biodiversity. The paper concludes by suggesting ways to enhance the compatibility of forest management for timber and biodiversity maintenance in tropical forests.
Disaggregating “Biodiversity”

Biodiversity refers to the natural variety and variability among living organisms, the ecological complexes in which they naturally occur, and the ways in which they interact with each other and with the physical environment. This definition and the elucidation below is based upon OTA (1987), Noss (1990) and Redford and Richter (1999). Climate, geology and physiography all exert considerable influence on broad spatial patterns of biotic variety; local ecosystems and their biological components are further modified by environmental variation (such as local climatic and streamflow fluctuations) and ecological interactions. This natural variety and variability is distinguished from biotic patterns or conditions formed under the influence of human-mediated species introductions and substantially human-altered environmental processes and selection regimes (Bailey 1996, Noss and Cooperrider 1994).

In discussions of diversity there is always the lurking danger of assuming that “more is better.” Were this the case, plowing roads through mature forests, clearcutting patches, and introducing alien species would all be reasonable. Although decisions about which species or community-types are most valuable are not straightforward, rarity is one obvious dimension that needs to be considered.

Biological diversity can be measured in terms of different components (landscape, ecosystem, community, population/species, and genetic), each of which has structural, compositional, and functional attributes (Table 1). Structure refers to the physical organization or pattern of the elements. Composition refers to the identity and variety of elements in each of the biodiversity components. Function refers to ecological and evolutionary processes acting among the elements.

Diversity of the landscape component refers to regional mosaics of land uses, land forms, and ecosystem types. Structure of this component could be described on the basis of the areas of different habitat patches, through the perimeter-area relations of these habitat patches, or on the basis of interpatch linkages. Landscape composition pertains to the identity, distribution, and proportions of different habitat types. Landscape function refers to issues such as patch persistence and interpatch flows of energy, species, and other resources.

Diversity of the ecosystem component refers to interactions between members of a biological community and their abiotic environment. Structure of this component might be measured through vegetative biomass and soil structural properties. Ecosystem composition refers to features such as biogeochemical standing stocks. Ecosystem function pertains to processes including biogeochemical and hydrological cycling.

Diversity of the community component refers to the guilds, functional groups, and patch types occurring in the same area and strongly interacting through trophic and spatial biotic relationships. Structure of this component
# Table I Components and attributes of tropical forest biodiversity that might be influenced by logging and other silvicultural activities

<table>
<thead>
<tr>
<th>Components</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Landscape</strong></td>
<td>Size and spatial distribution of habitat patches (e.g., seral stage diversity and area); physiognomy; perimeter-area relations; patch juxtaposition and connectivity; fragmentation</td>
<td>Identity, distribution, and proportion of habitat types and multi-habitat landscape types; collective patterns of species distributions</td>
<td>Habitat patch persistence and turnover rates; energy flow rates; disturbance processes (e.g., extent, frequency, and intensity of fires); human land use trends; erosion rates; geomorphic and hydrologic processes</td>
</tr>
<tr>
<td><strong>Ecosystem</strong></td>
<td>Soil (substrate) characteristics; vegetation biomass, basal area and vertical complexity; density and distribution of snags and fallen logs</td>
<td>Biogeochemical stocks; lifeform proportions</td>
<td>Biogeochemical and hydrological cycling; energy flux; productivity; flows of species between patches; local climate impacts</td>
</tr>
<tr>
<td><strong>Community</strong></td>
<td>Foliage density and layering; canopy openness and gap proportions; trophic and food web structures</td>
<td>Relative abundance of species and guilds; richness and diversity indices; proportions of endemic, exotic, threatened, and endangered species; proportions of specialists vs. generalists</td>
<td>Patch dynamics and other successional processes; colonization and extinction rates; pollination, herbivory, parasitism, seed dispersal and predation rates; phenology</td>
</tr>
<tr>
<td><strong>Species/Population</strong></td>
<td>Sex and age/size ratios; range and dispersion; infraspecific morphological variation</td>
<td>Species abundance distributions, biomass, or density; frequency; importance or cover value</td>
<td>Demographic processes (e.g., survivorship, fertility, recruitment, and dispersal); growth rates; phenomenology</td>
</tr>
<tr>
<td><strong>Genetic</strong></td>
<td>Effective population size; heterozygosity; polymorphisms; generation overlap; heritability</td>
<td>Allelic diversity; presence of rare alleles; frequency of deleterious alleles</td>
<td>Gene flow; inbreeding depression; rates of outbreeding, genetic drift and mutation; selection intensity; dysgenic selection</td>
</tr>
</tbody>
</table>

*Note: Modified from Noss (1990) and Redford and Richter (1999).*

Includes consideration of vegetative physiognomy and trophic structure. Community composition refers to relative abundances of species and guilds. Community functions include flows between patch types, disturbance regimes (such as fires and floods), successional processes, and species interactions. Diversity of the species/population component refers to the variety of living species and their component populations at the local, regional, or global scale. Structure of this component might be measured through population age structure or species abundance distributions. Species/population composition pertains to which particular species are present. Function of this component refers to demographic processes such as recruitment and death.

Diversity of the genetic component refers to the variability within a species, as measured by the variation in genes within a particular species, subspecies, or population. Structure of this component can be expressed on the basis of
heterozygosity or genetic distances between populations in different patches (that is, metapopulation genetic structure). Genetic composition refers to which alleles are present and in what proportions. Genetic functions can be expressed on the basis of gene flow, genetic drift, or loss of allelic diversity in small, isolated populations.
3 Disaggregating “Logging”

When properly planned and conducted, logging (=timber harvesting) is an integral component of forest management systems designed to promote sustained timber yields (STY), or the more all-encompassing goal of sustainable forest management (SFM). Unfortunately, logging in tropical forests all-too-often represents a timber “mining” activity carried out without regard for the renewability of this natural resource (for example, Putz and others 2000). Other silvicultural treatments designed to promote sustainability are often prescribed but are rarely applied. Due mostly to the desire to obtain voluntary third-party certification of good management from the Forest Stewardship Council (FSC), the transition from forest mining to forest management has finally started to occur in some areas in the tropics (Nittler and Nash 1999). Despite this progress, even within certified forests there are questions about how to most effectively and efficiently minimize the deleterious environmental impacts of logging and other silvicultural activities. This paper addresses many of these questions by reviewing the state of knowledge about biodiversity maintenance in exploited and managed tropical forests.

It is critical to recognize that forest interventions of all types, from harvesting of fruits for home consumption to clearcutting for timber, all have impacts on forests that need to be understood and often deserve mitigation (Peters 1996). Nevertheless, logging is often the most damaging and generally the most financially lucrative of such forest interventions (Pearce and others 1999). Unfortunately, discussions about the compatibility of logging with biodiversity conservation are complicated because logging is carried out over a huge range of intensities, using a variety of techniques which may be applied carefully or in ways that result in a great deal of avoidable damage. The following sections focus on the issues of harvesting intensities, yarding methods (=how timber is extracted from the stump to haul roads), and ways of reducing logging damage. A brief overview of other silvicultural treatments used in timber stand management is also provided.

Logging intensities

Trying to conclude anything about the compatibility of logging and biodiversity maintenance is made complicated by the wide range of logging intensities, yarding methods, and accompanying forest management practices to which tropical forests are subjected. The simple observation that logging intensities span more than two orders of magnitude (<1 m³/ha to >100 m³/ha) illustrates the challenge of generalizing (Figure 1).

Even within the same forest management unit, logging intensities vary greatly at different spatial scales. At a small scale (1-10 ha), the typical aggregated distributions of tropical trees (Hubbell 1979) leads to locally severe logging impacts unless harvesting controls are implemented (that is, “tree-marking rules” are followed, Uhl and Vieira 1989, Crome and others 1992, Cannon and others 1994). The localized but severe direct impacts of roads, log
landings, and skid trails hardly need to be emphasized (Guariguata and Dupuy 1997). At a slightly larger scale (10–100 ha), stands vary in stocking of commercial species and in their accessibility due to terrain or edaphic factors. Where logging is carried out in areas designated for each year of management (=annual coupes), logging impacts tend to be aggregated at larger scales.

Logging intensities also vary over time, tending to increase with local timber shortages, improved access, and greater willingness of markets to accept lesser-known species (Plumptre 1996). For example, in a remote part of the Bolivian Amazon loggers may harvest an average of only 0.3 $m^3$/ha of 1–3 species (Gullison and Hardner 1993) whereas in similarly stocked stands in the more accessible eastern portions of Amazonian Brazil, it is likely that 30–50 $m^3$ of 20–30 species would be harvested (Johns and others 1996).

The substantial number of studies conducted on the impacts of logging in tropical forests all concluded that soil impacts and damage to the residual forest all increase with increasing logging intensity (Ewel and Conde 1976, Sist and others 1998a). Proportions of both soil and residual trees damaged by logging range from 5 to 50 percent, depending on harvesting intensity, yarding method, and the care with which the operations are carried out. Interpretation of data pertaining to the relationship between logging intensity and residual stand damage is complicated by concomitant change in residual stand density; at the extreme, there is no residual stand in

<table>
<thead>
<tr>
<th>Figure I</th>
<th>Logging intensities ($m^3$/ha) for tropical forests. In most of these studies, as in most logging areas in the tropics, felling was with chainsaws and yarding with bulldozers or articulated skidders with rubber tires.</th>
</tr>
</thead>
<tbody>
<tr>
<td>$&lt;1\ m^3$/ha</td>
<td>50 $m^3$/ha</td>
</tr>
<tr>
<td>Bolivia</td>
<td>Venezuela</td>
</tr>
<tr>
<td>Suriname</td>
<td>Costa Rica</td>
</tr>
<tr>
<td>Venezuela</td>
<td>(Mason 1996)</td>
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<tr>
<td>Australia</td>
<td>(Crome et al. 1992)</td>
</tr>
<tr>
<td>Brazil</td>
<td>Malaysia</td>
</tr>
<tr>
<td>(Johns et al. 1996)</td>
<td>(Pinard and Putz 1996)</td>
</tr>
<tr>
<td>Australia</td>
<td>(Crome et al. 1997)</td>
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<tr>
<td>Uganda</td>
<td>(Chapman and Chapman 1997)</td>
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clearcuts. Further complicating assumptions
about logging damage is the fact that few
tropical forests are logged (or burned) only once
(Box 1).

Log yarding methods
In this paper, particular attention is paid to the
yarding phase of the timber harvesting process
because much of the direct damage to tropical
forest caused by logging occurs while logs are
being extracted from the stump to roads or
riversides from which they are then hauled or
towed out of the forest. Yarding methods
utilized in tropical forests range in technological
sophistication, the collateral damages with
which they are associated, and in yarding costs
(Conway 1982). Accurate cost figures for
yarding are difficult to obtain and problematic
to calculate (for example, should impacts on
future timber yields be included, and, if so, at
what discount rate?). For the same volume of
timber yarded, the range in biodiversity impacts
at the landscape, ecosystem, community,
population/species, and genetic levels varies
greatly along the continuum of technological
sophistication of yarding methods that stretches
from manual extraction to the use of helicopters
(Figure 2). Note that Figure 2 hides a great deal
of relevant variation. In particular, attempts to
include another dimension expressing yarding
costs (per cubic meter yarded to the roadside)
have so far failed due to the multitude of factors
that need to be considered, including labor
costs, technical capacity, and discount rates,
along with the values assigned to future harvest
yields, non-timber forest products, biodiversity
protection, and ecosystem services.

Yarding can be carried out in an environmen-
tally/silviculturally sensitive manner, or can be
extremely destructive. The rankings of
environmental impacts on Figure 2 are just
suggestions of the typical amounts of damage
associated with different yarding techniques.

Box 1
Few tropical forests are logged or burned only once

Logging and wildfires in tropical forests are causally linked (Holdsworth and Uhl 1997) and also share the
characteristic of being self-promoting processes. Although the environmental damage resulting from unplanned
logging of primary forest by untrained crews is justifiably receiving attention from researchers and policy-
makers, few logged stands are allowed to recover for the full cutting cycle or rotation stipulated in their man-
agement plans. Instead, forests are repeatedly entered by loggers who sequentially "high-grade" the best re-
mainning timber. Although the costs of road building are substantial, the first cut is generally the most lucrative.
As more timber species enter commercial markets, and mills begin to accept smaller and lower quality logs,
incentives for multiple entry logging increase. The same logging firms are sometimes involved in a series of
entries into the same forest, but more often firms with lower operating costs and smaller equipment take
advantage of the increased access provided during the first harvest. A familiar adage in Amazonia is "ma-
hogany builds the roads." Reliable data on frequencies of multiple entry logging, either legal due to waivers
issued by governmental agencies or illegal, are apparently rare, but the process is unquestionably common-
place.

As with multiple entry logging, wildfires seldom occur only once (Peres 1999, Cochrane and Schulze 1999). In
forests that are not naturally maintained by fire, the first fire opens the canopy and thereby subjects the under-
story to rapid drying. Fuel mass in the understory is augmented by trees killed by the first fire, and by grasses
and other plants that regenerate under conditions of reduced competition. The resulting fire frequencies far
exceed the historical rates. In the eastern Amazon, for example, fire frequencies in Paragominas have recently
doubled and now occur far too frequently to allow forest recovery. Human made savannas and grasslands
along with severely degraded forests due to frequent fires are becoming more the rule than the exception
throughout much of the wet and moist tropics. On the basis of either the area affected or the magnitude of
impacts, the direct damages attributable to tropical forestry are minor compared to those of wildfires.
For ground-based yarding, using the smallest yarding equipment possible contributes to reducing the deleterious impacts of logging. Which yarding equipment is used is important, but so is the care with which yarding operations are carried out, as indicated by the contrasts between conventional and reduced-impact logging (RIL) on Figure 2. Unfortunately, although the silvicultural, environmental, and economic benefits of planning of log extraction routes have been recognized for many decades (see Putz and others 2000 for a review), well planned logging operations are still the exception in tropical forests.

Most destructive yarding in tropical forests is carried out with bulldozers (=crawler tractors). Bulldozers are excellent devices for constructing roads but are unfortunately versatile enough to yard timber as well. The excessive damage to soils and residual trees during conventional bulldozer yarding, especially at high intensities on steep slopes logged during wet weather by untrained crews is well known, but bulldozers still yard much of the timber in commercial logging areas in the tropics.

With the loss of forest in accessible areas in much of the tropics, logging is increasingly being relegated to flooded, steep, rocky, or otherwise adverse terrain. Because ground based yarding from such sites can be prohibitively expensive, they have often been left as unlogged refugia within harvested areas. Their status as refuges is jeopardized by helicopter yarding, because helicopters can yard timber from even the most adverse sites. Obtaining timber from these sites is becoming more cost effective under some conditions. Although areas from which timber is harvested by helicopters are not dissected by skid trails, haul roads are still needed, and with them, all...
the problems associated with increased access. An equally important concern about helicopter yarding is that areas traditionally avoided by loggers are rendered accessible and are likely to be harvested. Because helicopters leave no obvious trails, it is going to be very challenging to monitor their harvesting impacts.

**Logging and other silvicultural treatments**

Logging can be either a cause of a great deal of avoidable damage or one of a series of silvicultural treatments designed to promote the regeneration and growth of commercial timber species while protecting ecosystem services and biodiversity. In logging areas where sustained yield of timber is a priority, various silvicultural treatments can be used in combination with the appropriate logging regime to promote the regeneration or growth of commercial species. Carrying out silvicultural treatments in conjunction with logging reduces their cost while simultaneously reinforcing the idea that logging itself can be silviculturally useful.

Silviculturally appropriate harvesting prescriptions (from a timber stand management perspective) run the full gamut from single tree selection to clearcutting (Smith and others 1997). For forests with ample stocks of trees of all sizes of the commercial species (that is, negative exponential size-class frequency distributions), “polycyclic” (=uneven aged) methods such as single tree and group selection methods are generally suitable. In such a forest, the next crop is derived from the mixture of trees of intermediate size (for example, 20–40 cm dbh) at the time of first cutting. In contrast, at the time of the initial harvest the trees in the next crop in “monocyclic” (=even-aged) systems are seeds, seedlings, or saplings. Clearcutting is the most obvious example of a monocyclic system, but in “shelterwood management” a single-aged stand develops after two stages of harvesting. The first shelterwood harvest opens the canopy and otherwise stimulates regeneration, the second cut releases the established seedlings and saplings. There are innumerable variations on the themes of monocyclic and polycyclic harvesting, and each is appropriate under different conditions and in forests for which there are different silvicultural goals (Lamprecht 1989). Even within the same forest, the silviculturally most appropriate harvesting regime sometimes can change over distances of less than 100 m (Pinard and others 1999).

The most common method attempted for controlling timber harvesting in tropical forests is to simply set a minimum stem diameter for felling. Although theoretically easy to implement and monitor, minimum diameter rules are often inimical to achieving silvicultural goals. The problem with this system is obvious where minimum diameter limits are set below the sizes at which trees start to reproduce (Appanah and Manof 1991, Plumptre 1995). Minimum diameter rules also do not prevent harvesting of clusters of trees and thereby creating silviculturally unsatisfactory conditions such as where commercial species are favored by small canopy gaps or vines proliferate in large ones.

Logging is often the most severe of silvicultural interventions, but there are other prescriptions designed to increase the stocking of commercial tree species, or to increase the growth of trees already present. Retaining seed trees in harvested stands is one possible way to increase stocking, but for species that require mineral soil seed beds or minimal competition for germination, establishment, and subsequent growth, seed tree retention needs to be combined with various other treatments. Seed beds can be modified and competition can be reduced by controlled burning, mechanical scarification, or herbiciding plants competing with seedlings of the crop species. Where natural regeneration fails, or where particularly
Biodiversity Conservation in the Context of Tropical Forest Management

High stocking levels are desired, many foresters have tried planting seedlings of commercial species in gaps or along lines cleared through the forest (see Box 2). For stands in which regeneration of the commercial species is already established, various thinning and weed control treatments are often prescribed to accelerate tree growth and otherwise for “stand improvement.” Thinning around potential crop trees, often referred to by tropical foresters as “liberation thinning,” and vine cutting are two commonly prescribed but less commonly applied silvicultural treatments. In experimental areas where liberation treatments have been applied to enhance volume increments of commercial species, treatments are often prescribed at 10-year or more frequent intervals (de Graaf and others 1999). As in the case of logging, all of these other silvicultural treatments have impacts on biodiversity, but they have been much less well studied.

Reducing the impacts of logging and other silvicultural treatments

Substantial attention has been given recently to “reduced-impact logging” (RIL, Box 3). Most of the practices in the various RIL guidelines that have been promulgated of late have long been recognized as being environmentally sound and silviculturally appropriate (Bryant 1914). Full application of RIL techniques would represent a major step towards sustainable forest management (SFM), but RIL alone does not guarantee sustainability. Especially where tree species being harvested only regenerate in large clearings, the required silvicultural interventions to assure sustained yield of the same species may often be substantial (for example, mimicking the impacts of slash-and-burn agriculture or hurricanes followed by fires, Snook (1996), Fredericksen (1998), Dickinson and Whigham (1999), Pinard and others (1999),

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**Box 2**

**Enrichment planting**

Despite scarcity of successes and millions of dollars wasted, enrichment planting in gaps or along cleared lines continues to be invoked as a way of restoring the commercial timber production potential of severely degraded forests. The principal problem with enrichment planting is failure to tend the out-planted seedlings (Dawkins 1961, Dawkins & Philip 1998). Admittedly there are cases where few silvicultural options remain. Unfortunately, enrichment planting is often utilized in forests that should not need such an expensive intervention. Furthermore, it is often not recognized that enrichment planting as often applied results in conversion of natural forests into plantations. Mason and Thiollay (in press) compared the impacts of enrichment planting with conventional selective logging in Venezuela and found the former to have greater negative impacts on birds. With these caveats issued, a description of a large-scale enrichment planting project in Sabah, Malaysia may be useful because the project implementers are avoiding some of the pitfalls of this technique and are trying to mitigate some of its deleterious environmental impacts.

Funds from the FACE (Forests Absorbing Carbon Dioxide Emissions) Foundation of The Netherlands, Rakyat Berjaya (a subsidiary of the Innoprise Foundation) have been used to plant nursery grown seedlings of commercial species in several thousand hectares of forests degraded by extremely intensive and poorly controlled logging in Sabah, Malaysia. The project has developed rapidly since its inception in 1992. Contract planters are now trained in dendrology so that they can recognize natural regeneration, and are paid for its retention at the same rate as for planted seedlings. Planters are also responsible for tending operations and are not paid for seedlings that do not survive 3 years. Site matching of species has also improved, as have nursery techniques for tending wildlings (=seedlings extracted from the forest) and raising large numbers of seedlings at relatively low cost. Finally, to promote retention of wildlife in the planting area, seedlings of fleshy-fruit trees are planted along with the mostly wind dispersed timber species in the Dipterocarpaceae. When the planted stands are ready to harvest in 50 or so years, the concessionaire hopes to harvest 50-100 trees per hectare, an intensity tantamount to clearcutting but perhaps preferable to the forest conversion to pulpwood plantation option that has been adopted in much of the region.
Disaggregating “Logging”

Box 3
Is reduced-impact logging (RIL) cheaper?

Many people believe that RIL has been proven cheaper than conventional logging (CL) and are therefore perplexed at the lack of its adoption by loggers. Adoption of RIL techniques is of particular importance in the tropics where few countries have regulations that are both explicit enough and well enough enforced to prevent exploitative harvesting.

Numerous studies over the past decades have shown that by planning skid trails and directional felling to facilitate yarding, logging damage can be substantially reduced (Bryant 1914, Nicholson 1958, Redhead 1960, Fox 1968, Nicholson 1979, Ewel and Conde 1980, Hendrison 1990, Johns and others 1996, Pinard and Putz 1996, Blate 1997, Elias 1997, Winkler 1997, Uhl and others 1997, Haworth 1999). Several experimental plot-based studies have demonstrated that due primarily to reduced yarding costs, such straightforward improvements in logging methods also translate into direct financial benefits to the loggers (Marn and Jonkers 1981, de Graaf 1986, Malvas 1987, Jonkers 1987, Hendrison 1990, Gerwing and others 1996, Bertault and Sist 1997, Barreto and others 1998, Holmes and others 1999, Boltz 1999). If RIL is cheaper than CL per cubic meter yarded to the roadside, then why haven’t loggers spontaneously adopted RIL techniques out of enlightened self interest? A partial answer to this question may be that under normal operating conditions, at industrial scales, and especially on adverse terrain in wet forests, RIL may not always be cheaper (Putz and others 2000, Hammond and others 2000). Data on the comparative costs of RIL and CL from the “Reduced-Impact Logging as a Carbon Offset Project” in Sabah, Malaysia may serve to elucidate this issue. Tay (1999) and Healey and others (in press) reported that the financial profits from logging were substantially lower in a 450 hectare area harvested according to RIL guidelines than in a comparable area subjected to CL. The principal reason for lower profitability of RIL was the reduction in yield caused by reductions in the proportion of the area logged due to restrictions on bulldozer access to steep slopes. Despite regulations against timber yarding from steep slopes in Sabah, the practice is commonplace and loggers consider the timber foregone when RIL guidelines are followed as lost profits and thus do not spontaneously adopt RIL out of enlightened self interest. These findings indicate that the issue of cost effectiveness of RIL under adverse conditions deserves scrutiny and that loggers not using RIL techniques may not be acting in a financially irrational manner.

In contrast to the apparent situation regarding RIL adoption in Southeast Asia, large vertically integrated forestry companies in Brazil have begun to invest heavily in RIL apparently out of enlightened self-interest. Although there is an additional incentive for adoption of more environmentally friendly logging in the form of better enforcement of regulations governing forest management operations, it appears that some companies have been convinced by the work of Baretto and others (1998), Holmes and others (1999), and others that their profits will increase if they adopt more efficient (and resource friendly) practices. Loggers in the region are also applying RIL techniques in response to an increase in demand for timber certified by the Forest Stewardship Council.

Given the differences in the results of comparative studies on the financial profitability of different logging techniques, adoption of RIL techniques under some conditions may require either subsidies (such as carbon offset funds) or stricter regulations with better enforcement. The conditions under which incentives for RIL adoption are needed, and the best form and magnitude of these incentives are yet to be determined.

Fredericksen and Mostacedo (2000). This silvicultural challenge notwithstanding, careful planning and implementation of harvesting guidelines would represent a big step towards sustainable forest management.

For all yarding methods, logging damage can be substantially reduced, and logging costs reduced as well, by proper design, construction, and maintenance of road networks. Much of the cost of harvesting timber and a large proportion of the hydrological damage (for example, stream sedimentation) due to logging is associated with roads (Bruijnzeel 1992). For ground-based yarding, efficient skid trail layouts are also essential for reducing logging.
damage and increasing yarding efficiency, but
improper road siting often restricts skid trails to
inappropriate places. Although the required
density of roads is lower where aerial extraction
techniques are used (such as skyline and
helicopter yarding), roads are still needed and
their locations can greatly influence logging
costs and environmental damage. Guidelines
for road construction are abundantly available
and are well outlined in the FAO Model Code of
Forest Harvesting Practices (Dykstra and
Heinrich 1996). Unfortunately, forest
engineering standards in most tropical logging
areas are extremely low and there are too few
experienced forest engineers involved in most
tropical logging operations. It is important to
note that because soil damage due to poor skid
trail design or improper use, for example,
reduces productivity, increases surface erosion,
and has various other deleterious
environmental impacts, adhering to RIL
guidelines has advantages regardless of
whether the forest is allowed to regenerate or is
replaced by an oil palm plantation or a maize
field (Congdon and Herbohn 1993, Nussbaum
and others 1995, Pinard and others 1996).

The impacts of other silvicultural treatments on
biodiversity (such as thinning and vine-cutting)
depend on the intensity with which they are
applied and on the proper designation of areas
deemed inappropriate for stand "improve-
ment." For example, vinecutting can enhance
tree growth but undoubtedly has negative
impacts on a wide variety of animals (Putz and
others in press). Both biodiversity impacts and
labor costs of vine cutting depend on whether
only selected future crop trees are liberated or
vine cutting is carried out as a blanket
prescription.
4

Impacts of Forest Management on Biodiversity

Background

Human activities are highly variable in their influence on the components and attributes of biodiversity. Any human activity that results in substantial resource extraction or modification will always entail significant, often unknown, and almost always unappreciated consequences for one or more biodiversity components, primarily by redirecting matter and energy flows. The cumulative redirection is enormous at the planetary scale as three examples illustrate: 1) Vitousek and others (1997) calculated that 40 percent of the Earth’s terrestrial primary productivity is being appropriated by humans, 2) 25 to 35 percent of the primary productivity of continental shelf marine ecosystems is consumed by humans (Roberts 1997), and 3) Postel and others (1996) report that humans now appropriate 26 percent of total evapotranspiration and use 54 percent of all runoff in rivers, lakes, and other accessible sources of water.

Despite these statistics on current human impacts and the observations of the long-term impacts of previous generations of our species (Denevan 1992), many in the conservation and sustainable development community still maintain it is possible to both use and preserve biodiversity (Huston 1993) with no costs to either side. This claim is made regardless of a history of human over-exploitation of resources that began in prehistory (Goudie 1990) and is manifested most recently in the negative effects of tropical logging (Bawa and Seidler 1998, Frumhoff 1995) and non-timber forest product exploitation (Coomes 1995, Homma 1992). This ahistorical and wishful thinking is extremely dangerous because it allows its adherents to believe that there exist easy, cost-free solutions to exploitation of the planet.

As societal concerns about the fates of tropical forests increase and human demands on forests change, so should the ways in which they are silviculturally treated. For example, a few decades back when the primary component of sustainability of interest to most forest decision-makers was sustained timber yields (STY), it was often recommended that logging be followed by silvicultural stand “improvement” treatments such as poison girdling of non-commercial trees, for a historical review see Dawkins and Philip (1998). Unfortunately, few of the tens of thousands of hectares of tropical forests that were treated according to the various silvicultural systems used in the 1950s (for example, the Malayan Uniform System and the Tropical Shelterwood System) remain, most having already been logged again, converted to oil palm plantations, or otherwise lost. Fortunately, there is now evidence from a lowland dipterocarp forest in Malaysia that at least the Malayan Uniform System (MUS) could achieve its silvicultural goals (Lee and others 1998) with mixed impacts on biodiversity, depending on the taxon in question. The forest treated by MUS and then studied 45 years later was, as intended, enriched in commercial timber trees, but also maintained much of the fauna and flora of primary forest. More studies of this sort are needed, and they need to be
better circulated so that more people will realize that at least “qualified” successes in tropical silviculture are attainable.

In the following sections (4.2–4.6) the impacts of logging and other silvicultural treatments on the components and attributes of biodiversity using the framework elaborated in section 2.0 (see Table 1) are outlined; details are provided in the appendices. Summarizing the impacts of forestry activities on biodiversity in tropical forests is an effort fraught with problems in large part due to their immense diversity. Even at the species level, the diversity is difficult to imagine and each of the literally millions of species in tropical forests responds to different logging impacts in distinct ways. While some general response patterns are obvious, and others have emerged from field research, the idiosyncrasies and apparent inconsistencies of species responses need to be recognized. For example, chimpanzee (Pan troglodytes) populations have been reported to increase (Howard 1991, Hashimoto 1995), decrease (White 1992), and not respond to logging (Plumptre and Reynolds 1994, see Appendix VI). In contrast, studies of terrestrial and bark-gleaning insectivorous birds have consistently reported negative impacts of logging (see Appendix IV). Many more examples of this sort are discussed briefly below, and both consistent and inconsistent patterns emerge.

**Landscape impacts (see Appendix I)**

Logging affects the landscape component of biodiversity by changing land forms and ecosystem types across large geographic areas (Box 4). Logging shifts the regional mosaic of land uses. Although the landscape component of biodiversity is the least sensitive to logging, changes in the size, spatial distribution, and connectivity of habitat patches across the landscape occur especially as the intensity of management interventions increases. These changes in the habitat mosaic alter species distribution patterns, forest turnover rates, and hydrologic processes. The most severe impacts described in this section, however, result from indirect consequences of logging such as increased access to remote areas, fragmentation, and altered fire regimes.

**Structure**: The size and spatial distribution of tropical forest patches and the juxtaposition and connectivity of different forest patches across the landscape are most affected by the indirect impacts of logging. One of these impacts—increased access to humans—is often deleterious. Particularly when logging roads penetrate far into the forest frontier in countries where there are numerous potential forest colonists, logging is almost invariably accompanied by increased hunting pressure (Robinson and others 1999, Robinson and Bennett 2000) and is often followed by deforestation as lands are cleared for agriculture (Kaimowitz and Angelsen 1998).

Another indirect impact of logging on this attribute is fragmentation of previously contiguous or otherwise connected forest patches. The resulting forest fragments, however, are typically not completely isolated for all species for all time. For example, wide logging roads may represent uncrossable barriers for some forest interior species but roadsides with secondary vegetation attract many large ungulates where they are more easily hunted (Robinson and Bennett (2000) in Sarawak, Malaysia). Whether logged stands are interspersed within species-rich forest or in a low diversity landscape dominated, for example, by pulpwood plantations, also influences long-term species maintenance. Furthermore, even small unlogged patches within harvest areas can serve as source populations for some species after logging. The degree of fragmentation depends on whether logging is dispersed over large areas, or is concentrated in small areas (Figure 3). Where
Box 4

The critical role of production forests in maintaining biodiversity in the tropics

Tropical rainforests harbor approximately 50 percent of all terrestrial biodiversity and much of that biodiversity occurs in the lowland forests which are most accessible for, and thereby most threatened by, logging and agricultural conversion. In Peninsular Malaysia, for example more than 50 percent of all mammal species occur below 350 m and a startling 80 percent occur below 650 m (Stevens 1968, MacKinnon and others 1996). Worldwide tropical forest loss is estimated as at least 17 million hectares/yr, an area the size of Cambodia (FAO 1993). Dramatic as it is, this global total is probably a gross underestimate given that recent figures show that Indonesia alone has lost 18 million hectares of tropical forests between 1985 and 1997 in just three of the major Outer Islands—Kalimantan, Sumatra and Sulawesi. Most of this forest loss has occurred in lowland forest and logging has played a major role in that destruction. In many areas forests have been cleared and converted for agriculture, plantations and transmigration. However, it is not logging activities per se that have caused forest and biodiversity loss but poor concession management and new access along logging roads. These factors have allowed shifting farmers and hunters to colonize and clear logged areas. Moreover, areas that have been logged (whether legally or illegally) are more vulnerable to wild fires during El Niño years; there is no doubt that poor logging practices contributed to the great Indonesian fires in 1998 when 5 million hectares of forest in Kalimantan burned, an ecological, economic and social disaster.

It is not just the reduction in total area of natural habitat that concerns conservationists, it is also the fact that remaining habitat is being broken into ever smaller fragments within which species’ populations are no longer viable. As habitats become more fragmented by clearance along new roads, protected areas will increasingly become “islands” and more and more species populations will be fated for local extinction. Fragmentation is particularly serious because few protected areas still cover a full range of altitudinal habitats as large areas of lowland forests, the most species-rich habitats and prime habitat for many large and wide-ranging mammals, have been excised from designated protected areas for logging (such as Kerinci, and Gunung Leuser National Parks in Sumatra). For Gunong Leuser National Park, for example, the main elephant, orangutan and tiger populations occur in the lower-lying production forests outside present park boundaries. Even more alarming is the recent increase in logging and other illegal activities within the boundaries of several National parks in Indonesia.

The loss of bird species on Java is an indicator of what can be expected in Borneo and other large islands as forests are cut and fragmented. Studies of forest bird distribution shows that whereas in Borneo, Sumatra, and New Guinea, maximum avian species richness is still found in lowland forests and species numbers decrease with altitude, Java is atypical in having a depauperate avian fauna in the hill zone between 300 and 1500 m. This pattern is interpreted as a result of long term deforestation since some predominantly lowland species have failed to survive in hill forests because they have been cut off from lower altitude populations which were a source of colonizers. Families of large birds such as malkohas have lost proportionally more species than families of small birds such as flowerpeckers. Moreover, Java has lost more species which are exclusive to lowland rainforests than species which can occupy secondary habitats, forest edge or open habitats. Small forest patches have lost more species than larger blocks, with extinction rates as high as 80 percent in 10-40 hectare plots compared to rates of 25 percent for areas over 10,000 hectares. These results have important implications for reserve design, forest management and biodiversity conservation. They emphasize that forests must be maintained as forests and that reserves must be large, cover a wide altitudinal range and be connected by corridors of natural habitats. This necessitates a landscape approach to biodiversity conservation by managing production forests adjacent to core protected areas to maintain both permanent forest cover and biodiversity so that production forests become buffer zones that effectively extend and supplement the conservation estate (MacKinnon and others 1996, MacKinnon and Phillipps 1993).

preservation of forest interior biodiversity is the priority, concentrating logging in small areas is generally the pattern recommended by conservation biologists (Noble and Dirzo 1997). Fortunately, due to associated cost savings, concentration of logging activities is one of the
Figure 3 Some impacts of logging on biodiversity as a function of distribution of logging activities (percent of area logged) and logging intensity (m³/ha of timber harvested)

<table>
<thead>
<tr>
<th>Logging Intensity</th>
<th>Area Logged (percentage)</th>
</tr>
</thead>
<tbody>
<tr>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>Concentrated Damage</td>
<td>Worst Case for Biodiversity</td>
</tr>
<tr>
<td>Moderate Yield</td>
<td>Maximum Yield</td>
</tr>
<tr>
<td>Large Area Impact-Free</td>
<td>Small Area Impact-Free</td>
</tr>
<tr>
<td>Low Impact</td>
<td>Dispersed Damage</td>
</tr>
<tr>
<td>Minimum Yield</td>
<td>Moderate Yield</td>
</tr>
<tr>
<td>Large Area Impact-Free</td>
<td>Small Area Impact-Free</td>
</tr>
</tbody>
</table>

Steps towards improved forest management that loggers may find acceptable.

The impacts of logging and other timber stand management activities on landscape structure are minimized when plantation products are used in place of those from natural forest. Setting aside reserves within logging areas may mitigate some of the deleterious impacts of logging and other silvicultural treatments. The specific location of reserves substantially influences their value in biodiversity conservation. Optimally, the full range of landscape features and habitats should be represented within protected areas.

Function: Logging may markedly alter several landscape level ecological processes subsumed under the functional attribute of the landscape component of biodiversity. For example, logging roads, and activities associated with their construction, can greatly influence the permanence of the forest fragments they create by altering landscape level disturbance regimes. In large part, disturbance is altered because roads and skid trails provide ready access to the forest for both colonists and fire (see below). High intensity and widespread logging, especially if not carefully controlled, also

Composition: Logging activities may directly and indirectly affect the identity, distribution, and proportion of habitat types in tropical forests. Forestry may directly affect composition of the landscape component of biodiversity by intentional creation of new types of habitats (such as forests converted into plantations). Furthermore, if silvicultural objectives are uniform across the landscape, inter-stand diversity is sacrificed by widespread application of the same stand “improvement” treatments. Perhaps most importantly, improved access provided by logging roads indirectly fosters post-logging habitat changes by human forest colonizers, weeds, and wildfires.

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impacts hydrological processes at all levels perhaps including the regional climate. Where logging roads are wide and logging intensities are high, landscape level movements of animals can be disrupted (Goosem 1997), as can gene flow in plants when pollinators are restricted to isolated fragments by inhospitable surroundings. It should be noted, however, that the impacts of roads depends on where and how they are constructed, and change with time as the road and its margins mature (Lugo and Gucinski 2000).

In this section the emphasis is on the impact of fires on the functional attributes of landscapes in full recognition of the importance of fires to all components and attributes of tropical forest biodiversity. Although fires have long played substantial roles in the ecology of tropical forests throughout the world (Goldammer 1990, Saldarriaga and West 1986), wildfires as exemplified by the recent Indonesian fires have recently increased in frequency, extent, and intensity in part due to widespread logging (Cochrane and Schulze 1999, Cochrane and others 1999). Canopy opening due to timber harvesting operations increases the rate of forest drying and thereby increases inflammability even in characteristically wet forests and swamps (Holdsworth and Uhl 1997). Perhaps equally importantly, vegetation proliferation along logging roads and skid trails greatly enhances fire penetration into forest interiors even where logging damage is slight or non-existent. Whether or not the extensive forest fires that have recently become common can be indirectly attributed to logging, it is clear that fire damage often exceeds logging damage in many tropical countries. In Bolivia in 1999, for example, fires burned an estimated 20,000 km\(^2\) whereas logging was carried out in perhaps only one-tenth of that area (W. Cordero, pers. comm.).

Although difficult to detect on satellite images, but see Souza and Barreto (in press) for recent improvements in relevant remote-sensing methods, the impacts of the understory burns typical for tropical forest fires are substantial. Especially in forests that have not burned for many decades to centuries, the effects of even a single understory burn are great, even if they are not fully realized for many years. Typically small trees are killed whereas large trees often receive only minor basal damage. Unfortunately, basal damage exposes trees to wood rotting organisms; a high incidence of heart rot is typical of forests that burn at low intensities at infrequent intervals (F. E. Putz, pers. obs.). Elevated rates of tree mortality rates for many years post-fire are also characteristic in seldom-burned forests. Such long-term data are difficult to obtain because accumulation of fuel in the form of fallen branches as well as post-fire proliferation of palms, grasses, and other fire-carrying plants render forests very fire prone. Surprisingly, the effects of tropical forest fires on biodiversity have not been extensively studied (Leighton and Wirawan, 1986).

**Ecosystem impacts (see Appendix II)**

The deleterious ecosystem-level impacts of logging on tropical forests are widespread, substantial, enduring, and well studied (especially compared to the landscape and genetic components). The ecosystem component of biodiversity is somewhat more sensitive to logging impacts than the landscape component in part because management activities are usually implemented at this scale. In contrast to the landscape component, most ecosystem-level impacts are a direct consequence of logging activities. Logging purposely removes biomass from ecosystems, but it also alters their vertical complexity and soil characteristics. Depending on the silvicultural objectives, changes in structural heterogeneity may be intended. Whether intended or not, the structural impacts of logging alter the relative proportions of lifeforms and biogeochemical stocks, as well as nutrient and hydrologic cycling, productivity and energy flows.
Structure: Logging may affect the structural attribute of the ecosystem component of biodiversity by changing the biophysical properties of soils, spatial heterogeneity of forest stands, and biomass. The extent and types of ecosystem damage to these structural attributes depend on logging intensity, yarding system, and the care with which the operations are conducted. Soil compaction, for example, is a major problem during ground-based yarding operations especially where skid trails are unplanned and yarding continues during wet weather. Where compaction is severe, soil permeability and bulk density often require many decades to recover. Exposure of mineral soil after litter layers and root mats are bladed off by bulldozers is also a concern. Mineral soil disruption during bridge building, road construction and maintenance, and skidding operations also represent forest management activities that affect ecosystem structure and thereby have biodiversity impacts.

Logging induced soil compaction negatively affects hydrology, which in turn alters the characteristics of watercourses. For example, water infiltration rates into soil, surface flow rates and volumes, and stream channel characteristics are all adversely affected by uncontrolled logging. Deposition of large volumes of unconsolidated sediments into streams during road construction and due to increased erosion from exposed mineral soils on log extraction routes can also greatly modify stream characteristics such as pool-riffle-run ratios. The impacts of such changes on aquatic organisms have been little studied in the tropics.

Another impact of logging on ecosystem-level structure is reduction in biomass and alteration of necromass. Losses of biomass due to forest management activities include the amounts in the removed timber, damage to trees in the residual stand that result in mortality, and any silvicultural treatments that result in tree death. Biomass losses range from 5–10 Mg/ha at the lowest logging intensities, to substantially greater amounts where heavy logging is followed by poison girdling of non-commercial trees in the residual stand. Necromass, including coarse woody debris, increases immediately after logging but may then decrease to levels below pre-logging conditions due to increased temperatures near the ground and associated increases in decomposition rates. Wildfires promoted by logging road construction and canopy opening, as well as controlled burns carried out for silvicultural purposes, can have obvious effects on biomass and necromass stocks in tropical forests.

Maintenance of healthy communities, species populations, and gene pools is predicated upon protection of hydrological functions, nutrient cycles, and other ecosystem properties. Fortunately, methods for mitigating the ecosystem-level impacts of logging on tropical forests are well known. Switching from ground-based to skyline yarding techniques, for example, greatly reduces damage to residual stands, soils, and streams but allows harvest on steep slopes. The opposite trend in technological change also can have environmental benefits; log yarding with draft animals or by manual means generally results in substantially less logging damage than yarding with bulldozers (Cordero 1995). To enjoy these potential benefits, the expertise of experienced forest engineers should be more often called upon where logging does have to occur.

Composition: One important way that logging affects the ecosystem-level compositional attribute of biodiversity is by changing biogeochemical stocks. For example, soil compaction reduces water holding capacity, which in turn leads to increased surface runoff. Limited storage capacity in natural streams is further reduced by sedimentation, which means that flow regimes can be greatly modified by logging, especially during the first years after logging is completed. Various RIL techniques,
Impacts of Forest Management on Biodiversity

such as installation of cross drains on skid trails, can greatly diminish these impacts.

Where large volumes of timber are harvested and post-logging forest recovery is retarded by soil damage, fire, or weed infestations, standing stocks of nutrients in biomass are greatly reduced. Storage of nutrients released from biomass in necromass and soil organic matter is generally brief in lowland tropical forests due to accelerated rates of decomposition and low cation-exchange capacities in the soil. The results can be substantial leaching and, if the forest is burned, volatilization of nitrogen and sulfur.

Function: Logging affects the functional attribute of ecosystem-level biodiversity by adversely affecting hydrological and biogeochemical fluxes as well as productivity. Reduced plant productivity results in part from impeded root growth, a consequence of logging-induced soil compaction. Because most of the available nutrients are usually found near the top of the soil profile, blading of the soil surface also diminishes nutrient availability in local areas and otherwise interferes with nutrient cycling. In more extensive portions of logging areas where RIL guidelines are not followed, nutrient cycling and hydrological functions are greatly modified by reduced canopy interception of rain and mist, decreased uptake of water and nutrients by the diminished biomass, and increased occurrence of surface erosion and landslides especially associated with improperly located and poorly constructed roads and skid trails.

Changes in carbon storage and flux associated directly and indirectly with logging and other silvicultural activities influence whether forests are net sources or sinks of "greenhouse" gases. For example, substantial logging-induced transfers of living trees to coarse woody debris can have substantial effects on understory structure and dynamics leading to more carbon release. The deleterious impacts of logging on forest carbon balance can be greatly diminished by application of RIL techniques (Pinard and Putz 1996). Substantial biodiversity benefits are also likely to result from RIL, but this and other co-benefits have not been well studied. Other silvicultural treatments such as fire management, weed control, thinning, and enrichment planting have various impacts on both greenhouse gas emissions and biodiversity that should be investigated.

Community impacts (see Appendix III)

Logging, especially if followed by silvicultural treatments such as liberation of future crop trees from competition, can substantially change the physiognomy, composition, and trophic structure of forest stands. To a large extent, these modifications represent the goal of forest "refinement" treatments applied to increase volume increments and relative densities of commercial timber species. This "stand domestication" by nature reduces species richness; rare, threatened and endangered species may become locally extinct especially if they have no perceived commercial value. These changes in composition and structure affect numerous community-level ecological processes including colonization, predation and mortality rates, pollination, seed dispersal, and timing and abundance of flower and fruit production.

Structure: The most obvious logging-induced impact on the structural attribute of community-level biodiversity is the change in proportions of successional stages in forest stands. Depending on harvesting intensities, planning of roads and skid trails, and training and supervision of workers, logging can result in large changes in the proportion of forest in mature, recovering, and early successional stages. In some severely disturbed areas, succession might be "arrested" by post-logging proliferation of vines, bamboo, and other non-arboreal growth forms. Silvicultural treatments such as thinning and vine-cutting can increase
the rate of succession and increase the proportion of stand growth concentrated in commercial species, but not without affecting biodiversity in more than the intended ways.

**Composition:** For the community component of biodiversity, logging affects composition by changing (often purposefully) the relative abundance of species and guilds inhabiting forest stands. Relative abundances of tree species with light demanding vs. shade tolerant regeneration, wind vs. animal dispersed seeds, vertebrate vs. invertebrate pollinated flowers, and thick vs. thin bark, for example, are all subject to change in logged and otherwise silviculturally treated forests. Likewise, representation of different guilds of animals (such as understory insectivores and arboreal folivores) is influenced by forestry activities. Depending on a great number of factors related to the intensities, spatial scales, and modes of forest intervention as well as characteristics of the focal taxa, effects of forestry activities can be negative, positive, or neutral. For example, in eight studies that considered the impacts of logging on frugivorous birds, two reported positive impacts at the guild level, three reported negative impacts, and three reported no change at all (Appendix IV).

**Function:** The functional attribute of community-level biodiversity includes numerous key ecological processes (such as pollination, herbivory, seed dispersal and predation) all of which are affected by logging especially under the most intensive management interventions. Many of the effects on these processes are a direct consequence of altered resource abundance (for example fruit for frugivores or young leaves for folivores), which in turn result from the logging-induced changes in community structure and composition. In addition to being influenced by resource-base changes, these ecological processes are also affected by changes in forest microclimates that result from exploitation, silvicultural treatment, and by hunting (see Box 5).

**Species impacts (see Appendices V and VI)**

The species component of biodiversity has received the most attention from researchers concerned about the impacts of logging and other silvicultural treatments in tropical forests. The most obvious species-level impact of logging is on the abundance and age/size distribution of harvested and damaged trees. Depending on the intensity of logging and the care with which it is carried out, the reproduction, growth, and survival of a great number of species can be adversely affected. In reviewing this literature, it is important to note that the taxa studied were not selected at random. Instead, in many cases, the species chosen were expected to be sensitive to, and thus good indicators of, forestry impacts.

**Structure:** The most immediate and direct impacts of logging on the structural attribute of the species component of biodiversity are suffered by the harvested tree species. Their populations are often left greatly depleted, especially in the larger size classes of reproductive individuals when management is based solely on minimum diameter felling rules. Because of the spatial clustering characteristic of many commercial timber trees, the richest patches of forest are generally the most severely disturbed, unless logging guidelines specify minimum spacing between harvested trees.

Changes in forest structure are suffered most by specialist species of the forest interior. After logging, many formerly shaded microenvironments in the forest interior become drier, brighter, warmer, and more easily exploited by some predators. For example, severe canopy opening adversely affects litter invertebrates and their predators. For species that are generalists in their diets and wide-ranging in their habitat use, such as many frugivorous canopy birds, the direct impacts of logging vary
Box 5

Hunting in logged tropical forests

The escalating scale of wildlife harvest in logged areas is an insidious problem that may undermine the biodiversity conservation potential of forests managed for wood production (Robinson and others 1999, Fimbel and others in press). Logging operations multiply the harvest of wildlife from tropical forests primarily by increasing access to previously remote areas. These same access roads serve as conduits for commercially traded meat and other wildlife products. Although the magnitude of the impacts of subsistence hunting of tropical forest animals (for food, feathers, and skins for example) by indigenous people are non-trivial (Robinson and Redford 1991, Redford 1992, Robinson and Bodmer 1999, Peres 2000), the impacts of commercial hunting are several orders of magnitude greater. Of particular concern is the tendency for hunters to target almost any species larger than 1 kg, including many species that are vulnerable to extinction due to their long life spans and low rates of population recovery (Bodmer and others 1997).

Hunting pressures increases in logging areas partially because of the number of people involved in logging operations as well as by the increased access provided by logging roads. In formerly isolated forest areas, loggers themselves hunt (Ruiz and others in press) or provide ready markets for meat and other wildlife products (Wilkie and others 1992, Bennett and Gumal in press). Local communities quickly shift toward commercial hunting to supply markets to which logging roads provide access. This shift initiates a positive feedback cycle wherein money gained from wildlife products buys better weapons with which to hunt, which in turn increases the harvest. Such escalations are unsustainable, however, and some species become locally extinct.

Loss of wildlife threatens the sustainability of tropical to the extent that the animals targeted by hunters play key roles in ecological processes including seed dispersal, seed predation, and herbivory (Redford 1992, Jansen and Zuidema in press). Even when animal species are not extirpated, their numbers may be reduced sufficiently such that they are rendered ecologically extinct. The loss of wildlife may have cascading effects on the structure and composition of the tropical tree community with deleterious consequences for recruitment of commercial timber species. Such dire predictions are most likely to be borne out in areas such as the Guaynas where a large proportion of canopy trees are dispersed by large animals (Hammond and others 1996). Finally, as human populations expand and forest landscapes become more fragmented, the potential for wildlife to recolonize defaunated areas will likely diminish.

While the deleterious consequences for wildlife of logging and other silvicultural treatments (such as vine cutting) deserve further investigation and mitigation, it should be recognized that the indirect impacts of logging resulting from increased access often far outweigh the initial damage done by even the worst predatory logging.

from being somewhat negative, to neutral, to positive. For the understory species that are adversely affected by logging, the effects may persist for decades (Wong 1985, but see Lee and others 1998). It is worth reiterating that the impacts of hunting on populations of large and slow-reproducing animals generally overwhelm any potential benefits that they might have enjoyed after the canopy was opened by logging (Bennet and Dahaban 1995, see Box 5).

Population sizes and structures of most species are also drastically modified by the fires that so often accompany uncontrolled logging. Species impacts of logging and stand refinement treatments are of particular concern in small forest management units. Private landowners with less than 100 hectares to manage, for example, may be unwilling to set aside 10 percent of their forest for species preservation. If their forests are surrounded by similarly managed or deforested areas, then blanket application of stand refinement treatments or heavy logging can take a heavy toll on commercial and non-commercial species alike.

Composition: Logging affects the composition attribute of species-level biodiversity by
changing the abundance and distribution of species. Unless logging is accompanied by other silvicultural treatments designed to foster their reproduction and growth, the abundance and population structure of the harvested tree species are greatly modified by logging (see Box 6). Logging impacts on tree populations continue for many years after logging is completed because damaged trees suffer high mortality rates, proliferation of weeds (such as vines) interferes with tree reproduction and survival, and population size reduction and fragmentation can decrease pollination levels and change patterns and intensities of seed dispersal and predation. Species composition of animals also changes in response to the direct impacts of logging (such as canopy opening) and the associated indirect impacts as well (for example increased fire frequency and intensity, hunting, and forest conversion).

Changes in species composition in response to forestry operations are by no means consistent across or even within taxa. Our review of the literature on primates, for example, revealed few cases of consistent responses of species to logging (Appendix VI). This variation can be attributed to differences in logging intensities as well as to differences in the duration of post-logging population monitoring. Silvicultural treatments other than logging, especially vine cutting and crown liberation of future crop trees, might have more consistent deleterious impacts on canopy animals, but such impacts have been little studied.

Small fragments of untouched forest that remain within even heavily logged forests serve as important refugia for plants and animals. Wildlife densities in these unlogged fragments can be very high during and shortly after harvesting, but then diminish as animals recolonize the surrounding matrix. Many "unplanned" reserves are on steep or otherwise adverse sites, which certainly influences their function as refugia. Implementation of

Box 6
Lesser-known species—Marketing challenges, silvicultural impediments, or desirable diversity?

Sustaining volumetric yields of timber is made challenging in tropical forests where many trees have little market value (Plumptre 1996). Particularly where just a few species of light-demanding trees have ready markets, species that are commercially lesser known or that have less desirable timber properties often render regeneration-enhancing treatments prohibitively expensive (Fredericksen 1998, Pinard and others 1999). Where markets for formerly lesser-known species have developed and they can therefore be harvested profitably, forest managers can financially justify creating the large canopy openings required for regeneration of the most valuable light-demanding timber species.

The same lesser known species that interfere with timber stand management treatments and present challenges for wood technologists and marketing firms represent an important component of forest diversity. One indication of their importance is that whereas tree species with wind dispersed seeds (such as Dipterocarpaceae and Meliaceae) dominate the international timber trade (ITTO 1996), a high proportion of species producing less well known timbers produce fleshy fruit and animal dispersed seeds (Jansen and Zuidema in press).

Silvicultural treatments applied to promote regeneration and growth of commercially desirable species might be made more feasible by increased market demand for what are currently lesser known timbers. While promoting sustained yield of the species being harvested, these silvicultural treatments intentionally result in at least locally substantial modification of pre-intervention stand structure and composition. Compromises between sustained yield and the more all-encompassing goal of sustainable forest management need to be informed by research on the direct and indirect impacts of increased timber harvesting (Fredericksen and others 1999).
helicopter yarding, and to a lesser extent other aerial yarding techniques, may place many of these unlogged patches in jeopardy.

**Function:** Demographic processes (such as survivorship, fertility, and recruitment) and growth rates are two key functional attributes of the species component of biodiversity that logging affects. Populations of many organisms are susceptible to large fluctuations after logging due to both the direct impacts on forest conditions (for example microclimate and fragmentation) and the indirect impacts of hunting, fire, and forest conversion. The proliferation of disturbance-adapted taxa in logged-over forests, some species of which are not native or were not previously common in the area, can have large but as yet little studied impacts on the resident flora and fauna.

**Genetic impacts (see Appendix VII)**

The genetic component of biodiversity is likely to be the most sensitive of all components to logging because of reductions in effective population size and interruptions in gene flow. At present, however, little is known about the genetic structure of any tropical organisms, even commercially valuable timber trees (Ledig 1992). Furthermore, the techniques required for assessing the genetic structure of populations are sophisticated and expensive. Except in a few cases, concerns about dysgenic selection, genetic drift, and other genetic problems are based on controversial theory that is rapidly developing as evidence accumulates.

**Structure:** Logging affects the structural attribute of the genetic component of biodiversity by reducing effective population sizes and heterozygosity. Effective population sizes of both commercial and non-commercial species are reduced by harvesting, other silvicultural treatments, forest fragmentation, weed proliferation, and wildfires. There are also good reasons to be concerned about the effects of logging and stand improvement treatments on dioecious species and in small forest management units in which population sizes of all species are correspondingly small. Allelic frequencies of commercial species change after removal of a large proportion of healthy reproductive adults. For species with high densities of advanced regeneration, genetic structure of their populations are unlikely to change dramatically after selective harvesting, unless collateral damage is severe. Timber stand improvement treatments also may affect the genetic structure of species targeted for removal (such as woody vines) as well as their associates, but these impacts have apparently not been studied. Given the high proportion of vines and other plants that resprout after cutting (=coppice), large impacts on genetic structure are unlikely.

**Composition:** The fact that most species are rare in tropical forests implies that allelic diversity will decrease with increasingly intensive management interventions. Unregulated harvesting of all merchantable individuals of commercial species, for example, has immediate impacts on allelic frequencies that continue to change due to decreased effective population sizes. Deleterious recessive genes may become more apparent due to dysgenic selection and heterozygosity may decline due to the "bottleneck" effect in the small, isolated populations that result from harvesting, forest fragmentation, and other direct and indirect impacts of forestry activities (Styles and Khosla 1976, Murawski and others 1994a and b, but see Newton and others 1996).

**Function:** Logging may affect the functional attribute of the genetic component of biodiversity by interrupting gene flow, which in turn influences outbreeding rates. Decreased effective population sizes coupled with losses of pollinators and seed dispersal agents can result in reduced gene flow and inbreeding depression in populations of both commercial and non-commercial species. Especially vulnerable are populations represented by scattered mature
individuals and very few juveniles (for example, many "long-lived pioneers" such as the mahoganies). Given the high proportion of tropical tree species that are dioecious or obligate outcrossers, only very severe reductions in effective population size are likely to have much effect on gene flow (Ghazoul and others 1998).
Overview of Biodiversity Conservation in Relation to Logging and other Silvicultural Treatments

Synthesis

A way of graphically displaying the impacts of logging on tropical forests (Figure 4) based upon the various components and attributes of biodiversity as described above was developed. Along the vertical axis of our framework the five components of biodiversity were arrayed in the order of increasing susceptibility to logging impacts (landscape, ecosystem, community, population/species, and genetic). The horizontal axis arrays a variety of approaches to silviculture in order of increasing intensity.

The impact of these silvicultural approaches on each biodiversity component using three categories were assessed. First, each biodiversity component was scored as "mostly conserved" for cases in which their attributes are expected to usually stay within their natural range of variation. Second, biodiversity components were scored as "affected" for cases in which their attributes are expected to frequently fall outside their natural range of variation. Finally, biodiversity components were scored as "mostly lost" for cases in which their

Figure 4  Expected effects of a range of forest uses on the components of biodiversity

Legend:
NTPF = Non-timber forest products
RIL = Reduced-impact logging
Reserves = Protected areas within logged units
Refinement = Silvicultural treatments such as liberation of future crop trees from competition, which can substantially change the physiognomy, composition, and trophic structure of forest stands which are applied to increase volume increments and relative densities of commercial timber species
Enrichment planting = Increasing the stocking of commercial species by planting seedlings (or seeds) in logging gaps or along cleared lines
CL = Conventional logging
attributes are expected to almost always fall outside their natural range of variation.

The scoring process used for Figure 4 was admittedly subjective; the scores were based on a reading of the literature and the authors’ experience. It is fully recognized that particular situations might warrant different scores, and it is emphasized that the purpose is to illustrate what is believed to be a useful analytical process rather than to obtain perfectly accurate scores. It is also emphasized that this framework serves to suggest hypotheses that seem to be important research topics for conservation biologists from a number of different disciplines.

The responses summarized on Figure 4 actually represent a multitude of impacts from a diversity of silvicultural approaches applied with varying degrees of concern for biodiversity over a large range of scales. Additionally, Figure 4 addresses neither the issue of profitability of different forest uses nor issues related to land-use capability, biodiversity value, or the capabilities and desires of local stakeholders. At least two other dimensions of this multi-dimensional topic are captured in Figure 5, on which it is indicated that every forest activity can be carried out over a range of intensities. Correspondingly, each forest-use activity generates timber volumes and financial profits that also range widely (Figure 6). Recognizing that forest use practices vary over time with, for example, market fluctuations and political change, and that biodiversity impacts are a function of a multitude of interacting factors operating at different temporal and spatial scales, it is hoped that Figures 4–6 present an accurate “snapshot” of the relationship between biodiversity conservation and forest management.

The impacts summarized in Figure 4 and the ranks and ranges of impacts and profits on Figure 5 are based on a combination of literature reviews and the authors’ subjective estimations. For example, under the range of stocking levels,
Figure 6 Generalized biodiversity impacts resulting from different approaches to forest management for timber. (CL = conventional logging; RIL = reduced-impact logging; refinement = any of a variety of silvicultural treatments applied to increase rates of timber volume increment; and, reserves = protected areas within logged units)

1) Tropical forests are often assumed to have developed their current structure, composition, and functional properties under a disturbance regime characterized by small, localized, and natural perturbations. The importance of unrecorded widespread cataclysmic anthropogenic and natural disturbances, especially those of past centuries, is seldom recognized.

2) It is often assumed that regenerating trees harvested for timber is simply a matter of protecting advanced regeneration (for example, seedlings, saplings, and poles) and providing small canopy openings in which they can mature.

3) Many people believe that devolution of authority over forests to indigenous groups and other rural communities will enhance efforts towards sustainable management.
Biodiversity Conservation in the Context of Tropical Forest Management

While these assumptions hold true in some tropical forests, many of the trees currently being harvested actually regenerated under conditions very different than those of today or those that will be created by harvesting alone. The most familiar example is provided by the neotropical and African mahoganies (Meliaceae spp); these trees and many others like them are the heritage of slash-and-burn agriculture, hurricanes, river meanders and/or catastrophic fires of centuries past. The problem confronting silviculturalists managing for such light-demanding species is that promoting regeneration requires intensive stand manipulations which are expensive and have great impacts on the biodiversity present immediately prior to intervention (Lugo 1999).

Despite accumulating evidence of substantial and long-term pre-historical and more recent impacts of humans on what were formerly considered “natural” ecosystems, it is important to recognize that the current rates, spatial scales, and intensities of human-induced perturbations often far exceed the resilience of tropical forests. Forest recovery after disturbances such as mechanical clearing for pastures, for example, is known to be extremely slow, especially if the pastures are large or intensively used. Natural regeneration in such areas can be accelerated through application of silvicultural treatments, but full recovery of diversity, from the genetic to the landscape level, will require centuries, if it occurs at all. For example, although leaf biomass may recover quickly, in abandoned agricultural clearings coarse woody debris accumulates at rates measured in centuries. Research progress on methods for afforestation and reforestation should not beguile us into believing that, once destroyed, tropical forests can be recreated.

The cause of social equity may be well served by devolution of management responsibility to local communities, but the biodiversity benefits are less secure. Some communities, such as the municipalities in Bolivia that have recently received control over substantial forest areas, may be indifferent about conservation (Kaimowitz and others 1998). In other cases, such as some communities in Nepal, devolution of forest control has had substantial conservation benefits. Obviously the process of devolution deserves scrutiny lest some turn out to be worse forest stewards than the logging companies and national governmental agencies they replace. Empirical evidence suggests that community behavior towards forests depends on the forest richness or forest poorness of their particular situation, but many other factors are likely involved and deserve investigation.

Conclusions

All consumptive use affects some component or attribute of biodiversity, commonly affecting not only the target resource but other factors as well (Redford and Richter 1999). The population/species component is most commonly understood to be affected by silvicultural activities although effects, some of which are subtle or cumulative, are undoubtedly often missed even in this comparatively well studied component. Of increasing importance is an understanding of how the community and ecosystem components have been (Runnels 1995) and are being affected by logging and other silvicultural activities (Noss and Cooperrider 1994, Vitousek and others 1997, Fimbel and others, in press).

Recognizing that all significant interventions in natural forests have biodiversity impacts, all silvicultural decisions necessarily represent compromises. Management for some goods or services necessarily involves management against some others. What are “weeds” to timber stand managers are food sources, rare species, carbon stores, or inter-crown pathways for other human and non-human stakeholders. The biodiversity compromises involved in deciding whether to cut vines, retain seed trees, or enhance seedling establishment by carrying out controlled burns should be informed by
research. Unfortunately, very little is known about how tropical forests can be managed in the most biodiversity friendly manner. Researchers have instead focussed on enumerating the deleterious environmental impacts of uncontrolled logging by untrained and unsupervised crews. To inform decisions about tropical forest management and to assure that biodiversity is protected to the maximum extent, more research is needed on how to maintain diversity in forests selected for logging. The large and rapidly growing body of literature on ecosystem management in both north and south temperate forests (such as Kohm and Franklin 1997, Lindenmayer 1999) should provide inspiration and starting points for tropical researchers intent on solving biodiversity related problems associated with forest management.

The primary conclusions to derive from these analyses are that:

- Different intensities and spatial patterns of timber harvesting, along with other silvicultural treatments, result in different effects on the different components of biodiversity.
- Some components and attributes of biodiversity are more sensitive than others to forest management activities.
- Only extremely limited use will protect all components (that is, large protected areas are essential for biodiversity conservation).

The capacity to mitigate the deleterious environmental impacts of logging and other silvicultural treatments should not be construed as constituting unilateral support for sustainable forest management as a conservation strategy. Such an endorsement is unwarranted given widespread illegal logging in the tropics, widespread frontier logging and logging of areas of high priority for biodiversity protection, the persistence of poor logging practices despite substantial efforts in research and training, and the general slow rate at which most loggers are transforming themselves from timber exploiters into forest managers. Nevertheless, even the most harshly treated forests maintain more biodiversity than tree farms for pulpwood, oil palm plantations, maize fields, or cattle pastures. Furthermore, logging is often the environmentally least damaging of land uses that are also financially viable (Pearce and others 1999). Given these conclusions, effective mechanisms for financing forest protection and environmentally sound forest management are needed. It might also prove useful to evaluate critically the limits to the assumption that well managed forests are less likely to be converted to other land uses than forests that are logged without apparent concern for sustainability.

By focusing on the deleterious environmental impacts of tropical forest management activities, we often lose sight of the fact that from a biodiversity maintenance perspective, natural forest management (that is, maintaining forests as forests) is preferable to virtually all land-use practices other than complete protection. The number of taxa with reportedly inconsistent, neutral, or positive responses to logging is an indication that timber harvesting is not necessarily incompatible with protection of many components and attributes of biodiversity. This conclusion derives even greater support from the fact that nearly all of the studies conducted to date on the biodiversity impacts of logging were carried out after unplanned logging by crews with no training in reduced-impact logging techniques and no incentives to reduce their impacts. As forest management practices improve under market pressure or pressure from land-owners, the deleterious environmental impacts of logging and other silvicultural activities are likely to be substantially reduced. Forests that are carefully managed for timber will not replace protected areas as storehouses of biodiversity, but they can be an integral
component of a conservation strategy that encompasses a larger portion of the landscape than is likely to be set aside for strict protection. In other words, forests managed primarily for timber, if managed properly, will supplement and effectively extend the conservation estate. Finally, it should be recognized that landscape management is consistent with the ecosystem approach emphasized by the Convention on Biological Diversity (CBD).
6 Recommendations

1. Given that all substantial forest interventions affect some component or attribute of biodiversity, the best way to assure biodiversity conservation is through the establishment and protection of large, properly located, and well managed reserves and protected areas.

2. In some parts of the world and some types of forest, there should be no logging at all.

3. Existing legislation and regulations should be enforced. Where they do not exist, or are weak they should be created or updated to reflect current conservation and sustainable resource management concerns.

4. Within forests used for logging and other silvicultural activities, biodiversity conservation is enhanced by setting aside a portion of the area for complete protection. Optimally, these reserves should be large, shaped so as to minimize edge effects, cover representative areas of all the ecosystem types present, and include features of special concern for biodiversity maintenance such as water courses, rock outcrops, and salt licks.

5. Where logging is to be carried out, reduced-impact logging guidelines (Dykstra and Heinrich 1996) should be fully implemented by well trained crews.

6. Researchers need to determine the financial and environmental costs and benefits of the components of reduced-impact logging and various other pre- and post-logging silvicultural activities. Particular attention should be paid to plant and animal species with attributes that render them susceptible to logging-induced extirpation (Martini and others 1994, Pinard and others 1999). Regional guidelines for certification need to be developed and regularly updated to reflect increasing knowledge about biodiversity so as to assure that certification efforts realize their conservation potential.

7. Measures to protect biodiversity should be included in all forest management plans. Particular attention should be paid to the retention of nest and den trees as well as species of great importance to biodiversity conservation.

8. Wildfires need to be controlled. Successful fire control programs will need to include public awareness building as well as implementation of existing technologies and methods. Research is needed for developing cost-effective fire control measures with minimal biodiversity impacts.

9. While continuing to promote and develop community-based wildlife management programs, hunting by loggers and market hunting of slow reproducing species should be prohibited. Enforcing existing laws will in many cases confer substantial protection to species threatened by over-hunting.
10. Training is needed for researchers who will develop biodiversity-sensitive silvicultural methods as well as cost-effective methods for monitoring the environmental impacts of silviculture. Training is also needed for the field crews that are responsible for implementing the recommended practices. Perhaps the most critical shortage is of forestry/conservation "research practitioners" who understand both the broad scientific and the more specific technical aspects of tropical forest management.

11. The biodiversity impacts of devolution, plantation conversion, certification, forest-based carbon offsets, and global trends in timber markets need to be monitored.

12. Linkages to the policy arena must be pursued. Ecosystems provide important services which typically are not internalized into the decision making process. Legislation and their regulations are an integral part of this process and should be reviewed, updated and enforced.
Appendix I —
Impacts of Logging (and other silviculture treatments where noted) on the Landscape Component of Biodiversity in Tropical Forests
Impacts of logging (and other silviculture treatments where noted) on the landscape component of biodiversity in tropical forests

<table>
<thead>
<tr>
<th>Category</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation, Soil &amp; Water</td>
<td>Fragmentation / roads → greater access to humans → increased deforestation (Kaimowitz &amp; Angelsen 1998)</td>
<td>50% forest area logged → species lost (Eastern Amazon: Nepstad et al. 1992)</td>
<td>Increased human colonization and fire &amp; decreased migration of animals and gene flow in plants (if pollinators are restricted to fragments)</td>
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<td></td>
<td>Increased spatial heterogeneity of habitats after 140 yrs of agroforestry, logging, &amp; charcoal production followed by 50 yrs regeneration (Puerto Rico: García-Montiel &amp; Scatena 1994)</td>
<td>Increase in mixed forest (w/ valuable timber spp) &amp; decrease of <em>Cynometra</em> forest patches after arboricide treatment from 1951-90 (Uganda: Plumpre 1996)</td>
<td>200 yrs necessary for recovery of spp composi- tion following logging damage (Malaysia, forest gap model: Kurpick et al. 1997)</td>
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<td></td>
<td>Complex mosaic of forest types &amp; disturbance w/ 85% swamp forest &amp; only 15% lowland forest escaping logging disturbance (W. Kalimantan, 8 yrs post-logging: Cannon et al. 1994)</td>
<td>Maintenance &amp; recovery of species composition in logged patches depends on landscape context (Liu &amp; Ashton 1999)</td>
<td>Impoundments due to undersized or failed culverts &amp; bridge abutments kill trees and may influence fauna (F.E. Putz, pers. obs.)</td>
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<td></td>
<td>Patches (1-100 ha) of unlogged forests in heavily logged area (500-1000 ha, 80-120 m³/ha) on steep slopes (&gt;35°) and in rocky areas (Sabah, Malaysia: Pinard &amp; Putz 1996)</td>
<td></td>
<td>Increased likelihood of large scale accidental fires due to canopy opening and roads (Amazonia: Cochrane &amp; Schulze 1999) or fuel accumulation from fire protection (Miombo woodlands, Africa: Chidumayo 1988)</td>
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<td></td>
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<td>Harvest gaps warmer than natural treefall gaps (Amazonia: Vitt et al. 1998)</td>
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<tr>
<th>Primates</th>
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<tr>
<td>Non-rode mammal</td>
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<td>Rodents &amp; Marsupials</td>
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<td>Bats</td>
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<td>Birds</td>
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<td>Herps</td>
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<td>Insects</td>
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| Soil organisms | |
|----------------|
Appendix II — Impacts of Logging (and other silviculture treatments where noted) on the Ecosystem Component of Biodiversity in Tropical Forests
Impacts of logging (and other silviculture treatments where noted) on the ecosystem component of biodiversity in tropical forests

<table>
<thead>
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<th>Category</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
</tr>
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<tbody>
<tr>
<td></td>
<td>➢ Reduction in large trees with hollows &amp; holes important for wildlife (Australia: Gibbons &amp; Lindenmayer 1996)</td>
<td>➢ Higher richness (trees &gt; 20 cm dbh) in logged vs. unlogged, but when equal numbers of trees sampled richness was equal (W. Kalimantan, 8 yrs post-logging: Cannon et al. 1998)</td>
<td>➢ Decreased evapotranspiration &amp; infiltration; increased surface flow ➔ nutrient losses (Brouwer 1996, Poels 1987, Bruijnzoon 1992), but nutrient losses don’t restrict productivity (Proctor 1992)</td>
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<td></td>
<td>➢ 2/3 basal area, 1/5 no of emergents (&gt;30 m), &amp; 1/3 commercial volume of unlogged forest (Venezuela, 19 yrs post-harvest of 10 trees/ha: Kammesheidt 1998)</td>
<td>➢ Tree spp distributions more related to geography than whether a patch was logged; richness higher in west of reserve and in logged patches (Uganda, 60 yrs post 'sustainable logging': Plumptre 1996)</td>
<td>➢ Soil bulk density 6 yrs post-logging 60% greater than in undisturbed forest (Malaysia: Malm &amp; Grip 1990); skid trails estimated to require &gt; 50 yrs to recover hydraulic conductivity (Malaysia: Kamaruzaman &amp; Kamaruzaman 1996)</td>
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<td></td>
<td>➢ Forest surface fires kill 40-95% of all trees w/ dbh &gt; 10 cm (Amazon: Nepstad pers obs)</td>
<td>➢ 36% biomass loss from increased tree mortality (Amazonia, 100 m from fragment edge, 10-17 yrs post-fragmentation: Laurance et al. 1997)</td>
<td>➢ 100-170 M tons of carbon/yr emitted from 1981-1996 (Asia: Houghton &amp; Hackler 1999)</td>
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<td></td>
<td>➢ 5-40% of forest floor disturbed by roads, skid trails &amp; tractor tracks (Cannon et al. 1994 Johns et al. 1996, Pinard &amp; Putz 1996, Sist et al. 1998a, Holmes et al. 1999)</td>
<td>➢ Epiphytic &amp; strangler Ficus spp suffer high rates of damage (Malaysia, post-logging: Lambert 1991)</td>
<td>➢ 0.5% of total global carbon emissions (Brazilian Amazon, 1996-97: Nepstad et al. 1999)</td>
</tr>
<tr>
<td></td>
<td>➢ Increase in organic fuels (56 to 180 tons/ha) on forest floor (eastern Amazon, post-logging: Uhl &amp; Kaufman 1990, Uhl &amp; Vieira 1989)</td>
<td>➢ Reduced densities of timber spp &amp; shift in dominance &amp; age structure of canopy spp (Puerto Rico, post 140 yrs agroforestry, logging, charcoal production + 50 yrs regeneration: Garcia-Montiel &amp; Scatena 1994)</td>
<td>➢ Post CL, stream suspended solids and turbidity were 12x &amp; 9x &gt; control and levels persisted at least 5 yrs; RIL levels were 2x &gt; control but recovered after 2 yrs (Malaysia: Zulkifli &amp; Anhar 1994)</td>
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<td></td>
<td>➢ Soil disturbance main factor limiting seedling regeneration &amp; establishment (Kalimantan: Gardingen et al. 1998)</td>
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Ecosystem Attributes

<table>
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<tr>
<th>Category</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
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<tbody>
<tr>
<td></td>
<td>➢ Reduction in large trees with hollows &amp; holes important for wildlife (Australia: Gibbons &amp; Lindenmayer 1996)</td>
<td>➢ Higher richness (trees &gt; 20 cm dbh) in logged vs. unlogged, but when equal numbers of trees sampled richness was equal (W. Kalimantan, 8 yrs post-logging: Cannon et al. 1998)</td>
<td>➢ Decreased evapotranspiration &amp; infiltration; increased surface flow ➔ nutrient losses (Brouwer 1996, Poels 1987, Bruijnzoon 1992), but nutrient losses don’t restrict productivity (Proctor 1992)</td>
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<td></td>
<td>➢ 2/3 basal area, 1/5 no of emergents (&gt;30 m), &amp; 1/3 commercial volume of unlogged forest (Venezuela, 19 yrs post-harvest of 10 trees/ha: Kammesheidt 1998)</td>
<td>➢ Tree spp distributions more related to geography than whether a patch was logged; richness higher in west of reserve and in logged patches (Uganda, 60 yrs post 'sustainable logging': Plumptre 1996)</td>
<td>➢ Soil bulk density 6 yrs post-logging 60% greater than in undisturbed forest (Malaysia: Malm &amp; Grip 1990); skid trails estimated to require &gt; 50 yrs to recover hydraulic conductivity (Malaysia: Kamaruzaman &amp; Kamaruzaman 1996)</td>
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<td></td>
<td>➢ Forest surface fires kill 40-95% of all trees w/ dbh &gt; 10 cm (Amazon: Nepstad pers obs)</td>
<td>➢ 36% biomass loss from increased tree mortality (Amazonia, 100 m from fragment edge, 10-17 yrs post-fragmentation: Laurance et al. 1997)</td>
<td>➢ 100-170 M tons of carbon/yr emitted from 1981-1996 (Asia: Houghton &amp; Hackler 1999)</td>
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<td>➢ 5-40% of forest floor disturbed by roads, skid trails &amp; tractor tracks (Cannon et al. 1994 Johns et al. 1996, Pinard &amp; Putz 1996, Sist et al. 1998a, Holmes et al. 1999)</td>
<td>➢ Epiphytic &amp; strangler Ficus spp suffer high rates of damage (Malaysia, post-logging: Lambert 1991)</td>
<td>➢ 0.5% of total global carbon emissions (Brazilian Amazon, 1996-97: Nepstad et al. 1999)</td>
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<td></td>
<td>➢ Increase in organic fuels (56 to 180 tons/ha) on forest floor (eastern Amazon, post-logging: Uhl &amp; Kaufman 1990, Uhl &amp; Vieira 1989)</td>
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<tr>
<td>Vegetation, Soil &amp; Water</td>
<td>Vegetation</td>
<td>Soil properties (i.e. BD, total porosity, sat hydr conductivity, &amp; resistance to penetration) required 12-52 yrs (skidtrails longest) to recover (Malaysia: Kamaruzaman &amp; Kamaruzaman 1996)</td>
<td>Woody vines increased w/in 100 m of edges but didn’t compensate for lost biomass from increased tree mortality Amazonia forest fragments, 10-17 yrs post fragmentation: Laurance et al. 1997)</td>
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<td>Limiting harvest gaps to &lt; 650 m² (&lt; 2 trees/gap) enhanced dipterocarp regen &amp; inhibited pioneer regen (Kalimantan: Gardingen et al. 1998)</td>
<td>200 yrs necessary for recovery of spp comp following logging damages (Malaysia, forest gap model: Kurpick et al. 1997)</td>
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<td>Harvest gaps much larger than natural treefall gaps &amp; thus have less litter and shade &amp; more intense light (Amazonia: Vitt et al. 1998)</td>
<td>Runoff as % precipitation ranged from 58% - 94% in plantations converted from forest depending on method (Malaysia: Malmer 1992)</td>
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<td>Tree basal area and commercial volumes recovered within 40 yrs post logging w/ MUS (Malaysia: Manokaran 1998)</td>
<td>Road &amp; skid trail impoundments may increase methane emissions (F.E. Putz, pers. obs.)</td>
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<td></td>
<td>Primates</td>
<td></td>
<td>Sediment loads have negative effects on downstream aquatic life (Australia: Campbell &amp; Doeg 1989)</td>
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<td></td>
<td></td>
<td>Long distance seed dispersal and species movements to/from different habitat patches likely impeded (no specific data)</td>
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<td></td>
<td>Non-rodent mammals</td>
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<td></td>
<td>Rodents &amp; Marsupials</td>
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<td></td>
<td>Bats</td>
<td>Bird spp decline w/ increasing disturbance (Lawton et al. 1998)</td>
<td>Species use of different habitat patches &amp; pollination &amp; seed dispersal rates likely altered (no specific data)</td>
</tr>
<tr>
<td></td>
<td>Birds</td>
<td></td>
<td>Species use of different habitat patches likely altered; depending on spp., pollination, predation, seed dispersal rates likely altered; (no specific data)</td>
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<td>Herps</td>
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<td>Community Attributes</td>
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<tr>
<td><strong>Structural Changes</strong></td>
<td><strong>Compositional Changes</strong></td>
<td><strong>Functional Changes</strong></td>
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<tr>
<td><strong>Insects</strong></td>
<td>Arthropod diversity decreases when structural complexity decreases (Holloway et al. 1992; Gardner et al. 1995) and/or b/c of changes in understory, litter, &amp; soil microclimate (Blau 1980, Kremen 1992, Spitzer et al. 1997, Camilo &amp; Zou in press)</td>
<td>Various ecosystem processes will be relatively unaffected (Vitousek 1990)</td>
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<tr>
<td><strong>Soil organisms</strong></td>
<td>25%-75% reduction in soil mycorrhizae (Malaysia, post-logging: Alexander et al. 1992)</td>
<td>Termite richness lower in clear-cuts, but higher in regenerating plots than 1° forest (Cameroon), but overall richness was similar in 1°, 3 yr and 17 yr secondary sites (Malaysia: Eggleton et al. 1995); changes in termite comp may be delayed after disturbance (Eggleton et al. 1997)</td>
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<td>Nematodes declined along disturbance gradient &amp; were 40% lower in clearcut vs. 1° forest (Bloemers et al. 1997)</td>
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<td>Soil fungi increased from 17 spp in 7-yr regrowth to 27 in 16-yr regrowth; soil microbial pop'n's positively correlated w/ tree density &amp; basal area (India: Arunachalam et al. 1997)</td>
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<td>Ectomycorrhizal diversity greater under open &amp; regenerating canopies than in undisturbed forest (Kalimantan, 9 months post clear-cut or partial logging, trees &gt; 50 cm dbh: Ingleby et al. 1998)</td>
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<td>Termite populations increase after conventional logging more than after RIL (Sabah, Malaysia, 80-120 m3/ha harvested: F.E. Putz, pers. obs.)</td>
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</table>

(MUS = Malayan Uniform System, a monocyclic approach to timber stand management)
Appendix III — 
Impacts of Logging (and other silviculture treatments where noted) on the Community Component of Biodiversity in Tropical Forests
## Impacts of logging (and other silviculture treatments where noted) on the community component of biodiversity in tropical forests

<table>
<thead>
<tr>
<th>Category</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
</tr>
</thead>
<tbody>
<tr>
<td>Geological</td>
<td>Greater density of understory vegetation 5-6 yrs post harvest (Venezuela: Mason 1996)</td>
<td>Assim t ase d s e e a a l r r i n g t i m e r d a n i n t e l t t a n t r n p s a d m n e a t e e c p e t i v l y e x c l u d e o t h e r p l a n t s (Uganda: Struhsaker 1997)</td>
<td>High degree of outcrossing and low density of mature individuals (neotropics: Murawski 1995, Stacy et al. 1996)</td>
</tr>
<tr>
<td>Geological</td>
<td>Remaining large trees inadequate to sustain future harvests (Philippines, 15 yrs post-SLS: Tombec &amp; Mendoza 1998)</td>
<td>High dominance of few spp in heavily disturbed habitats (e.g., roads) (Costa Rica: Guariguata &amp; Dupuy 1997)</td>
<td>Seed production lower in several timber species (Uganda; Plumptre 1996)</td>
</tr>
<tr>
<td>Geological</td>
<td>Reduced leaf biomass but increased leaf quality; reduced log distribution, regenerating stems, and overall tree size (Madagascar, RL, &lt; 10% forest: Ganzhorn 1995, Ganzhorn et al. 1990)</td>
<td>Vine spp. richness and densities recover to pre-MUS logging and treatment by 40 yrs (Malaysia: Gardette 1998)</td>
<td>Wind-dispersed seeds fall in higher numbers in gaps than in understory (Augspurger &amp; Franson 1988, Loiselle et al. 1996) ➔ higher colonization levels from seed for wind-dispersed spp.</td>
</tr>
<tr>
<td>Geological</td>
<td>Palm spp. richness in logged and MUS treated forest than in 1st forest (Malaysia: Nur Supardi et al. 1998)</td>
<td>Lichen diversity increased, moss diversity decreased 40 yrs post-logging and silvicultural treatment (Malaysia, MUS: Wolfe et al. 1998)</td>
<td>Reduced dispersal of seeds carried by large-bodied, gap-shy frugivores (Gorchov et al. 1993, Forget &amp; Sabatier 1997) or that are scatterhoarded by caviomorph rodents (Forget &amp; Milleron 1991; Forget 1990, 1993, 1994; Asquith et al. 1999) esp. if hunting present</td>
</tr>
</tbody>
</table>
### Vegetation

- Fewer large diameter vines 40 yrs post MUS than in 1st forest (Malaysia: Gardette 1998)
- Denning trees less available for arboreal rodents & marsupials after selective logging (Australia: Laurance & Laurance 1996; in Ochoa & Soriano in press)
- Repeated selective logging reduces abundance of large trees and shifts composition to earlier successional species (Australia: Horne & Hickey 1991)

- Mortality rates 4x higher for spp w/ dbh > 40 cm (S. India, 10-15 yrs post- selective logging: Pelissier et al. 1998)
- Tree mortality increased after liberation/refinement treatments (Costa Rica: Finegan & Camacho 1999)
- Vine cutting promotes tree growth & reduces logging damage but removes important intercrown pathways for wildlife (reviewed in Putz 1991)

### Primates

- Increased infant mortality for various spp. immediately after logging, but some spp recovered 6-12 yrs post logging (Malaysia: Johns & Johns 1995)
- Omnivore densities (1-25 trees harvested/ha, 1-50 yrs post-logging, Africa, Asia, Amazon) – 8 spp decreased, 13 spp increased, 5 spp showed no change; several spp showed opposite results in different studies and/or different sites (reviewed in Plumptre & Grieser Johns in press)
- Folivore/frugivore densities (as above) – 6 spp decreased, 5 spp increased, 3 spp showed no change; as above, several spp showed opposite responses in different studies/sites (Plumptre & Grieser Johns in press); hunting affects folivores more than logging itself (Africa: Oates et al. 1996)
- Frugivore/folivore densities (as above) – 3 spp decreased, 2 increased, 4 spp showed no change; again, several spp responded differently in different sites/studies (Uganda: Plumptre & Grieser Johns in press)
- Lemurs in dry forest (< 10% affected by logging) – increased sightings of all spp attributed to increased fruit & higher foliage quality on remaining trees (Madagascar: Ganzhorn 1995)
- Monkeys in Zaire - 4 spp preferred secondary forest: 3 others preferred primary forest (Zaire: Thomas 1991)
- Folivores - hunting has greater negative impact than logging (Africa: Oates et al. 1996)
- Primate densities decreased in response to logging & MUS (Malaysia: Laidlaw 1998)
- Herbivory & frugivory – enhanced by low intensity (<10% affected) logging disturbance (Madagascar: Ganzhorn 1995); data lacking for higher intensities, but presumed as variable as changes in spp densities
- Seed dispersal & predation – no specific data; probably varies with animal densities
- Vine cutting and liberation thinning increased locomotory costs and susceptibility to predators for non-volant arboreal animals (reviewed in Putz et al. 2000)
- Predation of small primates by raptors may increase in response to vine removal (Suriname: S. Boinski, pers comm.)
<table>
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<tr>
<th>Category</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
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<tbody>
<tr>
<td>Non-rodent</td>
<td>Browsers/grazers (e.g., Asian elephant, African buffalo, gaur, banteng, &amp;</td>
<td>If no hunting, increased herbivory; if</td>
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<td>mammals</td>
<td>sambar deer) increase (if no hunting) b/c of increased food quality</td>
<td>hunting decreased herbivory</td>
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<td>Oates et al. 1990, Waterman &amp; Kool 1994 &amp; quantity (Johns 1992); the only</td>
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<td>sensitive spp in this guild was the giant forest hog (Hylochoerus</td>
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<td>meintzehageni) (reviewed in Davies et al. in press)</td>
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<td>Carnivores – mixed responses &amp; few data; spp with restricted diets</td>
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<td>most sensitive (reviewed in Davies et al. in press)</td>
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<td></td>
<td>Species favored by hunters (e.g., duikers, pigs, peccaries), for special</td>
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<td>trade (e.g., elephants) and roadside feeders (e.g., Synceros caffer, Bos</td>
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<td>spp) decline (Davies et al. in press)</td>
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<td>Large, slow (moving &amp; reproducing) species decline (hunting a</td>
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<td>confounding factor) (Lahm 1994, Grieser Johns 1997)</td>
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<td></td>
<td>Species dependent on fruits/seeds of timber species (e.g. Sus barbatus,</td>
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<td>Caldecott 1991) on closed canopy (e.g., Muntiacus atherodes, Heydon</td>
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<td></td>
<td>1994; Giao et al. 1998), or with limited dietary flexibility due to</td>
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<td></td>
<td>energetic requirements (e.g., Tragulus spp. Heydon &amp; Bulloh 1997) decline.</td>
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<tr>
<td>Rodents and</td>
<td>Arbororeal marsupials - 1 sp declined &amp; 4 spp unaffected after 8-10</td>
<td>Increased seed predation rates (esp if</td>
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<tr>
<td>Marsupials</td>
<td>trees (50-55 m²) harvested/ha (Australia: Laurance &amp; Laurance 1996)</td>
<td>hunting) (e.g., Terborgh &amp; Wright 1994)</td>
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<tr>
<td></td>
<td>Small mammal populations most sensitive - according to models: (a)</td>
<td>Decreased predation on large seeds if large</td>
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<td></td>
<td>Sciuridae, Echimyidae &amp; other canopy dependent frugivores/granivores;</td>
<td>seed predators extirpated (Putz et al. 1990)</td>
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<td>and (b) spp w/ low RA and restricted geog range (Ochoa et al. 1993,</td>
<td>Increase in Muridae may increase</td>
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<td>Small mammal populations least sensitive - according to models: Didelphidae &amp; Muridae; semi-arboreal omnivores/predators increase (e.g., Didelphis spp. &amp; Philander opossum; see above refs)</td>
<td>(Hammond &amp; Thomas in press)</td>
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<td></td>
<td>Rodent spp richness increased soon after logging (Uganda, 9 trees/ha,</td>
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<td>62% canopy open: Isabirye-Basuta &amp; Kasenene 1987, Muganga 1989), but</td>
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<td>decreased by 16 yrs post (Lwanga 1994); Rodent spp. richness increased,</td>
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<td>2 dominant spp. decreased in abundance but increased in body mass after</td>
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<td>logging (China: Wu et al. 1996)</td>
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</table>
**Bats**

- Pioneer plant dispersers (e.g., *Carollia, Artibeus* & *Stenodermus*) increase.
- Phyllostominae (insectivores & vert predators) - decline (e.g., *Artibeus obscurus, Vampyressa bidens, Phyllostomus elongatus, Tonatia saurophila*) or disappear (e.g., *Vampyrum spectrum, Tonatia syvlica, Chrotopherus auritus*, & *Peropterix koppleri, Emballonuridae*) (Fenton et al. 1992, Brosset et al. 1996, Utrera 1996, review in Soriano & Ochoa in press)
- Reduced guild complexity & local extinctions after logging (5.8 m³/ha, trees > 40 cm dbh); effects exacerbated by enrichment strips (French Guiana: Brosset et al. 1996; similar findings in Venezuela: Ochoa in press; reviewed in Soriano & Ochoa in press)
- Open-forest, fast-flying spp w/ long, narrow wings do best b/c of increased gap area; dominated 6-yr post site w/ 6-7 trees/ha harvested (Australia: Crome & Richards 1988)
- Spp preferring medium to dense understory disappeared after brushing indicating high sensitivity to changes in forest structure (Miller in press)

**Increased dispersal of pioneer plant spp; decreased predation**

**Birds**

- Terrestrial & sallying insectivores (e.g., Tyrannidae) & other spp depending on forest interior (e.g., those in mixed-spp flocks) decrease w/ logging (Johns 1989a, 1991a, 1992a; Lambert et al. 1992; Thiollay 1992; Zakaria 1994; Bennet & Dahaban 1995; Fanshawe 1995; Grieser Johns 1996; Mason 1996; Owiunji & Plumptre 1998; reviews in Mason & Thiollay in press, Zakaria & Francis in press, & Plumptre in press)

**Herbivory & frugivory - reduced for fig-eating spp b/c of lower post-harvest density of Ficus spp (Lambert 1991)**

**Predation - expected to decrease b/c of reduced insect prey abundance (b/c of hotter/drier conditions) (Mason 1995)**

**Birds**

- Understory spp. - nectarivorous increased; insectivorous decreased; latter trend worsened by enrichment planting (Venezuela: Mason 1996)

**Seed dispersal - reduced b/c of avoidance of large openings in forest canopy (Stouffer & Bierregaard 1995a,b)**

**Colonization rates -expected to increase when refuges provided w/in logged areas (Johns 1996)**
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- Interior understory spp declined by 37-98%; generalists & spp associated w/ dense second growth, edges & large gaps increased; overall richness and abundance down 27-34% (French Guiana, 1-17 yrs post, 38% undergrowth & 63% canopy loss: Thiollay 1997); (other study: 3 trees/ha, 10 yrs post) - lower diversity b/c of more uniform habitat (Thiollay 1992)

- Flycatchers, wren-warblers, trogons & woodpeckers declined; nectarivores & some frugivores increased (Malaysia, 8 yrs post-logging: Lambert et al. 1992)

- Spp. richness lower than unlogged (Indonesia: Marsden 1998)

- 1/3 spp expected to disappear (Liberia, logged fragment: Kofron & Chapman 1995)

**Herps**

- Heliothermic lizards -- increased density of large lizards & altered comm structure (Amazonian treefall gaps: Vitt et al. 1998)

- No change in reptile community comp or abundance (<10% damage, 2-12 yrs post-logging: Bloxam et al. 1996)

- Lower amphibian diversity (Amazonia, high intensity logging or logging & fragmentation & roads: Pearman 1997, Vitt et al. 1998)

- Spp that live in leaf litter most severely affected especially after high intensity logging (e.g., monocyclic harvests); w/ less intense logging most spp persist in short term, but invasion of 2° forest by large predatory lizards may extirpate small frogs & lizards; some frog spp increase after logging b/c of ponding in skid trail ruts or in stream impoundments (reviewed in Vitt & Caldwell in press)

- Competition & Predation - increases in generalist spp (Vitt et al. 1998)

- Colonization - generalist spp (e.g, *Hyla geographica*) increase (Caldwell 1989)
<table>
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<tr>
<th>Insects</th>
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<tbody>
<tr>
<td>➢ Community comp of Araneae, Hymenoptera, Heteroptera, Homoptera,</td>
<td>➢ Short generation times &amp; high fecundity allow rapid recovery of</td>
</tr>
<tr>
<td>Coleoptera, Orthoptera &amp; Lepidoptera differed in 1° forest, logged</td>
<td>effected populations (temperate zone studies: Huhta 1976,</td>
</tr>
<tr>
<td>➢ Spp. restricted to understory, leaf litter &amp; soil most vulnerable (</td>
<td>➢ Negative cascading effects on insect-mediated processes (e.g.,</td>
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<tr>
<td>reviewed in Ghazoul &amp; Hill in press)</td>
<td>pollination, decomposition, nutrient cycling) are likely</td>
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<tr>
<td>➢ Geometrid moth diversity lowest in abandoned clearcut, highest in</td>
<td>(Ghazoul &amp; Hill in press)</td>
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<td>forest logged w/ MUS (Malaysia: Intachat et al. 1997; Ghazoul &amp; Hill</td>
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<td>in press)</td>
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<td>➢ Lower lepidopteran moth diversity after logging + conversion to</td>
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<td>plantation; Coleopteran dung &amp; carrion beetles less affected (</td>
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<td>Malaysia: Holloway et al. 1992; Ghazoul &amp; Hill in press)</td>
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<tr>
<td>➢ Dung beetle spp. composition in RIL area resembles 1° forest more</td>
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<td>than conventionally logged area (Malaysia: Davies in press)</td>
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<tr>
<td>➢ Butterflies, flying beetles, canopy beetles, canopy ants, leaf</td>
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<td>eater ants, termites and soil nematodes all declined along</td>
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<tr>
<td>disturbance gradient (Lawton et al. 1998)</td>
<td></td>
</tr>
<tr>
<td>➢ Butterfly richness, abundance &amp; evenness all decreased (</td>
<td></td>
</tr>
<tr>
<td>Indonesia, 5 yrs post-logging: Hill et al. 1995)</td>
<td></td>
</tr>
<tr>
<td>➢ Cassidinae beetle richness greater in logged forest and plantations</td>
<td></td>
</tr>
<tr>
<td>vs. 1° forest and logged forest community composition more like</td>
<td></td>
</tr>
<tr>
<td>plantations (Uganda: Nummelin &amp; Borowiec 1991)</td>
<td></td>
</tr>
<tr>
<td>➢ Opportunistic species increase after invasion of pioneer plants at</td>
<td></td>
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<tr>
<td>expense of closed forest spp (Holloway et al. 1992)</td>
<td></td>
</tr>
<tr>
<td>➢ Ant spp. richness similar in 1° forest and MUS forest 40 yrs after</td>
<td></td>
</tr>
<tr>
<td>treatment (Malaysia: Bolton 1998)</td>
<td></td>
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<tr>
<td>Category</td>
<td>Structure</td>
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<td>------------------</td>
<td>---------------------------------------------------------------------------</td>
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<tr>
<td><strong>Soil organisms</strong></td>
<td>➢ Fruiting body (sporome) density of mycorrhizal fungi higher 40 yrs. After MUS treatment than in 1° forest (Malaysia: Watling et al. 1998)</td>
</tr>
<tr>
<td><strong>Aquatic systems</strong></td>
<td>➢ Loss of instream &amp; riparian zone diversity; decreased abundance of fishes &amp; benthic invertebrates intolerant of (a) high sedimentation, (b) shifts in photosynthesis/respiration ratios, (c) invasive spp (review in Pringle &amp; Benstead in press)</td>
</tr>
</tbody>
</table>
Appendix IV —
Bird Responses (density based on inter-site comparisons) to Logging in Tropical Forests, Organized by Feeding Guild
Bird responses (density based on inter-site comparisons) to logging in tropical forests, organized by feeding guild

<table>
<thead>
<tr>
<th>Site</th>
<th>Budongo, Uganda</th>
<th>Kibale, Uganda</th>
<th>Tekam, Malaysia</th>
<th>Ulu Segama, Sabah</th>
<th>Ulu Segama, Sabah</th>
<th>Imatoca, Venezuela</th>
<th>Piste de St -Elie, French Guiana</th>
</tr>
</thead>
<tbody>
<tr>
<td>Logging intensity</td>
<td>20-80 m³/ha</td>
<td>21 m³/ha</td>
<td>18 trees/ha</td>
<td>79 m³/ha</td>
<td>90 m³/ha</td>
<td>5-8 m³/ha</td>
<td>10 m³/ha</td>
</tr>
<tr>
<td>Recovery (years since harvest)</td>
<td>25-50</td>
<td>25</td>
<td>6-7</td>
<td>3-4</td>
<td>9-10</td>
<td>1-6</td>
<td>1</td>
</tr>
<tr>
<td>Hunting</td>
<td>no</td>
<td>no</td>
<td>unknown</td>
<td>no</td>
<td>no</td>
<td>no</td>
<td>yes</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Guild</th>
<th>Frugivore</th>
<th>Frugivore/insectivore (arboreal)</th>
<th>Frugivore/insectivore (terrestrial)</th>
<th>Frugivore/granivore</th>
<th>Granivore</th>
<th>Granivore/insectivore</th>
<th>Insectivore (sallying)</th>
<th>Insectivore (terrestrial)</th>
<th>Insectivore (arboreal foliage gleaner)</th>
<th>Insectivore (understory foliage gleaner)</th>
<th>Insectivore (bark gleaning)</th>
<th>Insectivore/nectarivore</th>
<th>Predator</th>
<th>Predator/frugivore</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>+</td>
<td>+</td>
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</tbody>
</table>

Notes:

1 Density (calculated by same method within a study, but differed between studies). + = higher following logging, - = lower following logging, nc = no change, na = information not available; P ≥ 0.05. (Adapted from Mason and Thiollay, in press; Plumptre et al., in press; and Zakaria and Francis, in press).
Appendix V —
Impacts of Logging (and other silviculture treatments where noted) on the *Species Component* of Biodiversity in Tropical Forests
### Impacts of logging (and other silviculture treatments where noted) of the species component of biodiversity in tropical forests

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Vegetation</strong></td>
<td>Smaller size classes most affected by felling, skidding, and road construction</td>
<td>High grading large commercial trees caused local extirpation of <em>Caesalpinia echinata</em>, <em>Ceiba pentandra</em>, and <em>Aniba duckei</em>; Amazonia (Gentry &amp; Vasquez 1988, Grieser &amp; Johns 1997)</td>
<td>Reduced seed availability &amp; reduced seed dispersal by rodents of large seeded commercial species after logging (Guyana: Forget &amp; Hammond in press, Hammond &amp; Thomas in press)</td>
</tr>
<tr>
<td></td>
<td>62% seedling &amp; 59% sapling mortality (Malaysia: Borhan et al. 1987)</td>
<td>Harvesting figs promotes their regeneration &amp; may not have wildlife impacts due to scarcity of well-formed trees (Bolivia; Fredericksen et al. 1999)</td>
<td>For dioecious species, seed crop decreases with increasing distance of males to females (Mack 1997)</td>
</tr>
<tr>
<td></td>
<td>11% of trees uprooted (Brazil: Uhl &amp; Guimaraes 1989)</td>
<td>Populations of wild fruit trees are often decimated by destructive harvesting (Peru: Vazquez &amp; Gentry 1989)</td>
<td>Higher seedling mortality (esp. in very disturbed areas) in Sabah (Pinard et al. 1996)</td>
</tr>
<tr>
<td></td>
<td>&gt;50% of young trees killed (N. Queensland, Australia: Crome et al. 1992)</td>
<td></td>
<td>Reduced density of adults (i.e. breeding individuals) ⇨ fewer flowers per unit area (C. Peters, pers. comm.)</td>
</tr>
<tr>
<td></td>
<td>~ 60-70% of residual trees damaged by logging (Fox 1968; Nicholson 1979)</td>
<td></td>
<td>1% increase in tree mortality 3-4 yr post-logging (Silva et al. 1995, Finegan &amp; Camacho 1999)</td>
</tr>
<tr>
<td></td>
<td>Large and small trees killed with equal probability (Johns 1988; Crome et al. 1992)</td>
<td></td>
<td>4-fold increase in tree mortality 2 yrs post-logging (Thiollay 1992)</td>
</tr>
<tr>
<td></td>
<td>Species w/ type III population structure may be easily eliminated (C. Peters pers. comm.)</td>
<td></td>
<td>More resources available to residual trees b/c of reduction in canopy cover and stem density (Maitre 1991); but damaged trees may need to use resources to heal w/ net result of increase in aborted fruits (Stephenson 1981)</td>
</tr>
<tr>
<td></td>
<td>Spatially clumped pattern of conspecifics shifts to dispersed or random patterns ⇨ reproductive biology and logistics of future silviculture</td>
<td></td>
<td>Potentially higher seed predation levels (Schupp 1990, C. Peters, pers. comm.)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Temporary increase in growth rates depending on extent of canopy opening (Wan Razali 1989)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Mean dbh increment 2-3 x higher in logged vs. unlogged (Jonkers 1987)</td>
</tr>
<tr>
<td></td>
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<td></td>
<td>Lower growth rates after thinning + selective logging in (E. Malaysia: Primack et al. 1989)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Lower growth rate for <em>Araucaria cunninghamii</em> (Australia: Enright 1978)</td>
</tr>
</tbody>
</table>

**Primates**

See Appendix VI
<table>
<thead>
<tr>
<th>Group</th>
<th>Impact</th>
</tr>
</thead>
</table>
| **Non-rodent mammals**       | • Civet densities in logged forest were only 20% of 1° (Malaysia, 2-12 yrs post selective logging, no hunting: Heydon & Bulloh 1996); but increased when <33-50% forest impacted (Malaysia, 5-10 yrs post logging: Johns 1983b)  
• Tree shrew densities greater 40 yrs post MUS treatment than 1° forest (Malaysia: Laidlaw 1998)  
• Giant forest hog populations decline after logging (Davies et al. in press)  
• Elephant populations increase in response to logging (Africa & Asia: Grieser Johns 1997)  
• Mousedeer (Tragulus) populations reduced after logging (80-100 m³/ha, Malaysia: Heydon & Bulloh 1997) |
| **Rodents & Marsupials**     | • Squirrel & rat populations showed variable & species-specific responses to MUS (Malaysia, 40 yrs post-treatment: Laidlaw 1998) |
| **Bats**                     | • Vampire bat densities increase if cattle introduced to logged areas (Johns et al. 1985, Wilkinson 1985, Johns 1988, 1992a, Fenton et al. 1992) |
| **Birds**                    | • See Appendix IV                                                                          |
| **Herps**                    |                                                                                             |
| **Insects**                  |                                                                                             |
| **Soil organisms**           |                                                                                             |
Appendix VI —
Primate Responses (by feeding guild) to Logging in Tropical Forests
Primate responses (by feeding guild) to logging in tropical forests; 0 = no change; - = decrease after logging; + = increase after logging

<table>
<thead>
<tr>
<th>GUILD / SPECIES</th>
<th>Ulu Segama</th>
<th>Nanga Gaat</th>
<th>Tekam</th>
<th>Sg. Lalong</th>
<th>Kemasul</th>
<th>Pta da Castanha</th>
<th>Kibale</th>
<th>Budongo</th>
<th>Kalinzu</th>
<th>Kirindy</th>
<th>Lope</th>
</tr>
</thead>
<tbody>
<tr>
<td>Logging Intensity (trees/ha)</td>
<td>20</td>
<td>10</td>
<td>18</td>
<td>&gt; 20</td>
<td>&gt; 20</td>
<td>?</td>
<td>5.1 or 7.4</td>
<td>6-25</td>
<td>?</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Damage levels (% tree loss)</td>
<td>32-58</td>
<td>54</td>
<td>51</td>
<td>38</td>
<td>55</td>
<td>60</td>
<td>25 or 50%</td>
<td>&gt;50</td>
<td>?</td>
<td>11</td>
<td></td>
</tr>
<tr>
<td>Years since logging</td>
<td>6-12</td>
<td>0-4</td>
<td>0-12</td>
<td>20-30</td>
<td>20-30</td>
<td>11</td>
<td>12-28</td>
<td>1-50</td>
<td>4-20</td>
<td>1-15</td>
<td></td>
</tr>
</tbody>
</table>

**Omnivores**

- Cebus apella: +
- Cebus albifrons: +
- Lophocebus albigena: 0/-
- Cercopithecus lhoesti: -
- Cercopithecus ascani: 0
- Cercopithecus cephus: +/+
- Cercopithecus medius: +
- Cercopithecus mitus: 0/-
- Cercopithecus pogonias: 0
- Cercopithecus nictitans: 0
- Galago spp: -
- Gorilla gorilla: 0
- Macaca spp: 0
- Macaca spp: -
- Macaca fascicularis: 0
- Macaca nemestrina: 0
- Mandrillus sphinx: 0
- Microcebus spp: +
- Pan troglodytes: 0/-
- Perodicticus potto: 0
- Phaner furcifer: +
- Pongo pygmaeus: +
- Saimiri mystax: +
- Saimiri spp: +

**Folivores/frugivores**

- Alouatta seniculus: -
- Ateles paniscus: 0
- Callicebus moloch: +
- Callicebus torquatus: +
- Procolobus badius: 0/-
- Colobus guereza: +
- Colobus satanus: +
- Presbytis hosei: +
- Presbytis melalophos: +
- Presbytis obscura: +
- Presbytis rubicunda: -

Note: The table provides a comprehensive list of primate responses to logging in various tropical forests, categorized by feeding guild and detailing the effects of logging on species abundance and distribution.
<table>
<thead>
<tr>
<th>Frugivores/folivores</th>
<th></th>
<th></th>
<th>+</th>
<th>-</th>
<th>-</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hylobates lar</td>
<td></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Hylobates muelleri</td>
<td>0</td>
<td>-</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Lagothrix lagotricha</td>
<td></td>
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<tr>
<td>Pithecia albicans</td>
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</table>

|------------|---------------|---------------|---------------|---------------|---------------|---------------|-----------------------------------------------------------------|-------------------------------|----------------|-----------------------------|------------------|

* Modified from Plumptre and Grieser Johns (in press)
Appendix VII —
Impacts of Logging (and other silviculture treatments where noted) on the Genetic Component of Biodiversity in Tropical Forests
Impacts of logging (and other silviculture treatments where noted) on the genetic component of biodiversity in tropical forests

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Structure</th>
<th>Composition</th>
<th>Function</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vegetation</td>
<td></td>
<td>Genetic diversity decreased by 5.0-23.4% (logged 1 yr prior; regenerated stands showed no significant loss); effect on dioecious species perhaps underestimated (Wickneswari et al. 1997)</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>Outcrossing rates decreased in <em>Shorea megistophylla</em> (Murawski et al. 1994a,b); density of mature individuals likely an important factor</td>
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<tr>
<td></td>
<td></td>
<td>Reduced cross-pollination rates in <em>Shorea siamensis</em> after logging (Ghazoul et al. 1998) and reduced pollination success of <em>Dipterocarpus obtusifolius</em> after logging <em>S. siamensis</em> (Ghazoul 1999)</td>
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<tr>
<td></td>
<td></td>
<td>Outcrossing rates of <em>Carapa</em> spp. were lower after controlled experiments in logged vs. unlogged forests in French Guiana (Doligez &amp; Joly 1997) but were not significantly different in <em>Carapa</em> stands in Costa Rica (Hall et al. 1994) possibly because of very high stand densities in Costa Rica.</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Studies citing genetic erosion of <em>Swietenia mahoganii</em> from logging have not been substantiated in follow-up studies (reviewed in Lugo 1999)</td>
<td></td>
</tr>
</tbody>
</table>

Primates
Non-rodent mammals
Rodents & Marsupials
Bats
Birds
Herps
Insects
Soil organisms
**Glossary**

**Advanced regeneration**: seedlings or saplings present in the understory prior to logging or other silvicultural treatment.

**Aerial logging**: a timber yarding system using suspended cables, helicopters, or balloons to lift logs.

**Afforestation**: establishment of forests in areas that did not historically support them (e.g., in savannas or drained marshes).

**Bole**: trunk or main stem of a tree.

**Buffer zone**: vegetation maintained on the borders of roads, wetlands, or other ecologically sensitive areas to mitigate the impacts of management activities.

**Bulk density**: the weight per unit volume of a material (often soil).

**Carbon sequestration**: carbon storage.

**Coppice**: sprouting of trees from stumps or roots.

**Cutting cycle**: the time (years) between stand entries for timber extraction; there can be more than one cutting cycle in a “rotation.”

**Dendrology**: the study of trees including their identification.

**Discounting**: adjusting a future value by dividing by a function of the (generally compound) interest rate; used to calculate the “net present value.”

**Edaphic**: related to or caused by particular conditions in the soil.

**Enrichment planting**: increasing the density of desirable species by interplanting, planting in gaps, or planting along cleared lines.

**Fragmentation**: the process of breaking once continuous expanses of an ecosystem type (typically forest) into island-like patches in a matrix of other land uses.

**Gap**: generally refers to a hole in the canopy created by the death of a canopy tree, but there are also understory gaps and belowground gaps.

**Girdle**: to make an incision around a tree trunk at least as deep as the cambium with the intention of killing the tree; if combined with arboricide application then referred to as “poison girdling.”

**Group selection**: harvesting of trees in clusters generally no wider than twice the height of the mature trees to promote regeneration of moderately light-demanding species (a “polycyclic” system).

**Hauling**: carrying logs by truck (lorrie) from the forest to processing or exporting facilities.

**Heartrot**: decomposition of the central stemwood of living trees typically due to fungal.
infection after mechanical or fire-related damage

**Heterozygous**: having two different alleles at the same locus on homologous chromosomes; an index of genetic diversity

**High-grading**: removal of the best trees often leaving a residual stand dominated by trees of poor form and non-commercial species; “creaming”

**Highlead yarding**: a yarding system in which logs are skidded along the ground to a central point by a cable passing through a block at the top of a tower or spar tree (not to be confused with “skyline yarding”)

**Liana**: woody vine

**Liberation thinning**: a silvicultural treatment in which future crop trees are released from competition typically by girdling near neighbors (note: term used differently by many North American foresters)

**Logging intensity**: can refer to either the timber volume or number of trees harvested per unit area

**Natural forest management**: management of forests for timber, non-timber forest products, and environmental services by relying principally on natural regeneration (compare to plantation forestry)

**Neotropics**: American tropics

**Net present value**: the current value of some future cost or benefit

**Paleotropics**: tropical areas in Africa, Asia, Australia, and the Pacific Islands

**Plantation**: a stand of trees established by planting; tree farm

**Reduced-impact logging**: a set of techniques designed to avoid excessive damage to soil or to the residual stand during and after timber harvesting; typical components include pre-planned skid trails, directional felling, and engineering specifications for stream crossings

**Reforestation**: the recovery of forests in areas that were deforested; compare to “afforestation”

**Regeneration method**: a timber harvesting method that promotes development of a new age class; the principal methods include clearcutting, seed tree, shelterwood, selection, and coppicing

**Riparian**: a terrestrial area bordering a water body

**Rotation**: the period between establishment of regeneration and final harvesting in even-aged stands

**Shelterwood method**: a stand regeneration method that occurs in two phases, a first phase in which enough timber is harvested to promote development of a new age class in partial shade, and a final felling of the remaining mature timber after regeneration is established

**Silviculture**: the art and science of controlling the establishment, growth, composition, and health of forests and woodlands

**Skidding**: yarding logs with a rubber-tired self-propelled machine or bulldozer (“snigging” in Australia)

**Skyline yarding**: a log yarding system in which logs are partially or entirely suspended from a taut cable (not to be confused with “high-lead yarding”)

**Stand**: a contiguous group of trees with a more-or-less similar history of disturbance, species composition, size-class distribution, and structure
**Succession**: the gradual change in structure and composition as an area recovers from disturbance or on previously un-vegetated areas.

**Yarding**: the process of extracting logs from the stump to the roadside, river edge, or other site from which they are hauled.

**Thinning**: a silvicultural treatment designed to reduce stand density.

**Tractor**: can refer to a bulldozer (=powered vehicle with crawler tracks) or a farm tractor.

**Note**


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