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Towards sustainable management of mixed dipterocarp forests of South-east Asia: moving beyond minimum diameter cutting limits

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SUMMARY

Selective logging applied in tropical forests is based on one universal criterion: a minimum diameter cutting limit for all commercial timber species. Minimum diameter cutting limits in mixed dipterocarp forests of the Malesia region lead to high felling intensities (10–20+ trees ha^{-1}). Such extraction rates create massive stand damage (>50% of the remaining tree population), which has a negative impact on the regeneration and growth of many harvested dipterocarp species. As such, the minimum diameter cutting limit approach is seldom compatible with sustainable forest management. Where basic ecological characteristics of the commercial species are considered in timber harvesting prescriptions, mixed dipterocarp forests appear capable of sustained timber yields, habitat conservation, and providing other goods and services. This paper first presents the main silvicultural systems developed in mixed dipterocarp forests of Western Malesia and then reviews current knowledge of dipterocarp biology to finally develop guidelines aimed at improving the ecological sustainability of production forests of Western Malesia. These guidelines, a pragmatic reflection of science and 'best guess' judgement, include: (1) integration of reduced-impact logging practices into normal management operations; (2) cutting of eight trees ha^{-1} or less (with a felling cycle of 40–60 years to be determined according to local conditions); (3) defining minimum diameter cutting limits according to the structure, density and diameter at reproduction of target species; (4) avoiding harvesting species with less than one adult tree ha^{-1} (diameter at breast height [dbh] ≥ 50 cm over an area of 50–100 ha); (5) minimizing the size and connectivity of gaps (<600 m^2 whenever possible); (6) refraining from treatments such as understorey clearing; and (7) providing explicit protection for key forest species and the ecological processes they perform. Further refinement is encouraged to allow for local conditions, and for other forest types.

Keywords: Malesia, mixed dipterocarp forests, reduced-impact logging (RIL), sustainable forest management, silviculture, East Kalimantan, habitat conservation, Tebang Pilih Tanam Indonesia (TPTI)

INTRODUCTION

Selective logging is a common production system in mixed tropical forests. The densities of commercial timber species larger than a predetermined minimum stem size (the minimum diameter cutting limit) determine felling intensities. These limits accommodate processing technologies and market demands, rather than the biology and persistence of the harvested species. This neglect means that current diameter limit approaches are unlikely to provide ecologically sustainable forest management (Sist *et al.* 1998*b*, 2002; Putz *et al.* 2000; Sheil & van Heist 2000; Jennings *et al.* 2001). Where basic biological characteristics of the commercial species are considered in timber harvesting prescriptions, mixed dipterocarp forests appear capable of sustained timber yields, habitat conservation (most structural and compositional attributes), and providing other goods and services (Putz *et al.* 2000; Sist *et al.* 2003*a*). This paper aims to reduce the gap between forest exploitation and conservation for mixed dipterocarp forests, by suggesting science-based silvicultural harvesting guidelines that favour the sustained yield of timber products, while contributing to the conservation of wildlife habitat and overall protection of the production forest estate. In any overall conception of 'good forest management' there will be many biological, social, political and economic issues to consider (for example CIFOR [Centre for International Forestry Research] criteria and indicators; Prabhu *et al.* 1999). Many of these issues are contentious and poorly defined. While this complexity is an obstacle to the realization of sustainable forest management, there are recommendations that can be made now based on what we already know. Our focus is thus on one piece of the sustainability puzzle, namely the biology of the dipterocarps themselves, their sustained yield and the maintenance of productive dipterocarp forest habitat. This is not because other issues do not matter, but rather because it is better to propose partial improvements than to propose none.

In this paper we first review harvest-regeneration management systems applied to the aseasonal evergreen mixed dipterocarp production forests (henceforth referred to simply as dipterocarp forest) of the Western Malesian Region in South-east Asia. This region includes Borneo, Sumatra,

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Peninsular Malaysia and parts of the Philippines (see Whitmore 1984 for a definition of the Malesian Region). Our attention focuses on lower altitude forests (<600 m) in Borneo and Malaysia, although our discussion should be applicable wherever forests share similar floristic and stand structures (Ashton 1982; Whitmore 1984). We specifically exclude the more seasonal forests of continental Asia, as well as the forests of the Sahul shelf (Papuasia-New Guinea and Australia; Whitmore 1984), India and Sri Lanka (Ashton 1982; Whitmore 1984; Pascal & Pélissier 1996; Appanah 1998). Peat swamp and heath forests (*kerangas*) are also beyond the scope of this paper due to their distinct ecological features. We then look at the biology of dipterocarps, considering the main stages in the dipterocarp life cycle and requirements for their survival, growth and reproduction. Considering both the existing silvicultural systems in the region and the biology of dipterocarps, we finally develop recommendations for improving the ecological sustainability of these forests. These recommendations are based on reconciling ecological principles with harvesting.

SILVICULTURAL SYSTEMS IN MIXED DIPTEROCARP FORESTS IN WESTERN MALESIA

The dipterocarp forests of Western Malesia are the most productive tropical forest types in Asia, with considerable timber value (FAO [Food and Agricultural Organization of the United Nations] 2001). In Indonesia, for example, of the 41 million ha in forest concessions, 43% are still pristine forest, and 27% of the logged forest are still considered in moderate to good condition (FLB [Forest Liaison Bureau] 2002). Dipterocarps contribute most of the commercial timber. Extracted volumes vary from 50–100 $m^3\,ha^{-1}$, the forests of Borneo generally being the most productive with extracted volumes often exceeding 100 $m^3\,ha^{-1}$ (Nicholson 1979; Pinard & Putz 1996; Bertault & Sist 1997; Sist *et al.* 1998*b*). These forests are also some of the most species rich in the world (Whitmore 1984, 1990; Richards 1996), typically with 150–200 species of trees ($\geqslant$10 cm dbh) per hectare. Dipterocarpaceae are the dominant family representing 25% of stems (120 stems $\geqslant$ dbh 10 cm ha^{-1}), with large size stems contributing 75–80% of the canopy and emergent trees, and half (15 $m^2\,ha^{-1}$) of the total basal area (Whitmore 1984; Appanah & Weinland 1993; Sist & Saridan 1999). The preservation of such diverse and productive ecosystems through sustainable forest management practices remains one of the main challenges in the region.

Dipterocarp forests were among the first tropical forests where sustained timber production was attempted (Wyatt-Smith 1963; Appanah 1998; Dawkins & Philip 1998). Amongst the first management efforts was the Malayan Uniform System (MUS) introduced in 1948 (see Wyatt-Smith 1963). The MUS was characterized by the felling of all trees above 45 cm dbh, and poison girdling of all defective relics and non-commercial species down to 5 cm dbh. It required successive liberation release treatments (i.e. understorey clearing and liana cutting) 20, 35 and 55 years after logging (Wyatt-Smith 1963). This treatment aimed to convert uneven-aged, mixed forest into more timber rich even-aged stands.

By the mid-1970s, much of Peninsular Malaysia's lowland forest had been harvested, and in many instances converted to plantations. Harvesting subsequently shifted to the hill forests. In these steeper areas, the MUS was judged unsuitable due to patchy regeneration (Appanah 1998). Although the system was adapted (the Modified Malayan Uniform System) to include enrichment planting when natural regeneration was poor, the outcome was often unsatisfactory and the approach was abandoned (Appanah 1998). In the late 1970s, the Selective Management System was developed in Malaysia (Wyatt-Smith 1987). In this system, all commercial species with a dbh $\geqslant$ 45 cm for non-dipterocarps and $\geqslant$ 50 cm for dipterocarps, are felled as long as sufficient healthy stems remain to support another harvest in *c*. 30 years (Appanah 1998).

Based on the Malaysian silvicultural experiences, other South-east Asian countries developed and applied polycyclic (multi-aged) approaches similar to the Selective Management System (see Appanah 1998). These systems are all based on minimum diameter limit rules. The Indonesian cutting and planting system, known as Tebang Pilih Tanam Indonesia (TPTI), is an example relevant over the breadth of the Indonesian dipterocarp forests. In the TPTI all commercial trees above 60 cm dbh can be felled with a 35-year cutting cycle. Liberation cuttings, involving clearing of understorey vegetation, take place two, four and six years after logging, and target all woody climbers and non-commercial saplings in the understorey. This treatment is supposed to improve regeneration and growth of timber species. TPTI relies on leaving behind a minimum density of potential crop trees (25 ha^{-1}), which are sound stems of commercial species (20 cm dbh and above). If these are not present, enrichment planting is required three years after logging.

Companies incur significant costs while implementing the TPTI. Full implementation requires infrastructure and skills, including nurseries and qualified people that many concessionaires are not able or willing to support. Though procedures are normally implemented, our own observations show that the quality of treatments often fails to meet the intention of the original TPTI guidelines. For example while pre-harvesting inventories are carried out to obtain the annual allowance cutting, the resulting maps are not then used to plan skidtrail networks. Others activities such as post-logging liberation cuttings and enrichment planting are widely questioned (see later).

In some regions of Western Malesia, where unlogged forests are well stocked with timber (for example in East Kalimantan 16–23 merchantable stems ha^{-1}; Sist & Saridan 1999), selective logging operations often damage more than 50% of the stand, impacting both forest structure and productivity (Nicholson 1979; Pinard & Putz 1996; Bertault & Sist 1997). Such damage reduces available timber volume for the next cut in 30–40 years (Favrichon & Young Cheol 1998; Sist *et al*. 1998*b*, 2003*a*; Huth & Ditzer 2001). The sustainability of such systems is now widely questioned (for example in Sabah: Kleine & Heuveldop 1993; and Indonesia: Sist *et al*. 1998*b*, 2002, 2003*a*, *b*).

Over the last 15 years, timber demands have been changing and harvesting is no longer exclusive to red merantis. Almost all dipterocarps, except *Vatica* spp., are now considered commercial, and this trend seems to be continuing, with more and more species becoming valuable (dipterocarps and non-dipterocarps). In addition, many others forest products are recognized as valuable and increasing attention is being paid to the needs of local people (Sheil *et al*. 2003).

Finally, there is a growing awareness of the need to protect forest ecosystem functions and to maintain biological diversity in production forests. Although strictly protected areas appear the best means to conserve forest species, production forests can provide valuable forest habitat over larger areas (Grieser Johns 1997; Putz *et al.* 2000; Fimbel *et al*. 2001). Such issues have become increasingly important in the region, with most forest departments investing considerable efforts in conservation and biodiversity issues. This is manifesting itself in the exploration of revisions to regional codes of practice, including the adoption of criteria and indicators of sustainability, implementation of low impact methods, and the gazetting of national parks and other conservation areas.

BIOLOGY OF MALESIAN DIPTEROCARPS IN MIXED DIPTEROCARP FORESTS

Phenology and pollination

Malesian dipterocarp forests are renowned for their supra-annual masting cycle. Flowering is seldom annual, occurring on intermittent multi-year cycles (usually every few 3–5 years), which appear to follow El Niño events (Appanah 1993; Curran *et al*. 1999). Most dipterocarps reach reproductive size when individuals attain ⩾ 50 cm dbh (Burgess 1975; Nguyen-Thé & Sist 1998), although smaller trees can sometimes exhibit early flowering (Srivastava 1977). Significant fruit production is found only in large individuals (Appanah & Rasol 1990).

Dipterocarps are mainly out-crossing and self-incompatible. In dipterocarps, studies of breeding systems conducted so far have been based on very small sample sizes in very few species (Bawa 1998). The dipterocarps are pollinated by a wide variety of insects (Bawa 1998). Pollinator species can vary according to dipterocarp species, and perhaps to some degree by region and locality (Ashton 1982; Appanah 1990; Bawa *et al*. 1990; Bawa 1998). In Pasoh, Malaysia, Appanah & Chan (1981) suggested the importance of thrips as pollinators of red meranti (*Shorea*, section *muticae*). Elsewhere, moths, butterflies, and bees apparently serve as the primary pollination agents (see Bawa 1998 for a review). Flowering episodes, though near simultaneous, often occur in a staggered sequence, suggesting that the trees are 'sharing' and competing for pollinators (see for example Whitmore 1990). Available information suggests that pollinators of many such masting dipterocarps are not host specific (Kato 1996; Momose *et al*. 1998). These pollinator species need to have alternative food sources outside of the very short periods when the trees are in flower. Most pollinators associated with dipterocarps are not strong fliers (for example thrips and beetles) and do not cover long distances (Ghazoul *et al*. 1998).

Fruit, seeds and germination

Few fruits fall outside of mast years although not every potential tree fruits in every mast year. Dipterocarp fruits are winged, but generally disperse short distances (usually no more than 60–80 m) in closed-canopy forests (Burgess 1970, 1975; Tamari & Jacalne 1984; Whitmore 1984). Only in strong winds preceding rainstorms are fruits likely to be dispersed to greater distances (Nathan *et al*. 2002).

Pre-dispersal predation of unripe dipterocarp fruits by insects can have a severe impact on fruit production (Momose *et al*. 1996). Following maturation, the oil rich seeds are a significant source of food for a number of opportunistic species, including pigs and rodents. Pigs (mainly *Sus barbatus* in Borneo and *Sus scrofa* in peninsular Malaysia) are major consumers of dipterocarp seeds, and pig densities have been proposed as a cause of failed dipterocarp regeneration in Borneo (Curran *et al*. 1999; Ickes *et al*. 2001). It is only in masting years that such predators are satiated and a significant proportion of fruits given an opportunity to survive and germinate (Curran & Leighton 2000). During mast periods the levels of seed rain are lower in logged versus unlogged areas, due to fewer adult and fecund stems, making it easier for local seed predators such as rats and migratory pigs to consume most of them (Curran *et al*. 1999).

Dipterocarp seeds invariably germinate within a very few days or die; no viable seeds remain in the seedbank. However, the seedlings are capable of persisting in the understorey for several years. This 'seedling bank', along with 'advanced regeneration' in the form of saplings and poles, benefits from periodic canopy openings (Whitmore & Brown 1996; Brokaw & Busing 2000).

Recruitment and growth

Most dipterocarp seedlings and saplings can survive rather long periods in the understorey, exhibiting very little growth (Ashton 1998). However, they generally survive and grow better under increased light intensities, responding well to small gaps (Whitmore & Brown 1996). Most dipterocarp species require canopy openings no greater than those created by single-tree selection cutting practices to sustain their development (*c*. 500–600 m^2; Kuusipalo *et al*. 1996; Tuomela *et al*. 1996; van Gardingen *et al*. 1998).

Dipterocarps are generally dependent on ectomycorrhizal symbionts (one major class of beneficial root fungi) for initial establishment and survival, and later for good growth (Lee

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1998). Suitable mycorrhizal infection of juvenile trees can be limited in areas without infected adult dipterocarps (Alexander 1989; Alexander *et al.* 1992). Soil compaction, erosion, and nutrient loss, along with changes in hydrology and the removal of source trees, can all negatively impact mycorrhizal fungi, reducing beneficial tree infection and negatively influencing dipterocarp establishment, growth and survival (Alexander *et al.* 1992).

Most commercial dipterocarp species show a classic reverse J-shaped diameter (type I) distribution (Fig. 1) with high densities of young, relatively shade-tolerant individuals (Fox 1973; Ashton 1998). However, some dipterocarps (for example *Dipterocarpus crinitus*; Appanah & Weinland 1993) and important non-dipterocarp commercial species (such as *Agathis borneensis*) show a J-shaped diameter (type II) distribution (Fig. 1). These are relatively shade-intolerant species, which require abundant exposure to sunlight in their early development stages to survive and grow. Adult trees, with few juveniles present, are the main component of the population in forests subject to major but infrequent disturbance events. We observed that these light-demanding species existed before timber harvesting. It is wrong to simply propose that such species require more canopy opening than that created by logging. Harvesting operations do not replace natural disturbance regimes. Indeed such species are especially vulnerable to eradication when adults are removed and persistent regeneration is destroyed.

Dipterocarps exhibit a wide range of annual growth increments depending on species, location and environmental conditions. Growth rates of 0.30–0.50 cm yr^{-1}, and sometimes >1 cm yr^{-1}, appear normal for dipterocarp species (see Sist & Nguyen-Thé 2002).

Breeding and genetic processes

Research on the genetic variability of tree populations in dipterocarp forests is limited, but one fear is that the disruption of pollination processes following logging, through either the direct destruction of pollinators, their habitats, or increases in distance between reproductive trees, may negatively affect the genetic population structure and its intraspecific genetic diversity (Ghazoul & Hill 2001). However, the evidence remains unclear. Studies in Peninsular Malaysia (Wickneswari & Boyle 2000) on *Shorea leprosula* and in Brunei (Kitamura *et al.* 1994) on *Dryobalanops aromatica* reported no significant differences in out-crossing rates between logged-over and primary forest. Nonetheless, a loss of genetic diversity in five dipterocarp tree species immediately after timber extraction, the highest loss being for commercial timber species with low abundance, has been reported (Wickneswari & Boyle 2000).

If self-incompatibility systems are weak, as is apparently the case for the few species of dipterocarps studied (Bawa 1998), then inbreeding rates after harvesting may increase, especially in low density species or those shade intolerant species with limited seedling stock (Murawski & Hamrick 1990, 1991; Lee 2000; Obayashi *et al.* 2002). Jennings *et al.* (2001) suggest that the genetic diversity of shade-tolerant trees will be little affected by light harvesting, as advanced regeneration should serve as a genetic reservoir for the species being harvested.

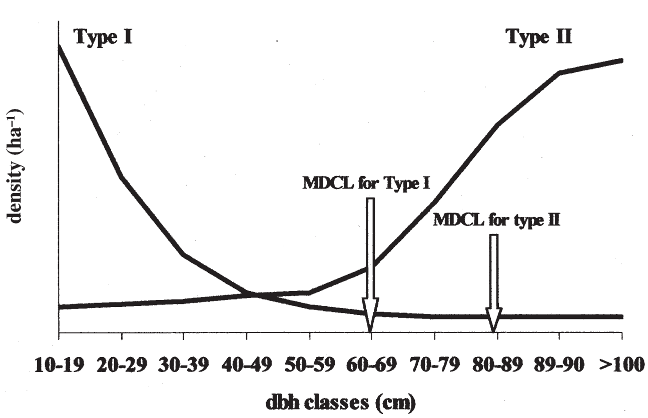

Figure 1 Ecological harvesting prescriptions according to population dbh structure. Type I = most dipterocarps (Appanah & Weinland 1993); Type II = species such as *Dipterocarpus crinitus* and *Agathis borneensis*. Arrows show the suggested minimum diameter cutting limit (MDCL) for each type.

TOWARDS A MORE SUSTAINABLE HARVEST OF MIXED DIPTEROCARP FORESTS

Reduced-impact logging (RIL) techniques as a prerequisite to any sustainable forest management system

The conservation potential of managed production has helped spur the development and implementation of timber harvesting practices generally referred to as 'low' or 'reduced-impact logging' (Dykstra & Heinrich 1996; Elias 1998; Sist *et al.* 1998*a*; Sabogal *et al.* 2000; Sist 2000). Reduced-impact logging techniques are widely recognized as an essential component of sustainable timber harvesting prescriptions (Ong & Kleine 1995; Pinard *et al.* 1995; Elias 1998; Putz *et al.* 2000; Fimbel *et al.* 2001).

These techniques generally act at the operational plan level by designing skidtrail layout, practising carefully controlled felling and skidding, and reducing damage to soils and residual trees (Dykstra & Heinrich 1996; Sist *et al.* 1998*a*). However, this level cannot be fully disconnected from the wider landscape-level of the forest management plan, which determines for example where the main roads should be sited and the setting aside of areas for biological or hydrological protection, or where the slope is too severe (Dykstra & Heinrich 1996; Sist *et al.* 1998*a*). Though we recognize the ecological value of connectivity between all forest areas in a larger landscape, our discussion focuses only on the operational plan level of RIL implementation (i.e. the annual coupe level, *c.*1000 ha). RIL methods limit environmental harm by reducing damage to soils and residual trees, and protecting areas from harvest (steep areas and riparian corridors) (Dykstra & Heinrich 1996; Sist *et al.* 1998*a*; Putz *et al.* 2000; Fimbel *et al.* 2001). Recovery depends on advance regeneration (saplings and young stems) remaining after the timber extraction processes are completed. Careful planning of skidtrails can significantly reduce the impact of logging on the regeneration strata. For example, in East Kalimatan and Sabah, RIL techniques have reduced the coverage of skidtrails from 14–17% of the harvested area (using conventional harvesting techniques) to 6–9 % (Pinard *et al.* 2000; Sist *et al.* 2003*b*). Moreover, the degree of soil disturbance on well-planned skidtrails is generally much lower than that in conventional harvests (Sist *et al.* 2003*b*). These improved soil conditions should benefit soil biology and subsequent dipterocarp establishment.

Unfortunately, RIL practices as currently formulated are insufficient to guarantee sustainability even in a narrow timber production sense (Sist *et al.* 2002). Improving the ecological sustainability of dipterocarp production forests requires additional measures, including those we propose below.

Logging intensity ≤ 8 trees/ha associated with a felling cycle of 40–60 years according to site conditions

Harvests guided only by minimum diameter limits run the risk of very high potential extraction rates in well-stocked mixed dipterocarp forests. Although planned skidtrails, directional felling, and other low-impact logging measures help to reduce logging damage, only with moderate extraction rates can damage levels be lowered to the 25–30% considered an upper limit to sustainable timber production (Johnson & Cabarle 1993; Favrichon & Young Cheol 1998; Huth & Ditzer 2001; Sist *et al.* 2003*a*). Several studies emphasize that this is achieved by the simple rule of limiting harvests to a maximum of eight trees per hectare (Bertault & Sist 1997; Sist *et al.* 1998*a*, 2003*b*). One practical way to remain under this logging intensity threshold, is to define a minimum spacing distance between harvested trees. In a homogeneous spatial distribution under maximum horizontal packing (triangular) the distance D between trees in metres is given by the formula:

$$D = \frac{200}{\sqrt{3x}}, \tag{1}$$

where x is the density in stems per ha. This is slightly longer than packing in a regular square lattice where

$$D = \frac{100}{\sqrt{x}}, \tag{2}$$

Following these two equations, in a homogeneous distribution, the minimum spacing distance for a maximum felling intensity of 8 trees ha^{-1} for each of these is 40.8 m and 35 m, respectively (Sist *et al.* 2003*b*).

Growth rates of 0.3 to >1 cm yr^{-1} necessitate 40–60 years between harvests to allow sufficient recruitment of medium-size residual stems (40–60 cm dbh) to occur into the harvestable size class (Favrichon & Young Cheol 1998; Huth & Ditzer 2001; Sist *et al.* 2003*a*). With heavy extraction (>8 trees ha^{-1}) rotation cycles will need to exceed 60 years to ensure sustainability (Huth & Ditzer 2001). In contrast, a simulation model based on six years of monitoring permanent sample plots in East Kalimantan suggests that applying RIL techniques with a harvesting intensity of 8 trees ha^{-1} can sustain a 40-year felling cycle with a yield of 66 m^3 ha^{-1} (i.e. 1.5 m m^3 ha^{-1} yr^{-1}; Fig. 2).

Cutting limits must be adapted to population structure and density

The density of reproductive trees remaining after logging is important for ecological sustainability (Martini *et al.* 1994; Pinard *et al.* 1999). Yet selective cutting systems have generally failed to consider the minimum diameter at which individuals of a species become reproductive and how fecundity relates to size (Fig. 1). Given that most dipterocarps are reproductively mature at diameters ≥ 50 cm dbh, it appears that minimum diameter limit cuts of dbh > 60 cm (outlined by TPTI) allow for the continuation of fruit production in most dipterocarp species, while also curtailing potential

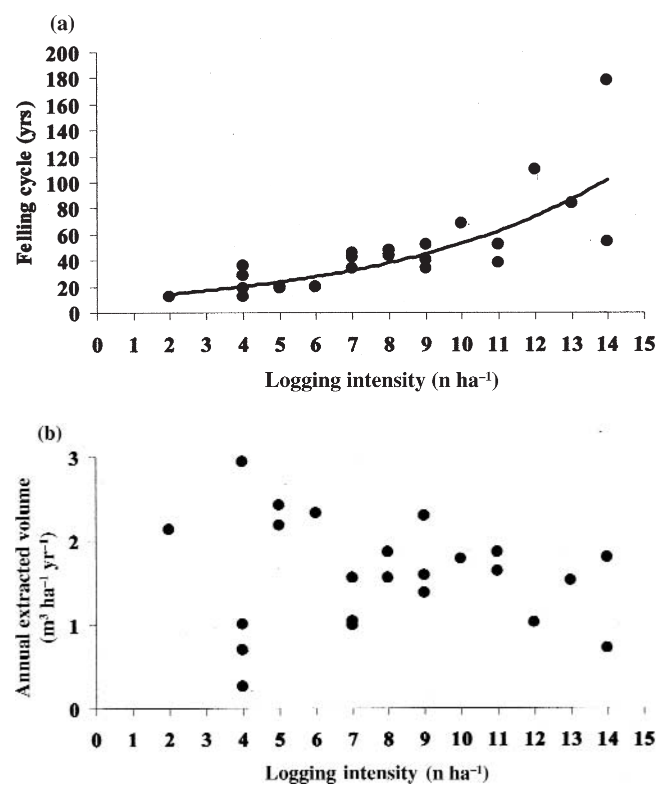

Figure 2 (*a*) Felling cycle length (Y) as a function of logging intensity (X = trees felled ha^{-1}); $Y = 10.3e^{0.162X}$ ($R^2 = 0.76$, F = 97.0, df = 32, $p < 0.001$). (*b*) Sustainable annual extracted volume under RIL regime as a function of logging intensity. Results of forest dynamics simulation based on six years monitoring of 23 subplots, 1 ha each, in Berau, East Kalimantan, logged with RIL techniques. (Source: Sist *et al.* 2003*a*.)

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harvest damage (Fig. 1). However, there is still a risk of reduced seed set, and seed predation surpassing production, in post-logged forest (cf. Curran *et al.* 1999). One means to reduce this risk is to retain some of the largest, most fecund stems in the stand (see below). For species showing a J-shaped diameter distribution (Type II; Fig. 1), logging applying the current minimum diameter cutting limit will drastically reduce the adult population density and will leave few young individuals (Fig. 1). Hence, for type II species, we recommend increasing the minimum diameter limit (dbh $\geqslant$ 80 cm in the case of *Agathis borneensis*) to retain at least one adult tree per hectare in the harvest area (Jennings *et al.* 2001).

Limit harvesting to species with a density >1 adult per hectare, especially if 'shade-intolerant'

Minimum diameter cutting limits threaten the over-harvest of rare species. This is especially true for species with broad but low density distributions (average of <1 adult tree ha^{-1}) and clumped, localized populations with low mean densities (Bawa & Ashton 1991). In mixed dipterocarp forests, individual species of dipterocarps typically exhibit densities of 0.05–2 adult trees ha^{-1} (dbh $\geq$ 50 cm; Poore 1968; Soepadmo 1995; Sist & Saridan 1999). Harvesting species that show an average of less than one adult tree ha^{-1} (>50 cm dbh for dipterocarps) can place a species at risk of local extirpation (Jennings *et al.* 2001). We suggest that harvesting should be limited to species with a density >1 adult ha^{-1} with densities being assessed at a scale of 50–100 ha.

Logging activities based on a uniform diameter cutting limit for all the species also run the risk of extirpating shade-intolerant species (Type II) with little advanced regeneration (Fig. 1), by removing most or all adult trees. These are light-demanding trees, the seedlings of which do not persist long under low-light conditions, and it is important to increase the minimum diameter limit to leave at least one or more individuals per hectare. These densities can be estimated from a pre-logging forest inventory in the annual coupe (i.e. these densities can be calculated over 50–100 ha). An additional argument for leaving reasonable adult densities (1 ha^{-1}) of each tree species are the feedback processes (Allee effects) that can reduce the viability of small and low density populations. Such effects could arise, for example, from reduced fecundity and less genetic variation of offspring caused by reduced pollen transfer between trees (Ghazoul *et al.* 1998).

Keep gap sizes below 600 m^2

In natural forests, gap openings tend to occur at low frequencies of approximately 1% of the forest area per year (Brandani *et al.* 1988; Riera & Alexandre 1988). Most involve one large tree fall, and openings seldom exceed 200 m^2 in size (Sanford *et al.* 1986; Brandani *et al.* 1988; Brown 1993). Evidence suggests that most important dipterocarp species do not require canopy openings larger than those created by single-tree felling (below 500–600 m^2) to regenerate (Kuusipalo *et al.* 1996; Tuomela *et al.* 1996; van Gardingen *et al.* 1998), and reach high/appropriate growth rates (Whitmore 1990; Ashton 1998; Clearwater *et al.* 1999). Techniques such as directional felling and pre-harvest climber cutting have a limited effect on reducing gap size (Cedergren 1996; Sist *et al.* 1998*b*; Parren & Bongers 2001). The most important factor in reducing gap size is limiting the number of trees felled per hectare. Another is to avoid felling large trees ($\geqslant$100 cm dbh). The retention of mature individuals is also an insurance against a decline in the frequency and quality of mass seed crops, and preserves a valuable part of the habitat. In addition, these very large individuals often have structural defects that reduce their timber value. Finally, as seed sources they represent local genetic stock that is likely to be better-adapted to local site conditions than most enrichment plantings, and is more desirable from a conservation perspective.

Conventional harvesting operations often involve several adjacent trees being felled, creating larger combined gaps (to >2000 m^2) that favour pioneer species regeneration (Swaine & Whitmore 1988). Large gaps are also prone to colonization by weedy species and climbers that can suppress regeneration and growth of more valuable species (Burgess 1975; Appanah & Putz 1984; Putz 1991; Perez-Salicrup 1998). Moreover, large canopy openings also significantly increase forest flammability, particularly during long periods of drought as periodically occurs in South-east Asia during El Niño events (Bertault 1991; Laumonier & Legg 1998; Dennis 1999).

In the mixed dipterocarp forest of East Borneo, applying a 600 m^2 gap limit by avoiding felling of large trees (>100 cm dbh) would reduce tree harvests from *c.* 14–16 to 10–11 stems ha^{-1} (Cedergren 1996; Sist & Saridan 1999). This suggests that a maximum recommended threshold of 8 trees ha^{-1} will often require leaving one or more stems in the merchantable size class (60–100 cm dbh), after retaining all stems over 100 cm dbh.

Minimize understorey slashing/thinning treatments

Mast fruiting events require large populations of pollinators to be available during the brief flowering period. A reduction in pollinator success due to key-habitat loss, and subsequent declines in dipterocarp reproduction success are potential consequences of heavy harvests under short rotations (see discussion of genetic processes above). Though poorly studied, the destruction of pollinator habitat during timber harvesting poses threats to tree regeneration. It is therefore essential to minimize logging impacts to maintain pollinators and the habitat elements that they require. Bee nesting sites (such as *Koompasia* trees and hollow stems) should be preserved (as is done by local shifting cultivator communities in much of the region). Also, understorey shrubs with non-specialized pollination types and non-seasonal and continuous flowering (for example Rubiaceae, Annonaceae and Euphorbiaceae) should not be cut back and cleared as they provide both habitat and food for many generalist pollinators to persist during the long period between mast years.

Indeed, damaging general blanket prescriptions such as understorey clearing, as required in TPTI, are likely to have undesirable impacts on ecological processes, habitat values, and biodiversity in general. Such treatments should not be undertaken unless specifically justified (such as in areas infested by aggressive climbers and weeds; see Sheil 2002 and Sheil *et al*. 2003 for more detail).

WHAT ARE THE COSTS ASSOCIATED WITH MORE SUSTAINABLE HARVESTS?

One argument that is likely to arise in assessing our proposed measures relates to reduced revenues (harvesting threshold at first felling cycle), and the additional needs for training, planning and supervision. While we have not been able to do a specific economic analysis of our recommendations, several studies in different continent have demonstrated that RIL was less costly and more profitable than conventional logging. These benefits relate to time saved, greater efficiency in log extraction (Dwiprabowo *et al*. 2002; Holmes *et al*. 2002). However, the comparison of the costs of RIL and conventional approaches in a 450 ha area in Sabah indicates that the financial profits from logging were substantially lower with RIL. The main reason was the reduction in yield due to protected exclusion areas in RIL (steep slopes and buffer zones either side of rivers; Putz *et al*. 2000). RIL aims to promote long-term yields while conventional logging arguably does not. Therefore, comparison of the two systems in terms of returns must be made using the same timeframe, considering how the two methods can sustain successive felling cycles. Moreover, the loss of forest lands and environmental services associated with conventional harvesting should also be considered in any cost-benefit analysis. For Sabah, yield simulation models (Huth & Ditzer 2001) suggest that logging cycle lengths between 20 and 60 years have nearly the same financial returns (3% annual discount rate, harvesting cost equal to 20% of the merchantable yields, financial returns calculated in 400 years). Longer cycles have a lower return. A 100-year logging cycle leads to a reduction of 10% in the financial return compared with shorter cycles (20–60 years). With conventional logging, cycles will need to exceed 60 years to ensure yield sustainability, whereas RIL can maintain shorter cycles (Huth & Ditzer 2001; Sist *et al*. 2003*b*). Most of the Asian countries have undertaken reforms of their forest legislation to achieve sustainable forest management (Poore & Chiew 2000). These reforms have usually obligated forest managers to apply RIL techniques to limit logging impact on forest ecosystems (FLB 2002). In these conditions, any concerns about the costs of training, planning and supervision associated with our proposed regulations become less relevant as the associated costs are already obligatory.

NEXT STEPS

Our suggestions are based on scientific reasoning and judgments. They are neither perfect nor intended as the final word. The point is that these recommendations can be useful now. By making them explicit they can be applied and discussed, but it cannot be claimed that advice on improved practices ‘does not exist’. Further adaptations, additional guidelines and operational procedures should all be developed to help current forest management better reflect our ecological knowledge. We have suggested a few initial steps, but others are also needed. We would especially encourage greater training and reform in management procedures. Knowledgeable and committed staff should ultimately take control of forest management based less on prescriptive rules than on observing and responding to local needs. In the shorter term, we note that there is little point in identifying improved guidelines if guidelines are not implemented.

CONCLUSIONS

At present, selective harvesting systems in the Malesian dipterocarp forests ignore tree biology and appear likely to be unsustainable. This is in part attributable to the fact that tree ecology has received limited consideration in the development of harvest-regeneration protocols. Our recommendations aim to maintain productivity, tree diversity, viable habitat and ecological functions. These goals also serve the increasing demand for multiple benefits from tropical forests, including a wider range of timbers and non-timber products, as well as the less tangible benefits of tropical forests for biodiversity conservation and recreation. Our specific recommendations are that (1) reduced-impact logging practices be integrated into normal management operations; (2) only 8 trees ha^{-1} or less be cut with a felling cycle of 40–60 years to be determined according to local conditions; (3) minimum diameter cutting limits be defined according to the structure, density and diameter at reproduction of target species; (4) harvesting species with less than 1 adult tree ha^{-1} (dbh $\geq$ 50 cm over an area of 50–100 ha) be avoided; (5) the size and connectivity of gaps ($<600\ m^2$ to the extent possible) be minimized; (6) treatments such as understorey clearing be refrained from; and (7) explicit protection be provided for key forest species and the ecological processes they perform (Table 1).

Our recommendations are intended to promote the elusive goal of sustainable forest management in mixed dipterocarp forests. They draw upon existing ecological information and our own scientific reasoning and judgement to develop practical actions that forestry departments can take to promote sustained yields, and donors and stakeholders can endorse to help conserve production forests of the region. Adoption and implementation of such guidelines is urgently needed. It is worth re-emphasizing, however, that they are not a final solution. Local adaptation and fine-tuning is encouraged, especially in other forest types.

We were encouraged to see that the Director General of Forest Development in Indonesia decreed on 14 March 2002, that concession companies are now required to apply low-impact harvesting techniques in mixed dipterocarp forests,

Table 1 Recommended modifications to the minimum diameter cutting limit rule in mixed dipterocarp forests and the ecological justification for these actions.

Principle and practices	*Justification*
1. RIL techniques as a pre-requisite to any sustainable forest management system	Low impact logging practices minimize damage to the residual stand during and following the harvest operation (Sist *et al.* 1998*a*)
2. Logging intensity $\leqslant$ 8 trees ha^{-1} associated with a felling cycle of 40–60 year felling cycle according to site conditions. Leave a minimum distance of 35–40 m between cut stems	Higher extraction rates lead to high damage levels (>40% of the original stand), increasing cycle length (>60 years). RIL techniques with a harvesting intensity of 8 trees ha^{-1} can sustain a 40-year felling cycle with a yield of 66 m^3 ha^{-1} (Sist *et al.* 2003*a*). In a homogeneous distribution, the minimum spacing distance for a maximum felling intensity of 8 trees ha^{-1} is 35–40 m (see text)
3. Cutting limits must be adapted to population structure and density: harvesting dipterocarps $\geqslant$ 60 cm dbh and refining cutting diameters according to dbh structure of the species	Most dipterocarps become reproductively mature at approximately 50 cm dbh and show a type I distribution. For species showing a J-shaped diameter distribution, applying a 60 cm minimum diameter cutting limit may drastically reduce the adult population (see Fig. 1 and text)
4. Limit harvesting to species with a density > 1 adult ha^{-1}, especially if shade-intolerant	Harvesting species with low densities (rare species) can negatively impact long-term regeneration of these species (Jennings *et al.* 2001)
5. Keep gaps small (maximum of 600 m^2) by creating single tree felling gaps and avoiding harvesting trees with dbh > 100 cm	Large gaps create environmental conditions unfavourable to dipterocarp establishment and growth while favouring that of non-commercial competitors. Multiple gaps are common in logged-over forest and much bigger than single ones. Favouring single gap formation during felling will therefore limit gap size. Leaving trees larger than 100 cm dbh minimizes gap sizes, ensures the retention of mature trees genetically well-adapted to the site conditions, and maintains wildlife habitat
6. Refrain from general treatments such as understorey slashing/thinning	Understorey plants provide important habitat for pollinators, helping to maintain healthy populations of these invertebrate species
7. Provide explicit protection for key tree species in the stand (e.g. those supporting bees nests, figs, etc.)	Many plants and animals are known to serve important roles in forest ecological processes, e.g. figs serve to maintain many frugivorous species in periods where fruit availability is low

and must limit extraction rates to 8 trees ha^{-1} $\geqslant$ 60 cm dbh (FLB 2002). The universal application of sustainable management practices will take time, owing to the numerous technical, political, social and economic factors that require attention, but progress can be achieved step by step.

Finally, the development of harvesting systems has historically been the domain of foresters. Tropical forestry has been dominated by western silvicultural concepts that mainly aimed to favour a limited number of species while eliminating the invaluable one. In contrast, we believe that the extreme high diversity of tropical forest should be maintained and favoured as it represents an important biological and potential economical value for the future. There is a need to draw in a wider range of expertise to identify opportunities for improvements in tropical forest management and resource protection. Conservation biologists and ecologists need to consider applied research to refine and augment the recommendations presented above, so that managers have the necessary tools to conserve and protect the multiple resources in dipterocarp forests (Sheil & van Heist 2000). This is not the first paper to urge for the better integration of ecology into tropical forest management (Kleine & Heuveldop 1993; Sheil & van Heist 2000; Sist *et al.* 2002). The guidelines we present here should be viewed as one more evolutionary step along the sustainable forest harvesting systems pathway. There is a need to further test and refine these techniques, in an effort to minimize their costs while maximizing their conservation benefits. We hope that forestry stakeholders, including forest departments, donors and certification groups, can rapidly endorse, apply and build upon our suggestions to help conserve the tropical forests.

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References

Alexander, I. (1989) Mycorrhizas in tropical forests. In: *Mineral Nutrients in Tropical Forest and Savanna Ecosystems*, ed. J. Proctor, pp. 169–188. Special Publications Series of the British Ecological Society 9. Oxford, UK: Blackwell Scientific Publications.

Alexander, I., Ahmad, N. & See, L.S. (1992) The role of mycorrhizae in the regeneration of some Malaysian forest trees. *Philosophical Transactions of the Royal Society of London Series B* **335**: 379–388.

Appanah, S. (1990) Plant-pollinator interactions in Malaysian rain forests. In : *Reproductive Ecology of Tropical Forest Plants*, ed. K.S. Bawa & M. Hadley, pp. 85–101. Man and Biosphere Series, Volume 7. Paris, France: Parthenon Publishing Group.

Appanah, S. (1993) Mass flowering of dipterocarp forests in the aseasonal tropics. *Journal of Biosciences* **18**: 457–474.

Appanah, S. (1998) Management of natural forests. In: *A Review of Dipterocarps, Taxonomy, Ecology and Silviculture*, ed. S. Appanah & J.M. Turnbull, pp. 133–149. Bogor, Indonesia: CIFOR.

Appanah, S. & Chan, H.T. (1981) Thrips: the pollinators of some dipterocarps. *Malaysian Forester* **44**: 234–252.

Appanah, S. & Putz, F.E. (1984) Climber abundance in virgin dipterocarp forest and the effect of pre-felling climber cutting on logging damage. *Malaysian Forester* **47**: 335–342.

Appanah, S. & Rasol, M. (1990) Smaller trees can fruit in logged dipterocarp forests. *Journal of Tropical Forest Science* **3**: 80–87.

Appanah, S. & Weinland, G. (1993) A preliminary analysis of the 50-hectare Pasoh demography plot: I. Dipterocarpaceae. Research Pamphlet 112, Forest Research Institute, Kuala Lumpur, Malaysia.

Ashton, M.S. (1998) Seedling ecology of mixed-dipterocarp forest. In: *A Review of Dipterocarps, Taxonomy, Ecology and Silviculture*, ed. S. Appanah & J.M. Turnbull, pp. 90–98. Bogor, Indonesia: CIFOR.

Ashton, P.S. (1982) Dipterocarpaceae. *Flora Malesiana I* **9**(2): 237–552.

Bawa, K.S. (1998) Conservation of genetic resources in the Dipterocarpaceae. In: *A Review of Dipterocarps, Taxonomy, Ecology and Silviculture,* ed. S. Appanah & J.M. Turnbull, pp. 45–56. Bogor, Indonesia: CIFOR.

Bawa, K.S. & Ashton, P.S. (1991) Conservation of rare species trees in tropical rainforests: a genetic perspective. In: *Genetic and Conservation of Rare Plants*, ed. D.A. Falk & K.F. Holsinger, pp. 62–71. New York, USA: Oxford University Press.

Bawa, K.S., Ashton, P.S. & Nor, S.M. (1990) Reproductive ecology of tropical forest plants: management issues. In: *Reproductive Ecology of Tropical Forest Plants*, ed. K.S. Bawa & M. Hadley, pp. 2–13. Man and Biosphere Series, Volume 7. Paris, France: Parthenon Publishing Group.

Bertault, J.-G. (1991) Quand la forêt tropicale s'enflamme. Près de 3 millions d'hectares détruits à Kalimantan. *Bois et Forêts des Tropiques* **230**: 5–14.

Bertault, J.-G. & Sist, P. (1997) An experimental comparison of different harvesting intensities with reduced-impact and conventional logging in East Kalimantan, Indonesia. *Forest Ecology and Management* **94**: 209–218.

Brandani, A., Hartshorn, G.S. & Orians, G.H. (1988) Internal heterogeneity of gaps and species richness in Costa Rican tropical wet forest. *Journal of Tropical Ecology* **4**: 99–119.

Brokaw, N. & Busing, R.T. (2000) Niche versus chance and tree diversity in forest gaps. *Trends in Ecology and Evolution* **15**: 183–188.

Brown, N. (1993) The implication of climate and gap microclimate for seedling growth conditions in a Bornean lowland rain forest. *Journal of Tropical Ecology* **9**: 153–168.

Burgess, P.F. (1970) An approach towards a silvicultural system for the hill forests of the Malay Peninsula. *Malaysian Forester* **33**: 126–134.

Burgess, P.F. (1975) Silviculture in the hill forests of the Malay Peninsula. Malaysian Forest Department Research Pamphlet 66, Kuala Lumpur, Malaysia.

Cedergren, J. (1996) A silvicultural evaluation of stand characteristics, pre-felling climber cutting and directional felling in a primary dipterocarp forest in Sabah, Malaysia. Acta Universitatis Sueciae, Silvestria 9, Doctoral thesis, Swedish University of Agricultural Sciences, Umeå, Sweden.

Clearwater, M.J., Nifinluri, T. & van Gardingen, P.R. (1999) Growth response of wild *Shorea* seedlings to high light intensity. In: *Management of Secondary and Logged-over Forests in Indonesia*, ed. P. Sist, C. Sabogal & Y. Byron, pp. 54–64. Bogor, Indonesia: CIFOR.

Curran, L.M. & Leighton, M. (2000) Vertebrate responses to spatiotemporal variation in seed production of mast-fruiting Dipterocarpaceae. *Ecological Monographs* **70**: 101–127.

Curran, L.M., Caniago, I., Paoli, G.D., Astianti, D., Kusneti, M., Leighton, M., Nirarita, C.E. & Haeruman, H. (1999) Impact of El Niño and logging on canopy tree recruitment in Borneo. *Science* **286**: 2184–2188.

Dawkins, H.C. & Philip, M.S. (1998) *Tropical Moist Forest Silviculture and Management. A History of Success and Failure*. Wallingford, UK: CAB International.

Dennis, R. (1999) *A Review of Fire Projects in Indonesia (1982–1998)*. Bogor, Indonesia: CIFOR: 105 pp.

Dwiprabowo, H., Grulois, S., Sist, P. & Kartawinata, K. (2002) Cost-benefit analysis of reduced-impact logging in a lowland dipterocarp Forest of Malinau, East Kalimantan. In: *Forest, Science and Sustainability: The Bulungan Model Forest*, Technical Report, Phase I (1997–2001), ITTO Project PD 12/97 Rev.1 (F), pp. 39–55. Bogor, Indonesia: CIFOR, MOF and ITTO.

Dykstra, D. & Heinrich, R. (1996) *FAO Model Code of Forest Harvesting Practice*. Rome, Italy: FAO: 85 pp.

Elias (1998) Reduced-impact timber harvesting in the tropical natural forest of Indonesia. FAO Forest Harvesting Case 11, FAO, Rome, Italy.

FAO (2001) *State of the World's Forests*. Rome, Italy: FAO.

Favrichon, V. & Young Cheol, K. (1998) Modeling the dynamics of a lowland mixed dipterocarp forest stand: application of a density-dependent matrix model. In: *Silvicultural Research in a Lowland Mixed Dipterocarp Forest of East Kalimantan, the Contribution of STREK Project*, ed. J.-G. Bertault & K. Kadir, pp. 229–246. Montpellier, France: Cirad-Forêt, FORDA, INHUTANI I.

Fimbel, R.A., Grajal, A. & Robinson, J.G., eds. (2001) *The Cutting Edge: Conserving Wildlife in Managed Tropical Forest*. New York, USA: Columbia University Press.

FLB (2002) Revising the silviculture system? *Forests Initiatives Newsletter* **10**: 4–5.

Fox, J.E.D. (1973) Dipterocarp seedling behavior in Sabah. *Malaysian Forester* **36**: 205–214.

Ghazoul, J. & Hill, J. (2001) The impacts of selective logging on tropical forest invertebrates. In: *The Cutting Edge: Conserving Wildlife in Managed Tropical Forest*, ed. R.A. Fimbel, A. Graja & J.G. Robinson, pp. 251–278. New York, USA: Columbia University Press.

Ghazoul, J., Liston, K.A. & Boyle, T. (1998) Disturbance-induced, density dependent seed selection in *Shorea siamensis* (Dipterocarpaceae), a tropical forest tree. *Journal of Ecology* **86**: 462–473.

Grieser Johns, A. (1997) *Timber Production and Biodiversity*

Conservation in Tropical Rain Forests. Cambridge, UK: Cambridge University Press.

Holmes, T.P., Blate, G.M., Zweede, J.C., Rodrigo Pereira, Jr, R., Barreto, P., Boltze, F. & Bauch, R. (2002) Financial and ecological indicators of reduced impact logging performance in the eastern Amazon. *Forest Ecology and Management* **163**: 93–110.

Huth, A. & Ditzer, T. (2001) Long-term impacts of logging in a tropical rain forest – a simulation. *Forest Ecology and Management* **142**: 33–51.

Ickes, K., Dewalt, S.J. & Appanah, S. (2001) Effects of native pigs (*Sus scrofa*) on woody understorey vegetation in Malaysian lowland rain forest. *Journal of Tropical Ecology* **17**: 191–206.

Jennings, S.B., Brown, N.D., Boshier, D.H., Whitmore, T.C. & do C.A. Lopes, J. (2001) Ecology provides a pragmatic solution to the maintenance of genetic diversity in sustainably managed tropical rain forests. *Forest Ecology and Management* **154**: 1–10.

Johnson, N. & Cabarle, B. (1993) *Surviving the Cut: Natural Forest Management in the Humid Tropics*. Washington DC, USA: World Resources Institute.

Kato, M. (1996) Plant-pollinator interactions in the understory of a lowland mixed dipterocarp forest in Sarawak. *American Journal of Botany* **83**: 732–743.

Kitamura, K., Rahman, M.Y.B.A., Ochai, Y. & Yoshimaru, H. (1994) Estimation of the outcrossing rate on *Dryobalanops aromatica* Gaertn.f. in primary and secondary forests in Brunei, Borneo, Southeast Asia. *Plant Species Biology* **9**: 37–41.

Kleine, M. & Heuveldop, J. (1993) A management planning concept for sustained yield of tropical forests in Sabah, Malaysia. *Forest Ecology and Management* **61**: 277–297.

Kuusipalo, J., Jafarsidik, Y., Adjers, G. & Tuomela, K. (1996) Population dynamics of tree seedlings in a mixed dipterocarp rainforest before and after logging and crown liberation. *Forest Ecology and Management* **81**: 85–94.

Laumonier, Y. & Legg, C. (1998) Le suivi des feux de 1997 en Indonésie. *Bois et Forêts des Tropiques* **258**: 5–18.

Lee, S.S. (1998) Root symbiosis and nutrition. In : *A Review of Dipterocarps, Taxonomy, Ecology and Silviculture*, ed. S. Appanah & J.M. Turnbull, pp. 100–114. Bogor, Indonesia: CIFOR.

Lee, S.L. (2000) Mating system parameters of *Dryobalanops aromatica* Gaertn. f. (Dipterocarpaceae) in three different forest types and a seed orchard. *Heredity* **85**: 338–345.

Martini, A.M.Z., de la Rosa, N. & Uhl, C. (1994) An attempt to predict which Amazonian tree species may be threatened by logging activities. *Environmental Conservation* **21**: 152–162.

Momose, K., Nagamitsu, T. & Inoue, T. (1996) The reproductive biology of an emergent dipterocarp in a lowland rain forest in Sarawak. *Plant Species Biology* **11**: 189–198.

Momose, K., Yumoto, T. Nagamitsu, T., Kato, M., Nagamasu, H., Sakai, S., Harrison, R.D., Itioka, T., Hamid, A.A. & Inoue, T. (1998) Pollination biology in a lowland dipterocarp forest in Sarawak, Malaysia. I. Characteristics of the plant-pollinator community in a lowland dipterocarp forest. *American Journal of Botany* **85**: 1477–1501.

Murawski, D.L. & Hamrick, J.L. (1990) Mating systems of two bombacaceous trees of a neotropical moist forest. *Oecologia* **82**: 501–506.

Murawski, D.L. & Hamrick, J.L. (1991) The effect of density of flowering individuals on the mating systems of nine tropical tree species. *Heredity* **67**: 167–174.

Nathan, R., Katul, G.G., Horn, H.S., Thomas, S.M., Oren, R., Vissar, R.A., Pascala, S.W. & Levin, S.M. (2002) Mechanisms of long-distance dispersal of seeds by wind. *Nature* **418**: 409– 413.

Nguyen-Thé, N. & Sist, P. (1998) Phenology of some dipterocarps. In : *Silvicultural Research in a Lowland Mixed Dipterocarp Forest of East Kalimantan, the Contribution of STREK Project*, ed. J.-G. Bertault & K. Kadir, pp. 95–110. Montpellier, France: Cirad-Forêt, FORDA, INHUTANI I.

Nicholson D.I. (1979) The effects of logging and treatment on the mixed dipterocarp forests of Southeast Asia. Report MISC/79/8, FAO, Rome, Italy: 65 pp.

Obayashi, K., Tsumura, Y., Ihara, U.T., Niiyama, K., Tanouchi, H., Suyama, Y., Washitani, I., Lee, C.T., Lee, S.L., & Muhammad, N. (2002) Genetic diversity and outcrossing rate between undisturbed and selectively logged forests of *Shorea curtisii* (Dipterocarpaceae) using microsatellite DNA analysis. *International Journal of Plant Sciences* **163**: 151–158.

Ong, R., & Kleine, M. (1995) DIPSIM. A dipterocarp forest growth simulation model for Sabah. Forest Research Center Research Paper 2, Forest Department, Sabah, Malaysia: 94 pp.

Parren, M. & Bongers, F. (2001) Does climber cutting reduce felling damage in southern Cameroon? *Forest Ecology and Management* **141**: 175–188.

Pascal, J.-P. & Pélissier, R. (1996) Structure and floristic composition of a tropical evergreen forest in south-west India. *Journal of Tropical Ecology* **12**: 191–214.

Perez-Salicrup, D.R. (1998) Effects of liana cutting on trees and tree seedlings in a tropical forest in Bolivia. Ph.D. Dissertation, Department of Biology, University of Missouri, St Louis, USA.

Pinard, M.A. & Putz, F.E. (1996) Retaining forest biomass by reducing logging damage. *Biotropica* **28**: 278–295.

Pinard, M.A., Barker, M.G. & Tay. J. (2000) Soil disturbance and post-logging forest recovery on bulldozer paths in Sabah, Malaysia. *Forest Ecology and Management* **130**: 213–225.

Pinard, M.A., Putz, F.E., Rumìz, D., Guzmàn, R. & Jardim, A. (1999) Ecological characterization of tree species for guiding forest management decisions in seasonally dry forests in Lomerio, Bolivia. *Forest Ecology and Management* **113**: 201–213.

Pinard, M.A., Putz, F.E., Tay, J. & Sullivan, T. (1995) Creating timber harvesting guidelines for a reduced-impact logging project in Malaysia. *Journal of Forestry* **93**: 41–45.

Poore, D. & Chiew, T.H. (2000) Review of Progress towards the Year 2000 Objective. ITTO Technical Report, ITTC (XXVIII)/9/Rev.2.

Poore, M.E.D. (1968) Studies in Malaysian rain forest. I. The forest on Triassic sediments in Jengka Forest Reserve. *Journal of Ecology* **56**: 143–196.

Prabhu, R., Colfer, C.J.P. & Dudley, R.G. (1999) *Guidelines for Developing, Testing, and Selecting Criteria and Indicators for Sustainable Forest Management*. The Criteria and Indicators Toolbox Series 1. Bogor, Indonesia: CIFOR.

Putz, F.E. (1991) Silvicultural effects of lianas. In: *The Biology of Vines*, ed. F.E. Putz & H.A. Mooney, pp. 439–501. Cambridge, UK: Cambridge University Press.

Putz, F.E., Redford, K.H., Robinson, J.G., Fimbel, R. & Blate, G.M. (2000) Biodiversity conservation in the context of tropical forest management. The World Bank Environment Department, Biodiversity Series Impact Studies Paper 75. The World Bank, Washington DC, USA.

Richards, P.W. (1996) *The Tropical Rain Forest*. Second edition. Cambridge, UK: Cambridge University Press.

Riera, B. & Alexandre, D.Y. (1988) Surface des chablis et temps de renouvellement en forêt tropicale. *Acta Oecologica* **9**: 211–220.

Sabogal, C., Silva, J.N.M., Zweede, J., Pereira Jr, R.P. & Barreto, P. (2000) Diretrizes Técnicas para exploração de impacto reduzido em operações florestais de terra firme na Amazônia Brasileira. Embrapa Amazônia Oriental, Documentos 64, Belém, Brazil.

Sakai, S., Momose, K., Yumoto, T., Kato, M. & Inoue, T. (1999) Beetle pollination of *Shorea parvifolia* (section Mutica, Dipterocarpaceae) in a general flowering period in Sarawak, Malaysia. *American Journal of Botany* **86**: 62–69.

Sanford, R.L., Jr, Braker, E.H. & Hartshorn, G.S. (1986) Canopy opening in a primary neotropical lowland forest. *Journal of Tropical Ecology* **2**: 277–282.

Sheil, D. (2002) Biodiversity research in Malinau. In: *Forest, Science and Sustainability: The Bulungan Model Forest*, Technical Report, Phase I (1997–2001), ITTO Project PD 12/97 Rev.1 (F), pp. 55–107. Bogor, Indonesia: CIFOR, MOF and ITTO.

Sheil, D., Liswanti, N., van Heist, M., Basuki, I., Syaefuddin, Samsoedin, I., Rukmiyati & Agung, M. (2003) Local priorities and biodiversity in tropical forest landscapes: asking people what matters. *Tropical Forest Update* (in press) [www document]. URL http://www.itto.or.jp

Sheil, D. & Van Heist, M. (2000) Ecology for tropical forest management. *International Forestry Review* **2**: 261–270.

Sist, P. (2000) Reduced-impact logging in the tropics: objectives, principles and impacts. *International Forestry Review* **2**: 3–10.

Sist, P. & Nguyen-Thé, N. (2002) Logging damage and the subsequent dynamics of a dipterocarp forest in East Kalimantan (1990–1996). *Forest Ecology and Management* **165**: 85–103.

Sist, P. & Saridan, A. (1999) Stand structure and floristic composition of a primary lowland dipterocarp forest in East Kalimantan. *Journal of Tropical Forest Science* **11**: 704–722.

Sist, P., Dykstra, D.P. & Fimbel, R. (1998*a*) Reduced-impact logging guidelines for lowland and hill dipterocarp forests in Indonesia. CIFOR Occasional Paper 15, Bogor, Indonesia: 19 pp.

Sist, P., Nolan, T., Bertault, J.-G. & Dykstra, D.P. (1998*b*) Harvesting intensity versus sustainability in Indonesia. *Forest Ecology and Management* **108**: 251–260.

Sist, P., Picard, N. & Bertault, J.-G. (2002) Why minimum diameter cutting alone cannot fit with RIL objectives. In: Proceedings of the International Conference on the Application of Reduced Impact Logging to Advance Sustainable Forest Management: Constraint, Challenges and Opportunities, 26 February–1 March 2001, Sarawak, Malaysia.

Sist, P., Picard, N. & Gourlet-Fleury, S. (2003*a*) Sustainable cutting cycle and yields in a lowland mixed dipterocarp forest of Borneo. *Annals of Forest Science* (in press).

Sist, P., Sheil, D., Kartawinata K. & Priyadi H. (2003*b*) Reduced-impact logging and high extraction rates in mixed dipterocarps forests of Borneo: the need of new silvicultural prescriptions. *Forest Ecology and Management* **179**: 415–427.

Soepadmo, E. (1995) Plant diversity of the Malesian tropical rainforest and its phytogeographical and economic significance. In: *Ecology, Conservation and Management of South East Asian Rainforests*, ed. R.B. Primack & T.E. Lovejoy, pp. 19–40. USA: Yale University Press.

Srivastava, P.B.L. (1977) Precocious flowering (paedogenesis) in Dipterocarpaceae. *Malaysian Forester* **40**: 251–253.

Swaine, M.D. & Whitmore, T.C. (1988) On the definition of ecological species groups in tropical rain forests. *Vegetatio* **75**: 81–86.

Tamari, C. & Jacalne, D.V. (1984) Fruit dispersal of dipterocarps. *Bulletin of the Forest and Forest Product Research Institute* **325**: 127–140.

Tuomela, K., Kuusipalo, J., Vesa, L., Nuryanto, K., Sagal, A.P.S., &. Adjers, G. (1996) Growth of dipterocarps seedlings in artificial gaps: an experiment in a logged-over rainforest in South Kalimantan, Indonesia. *Forest Ecology and Management* **81**: 95–100.

Van Gardingen, P.R., Clearwater, M.J., Nifinluri, T., Effendi, R., Ruswantoro, P.A., Ingleby, K. & Munro, R.C. (1998) Impacts of logging on the regeneration of lowland dipterocarps forest in Indonesia. *International Forestry Review* **77** : 71–82.

Whitmore, T.C. (1984) *Tropical Rain Forests of the Far East*. Second edition. Oxford, UK: Clarendon Press: 352 pp.

Whitmore, T.C. (1990) Tropical rain forest dynamics and its implications for management. In: *Rain Forest Regeneration and Management*, ed. A. Gomez-Pompa, T.C. Whitmore & M. Hadley, pp. 67–89. Man and the Biosphere Series, Volume 6. Paris, France: Parthenon Publishing Group.

Whitmore, T.C. & Brown, N.D. (1996) Dipterocarp seedling growth in rain forest canopy gaps during six and a half years. *Philosophical Transactions of the Royal Society of London B* **351**: 1195–1203.

Wickneswari, R. & Boyle, T. (2000) Effects of logging and other forms of harvesting on genetic diversity in humid tropical forests. In: *Forest Conservation Genetics, Principles and Practices*, ed. A.Young, D.Boshier & T.Boyle, pp. 115–122. Wallingford, UK: Cabi Publishing.

Wyatt-Smith, J. (1963) Manual of Malayan silviculture for inland forest. Malayan Forest Record 23, Forest Department, Kuala Lumpur, Malaysia.

Wyatt-Smith, J. (1987) The management of tropical moist forest for the sustained production of timber: some issues. IUCN/IIED Tropical Forestry Policy Paper 4.