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Retaining Forest Biomass By Reducing Logging Damage[1]

Michelle A. Pinard[2]

Department of Botany, University of Florida, 220 Bartram Hall, Gainesville, Florida 32611-8526, U.S.A.

and

Francis E. Putz[3]

Center for International Forestry Research (CIFOR), P.O. Box 6596, JKPWB, Jakarta 10065, Indonesia

ABSTRACT

Global concern over rising atmospheric concentrations of carbon dioxide is stimulating development and implementation of policies aimed at reducing net greenhouse gas emissions by enhancing carbon sinks. One option for reducing net emissions is to lessen damage to residual forests during selective logging, thereby retaining additional carbon in biomass. A pilot carbon offset project was initiated in Sabah, Malaysia, in 1992 in which a power company provided funds to a timber concessionaire to implement guidelines aimed at reducing logging damage; in doing so, the utility gained potential credit towards future emissions reduction requirements. To quantify the carbon retained due to this effort, we compared dipterocarp forests logged according to reduced-impact logging guidelines with forests logged by conventional methods in terms of the above- and below-ground biomass both before and after logging. Prior to logging, the forest stored approximately 400 Mg biomass ha^{-1}, 17 percent of which was belowground. High volumes of timber were removed from both of the logging areas (mean CNV = 154, RIL = 104 m^3ha^{-1}). Forty-one percent of the unharvested trees <60 cm DBH were severely damaged (uprooted and crushed) from logging in conventional logging areas in contrast to 15 percent in reduced-impact logging areas. Approximately 18 and 12 percent, respectively, of the remaining residual trees in conventional and reduced-impact logging areas suffered less severe damage (*e.g.*, crown or bark damage). Mortality rates of the less severely damaged trees in all DBH classes were higher during the first year in conventional logging areas than in reduced-impact logging areas. One yr post harvest, conventional and reduced-impact logging areas contained biomass equivalent to about 44 percent and 67 percent of pre-logging levels, respectively. Approximately 62 percent of the difference in carbon retention was due to fewer trees killed in the reduced-impact logging areas; the remaining 38 percent was due to a lower mass of branches, stumps and waste wood from felled trees in reduced-impact logging areas. Mortality of damaged trees in both areas may contribute to net decreases in biomass for several years after logging. More and larger trees remained undamaged where reduced-impact logging was practiced, hence future biomass increment and yields of marketable timber are expected to be greater in the reduced-impact logging areas than in conventional logging areas.

SEDUTAN

Kebimbangan sejagat yang semakin meningkat mengenai ketebalan karbon dioksida di atmosfera telah merangsang pembangunan dan pelaksanaan polisi-polisi yang bertujuan mengurangkan pengeluaran gas "rumah hijau" dan menambah penyerapan karbon. Satu cara meningkatkan pengasingan karbon dioksida adalah dengan mengurangkan kesesakan pada pentas balak semasa membalakpilih yang akan mengekalkan karbon dalam biomass hutan. Satu projeck karbon ofset pertama tolah dimulakan di Malaysia pada tahun 1992. Sebuah syarikat tenaga membekalkan dana kepada sebuah pemegang konsesi balak bagi melaksanakan garispanduan yang bertujuan mengurangkan kerosakan pembalakan dan dengan itu, kebergunaan ini mendapat penghargaan dalam usaha pengurangan keluaran gas karbon masa depan. Bagi menyatakan kuantiti karbon yang dikekalkan disebabkan usaha ini, kami relah membanding hutan yang dibalak mengikut garispanduan pembalakan kesan berkurangan (RIL) dengan pembalakan konvensional dari segi biomass atas dan bawah tanah sebelum dan selepas pembalakan. Sebelum pembalakan, hutan dipterocarp menakung lebih kurang 400 Mg biomass ha^{-1}, 17 peratus adalah bawah tanah. Isipadu balak yang tinggi telah dikeluarkan dari dedua-kedua kawasan balak (min = 129 m^3 ha^{-1}). Empat puluh satu peratus pokok tidak dibalak (atau bakinya) <60 cm DBH telah mengalami kerosakan teruk (tercabut dan musnah) menggunakan kaedah konvensional berbanding 15 peratus di kawasan pembalakan kesan berkurangan (RIL). Kadar mortaliti pokok-pokok yang mengalami kerosakan yang sikit adalah lebih tinggi pada tahun pertama di kawasan pembalakan cara konvensional berbanding kawasan pembalakan kesan berkurangan. Setahun selepas pembalakan, kawasan-kawasan konven-

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[2] Mailing address: Tropical Research and Development, Inc., 7001 S. W. 24th Avenue, Gainesville, Florida 32607, U.S.A.

[3] Mailing address: Department of Botany, University of Florida, 220 Bartram Hall, Gainesville, Florida 32611-8526, U.S.A.

sional mengandungi biomass bersamaan 44 peratus paras pra-pembalakan. Disebaliknya kawasan-kawasan pembalakan kesan berkurangan mengandungi biomass adalah 67 peratus paras pra-pembalakan. Lebih kurang 62 peratus perbezaan dalam penakungan karbon akibat berkurangnya insiden kerosakan teruk pada kawasan pembalakan kesan berkurangan (RIL), yang tertinggal adalah dahan-dahan, tunggul dan kayu buangan daripada pokok yang telah dibalak. Kadar mortaliti pokok yang rosak dikedua-dua kawasan akan menyumbang kepada pengurangan bersih dalam biomass untuk beberapa tahun selepas pembalakan. Mamandangkan lebih banyak pokok yang tidak mangalami kerosakan hasil pembalakan kesan berkurangan berbanding kawasan pembalakan konvensional, peningkatan biomass dan hasil balak yang boleh dipasarkan dijangka bertambah di kawasan pembalakan kesan berkurangan (RIL).

Key words: *biomass; carbon offsets; carbon storage; conservation; dipterocarp; harvesting; logging damage; Malaysia; tropical forests.*

UNCONTROLLED LOGGING AND RISING ATMOSPHERIC CONCENTRATIONS of "greenhouse" gases are distinct problems with somewhat overlapping solutions. Many logging operations in the tropics involve unregulated and unsupervised selective cutting; though only a small proportion of the trees are harvested, a large proportion of the forest is damaged (*e.g.*, Johnson & Cabarie 1993). Heavily damaged residual forests yield little timber and thus are at high risk of conversion to other types of land use. Open canopies and heavy vine loads, typical of many heavily logged forests, increase forest vulnerability to fire and further degradation (*e.g.*, Uhl & Buschbacher 1985). Appropriate timber harvesting methods exist (*e.g.*, FAO 1980) but incentives to implement better practices are lacking in many countries (Gillis & Repetto 1988). Policies aimed at reducing greenhouse gas emissions may provide a financial incentive for better logging.

In 1992, 52 nations signed a resolution to adopt policies to mitigate climate change by limiting emissions and enhancing greenhouse gas sinks and reservoirs (UNCED 1992). The resolution supported cost-effective approaches to reducing net emissions and recommended cooperation between nations such as in joint implementation programs that allow greenhouse gas emissions in one nation to be offset by reduced emissions or increased sequestration in another.

A wide range of opportunities exist for carbon offset programs in forestry. For example, estimates of the potential for carbon sequestration have been published for the following activities: preserving old growth forests (Harmon *et al.* 1990), controlling forest fires (Faeth *et al.* 1994), creating plantations and reforesting degraded lands (Schroeder 1992), increasing rotation times in plantations (Cropper & Ewel 1987, Hoen & Solberg 1994) and reducing logging damage (Putz & Pinard 1993). Forestry-based offsets increase terrestrial carbon storage either by expanding forest cover or by maintaining or improving existing forest for carbon storage. This paper explores the potential for increasing carbon retention in managed forests by reducing avoidable logging damage. By improving harvesting practices, fewer trees are killed or damaged during logging and more carbon remains in the forest in living trees. If residual stands contain more trees of larger diameter than areas conventionally logged, future yields of timber are also likely be higher.

Techniques for controlling logging damage are well-known (*e.g.*, Hendrison 1990) and can be readily applied by trained field staff (*e.g.*, Marn *et al.* 1981). Competent supervision of operations can result in improved efficiency, thus lower harvesting costs (Marn & Jonkers 1981). The environmental benefits of reduced-impact logging go beyond carbon retention and include soil and nutrient conservation (Kasran 1988, Shariff *et al.* 1989), water retention (Nik & Harding 1992), maintenance of forest structure (Hendrison 1990), and preservation of existing biodiversity. Sustainable forest management is based on sound silviculture, a component of which is proper harvesting.

In this paper, we describe the results of a reduced-impact logging project in dipterocarp forest in Sabah, Malaysia. We have three objectives for this paper. The first objective is to describe forest biomass stores both before and after logging. The second is to compare logging damage in forest logged by conventional methods and in forest logged according to reduced-impact logging harvesting guidelines. The third objective is to quantify the carbon retained in biomass due to implementation of the harvesting guidelines.

BACKGROUND INFORMATION ON THE REDUCED-IMPACT LOGGING PROJECT

Commercial forests in Sabah are selectively logged (*e.g.*, Kleine & Heuveldop 1993) with all mature trees (>60 cm DBH) of commercial species felled during the first harvest. Trees in the Dipterocarpaceae represent 90 percent of the total volume of commercial timber extracted (Sabah Forestry De-

TABLE 1. *Reduced-impact logging planning and harvesting guidelines (condensed from Rakyat Berjaya-Innoprise Corporation, unpublished document, version 4; road specifications not included).*

Harvest plan	Formal plan to be prepared based on stock map (1:5000 scale) showing locations of commercial trees, proposed roads, skid trails, stream crossings, buffer zones, restricted or excluded areas, rocky or otherwise inaccessible areas, and logging unit boundaries. Stream buffer zones to be demarcated on all streams >5 m between banks; buffer zone width varies with stream width. All trees to be felled are to be marked with record number and with a vertical paint blaze to indicate the direction of fall. Potential crop trees of good form and >20 cm DBH to be paint marked with a blue ring if they are at risk of being damaged from felling or skidding.
Pre-felling vine cutting	All vines >2 cm DBH to be severed at least 12 mo prior to harvesting; vines in riparian reserves, buffer zones or excluded areas, figs, and root climbers are not to be cut.
Skid trail planning	Skid trails to be located on ridges and designed to minimize skidding distances, skidding on steep slopes, skidding downhill, and stream crossings. Skid trails to be marked in field prior to harvesting and their endings clearly marked. Side cutting is permitted only on slopes >20°. Construction of skid trails that cannot easily be drained is to be avoided.
Tree felling	Decisions on felling directions to be based on safety to feller, ease of skidding, and avoidance of damage to harvested tree and potential crop trees. Trees to be felled within 10° of indicated direction. Fellers to be trained in use of wedges and equipped with them.
Skidding	Use of the bulldozer's blade is permitted only during skid trail construction on slopes >15°. Skid trail gradients not to exceed 20° except over short distances. Skidding should not occur on slopes >35°.
Landings	Where possible all log landing operations to be carried out on existing roads. Where required, landings to be located on ridges and are not to exceed 0.18 ha.
Closing operations	Roads and skid trails to be drained, temporary stream crossing structures to be removed, and landings should be reshaped to assure adequate drainage. Available logging debris from perimeter to be redistributed on landing surface.

partment 1989). In a typical logging operation in Sabah, logs are skidded to the roadside or log landing (flat, cleared area for storing logs) by bulldozers. On average 8–15 trees are felled per ha, representing 50–120 m^3 of timber. Damage to the forest is extensive; as much as 30–40 percent of the area is traversed by bulldozers (Chai 1975, Jusoff 1991) and 40–70 percent of the residual trees are damaged (Fox 1968, Nicholson 1979). Typically, little pre-harvest planning is carried out and the activities of fellers and bulldozer operators are not well-coordinated.

Current forest management practices in Sabah are not sustainable because the volumes of timber extracted, the area logged each year, and damage to advanced regeneration are all too high (Sabah Forestry Department 1989). A new silvicultural system is clearly needed in Sabah and is presently under development by the Sabah Forestry Department (Kleine & Heuveldop 1993, Udarbe *et al.* 1994). As is true for many tropical countries, however, lack of forestry department staff and difficulties in enforcing regulations over large and dispersed tracts of forest can render even the best regulations ineffective (Jabil 1983). Programs that provide concession holders with incentives for better management practices may help stimulate change in the industry.

In 1992, the Reduced-Impact Logging (RIL) Project was established between Innoprise Corporation, a timber concessionaire in Sabah, Malaysia, and New England Electric system, a coal-burning utility in Massachusetts, U.S.A. New England Electric provided funds to Innoprise for training staff and implementing harvesting guidelines (Table 1) aimed at reducing logging damage in 1400 ha of their concession (total concession area is ≈ 1 million ha with annual logging of about 20,000 ha). The carbon retained in the forest due to these efforts could be claimed by the utility as a carbon offset. Contemporary conventional selective logging practices in the area provide the baseline for comparison.

The 1400–ha experimental area dedicated to the project is divided among two commercial forest reserves in southeastern Sabah, a 450–ha tract in Ulu Segama Forest Reserve (5°0′N, 117°30′E, 150–

750 m a.s.l.) and a 950–ha tract in Kalabakan Forest Reserve (4°25'N, 117°29'E, 150–900 m a.s.l.). This paper is based on data from Ulu Segama only. The project began in May 1992 when woody vines were cut in Ulu Segama; logging is expected to be completed in the second tract, Kalabakan, by December 1995. The logging crews and forest rangers working in the experimental area were trained by foresters from the Queensland Forest Service and by expert fellers from Sweden. The harvesting guidelines (Table 1) were based on best management practices recommended in Indonesia, Malaysia, and Australia. (For more details on implementation of the project and development of the harvesting guidelines see Pinard *et al.* 1995).

STUDY SITE

The experimental area in Ulu Segama supports dipterocarp hill forest, spectacular both for its stature and its high density of big trees. Prior to logging, canopy height averages ≈45 m, but emergent trees reach heights of 70 m. The terrain consists of series of steep ridges; over 75 percent of the area occurs on slopes exceeding 20° and generally <200–300 m long (Pinard 1995). Soils are varied but primarily are Ultisols derived from Tertiary sediments (Ohta & Effendi 1992). Mean annual rainfall is approximately 2700 mm and mean daily temperature is 26.7°C (Danum Valley Field Centre Records, 1986–1993).

METHODS

PRE-LOGGING MEASUREMENTS.—Forest biomass and stand structure before logging were measured to allow comparison of the effects of logging treatment on carbon stores. Prior to logging, four logging units (30–50 ha each) were randomly selected from the experimental area to be logged according to the reduced-impact logging guidelines (hereafter RIL units); four additional units were randomly selected from an adjacent area destined to be logged conventionally (Fig. 1). Units logged conventionally or by the RIL guidelines were paired according to topography and logging schedule to reduce variability of logging impacts on the residual stand due to differences in soil moisture content and slope. The conventional logging units were harvested by crews not involved in the RIL project. A crew that was trained with funds from the power company and was experienced with directional felling and proper log extraction techniques harvested the RIL units according to the RIL guidelines.

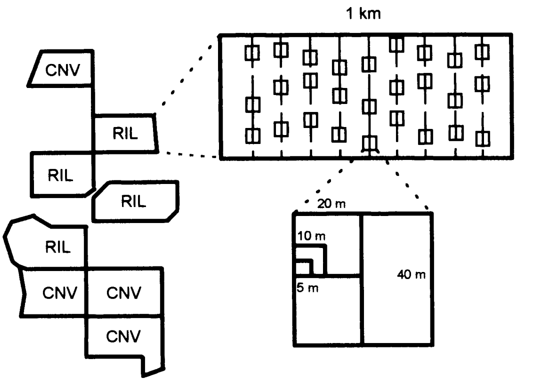

FIGURE 1. In eight randomly selected logging units (30–60 ha each; RIL- units logged according to the RIL guidelines, CNV- units logged conventionally), permanent plots (1600 m^2) were established to sample trees and woody vines. Stems were tagged and measured in nested plots as follows: 1600 m^2, trees ≥60 cm DBH, 800 m^2, stems 20–40 cm DBH; 400 m^2, stems 10–20 cm DBH; 100 m^2, stems 5–10 cm DBH; 25 m^2, trees 1–5 cm DBH, vines 2–5 cm DBH.

EXPERIMENTAL DESIGN.—Within each unit twenty to thirty-five 1600 m^2 plots (40 × 40 m for six units or 20 × 80 m for two units, approximately 10 percent of each logging unit area) were located according to a stratified random design (Fig. 1), avoiding areas within 20 m of permanent streams, within 10 m of a logging unit boundary or a main road, steep rocky areas (slopes >45°), and landslides. In the eight logging units, a total of 216 plots were established. No plots were established at 49 points dismissed due to exclusions listed above.

ABOVEGROUND BIOMASS.—All trees >60 cm diameter in each plot were tagged and their diameter measured at 1.3 m or above buttresses (hereafter, DBH). Nested subplots were used for smaller trees and lianas (Fig. 1). All commercial trees tagged in the plots were identified to species or timber species group. Stem and bark damage were described and any other tree characteristics that might be mistaken for logging damage were noted. Lianas were tagged and measured only in the four units to be logged conventionally because most of the lianas were cut prior to plot establishment in the units to be logged according to the reduced-impact logging guidelines.

Aboveground tree biomass was estimated allometrically using tree inventory data and stem volume–diameter–height relations calculated for 15 local species groups in the Ulu Segama Forest Reserve (Forestal International Ltd. 1973) and a Biomass Expansion Factor (BEF) developed for "good" hill dipterocarp forest in West Malaysia (see Brown *et al.* 1989). The BEF for good hill forest was selected over the factor developed for other Malaysian dipterocarp forest types because the basal area for good hill forest (28.5 m^2 ha^{-1} for trees > 15 cm DBH) most closely matched that of the study site. Wood densities were available for 120 of the 124 species or species groups recorded in the plots (Burgess 1966). To convert wood densities determined at 12 percent moisture content (air-dry weight) to density at dry weight, we applied a regression developed by Reyes *et al.* (1992). For non-dipterocarp species whose wood density was not known, we used the arithmetic mean of the known non-dipterocarp species (0.503 g cm^{-3}, $N = 48$ species). For dipterocarp species whose wood density was not known, we used the mean value of the known species within that genus (or section of the genus, when applicable). Biomass of lianas >2 cm DBH was estimated from basal area using a regression equation developed for Venezuelan liana species (Putz 1983).

To supplement the available stem volume equations, which we judged were inappropriate for trees <10 cm DBH, we harvested 40 randomly selected trees 1–10 cm DBH representing a mixture of species and determined their aboveground biomass. Our sampling was conducted in primary forest within one km of our study sites. Tree biomass was regressed against DBH. To determine total small tree biomass, we applied the DBH-biomass equation to the trees (1–10 cm DBH) in the permanent plots.

Shrub, herb, palm, and herbaceous vine biomass was measured in three RIL and three conventional logging units using 1 m^2 circular clip plots ($n = 15$ per unit, $N = 45$ per treatment) located in a stratified random fashion to include the range of topographical conditions. Each plot was considered a sample and logging unit divisions were assumed to be inconsequential to the estimate. In the clip plots, all above-ground plant biomass (<1 cm diameter at base) was cut at the soil surface, weighed and then a subsample oven-dried at 70°C to constant mass. For self-supporting species, only plants rooted inside plots were included. For vines, all stems and leaves occurring over the clip plots were collected regardless of the rooting site. Palms (primarily stemless rattans) that occurred in the plots were also clipped and collected.

A conversion factor of 50 percent is frequently used to estimate carbon content of plant tissues (*e.g.*, Hoen & Solberg 1994). To determine whether woody tissue in our site was similarly 50 percent carbon, we tested a small number of wood samples randomly collected from fresh logging debris for carbon content. Twenty samples, approximately 45 cm^3 each, were collected from log and branch debris. The samples were split into small pieces, oven-dried, ground and sieved. Carbon content was determined using a Carlo-Erba NA 1500 Carbon-Nitrogen Elemental Analyzer (Isotope Ratio Mass Spectrometer, Department of Soil Science, University of Florida, Gainesville, FL). Carbon content averaged 49.2 percent (SD = 1.1%, $N = 20$; statistically different from 50%, $t = 3.55$, df = 19, $P < 0.005$). We assume all plant tissues to be 49.2 percent carbon by dry weight, though we recognize that certain tissues often have carbon contents that are above or below this percentage (*e.g.*, seeds and fine roots, respectively; Golley 1969, Williams 1986).

BELOWGROUND BIOMASS.—Pre-logging root biomass was sampled in the eight logging units using a stratified random design, traversing terrain typical for each unit ($n = 10$ pits per unit, total $N = 40$ pits per treatment, logging unit divisions were disregarded in the analyses). Coarse roots (>5 mm diameter) were sampled in 50 × 50 cm monoliths of soil extracted to 50 cm with a sledge-driven flat blade. Roots were separated from the soil in the field, washed, live and dead roots separated, and sorted into four diameter classes in the lab (5–15, 15–50, 50–150, and >150 mm diameter); live roots were weighed and subsampled for dry weight determination. Dead roots were weighed and subsampled in a subset of the samples ($N = 56$). We did not sample deeper than 50 cm in the soil profile for coarse roots and consequently underestimated carbon stored in coarse roots. Fine root (<5 mm diameter) mass was estimated using 50 mm diameter cores taken to 10 cm depth; two cores were taken at each sampling site, combined, soaked in water, and agitated. Roots were then separated from soil, oven dried, and weighed. Due to difficulties in confidently differentiating live and dead roots, only total fine root mass values are reported. A proportion of the early samples were not included as only live roots were dried and weighed.

To determine coarse root biomass directly beneath trees where core sampling was impractical (hereafter, butt roots), 14 partially uprooted trees (20–130 cm DBH) along roadsides and skid trails within the 1993 logging area were opportunistically sampled to establish the relationship between butt root mass and DBH. Coarse roots >10 mm diameter within 1 m of the bole of the tree were separated from the soil, cut into pieces <50 kg, washed, weighed, and subsampled for dry weight determination. Butt root mass was log-transformed and regressed against DBH. To determine total root mass, we applied the DBH − butt root mass equation to trees in the permanent plots and calculated the mean butt root biomass per ha across the eight logging units. Coarse and fine root biomass are expressed on a per ha basis, the calculation of which excluded areas occupied by butt roots.

DAMAGE ASSESSMENT AND NECROMASS PRODUCTION.—Permanent plots were recensused for tree damage and survival at 5–30 days after logging and again 8–12 mo later. All trees and vines were relocated and assessed for damage. Although numerous damage classes were used in the field, here we compress them into the following: destroyed (uprooted and crushed), snapped-off below crown, and other damage (includes crown, stem, bark, or root damage of varying severity).

From the damage assessment data we estimated (by logging unit) the following parameters: timber volume extracted; necromass produced from the branches, leaves, stumps and butt roots of harvested trees; necromass produced from trees destroyed during harvesting; and, necromass produced from damaged trees that died within the first 8–12 mo after logging. Aboveground and butt root biomass were included in these calculations.

The biomass of shrubs and herbs in logged forest was measured using 1 m^2 clip plots (n = 15) randomly located along transects dispersed through each of seven logging units (3 RIL, 4 conventional logging) 1 yr after logging; the sampling protocol was similar to that used for pre-logging measurements. To determine the biomass of colonizers and resprouted plants in areas with soil disturbance (*e.g.*, skid trails, log landings and roads) one yr after logging, we sampled skid trails in the same seven logging units using 1 m^2 clip plots (n = 10 per unit, N = 70) located in a stratified random manner. Although skid trails and other areas with soil disturbance covered a relatively small percentage of the total area (conventional logging areas, mean = 11.9%, SD = 2.7, N = 4; RIL areas, mean = 3.5%, SD = 1.6, N = 4; Pinard 1995), biomass in these areas was expected to be more variable than that in the rest of the forest and so sampling intensity was higher. As with pre-logging shrub and herb biomass measurements, logging unit divisions were disregarded in the analyses. Pre- and post-logging measurements of shrub and herb biomass are not paired as sampling points were located randomly in logging units.

Coarse root mass (both living and dead) was measured three mo after logging in four logging units (2 RIL, 2 CNV) following the protocol described for the pre-logging measurements. Coarse root pits were located randomly on skid trails (10 pits per logging unit, N = 40) and in other areas of disturbed forest (10 pits per unit, N = 40). The difference between mean coarse root biomass before and three mo after logging (corrected for proportion area in skid trails and disturbed forest) was considered to have entered the necromass pool (if $\text{biomass}_{\text{before}} > \text{biomass}_{\text{after}}$ at $\alpha = 0.05$). As with understory biomass, logging unit divisions were disregarded and thus pre- and post-logging samples were not paired for statistical comparisons. We do not include post harvest fine root mass and assume that fine root mass one yr after logging is similar to mass before logging.

For all of statistical comparisons we use a significance level of 5 percent but report test statistics when P values are between 0.1 and 0.05. T-tests are two-tailed using pooled variances unless stated otherwise. For t-tests using separate variances, degrees of freedom were calculated following Brownlee (1965 in Wilkinson 1990). For treatment comparisons based on the aboveground biomass plots, rather than using a global analysis of variance, we use separate t-tests for each diameter class. Nested subplot size was selected for sampling convenience, not to allow equal variances among the diameter classes. The term biomass always refers to living plant material.

RESULTS

PRE-LOGGING CONDITIONS.—Stand structure in RIL units was similar to that in conventional logging units prior to logging (Fig. 2). The mean number of harvestable trees per hectare (commercial species with DBH ≥ 60 cm DBH) ranged from 14.4 to 26.9 (mean = 19.0, SD = 3.88, N = 8); densities in RIL units did not differ from those in conventional logging units (t = 0.244, df = 6, $P > 0.8$). Total basal area of trees ≥ 10 cm DBH ranged from 24.9 to 33.1 with an overall mean of 27.5 $m^2\ ha^{-1}$ (SD = 2.86, N = 8). Tree densities for the two treatment areas did not differ for any diameter class (t-tests, $\alpha = 0.05$). In the conventional logging units, density of lianas >2 cm DBH averaged 586 stems per ha (SD = 211, N = 4) with about 86 percent of the stems <5 cm DBH.

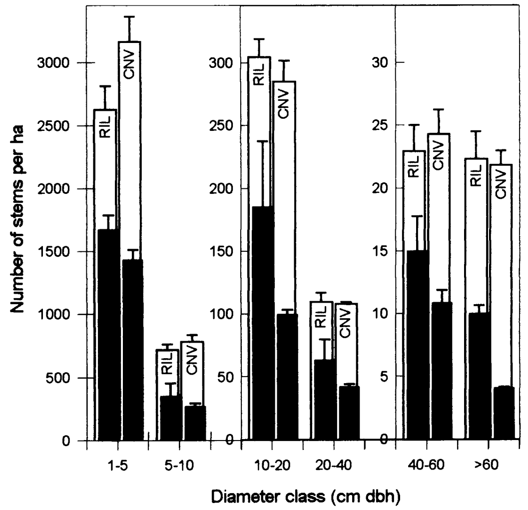

FIGURE 2. Stem density (mean ± SE) for logging units prior to logging (open bars) and after logging (black bars; RIL—units logged according to the RIL guidelines, CNV—units logged conventionally; Note different y-axes). Pre-logging stem densities for trees >5 cm DBH did not differ for the two treatments (*t*-tests with pooled variances, $\alpha = 0.05$; trees 1–5 cm DBH, $t = 2.26$, df = 6, $P = 0.07$, $\text{mean}_{CNV} = 3159$ trees ha^{-1} ± 140.4; mean_{RIL} = 2623.3 trees ha^{-1} ± 187).

TABLE 2. *The 10 most common taxa of trees >1 cm DBH based on density and basal area (BA m² ha⁻¹) before logging in the eight logging units (all plots,* N = *170).*

Species or species group	% Stems	% BA
Parashorea tomentella	12.2	24.1
Shorea johorensis	7.8	23.8
Eugenia spp.	5.8	3.4
Lauraceae	5.5	2.2
Diospyros spp.	5.3	3.4
Annonaceae	4.0	5.4
Shorea parvifolia	1.4	3.5
Shorea leprosula	1.4	2.6
Dryobalanops lanceolata	1.4	2.4
Shorea section *Shorea*	1.1	1.7

Of the 6298 trees ≥10 cm DBH in the plots, 59.3 percent were identified to species or species group (representing 83.3% of the total basal area). Dipterocarpaceae was well-represented in the study area, comprising 29.6 percent of the tagged trees ≥10 cm DBH and 67.9 percent of the basal area (Table 2). The forest was dominated by two dipterocarp species, ***Parashorea tomentella*** and ***Shorea johorensis,*** which together made up 20 percent of the stems ≥10 cm DBH and 47.8 percent of the basal area. The 10 most abundant species or species groups were represented similarly in the RIL and conventional logging units.

Total biomass in the two treatment areas averaged about 400 Mg ha^{-1} with approximately 17 percent occurring belowground (Table 3). For each diameter class, aboveground biomass per ha was equivalent for the two treatments (Table 3). Approximately 59 percent of the initial aboveground biomass was in trees ≥60 cm DBH. Small trees (<10 cm DBH) contributed approximately 4 percent of total aboveground biomass (Table 3). The diameter-biomass regression equation developed for all species of small trees (1–10 cm DBH) was as follows: $\text{Log}_e(\text{Dry Weight}) = 0.539 \times \text{DBH} - 1.25$ ($R^2 = 0.93$, $SE_{slope} = 0.021$, $SE_{intercept} = 0.119$, $P < 0.001$, $N = 40$). Understory plant biomass contributed approximately 1 percent of total aboveground biomass and was similar in RIL and conventional logging units (Table 3). Approximately 2 percent of the aboveground biomass in conventional logging areas was in vines; small vines (2–5 cm DBH) contributed about 56 percent of the vine biomass.

Total belowground biomass averaged approximately 66 Mg ha^{-1} in the two treatment areas, about 40 percent was in butt roots, about 56 percent was in coarse roots (Table 3). Estimated biomass in butt roots for trees ≥20 cm DBH increased with diameter according to the following relationship: $\text{Log}_{10}(\text{Dry Weight}) = 0.014 \times \text{DBH} + 1.51$ ($R^2 = 0.88$, $SE_{slope} = 0.001$, $SE_{intercept} = 0.10$, $P < 0.001$, $N = 14$). Application of the above regression equation to trees ≥20 cm DBH in the plots used for aboveground biomass yields an overall butt root biomass estimate of 25.6 Mg ha^{-1} (SD = 5.84, N = 8). Coarse root biomass (>5 mm diameter) was extremely variable (overall C.V. = 95%; Table 3), owing to the presence of widely dispersed, large roots of the canopy trees that may extend more than 35 m away from the tree's stem (see Baillie & Mamit 1983 for discussion). Coarse root mass in the two treatment areas was not different prior to logging (Table 3). Mean fine root mass (<5 mm) in the upper 10 cm of soil also did not differ in the two treatment areas (Table 3).

Details of logging.—Logging started in July 1993 and ended in March 1994 (Table 4). The time

TABLE 3. *Above- and below-ground biomass for the two logging treatment areas before logging. Means (Mg ha^{-1}) presented with SD and N noted parenthetically. For trees, vines, and butt root mass, SD describes variation among four logging units[a] and does not incorporate error in biomass equations. No significant differences were detected between treatments (t-tests, P < 0.05).*

Before logging	Conventional logging	Reduced-impact logging
Trees >60 cm DBH	190 (35, 4)	190 (53, 4)
Trees 40–60 cm DBH	53 (20, 4)	46 (6.5, 4)
Trees 20–40 cm DBH	46 (2.5, 4)	46 (6.3, 4)
Trees 10–20 cm DBH	21 (2.7, 4)	23 (2.8, 4)
Trees <10 cm DBH	13 (2.0, 4)	12 (2.0, 4)
Vine biomass	7.6 (3.8, 4)	7.6 (3.8, 4)[b]
Understory biomass	2.87 (1.50, 45)	2.94 (1.67, 45)
Butt root mass	26.8 (6.2, 4)	24.5 (5.7, 4)
Coarse roots (alive)[c]	35.9 (33.0, 40)	39.4 (38.7, 40)
Coarse roots (dead)[c,d]	1.6 (2.6, 30)	1.8 (3.5, 26)
Fine root mass	2.57 (1.30, 31)	2.74 (1.43, 18)
Total mean (SD) biomass before logging	399 (40)[e]	394 (59)[e]

[a] Each logging unit considered a replicate and subsampled with 10–27 plots.
[b] Assumed to be equivalent to conventional logging units, no statistical comparison made between treatments.
[c] Log-transformed data used for statistical comparison.
[d] Not included as biomass.
[e] Variance for sum of means calculated using a weighted estimate: $\Sigma_1{}^k ((k(w_i)s_i{}^2)/(n_i))$, where k = # of components, w_i = mean of component/sum of means.

required to log a unit varied from 1–24 weeks. Logging in two of the RIL units was prolonged due principally to wet weather. No dry season occurred during the study period (unpubl. data) and environmental conditions during logging were fairly similar for all units.

A portion of each of the RIL logging units (mean = 44%, SD = 18.9, N = 4, range 12–63%) was deemed unloggable by the rangers due to steep terrain, unstable substrates, lack of commercial trees, or inaccessibility. Because the principal comparison of our study involves impacts of two harvesting methods, we eliminated these unlogged areas (and any influenced plots) from our analysis. Difficulties arose when trying to identify these areas *a posteriori*, but we used the following criterion: if neither a skid trail nor a stump of a harvested tree was inside a plot or within 30 m of any plot boundary, the plot was considered to be within an unloggable area. By this definition, 48 of the 114 plots in the RIL units were eliminated; none of the 104 plots in the conventional logging units were eliminated.

Volume of timber extracted ranged from 54–175 m^3 ha^{-1} and averaged 154 (SD = 20) and 104 (SD = 51) in conventional logging and RIL areas, respectively (Table 4). The two treatments did not differ in terms of volume removed (Table 4) or associated biomass converted into logging debris (Table 5). The Pacific Hardwoods mill that converts the timber extracted from Ulu Segama into lumber, veneer, and blockboard, does so with about 50 percent efficiency (Eng W. H., pers. comm.). Therefore, in addition to the biomass converted to necromass in the forest, we included 50 percent of the biomass in extracted timber in the necromass

TABLE 4. *Dates of logging and volumes of timber removed from reduced-impact logging units (RIL) and conventional logging units (CNV) in Ulu Segama Forest Reserve. Volumes are based on one hundred seventy 1600 m^2 plots distributed among the eight units and were not different in the two treatments (separate variances, t = 1.81, df = 3.9, P = 0.15). Only loggable areas were included in calculations.*

Unit no.—treatment	Dates logged	Volume extracted (m^3 ha^{-1})
32—RIL	17 Jul '93–6 Aug '93	99.3
41—CNV	17 Jul '93–10 Sep '93	134.1
36—RIL	7 Aug '93–16 Aug '93	54.0
38—CNV	7 Aug '93–21 Sept '93	173.3
30—RIL	10 Oct '93–4 April '94	175.1
23—CNV	10 Oct '93–11 Nov '93	138.8
35—RIL	24 Nov '93–21 April '94	86.6
39—CNV	24 Nov '93–21 Dec '93	167.8

TABLE 5. *Mean (SD) Mg biomass ha^{-1} converted into necromass. SD described variation among four logging units and does not incorporate error in biomass equations.*

	Conventional logging units	Reduced-impact logging units
50% of extracted timber	32.22 (4.4)	25.50 (11.12)
Branches, stumps, and butt roots of extracted trees[a]	67.14 (9.76)	45.93 (22.96)
Destroyed trees (uprooted and crushed)	67.49 (45.68)	14.28 (9.56)
Damaged trees dead within one yr after logging	7.20 (6.90)	4.01 (5.00)
Lianas destroyed	5.05 (3.23)	6.61 (3.3)
Understory plant death[b]	1.74 (1.77)	1.78 (1.94)
Coarse root death (excluding butt roots)[b]	10.8 (42.39)	10.4 (48.47)
Total necromass produced	192 (37)[c]	108.5 (22.5)[c]
Mean (SD) difference between two logging methods	86 (43) Mg necromass ha^{-1}	

[a] Treatment comparison *t*-test with separate variances, $t = 1.79$, df $= 3.8$, $P = 0.15$.
[b] Represented as the difference between biomass before logging and biomass at one yr after logging.
[c] Variance for sum of means calculated using a weighted estimate: $\Sigma_1^k ((k(w_i)s_i^2)/(n_i))$, where k = # of components, w_i = mean of component/sum of means.

pool. Most scrap at the mill is burned to produce electricity.

DAMAGE ASSESSMENT AND NECROMASS PRODUCTION.— For all DBH classes, proportionally more trees were damaged from logging in conventional logging areas than in RIL areas (one-tailed *t*-tests, arcsine transformed data, $\alpha = 0.05$; Figs. 3 & 4). The proportion of trees damaged differed by DBH class (ANOVA on arcsine transformed data, $F = 3.45$, df = 5, 36, $P < 0.02$) following a general pattern of more damaged trees in smaller DBH classes (Fig. 3). There was no interaction in the proportion of trees damaged between logging method and DBH class ($F = 1.4$, df = 5, 36, $P = 0.25$).

The percentage of trees destroyed during log-

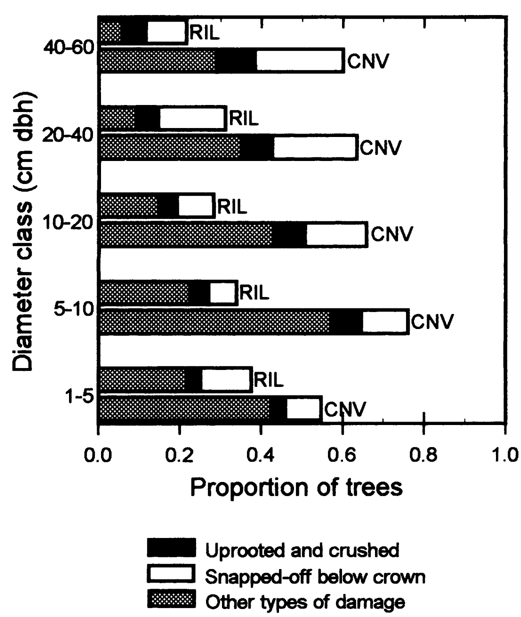

FIGURE 3. Mean proportion of trees completely destroyed, snapped-off, or otherwise damaged (stem, bark, crown, or root) during logging in four units of each treatment. The proportion of trees snapped-off or otherwise damaged did not differ for the two treatments (*t*-tests, arcsine transformed data, $\alpha = 0.1$). (Destroyed trees do not include harvested trees.)

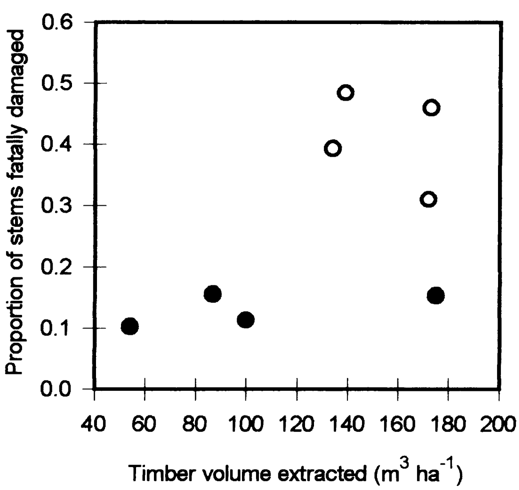

FIGURE 4. Mean proportion of stems (1–60 cm DBH) fatally damaged plotted against mean timber volume extracted (m^3 ha^{-1}). Open circles represent units logged conventionally; solid circles represent units logged according to RIL guidelines.

TABLE 6. *Percentage of trees dead at 8–12 mo after logging for each treatment (RIL = reduced-impact logging, CNV = conventional logging). All damaged trees in the four logging units were pooled for each treatment to generate mortality figures; sample sizes (*i.e.*, number of trees) are noted parenthetically. All trees uprooted or uprooted and crushed are assumed dead.*

	Snapped-off		Other damage		Undamaged	
DBH class	CNV	RIL	CNV	RIL	CNV	RIL
≥60 cm	14.3% (7)	28.6% (7)	2.7% (37)	0.0% (27)	0.0% (72)	0.0% (101)
40–60 cm	22.0% (18)	42.9% (7)	10.0% (40)	0.0% (15)	0.0% (90)	0.5% (94)
20–40 cm	21.7% (60)	12.1% (33)	8.2% (171)	0.0% (100)	0.9% (352)	0.3% (382)
10–20 cm	17.9% (84)	21.1% (38)	8.2% (171)	2.9% (69)	0.5% (414)	0.0% (509)
5–10 cm	11.3% (53)	10.0% (20)	8.6% (93)	2.9% (34)	1.4% (283)	0.6% (349)
1–5 cm	23.8% (21)	22.7% (22)	6.0% (100)	1.5% (67)	0.5% (374)	0.4% (282)

ging was higher in units logged conventionally than in units logged according to the RIL guidelines for all DBH classes (Fig. 3, one-tailed *t*-tests, arcsine transformed data, $\alpha = 0.05$); the mean values by DBH class ranged from 17–57 percent in conventional logging areas in contrast to 2–22 percent in RIL areas (Fig. 3). The biomass in these destroyed trees was assumed to enter the necromass pool (Table 5).

The proportion of trees snapped-off (below crown) ranged from 3.5–10 percent across the DBH classes (Fig. 3) and was higher in conventional logging than RIL areas for only one of the six diameter classes, trees 10–20 cm DBH ($t = 1.77$, df = 6, $0.01 < P < 0.05$; one-tailed *t*-tests, arcsine transformed data). Snapped-off trees were distinguished from other severely damaged trees because we expected a proportion of these would resprout and would not entirely enter the necromass pool immediately.

The incidence of minor to moderate damage (*e.g.*, crown or bark damage) was higher in conventional logging units than in RIL units for three diameter classes, 40–60 cm DBH (arcsine transformed data, $t = 1.97$, df = 6, $P < 0.05$), 10–20 cm DBH ($t = 2.17$, df = 6, $P < 0.05$), and 5–10 cm DBH ($t = 4.10$, df = 6, $P < 0.05$); the other three diameter classes did not differ (one-tailed *t*-tests on arcsine transformed data, $\alpha = 0.05$; Fig. 3).

REASSESSMENT.—At 8–12 mo after logging, many of the damaged trees were dead (Table 6). Overall, 18 percent of the trees (>5 cm DBH) snapped-off below the crown had not resprouted, and thus were considered dead. In general, trees snapped off at a height >10 m resprouted regardless of logging treatment. The mortality rates for trees receiving other types of damage ranged by DBH class from 0 to 3 percent in RIL areas and from 3 to 10 percent in conventional logging areas (Table 6). The percentage of these damaged trees that died during the first year after logging was higher in conventional logging areas than in RIL areas for all diameter classes (Table 6). The two logging treatments were not compared statistically because none of the damaged trees in many logging units had died. Although many of the damaged trees were expected to die soon, only the proportions that died before the recensus were incorporated into the necromass pool (Table 5).

Between the time the plots were established and the recensus (approximately 18 mo after establishment), the undamaged trees (>5 cm DBH) had an average mortality rate of 0.5 percent. The mortality rates for undamaged trees appears similar for the two treatments (Table 6).

Shrub and herb biomass 12 mo after logging was less than before logging both on skid trails (separate variances, $t = 12.64$, df = 133.5, $P < 0.001$; Tables 3 & 7) and in otherwise disturbed forest ($t = 8.97$, df = 193, $P < 0.001$; Tables 3 & 7). Biomass on skid trails was greater in RIL units than in conventional logging units (Table 7). Biomass in other areas of disturbed forest did not differ for the two treatments (Table 7). The difference between shrub and herb biomass before logging and at 12 mo after logging was considered necromass (Table 5).

Three mo after logging, coarse root biomass (exclusive of butt roots) on skid trails did not differ between the logging treatments (Table 7) but was less than pre-logging levels (log-transformed data, $t = 15.2$, df = 118, $P < 0.001$; Tables 3 & 7). This decline is probably due to both root death and excavation and relocation from bulldozer activities. Dead coarse root mass on skid trails did not differ between treatments (Table 7) and was similar to

TABLE 7. *Above- and below-ground biomass (and necromass) for the two logging treatment areas 8–12 mo after logging or, in the case of coarse roots, three mo after logging. Means ($Mg\ ha^{-1}$) presented with SD and N noted parenthetically. For trees, vines, and butt root mass, SD describes variation among logging units and does not incorporate errors in biomass equations. Statistical test results are given for treatment comparisons: pre- and post- harvest comparisons are presented in text.*

After logging	Conventional logging	Reduced-impact logging	Statistical test results
Trees >60 cm DBH	49 (15, 4)	100 (16, 4)	$t = 4.12$, df = 6, $P = 0.004$
Trees 40–60 cm DBH	37 (13, 4)	41 (4.9, 4)	$t = 0.58$, df = 3.8, $P = 0.59$[a]
Trees 20–40 cm DBH	29 (5.0, 4)	42 (7.0, 4)	$t = 2.9$, df = 6, $P = 0.03$
Trees 10–20 cm DBH	11 (2.7, 4)	16 (3.6, 4)	$t = 2.22$, df = 6, $P = 0.068$
Trees <10 cm DBH	6.9 (1.2, 4)	9.8 (1.8, 4)	$t = 3.0$, df = 6, $P = 0.02$
Vines	2.6 (1.1, 4)	0.99 (0.49, 4)[b]	no test performed
Understory (skid trails)	0.30 (0.38, 40)	0.82 (0.97, 30)	$t = 2.81$, df = 35.6, $P = 0.008$
Understory (disturbed forest)	1.24 (1.02, 60)	1.17 (1.03, 45)	$t = 0.33$, df = 103, $P = 0.75$
Butt roots	11.53 (3.0, 4)	17.39 (2.73, 4)	$t = 2.97$, df = 3.4, $P = 0.05$[a]
Coarse roots (skid trails–alive)	2.80 (6.08, 20)	1.81 (3.75, 20)	$t = 0.22$, df = 38, $P = 0.83$[c]
Coarse roots (skid trails–dead)	2.58 (3.32, 20)	8.28 (15.0, 20)	$t = 1.79$, df = 38, $P = 0.08$[c]
Coarse roots (disturbed forest–alive)	28.1 (30.2, 20)	30.0 (36.8, 20)	$t = 0.38$, df = 38, $P = 0.71$[c]
Coarse roots (disturbed forest–dead)	0.98 (1.43, 20)	4.08 (14.8, 20)	$t = 0.73$, df = 38, $P = 0.48$
Mean (SD) total biomass after logging[d]	176 (34)[e]	264 (40)[e]	

[a] Separate variances.
[b] Calculated as 13 percent of pre-logging vine biomass.
[c] Log-transformed data.
[d] Adjusted for area in skid trails RIL = 3.5 percent, CNV = 12 percent. Dead coarse roots not included. Fine root mass included assumed equivalent to pre-logging mass.
[e] Variance for sum of means calculated using a weighted estimate: $\Sigma_1^k\ ((k(w_i)s_i^2)/(n_i))$, where k = # of components, w_i = mean of component/sum of means.

dead root mass before logging in conventional logging areas (log-transformed data, t = 1.07, df = 58, P = 0.29; Tables 3 & 7). In RIL areas, dead coarse root mass was possibly higher three mo after logging than before logging (log-transformed data, t = 1.64, df = 54, P = 0.11; Tables 3 & 7)

In disturbed forest (not skid trails) three mo after logging, coarse root biomass did not differ between treatments (Table 7) and was similar to pre-logging estimates (log-transformed data, t = 1.6, df = 118, P = 0.11; Tables 3 & 7). Dead coarse root mass in logged-over forest did not differ from pre-logging mass (log-transformed data, t = 1.35, df = 118, P > 0.18; Tables 3 & 7), nor did treatments differ (Table 7). Because conventional logging units had proportionally more area with disturbed soil or skid trails than did RIL units (approx. 12% and 3.5%, respectively; unpubl. data), the calculated total standing stock of coarse root biomass in conventional logging units was less than in RIL units (Table 7). Our estimates of necromass produced from coarse root death ($\text{biomass}_{\text{before}}$ − $\text{biomass}_{\text{after}}$) are associated with relatively large standard deviations (Table 5) as they were calculated as the sum of the variances for the pre- and post-harvest biomass estimates.

Sixty-seven percent of the vine stems were killed during logging in conventional logging units contributing an average of 4.68 Mg biomass per ha (SD = 0.18) to the necromass pool. Mortality was evenly distributed across diameter classes (ANOVA, F = 0.49, df = 3, 12 P > 0.6). Vines in the RIL units were neither tagged nor measured prior to cutting. To estimate vine biomass killed in the RIL areas we assume that vines cut were killed (87% of stems, unpubl. data) and that they represented 87 percent of the total vine biomass (Tables 3, 5 & 7).

One year after logging, forest areas logged by conventional methods and according to RIL guidelines contained approximately 44 percent and 67 percent of their pre-logging biomass, respectively (Tables 3 & 7). The difference between the two methods in necromass produced was 76 Mg ha^{-1} (37 Mg C ha^{-1}, Table 5). The greater number of residual trees destroyed during logging in conventional logging areas was responsible for approximately 62 percent of the difference between the two methods; the difference in debris produced from trees felled accounted for approximately 25 percent of the difference. The standard deviation associated with the estimate of the difference between the two methods is largely related to variation in coarse root death.

DISCUSSION

Implementation of RIL harvesting guidelines substantially reduced logging damage. The residual forest in the two treatment areas is dramatically different, hence each forest's potential for both short- and long-term carbon storage also differs. In the following sections, we compare our biomass estimates to other dipterocarp forests and briefly discuss estimation methods. We compare levels of logging damage recorded at our sites with other selective cutting operations and discuss ecological implications of reductions in damage for forest recovery. We also discuss the amount of carbon retained due to implementation of the RIL guidelines, how it could be increased, and how it relates to power plant emissions and other offset options. Finally we identify several issues relevant to future efforts to offset carbon through reduced-impact logging and suggest topics needing further research.

Residual forest biomass.—Pre-logging aboveground biomass estimates for our sites (291–400 Mg ha^{-1}, mean = 330) are higher than average moist forest biomass in southeast Asia (mean = 225 Mg ha^{-1}, N = 204 stand inventory data sets; Brown *et al.* 1991) but are comparable to estimates for unlogged forests in Sarawak (280–405 Mg ha^{-1}, Brown *et al.* 1991). Big trees (>60 cm DBH) made up about 59 percent of the pre-logging biomass at our sites. Degraded forests tend to have few big trees and, consequently, have much lower stores of biomass (see Brown *et al.* 1991).

We calculated tree biomass using published regression equations and conversion factors. Both stem volume equations and biomass expansion factors (BEF) are associated with standard errors but these errors were not incorporated into our estimates. We assume that the variance inherent in calculated estimates apply equally to the two treatments. Stem volume equations used in this study were generated from trees within the Ulu Segama area (Forestal International Ltd. 1973). The BEF, however, was based on data taken from Peninsular Malaysia, Indonesia, Cambodia and Brazil; we did not harvest trees to determine whether or not the selected BEF was appropriate for our site. We also made no provision for hollow trees and therefore may have overestimated forest biomass.

Our estimates of necromass produced from logging were based on a relatively large sampling area but did not incorporate the complete necromass pool. No effort was made to measure necromass inputs from trees damaged but not killed (*e.g.*, crowns of snapped off trees, or branches from trees subjected to crown damage) making our estimate conservative. Also, trees snapped-off below the crown which had resprouted at the 8–12 mo census were considered alive, although many of these trees will probably die within the second yr post-harvest (Putz & Brokaw 1989).

Data published on belowground biomass in tropical moist forests are sparse and, generally, based on few samples. For example, Edwards and Grubb (1977) excavated roots from two pits (10 × 5 m) to a depth of about 25 cm. Sim and Nykvist (1991) excavated roots from seven pits (0.5 × 0.5 m) to 50 cm depth. Our estimate (about 17% of aboveground biomass) falls close to the mean of reported values for tropical moist forests (mean = 19%, range = 7–41%, N = 7; Ogawa *et al.* 1965, Hozumi *et al.* 1969, Jenik 1971, Klinge & Rodrigues 1974, Edwards & Grubb 1977, Bullock 1981, Sim & Nykvist 1991). Although we probably underestimate coarse root biomass by sampling to 50 cm depth, the bias introduced may not be substantial; in dipterocarp forest on similar terrain in Sarawak, most coarse roots were found to be at 15–40 cm below the surface (Baillie & Mamit 1983). For fine roots, our sampling of the upper 10 cm probably included 55–60 percent of total fine root mass (Green 1994). Bias in our butt root measures is harder to predict. Uprooted trees along roads and skid trails may not have complete root systems, and do not represent a random sample from the population; furthermore, we made no effort to separate live and dead sections of root.

LOGGING DAMAGE.—In this study, there was no correlation between the proportion of stems fatally damaged and timber volume extracted (R^2 = 0.39, P = 0.37, N = 8; Fig. 4). This result is contrary to the more general finding that logging damage and volume extracted are positively correlated (*e.g.*, Nicholson 1979). Across the broad range of volumes extracted in RIL units, fatal damage remained less than 20 percent of the stand, lending support to the conclusion that treatment differences in logging damage were due to logging technique, not harvesting intensity.

Relative to other selectively logged tropical forests, the amount of timber removed from our study site was high, as was the level of logging damage. First cuts in Amazonian moist forest generally take $<$50 m^3 ha^{-1} (Uhl & Vieira 1989, Thiollay 1992, Verissimo *et al.* 1992); in African forests generally $<$30 m^3 ha^{-1} of timber is harvested (Nwoboshi 1987, Ola-Adams 1987, Klo & Ekwebelam 1987, Wilkie *et al.* 1992, White 1994). Even though the lack of standard methodologies precludes direct comparisons of results, for four studies where logging damage was reported, damage to residual trees $>$10 cm DBH averaged 11 percent (Gabon—White 1994), 18 percent (Nigeria—Ola-Adams 1987), 26 percent (Brazil—Uhl & Vieira 1989) and 43 percent (Brazil—Verissimo *et al.* 1992). The damage recorded in our conventional logging areas (approximately 66%), though higher than the figures from Amazonia and Nigeria, is similar to figures reported for other sites in Sabah (Fox 1968, Chal & Udarbe 1977), Sarawak (Nicholson 1979, Mam & Jonkers 1981), and West Kalimantan, Indonesia (Cannon *et al.* 1994).

Implementation of RIL guidelines in our study area was associated with a reduction in damage to the residual stand, both in extent and severity. In reduced-impact logging areas, $\approx$27 percent of trees $>$10 cm DBH were damaged and $\approx$19 percent were dead within the first year after logging compared with $\approx$54 percent damaged and $\approx$46 percent dead in conventional logging areas. Efforts to control damage in tropical moist forest in Suriname (Hendrison 1990) and Indonesia (J. G. Bertault & P. Sist, pers. comm.) also reduced damage by about half as compared with uncontrolled or conventional logging. The slopes in our sites, on average, exceeded those recommended for ground-based skidding. Switching to an aerial yarding system (*e.g.*, skyline cable yarding), as is generally recommended for slopes greater than about 20 degrees (Dykstra 1994), could further reduce damage, as might further training of fellers and bulldozer operators.

RIL areas had about 25 percent fewer severely damaged residual trees (all DBH classes) than conventional logging areas. Often severe damage (*e.g.*, uprooted, crushed, or snapped-off) is associated with skidding operations and felling trees laden with lianas (Fox 1968, Appanah & Putz 1984). Vine-cutting, planning skid trail locations, and controlling skidding operations may have been instrumental in reducing severe damage in RIL areas. Reductions in less severe damage (*i.e.*, crown and bark damage) in RIL areas may have been related to directional felling. Directing trees onto skid trails or into gaps created by previously felled trees further reduced overall gap size and felling damage (Hendrison 1990).

IMPLICATIONS FOR FOREST RECOVERY AND CARBON STORAGE.—RIL areas contained nearly 100 Mg more biomass per ha than conventional logging areas one yr after logging. If both forests were ultimately to recover pre-logging biomass stores, then, regardless of conditions immediately following logging, the net difference in stored biomass, at this ending point, would be zero. Given that these are production forests, repeated cutting cycles or conversion to plantations are their probable fates; they are unlikely to be abandoned for the 200 or 300 years probably needed to fully recover biomass. The timescale relevant to this discussion, therefore, may be through the next cutting cycle (generally stated as 60 yr but undoubtedly will be shorter). During this period, differences in growth and mortality rates and other responses to logging could increase or decrease the difference between the two treatments in biomass stores. We expect biomass to continue to decline in both areas for 2–6 years after logging. Following stabilization of mortality rates, we expect biomass accumulation rates to be greater in RIL areas than in conventional logging areas. The rationale behind our predictions is outlined below.

Mortality rates in logged forest are often relatively high for several years after logging relative to pre-harvest levels (Wan Razali 1989). Elevated mortality rates may be due to any or all of the following: damage incurred during logging; increased exposure and edge effects (*e.g.*, Young & Hubbell 1991); increased incidence of mechanical damage from vines (Putz 1991) and falling debris (Wan Razali 1989); and competition with fast growing trees and vines (Fox & Chai 1982). Conditions in conventional logging areas (*i.e.* proportionally more damaged trees and greater degree of crown exposure) are expected to be associated with higher mortality rates (Korsgaard 1992). The difference in mortality during the first year of post harvest observations supports this conjecture.

Growth rates in logged forest have been found to be correlated with crown exposure (Korsgaard 1992, van Daalen 1993) and, in general, increased growth rates are frequently observed in residual trees following selective logging (Jonkers 1987, Wan Razali 1989), thinning operations (Fox & Chai 1982, Korsgaard 1992), or natural gap formation (Brown & Whitmore 1992). Though fewer in number, the undamaged residual trees in conventional logging areas may show larger growth increments after logging than trees in RIL areas because of the more open canopy conditions after conventional logging. Overall biomass accumulation, however, is expected to be greater in RIL area than in conventional logging areas because of several characteristics of logged dipterocarp forest as discussed below.

First, large canopy openings can lead to extensive vine and pioneer tree invasions (*e.g.*, Chai & Udarbe 1977, Cannon *et al.* 1994). Residual trees infested with vines or overtopped by pioneer trees may experience reduced growth rates (Lowe & Walker 1977, Putz *et al.* 1984). Vine invasions in RIL areas are expected to be less common than in conventional logging areas due to vine cutting before logging and more closed canopy conditions after logging (Appanah & Putz 1984). Pioneer trees may be more likely to colonize conventional logging areas because of both more open canopy and more soil disturbance (Chai & Udarbe 1977). Pioneer trees, because of their low wood densities and short life spans, may not accumulate as much biomass per unit area as similar-sized persistent forest species (Jordan & Farnworth 1980). Second, RIL areas contain more undamaged trees and more trees in the larger DBH classes than conventional logging areas and so the residual trees in RIL sites will probably have larger volume increments than residual trees in conventional sites. Third, sites with scraped and compacted soils accumulate less biomass than sites free of heavy soil disturbance (*e.g.*, Maycock & Congdon 1992), and proportionally more soil was severely damaged in conventional logging areas. For the skid trails that were opened in RIL areas, higher biomass one yr after logging relative to skid trails in conventional logging areas, may reflect less severe soil disturbance due to controlled logging (*e.g.*, restrictions on soil scraping and wet weather logging). The effect of a larger input of nutrients from logging debris in conventional areas as compared with RIL areas is difficult to predict. The input may stimulate tree growth but could lead also to nutrient immobilization by microbes, decreasing nutrient availability for trees (Lodge *et al.* 1994).

To summarize, for some time after logging, we expect carbon stored in both RIL and conventionally logged forests to decline from levels immediately following logging because of high mortality rates and decay of logging debris. If carbon accumulation rates are higher in RIL areas due to low mortality rates and small quantities of decaying logging debris, they will become net sinks for carbon in fewer years after logging than the conventional logging areas.

CARBON OFFSETS THROUGH REDUCED-IMPACT LOGGING.—In this pilot project we demonstrated that implementation of RIL guidelines in dipterocarp forests, that would otherwise be logged in an uncontrolled and destructive manner, could result in the short-term retention of, on average, about 42 Mg C ha^{-1} at a cost of approximately U.S.$300 ha^{-1} (J. Tay, pers. comm.). If the carbon "savings" was considered through the next rotation (*e.g.*, 40–60 yr), the difference in carbon stored in RIL areas versus conventional logging areas is expected to be greater than 42 Mg ha^{-1}. How policy makers will translate this effort into carbon credits is uncertain (Dixon *et al.* 1993, USDOE 1994). Without doubt, however, the time profile of emission reductions or carbon sequestration will be important for determining the consequences of the action for climate change (*e.g.*, Price & Willis 1993).

Forestry-based carbon offset programs, like the RIL Project, can supplement but not replace other efforts such as energy conservation, fuel switching, and increased power plant efficiencies. For example, application of our estimate (42 Mg ha^{-1}) to the project area (1400 ha) yields 58,800 Mg C, equivalent to about 17 percent of the annual emissions from a 200 MW coal-burning energy plant (Freedman *et al.* 1992). Given the ubiquity of poor timber harvesting practices, considerable scope exists for application of reduced-impact logging in other tropical, subtropical, and temperate forests. This approach to offsetting carbon may not be appropriate for forests with large proportions of their ecosystem carbon stored in fallen logs and soil organic matter because harvesting operations can result in large net losses in carbon over time (Harmon *et al.* 1990). Principally, a forest's potential for retaining carbon by altering harvesting practices is a function of the forest's biomass, the baseline to which the damage-controlled site is compared, possibilities for damage reduction, and the volume of timber extracted. In the pilot project in Sabah, about 36 percent (or 15 Mg C ha^{-1}) of the additional carbon retained in RIL areas was related to volume extracted and debris from felled trees (*i.e.* treetops, stumps, butt roots). Reductions in net volume extracted are not inherent to RIL operations but are related to areas in streamside buffer zones, terrain, expertise of operators, and supervision of field operations. As the project expands in Sabah, we expect differences related to number of trees felled per ha to disappear, as will this proportion of the carbon savings.

As policies supporting forestry-based carbon offset projects develop, so will a system for evaluating potential projects, their credibility, reliability, and verifiability (Dixon *et al.* 1993). Describing the costs and benefits of reduced-impact logging as a harvesting technique is complicated by externalities and undervalued environmental services (Kramer *et al.* 1992). Assessment of the cost-effectiveness of applying RIL techniques for offsetting carbon will require an even more complex analysis. Reduced-impact logging carbon offset programs may be attractive to power companies because, relative to many tree planting programs, the carbon yield comes earlier and less risk is involved. The risk of losing the investment to pests, fire or disease are small relative to that for trees in industrial monoculture plantations with rotations of 5–20 yr.

Expansion of the RIL approach to carbon offsetting is predicated on international acceptance of joint implementation. Hesitancy is coming from developing countries suspicious about the motivation of wealthy countries. Also, as nations industrialize they will develop their own need for reducing net emissions. Although it would not be sensible for developing countries to sell all of their inexpensive offset options to the industrialized nations, poorly managed forests abound and the world's supply of forestry-based carbon offset options is not in jeopardy.

Researchable issues.—Policy makers will look to ecologists to provide estimates of impacts of forestry-based carbon offset programs. Particularly lacking are data on the biomass for very large trees and woody root biomass. For many tropical trees, little is known about the effects of mechanical damage on growth rates, wood quality, fruit production, mortality rates, and pathogen attack. Foresters promote vine cutting as a useful tool for reducing logging damage but the implications of vine cutting on wildlife species, particularly frugivores and foliovores, should be investigated. Logging stimulates leaf production in some species (*e.g.*, Johns 1988) but few data exist describing changes in fruiting phenology or fruit abundance following logging (but see Wong 1983, Johns 1988). The incidence of weed invasions in logged-over forest appear to be related to gap size, soil disturbance, and pre-logging species composition. Research directed towards elucidation of these relationships could be useful for predicting impacts of harvesting clusters of trees versus scattered individuals and trees growing in areas with climbing bamboo (*Dinochloa* spp.), and the importance of minimizing soil disturbance. Further efforts to quantify the impacts of forest management activities on carbon storage or sequestration rates through models (*e.g.*, Cropper & Ewel 1987, Dewar 1990) and the validation of models will contribute to the database from which proposed carbon offset projects can be assessed.

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