PAPERS

Setting critical limits to ecological indicators of sustainable tropical forestry

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SUMMARY

Ecologically sustainable forest management necessitates monitoring forest conditions and the environmental impacts of management activities. Through adaptive management, assessments that reveal excessive environmental impacts require revision of the management plan to ensure that ecological functions and attributes of forest ecosystems are maintained within acceptable limits. But how are these acceptable limits determined? And how are 'normal' or 'natural' conditions defined when these are temporally or spatially variable. In order to determine the desired biological state of a forest, indicator baseline values, together with some measure of variability under natural conditions are needed. A useful Ecological Criteria and Indicator set should therefore provide objective standards for evaluation, and these standards should account for spatial and temporal variation in indicator states as well as resilience to anthropogenic disturbance. We present a framework within which the standards for assessing ecologically sustainable forest management might be defined. We draw upon published literature to determine the responses of selected indicators to natural and anthropogenic disturbances. Using these values we can begin to predict expected ecological conditions of a sustainably managed forest given information on forest type, disturbance intensity and time since disturbance.

Keywords: biodiversity, criteria and indicators, forest ecology, forest management.

INTRODUCTION

Evaluation of biodiversity using Criteria and Indicators

Forest certification aims to promote sustainable forest practice through independent evaluation of forest management against a number of objective and unambiguous requirements (Principles and Criteria). In the evaluation process the requirements translate into a number of Indicators and Verifiers to ascertain whether management planning and activities meet the requirements set out. This review uses the ecological C and I set, including the verifiers and definitions therein, developed by CIFOR and described by Boyle *et al.* (1998). An **Indicator** is any variable or component of the forest ecosystem or relevant management system that is used to infer attributes of the sustainability of utilisation of the resource. **Verifiers** are the data or information that need to be collected for the assessment of any particular indicator. The aim of an ecological Criteria and Indicator set (C and I) is to provide an objective and unambiguous set of Indicators and Verifiers against which environmentally sustainable forest management may be evaluated.

But how should we determine what is an acceptable desired state in terms of environmental sustainability? On the premise that maintenance of pre-disturbance ecological attributes is the goal of ecological sustainability a further two questions are demanded in answer: 1) What are the original (or desired) conditions of the verifiers to be used in any assessment process; and 2) What is the operational time-scale for assessment?

The second of these questions is easier to deal with, as the relevant time-scale can be set according to the local context. We use 'sustainability' in this context to refer to a representative tropical forest cutting cycle of, say, 40 years, by which time data generated by ecological verifiers should be statistically identical to equivalent data from undisturbed forest. Thus the goal of a forest manager would be to secure the desired value for each verifier in the C and I set by the end of the logging cycle (i.e. 40 years after the initial harvest).

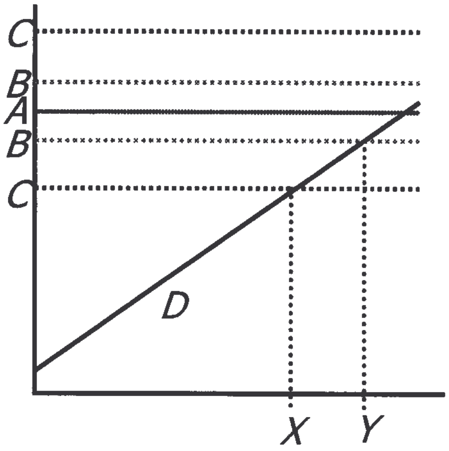
While recognizing that a return to the exact original state is impossible, ecological sustainability infers the conservation of certain key functions and parameters of the ecological system. The CIFOR C and I set have been designed with this in mind. It is therefore necessary to place values on the original state of the selected verifiers (Question 1 above). The 'baseline' values need to include a measure of the average condition of the verifiers, and an associated measure of variability, such as standard deviation, to account for natural spatial and temporal heterogeneity over a representative period of time (e.g. the length of a single cutting cycle). The goal for recovery can then be defined as the set of verifier values that lie within one standard deviation of the mean value. If less rigour is needed, the acceptable range of verifier states may encompass values that lie within two (or other multiple) standard deviations of the mean. Thus recovery may be more formally defined as the point at which the state of the verifier in logged forest is statistically identical to the state of the verifier in unlogged forest. The variability of values (i.e. its standard deviation) of a particular verifier in natural undisturbed forests will determine its utility for assessing anthropogenic impacts. Clearly, verifiers that generate a set of highly variable results under similar conditions are of little value for the rapid assessment of disturbance impacts, and probably of little value at all.

Finally, information on the rate and directionality of indicator responses to disturbance as measured by verifiers is needed, to which data generated from the implementation of C&I can be compared for an assessment of sustainability. Information on the patterns of response can be useful in determining whether particular verifiers should be rejected from the C&I set. A clear and ubiquitous pattern of recovery enhances the value of a verifier as a monitoring tool while the absence of any clear pattern indicates that the verifier concerned is unlikely to be useful in evaluating sustainability of forest practice.

Critical time-thresholds

Different verifiers have different patterns of recovery. Such variation can form the basis for a series of checks, using different verifiers, conducted at intervals through the regeneration phase of the cutting cycle. To satisfy the condition of sustainability all verifiers, except for the special case of landscape pattern verifiers, are expected to have achieved an acceptable state by the end of the 40-year harvesting and growth cycle. Thus a *critical time-threshold* might be defined for each verifier as the expected time after disturbance at which the verifier value is statistically indistinguishable from the undisturbed (baseline) value (or the prescribed desired condition, if different from the undisturbed state). If verifier recovery rates are known it will be possible to predict the critical time-thresholds. Rapidly recovering verifiers provide early indications of unsustainable practice (if the expected targets are not met), while long-term verifiers confirm the sustainability of practice at the end of the regeneration cycle. In this way unsustainable practices may be identified and modified as information is generated by verifiers at intervals throughout the logging cycle. This opportunity for adaptive management provided by the CIFOR C&I set is a primary advantage over other C&I systems.

A critical time-threshold for any verifier depends on verifier rate of recovery and its natural variability. Given linear recovery rates the recovery time of a verifier that has a highly variable natural state is less than a verifier with little variation (Figure 1). Setting critical time-thresholds for verifier recovery should account for differences in natural variability, possibly by designing sampling regimes such that standard errors are used to compare verifier states, or power analysis used to quantify the confidence placed in the data. The actual critical time threshold set may, ultimately, be a subjective decision informed by available information (such as that contained in this review or generated by the implementation of C&I in the FMU). Critical thresholds might even be reached by expert and stakeholder consensus. In any case, having set and justified critical time thresholds a transparent assessment system for adaptive management can be constructed and which can be checked by third party certifiers. Ultimately, verifiers can be placed into a framework according to whether the information they provide is useful for assessing sustainability early or late in the regeneration cycle (Table 1).



Where ***A*** indicates the mean value for a verifier in a natural and anthropogenically undisturbed state, ***B*** and ***C*** represent the upper and lower boundaries of acceptable variability around ***A*** under moderately and highly variable states respectively, and ***D*** indicates the constant recovery exhibited by verifier values obtained from a disturbed forest, then ***X*** and ***Y*** represent the two different critical time-thresholds obtained under ***C*** and ***B*** respectively.

FIGURE 1 *Representation of different critical time-thresholds for the same verifier given differing variability in the natural undisturbed state. The variability of the states of a verifier under natural conditions can increase uncertainty with regards to setting an appropriate critical time-threshold for the verifier. Full recovery is expected to occur sooner where much variability about the mean is recorded.*

TABLE 1 *A framework of critical time-thresholds for categorizing expected verifier recovery times*

Critical time-threshold	Years since logging
very early	<5
early	5-10
early-mid	10-20
mid-late	20-30
late	30-40

Objectives

To propose critical time thresholds for verifiers and to assess the usefulness of the proposed verifiers for ecological C and I monitoring two values are needed for each verifier.

1. ***The baseline (or target)*** – the average value of the verifier in the desired state. Associated with the baseline value should be a measure of variability to reflect temporal and spatial variation.
2. ***The pattern of recovery*** – the expected rate and direction of change in verifier values through a time series following forestry operations to the desired target state.

An initial appraisal of verifier baseline values and patterns of recovery can be obtained from current studies of ecological responses to logging. Drawing upon such information is problematic given that studies are rarely co-ordinated and vary in their management and harvesting regimes, the sampling protocols used, and the location and forest types in which they are conducted. Few studies monitor ecological responses over long time-series, and most use chronosequences that assume equivalent starting conditions between sample plots. Despite these deficiencies a review of published studies provides at least preliminary information needed to set baseline values and determine recovery patterns, allowing initial assessment of the utility of verifiers. Thus we aim to quantify baseline values, patterns of recovery, and variability of selected verifiers derived from the CIFOR C and I set (Boyle *et al.* 1998).

FOREST STRUCTURE

Verifiers: Basal Area
Canopy Openness

Variation among regions and ecosystems

Values for forest structure verifiers may have regional or even global applicability provided attention is paid to forest type (Whitmore 1990, Wadsworth 1997). Data on forest structure verifiers from similar forest formations around the globe can therefore be compared, and if shown not to be greatly variable in natural undisturbed conditions can be used to derive generic baseline values. Verifier recovery patterns can also be reviewed to determine their generality. Here we use two examples, the basal area and canopy cover verifiers, to illustrate these points.

Canopy cover and basal area under undisturbed forest conditions

Values for canopy cover across studies in undisturbed tropical wet forests around the globe average 91±5% (Table 2) giving a reliable baseline value with limited apparent variability. Basal area estimates from five studies (Rundel and Boonpragob 1995, Pascal and Pelissier 1996, Pinard and Putz 1996, Kammesheidt 1998, Sist *et al.* 1998) in lowland moist tropical forest gives a mean basal area value of 35±3.4m^2/ha. Variability in these verifiers across forest types is likely to be much greater than within the single tropical wet forest type, and baseline values will need to be calculated accordingly. More problematic is variability within sites due to topography or edaphic conditions which can give rise to different forest formations on local scales. This has implications for planning of monitoring protocols to account for local spatial heterogeneity.

TABLE 2 *Canopy cover of undisturbed tropical rain forests from different regions*

Location	Canopy cover	Reference
Gabon	93	White 1994
Amazon	92	Johns 1996
Costa Rica	91.4 ± 4.5	Webb 1997
Guiana	87-92	Thiollay 1992
Amazon	80	Uhl & Viera 1989
Panama	95	Lang & Knight 1983
Venezuela	94.1 ± 1.5	Mason 1996

Patterns of canopy cover and basal area recovery

Initial impact

Damage to the stand and canopy loss is generally related to the number of stems harvested per hectare (Webb 1997) but can also be strongly influenced by the nature of the logging operations. The volume of timber extracted varies across regions according to the stocking of commercially valuable stems and regional market demands. In general volumes harvested from Asian dipterocarp forests tend to be around 4-8 trees/ha or equivalent to 20-30m^3/ha, but may be as high as 100m^3/ha in forests where a high proportion of stems are commercially valuable (Whitmore 1990). Extraction and damage in Latin America tend to be less than in SE Asia but this varies among different countries and forest types (Uhl and Viera 1989, Johns 1996, Mason 1996, Webb 1997, Kammesheidt 1998). Logging intensity in Africa is generally the lowest of the three tropical regions due to high costs, low local demand and a limited number of species used for the export market (White 1994, Hawthorn 1995, Chapman and Chapman 1997). The great variability in logging intensity makes it difficult to establish generic recovery patterns of forests.

Damage is also caused to the forest by the extraction process and can be considerable, ranging from approximately 20 to 70% of stem mortality (Kurpick *et al.* 1997). In Asia, where extraction levels are generally high, the damage to the remaining stand is about 50% for conventional logging operations (Kurpick *et al.* 1997). Reduced impact logging (RIL) techniques (directional felling, planned skid trails etc.) can reduce damage to the remaining stand by 50% (Sist *et al.* 1998). However, the relationship between volume harvested and damage may not always be linear. Sist *et al.*

(1998) recorded a large increase in damage when over eight stems/ha were cut in RIL systems.

Studies from Latin America report reductions in canopy cover from 10 to 40% (Uhl and Viera 1989, Johns 1996, Mason 1996, Webb 1997). In Africa canopy cover reduction by logging is relatively low (<10%) (Wilkie *et al.* 1992, White 1994, Hawthorn 1995) while in Asian dipterocarp forest it is high, corresponding to the high levels of extraction (Johns 1996, Pinard and Putz 1996, Sist *et al.* 1998, Abdulhadi *et al.* 1981, Cannon *et al.* 1994, 1998).

Recovery

If logging is light canopy cover can return to pre-disturbance levels relatively quickly by crown extension by remaining trees. Gaps may also be closed by advanced regeneration or fast growing pioneers. Mason (1996) found canopy cover in a Venezuelan forest to be 90.9 ± 7.3% in plots logged 6 years previously (3-7 stems/ha), compared to 71.9 ± 25.5% in plots logged 1 year before, and 94.1 ± 1.5% in unlogged forest, demonstrating recovery within a few years. Chapman and Chapman (1997) found no differences in forest floor light levels (a surrogate measure for canopy openness) between forest logged at 5-7 stems/ha 25 years previously and unlogged forest in Uganda, indicating apparently full recovery of canopy cover by this point.

The recovery of basal area after logging occurs by growth of remaining trees and new recruitment. Increased light levels following canopy opening in logged sites can promote both these processes, but there is evidence that increased growth rates are short lived. Silva *et al.* (1995) found that the basal area of logged forest in the Brazilian Amazon increased from 20.3 to 25.9m²/ha in 2 to 13 years after logging (real-time series) compared to an unlogged basal area of approximately 36m²/ha. Diameter increments in recently logged stands were double those of unlogged forest levels, but growth in stands logged 6-12 years previously were similar to that of unlogged forest implying that the stimulus in growth due to increased light levels lasts only a few years. By extrapolating from this rate of basal area increment we calculate that 30 years is required for this forest to reach pre-disturbance basal area values. Differences in stand structure due to variable logging intensities were still obvious 25 years later at Kibale, Uganda (Chapman and Chapman 1997): basal area of moderately logged stands (5-7.4 stems/ha) was 35-38% of unlogged forest basal area while lightly logged forest (1-3 stems/ha) was 68% of unlogged levels, implying high initial damage and slow recovery.

The time for basal area recovery depends greatly on initial damage. No significant difference in either basal area or stem density between logged and unlogged forest was recorded six years after low-impact logging (8.5 trees per ha) using elephants in the Western Ghats (Pelissier *et al.* 1998). The damage associated with logging in this case was very low and may represent the minimum critical time-threshold for basal area. Stem density was closer to the unlogged state (4.8% difference) than basal area (12.9% difference) implying that stem density recovers faster than basal area. In contrast, basal area in forest logged (10 stems/ha) in Kalimantan using mechanized extraction, was only 50% of unlogged forest basal area values eight years after logging (Cannon *et al.* 1998). Likewise, Kammesheidt (1998), assessing recovery from logging (10 stems/ha) in Venezuelan lowland moist forest, found that basal area increased from 17.8m²/ha at a 5-year-old logged site to 22.2m²/ha at a 19-year-old logged site, which was still only 67% of the basal area of unlogged forest.

Conclusions

Canopy cover can recover relatively quickly (within about 10 years) to values observed in unlogged forest, but can take much longer if disturbance is excessive. Canopy cover can therefore be considered an early threshold verifier within the framework of Table 1. Basal area recovers more slowly and can, perhaps optimistically, be termed a medium-late critical time-threshold verifier (20-30 years).

BIRDS

Verifiers Species Richness
Abundance
Abundance of Selected Avian Guilds

Response to logging

Bird species richness has been found to decline in response to logging (Johns 1991; Thiollay 1997; Marsden 1998) and to increase (Kofron and Chapman 1995; Johns 1996; Table 3). In Sabah higher species richness in disturbed forest was due to an increased number of common edge species (Johns 1996) highlighting the problem of using species richness indicators where all species have equal value. Species richness alone may therefore not be a good indicator of the recovery of forest biodiversity.

The effect of logging on abundance is also variable. Of 117 species recorded in forest in Venezuela, Mason (1996) found that 56% were more common and 44% less common in 1-6 year post-logging (3-7 stems/ha) forest. Thiollay (1997) recorded a 27% decrease in overall abundance in logged forest in the French Guianian Amazon, and Johns (1991) recorded similar changes in the abundance of birds across logged and unlogged forest in the Brazilian Amazon. Mallory and Brokaw (1997), however, concluded that there was no change in bird abundance in Belize 10 months after logging (6 stems/ha).

A number of studies have found species diversity (a measure that combines richness and evenness parameters) to be lower in logged forest and disturbed habitats, due to greater dominance by fewer generalist species. Marsden (1998) found significantly higher species diversity (Shannon index) in unlogged forest in Serum compared with 6-year-old logged forest. In Mason's 1996 study in Venezuela, species diversity in unlogged forest was higher than 1-year-old logged forest (7 stems/ha) but lower than 5-year-old logged forest (with logging intensity of 3 stems/ha). Thiollay (1997) recorded only slight differences in species diversity following logging (3 stems/ha) despite large changes in species richness.

In general, forest avifaunas appear to lose few species due to logging. However, certain species may be absent from logged forest. In Liberia 38% of forest interior specialists were not present in selectively logged forest (Kofron and Chapman 1995). Thus, differences between unlogged and logged forests in bird species composition are often clearer when individual guilds are examined. The abundance of insectivorous birds, for example, generally declines in disturbed habitats, while nectarivore abundance and species richness increases (Mason 1996; Canaday 1997).

The insectivore guild can be subdivided into several categories that respond to disturbance based on foraging location. Most studies (Table 4) report canopy foliage-gleaners to be less common in disturbed forest, although Andrade and Rubio-Torgler (1994) found no differences across primary and secondary forest. Bark-gleaners also tend to be less abundant in logged forest (Johns 1989, 1996, Owiunji and Plumptre 1998), with some exceptions (Andrade and Rubio-Torgler 1994). Terrestrial species show the greatest reductions in abundance, although Lambert (1992) found approximately equal numbers in logged and unlogged forest.

Recovery

The available data suggest a relatively slow recovery of bird species communities. Comparing 1- and 10-year-old logged plots Thiollay (1997) reported that overall species richness and species diversity were higher in the 10-year plot, but abundance was lower and there was no evidence of recovery by insectivorous species.

At logged forest at Sungei Tekam in Malaysia a high proportion of foliage-gleaning birds had re-appeared in logged forest by 12 years, although terrestrial insectivores showed little recovery (Johns 1989). Re-establishment of an understorey canopy (which occurred by about 10 years in his study) was a major factor in the re-colonization of the logged forest by vulnerable species.

TABLE 3 *Species richness and abundance of bird guilds as a percentage of unlogged forest values*

Site (Disturbance)	Insectivore	Frugivore	Nectarivore	Reference
Amazon (1-10 yrs, 3 stm/ha)	7-28	47-63	65-90 n.s.	Thiollay 1997
	25-66	58-67	57-100 n.s.	
Columbia (edges)	75	79	131	Restrepo & Gomez 1998
Sabah (6-12 yrs, 118m^3/ha)	72-105	100	100-129	Johns 1996
	22-60	228	123	
Amazon (fragment)	-	-	267	Stouffer & Bierregaard 1995
	42	61	137	
Amazon (11yrs, 3-5 stm/ha)	96	92	125	Johns 1991
Uganda (5-40 yrs)	77	300	146	Owiunji & Plumptre 1998
Sabah (8yrs, 90m^3/ha)	83	132	245	Lambert 1992

TABLE 4 *Abundance and species richness of insectivorous birds in logged and secondary forest*

Data presented as % abundance of undisturbed forest values

Location		Terrestrial	Bark-gleaning	Foliage-gleaning	Sallying	References
Sabah	(6 yrs, 118m^3/ha)	15	77	60	153	Johns 1996
	(12 yr, 118^3/ha)	24	53	48	126	
Brazil	(11 yrs, 3-5 st/ha)	114	104	64	142	Johns 1991
Uganda	(5-40 yrs)		49	87	220	Owiunji & Plumptre 1998
Guiana	(1 yr, 3 st/ha)	6	118		38 (b)	Thiollay 1997
	(10 yrs, 3 st/ha)	9	111		3	
Sabah	(8 yrs, 90m^3/ha)	97	105	200	34	Lambert 1992
Malaysia	(0.5 yrs)	38	64	73	85	Johns 1989
Colombia	(1-5 yr)		25	300 (a)	31 (b)	Andrade &
	(7-17 yr)	13	600	27		Rubio-Torgler 1994

(a - canopy foliage gleaners, b - ant followers)

Conclusions

Species richness or measures of overall abundance are *not* good indicators of forest recovery, given that there are no consistent patterns among the studies reviewed. The invasion of logged forest by species adapted to disturbed habitats can offset any loss of forest specialists giving similar or even higher richness and abundance values. Diversity and similarity indices of bird assemblages are less affected by colonizing generalist species but clear trends in initial changes and subsequent recovery remain ambiguous and highly dependent on the nature and intensity of the disturbance event.

Of all bird guilds, insectivores show the clearest responses to logging, with almost all studies showing a decline in species richness and abundance following logging. Nectarivore and frugivore abundances are often higher in logged forest, though frugivore responses are more variable. Insectivores therefore appear to be the best avian group for monitoring forest recovery. Within the insectivore guild different species exhibit different responses – some species may increase even if most decrease. Monitoring may need to focus on select groups of feeding guilds that show greatest sensitivity to disturbance.

The available data indicate that the recovery of insectivores occurs in the mid-long term after logging. Recovery rates of insectivore feeding guilds depends on the state of the vegetation at which they forage. There is no evidence that terrestrial insectivores recover in the short-term, possibly due to the dense understorey and vine tangles that often characterize heavily disturbed forest and may impede movement of these birds.

We therefore propose that foliage-gleaning insectivores represent early-mid time-threshold verifiers (i.e. expected recovery of 10-20 years). Terrestrial insectivores, which may not recover until the understorey has been shaded out, are likely to be mid-late verifiers. Similarly, bark-gleaning species may not recover until late in the rotation. Nectarivores, dependent on short-lived floral resources that are stimulated by canopy opening, are expected to be abundant soon after logging but to decline rapidly as floral resources disappear.

BUTTERFLIES

Verifier Number of large butterfly species

Species richness

Butterflies have frequently been proposed as indicators of environmental change due to their sensitivity to microclimate and the specificity of their plant interactions (Kremen 1992). Forest and non-forest habitats generally have relatively discrete butterfly faunas with few shared species (Beccaloni and Gaston 1995). However, as with birds, conclusions about the richness and abundance of butterflies in disturbed and undisturbed forests vary (Table 5).

Species assemblages and community composition

Pinheiro and Ortiz (1992) identified distinct butterfly communities from gallery forest, savanna-like scrub and early succession vegetation, and distinguished other communities in the ecotones where different habitats meet. Kremen (1992) noted that butterfly assemblages were excellent indicators of topographic/moisture gradients but were less closely associated with gradients in disturbance (25-year-old and 2-year old logged forests). He concluded that the effect of topography on species composition masked the affects of logging. Furthermore, Kremen (1994) found that the genus *Henotesia* (Satyrinae, Nymphalidae) was a better indicator of disturbance than all species grouped together. Beccaloni and Gaston (1995) found a strong positive correlation

TABLE 5 *Percentage change in butterfly species richness and abundance in comparison to undisturbed forest*

Forest type and site	Sp. richness	Abundance	Reference
Monsoon, Indonesia			
logged	78	63	Hill *et al.* 1995
mod disturbed	110	116	Hamer *et al.* 1997
heavily disturbed	142	125	
Semi-evergreen, Trinidad			
disturbed (transect)	127	150	Wood & Gillman 1998
disturbed (trap)	100	807	
Evergreen, Trinidad			
disturbed (transect)	181	140	Wood & Gillman 1998
disturbed (trap)	73	193	
Montane rain, Vietnam			
transition	117	119	Spitzer *et al.* 1993
ruderal	110	177	
gap	147	295	Spitzer *et al.* 1997
Rainforest, Ecuador (Nymphalid fruit feeders - traps)			
logged	103	122	DeVries *et al.* 1997
secondary	118	118	
edge	105	411	

between species of the Ithomiine, another subfamily of the Nymphalidae, and species richness of all other butterflies in Neotropical forests.

Several studies report that endemic and restricted range species are more frequently associated with undisturbed forests (e.g. Thomas 1991, DeVries *et al.* 1997). Montane forest butterflies in Vietnam had smaller geographic ranges than species occupying adjacent agricultural and transition vegetation (Spitzer *et al.* 1993). Hill *et al.* (1995) reported that six species endemic to the Malukus were restricted to undisturbed forest, and Lewis *et al.* (1998) found species richness of endemic butterflies on Grande Comore was highest in mature forest, while richness of non-endemics was highest in disturbed habitats. Although de Vries *et al.* (1997) recorded much the highest abundance in disturbed forest and forest edge habitats, undisturbed forest habitats again had the most rare species.

There are few data on the pattern of recovery of butterfly communities following logging or other disturbance. In the short term (i.e. <5 years), species composition appears to be altered in most cases (Spitzer *et al.* 1997, Kremen 1992, Kremen 1994, Wood and Gillman 1998). De Vries (1987) recorded similar numbers of rare species in 15-20 year-old secondary growth as in primary forest indicating some recovery. At least partial recovery is also suggested in Kremen's 1992 study, as shown by the difference between forest logged two- and 24-years ago; but without direct comparisons with unlogged forest it is impossible to confirm this.

Conclusion

In contrast to birds there seems to be no single guild that stands out as a useful indicator group. This may be due to the relatively few number of feeding guilds among butterfly species reflecting the lower diversity of resources exploited. However, and as with birds, measures of species richness, abundance and even species diversity do not make good indicators of recovery after logging given their erratic response to disturbance.

Responses to disturbance are largely species-specific and butterfly indicators need to be taxonomically defined. Rare and geographically restricted species are more common in closed forest habitat and seem most sensitive to disturbance. These species often belong to species-rich taxa with high beta-diversity which, in general, form good indicator assemblages (Kremen 1992). However, species identification becomes necessary which increases the effort required for assessment. Nevertheless, indicator groups comprising selected genera or families have been identified for some regions, and there is evidence that these may be usefully applied as verifiers of forest recovery. Particular restricted range species or butterflies associated with canopy or open habitats may form potentially useful indicators.

DEAD WOOD AND DECOMPOSITION

Verifiers	volume of standing and fallen dead wood state of decay of dead wood abundance of small woody debris soil conductivity and pH

Dead wood

Dead wood is widely recognized for its importance as a habitat for a variety of invertebrates and as a foraging substrate for vertebrates (Sturtevant *et al.* 1997, Ratcliffe 1994). Selective logging often produces large amounts of woody debris from residues of felled trees and other toppled stems. Mortality rates after logging usually increase as trees damaged in the operation or affected by the opening of the stand die. As the stand structure is altered, leaving fewer large trees, the generation of large-dimension dead wood is reduced until the stand recovers its pre-logging size class distribution. Chronosequence studies in temperate forests have observed a 'U-shaped' response in the abundance of dead wood after clear-felling (Sturtevant *et al.* 1997) (Figure 2). Following logging there is much woody debris that decomposes slowly, but little additional input from the regenerating stand. As the stand ages coarse woody debris (CWD) begins to increase in volume and mean diameter.

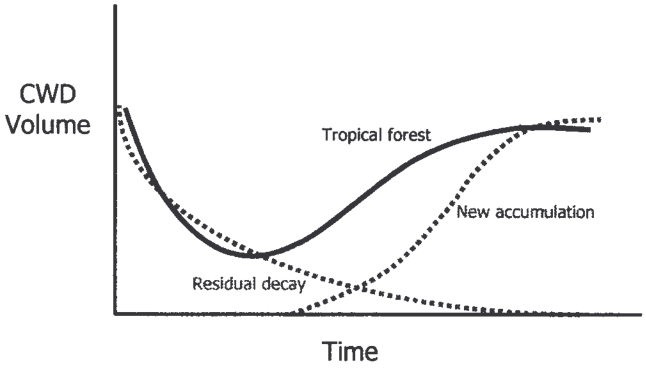


FIGURE 2 *Hypothetical response of CWD after clearfell logging (dashed) and expected response after selective logging in tropical forests.*

In tropical forests at least two tropical studies show U-shaped CWD patterns (Saldarriaga *et al.* 1986, Brown and Lugo 1990). While selective-felling and clear-felling are very different, the response of dead wood volumes is similar (Brown and Lugo 1990). However, after selective logging the residual stand will continue to produce dead wood without the time-lag observed in clear-felled areas. A short-lived initial increase in dead wood after logging due to logging-induced mortality may also be observed. The 'U-shaped' response is therefore likely to be flattened (due to continued production of dead wood by the residual stand) with perhaps a short-lived peak after logging (Figure 2). The time scale for this process is expected to be shorter in the tropics owing to higher decomposition and growth rates. However, this conclusion should be treated with caution given the slow increase in basal area following logging reported in some studies.

Data from temperate and boreal forests suggest that lowest volume of CWD (during which logging residues have disappeared but new production of dead wood is low) will occur at about 20 and 50 years respectively. In the tropics logging-induced mortality will occur for a few (1-5) years after logging while logging residues will have decayed within about 15 years. Recovery of dead wood to pre-logging levels depends on stem density and basal area. The rate of recovery of all dead wood characteristics (volume, size and decay state distributions) will therefore be relatively slow and full recovery under sustainable management is expected by the mid-late stage. The pattern of recovery can be monitored through the distribution of dead wood in different decay state classes.

Decomposition and soil nutrient content

Decomposition rates appear little affected by disturbance except in extreme conditions where canopy opening results in drying of litter and soil. Burghouts *et al.* (1992) recorded no difference in decomposition across logged and undisturbed forest, while Zou *et al.* (1995) found slightly higher decomposition in secondary forest. If this trend is true then significant differences beyond the earliest stages of regeneration would indicate slow recovery. Decomposition might therefore be used as an early time-threshold verifier. Soil acidity and organic carbon also recover rapidly following disturbance and these may be classed as very early stage verifiers. Enright (1978) found that pH, organic matter and nitrogen content of the soil recovered within two years of heavy logging damage in *Araucaria* forest in Papua New Guinea. Other nutrients took longer to recover. In Australia Herbohn & Congdon (1998) found no consistent differences in total nutrient accessions in small litter fall (leaves and woody debris <2cm) between forest logged 25 years previously and unlogged rain forest. Several other studies also show rapid recovery of soil nutrients following disturbance by logging (Folster & de las Salas 1976, Ramakrishan and Torky 1981). Acidity is easy to measure but its link to biodiversity is not as intuitive as other verifiers, and it is important to justify the significance of this and other decomposition verifiers if they are to be adopted enthusiastically. Whether the effort required to monitor decomposition rates and soil nutrients is justified is arguable, although decomposition is clearly an important process in the ecosystem.

DISCUSSION AND CONCLUSIONS

The lack of long-term studies limits our abilities to estimate recovery patterns. The few long-term studies are mostly based on chronosequences where differing extraction rates and methods as well as initial forest conditions complicate comparisons across sites. Most studies fail to report initial damage or logging intensity, so drawing conclusions about recovery are difficult. Likewise, information on the natural variability of verifiers is sparse. Quantifying the natural variation in species diversity and abundance for birds and butterflies is complex and has not been attempted within the scope of this review. Temporal and spatial variation are often different across forest types, and measurement is highly sensitive to sampling effort. High regional variability in verifier states suggest that baseline values be set using data generated within the FMU, although regional averages will provide a quality control mechanism for FMU datasets.

Choice of verifier

In general forest structure verifiers show clear patterns of recovery involving various time-thresholds throughout the regeneration cycle. It is therefore possible to propose critical time-thresholds with reasonable confidence for these verifiers, which are, furthermore, relatively easy to measure.

Species richness and abundance do not, in general, make good verifiers as species respond differently to disturbance. Diversity and similarity indices of particular assemblages show clearer recovery patterns. Among birds, for example, insectivores and nectarivores show clear diversity and abundance trends. Butterfly response patterns are more difficult to identify. Nevertheless, some tentative proposals have been made for critical time-thresholds based the limited data available.

Although there are few data on dead wood in tropical forests there is some evidence for relatively predictable recovery patterns. It is possible to draw conclusions about the nature and time scale to the response of these verifiers (volume, decay, size) from studies in temperate forest, although extension to tropical environments remains to be tested.

Critical time-thresholds for recovery

The concept of a critical time-threshold for recovery provides a framework for estimating the time a particular verifier state may take to recover to the ‘undisturbed state’. Using this framework we have identified trends in recovery of certain verifiers and suggested possible time durations over which they are expected to recover – i.e. critical time-thresholds.

Each verifier can be allocated to a particular critical time-threshold (as in the categories in Table 1), and so each time-threshold category will represent a suite of indicators to assess sustainability at that point in the regeneration cycle. This allows for a system of continual assessment. In proposing critical time-thresholds we suggest that scientific exactitude must be balanced with pragmatism as to what data are available and what is feasible for forest managers to accomplish.

Future research requirements

The review of the literature for this paper has been intensive but is by no means comprehensive. Especially for forest structure verifiers, and to a lesser extent forest dynamics verifiers, there will be much more information available in the 'grey' and unpublished literature as well as other published studies not included in this review. However, it would seem that long-term recovery data will always be scarce, and this gap will only be filled by further field work.

A Bayesian statistical analysis of expert opinions on verifier critical time-thresholds will allow integration of the Precautionary Principle with data available in published literature. This will be especially important in areas where the literature suggests patterns of long-term response but where data to support or reject hypotheses are limited. A good example is bird species verifiers: there are clear indications of unidirectional patterns of recovery in various guilds but little data on long-term recovery.

Field tests of verifiers should be designed bearing in mind current gaps in the literature. Time series studies should include the whole cutting cycle and not just the first few years after logging. This is problematic given that mechanized logging is comparatively recent in some areas, and that it is hard to control subsequent legal or illegal extraction or conversion to non-forest uses. The temptation is to concentrate on the more recently logged stands that are easier to identify and give more accurate results, albeit of limited value.

SUMMARY

Table 6 gives the proposed critical time-threshold of each verifier and ranks their usefulness in monitoring biodiversity based on the following characteristics: clarity of response (in a single direction in most studies), feasibility of predicting recovery period from available data, likelihood that further data would confirm recovery patterns if data were scarce, and feasibility of predicting recovery period from general ecological information available if data were scarce. Verifiers were ranked as positive potential (+), value unclear or doubtful (?) and of little or no use (-).

TABLE 6 *Review of selected ecological verifiers from the CIFOR ecological C and I set including some additional verifiers discussed in detail elsewhere (Ghazoul unpublished)*

Verifier	Value	Critical time-threshold	Notes
basal area	+	mid-late	
density of seedlings	+	very early	
density >10cm DBH	+	early-mid	
density >60cm DBH	+	late	
canopy openness	+	early	
bird species richness/abundance	-	n/a	
bird species diversity	?	?	
insectivore richness/abundance	+	mid-late	guilds recover at different rates
foliage gleaning species	+	early-mid	
terrestrial and sallying species	+	mid-late	
bark-gleaning species	+/?	late	
nectarivore richness/abundance	+	early	
butterfly richness/abundance	-	n/a	
butterfly species of open habitat	?	early	further research needed
restricted range butterfly species	?	?	as above
volume of dead wood	+	all stages early to late	
abundance of small woody debris	?	?	
soil pH	?	(very early)	

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