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FIVE YEARS OF POST-FIRE VEGETATION SUCCESSION IN A CARIBBEAN CLOUD FOREST (CORDILLERA CENTRAL, DOMINICAN REPUBLIC)

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Resumen. Después de un incendio provocado en un bosque nublado de la Cordillera Central Dominicana, la composición de especies fue monitoreada durante cinco años, así como la cubierta y la altura de la vegetación. En el desarrollo de la cubierta vegetal se observó una fase de inercia de un año a año y medio, antes de incrementar hasta valores de aproximadamente 90%, alrededor de cinco años después del incendio. En la vegetación succesional, árboles colonizadores y arbustos eran dominantes, y la inmigración de especies del bosque nublado maduro fue baja. Las tasas de crecimiento longitudinal de las especies de árboles colonizadores alcanzaron valores de hasta 1 m por año. La riqueza de especies llegó a un pico, alrededor de tres años después del incendio. Después, las especies colonizadoras de poca longevidad desaparecieron, y la riqueza de especies descendió.

Abstract. During the 5 years after a human-made fire in a cloud forest of the Cordillera Central of the Dominican Republic, species composition, vegetation cover and height were monitored in a plot of 10 x 30 m. Forty-four of 92 tree and shrub individuals, representing 8 of 22 species, survived the fire, mainly by sprouting from subterranean buds. Development of vegetation cover showed a lag phase of 1 to 1.5 years and then increased up to about 90% five years after the fire. Colonizing tree and shrub species were dominant in the post-fire vegetation, while immigration of species from mature cloud forest was very low. Longitudinal growth rates of colonizing tree species reached values of up to 1 m per year. Species richness reached a peak about three years after the fire. Afterwards, short-lived colonizing species disappeared and species richness decreased. *Accepted 3 November 2000.*

Key words: Cloud forest, post-fire succession, colonization, vegetation cover, species richness.

INTRODUCTION

In permanently humid environments of the tropics natural fire frequency is low, due to the almost continuously high humidity of air, soil and plant biomass (Mueller-Dombois & Goldammer 1990). However, in many parts of the humid tropics man-made fires are today an important factor in shaping ecosystems and landscapes. In the Dominican Republic, mountain ranges with cloud forest and pine forest vegetation are much more affected than the created landscapes of the densely populated flat valleys, which are used for intensive agriculture and livestock. In these mountain ecosystems, fire is used for forest clearing or slash-and-burn agriculture, and sometimes fires are set in the context of social and political conflicts.

It is accepted that fires cause damage in mountain forest ecosystems (Koonce & González-Cabán 1990). However, little detailed information is available about the effects of fire on vegetation cover, structure and composition, and about the temporal dynamics of vegetational responses to fire in the broadleaved mountain forests of the Caribbean zone. In this paper the results of the monitoring of the first 5 years of vegetation succession after a human-made fire in a cloud forest of the Dominican Cordillera Central are presented. This monitoring was carried out in the context of an investigation program in the Scientific Natural Reserve Ebano Verde, a protected cloud forest area which is managed by the Fundación PROGRESSIO, in coordination with the National Park Administration of the Dominican Republic (DNP).

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STUDY SITE AND FIRE CONDITIONS

Due to their exposure to the north-eastern trade winds, the northern and eastern slopes of the Dominican Cordillera Central are covered with cloud forest, at altitudes of approximately 1000 to 2000–2200 m. In the central and eastern parts of this mountain range these cloud forests are dominated mainly by *Didymopanax tremulus* (= *Schefflera tremula*) and *Magnolia pallescens* (Hager & Zanoni 1993, García *et al.* 1994). Other common tree species are *Byrsonima lucida, Clusia clusioides, Cyrilla racemiflora, Haenianthus salicifolius* var. *obovatus, Ocotea leucoxylon* and, on slopes and in valleys, the palm *Prestoea montana* (= *Prestoea acuminata*). Patches of fern thicket dominated by *Dicranopteris pectinata* are present in many parts of these forests, and are considered by some authors to be caused by fires of human origin (Ciferri 1936, García *et al.* 1994, Dalling 1994, the latter referring to Jamaica).

The study area was located within this cloud forest zone, at an altitude of approximately 1300 m, about 2 km south of the peak of Casabito, near the limit of the Scientific Natural Reserve Ebano Verde (Fig. 1). Mean annual precipitation is estimated to be in the order of 3000 mm, without a dry period, and mean annual temperatures are in the range of 16–18 degrees C, with relatively low annual and daily fluctuations, according to data at Casabito from July 1993 to June 1997. In the summer of 1992, several hectares of forest and woodland were burned by a human-made crown fire on a steep southern slope. Distribution of dead trunks and other remnants of burned vegetation suggested that before the fire there had been a closed forest stand on the ridge and upper part of the slope, and an open woodland with fern thicket in the understory on the lower part of the slope and in the valley bottom. A similar landscape pattern is frequently found in the surrounding area due to human penetration and logging along the valleys, and possibly also due to landslides triggered by fluvial erosion in the

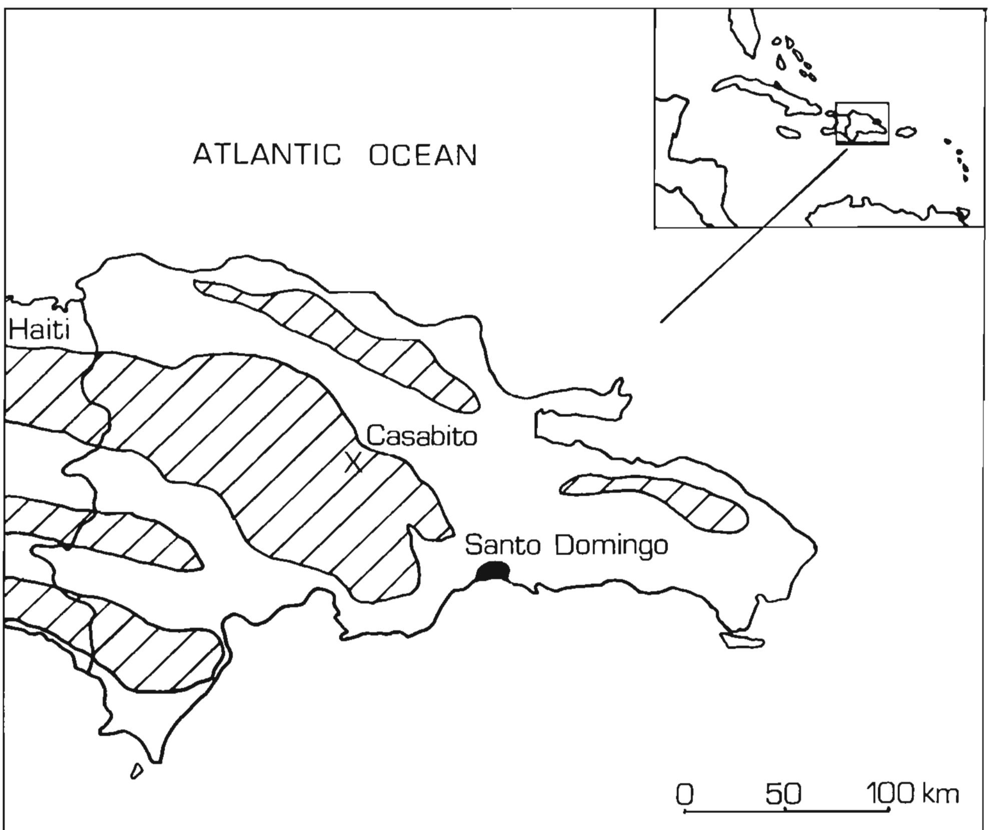

FIG. 1. The study area.

valleys. According to local people, the fire was stopped after some hours by heavy rainfall. Our observations in March 1993 showed that nearly all leaves and small branches of trees and shrubs were consumed by fire, as well as the above-soil biomass of the herbaceous vegetation.

METHODS

In March 1993, on the burned area a plot of 10 x 30 m was marked and divided into 75 subplots of 2 x 2 m. Remnants of trees which had been present before the fire were identified with the help of local personnel, using local names. Three out of 19 local names could not be matched to scientific names, but differences in appearance and in the aspect of the bark and of the wood suggested that the three taxa that could not be identified were different species. All remnants of trees were counted and classified according to their DBH (diameter at breast height, 10 cm or more/less than 10 cm).

Living plants present in the plot were counted, and species identified. The height of each tree and shrub individual was measured. In each 2 x 2 m subplot cover values for each species present were estimated, according to a modified Braun-Blanquet scale (0–1%: +, 1–5%: 1, 5–25 %: 2, 25–50%: 3, 50–75%, 4, 75–100%: 5). Mean cover values for each species were calculated for the whole plot. Overall vegetation cover for each subplot was estimated on a scale of 1–10 (0–10%: 1, 10–20%: 2, and so on), and overall vegetation cover of the whole plot was calculated as the mean value of all subplots. The same procedure of vegetation monitoring was repeated in July 1993, November 1993, March 1994, July 1994 and November 1994. Subsequently, monitoring continued in an annual pattern (November 1995, November 1996 and November 1997). Since it became extremely difficult to count individuals of herbaceous species and climbers from November 1995 onward, due to the presence of creeping species and the increase in vegetation cover, the procedure was modified: individuals of herbaceous species, herbs and climbers were not counted, but only cover values were assigned to these species for each subplot, calculating the cover of each species for the whole plot as the mean value.

From November 1996 onward it became difficult to measure the exact height of all tree and shrub individuals because of the height they reached. For this reason, all individuals were assigned to height classes (0–1m, 1–2 m, 2–3 m, and so on), and only height classes were taken into account for the monitoring of vertical structure and vertical growth. In addition, from November 1995 onward DBH was recorded for each tree or shrub individual with a diameter at breast height of more than 2 cm.

At various times and at various sites of the burned area, dominant species and vegetation structure were checked visually to ascertain that the 10 x 30 m plot was representative of the whole area. Taxonomically, in this paper we follow Liogier (1981) and Liogier (1983–1996) for spermatophytes and Proctor (1989) for ferns.

RESULTS

Structural parameters. The development of total vegetation cover showed a lag phase during the first 15–16 months, until November 1993. After this point a steady increase in vegetation cover was observed, reaching 84% in November 1996, 52 months after the fire (Fig. 2, above). Further increase was slow. From November 1994 a distinction could be made between a herbaceous layer and a shrub layer, comprising individuals taller than 1.5 m. Cover of this shrub layer increased rapidly and was 46% in November 1997.

Although vegetation cover was relatively poor during the first 1–2 years of post fire succession, no evidence of soil erosion was observed, probably due to the presence of an organic topsoil horizon which was not or only slightly altered by fire and which had preserved its structure and infiltration capacity.

Of 73 individual shrubs and trees, excluding tree ferns, that were recorded on the burned plot in March 1993, 18 (= 25%) had survived at the end of the period in November 1997. All of them had sprouted from subterranean regeneration buds or from buds located no more than 10 cm above ground. Adding 30 individuals of shrubs which had not been recorded in March 1993 because their above-ground biomass had been consumed by fire, but subsequently had sprouted from subterranean buds, the survival rate was 30% in November 1997. This means that mortality was 75% and 70% respectively.

The individual numbers of trees and shrubs increased rapidly during the first two years after the fire, reaching a first peak of 766 stems in July 1994, corresponding to a density of 25 533 individuals/ha (Fig. 2, below). Between July and November 1994 individual numbers decreased to 685 stems (22 833/ha), afterwards increasing again slowly to a second maximum in November 1996. Between November 1996 and November 1997 a marked decrease was observed, and the total number of stems had fallen to 596 (19 867/ha), the lowest value since March 1994 (Fig. 2, below).

The temporal sequence of the height class distribution of trees and shrubs is shown in Fig. 3. At the end of the monitoring period the tallest individuals had grown to 4–5 m. While initially most individuals belonged to the lowest height class (0–1 m), from November 1996 individual numbers were highest in the class of 1–2 m. Numbers of individuals smaller than 1 m had begun to decrease from November 1994, which indicates that the colonization rate of the area by tree and shrub individuals had begun to slow down, three to four years after the fire event.

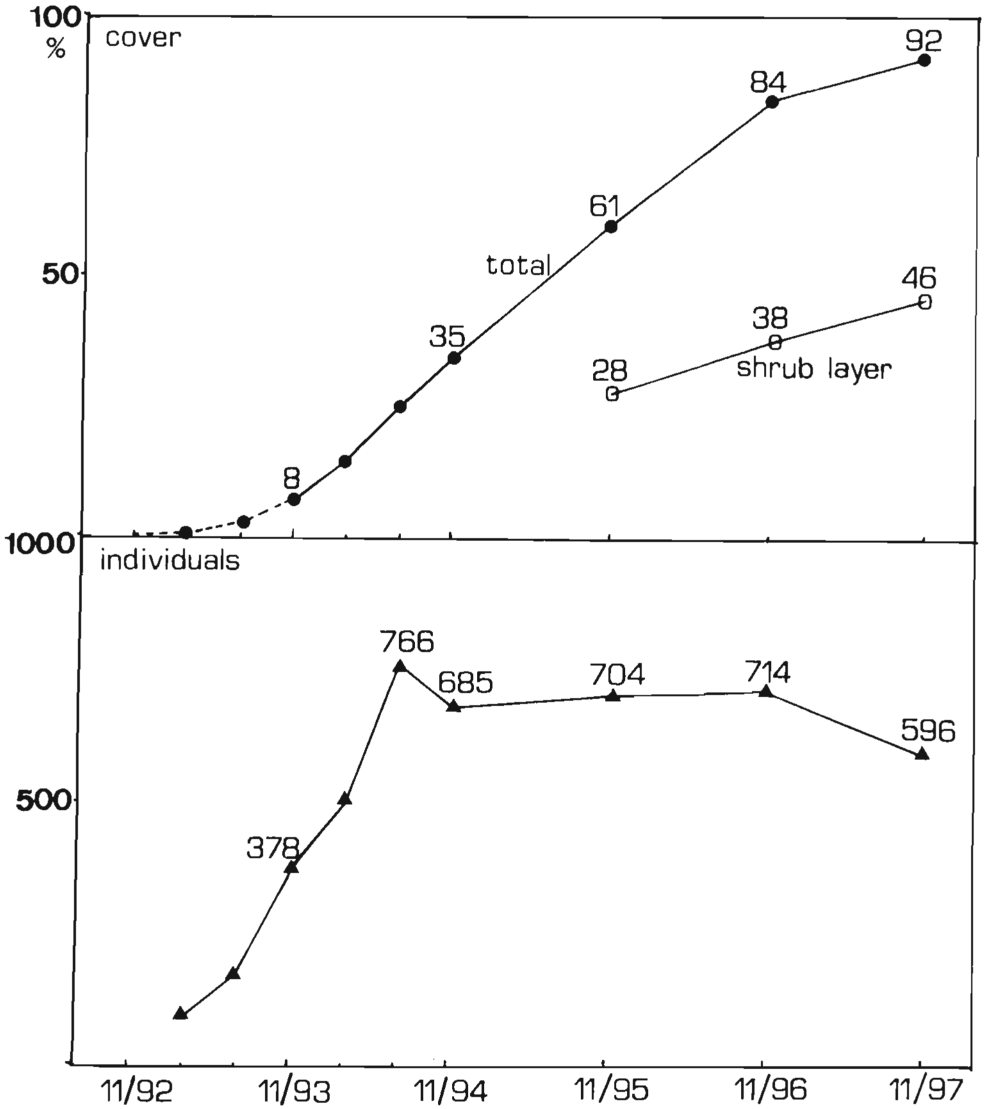

FIG. 2. Above: development of vegetation cover; below: individual numbers of tree and shrub species.

In November 1995, five individuals (= 1.5 % of the 340 individuals of tree species present in the plot) had reached a DBH of 2 cm or more. One year later, in November 1996, 20 individuals (5.3% of 379) had grown to this diameter class, and in November 1997, at the end of the monitoring period, 45 individuals (13.9% of 332) had reached a DBH of 2 cm or more. The DBH of one of them exceeded 5 cm at this time.

Floristic aspects. Of the 19 tree and shrub species (excluding tree ferns) that had been present on the plot before the fire event and had been recorded as remnants in March 1993, only 6 (= 32%) were present at the end of the monitoring period, implying a loss of 68% of the species. Adding three species which had not been recorded as remnants in March 1993 because their above-ground biomass had been entirely consumed by fire, but had subsequently sprouted, the species survival rate was 36% and the loss 64%.

Sprouting after fire was highly selective for species (Chi squared test significant at the 0.1% level). Fifteen of the 18 individuals that had been recorded as remnants in March 1993 and had sprouted subsequently were of the species *Ocotea leucoxylon*, *O. foeniculacea*, and *O. nemodaphne*. Sprouting was also selective for individuals with a DBH of less than 10 cm (Chi squared test significant at the 2.5% level). Only two of the 18 surviving individuals mentioned above had a DBH of 10 cm or more.

Four of the nine tree and shrub species that had resprouted and were present in November 1997 are endemic to La Hispaniola (*Psychotria plumierii*, *Hieronyma montana*, *Tabaebuia vinosa*, and *Magnolia pallescens*). In the case of *Magnolia*, only one out of 18 individuals that had been recorded as remnants in March 1997 survived the fire event. Other tree species that are common in the nearby cloud forest (*Cyrilla racemiflora*, *Didymopanax tremulus*, *Haenianthus salicifolius*) did not show vegetative regeneration from individuals that had been present before the fire event. Except for *Ocotea leucoxylon*, one of the species with the highest sprouting rate, and *Psychotria plumierii*, regeneration buds were located mostly or exclusively

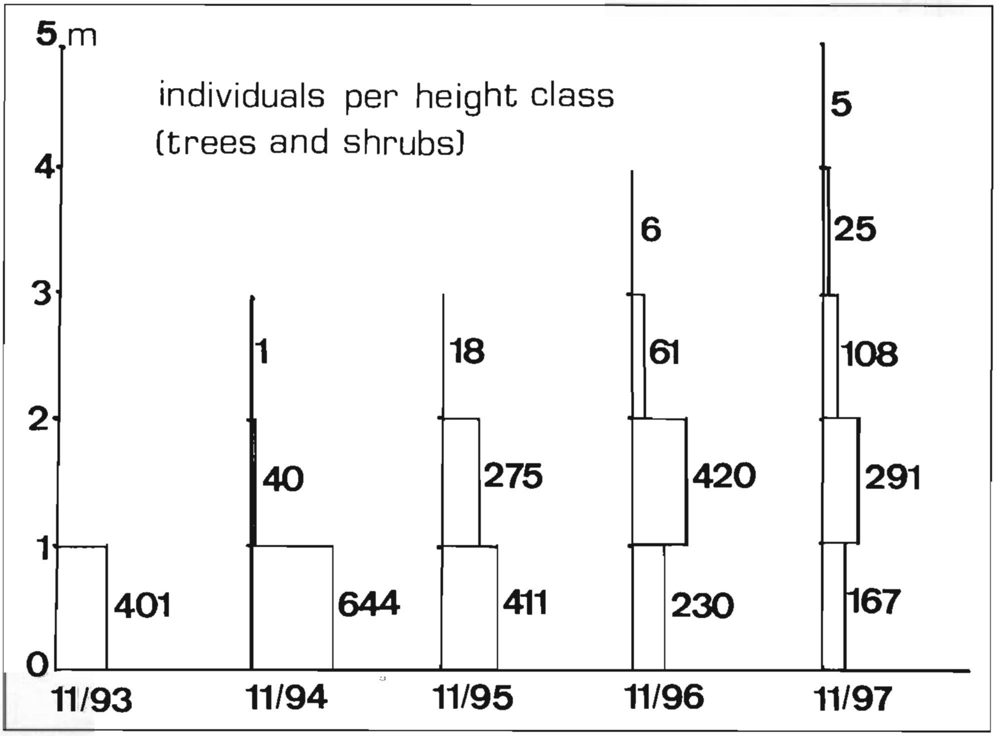

FIG. 3. Individual numbers per height class (trees and shrubs).

above ground (May 1997a), where temperatures are high and damage to the regeneration tissues must be expected. Some individuals of the palm Prestoea montana on sites on the lower part of the slope, outside of the plot, as well as some individuals of tree ferns (*Cyathaea* sp., *Alsophila* sp.) were observed to sprout in the crown. This feature was also observed in some individuals of broad-leaved tree species (*Magnolia pallescens, Cyrilla racemiflora, Chaetocarpus* sp.), but only near the edge of the burned area, where fire temperatures are expected to have been lower.

The total species number increased rapidly until July 1994, approximately two years after the fire, and continued increasing more slowly until reaching a peak in November 1995. After this point, species numbers declined (Fig. 4, above).

Changes in species numbers during the succession process can be regarded as a result of the number of species present in the previous observation, number of species lost, and number of immigrant species. Numbers of immigrant species were high until November 1994 and then slowed down, whereas numbers of species lost peaked in November 1995 and then remained nearly constant (Fig. 4, below).

From July 1993 until the end of the study period, the colonizing trees *Brunellia comocladifolia* and *Myrsine coriacea* were among the most abundant species, whereas the shrub *Psychotria berteriana* was the species with the highest individual numbers from November 1993 to November 1996 (Table 1). The tree *Trema micrantha* was very abundant at the beginning of the period, but its population began to decrease in

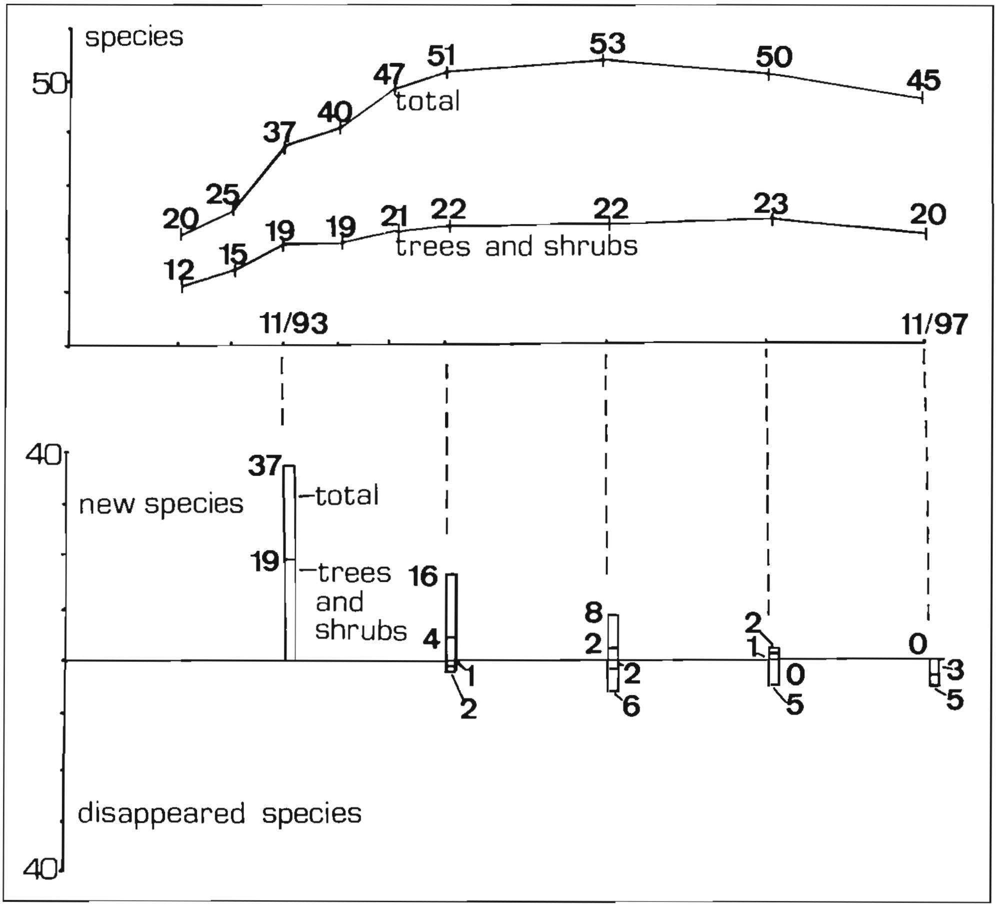

FIG. 4. Above: total species numbers and numbers of tree and shrub species; below: new species and lost species.

TABLE 1. Populations of tree and shrub species.

	numbers of individuals								
Taxa	3/93	7/93	11/93	3/94	7/94	11/94	11/95	11/96	11/97
Solanum torvum S	7	3	3	2	1	1	.	.	.
Bocconia frutescens S	.	1	2	1	1	1	.	.	.
Trema micrantha S	31	68	73	66	53	46	13	4	.
Brunellia comocladifolia S	20	37	68	67	92	92	81	70	67
Baccharis myrsinites S	1	4	5	24	34	39	66	56	65
Myrsine coriacea S	.	23	64	68	186	141	147	185	161
Solanum crotonoides S	.	1	3	3	3	3	3	7	6
Psychotria berteriana S/R	1 (R)	.	83	149	239	123	202	207	145
Solanum rugosum S	.	.	24	37	47	49	56	35	10
Clidemia umbellata S	.	.	7	16	30	20	49	47	39
Cecropia peltata S	.	.	6	11	15	17	16	19	17
Miconia dodecandra S	.	.	.	.	2	3	8	14	14
Miconia laevigata S	.	.	.	.	1	1	1	3	3
Byrsonima lucida S/R	4 (R)	.	.	.	.	1	1	1	.
Rondeletia conferta S	.	.	.	.	.	.	1	1	.
Palicourea eriantha S	.	.	.	.	.	.	.	4	6
Cyathaea spec. R	20	21	23	24	24	24	24	24	24
Ocotea leucoxylon R/S	11	16	14	14	14	12	11	17	17
Ocotea foeniculacea R	4	5	12	10	10	7	8	8	10
Psychotria plumierii R	.	4	7	7	8	9	10	8	5
Hieronyma montana R	.	.	1	4	3	3	3	3	3
Tabaebuia vinosa R	.	2	1	1	1	1	1	1	1
Ocotea nemodaphne R	.	1	1	1	1	1	1	1	1
Magnolia pallescens R	1	4	4	.	.	.	1	1	1
Myrcia splendens R	4	.	.	2	1	1	1	1	1

S: recovery by seedlings, R: recovery by vegetative regeneration (sprouting).

March 1994, and by November 1997 this species had completely disappeared from the plot. A similar pattern was observed in the tall herbaceous species *Phytolacca icosandra* and *Erechtites valerianaefolia*, which dominated the aspect of the vegetation from July 1993 to July 1994 and then disappeared completely between November 1994 and November 1995. In contrast to *Trema*, the two latter species flowered and fruited, while in *Phytolacca icosandra* heavy insect damage was observed in July 1994, which possibly played a role in the rapid decline of the population of this species. From November 1995 onwards another herbaceous species, *Rhynchospora elongata*, dominated the herbaceous layer of the vegetation, whereas in the shrub layer *Baccharis myrsinites* became co-dominant, together with *Brunellia* and *Myrsine*.

The fern *Dicranopteris pectinata*, which forms dense thickets in the area that are said to be due to fire (Ciferri 1936, García *et al.* 1994), never played a prominent role in the post-fire vegetation during the whole observation period (Table 2). Regeneration buds of this species survived in some concave microsites that were starting points of vegetative colonization. Colonization by individuals that had developed from spores was not observed in the plot.

Tree ferns (*Cyathaea* sp.) were present in the plot from the beginning, but their vertical growth was very slow. The stems of the tallest individuals reached about 40 cm in November 1997, five years after the fire, which is equivalent to an annual average increase of only 8 cm. Colonization of the plot by seedlings of shade-tolerant species, which are present in well-developed cloud forests nearby, was poor, except for the shrub *Psychotria berteriana*, which is also common in secondary growth pioneer forests (May 1994), and *Rhynchospora elongata* (Cyperaceae), which is abundant in nearby cloud forest. Some individuals of *Ocotea leucoxylon* and *Palicourea eriantha* were observed

TABLE 2: Abundance/cover values of herbaceous species, climbers and epiphytes.

Taxa	3/93	7/93	11/93	3/94	7/94	11/94	11/95	11/96	11/97
Phytolacca icosandra S	1	2	2	2	1	1	.	.	.
Erechtites valerianaefolia S	+	1	2	2	2	1	.	.	.
Emilia fosbergii S	.	+	+	1	1	+	+	.	.
Cyperus sphacelatus S	.	.	+	+	+	+	+	+	.
Portulacca oleracea S	.	.	+	+	.	.	.	.	.
Sida cf. *rhombifolia* S	.	.	.	+	+	+	.	.	.
Gnaphalium purpureum S	.	.	.	+	+	1	+	.	.
Conyza canadensis S	.	.	.	.	1	1	+	.	.
Eupatorium odoratum S	.	.	.	.	+	1	+	.	.
Dilomilys montana S	.	.	.	.	+	+	+	.	.
Ampelocissus robinsonii S	.	.	.	.	1	+	.	.	.
Potomorphe peltata S	.	.	.	.	.	.	+	+	.
Smilax havanensis R	1	1	1	1	1	1	1	1	1
Dicranopteris pectinata R	+	1	1	1	1	1	1	1	1
Odontodenia polyneura S	+	1	1	1	1	1	1	1	1
Passiflora sexflora S	+	1	2	1	1	1	1	1	+
Ipomoea furcyana S	+	+	+	+	+	+	+	+	+
Rhynchospora elongata S	.	1	2	2	2	2	3	3	2
(unknown fern) R	.	1	1	1	1	1	1	1	1
Lobelia rotundifolia S	.	.	1	1	1	1	1	1	1
Iresine diffusa S	.	.	1	1	1	1	1	1	1
Coccocypselum herbaceum S	.	.	1	1	1	1	1	1	1
Ichnanthus pallens S	.	.	1	1	1	1	1	1	+
Pteridium aquilinum R	.	.	+	+	+	+	+	+	+
Homolepis glutinosa S	.	.	.	1	1	1	2	2	2
Andropogon bicornis S	.	.	.	.	1	1	1	1	1
Cissampelos pareira S	.	.	.	.	+	+	+	+	+
Odontosoria uncinella S	.	.	.	.	+	+	+	+	+
Tetrazygia crotonoides S	.	.	.	.	.	1	1	1	1
Gleichenia bifida R	.	.	.	.	.	1	1	1	1
Odontosoria aculeata S	.	.	.	.	.	.	1	1	1
Blechnum tuerckheimii S	.	.	.	.	.	.	1	1	1
Mikania sp. S	.	.	.	.	.	.	+	+	+
Spermacocce verticillata S	.	.	.	.	.	.	+	+	+
Blechnum occidentalis S	.	.	.	.	.	.	+	+	+

S: recovery by seedlings, R: recovery by vegetative regeneration (sprouting).

to colonize the plot by seeds, in addition to some individuals of *Renealmia jamaicensis*, a herbaceous species found in well-developed forest nearby. One single individual of the small tree *Rondeletia conferta*, an endemic species, was observed, as well as of the tree *Byrsonima lucida*. Both individuals later disappeared.

DISCUSSION AND CONCLUSIONS

Five years after the fire, species composition of the pioneer vegetation was very different from that of nearby mature forest. None of the dominating pioneer trees and shrubs was present in significant numbers in mature cloud forest vegetation. Only a few species that were typical of a well-developed cloud forest and were present around the site were observed, and individual numbers of these species were low. This is in line with the results of Miyagi *et al.* and Tangawa *et al.* (in Goldammer & Seibert 1990) from lowland dipterocarp forests in East Kalimantan, where post-fire successional vegetation was composed mainly of pioneer species, and dipterocarp species dominating the undisturbed forests were rare. In this sense, tropical broad-leaved forests are very different to some fire-adapted communities such as sclerophyll scrub and woodlands from the Mediterranean basin, where within a time span of some years, pre-fire species composition is reestablished (Trabaud 1987).

Tree and shrub species dominating the successional vegetation five years after the fire had been present as small seedlings during the initial stages of the succession process. On the other hand, the herbs that dominated the first succesional stages (*Phytolacca icosandra, Erechtites valerianaefolia*), as well as some other small herbaceous species present in these stages (*Emilia fosbergii, Cyperus sphacelatus, Conyza canadensis*, and others) are frequently observed in nearby recently disturbed areas such as new landslides or agricultural land abandoned for a few years, whereas they are virtually absent from mature cloud forest and are only occasionally found in secondary forest. In our burned area, they were eliminated after a few years, as well as some tree and shrub species (e.g., *Trema micrantha*). The initial colonization and subsequent decline of ephemeral, shade-intolerant species is reflected by a peak in total species number approximately three years after the fire. The decline in individual numbers after reaching a peak is probably due to the beginning of intra- and/or interspecific competition, four to five years after the fire, at total cover values of 80 to 90%, and a shrub layer cover of about 40%. Similar patterns – a peak and subsequent decline in numbers of both individuals and species during early succession stages – are reported by many authors, such as Hunter (1989) from Costa Rica after clear-cutting. This author interprets the rapid colonization of the plot by opportunistic species and/or the rapid sprouting as a wound-healing process which takes place during the first stages of succession.

In this sense, our results suggest that post-fire succession in this ecosystem follows the pattern of a "competitive hierarchy", as observed by Swaine & Hall (1983) in their study. However, as species composition five years after the fire was very different from the mature forest surrounding the burned area, it must be supposed that a long-term succession process takes place which follows another pattern. Post-fire secondary forest is dominated by the relatively short-lived *Brunellia comocladifolia* and *Myrsine coriacea*. There is evidence from older secondary forest stands nearby that *Brunellia* begins to decline after 30–40 years (May 1997c). In these forests, other, more shade-tolerant species which are not observed in the younger stands of secondary forests, develop in the understory. This suggests that these secondary forests will be replaced by other communities, more closely related to mature forest.

The height of the shrub layer indicates a longitudinal growth rate of the dominant species *Brunellia comocladifolia* and *Myrsine coriacea*, as well as of *Cecropia peltata* and *Miconia dodecandra*, of up to one meter per year. Diameter growth rates of the fastest growing individuals are slightly below 1 cm per year. Both values are in agreement with the estimations of May (1994) for the first two species in an old field secondary forest in the same cloud forest area. It has to be stressed that growth rates of some species were considerably lower, such as of the tree ferns, which reached maximum values of longitudinal growth of 8 cm per year. Longitudinal growth rates of the relatively fast-growing species *Brunellia* and *Myrsine* are considerably below the growth rates observed by Swaine & Hall (1983) of up to 4 m per year for secondary tree species after clearing in a West African evergreen upland forest. This difference probably is due to the lower temperatures in the cloud forest zone of Casabito.

As mentioned above, Ciferri (1936) and more recently García *et al.* (1994) refer to fern thickets from the Dominican Cordillera Central as communities that were created by fires, as does Dalling (1994) from the Blue Mountains in Jamaica. Although *Dicranopteris pectinata*, the dominant species of these communities in the Dominican Republic, and *Gleichenia bifida*, mentioned by Dalling from Jamaica, are both present in our plot, obviously the fern thicket present in the lower part did not resist the fire. Colonization and/or sprouting of both species was poor, and there is no evidence that *Dicranopteris pectinata* or *Gleichenia bifida* will cover considerable areas of the burned stand in the near future. Tree species composition, as well as their height and growth rates, suggest that in our case succession will lead first to a secondary forest similar to the stand on an abandoned agricultural plot studied by May (1994), and not to a fern thicket. Obviously fire is not the only condition required by the thickets of *Dicranopteris pectinata*. Relatively slow resprouting and absence of post-fire colonization of this species makes it doubtful if these communities are directly related to fire at all. In the light of the results presented here it might even be possible to eliminate *Dicranopteris* thickets by prescribed fire, and to substitute them by pioneer tree communities.

In the Casabito area it was observed that the cloud forest trees *Magnolia pallescens, Didymopanax tremulus*, and *Cyrilla racemiflora*, which sprouted only poorly and did not colonize the monitoring plot, sprout vigorously from aerial buds after mechanical disturbance, while the same occurred on the border of the burned area, where fire temperatures had been lower and the aerial buds had not been affected. Sprouting from roots provides better opportunities of surviving intense fires than sprouting from aerial buds (Kauffman & Uhl 1990), due to the much lower temperatures during fire some centimeters below the soil surface compared with temperatures at soil surface (Suárez de Castro 1957 and Zinke *et al.* 1970, quoted in Sánchez 1976; Coutinho 1990). Sprouting from subterranean buds is not common in most cloud forest species at Casabito (May 1997a). This holds for the three species mentioned above, and might be related to the shallow and extensive root system in the case of *Magnolia* (May 1997b). In contrast, the *Ocotea* species, which showed much higher sprouting rates, regenerated mainly from subterranean buds (May 1997a).

On the other hand, the capacity of mature cloud forest species such as *Magnolia* and *Cyrilla* to sprout from aerial buds as a response to mechanical disturbance suggests that they are probably better adapted to destruction caused by hurricanes and the severe rains and landslides associated with such events (Larsen & Torres Sánchez 1992), such as defoliation, breaking of stems and twigs, and uprooting (Dittus 1985). In fact, Fangi & Lugo (1991) from Puerto Rico and Whigham *et al.* (1991) from Yucatán, two areas where hurricanes are relatively frequent, observed a low mortality rate in trees of 3% and 13% respectively, in contrast to the mortality rates of 70% and more in the present case. In contrast to fires (without the presence of humans), hurricanes and heavy storms can be expected to occur at comparatively shorter intervals in mountain and lowland forest ecosystems in the Caribbean.

The lack of evidence for soil erosion during the first post-fire succession stages, when vegetation cover was still poor, makes it questionable if the assumptions of rapid soil erosion caused by a single forest fire are realistic under cloud forest conditions, where there is often an organic soil horizon of some centimeters that is difficult to alter by crown fires alone. Probably this is different in the case of repeated fires and accelerated loss of organic soil matter, or in the case of post-fire cultivation and/or grazing by cattle.

With regard to ecosystem management, the recovery rates of vegetation cover and vertical structure, as well as the absence of evidence for soil erosion, make it doubtful if reforestation of cloud forest sites, after one single fire, aimed at soil and water protection would be useful. On the other hand, the poor resistance to fire and the slow immigration rates of most of the mature forest species suggest that in cloud forest ecosystems the medium-term effects of a single fire on species richness might be much more severe than on soil and hydrological conditions. Thus an artificial reintroduction of late successional species in the developing post-fire secondary vegetation could be an important measure in restoring local biodiversity.

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