TROPICS Vol. 23 (2) 83-90 Issued September 1, 2014

ORIGINAL ARTICLE

Secondary succession of mixed plantations established to rehabilitate abandoned pasture in the Peruvian Amazon

Shigeo Kobayashi[1*], Manuel Soudre[2] and Arberto Ricse[2]

[1] Graduate school of Asian and African Area Studies, Kyoto University
[2] Instituto Nacional de Innovación Agraria (INIA), Peru
* Corresponding author : skobayashi@asafas.kyoto-u.ac.jp

ABSTRACT Secondary succession or facilitation processes carried out at sites established for rehabilitating abandoned pastures and degraded forests (prurmas) are instrumental in their return to original forest status. An understanding of these secondary succession processes contributes to the rehabilitation of degraded forest ecosystems and to the livelihoods of local communities, and aids in conserving biodiversity. We studied secondary succession in mixed species plantations that were established to rehabilitate abandoned land. The initial vegetation in these abandoned pastures and croplands was grassland composed of three dominant species: *Rottboellia exaltata, Imperata brasiliensis*, and *Brachyaria decumbens*. After tree planting and weeding had been carried out, the site was first invaded by *R. exaltata* and *Baccharis floribunda*. These two species, which depend solely on sexual and not vegetative reproduction, facilitated secondary succession and elevated species diversity by enabling subsequent invasion by several species. By contrast, *B. decumbens, I. brasiliensis*, and *Hyparrhenia rufa* depend mainly on vegetative reproduction involving rhizomes and tillers, and subsequent invasion by other species was relatively less in stand types dominated by these three species. We found that further adequate rehabilitation techniques were necessary for the respective vegetation types.

Key words: abandoned pasture, secondary succession, grasslands, rehabilitation, Peru

INTRODUCTION

Anthropogenic impacts and land-use conflicts between forests and agricultural land, caused by population increase, are accelerating the degradation of forest ecosystems. Every year, 16.1 million hectares (ha) of the world's primary forests are disappearing (FAO, 2001). Specifically, 15.2 million ha of this annual loss of tropical forests is resulted from 14.2 ha of land use changes and 1.0 million ha of deforestation. These degraded forests generally fail to regenerate naturally, because adequate rehabilitation techniques are not applied, especially in tropical forests (Nihon Ringyo Chousakai 1989, ITTO, 1991), necessitating silvicultural treatment for their recovery. This is an issue requiring attention, because rehabilitated forestland could potentially contribute to the sustainable use of forest resources, conservation of primary tropical forests, and ecosystem services improvement. The process of secondary succession provides important information for the rehabilitation of degraded tropical forests and land. In particular, the availability of information on natural secondary succession processes contributes to the development of rehabilitation techniques (Kobayashi, 2004). Among the groups of species that appear in secondary succession, pioneer species play key roles in the land dynamics and recovery process of degraded forests (Kobayashi et al., 1995). In this study, we seek to answer the following questions. (1) Do rehabilitated forests provide timber and forest ecosystem services? (2) Can we expect newly established rehabilitated forests to return to their original forest status? (3) Should an original forest stand be the final goal for rehabilitating degraded forest ecosystems? To answer these questions, we surveyed initial vegetation recovery and classified undergrowth vegetation types to predict changes in several types of abandoned pasture and secondary forest. We investigated the influences and roles of vegetation types on secondary succession affected by anthropogenic activities, i.e., facilitation or competition processes relating to environmental changes (Holmgren et al., 1997; Li and Wilson, 1998), their interactions (Callaway and Walker, 1997), the allocation pattern of photosynthesis products of each indicated species (Kawano 1974), and biodiversity of each forest (Kobayashi 1984a). The aims of this study were to: (1) investigate dynamic processes of vegetation recovery in the context of soils found in abandoned agricultural and pasture lands, (2) determine changes in biomass composition and species diversity, and (3) define vegetation types in terms of facilitators or competitors in processes of secondary secession. Instead of only focusing attention on the decrease in forests in Peru, it is necessary to attend to previously neglected degraded forests in each country located along a tributary of the Amazon River. In this study, we focused on the region around the Ucayali River, an Amazon tributary in Peru, and the area of Pucallpa which is emerging as a new front for natural forest logging.

SITE DESCRIPTION AND METHODS

Study site

The Ucayali region in the Peruvian Amazon has a population of 391,000 and occupies an area of 102,500 km^2. It covers seven different ecological zones and four transitional ecological zones (ONERN, 1976 &1981; Soudre et al., 2001), with tropical rain forest accounting for the largest area in this territory, followed by sub-montane rain forest. Seventy-five percent of the Ucayali forest is deforested and degraded, and requires rehabilitation. Our study sites were located at Campo Verde in Pucallpa, along the eastern tributaries of the Ucayali River, including the Aguaytia River. Deforestation in this region stands at 30,787 ha annually, and is the most intensive in Federico Basadre, Nueva Requena (along the Aguayia River), Tournavista (along the Pachitea River), and in areas near the Neshuya River (Fujisaka et al., 2000). The land was formed from residual material and old alluvial deposits during the Tertiary period of the Cenozoic era. Geomorphologically, it is defined as dissected alluvium (Rasanen, 1993). Topographically, little elevation exists. The relief is flat with slightly undulating slopes varying between 1 % and 8 %. Drainage is generally good, although some areas in the region near Nueva Requena experience poor drainage. The study area, Campo Verde, does not have a meteorological station. However, it is located midway between the Pucallpa (8° 23′ S, 74° 34′ W) and San Jorge (8° 30′ S, 74° 52′ W) meteorological stations. Therefore, climatic information for the study site was interpolated using data from these two stations. The average annual precipitation in the area is 1800 mm, with the majority of the precipitation occurring during the wet season between November and April, and less precipitation occurring during the dry season between May and October. The average annual temperature is 25.2℃, with an average maximum of 30.9℃ and an average minimum of 19.6℃. During the last decade, the average monthly temperature oscillated between 23℃ and 26℃. The average evapotranspiration potential is 1200 mm and the average relative humidity is 77 %.

Study methods

Three experimental sites, each being 40 m × 40 m × 3 replications in area, were established to rehabilitate degraded abandoned pastures and agriculture lands through mixed plantations of *Tabebuia serratifolia* (tahuari), *Calycophyllum spruceanum* (capirona), *Amburana cearensis* (ishipingo), *Terminalia oblonga* (yacushapana), *Cedrelinga catenaeformis* (tornillo), and *Schizolobium amazonicum* (pashaco). Three sites were scattered in the area of about 31 km^2 in Campo Verde. Each tree was planted in a 3 m × 3 m space within these plantations. Three 40 m × 40 m plots, each containing five quadrats (measuring 2 m × 2 m), were demarcated in each site in August 1997. The plots were originally dominated by *Imperata brasiliensis* (cashupsha), *Rottboellia exaltata* (arrocillo), and *Baccharis floribunda* (sachahuaca). Another two grass species, *Brachiaria decumbens* (pasture weed) and *Hyparrhenia rufa* (yarahua) were frequently observed at near the study sites. We had also selected three 800-m^2 area each representing the area where initially dominated by one of the grass species, *Imperata brasiliensis* (cashupsha), *Rottboellia exaltata* (arrocillo), or *Baccharis floribunda* (sachahuaca), for monitoring the distribution area change of these grass species and estimating secondary succession. The 800-m^2 was consisted of former three sites (2 m × 2 m × 5 replications × 3 replications = 60 m^2) and added one controlled plot (no treatment: 2 m × 2 m × 5 replications = 20 m^2). Species composition in the 800-m^2 monitoring areas and sampling points set in each area was recorded using Braun-Blanquet's (1964) method at the following times: in June 1998 (one month after tree planting) and in June 1999 (one year after planting). A species composition table was drawn up to examine the relationships between vegetation types and litter, grass biomass, soil condition, and landform.

Above ground biomass, accumulated litter, soil condition, and landform were annually surveyed for grass species occurring in the vicinity or inside of each quadrat, plot, and site. The above ground biomass of grass was surveyed in the vicinity of each quadrat (sample area = 1 m × 1 m) or in 10 temporary quadrats (1 m × 1 m) in the plot. The litter amount was surveyed at each quadrat (sample area = 0.5 m × 0.5 m). Ratios of dry weight (obtained through heating at 80℃ over 72 hours) to fresh weight were determined for each grass and litter type, and fresh weights were converted to dry weights. The soil profile at 30 cm depth was measured at a thickness of A_0 and A horizons, and a hardness of A and B horizons, using a Yamanaka soil penetrometer. Slope direction, slope incline, and relief index were measured using a clinometer. We commenced this study in August 1997, and repeated these observations in June 1998 and June 1999.

We evaluated species abundance in each 2 m × 2 m quadrat and in each observation sites using Braun-Blanquet's cover-abundance scale (Braun-Blanquet, 1964). The scale was converted to dominance values which are necessary for the calculation of diversity indices, as shown below.

Braun-Blanquet's cover-abundance scale: dominance value of *i* th species d_i

r: 0.01
+: 0.001
1: 0.05
2: 0.175
3: 0.375
4: 0.625
5: 0.875

We employed the following diversity indices:

Mean diversity, $H' = 3.3219 (\log_{10} D' - 1/D' \sum d_i \log_{10} d_i)$, (Lloyd and Ghelardi, 1964)

Total diversity, $H'D' = H' \times D'$ (Pielou, 1975)

Evenness, $J' = H' / \log_2 S$ (Pielou, 1975)

where S denotes the total number of species recorded, and D' denotes the sum of dominance values ($\sum d_i$).

We harvested the following five dominant grasses to assess their allocation patterns: *Imperata brasiliensis*, *Rottboellia exaltata*, *Baccharis floribunda*, *Brachyaria decumbens*, and *Hyparrheria rufa*. Of these five species, *R. exaltata* and *B. floribunda* were individually harvested, while each of *I. brasiliensis*, *B. decumbens*, and *H. rufa* were harvested in one 1 m × 1 m sampling site because these species developed rhizome and it is difficult to distinguish individual. We dissected each plant into its photosynthetic organ (leave/blade), intake organ (root), support organ (stem), sexual reproductive organ (flower and fruit), and vegetative reproductive organ (rhizome), and obtained their fresh weights. We then calculated the reproductive effort (sexual and vegetative reproductive organ/total individual weight) for each species using the data set.

RESULTS

Vegetation recovery and undergrowth vegetation types related to soil condition

The initial vegetation before tree planting was classified into three vegetation types dominated by *I. brasiliensis*, *R. exaltata*, and *B. floribunda*. One year after the trees were planted, the soil condition had changed in terms of the hardness of the A and B horizons (Table 1). The A and B horizons were compacted from 6.04 mm to 9.25 mm and from 14.5 mm to 16.3 mm, respectively.

The total number of species increased from 8.23 to 10.73, and the total dominance value (D') also increased from 85.5 to 128.4, resulting in increased mean diversity, total diversity, and evenness. In areas where the original

Table 1. Vegetation changes observed one month and twelve months after planting in Campo Verde, Pucallpa

	1 month after planting	12 months after planting
Slope degree	0.95	0.95
Thickness of A_0 horizon (cm)	3.38	2.82
Thickness of A horizon (cm)	4.77	6.36
Hardness of A horizon (mm)	6.04	9.25
Hardness of B horizon (mm)	14.5	16.27
Undergrowth biomass (dry t/ha)	4.63	8.19
Litter (dry t/ha)	8.36	5.59
Total species number (/4 m^2)	8.23	10.73
Total dominance value	85.52	128.35
Mean diversity	1.09	1.43
Total diversity	100.92	202.1
Evenness	0.36	0.43

Note: All data are averages and are presented without conducting statistic tests. April/98: clearing; May/98: planting; every two months: weeding at an average of 20 points.

dominant vegetation type was *Imperata*, we found the largest amount of undergrowth biomass (9397.9 kg/ha) the lowest total dominance value (115.66), and the lowest diversity indices in 1999 (Table 2). In areas where the initial dominant vegetation type was *Baccharis*, we found the smallest amount of undergrowth biomass (6139.8 kg/ha) and the highest litter, mean diversity, total diversity, and evenness.

Lastly, in areas where the initial dominant vegetation type was *Rottboellia*, we observed the highest total dominance value. These results indicate a relationship between mean diversity and soil hardness of the A horizon for these three grass species (Figure 1). In 1998, the *Baccharis* and *Rottboellia* vegetation types were found to be associated with soft soil with a hardness of$<$6 mm at the soil surface, whereas the *Imperata* type was found to be associated with hard compacted soil with a hardness$>$10−15 mm at the soil surface.

Energy allocation patterns and reproductive efforts

We sampled five dominant grass species in each area to compare their allocation patterns of photosynthetic products (Table 3). The estimated biomass including their roots for *I. brasiliensis*, *B. decumabens*, *H. rufa*, *R. exaltata*, and *B. floribunda*. The estimated biomasses, including roots, for these species were: 42.2, 39.2, 34.5, 6.9 and 24.8 t/ha, respectively. The highest proportionate allocation of photosynthetic products to the photosynthetic organ

Table 2. Vegetation changes by types one month (in 1998) and twelve months (in 1999) after planting

	1998				1999			
	Control	*Imperata*	*Rottboellia*	*Baccharis*	Control	*Imperata*	*Rottboellia*	*Baccharis*
Slope degree	1	2	0	1	0.75	2	0	1
Thickness of A_0 horizon (cm)	2.25	4.3	2.15	5.7	3.59	3.2	2.33	4.3
Thickness of A horizon (cm)	3.75	8.8	3.2	4	6.25	9.4	5.8	5.8
Hardness of A horizon (mm)	4.1	13.4	3.12	4.44	6.88	11.24	11.06	9.08
Hardness of B horizon (mm)	14.92	14.88	13.58	13.62	12.07	14.94	0	0
Undergrowth biomass (dry t/ha)	13.36	0.9	0.83	0.87	8.47	9.4	9.38	6.14
Litter (dry t/ha)	4.51	9.65	6.14	14.95	4.63	4.31	5.55	8.12
Total species number (/4m^2)	9.5	6.8	6.2	10.6	9.08	9.2	11.2	13.9
Total dominance value	139.77	70.04	63.8	53.66	145.7	115.66	128.96	116.61
Mean diversity	1.14	0.22	1.25	1.81	1.46	0.93	1.48	1.79
Total diversity	196.14	16.59	78.68	98.82	234.47	117.94	203.05	233.44
Evenness	0.34	0.09	0.48	0.54	0.47	0.29	0.43	0.48

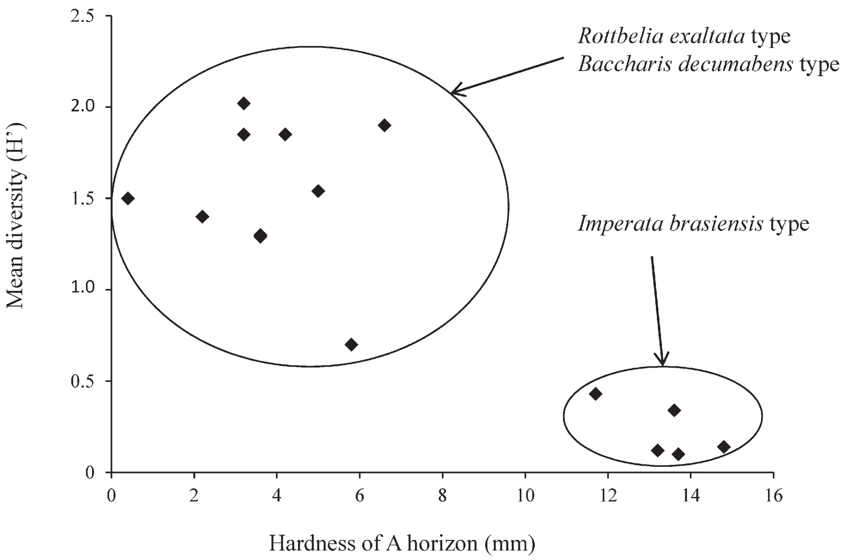

Fig. 1. Relationship between A horizon soil hardness and mean diversity of *R. exaltata, B. decumabens* and *I. brasiliensis* tuypes in 1998.

occurred in *I. brasiliensis*. In *H. rufa*, 50 % of these products were allocated to the intake organ, and in both *R. exaltata* and *B. floribunda*, high proportions were found in the support organs at 58.1 % and 49.8 %, respectively. At the time of sampling, *I. brasiliensis* had no flowering organs and *R. exaltata* and *B. floribunda* had no vegetative organs. The reproductive effort [(sexual organ weight + vegetative organ weight)/individual weight × 100 %] of each grass species differed significantly. *I. brasiliensis* showed the highest reproductive effort with 58.9 % associated with the vegetative organ, whereas only 4 % was associated with the sexual organ in *B. floribunda*.

Secondary succession processes of abandoned pastures

Changes in vegetation between 1997 and 1999 were

Table 3. Energy allocation patterns and reproductive efforts of five grass species

Species name (common name)	Biomass	Stem Number	Photosynthetic Organ	Intake Organ	Support Organ	Sexual Reproductive Organ	Vegetative Reproductive Organ	Reproductive Effort
	(fw t/ha)	(/m^2)	(%)	(%)	(%)	(%)	(%)	(%)
Imperata brasiliensis (Cashupsha)	42.2	375	25.7	10.5	5.5	0	58.7	58.7
Brachyaria decumabens (pasture weed)	39.2	n	17.8	38.6	31.7	0.001	11.9	11.9
Hyparrheria rufa (Yarahua)	34.5	n	14.5	53.4	14.2	3.1	14.8	17.9
Rottbellia exaltata (Arrocillo)	6.9	83	12.8	11.3	58.1	17.9	0	17.9
Baccharis floribunda (Sachahuaca)	24.8	1.4	20	26	49.8	4.2	0	4.2

Note: Photosynthetic organ = leaves/blades, intake organ = roots, support organ = stem, reproductive organ = seeds and rhizomes, n: no main stem.

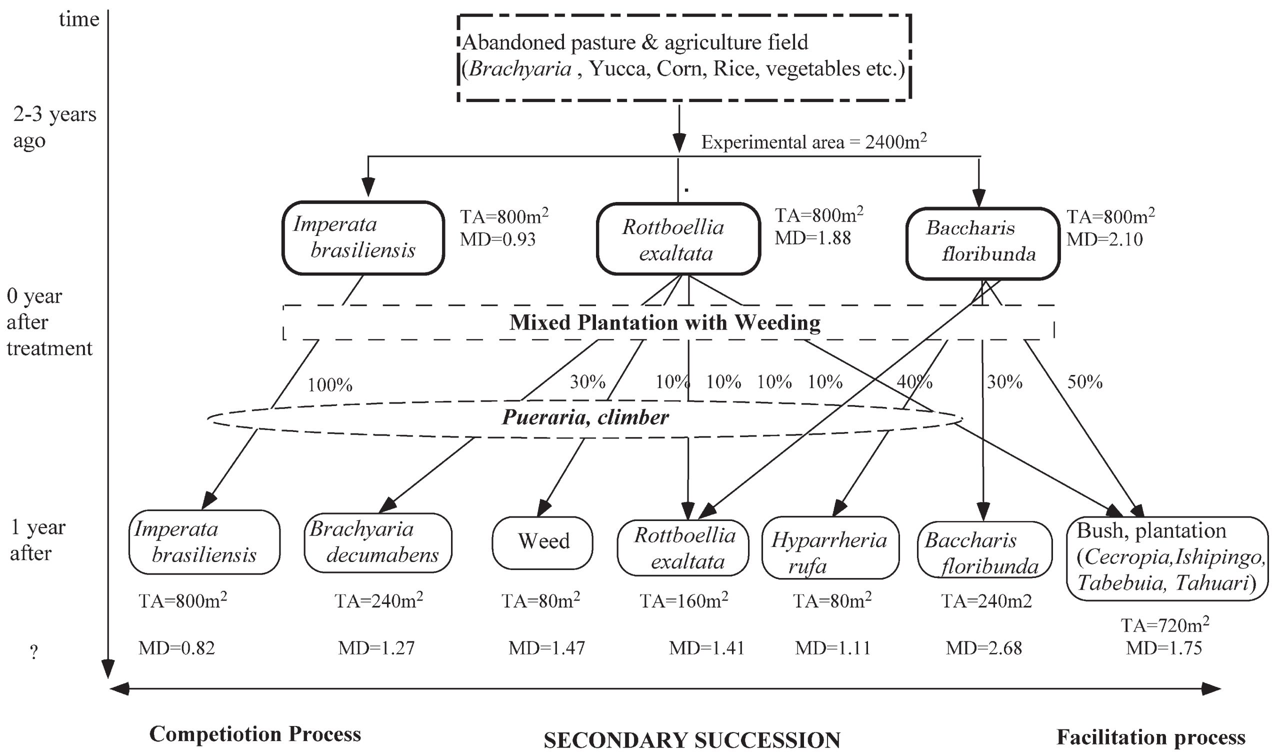

Fig. 2. Secondary succession process based on changes in dominance area and mean diversity.

recorded in three observation areas (Fig. 2). Portions of these areas occupied by each dominant species were separately estimated by observing changes in the number of sampling points dominated by the species and estimating mean diversity. At the initial stage of treatment, the *Imperata*, *Rottboellia*, and *Baccharis* vegetation types were dominant. A further five vegetation types appeared one year after planting, which included *Brachiaria*, *Hyparrhenia*, other weeds, and bush types (these dominated by the planted trees). *Imperata* occupied 100 % of its original area and its distribution range remained unchanged. The mean diversity of the *Imperata* stand decreased. The *Rottboellia* stand decreased from 800 m^2 to 160 m^2 with the remaining area of 640 m^2 showing an expansion into four vegetation types, including *Brachiaria*, weeds, and bush types. The area dominated by *Baccharis* remained at 1200 m^2 and expanded into four vegetation types, including other *Baccharis* types, *Hyparrhenia*, *Rottboellia*, bushes, and planted trees. This area originally contained 50 % of bushes and planted trees and showed the highest mean diversity one year after treatment. The *Rottboellia* and *Baccharis* types were seen to progress to bushes and planted trees (40 % and 50 %, respectively).

DISCUSSION

Vegetation recovery related to energy allocation patterns and reproductive efforts of five grass species in Campo Verde, Pucallpa

Significant differences were noted between the allocation patterns and reproductive efforts of the five grass species, especially in their supporting organs. Their reproductive efforts indicated that some of their strategic aspects were being used to facilitate succession (Kawano 1974, Tilman 1997). *I. brasiliensis* showed the highest reproductive effort at 59 % of the vegetative organ, while *B. floribunda* only showed a 4 % sexual organ allocation. The reproductive effort of each species contributes to maintaining and/or expanding its population (Kawano, 1984). *Baccharis* type decreased its area size and increased its mean diversity, whereas *Imperata* remained stable and had the lowest mean diversity. *Rottboellia* type showed a medium changing ratio of mean diversity and area size. These results are similar to those obtained for a tropical seasonal forest in Thailand where banana was a facilitator and bamboo a competitor (Kobayashi et al., 1995). Environmental factors significantly affect the plant strategies (Kobayashi, 1984b; Wilson and Keddy, 1988). Grime (2001) classified these factors into three types in relation to the evolution of plant strategies: the S-type (stress-tolerant: physical and nutrition environments), the R-type (ruderal: environmental disturbance), and the C-type (competitive: competition for resources among organisms). The idea is related to r/K selection (Pianka, 1970). *Baccharis* and *Rottboellia* types showed a preference for soft soil of $<$6 mm hardness at the soil surface that could be categorized as a middle C-type, according to Grime (2001). *Imperata* type, by contrast, was distributed in hard compacted areas of $>$10−15 mm soil hardness that could be classified as a S-type. Therefore, both environment conditions and competition between plants must be considered in relation to secondary succession. The reproductive effort of a species significantly affected the areas it occupied and its mean diversity, and this correlated with either a facilitation or competition process (Tilman et al., 1996; Callaway and Walker, 1997; Holmgren et al., 1997; Tilman, 1997; Li and Wilson, 1998). *I. brasiliensis* has a tillering structure associated with its predominantly vegetative reproduction method, with sexual reproduction occurring only after exposure to fire (FAO-RAP, 2003). *R. exaltata* and *B. floribunda* populations flower frequently at the initial stages, and these species were found to have invaded all populations except for *I. brasiliensis*. From our observations during the study, we concluded that *B. floribunda* was a typical succession facilitator for controlling the direction of secondary succession in abandoned pasture, as well as a catalyst for increasing biodiversity. By contrast, *I. brasiliensis* was found to be a secondary successional competitor.

Secondary succession processes based on changes in dominance and mean diversity

We examined the early phase of vegetation recovery in abandoned pastures one year after planting. We found that intensive forest clearing to create pasture affected vegetation recovery during the initial phase. As previous studies have also shown, we observed relationships between grass types and environmental gradients such as soil hardness (Ito, 1979; Wilson and Keddy, 1988; Currie, 1991). This is a similar situation to that seen in newly created primary non-vegetated sites where seed sources are the most important agents for vegetation recovery, and wind dispersal is an important factor (Nakashizuka et al., 1993). At the initial phase of treatment, *Imperata*, *Rottboellia*, and *Baccharis* vegetation types were observed. Subsequently, seven other vegetation types appeared such as *Brachiaria*, *Hyparrhenia*, weeds, bushes, and planted tree types (*Cecropia*, ishipingo, tabebuya, and tahuari). Over a period of one year of mixed plantation, *Pueraria phaseoloides* and other climbers, combined with soil condition, further influenced secondary succession. Imperata type was found to be associated with the competition process because it showed the smallest mean diversity and no change in area size (Kobayashi, 2004). Vegetation suppressed by this process would require treatment for successful rehabilitation (enrichment and planting). *Baccharis*, by contrast, was part of the facilitation process because of its higher mean diversity and recruitment of woody shrubs. In the facilitation process, vegetation is expected to follow its natural course of secondary succession but may sometimes require treatment to accelerate natural regeneration. Similar processes have been observed in several other forest types in different regions. These include: mixed Dipterocarp forests in Samarinda, East Kalimantan, Indonesia, and Pasoh in Malaysia; a mixed deciduous forest in Maeklong, Thailand; a peat swamp forest in Belait, Brunei, Darussalam; a subtropical moist forest in Misiones, Argentina; and a teak (*Tectona grandis*) plantation in Thom Pha Phun, Thailand (Kobayashi, 2004 & 2007). The facilitation process is necessary to accelerate natural regeneration and enrichment planting (Kobayashi, 2004). The competition process can be introduced in large-scale and catalytic plantations (Nambir and Brown, 1997). This facilitation-competition process supports the diversity–productivity hypothesis (Tilman et al., 1996), but does not support Tilman's statement that "species-rich sites are more resistant to invasion" (Tilman, 1997). Accelerated natural regeneration methods, such as "umbrella" natural regeneration, "side-effect" natural regeneration, and forest patch improvement, are closely related to assisted natural regeneration (FAO-RAP, 2003). Enrichment planting techniques, such as line planting and gap planting, have also been studied (Kobayashi, 2004). Our study suggests that large-scale planting using site matching, direct sowing, and mixed planting can act as a catalyst. We have addressed both the development of techniques as well as local community incentives for the rehabilitation of abandoned pastures. Forest recovery takes a long time, and fires and illegal logging are likely to occur during this period. Local community incentives should focus on ecological forestry resources, such as fallow products and non-wood forest products, on which forest dwellers depend. Local communities can then use these resources during the facilitation process in secondary succession.

ACKNOWLEDGEMENTS This study is part of a cooperative research program jointly organized and supported by INIA, the Center for International Forestry Research (CIFOR), E-3 research on rehabilitation in the landscape of degraded tropical forests (supported by the Global Environment Research Fund), and the Ministry of the Foreign Affairs, Japan.

REFERENCE

Braun-Blanquet J. 1964. *Pflanzensoziologie*. Springer Verlag, Wien, Germany.

Callaway RM, Walker LR. 1997. Competition and facilitation: a synthetic approach to interactions in plant communities. *Ecology* 78: 1958−1965.

Currie DJ. 1991. Energy and large-scale patterns of animal-and plant-species richness. *American Naturalist* 137: 27−49.

FAO. 2001. *State of The World's Forest. Rome*. (Japanese version, by Noubunkyo, 2002, pp.311)

FAO-RAP (Food and Agriculture Organization of the United Nations Regional Office for Asia and the Pacific). 2003. *Advancing assisted natural regeneration (ANR) in Asia and the Pacific*. RAP Publication 2003/19. 44 pp.

Fujisaka S, Escbar G, Veneklaas EJ. 2000. Weedy fields and forests: interactions between land use and the composition of plant communities in the Peruvian Amazon. *Agriculture, Ecosystems and Environment* 78: 175−186.

Grime JP. 2001. *Plant strategies, vegetation processes, and ecosystem properties*. Wiley. 417 pp.

Holmgren M, Scheffer M, Huston MA. 1997. The interplay of facilitation and competition in plant communities. *Ecology* 78: 1966−1975.

Ito S. 1979. Concepts of diversity in plant sociology. *Biological Science* 31: 200−206. (in Japanese)

[ITTO] International Tropical Timber Organization. Forestry Agency. 1991. *Beyond the guidelines – an action program for sustainable management of tropical forests*. ITTO Technical Series 7, Yokohama.

Kawano S. 1974. Speciation and adaptation. *Evolutionary biology in plants II*. Sanseido. 407 pp. (in Japanese)

Kawano S. 1984. Life history and evolution of forest floor vegetation. In: Kawano S (ed) *Plant history and evolution-2*. Baifukan, Tokyo, 1−19. (in Japanese)

Kobayashi S. 1984a. Biodiversity of forests. *Research of Species Biology* 8: 35−45. (in Japanese)

Kobayashi S. 1984b. Life history of *Disporum smillacinum* and its population dynamics. In: Kawano S (ed) *Plant history and evolution-2*. Baifukan, Tokyo, 87−112. (in Japanese)

Kobayashi S. 2004. Landscape rehabilitation of degraded tropical forest ecosystems: case study of the CIFOR/Japan project in Indonesia and Peru. *Forest Ecology and Management* 201: 13−22.

Kobayashi S. 2007. An overview of techniques for the rehabilitation of degraded tropical forests and biodiversity conservation. *Current Science* 93: 59−66

Kobayashi S, Yarwudhi C, Punyaratana P. 1995. Population dynamics of wild banana (Musa acuminata) related with the succession of the logged-over forest. *Proceedings of the International Workshop on the Changes of Tropical Forest Ecosystems by El Niño and Others*. JSTA, NRCT, & JISTEC, pp. 265−270.

Li X, Wilson SD. 1998. Facilitation among woody plants establishing in an old field. *Ecology* 79: 2964–2705.

Lloyd M, Ghelardi RJ. 1964. A table for calculating the "equitability" component of species diversity. *Journal of Animal Ecology* 33: 217−225.

Nakashizuka T, Iida S, Suzuki W, Tanimoto T. 1993. Seed dispersal and vegetation development on a debris avalanche on the Ontake volcano, central Japan. *Journal of Vegetation Science* 4: 537−542.

Nambir EKS, Brown AG. 1997. *Management of soil, nutrients and water in tropical plantation forests*. CSIRO. Canberra, Australia. 571 pp.

Nihon Ringyo Chousakai. 1989. *Midori no booklet No. 1*. 84 pp. (in Japanese)

[ONERN] Oficina Nacional de Evaluacion de Recursos Naturales (Office of National Assessment of Natural Resources; Peru). 1976. Ecological map of Peru (explanatory guide). Lima, Peru, 146 pp.

[ONERN] 1981. Peruvian major land use map, scale 1:1,000,000. Lima, Peru.

Pianka ER. 1970. On r- and K-selection. *American Naturalist* 104: 592−597.

Pielou EC. 1975. *Ecological diversity*. Wiley-Interscience.

Rasanen, M. 1993. The geohistory and geology of the Peruvian Amazon. pp. 46−67. In: Kalliola R, Puhakka (eds) *Peruvian Amazon. Tropical humid vegetation in the sub-Andean plain*. PAUT, and ONERN, Jyvaskyla, Finland. 265 pp.

Soudre M, Ricse A, Carbajal Y, Kobayashi S, Sabogal C, Alegre J. 2001. Adaptability of six native forest tree species to degraded lands in Pucallpa, Peruvian Amazon. In: Kobayashi S et al. (eds) *Rehabilitation of degraded tropical forest ecosystems*. CIFOR. pp. 123−128.

Tilman D. 1997. Community invasibility, recruitment limitation, and grassland biodiversity. *Ecology* 78: 81−92.

Tilman D, Wedin D, Knops J. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* 379: 718−720.

Wilson SD, Keddy PA. 1988. Species richness, survivorship, and biomass accumulation along an environmental gradient. *Oikos* 53: 375−380.

Received: 22 July, 2013
Accepted: 29 April, 2014