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INFLUENCE OF FERTILIZATION ON NATIVE PLANTS AND EXOTIC PASTURE GRASSES ON THE FASCINED LANDSLIDE SLOPES IN MIKURA-JIMA ISLAND, JAPAN

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ABSTRACT: Typhoon 9512 hit Mikura-jima Island, Izu Islands, Japan in 1995, producing many landslide slopes. Torino-o No.3 landslide slope was fascined with fallen trees and planted with seedlings of native species (*Alnus sieboldiana* tree and *Miscanthus condensatus* grass) as part of vegetation recovery efforts, and was subsequently seeded by helicopter in 2002 with exotic pasture grasses. To elucidate the influence of fertilization on the growth of transplanted seedlings of native species, exotic pasture grasses and newly invading trees and herbs, we designated seven plots in 2004, and surveyed vegetation every summer until 2013: four fertilized plots (fowl manure applied at 0.2 g/m^2) and three non-fertilized plots. The native species seedlings grew steadily irrespective of fertilization, but with self-thinning in *A. sieboldiana*. Exotic pasture grasses decreased gradually and almost disappeared in 2013. Fertilizing temporarily enhanced the growth of some species, but seemed to induce species competition, judging from the delay of invading of native trees and the increase of climbing plants in herbaceous layer in fertilized plots.

Keywords: Exotic pasture grasses, Fertilization, Landslide slope, Mikura-jima Island, Vegetation recovery

1. INTRODUCTION

Vegetation recovery in insolated ecosystems, such as on an island, should be carried out in ways that restore and do not disrupt the endemic ecosystem. In an insular ecosystem, habitat for wildlife can easily be lost due to disaster or development. Island ecosystems are generally rich in endemic species but tend to be homogeneous in habitat, making them vulnerable to invasive species [10], [13].

Mikura-jima Island (area, 20.58 km^2; location, ca. 180 km south of Tokyo) is an isolated island on the Izu Islands in the Pacific Ocean off the main island of Japan, where volcanic activity ceased ca. 7,000 years ago, leaving a long period for the development of the terrestrial ecosystem. It is covered with flora typical of that of the Izu Islands, has cliffs facing the sea with a large range in elevation (the highest peak is 850.8 m), and has many streams with abundant water [6], [12]. Mikura-jima Island was spared the deforestation for tourism or industry that took place on the other of the Izu Islands.

In September 1995, Typhoon 9512 hit Mikura-jima Island (Fig.1). It produced many landslide slopes among the old-growth forests of evergreen broad-leaved trees (e.g., *Castanopsis sieboldii* (Makino) Hatusima ex Yamazaki et Mashiba, *Machilus thunbergii* Sieb. et Zucc., *Eurya japonica* Thunb. and *Buxus microphylla* Sieb. et Zucc.) [1]. The thin layer of volcanic ash soil was washed away, and the bedrock was exposed on the landslide slopes. After the typhoon, vegetation recovery efforts were begun with projects to build fascines (using fallen trees) to retain the soil and to transplant locality-certified seedlings of native species (*Alnus sieboldiana* Matsum. and *Miscanthus condensatus* Hack.) [9].

We focus on Trino-o No.3 landslide slope (Fig.2) which is located on the northwest side of

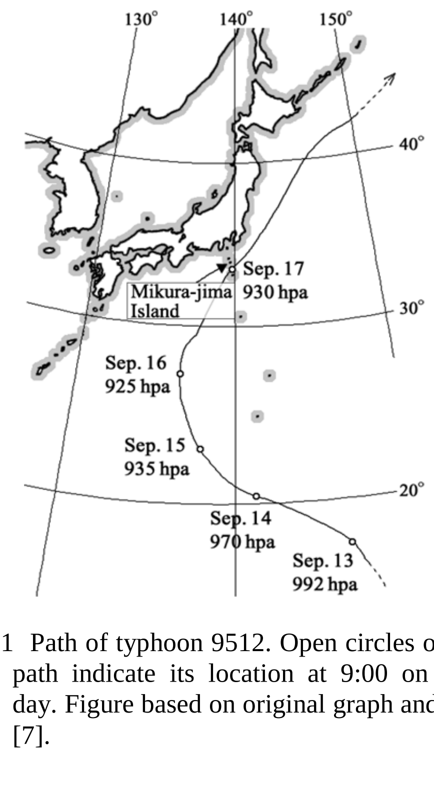

Fig.1 Path of typhoon 9512. Open circles on the path indicate its location at 9:00 on each day. Figure based on original graph and data [7].

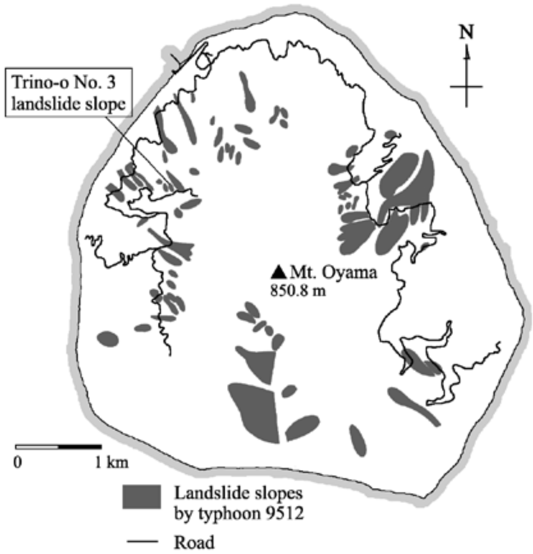

Fig. 2 Locations of landslide slopes formed by typhoon 9512 on Mikura-jima Island.

the island. On this slope, exotic pasture grasses were seeded by helicopter after the locality-certified seedlings had become established (Fig.3 and Fig.4); subsequently, the vegetation of the slope transitioned to a grassland community dominated by exotic pasture grasses in the next year [1], [9]. Then, in 2004, study plots were established, and one treatment group was fertilized once with fowl manure at 0.2 g/m^2 in order to support the growth of planted native seedlings which had inferior growth (Table 1).

In Japan, exotic pasture grasses have come to be used frequently for revegetation, because of their rapid and homogeneous germination and growth. Recently, ecological conservation strategies call for the use of native species, except in urban areas. Therefore, the unintended introduction of non-native pasture grasses to Trio-o No.3 landslide slope provides a valuable study site for examining the competition between native and non-native grass species. It is noteworthy in the sense that it is difficult to design an exotic grassland in an insular ecosystem, even in the context of academic research. If we monitor the succession and the growth of each plant species, we will be able to obtain a rare data set for examining the succession of vegetation after seeding exotic pasture grasses on landslide slopes on islands, as a part of vegetation recovery efforts.

The strategy of using native plant species remains unproven: since their germination and growth are inferior to exotic pasture grasses, they cannot rapidly achieve even ground surface cover. For future disaster prevention, it is desirable to transition from exotic pasture grasses to native species. However, exotic pasture grasses have been reported to prevent vegetation recovery: they prevent the establishment of trees [1], [4], change

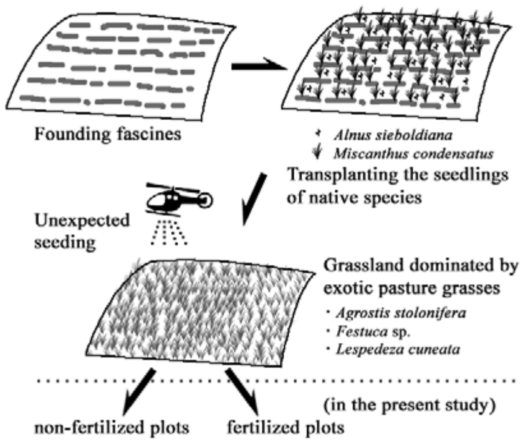

Fig.3 A schematic depiction of vegetation restoration on Torino-o No.3 landslide slope

the direction of early succession [15], and may dominate the vegetation for long periods, e.g., 20 years [5], [8], [11], [15].

In the present study, we monitored the vegetation recovery on Trino-o No.3 landslide slope of Mikura-jima Island for 10 years, from 2004 to 2013. To elucidate the influence of fertilization on the growth of locality-certified seedlings of native species, exotic pasture grasses and vegetation including newly invading trees and herbs, we investigated in both non-fertilized and fertilized plots.

Table 1 History of vegetation restoration activities on Torino-o No.3 landslide slope

Year	Work done
1995	Landslides and loss of vegetation caused by typhoon 9512 precipitation: 648.5 mm maximum wind speed: 67.8 m/s
1996	Start of transplanting trees (on the lower parts of the slope)
2000	Installation of founding fascines (1,100 m^2, 19 steps) Transplantation of 800 *A. sieboldiana* seedlings (average of one seedling per square meter)
2002	Transplantation of 1,750 *Miscanthus condensatus* seedlings (average of 1.5 seedlings per square meter) Unexpected seeding of exotic pasture grasses by helicopter in fall
2003	Shift in vegetation to a grassland dominated by exotic pasture grasses
2004	Fertilization by hand to boost growth of planted native seedlings (fowl manure applied at 0.2 g/m^2) Establishment of monitoring plots

a) Founded fascines in 2002 b) Vegetation in 2004

c) Vegetation in 2005 d) Vegetation in 2013

Fig.4 Vegetation recovery on Torino-o No.3 landslide slope. a) Founded fascines to retain the soil and transplanted seedlings of native species in 2002. b) Vegetation dominated by exotic pasture grasses in 2004. c) Vegetation in which the seedlings of native species were restored in 2005, 1-year after fertilizing. d) Vegetation covered by trees (exotic pasture grasses had almost disappeared) in 2013.

2. METHODS

2.1 Site and Monitoring Plots

Mikura-jima Island is located ca. 180 km south of Tokyo in the Izu Islands. Over the past 5 years, the average annual precipitation was 2,910 mm and the average temperature was 17.9 °C. Trino-o No.3 landslide slope is located at elevations ranging from 500 to 550 m above sea level with average slope of 27° and faces north. The bedrock around the site is andesite and basalt. Surrounding intact forest is dominated by evergreen broad-leaved trees, mainly *Castanopsis sieboldii*.

The vegetation recovery activities conducted at this site are listed in Table 1. The seeding of exotic pasture grasses occurred in 2002, and the amount of seeds and grass species is unknown; based on subsequent surveys, the species included bent grass (*Agrostis stolonifera* L.), a fescue (*Festuca* sp.) and a sericea lespedeza (*Lespedeza cuneata* (Dum. Cours.) G. Don).

In August 2004, seven plots of ca. 3 m × 2 m (5.5-7.4 m^2) were established. To elucidate the effect of fertilization, we set three non-fertilized plots (C1 to C3) and four fertilized plots (F1 to F4). In subsequent years, growth and vegetation species were investigated in summer (early August).

2.2 Growth of transplanted species and exotic pasture grasses

Among the tree layer, transplanted trees of *A. sieboldiana* and other species were measured for plant height (H) and diameter at the base (D_0), and the parameter D_0^2H was calculated as an estimate of aboveground biomass. Since a phenomenon like self-thinning was observed in *A. sieboldiana*, we estimated the regression expression as follows:

$$1/x = a + by \quad (1)$$

where x is the mean plant density of *A. sieboldiana*, y is the mean individual D_0^2H, and a and b are constants [16]. Data of other invading tree species were analyzed in the same way.

For the herbaceous layer, plant height was measured for *M. condensatus*, and the coverage percentage was determined for the exotic pasture grasses.

2.3 Vegetation survey

For the herbaceous layer, two layers were established based on height. The first herbaceous layer, taller than 1 m, was omitted from the survey because it comprised only *M. condensatus* growing along the fascines. For the second herbaceous layer, comprising exotic pasture grasses and invading native species, the coverage percentage of each species was determined.

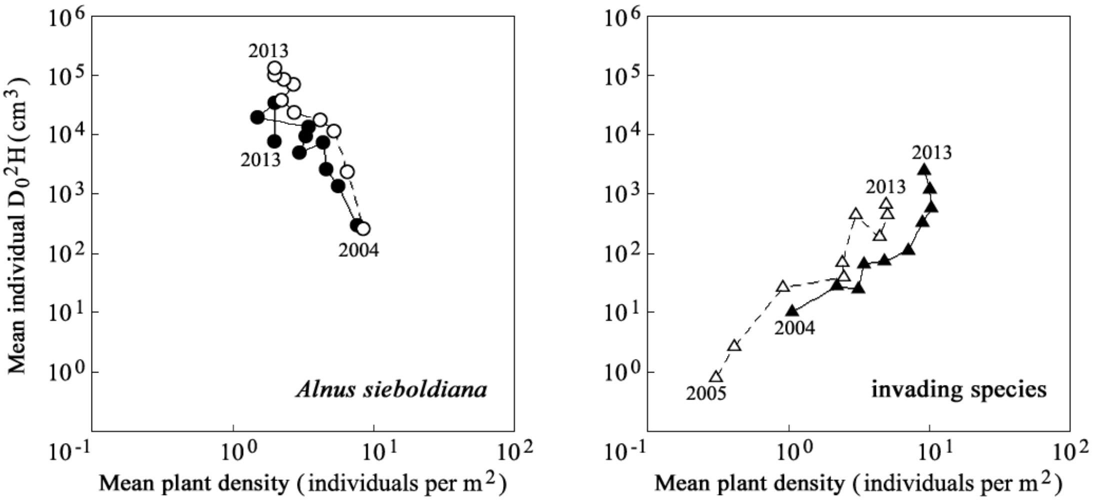

Fig.5 Changes in the density and the average D_0^2H per individual of trees in Torino-o No.3 landslide slope. The solid dots and solid line indicate non-fertilized plots, and the open dots and broken line indicate fertilized plots.

Since several growth forms were observed in the second herbaceous layer, we classified the growth forms into four categories: b (branching form), e (erect form), l (climbing form) and t (tussocks). Each species was classified into one of these growth forms, and the coverage percentage for each growth form was summed.

Coverage percentage data were arcsine transformed prior to statistical analysis since they are considered to have a binomial distribution. Statistical operations were conducted by calculations in a spreadsheet (Microsoft Office Excel 2003).

3. RESULTS

3.1 Growth of trees

Figure 5 shows the growth of *A. sieboldiana* and other invading trees over a period 10 years. In each of non-fertilized and fertilized plots, a phenomenon of self-thinning was confirmed in *A. sieboldiana*: the mean plant density decreased yearly, while individual D_0^2H increased and tended to reach a plateau. Although individual D_0^2H tended to be greater in fertilized plots, analysis of co-variance detected no significant differences in the regression coefficients (F-test, p = 0.108 and 0.266 for a and b, respectively) between the non-fertilized and fertilized plots.

In contrast, other invading trees increased in both density and individual D_0^2H, but the density appears to have reached a plateau in 2011 in each non-fertilized and fertilized plot. Invading trees were detected in 2004 in non-fertilized plots and in 2005 in fertilized plots. Both mean plant density and individual D_0^2H tended to be smaller in fertilized plots, and significant differences were detected in 2011 and 2012 for mean plant density and in 2013 for individual D_0^2H (Wilcoxon's rank sum test, $p < 0.05$).

3.2 Growth of *Miscanthus condensatus*, exotic pasture grasses and other invading herbs

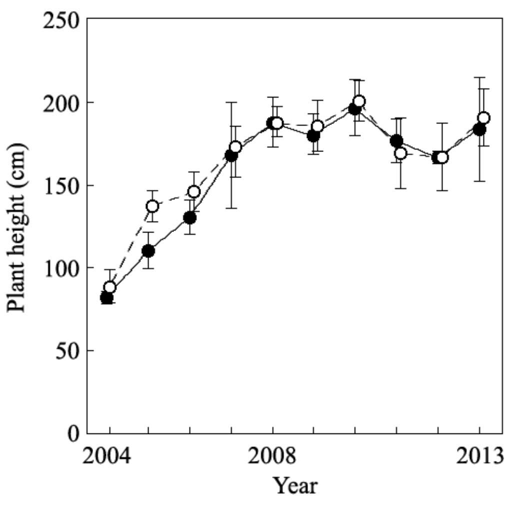

Fig.6 Changes in the height of *Miscanthus condensatus* on the Torino-o No.3 landslide slope. The solid circles and solid line indicate non-fertilized plots, and the open circles and broken line indicate fertilized plots. Error bars indicate ± 1 standard deviation (SD) (n=3 for non-fertilized plots; n=4 for fertilized plots).

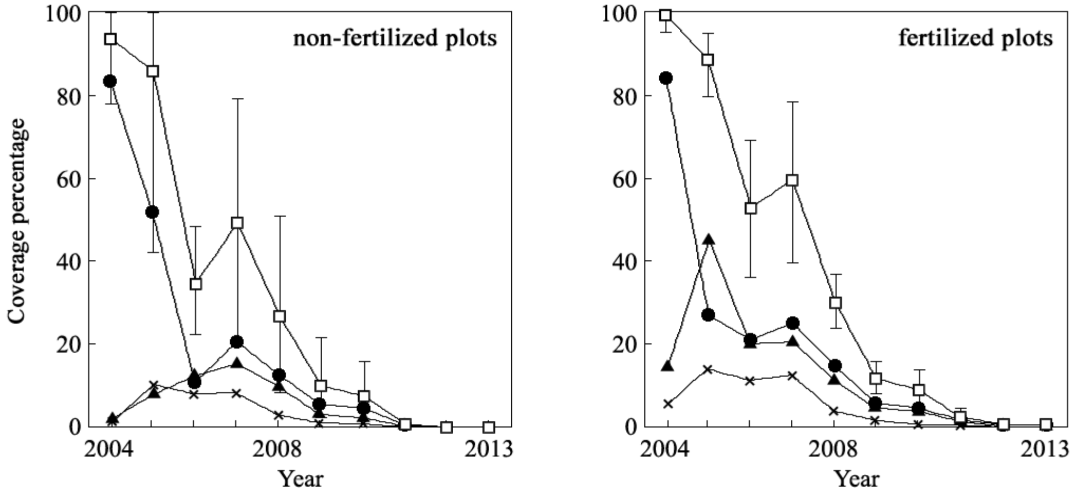

Fig.7 Changes in the coverage of exotic pasture grasses on Torino-o No.3 landslide slope. □, ●, ▲ and × indicate the total, *A. stolonifera*, *Festuca* sp. and *L. cuneata*, respectively. Error bars indicate ± 1 SD (n=3 for non-fertilized plots; n=4 for fertilized plots).

Figure 6 shows the change in mean plant height of *M. condensatus*. Only in 2005, the mean plant height was significantly greater in fertilized plots than in non-fertilized plots (t-test, $p < 0.05$). There were no significant differences in the following years, and plant height tended toward a plateau level in 2008 around which it fluctuated (from ca. 170 to 200 cm).

Coverage percentage of exotic pasture grasses is shown in Fig.7. After 2005, the total coverage percentage of exotic pasture grasses decreased with the exception of in 2007, and no significant differences were detected between non-fertilized and fertilized plots (Wilcoxon's rank sum test). By species, *A. stolonifera* decreased yearly, while *Festuca* sp. and *L. cuneata* leveled off but showed decreases after 2008. *Festuca* sp. increased rapidly in fertilized plots in 2005 (the year after the fertilization), but no changes were observed in non-fertilized plots. In 2013, exotic pasture grasses were absent from non-fertilized plots. The remnants were still extant in one of the fertilized plots, but the coverage percentage was less than 1%.

3.3 Growth form of invading herbs

In the monitoring plots of Trino-o No.3 landslide slope, volunteer seedlings of trees and composite species were representative of e (erect form), *Paederia foetida* L. was representative of l (climbing form), *Viola grypoceras* A. Gray var. *hichitoana* (Nakai) F. Maek. was representative of b (branching form), and ferns such as *Stegnogramma pozoi* (Lag.) K. Iwats. were representative of t (tussocks form).

Figure 8 indicates the changes in coverage of invading herbs with respect to growth forms. In non-fertilized plots, e dominated from 2006 to 2008, but gave way to t. Significant differences were detected in 2006 (e > b, l, t), in 2012 (t > b), and in 2013 (t >e, b, l) (Tukey's HSD, $p < 0.05$). In contrast, in fertilized plots, l dominated from 2005 to 2007, but gave way to t. Significant differences were detected in 2005 and 2006 (l > b, t), in 2009 and 2010 (e, l, t > b), in 2011 (t > e, b; l > b), and in 2012 and 2013 (t > l, e, b; l > b) (Tukey's HSD, $p < 0.05$).

4. DISCUSSION

4.1 Growth of *Alnus sieboldiana*, *Miscanthus condensatus* and exotic pasture grasses

In *A. sieboldiana*, the mean plant density decreased yearly, while the individual D_0^2H increased and tended to approach a plateau (Fig.5). Analysis of co-variance did not detect significant differences in the regression coefficients between the non-fertilized and fertilized plots. Previously, we reported that individual D_0^2H was greater in fertilized plots than in non-fertilized plots based on data collected from 2004 to 2009 [1], but those differences diminished by 2013. This extended data set indicates that a single fertilization increased individual D_0^2H of *A. sieboldiana* with effects apparent for several years but diminishing in 10 years. However, the decrease in mean plant density cannot be explained, because the plots were not covered completely by *A. sieboldiana*; that is, the self-thinning is not expected to be caused by competition between individuals. The Trino-o No.3 landslide slope might be a severe environment for *A. sieboldiana*, as we observed lodging or slanting of trees in 2007 [1].

Average plant height of *M. condensatus* was significantly higher in fertilized plots than in non-

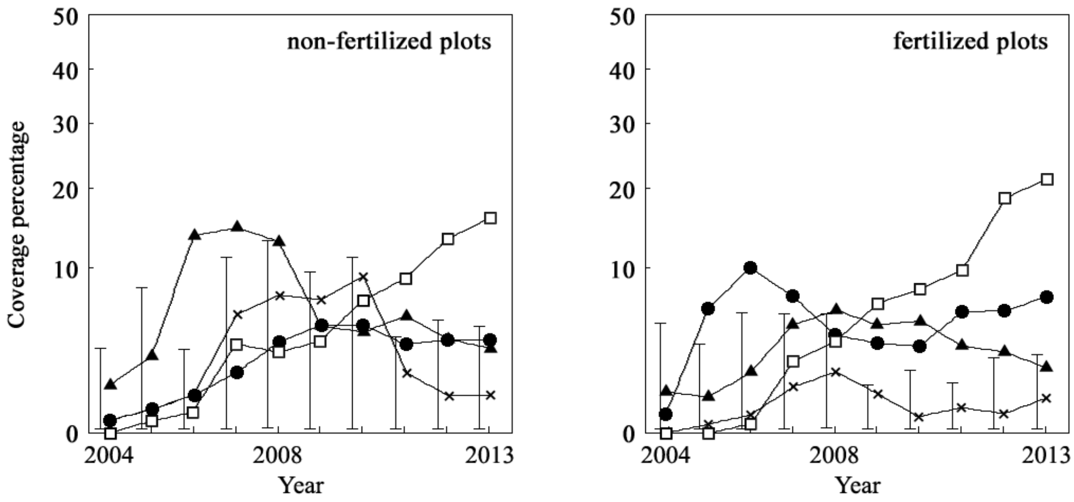

Fig.8 Changes in the coverage of invading herbs in the herbaceous layer on the Torino-o No.3 landslide slope with respect to growth form. ▲, ●, × and □ indicate erect form (e), climbing form (l), branched form (b) and tussock form (t), respectively. Each bar indicates the 95% confidence interval of difference between means based on Tukey's HSD.

fertilized plots only in 2005, the year immediately following fertilization. There were no significant differences in the subsequent years (Fig.6). Therefore, the effect of fertilization was temporary for *M. condensatus*, lasting only a year in the present study.

Exotic pasture grasses tended to decrease yearly, and *Festuca* sp. increased rapidly in fertilized plots only in 2005 (the year following fertilization). Generally, excessive fertilization causes gramineous species to put out many offshoots and to form a large tussock, and then to decrease in density [14]. It is suggested that *Festuca* sp. is more susceptible to such effects of fertilization than *A. stolonifera* because the latter grows as a carpet-like community by putting down filiform stolons.

The length of time needed to transition from a community of exotic pasture grasses to one of native vegetation is said to range from over 5 to 10 years [5], [8], [11], [15], and over 20 years would be needed for harsh habitats, such as when the substrate is bare bedrock or wind-beaten bare ground [5], [11]. Simultaneous transplanting of exotic pasture grasses and trees is reported to accelerate the changes to the surrounding vegetation [11]. As a nitrogen-fixing pioneer tree, *Alnus sieboldiana* is hypothesized to improve the soil and protect herbaceous layer species by shading, thus facilitating the growth of *M. condensatus* and other invading native species.

4.2 Change of Growth forms

Two phenomena are remarkable with regard to growth forms. First, different growth forms dominated the early period non-fertilized and fertilized plots; second, t (tussocks form) dominated after 10 years in all plots (Fig.8).

In fertilized plots, l (climbing form) dominated in the early period. In contrast, e (erect form) dominated in the early period in non-fertilized plots. Climbing plants have an advantage in the competition: they can grow rapidly because they do not need to invest in supporting organs and can bring down other plants by winding or leaning [2]. Fertilization is reported to induce species competition within the community in the early period because it improves the growth of some nutrient-limited species [3]. Since fertilization temporarily increases the growth of *M. condensatus* and *Festuca* sp., severe competition could result within the herbaceous layer. The high abundance of climbing plants suggests that plants of other growth forms could hardly germinate or establish seedlings under such severely competitive conditions.

Plants of tussocks form in the study sites were mainly ferns: it is rare and unprecedented that the grasslands dominated by exotic pasture grasses give way to ferns in 10 years. The spores of ferns might have been carried by rainfall or wind from the surrounding intact forest, but their growth and propagation are not competitive enough to generate these shifts in numbers. In Fig.8, only t (tussocks form) gradually and steadily increased during the 10 years of observation in both the non-fertilized and fertilized plots, and the decreasing exotic pasture grasses (*Festuca* sp. and *A. stolonifera*) are also of type t (tussocks form): this commonality suggests that ferns gradually replaced exotic pasture grasses by occupying the same niche in the herbaceous layer.

5. CONCLUSION

From the results obtained, we can conclude that the community of exotic pasture grasses in this study sites was almost succeeded by native vegetation in 10 years, irrespective of fertilization. Effect of fertilizing was observed temporarily in some fertilizer-sensitive species (*A. sieboldiana*, *M. condensatus* and *Festuca* sp.), but enhanced species competition, judging from the delay of invading of native trees from surrounding forest and the increase of climbing plants. Therefore, it is not considered that fertilization should be recommended in vegetation recovery process unless there is certain urgent necessity to support the transplanted seedlings with inferior growth.

In the present study, we focused on the vegetation recovery on a unique site in an island, the fascined landslide slope where exotic pasture grasses were seeded after the locality-certified seedlings of native species had become established. The influence of seeding exotic pasture grasses on vegetation recovery itself is yet not revealed, which will be of practical value in insular ecosystems: it remains to make comparison with the vegetation recovery on the fascined landslide slopes free from seeding exotic pasture grasses.

6. ACKNOWLEDGEMENTS

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7. REFERENCES

[1] Arase T, Okano T, Kimura T and Inoue S, "Influence of seeding exotic pasture grasses on the vegetation recovery of landslide slopes formed by typhoon 9512 in Mikura-jima Island, Izu Islands, Japan", J. of the Japanese Society of Revegetation Technology, Vol. 35, Feb. 2010, pp. 448-461

[2] Bailey DR, "Weed control in tropical pastures", Tropical forege legumes, Skerman PJ, Cameron DG and Riveros F Eds. Rome: FAO, 1988, pp. 136-147.

[3] Bazzaz FA, Plants in changing environment. Cambridge: Cambridge University Press, 1996, 320 pp.

[4] Holl KD, Loik ME, Lin EH and Samuels IA, "Tropical montane forest restoration in Costa Rica: overcoming barriers to dispersal and establishment". Restoration Ecology, Vol. 8, 2000, pp. 339-349.

[5] Imamoto H, Goto K, Shirai A and Washitani I, "Effect of the introduction of exotic grasses to solid rock slopes on the vegetation succession of the site". Ecology and Civil Engineering, Vol. 6, Aug. 2003, pp. 1-14.

[6] Issiki N, Geology of the Mikurajima, Inambajima and Znisu districts. Yatabe: Geological Survey of Japan, 1980, 35 pp.

[7] Japan Weather Association Ed., the yearbook of weather in 1996. Tokyo: the Printing Bureau, 1996, 265 pp.

[8] Kodama S, Wada M, Shima K, Chiba K, "Effects of different introduced plant species on invasion and establishment of other plant species in the artificial slopes". J. of the Japanese Society of Revegetation Technology, Vol. 28, Aug. 2002, pp. 139-142.

[9] Kyoju no Kai Ed., Saving earth (1) Practices to restore nature in Mikura-jima Island. Tokyo: Kyoju no Kai, 2004, 48 pp.

[10] Loope LL, Hamann O and Stone CP, "Comparative conservation biology of oceanic archipelagoes –Hawaii and the Galapagos". BioScience, Vol. 38, 1988, pp. 272-282.

[11] Nakano Y and Futami T, "Observation on plant succession at the past cut slope landscaping sites in Miyakejima Island". J. of the Japanese Society of Revegetation Technology, Vol. 30, Nov. 1987, pp. 383-388.

[12] Ohba T, "Die Vegetation von Mikura-Insel leight 200km sudlich von Tokyo". Bull. of Kanagawa Prefectural Museum, Vol. 1, Feb. 1971, pp. 1-53.

[13] Ono M, Wildlife in solitary island. Tokyo: Iwanami Shoten, 1994, 239 pp.

[14] Sato K, Nishimura N and Takahashi M, Eco-physiological studies on the maintenance of density of forage crop swards, I". Grassland Science, Vol. 11, Jun. 1965, pp. 14-19.

[15] Swada Y, Kubota K, Yashiro Y, Nishiwaki A and Tsuda S, "Vegetation established by the aerial seeding of pasture plants after a forest fire in a secondary forest in Shiga Prefecture, Japan". Japanese J. of Conservation Ecology, Vol. 13, May 2008, pp. 29-36.

[16] Yoda K, Forest ecology. Tokyo: Tsukiji Shokan, 1971, 331 pp.

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