【Article】

Plant Species Response against Mowing in Southwestern Japan

Justin ONDOPA[1*], Yoshitaka TAKAHASHI[2], Akira KIKUCHI[3] and Nobukazu NAKAGOSHI[1,3]

[1]*Graduate School for International Development and Cooperation, Hiroshima University,*
1-5-1 Kagamiyama, Higashi-Hiroshima, 739-8529, Japan

[2]*National Agriculture Research Center for Western Region, Kawai-cho, Oda, 694-0013, Japan*

[3]*Faculty of Integrated Arts and Sciences, Hiroshima University,*
1-7-1 Kagamiyama, Higashi-Hiroshima, 739-8521, Japan

Correspondent author: Nobukazu NAKAGOSHI
e-mail address: nobu@hiroshima-u.ac.jp

Abstract

This paper tried to study the patterns of plant species diversity in a herbaceous vegetation under mowing management in southwestern Japan. The investigation was conducted to examine species response from community-level mowing to assess species diversity and richness in two large quadrats. Within the two quadrats, six 4 m^2 plots each were diligently investigated and 110 vascular plant species were recorded. Analysis on diversity revealed that total number of species increased from 2001. There was an increase in rare and newly appeared species recorded after mowing (ANOVA $p<0.01$). After 2001, species diversity increased in both Q1 and Q2 of each year but a high diversity was seen in Q2 of 2003 and 2004 ($p<0.05$). It was also seen that the increase in species diversity and richness was determined by to biomass composition and invasion of new species after mowing. A high biomass level saw a decrease in species richness. Using DCA ordination, each plot was classified to determine the mean species richness which showed an increase from 2003 to 2004. It is from this finding that mowing is seen to sustain species diversity and richness and should be continued in order to conserve the unique biodiversity of Mt. Sanbe. These results suggest that management with mowing is an effective method although it reduced above-ground biomass, and periodic mowing proved to be successful in biodiversity conservation.

Key Words: biomass, mowing management, species diversity, richness, rare species, newly appeared species

1. Introduction

In the recent times, there has been increasing interest in the dynamics of semi-natural grasslands either for agricultural interests or nature conservation. Among others, several studies done focused on different aspects ranging from flora (Cowie et al. 1992) and fauna (Gerstmeier and Lang, 1996; Dithlogo et al. 1992) to management (Walker and Warner, 2000; Kollmann and Bassin, 2001; Barbaro et al. 2004), while others on succession (Numata, 1961; Itow, 1972) and species restoration (Pykälä, 2003). Studies on species diversity and richness at different biomass levels are rare in areas managed with mowing. However, Grime (1973, 1979) initially did a widely accepted empirical work on species richness-standing crop (biomass) but factors of disturbance that may associate such as mowing were not explicitly stated. Even few studies that followed instead focused generally on plant ecology (Willems, 1980) while others emphasized on species richness in wetlands (Wheeler and Giller, 1982; Vermeer and Berendse, 1983; Moore and Keddy, 1989). Scientifically, to evaluate the vegetative species, it is of course erroneous to apply findings of one vegetation type to assume species richness and diversity pattern of another. This is because in a certain vegetation type there are many species of different origin with different growth traits and response to disturbance. Thus it is quite difficult to assess species richness at species or community levels without separate devoted studies.

In Europe, for example, many studies have been done on species diversity (Chippindale and Milton, 1934; Champness and Morris, 1948; Forbes et al. 1980; Leps et al. 1982; Grime et al., 1988) including plant species restoration and communities (Bakker, 1989; WallisDe Vries et al. 1998; Bakker and Berendse; 1999). In contrast, only few studies have been done since the emergence of grasslands in Japan. Furthermore, grasslands dramatically declined after the 1960s and studies devoted to species diversity and response to management are rare and there is insufficient knowledge. However, a small number of studies committed instead focused on secondary succession (Numata, 1961; Ohga & Numata 1965; Itow, 1972), landscape patterns (Kamada and Nakagoshi, 1996), patterns of species diversity (Kitazawa and Ohsawa; 2002) and trampling gradients (Ikeda, 2003) while those specifically done on species richness and diversity under mowing are rare. From a community level standpoint, one way to evaluate and ascertain whether a certain management regime has been successful is to examine the abundance of species of different origin. So far, there is no steady information on mowing as an effective management practice by examining species diversity. In addition, several studies committed to examine species diversity came up with inconsistent results.

For example, it was reported that mowing increases species richness and diversity (Abul-Faith and Bazzaz, 1979; Hayashi, 1994; Kobayashi et al. 1997); but other studies found that it decreases species diversity (Hils and Vankat, 1982). Mowing is an important grassland management technique and these inconsistencies reflect that our understanding on species richness and diversity under mowing conditions are inadequate. It implies that more studies need to be devoted to grassland dynamics and the mechanisms underlying diversity. These discrepancies, apart from the frequency and intensity of disturbance, are sometimes due to biomass of the component species. Even at moderate intensities it is reasonably difficult to determine species richness without considering the factors of individual species biomass. In this context, this study was done to examine species richness and diversity from annual mowing and biomass composition. A four year vegetation database of after mowing was thoroughly analyzed to assess the species composition. In Japan since the 1960s urbanization, most of the grassland areas decreased due to abandonment which resulted in landscape homogeneity followed by loss of biodiversity (Kamada et al. 1991, Nakagoshi and Ohta, 1992). According to the Japan Red Data Book, there was a consequent increase in the number of threatened species in southwestern Japan (e.g. *Pulsatila cernua* and *Patrinia scabiosaefolia).* Similarly, other studies found that secondary succession by certain successive species could have detrimental effects on less competitive species (Hayashi, 1977; Zhou, et al., 2002). Arguably, these effects can result in gradual loss of other biological species. Compared with studies done on succession and threatened species, no specific research has been devoted to species richness and diversity in this area. Therefore, from the conservation view, it is equally important to carry out studies from all levels and perspectives if conservation must be seriously addressed.

It is from the conservation standpoint that to sustain and promote biodiversity in semi-natural grassland vegetations more studies should be done. The aim of this study was (1) to examine the species richness pattern of herbaceous species under mowing in semi-natural grasslands and (2) to suggest appropriate measures for plant species conservation.

2. Methods

2.1 Study site

Field survey and investigation was conducted in Mt. Sanbe (1, 126 m asl), Shimane prefecture, southwestern Japan **(Fig.1)**. There are about 680 ha of semi-natural grassland, 33% of it is utilized for pasture and meadow. At present, the semi-natural grassland is distributed along the foot belt of the mountain namely the east and west grassland (see Fig.1). Vegetation survey and sampling were conducted in a 40.1 ha portion of the east grassland (37º07’55”N, 132º38’7”E). The east grassland is mainly maintained with mowing and grazing while the west is managed with periodic burning and grazing. In the east, mowing is done annually in November to avoid severe cold-related fatality on plants and to sustain biodiversity.

General vegetation composition includes *Miscanthus sinensis* (meadows) and *Zoysia japonica* (pasture) types, with patchy communities of *Rosa wichuraiana* and *Pleioblastus distichus* var. *nezasa* species. Comparing species, the area appeared to be relatively heterogeneous with common and rare species nurturing from mild leaching organic clay loam soil types with an estimated 100% vegetation coverage. The area has a mean annual rainfall of 2,000 mm while temperature variation is within 12-13

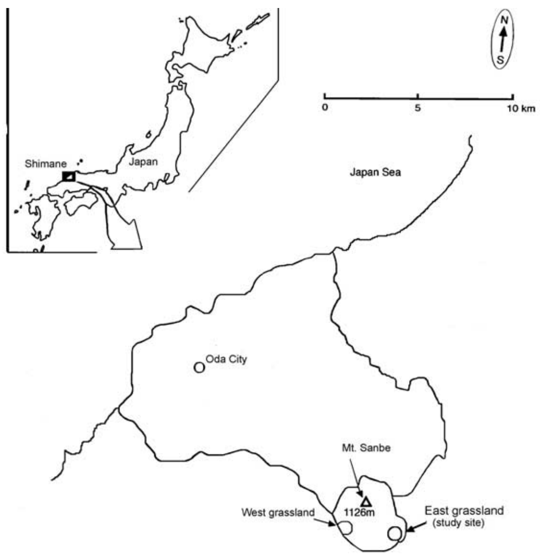

Fig. 1. Map of study site showing the east and west grasslands. The east grassland is managed with mowing and grazing while the west one is managed with grazing and periodic burning.

ºC. During winter snow fall reaches over 50 cm. The grassland in this area was once described as 'Orthoseral' and 'Plagioseral' type featuring *M. sinensis, Imperata cylindrical* var. *Koenigii, Pleioblastus distichus* var. *nezasa*, *Pleioblastus distichus* var. *nezasa, and Z. japonica* species Numata (1974).

2.2 Management history

Characterized by a pastural landscape, Mt. Sanbe has a management history of 400 years. Historically volcanic activities and changes in climatic patterns had created suitable environmental habitats quickly invaded by grass and other herbaceous species. Mt. Sanbe was vastly covered with *Sasa* type grass from its peak down to the foot, part of which was grazed by the Japanese Black cows from 1603 to 1867. Until the 1950s, the rise in mechanized farming decreased grazing cattles to in-house concentration which consequently affected grassland diversity. Since then, most verges and steep slopes up to the top of the mountain were reclaimed by secondary pine and oak forests cornering grasslands to the bottom. After a quarter centaury obsolescence cattle grazing was revived in 1989 at the west grassland and expended to the east which is still continuing at this time. This has restored some ecological hope in grassland management and conservation of biodiversity including pastural landscapes.

2.3 Field survey and data collection

Initial field survey was done in the east grassland in October, 2001. Continuous vegetation survey was conducted in June and October of 2002, 2003 and 2004. These two months were chosen because (the month of) June is the peak growth period (of plants) while October is the time when most plants have stabilized in phytosociology. With mowing done in November each year, this study has a balanced vegetation database of early growth and stabilized periods. To facilitate the vegetation investigations two big quadrats (Q), a 20 m×20 m (Q1) and a 25 m×15 m (Q2), were plotted at different locations along the slope. Although Q1 and Q2 were both treated with mowing, the database of 2001 is the before mowing one. The following 3 years investigation was done after annual mowing to make a parallel and simultaneous analysis on species diversity in the two quadrats. The differences in the quadrats sizes are due to the distribution range of target habitats which were decided to be investigated. Within the two quadrats, six 4 m^2 plots each were carefully investigated to account for low growing tiny species. All together, a total of twelve 4 m^2 plots were investigated. Using DCA ordination each plot was classified to determine the average number of species based on physiognomy and phytosociology of each species. The canopy coverage (C: %) and plant height (H: *cm*) for each species identified in each plot were measured with scale rulers, tape measure and graduated poles.

2.4 Data analysis

After diligently identifying species of tiny juvenile tree shrubs and herbs in each plots, a total of 110 species of 53 different families recorded were used in the analysis using multivariate ordination and classification methods (PC-ORD Version 4). To calculate the biomass of each community, samples were clipped to ground level, dried to the constant biomass at 70℃ for more than 3 days in an air forced draught oven and weighed. Species richness and the mean number of species in each quadrat were analyzed using SPSS statistical package 11.0. Species diversity (*H*') in each community was calculated using the Shannon-Wiener Index.

$$H' = -\sum_{i=1}^{n} (\mathrm{P}i \log \mathrm{P}i)$$

where Pi is the proportion of i^{th} species relative to the total n number of species in a quadrat, expressed in logarithm of this proportion (logPi).

The relative importance values or the Summed Dominance Ratio: SDR, (after Numata, 1974) were obtained from the phytosociological characters as follows:

$SDR_2 = (C+H)/2$,

$C = (c/c_{max}) \times 100$

$H = (h/h_{max}) \times 100$,

Where C and H are coverage and height, respectively. $_{max}$ is the maximum coverage and height of the tallest species per plot. To compare species diversity indices in each plot, one-way species analysis (ANOVA) was performed.

3. Results

3.1 Species diversity per plot per year

Within the study period, a total of 110 species of 53 families were recorded. The family class with the highest number of species was Compositae (13 spp) followed by Leguminosae (11 spp) and Gramineae (10 spp). Family Rosaceae had 9 species while Liliaceae had six. Family Caprifoliaceae and Hydrangeaceae had 4 species each while 3 species each were recorded from Labiatae and Lauraceae families **(Table 1)**. The remaining 44 families had less than 2 species recorded. During the study period a relatively similar pattern was seen in the total number of species found. In 2001, about 58 species (Q1, 36 spp; Q2, 22 spp) were recorded, 56 spp in 2002 (Q1, 26 spp; Q2 30 spp), 60 spp in 2003 (Q1, 32 spp; Q2, 28 spp) and 56 spp in 2004 (Q1, 29 spp; Q2, 27 spp).

Basically, there was no wide variation in species distribution. Based on this fluctuating pattern, we considered that not all of the species were common and categorized them into three categories. From the total number of species (110 spp) initially documented, species that were present throughout the investigation period (3 years) were classified as "common" species, while those scarcely distributed with reduced population size as "rare" and newly appeared fresh records as "new" species. The categorizing of species was done in chronological order of investigation for each year based on the initial before-mowing 2001 database (2 quadrats). It was seen that there was an increase in the number of rare and new species while the common species distributed were basically fluctuating throughout in each plot **(Fig. 2)**. It shows that, after mowing in 2002, a slightly increased number of common species were seen in Q2 compared to 2001 investigation, similarly species recorded in Q1 also increased. With regard to species diversity concerning rare and new species, an increase in the number of both categories was seen (ANOVA p<0.01). A high number of new species (7 spp) was recorded in 2002 in both quadrats, compared to 2003. In the case of rare species, a steady increase was seen. There were 12 species in 2003, followed by 11 in 2004 and 10 species in 2002. However, both rare and new species recorded in 2004 of both Q1 and Q2 were quite less compared to 2002 and 2003 figures due to high coverage by common species in that year. Relatively, compared to 2001 figures, which is the initial (June) database before mowing (in November), the number of common species in both Q1 and Q2 gradually increased especially in 2002 and 2003 **(Fig. 2)**. These results suggest that there was an increase in the number of dif-

Table 1. Selected families with the highest number of species found, families with less than two species recorded are not shown.

Family	Species	SDR2*	Family	Species	SDR2*
Compositae	*Artemisia japonica*	37	Rosaceae	*Agrimonia japonica*	7
	Artemisia princeps	5		*Potentilla fragarioides* var. *major*	14
	Artemisia stolonifera	30		*Potentilla freyniana*	11
	Aster scaber	32		*Prunus jamasakura*	10
	Atractylodes japonica	32		*Prunus* sp	7
	Chrysanthemum makinoi	15		*Rosa wichuraiana*	5
	Cirsium japonicum	30		*Rubus palmatus*	7
	Eupatorium lindleyanum	21		*Rubus parvifolius*	21
	Gnaphalium japonicum	12		*Sorbus japonica*	25
	Ixeris dentata	30	Liliaceae	*Allium thunbergii*	21
	Leibnitzia anandria	10		*Disporum smilacinum*	14
	Pertya scandens	7		*Hosta montana*	12
	Solidago virga-aurea var. *asiatica*	20		*Polygonatum falcatum*	7
Leguminosae	*Albizia julibrissin*	10		*Polygonatum odoratum* var. *pluriflorum*	20
	Castanea crenata	30		*Smilax china*	70
	Indigofera pseudo-tinctoria	5	Caprifoliaceae	*Lonicera japonica*	10
	Lespedeza bicolor f. *acutifolia*	60		*Viburnum dilatatum*	5
	Lespedeza cuneata	15		*Viburnum erosum*	5
	Lespedeza pilosa	7		*Weigela hortensis*	18
	Medicago hispida	10	Hydrangeaceae	*Cardiandra alternifolia*	10
	Pueraria lobata	5		*Deutzia crenata*	5
	Quercus mongolica var. *grosseserrata*	7		*Hydrangea paniculata*	5
	Quercus serrata	5		*Schizophragma hydrangeoides*	7
	Trifolium repens	12	Labiatae	*Perilla frutescens* var. *japonica* f. *viridis*	12
Gramineae	*Agrostis clavata*	30		*Plectranthus inflexus*	5
	Arundinella hirta	80		*Prunella vulgaris* var. *lilacina*	10
	Festuca parvigluma	12	Lauraceae	*Cinnamomum japonicum*	10
	Imperata cylindrica var. *koenigii*	34		*Lindera obtusiloba*	5
	Microstegium japonicum	7		*Lindera umbellata*	7
	Miscanthus oligostachyus	50			
	Miscanthus sinensis	100			
	Paspalum thunbergii	13			
	Themeda japonica	40			
	Zoysia japonica	20			

*calculated from phytosociology of each species by plot parameters.

ferent species distributed partly due to mowing that destabilized several common species from dominancy. The rare species were found in habitats where the common ones suffered much detrimental effects from mowing. Similarly, new species were recorded in bare grounds or new microsites created from mowing which swept away litter accumulation and other organic debris in the process.

Although new species colonizing strength appeared to be relatively low with the numbers recorded, there was a positive indication that mowing enables species diversity. This finding of increase in the number of species meant that there was a positive effect on species diversity as indicated by the diversity indices (**Fig. 3**). The figure shows that, although seemingly constant, Shannon diversity indices (*H*') increased in each plot per year. Most of the plots in Q2 of each year had higher species diversity compared to those of Q1. One of the factors seen in Q1 was the robust growth ability of common species that diversified quickly after mowing which affected the chances of new species from colonizing compared to Q2. In addition, species diversity indices of 2001 were quite constant and there was an increase

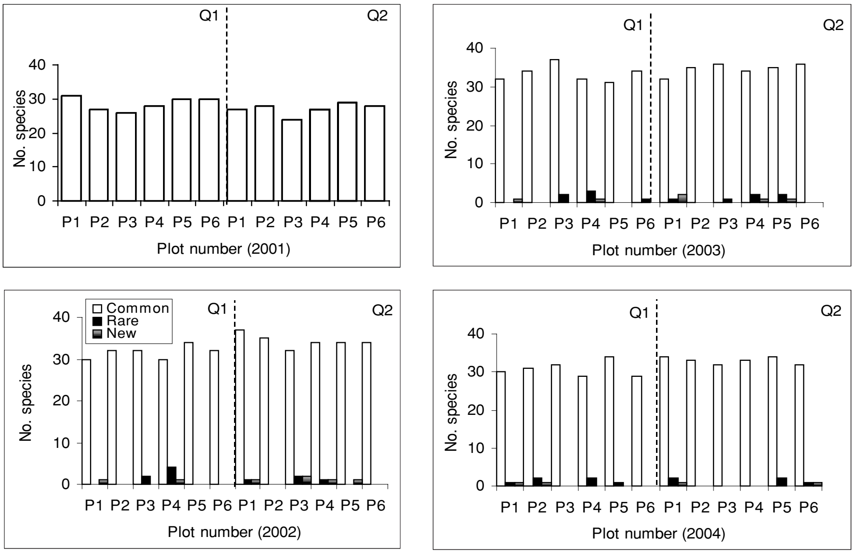

Fig. 2. Fluctuating pattern of common species and the distribution of new and rare species. Species categorizing was done from 2001 database so there was no new and rare species categories in 2001. Q1 is quadrat 1 (20 m×20 m) and Q2 is quadrat 2 (25 m×15 m) while P1, P2 etc. in each captions represent plot numbers of each quadrat. (Example; P1 under Q1 of 2001 = plot 1 of Q1). 2002, 2003 and 2004 are after-mowing database and only 2001 is before-mowing data.

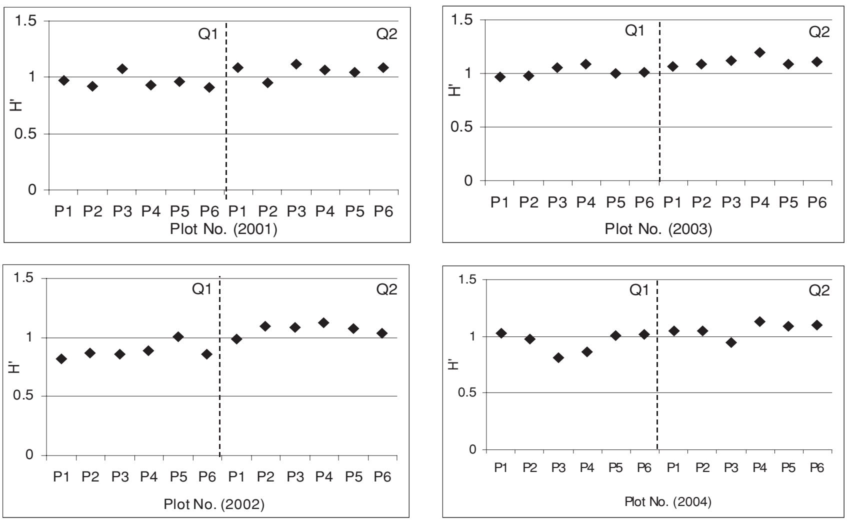

Fig. 3. Species diversity indices of each plot per investigation period. *H*' in 2002 was initially low while the figures increased in Q2 of 2003 and 2004 which continue show an increasing trend.

in common species in Q1 which remained fairly steady even after mowing. This reduced the chances of rare species from sustaining further and prevented new species from encroaching which resulted in low species diversity in Q1 plots. Moreover, although equally treated with mowing, the reduced number of species recorded in Q1 was partly due to speedy recolonization by certain common species such as *M. sinensis* weakening low growth species.

Meanwhile, the concept of differences in quadrant size playing a pivotal role in species spatial distribution patterns was virtually non existent. Although the quadrats sizes were quite different, the plots within them were of equal sizes and there were not much spatial differences, few rosette growth and clumping effects were found in a few low growing species such as *Ixeris dentate* and *Viola grypoceras*. Species invasion after disturbance was seen to be in two paces where low growing species quickly invaded after mowing but where gradually dwarfed and affected by tall growing ones such as *Miscanthus sinensis, Arundinella hirta, Pteridium aquilinum* var. *latiusculum* and *Carex floribunda*. Furthermore, there were not major differences in species due to area size effect but species heterogeneity was evident from one quadrat to another. One factor owing to this evidence was disturbance where mowing effectively destabilized potential dominants. In addition, the quadrats and the plots investigated were randomly plotted to survey enough species. This proved to be quite true in this study that species-area relationship did not have much effect on species diversity **(Fig. 4)**. Combining Q1 (6 plots) and Q2 (6 plots) species dataset for each year, it indicates that there was an increase in the average number of species with an increase in the number of plots. Despite the differences in area size, a corresponding positive increase in species was seen but tends to be constant in the end. This is the basis of rare and new species recorded which were quite steady. Moreover, it was obvious that the average number of species was high in 2003 followed by 2004 while a low number was seen in 2001.

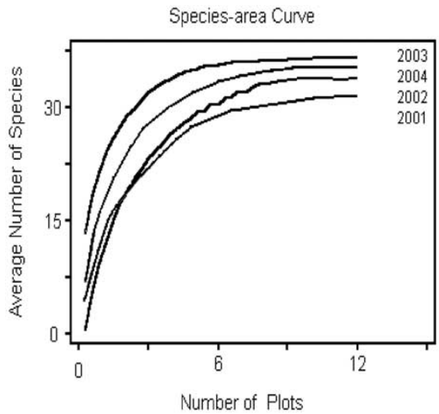

Fig. 4. Species-area relationship of each period showing the changes in mean number of species recorded. An increasing trend was initially seen but tend to stabilize in the end. This is a combined analysis of the two quadrats by year so there are total of 12 plots (6 plots/quadrat)

3.2 Effect of mowing and species richness

According to this study (see Fig.2), although can be inconsistent with other similar studies, mowing was an effective management technique in managing biodiversity and species richness. Analysis against biomass data from each plots per quadrat indicated that there was an increase in species richness **(Fig. 5)**. It shows that species richness generally increased in all plots, that is Q1 and Q2, after mowing but only a high level was seen in 2003 and 2004. Species richness was initially low at a low biomass level but increased at an intermediate stand then tend to decrease with an increased biomass. This is the cornerstone of species richness-biomass relationships, where high species richness can be seen at moderate

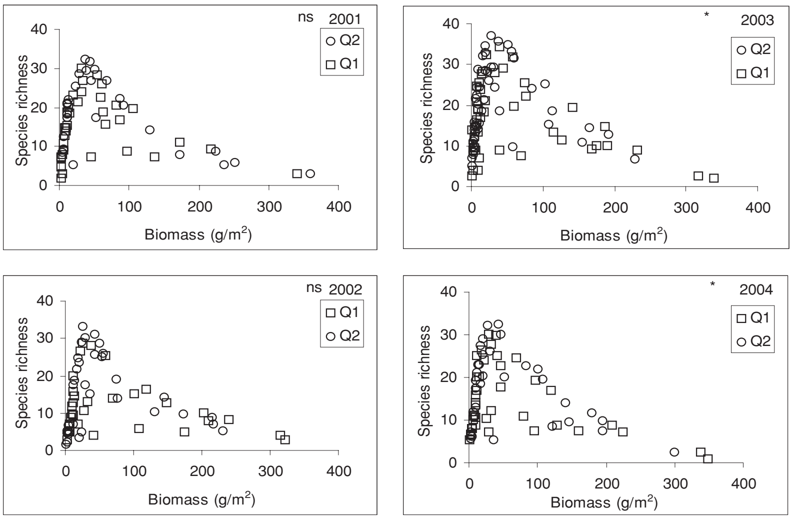

Fig. 5. Species richness plotted against biomass. A high species richness was seen in 2003 and 2004. Plots in Q2 were high in species richness compared to Q1.ns (not significant) and *(significant at 0.05).

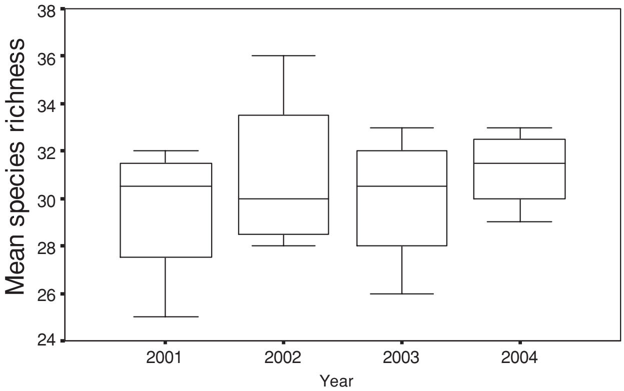

Fig. 6. Mean species richness of each period. An increased scenario of species was seen from 2003. Mean refers to the average values of species per plot for the two quadrats in each year.

levels. Mowing destabilized most common species which consequently had a great impact on the biomass composition of each plot. Although there appeared to be an even distribution of species richness in all, Q2 plots had higher species richness compared to Q1. The one way species analysis (ANOVA) showed significantly high species richness in Q2 (p<0.05) than that of Q1. One factor that gave rise to this scenario was the biomass level and the ability to reinvade and establish by individual species after mowing. It was quite evident that in Q2, mowing seemed to have affected most common species which in turn encouraged less competitive ones. On the other hand, although lower species were seen, the quick recovery of common species such as *M. sinensis, A. hirta, P. aquilinum* var. *latiusculum* and *C. floribunda* after mowing hampered less competitive species.

The positive change in species was also seen from the analysis of mean species richness for each year **(Fig. 6)**. It shows that there was no outlier of the minimum and maximum values while the mean values tend to increase. This adds significance to our finding of increase in species richness under mowing treatment. As can be seen the mean values increased from 2003 to 2004.

4. Discussion

4.1 Mowing and plant species

This study tried to understand whether mowing is a best management practice concerning plant species diversity. From the overview of our findings, there was enough evidence to state that mowing is a suitable management option that can sustain plant diversity. Mowing disturbance was highly favored by species from Compositae (13 spp), Leguminosae (11 spp), Gramineae (10 spp) families followed by species from Rosaceae (9 spp) and Liliaceae (6 spp). Our finding of several species of 53 different families which showed an increase in species richness indicates that mowing played an effective role. This result is similar to those demonstrated in other studies (Abul-faith and Bazzaz, 1979; Hayashi, 1994). The positive relationship between mowing and species diversity indicates that mowing had a profound negative effect on dominancy affecting species such as *M. sinensis, A. hirta*, and *P. aquilinum* var. *latiusculum*.

Vegetative regeneration after mowing differed among plants owing to their ability to survive and regrowth traits after defoliation. Plants with low growth forms or rosettes recovered better than those with erect growth forms (Parr and Way 1988; Mitchley and Willems, 1995). However, when the common species grew taller with lateral branches most low-growth rosettes were suppressed in the latter growth part. It was also seen that most tree and coppice species were severally affected while it supported hemicryptophytes (life-form) with buds or shoot apices situated along the soil surface. Most of which were perennials with rhizomes, bulbs or stem tubers buried horizontally along the soil surface. The total number of species recorded in each investigation period indicated an increasing trend from the initial figures in 2001 (**Fig. 2**). A high number of species was seen from 2003 to 2004 in both Q1 and Q2 followed by 2002 which showed an increased species. Nearly all of the Q2 plots had high number of species compared to Q1. However, a more comparative analysis showed that nearly all the plots in both Q1 and Q2 of 2003 had the highest number of species. Furthermore, there were differences in the distribution of rare and new species. This suggest that although mowing enabled coexistence among different species by weakening potential dominancy, competitive traits of certain common species remained a threat to less competitive ones.

The slightly steady pattern of species distribution seen in most plots of Q1 was due to quick recovery by common species that consequently affected slow growing low species, especially rare ones, some of which were eliminated. This loss of rare species was however compensated with newly recorded species in each plot per quadrat but the tradeoff was low. Regarding newly recorded species, a highest number (5 spp) was seen in Q2 of 2002 followed by Q2 of 2003 (4 spp) and Q2 of 2004 (2 spp). This was seen to play an important role in estimating species diversity indices of each plot. Mowing caused detrimental damages to most of the common species but simultaneously affected rare ones such as *Hydrocotyle javanica, Akebia trifoliate, Paederia scandens* var. *mairei*, and *Pulsatilla cernua*, species which did not effectively reinvade. This suggest that even in managed environments competitive exclusion may continue to exist until such time where dominants are completely destabilized. This logic is arguably true because environments are neither spatially uniform nor temporally constant and their constituent plant populations are subject to fluctuating competition from other species and variable levels of impact from disturbance (Crawley, 1997). In addition, species able to tolerate regular disturbance do not remain constant in environments with adequate resources.

4.2. Disturbance as a factor for controlling biodiversity

Various management regimes including mowing, grazing, trampling and periodic burning can lead to the maintenance of diverse plant communities (Kitazawa and Ohsawa, 2002), but the opposite can happen with extreme scenarios such as abandonment and over grazing (During and Willems, 1984). The moderate mowing conducted in this study enabled high species richness but there was a competitive change in species where perennials tend to lagged annuals. This finding was similarly reported in other studies in Japan (Ikeda, 2003; Kitazawa and Ohsawa, 2002). Each of the different species and their diversity patterns were not the result of temporal factors but spatial heterogeneity established under the disruptive influences of recurrent mowing management. From the species level standpoint, the increase in species diversity indices (*H*') from 2001 to 2004 implies that mowing can sustain biodiversity *(Fig. 3)*. It indicates that Q2 of 2003 and 2004 had high species diversity values. Although mowing promoted branching, horizontal lateral tubes and procumbent tussocks, there was obvious increase in species diversity in Q1 and Q2 from 2003 to 2004. In addition, the action of mowing increment much needed resources such as light and moisture for low growing species.

Furthermore, biomass composition of each plots played a significant role in biodiversity and species richness. Clipping and removal experiments indicated that reduced or increased level of biomass is unsuitable for species richness **(Fig. 5)**. An initial low biomass saw low species richness that reached a peak at moderate biomass level but again decreased with increased biomass. Although mowing tends to decrease above-ground biomass, it is the opinion of this study that mowing over a long period is good not only to keep the existing species but also to assist new invasion. This could relax interspecies competition for resources which eliminate weak ones and increase in coexistence. The combined mean values on species richness per plot per quadrat for each year also justified the increasing trend seen from mowing **(Fig. 6)**. The increased mean values of 2004 suggest that mowing is a good alternate option to promote biodiversity. Given the fact that grasslands have decreased across Japan, it is the arguable perspective of this study to state that mowing should be continued over a long period of time. This is not only to conserve the grasslands but also to protect individual species which might become important food and genetic resources.

The high species diversity and abundance of species of different life origin seen characterizes long management history. According to the local farmers and catalogues from Mt. Sanbe Shizenkan Field

Museum, the area was grazed for 400 years. After the 1960s, mowing was considered as an effective practice to conserve grassland biodiversity. Furthermore, mowing did not only sustain biodiversity but encouraged local tourism and livestock through fodder collection. Mowing also prevented gradual invasion from tree species which effectively drive herbaceous species to extinction. Such events have occurred in other areas such as Chiba where grassland cessation has largely transpired. Therefore, it is our strongest opinion that abandonment of managed grasslands should be avoided.

5. Conclusion

Periodic burning, grazing and mowing management activities aimed at sustaining community or species level biodiversity must be maintained. In Mt. Sanbe, mowing in the east grassland should be continued for a long period of time in order to destabilize the fast-growing common species such as *M. sinensis* and *C. floribunda*. It was obvious that dominancy from these species was a potential threat that can lead to gradual vegetation succession and reduce biodiversity and should be discouraged. Natural species once extinct through landuse alteration and abandonment activities and succession are hardly restored because of weak dispersal and invasive abilities. Thus, it is from this opinion that mowing be continued to conserve the biodiversity of herbaceous plant communities. In addition, further collaboration studies with researchers from different specialist fields but concerning biodiversity, can give good results and this is highly recommended.

Acknowledgement

We would like to thank the Mt. Sanbe Youth Volunteers for assisting field survey and Mrs. Yasuko Takahashi for her help in sampling and lunch. Many thanks are also due to the Nakagoshi and Isagi lab. members of Hiroshima University for their support in this study.

References

Abul-Faith, H.A. and Bazzaz, F.A. (1979), The Biology of *Ambrosia trifida* L. I. Influence of species removal on the organization of the plant community, *New Phytol*, 83, 813-816.

Bakker, J.P. (1989), *Nature Management by Grazing and Cutting*, Kluwer Publishers. Dordrectht.

Bakker, J.P. and Berendse F. (1999), Constraints in the restoration of ecological diversity in grassland and heathland communities, *Trends Ecological Evolution*, 14, 63-68.

Barbaro, L., Dutoit T., Anthelme, F. and Corcket, E. (2004), respective influence of habitat conditions and management regimes on prealpine calcareous grasslands, *Journal of Environmental Management*, 72, 261-275.

Champress S.S. and Morris, K. (1948), The population of buried viable seeds in relation to contrasting pasture and soil types, *Journal of Ecology*, 36, 149-173.

Chippindale, H.G. and Milton, W.E.J. (1934), On the viable seeds present in the soil beneath pastures, *Journal of Ecology*, 22, 508-531.

Cowie, N.R., Sutherland, W.J., Ditlhogo, M.K.M. and James, R. (1992), The effect of conservation management of reed beds. II. The flora and litter disappearance, *Journal of Applied Ecology*, 29, 277-284.

Crawley, M.J. (1997), The structure of plant communities, In: Plant Ecology, 2nd eds., Blackwell Scientific Publications, Oxford. pp. 475-531.

Dithlogo, M.K.M., James R., Lawrence B.R. and Sutherland, W.J. (1992), The effect of conservation management of reed beds, I. The invertebrates, *Journal of Applied Ecology*, 29, 265-276.

During, H.J. and Willems, J.H. (1984), Diversity models applied to a chalk grassland, *Vegetatio*, 57, 241-258.

Forbes, T.J., Dibb, C., Green, J.O., Hopkins, A. and Peel, S. (1980), Factors affecting the productivity of permanent grassland; a national farm study. Joint permanent pasture group, Hurley, Maidenhead Berks.

Gerstmeier, R. and Lang, C. (1996), Beitrag zu Auswirkungen der Mahd auf Arthropoden. Zeitschrift für Ökologie und *Naturschutz*, 5, 1-14.

Grime, J.P. (1973), Competitive exclusion in herbaceous vegetation, *Nature*, 242,: 344-347.

Grime, J.P. (1979), *Plant Strategies and Vegetation Processes*, John Wiley & Sons, Chichester.

Grime, J.P., Rodgson, J.O. and Runt, R. (1988), *Comparative Plant Ecology*: *a functional approach to common British species*, Unwin hyman, London.

Hayashi, I. (1977), Secondary succession of herbaceous communities in Japan, *Japanese Journal of Ecology*, 27, 191-200.

Hayashi, I. (1994), Experimental community ecology in *Miscanthus sinensis* grassland. Change of species composition according to mowing frequency, *Japanese Journal of Ecology*, 44, 161-170. (in Japanese with English summary).

Hils, M.H. and Vankat, J.L. (1982), Species removal from a first-year old-field plant community. *Ecology*, 63, 705-711.

Ikeda, H. (2003), Testing the intermediate disturbance hypothesis on species diversity in herbaceous plant communities along a human trampling gradient using a 4-year experiment in an old-field, *Ecological Research*, 18: 185-197.

Itow, S. (1972), A study of plant succession in abandoned grazed-grasslands, In *Gassland Ecosystem Studies*, *JIBP Grassland Project, Project Report*, 1971. M. Numata eds., pp. 32-38. Japanese Committee for the International Biological Program, Tokyo.

Kamada, A. and Nakagoshi, N. (1996), Landscape structure and the disturbance regime at three rural regions in Hiroshima Prefecture, Japan, *Landscape Ecology*, 11, 15-25.

Kamada, A., Nakagoshi, N. and Nehira, K. (1991), Pine forest ecology and landscape management: A comparative study in Japan and Korea, In: Nakagoshi N. and Golley F.B eds, *Coniferous Forest Ecology from an International Perspective*, SPB Academic Publishing, The Hague. pp. 43-62.

Kitazawa, T. and Ohsawa, M. (2002), Patterns of species diversity in rural herbaceous communities under different management regimes, Chiba, central Japan, *Biological Conservation*, 104, 239-249.

Kobayashi, T., Hori, Y. and Nomoto, N. (1997), Effects of trampling and vegetation removal on species diversity and micro-environment under different shade conditions, *Journal of vegetation Science*, 8, 873-880.

Kollmann, J. and Bassin, S. (2001), Effects of management on seed predation in wildflower strips in northern Switzerland, *Agriculture, Ecosystems and Environment*, 83, 285-296.

Leps, J., Osbornova-Kosinova, J. and Rejmanek, M. (1982), Community stability, complexity and species life history strategies, *Vegetatio*, 50, 53-63.

Mitchley, J. and Willems, J.H. (1995), Vertical canpy structure of Dutch chalk grasslands in relation to their management, *Vegetation*, 117, 17-27.

Moore, D.R.J. and Keddy, P.A. (1989), The relationship between species richness and standing crop in wetlands: the importance of scale, *Vegetatio*, 79, 99-106.

Nakagoshi, N. and Ohta, Y. (1992), Factors affecting the dynamics of vegetation in the landscapes of Shimokamagari Island, southwestern Japan, Landscape Ecology, 7: 111-119.

Numata, M., (1961), Some problems in the secondary succession and judging the serial condition and trend, *Biological Science*, 13, 658-659. (in Japanese).

Numata, M., (1974), Grassland vegetation, In: Numata M. (ed), *The Flora and Vegetation of Japan*, Kodansha Scientific Publications, Tokyo. pp. 125-134.

Ohga, N. and Numata, M. (1965), Ecological analysis of successional development from herbaceous stage to arborescent stage, I. Establishment of *Pinus thunbergii* stand, *Journal of Collective Arts and Sciences*, Chiba University, 4, 263-275, (Japanese with English summary)

Parr, T.W. and Way, J.M. (1988), Management of roadside vegetation: the long term effects of cutting, *Journal of Applied Ecology*, 25, 1073-1087.

Pykälä, J. (2003), Effects of restoration with cattle grazing on plant species composition and richness of semi-natural grasslands, *Biodiversity and Conservation*, 12, 2211-2226.

Vermeer, J.G. and Berendse, F. (1983), The relationship between nutrient availability, shoot biomass and species richness in grassland and wetland communities, *Vegetatio*, 53, 121-126.

WallisDeVries, M.F., Bakker, J.P. and Van Wieren S.E. (1998), *Grazing and Conservation Management*, Kluwer Academic Publishers, Dordrecht.

Walker, J.W. and Warner, R.E. (2000), Grassland management for the conservation of songbirds in the Midwestern USA, *Biological Conservation*, 94, 165-172.

Wheeler, B.D. and Giller, K.E. (1982), Species richness of herbaceous fan vegetation in Broadland, Norfolk in relation to the quantity of above-ground plant material, *Journal of Ecology*, 70, 179-200.

Willems, J.H. (1980), Observations on northwest European limestone grass communities, an experimental approach to the study of species diversity and above-ground biomass in chalk grassland, *Proc. K. Ned. Akad. Wet. Series* C., 83, 279-306.

Zhou, J., Naito, K. and Takahashi, Y. (2002), Secondary vegetation and succession of abandoned grazed grasslands in upland southwestern Honshu, Japan, *Vegetation Science*, 19, 11-23.