

Potential roles of small and linear habitat fragments in *satoyama* landscapes for conservation of grassland plant species

Tomoyo Koyanagi · Yoshinobu Kusumoto ·
Shori Yamamoto · Kazuhiko Takeuchi

Published online: 14 September 2012
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Abstract By comparing with the historic semi-natural grassland and woodland vegetation data compiled in the 1970s and 1980s, we aimed to reveal potential roles of small and linear habitat fragments remaining in the urbanized *satoyama* landscape (i.e., agricultural landscape) in Japan for conservation of grassland plant species. TWINSpan cluster analysis differentiated the current vegetation from historic one. Current vegetation of *Miscanthus* grassland was classified into two different groups and one of the two was equivalent to current understory vegetation of fragmented woodland. The linear woodland edge vegetation along the roads was classified into one group and separated from the other current vegetation groups. The current vegetation groups were characterized by higher richness of exotic species than historic vegetation groups. Detrended correspondence analysis (DCA) revealed that there were no groups of current vegetation that is equivalent to historic woodland understory vegetation. The vegetation quality of the current group of *Miscanthus* grassland and secondary woodland remaining on urban public properties, and the group of linear roadside vegetation were almost equivalent to that of historic semi-natural grassland. Both of the small and linear habitat fragments might have functioned as habitats for grassland species under regular mowing management. Although small and linear habitat fragments would not be sufficient for sustaining grassland populations in the future, these habitats can serve as key reservoirs for grassland species recovery in the conservation and restoration of grassland communities in the urbanized *satoyama* landscape.

T. Koyanagi (✉)
Faculty of Human Sciences, Waseda University, 2-579-15 Mikajima, Tokorozawa, Saitama 359-1192,
Japan
e-mail: t-koyanagi@aoni.waseda.jp

Y. Kusumoto · S. Yamamoto
Biodiversity Division, National Institute for Agro-Environmental Sciences, 3-1-3, Kannondai, Tsukuba,
Ibaraki 305-8604, Japan

K. Takeuchi
Integrated Research System for Sustainability Science (IR3S), The University of Tokyo, 1-1-1, Yayoi,
Bunkyo-ku, Tokyo 113-0032, Japan

Keywords Urban *Miscanthus* grassland · Open secondary woodland · Roadside vegetation · Regional flora · Species diversity · Habitat fragmentation

Introduction

Conservation of urban biodiversity is a major challenge for preventing further loss of biodiversity at local, regional and global scales, and for supporting human well-being through enhancing ecosystem functioning (Alberti and Marzluff 2004; Puppim de Oliveira et al. 2011; Werner 2011). In urban landscapes, especially in developed countries, habitats with high conservation value (i.e., “hotspots”) have mostly deteriorated (e.g. Pysek 1998; Godefroid 2001; Tait et al. 2005). From a conservation point of view, however, it is important to focus not only on hotspots but also on remaining habitat fragments, because these fragments may be relatively important for maintaining and enhancing regional species diversity in highly fragmented landscapes (Cousins 2006; Cousins and Eriksson 2008; Ramalho and Hobbs 2011). Green space that is managed in urban areas to promote recreation and people’s well-being also may support local biodiversity (Bolund and Hunhammar 1999; Colding et al. 2006; McGranahan et al. 2005). Ecosystem services in urban areas have received considerable attention, and numerous studies have revealed the function of green spaces in biodiversity conservation (e.g., Muratet et al. 2008; Sandström et al. 2006; Sattler et al. 2010). Not only small fragmented habitats in green spaces but also linear landscape components such as roadsides, railway-sides, and riverbanks can have important roles for species conservation in urban landscapes and their surroundings (Cousin 2006; Hovd 2008; Noordijk et al. 2011). It is essential to focus on these small and linear habitat fragments for preventing further loss of urban biodiversity.

Satoyama landscapes, the traditional ‘socio-ecological production landscape’ of Japan, consist of a mosaic of grasslands and woodlands, rice paddy fields, crop fields, streams and ponds (Fukamachi et al. 2001; Takeuchi et al. 2003). There are various types of socio-ecological production landscapes similar to *satoyama* landscapes not only in Asia but also in Europe, Africa and many other areas in the world, characterized by human-influenced nature environments with high biodiversity (The Satoyama Initiative 2010). *Satoyama* landscapes once covered more than 40 % of the whole land area in Japan (Ministry of the Environment 2002), but have undergone dramatic changes by intensification of agricultural system and urbanization during the past 100–150 years. Negative impacts of rapid urbanization and agricultural intensification on biodiversity can be recognized many parts of the socio-ecological production landscapes in the world including *satoyama* landscapes (The Satoyama Initiative 2010).

Semi-natural grassland was one of the main components of *satoyama* landscapes. Japanese people used to maintain a vast area of semi-natural grassland to produce fodder for farm animals and organic fertilizer for crop fields and rice paddy fields (Sprague 2001; Takeuchi et al. 2003). The total area of semi-natural grassland in Japan was estimated to have decreased from more than 10 % of the whole land area in the 1880s to less than a few percent in recent years (Himiyama 1995; Shoji 2003). Many grassland species have been designated as rare or threatened because of the drastic loss of habitats for these species by rapid urbanization and agricultural intensification, especially in the urbanized *satoyama* landscapes surrounding metropolitan areas (Ide 2006).

Increasing recognition of the biodiversity value of semi-natural vegetation in *satoyama* landscapes especially since the 1990s has made it possible to retain green spaces for conserving urban biodiversity (Takeuchi 2006; Takeuchi et al. 2003). In the course of development, some of the historic semi-natural woodland and grassland areas have been left in place (e.g., public parks), and these areas may have retained their functional role as habitats for grassland species.

It is urgently needed to clarify whether populations of grassland plant species that were common in the historic *satoyama* landscapes can be sustained in the remaining small and linear habitats in the present urbanized *satoyama* landscapes. The remaining populations of grassland species could be important sources for not only maintaining but also enhancing local and regional biodiversity in urbanized *satoyama* landscapes.

Because most monitoring efforts have been confined to relatively rare natural or semi-natural habitats (Japan Wildlife Research Center 2006), information on the historic reference vegetation of *satoyama* landscapes tends to be lacking. In the early 1970s, phytosociology was introduced in Japan and nationwide vegetation surveys based on the methods of Braun-Blanquet (1964) were conducted (Miyawaki 1977). The regional flora data compiled in the 1970s and 1980s are the oldest data based on the unified phytosociological methodology, and these data could be critical for identifying references of the target semi-natural grassland vegetation in *satoyama* landscapes.

In this study, we aimed to reveal the potential roles of small and linear habitat fragments of semi-natural vegetation in the conservation of grassland plant species in highly urbanized *satoyama* landscape. We clarified whether the current semi-natural vegetation remaining in the landscape is comparable to that of historic semi-natural grassland. The specific objectives were: (1) to compare the species composition in the current small and linear grassland and open secondary woodland vegetation remaining in the urbanized *satoyama* landscapes with those of historic reference semi-natural vegetation; (2) to reveal richness patterns of grassland species and exotic species in the current and historic vegetation; and (3) to evaluate the vegetation quality and the roles of small and linear habitats for preventing further loss of grassland species diversity in the urbanized *satoyama* landscape.

Materials and methods

Study area

This study was conducted in southern Tsukuba City, Ibaraki Prefecture, in the Kanto region, Japan (Fig. 1). The study area was located in the *Tsukuba–Inashiki* upland region, where the elevation ranges from approximately 20 to 30 m above sea level. The mean annual temperature is 13.5 °C (2.3 °C in January and 25.2 °C in August), with a mean annual precipitation of 1,230 mm at Tsukuba (Tateno) meteorological observatory (Japan Meteorological Agency 2007).

Tsukuba Science City has been developed in the study area since the 1970s, and a number of research and educational institutes were constructed with both public and private housings. Semi-natural habitats for grassland species, which include both semi-natural grassland and open secondary woodlands, were widespread around the study area in the 1880s (Fig. 1). However, nearly all these habitats have disappeared particularly during the last 50 years, due to rapid urbanization accompanied with the development of Tsukuba Science City (Sakiyama and Itoga 1994). Most of the woodland areas that escaped land-use conversion outside the planning areas of Tsukuba Science City were abandoned and underwent further succession to mixed deciduous-evergreen forest with a fully developed canopy (Inui 1992). In the present highly fragmented landscape, limited numbers of semi-natural habitats for grassland species remain, mostly in areas under regular management on public properties within Tsukuba Science City. In addition to the small habitat fragments, linear roadside vegetation along the edges of remnant woodlands could be another key habitat for grassland species because the vegetation has been maintained by the regular management (mowing) to improve traffic safety.

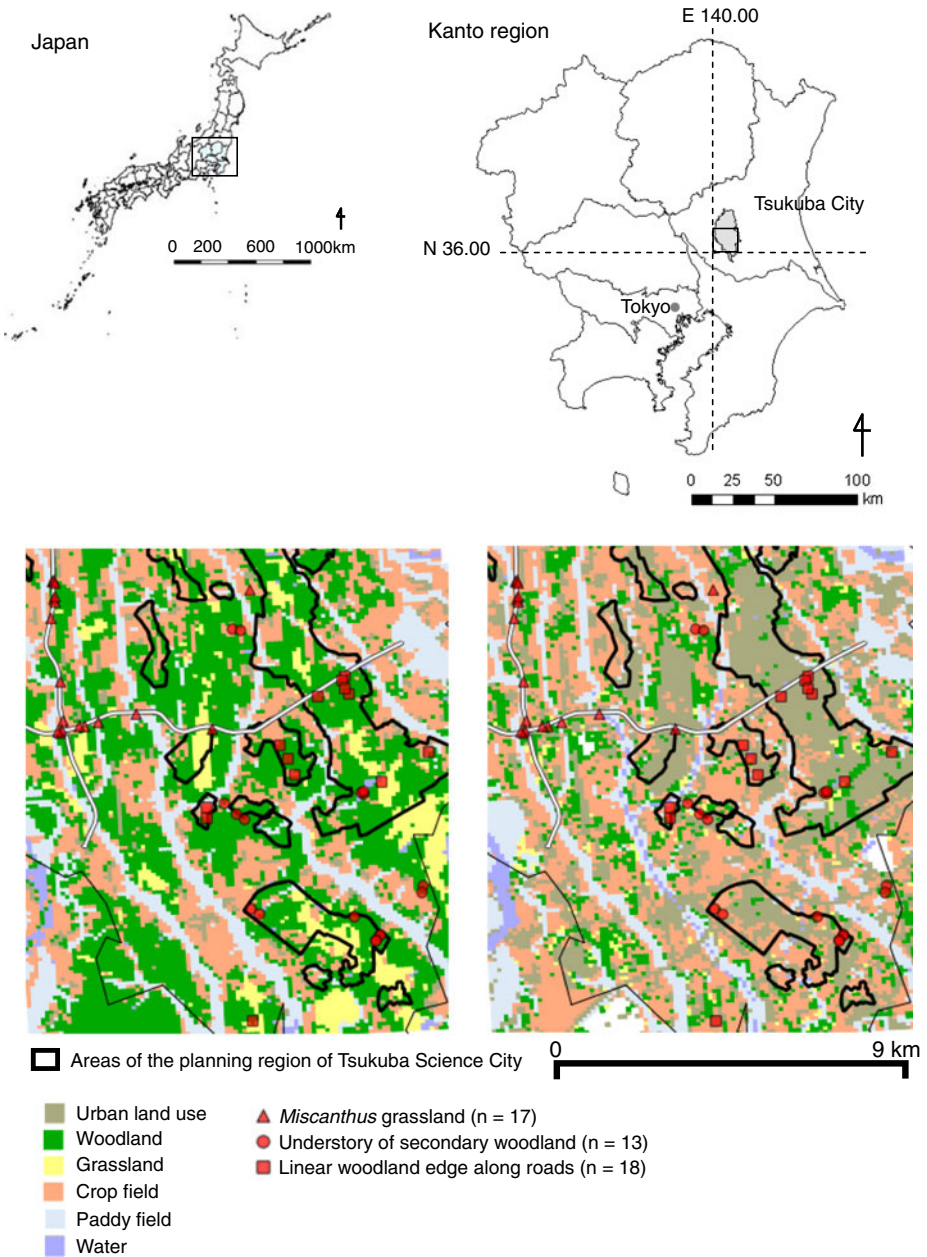


Fig. 1 Location of the study area and the sampled sites, with information on land use in the 1880s and 1990s

Target semi-natural vegetation

The study area belongs to the warm-temperate, monsoonal, mixed evergreen-broadleaved forest region, with the climax dominants *Castanopsis cuspidata* var. *sieboldii*, *Quercus myrsinifolia*, and *Machilus thunbergii*. The vegetation of semi-natural grasslands maintained by the traditional

agricultural management (i.e., burned in early spring and mown once in summer) found in this region belongs to the *Pleiolobus chino*–*Miscanthus sinensis* phytosociological community (Miyawaki 1986). We defined characteristic species in this community as typical grassland species in the study area. The open secondary woodland, which consists mainly of *Pinus densiflora*, associated with *Quercus acutissima* and *Quercus serrata*, is another type of habitat for grassland species in the study area. These woodlands were cut once every 20 to 30 years, and the understory layer was mowed to produce organic fertilizer or livestock feed. These management practices maintained high light availability which enabled many grassland species to coexist in the understory vegetation (Fujii and Zinnai 1979; Yamamoto and Itoga 1988).

Vegetation sampling

We targeted fragmented grassland vegetation dominated by *M. sinensis* and the understory vegetation of managed open secondary woodland remaining on urban public and private properties in the study area (i.e., in Tsukuba Science City and surrounding areas) (Fig. 1). We also targeted the linear woodland edge vegetation along the two major roads running north–south and east–west maintained by public mowing management for traffic safety. We conducted vegetation surveys at 17 grassland and 13 woodland sites, and 18 sites of roadside along the edges of woodlands during summer in 2006, 2007 and 2008. The surveys were conducted in areas that used to be semi-natural grasslands or open woodlands within the study area (Fig. 1) and where the grassland and understory of woodland vegetation plots was dominated or sub-dominated by *M. sinensis*.

In order to compare with historic vegetation data, the vegetation survey was based on the phytosociological method of Braun-Blanquet (Mueller-Dombois and Ellenberg 1974) which was also used in past surveys (Miyawaki and Suzuki 1974; Miyawaki et al. 1975; Miyawaki 1986; Sakurai 1982). We recorded all species using cover rank values (5: 100–75 %, 4: 75–50 %, 3: 50–25 %, 2: 25–10 %, 1: 10–1 %, +: < 1 %, r: appeared only once) at each surveyed site. Depending on the entire area of each site, the total area surveyed ranged from 15 to 400 m² at *Miscanthus* grassland sites and from 20 to 75 m² at secondary woodland sites. Each survey plot was surveyed starting from approximately the center of each fragmented habitat and survey area was increased until new species were no longer encountered. We also surveyed the woodland edge vegetation along roads within 1–2 m from the pavement edge. The surveyed lengths of roadside vegetation ranged from 10 to 40 m depending on the length of woodland edges.

Historic vegetation data

We gathered vegetation data for the *P. chino*–*M. sinensis* community and pine woodland community representing the historic grassland and secondary woodland vegetation, respectively, from the regional flora compiled in the 1970s and 1980s (Miyawaki and Suzuki 1974; Miyawaki et al. 1975; Miyawaki 1986; Sakurai 1982). We gathered the historic vegetation data conducted in *Tsukuba–Inashiki* and *Shimousa* upland regions both located in the eastern part of the Kanto plain and under the almost same geographical conditions. Historic vegetation data of *Miscanthus* grasslands has some variation, including semi-natural grassland maintained by the traditional management practices and grassland established on open disturbed habitats after abandonment of cultivation or land reclamation (Miyawaki 1986). By referring the information on the management of each surveyed vegetation plots, we selected vegetation data for 9 *Miscanthus* grassland sites representing the past semi-natural grassland community. To represent vegetation of the managed open secondary woodland, we extracted understory vegetation data of the pine woodland sites that had a sparse shrub layer with ≤ 40 % cover, 20 woodland sites in total.

Data analysis

Data analysis consisted of three steps. First, to identify and characterize different vegetation types, we classified historic and current vegetation data using a classification analysis called TWINSpan (Two-way Indicator Species Analysis; Hill 1979b). Secondly, we examined differences in species richness and diversity patterns among historic and current vegetation groups. Finally, we analyzed species compositional data using an ordination method called detrended correspondence analysis (DCA; Hill 1979a) to clarify the differences in vegetation quality among the historic and current vegetation groups.

We used both historic and current vegetation data (77 sites) in the TWINSpan classification using presence–absence data of species at each site. All species recorded in less than three sites in the historic and current vegetation data were excluded from the TWINSpan analysis. We classified the recorded species into five species groups (grassland species, exotic species, native annuals, native perennials, and woody species), and compared the richness values of all species and each species group for both historic and current vegetation groups identified by TWINSpan. We performed generalized linear models (GLMs) with a logarithmic link function that follows Poisson distribution to investigate the differences in richness values among the historic and current vegetation groups. We then performed pair-wise comparison of richness values between all pairs of vegetation groups using simultaneous inference methods (Hothorn et al. 2008). *P*-values were adjusted for multiple comparisons using the conservative Bonferroni method. Finally, we performed detrended correspondence analysis (DCA) with species compositional data of the historic and current vegetation simultaneously using log-transformed cover rank values (“+”: <1 %” and “r: appeared only one time” were transformed into 0.1 and 0.01, respectively; Mueller-Dombois and Ellenberg 1974). The relationships between DCA axes 1 and 2 scores and richness values of all species and different species groups were tested by Spearman’s rank correlation test. We also tested the significance of differences in the scores of DCA axes 1 and 2 among the vegetation groups identified by TWINSpan using Tukey’s multiple comparison test. The analytical program PC-ORD for Windows version 4 (McCune and Mefford 1999) was used for TWINSpan and DCA. All other statistical tests were performed with R version 2.12.2 (R Development Core Team 2011).

Results

A total of 285 plant species were recorded in the historic and current vegetation data. We also recorded some plants that could not be identified to the species level (13 individuals in Poaceae spp., 2 individuals in Compositae spp., 1 individual in Liliaceae spp., 17 individuals in Carex spp., 7 individuals in Rumex spp., 5 individuals in Sonchus spp., and 1 individual in Solanaceae spp.). These unidentified plant individuals were not included in the following data analyses. All the recorded typical grassland species are listed in Appendix 1. A total of 39 and 52 grassland species were recorded in the historic and current vegetation, respectively, and 35 grassland species were common to both vegetation. Four species (*Patrinia villosa*, *Achillea alpina* var. *longiligulata*, *Bupleurum stenophyllum* and *Gentiana scabra* var. *buergeri*) were recorded only in the historic vegetation data.

TWINSpan cluster analysis classified the 77 vegetation samples into five vegetation groups, differentiating the current vegetation from the historic one (Fig. 2, Table 1). The current vegetation was classified into three groups (C-L, C-GW, and C-G). Almost half of

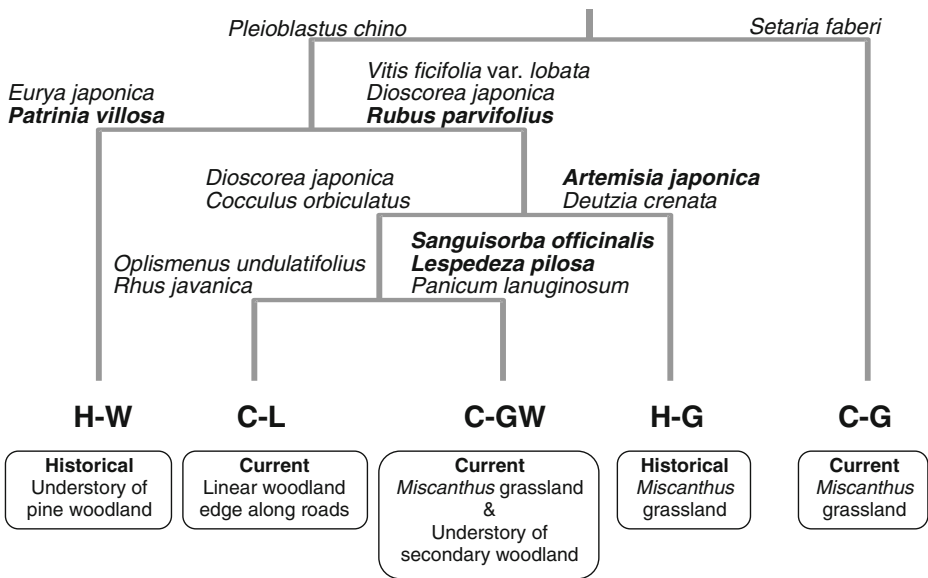


Fig. 2 TWINSpan classification of the historic and current vegetation. Indicator species are shown along the branches. The typical grassland species are shown in boldface. Correspondence between the historic and current vegetation types and vegetation groups identified by TWINSpan are also shown in Table 1

the current *Miscanthus* grassland vegetation was classified into the same group with the understory vegetation of secondary woodland (C-GW) characterized by typical grassland species (*Sanguisorba officinalis* and *Lespedeza pilosa*) and also exotic species (*Panicum lanuginosum*). The other current *Miscanthus* grassland vegetation was classified into one group (C-G) characterized by annual species (*Setaria faberi*). Most of the current linear woodland edge vegetation along the road was classified in to one group (C-L) characterized by native perennial grass (*Oplismenus undulatifolius*) and broadleaved tree (*Rhus javanica*). The historic semi-natural grassland vegetation and the understory vegetation of pine woodland were classified into two different groups (H-G and H-W, respectively). The characteristic species of H-G were typical grassland species (*Artemisia japonica*) and deciduous shrub (*Deutzia crenata*), whereas those of H-W were ever-green shrub (*Eurya japonica*) and typical grassland species (*Patrinia villosa*).

Richness values of all species were significantly higher in H-G, C-L and C-GW compared to C-G (Fig. 3a). The richness values of grassland species were also higher in H-G and C-GW compared to the other three groups. Exotic species richness was significantly higher in the current vegetation groups of C-GW and C-G compared to the historic ones (Fig. 3b). In C-G, richness of native annuals was significantly higher than the other groups, whereas richness of woody species was the lowest among the groups.

The historic and current vegetation groups identified by TWINSpan classification were roughly ordered along axis 1 of the DCA ordination (Fig. 4a). The distribution of plots based on the current and historic vegetation types are shown in Appendix 2. The scores of axis 1 (eigenvalue=0.433) were positively correlated with the richness of exotic species ($r=0.577$, $P<0.0001$) and native annuals ($r=0.582$, $P<0.0001$), and negatively correlated with those of woody species ($r=-0.599$, $P<0.0001$). The scores of axis 2 (eigenvalue=0.255) were positively correlated with the richness of grassland species ($r=0.313$, $P=0.006$). The average scores of DCA axis 1 were significantly

Table 1 Correspondence between the historic and current vegetation types and vegetation groups identified by TWINSpan (see, Fig. 2). The values show the numbers of surveyed sites

Vegetation type	Vegetation groups classified by TWINSpan				
	H-W	C-L	C-GW	H-G	C-G
Current vegetation					
<i>Miscanthus</i> grassland	0	1	6	0	10
Understorey of secondary woodland	0	2	11	0	0
Linear woodland edge along roads	1	15	2	0	0
Historic vegetation					
<i>Miscanthus</i> grassland	0	0	0	7	2
Understorey of pine woodland	16	2	0	2	0

higher in C-G and lower in H-W among the groups (Fig. 4b). There are no significant differences in the scores between C-L and H-G and between C-GW and H-G. The

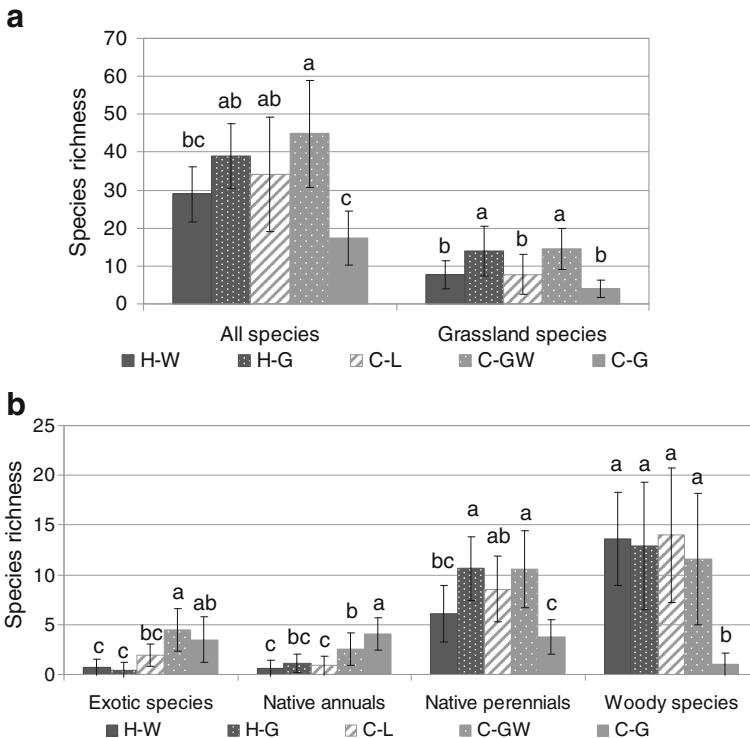
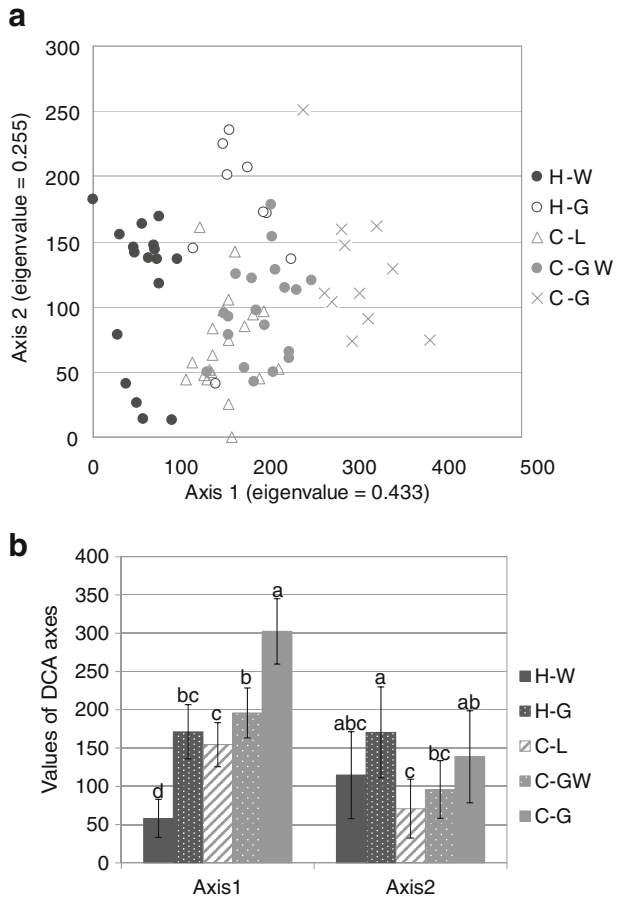


Fig. 3 Average richness values of **a** all species and grassland species, and **b** exotic species, native annuals, native perennials and woody species of the historic and current vegetation groups classified by TWINSpan (see Fig. 2). Differences among the groups for each of the richness value were tested for significance using pair-wise comparison between any two vegetation groups using simultaneous inference methods (Hothorn et al. 2008). *P*-values were adjusted for multiple comparisons using the conservative Bonferroni method. The error bars represent standard deviation within the groups. Different letters showed significant differences among the groups ($P < 0.05$)

Fig. 4 **a** DCA ordination showing all the vegetation plots of the historic and current vegetation groups classified by TWINSpan (see Fig. 2). **b** Average scores of DCA axes 1 and 2 of the five vegetation groups. Differences among the groups for each of the richness value were tested for significance using Tukey’s multiple comparison test. Different letters show significance of differences among the groups ($P < 0.05$). The error bars represent standard deviation within the groups



average scores of DCA axis 2 were significantly lower in C-L and C-GW compared with H-G but not significantly different with H-W (Fig. 4b).

Discussion

This study revealed that small remnant *Miscanthus* grassland and secondary woodland habitats on urban public and private properties could function as habitats for grassland species. The linear woodland vegetation along the roads could also serve as habitats for grassland species in the study area. Although these habitats have been invaded by exotic species, the vegetation quality (i.e. species compositions represented by DCA axis 1) showed no significant differences compared with those of the historic reference vegetation of semi-natural grassland. Among the 39 grassland species recorded in the historic vegetation data, four species were absent in the current vegetation surveyed in the study area (Appendix 1). One of the species, *Bupleurum stenophyllum*, and the other two species, *Platycodon grandiflorus* and *Cynanchum paniculatum*, have been listed as VU (vulnerable), VU and NT (near threatened), respectively in the national RDB (Ministry of the Environment 2000). In addition to the above species threatened

at the national level, some other species that use to be common in past *sato-yama* landscapes (e.g. *Sanguisorba officinalis*, *Patrinia scabiosaefolia*, *Adenophora triphylla* var. *japonica*) might become rare or threatened in the near future (Okubo 2002; Ide 2006).

The current vegetation group C-GW consisted both of the *Miscanthus* grassland and the understory of secondary woodland vegetation. This vegetation would be nearly equivalent to the historic reference semi-natural grassland vegetation (H-G) but not to the historic understory vegetation of pine woodland (H-W). Although the current vegetation of C-GW was invaded by exotic species, grassland species richness in this group was equally high compared with that of historic semi-natural grassland (H-G). Among the 19 sites classified into C-GW, 3 *Miscanthus* grassland sites and 10 secondary woodland sites are located on urban public properties (i.e. inside the areas of national institutes and public housing estates) and are maintained by regular management. The other *Miscanthus* grassland and secondary woodland sites are private and might have been managed regularly by the landowners. On the urban public and private properties, most of the remaining semi-natural vegetation has been managed regularly (i.e. mowing aboveground vegetation once or twice during the summer) to keep the open environment for esthetic preference or other purposes of people's use. Because of the management practices, the current vegetation of these sites (not only *Miscanthus* grassland sites but also secondary woodland sites) has been kept at the earlier successional stage compared to historic understory vegetation of pine woodland (H-W) (Fig. 4b, represented by the score of DCA axis 1). The fact that most of these habitats were quite small (i.e. less than 100 m²) and surrounded/bordered by roads and built-up areas explain why these habitats were highly invaded by exotic species and other native ruderal annals.

Linear roadside vegetation has received increasing attention for conserving grassland plant species (e.g., Tikka et al. 2000; Cousin and Eriksson 2002; Jantunen et al. 2007). However, such vegetation has experienced the transition from traditional agrarian management (i.e. subscribed burning in early spring and mowing once in summer) to current road verge management with frequent mowing (i.e. two or three times during summer) and additional disturbances such as chemical pollution by automobile fumes and soil fertilization by adjacent land-uses (Tikka et al. 2000; Jantunen et al. 2007). This drastic change in disturbance regime has resulted in the observed changes in species compositions and grassland species richness from those of the reference semi-natural grassland (Koyanagi et al. 2012). In this study, the group of linear woodland edge vegetation was differentiated from the other vegetation types (*Miscanthus* grassland and understory of secondary woodland) and categorized into one group C-L. Although the current vegetation of group C-L was almost at the same successional stage as the historic semi-natural grassland (H-G) and current *Miscanthus* grassland and understory of woodland vegetation (C-GW), grassland species richness was significantly lower in C-L than H-G and C-GW. The linear vegetation of roadsides can serve as dispersal corridor for not only grassland species but also exotic species (Tikka et al. 2000; Thiele and Otte 2008). In addition to the exotic species invasion, linear woodland edge vegetation along the roads might have been suffering further loss of grassland species in the future (Koyanagi et al. 2009, 2012).

The current vegetation group C-G was characterized by annual species and was significantly different from historic vegetation groups of semi-natural grassland (H-G) and understory of pine woodland (H-W). All *Miscanthus* grassland sites classified into C-G were located on private properties, appeared to have been established by land reclamation, and were kept open for future housing developments (Koyanagi et al. 2007). Because *M. sinensis* is dominant, the vegetation of these grassland sites can also be classified as the *P. chino*–*M. sinensis* community to which the historic reference semi-natural grassland belonged

(Miyawaki 1986), but with fewer grassland species. Most of the remaining *Miscanthus* grasslands classified into group C-G might be transient and established before future development of housing and other buildings by land reclamation. The vegetation was characterized by annuals and exotic species at earlier successional stage having little conservation value in the present urbanized landscape.

Although populations of grassland species still exist in the present urbanized *satoyama* landscape, the remaining habitats appear to be too small and fragmented for maintaining and conserving grassland species diversity in the future. The previous studies conducted in the same study area have revealed that there were significant time-lags of grassland species decline after the landscape changes occurred during the past 50 years (Koyanagi et al. 2009, 2012). This may be evidence of “extinction debt” in the landscape (Kuussaari et al. 2009). The area and quality of remaining habitats in the current landscape are not enough to sustain the grassland populations in the future unless habitats are restored. In Japanese *satoyama* landscapes, the earlier conservation policies have primarily considered the secondary woodlands and the species associated with these habitats (e.g. forest plant spring ephemerals). Community-based activities for management of secondary woodlands have intensified since the late 1990s, especially in the urban and rural landscapes surrounding metropolitan areas (Yamasaki et al. 2000; Okubo et al. 2005; Ide 2006). Conservation of grassland species can be possible by creating open environment such as well-managed secondary woodlands, and therefore could be part of the objectives of the existing policies and people’s activities in *satoyama* landscapes. Various remnants of the past semi-natural habitats with grassland plant species growing in abandoned overgrown sites can be still found in the present highly fragmented landscape. Although the existence of extinction debt has been suggested in the landscape (Koyanagi et al. 2009, 2012), there is still opportunities to prevent local extinctions of grassland species by restoring semi-natural habitats suitable for recolonization (Cousins 2009; Kuussaari et al. 2009). Resuming extensive management of the abandoned woodlands that still retain a pool of some grassland species in the roadside vegetation could be an effective conservation and restoration measures to enhance grassland species diversity in the region. Improving the management conducted both in the roadside and other small habitats in public properties can be another effective way of promoting grassland species recolonization and regeneration (Jantunen et al. 2007). Mowing management has been conducted two or three times in summer, and mown vegetation is removed by the public sector. This management regime might have negative impacts on flowering success and seed production of early and late summer-flowering grassland species, so conducting extensive mowing after their flowering and seed setting (i.e. late summer or autumn) is recommended. Although the current areas of remaining habitats are not enough for sustaining grassland species population, the small and linear habitats on urban public and private properties can serve as key reservoirs for grassland species recovery that contribute to enhance regional species diversity in the highly fragmented urbanized *satoyama* landscape. Utilization of historic or past vegetation data can be important tool for describing potential roles of remaining habitat fragments for species conservation and biodiversity management, especially in highly urbanized *satoyama* landscapes and any other ‘socio-ecological production landscapes’ around the world.

Acknowledgments We thank K. Fujiwara and T. Fuji, who compiled the regional flora, for providing detailed information on the historic vegetation data. T. Nakamura kindly provided additional historic vegetation data in the study area. We also thank S. Yamada and Y. Tokuoka for helping with vegetation surveys. T. Furukawa kindly edited English and gave us informative comments to improve our manuscript. This study was financially supported by the 21st Century COE Program, Biodiversity and Ecosystem Restoration, with additional support from JSPS research fellowship to T. Koyanagi.

Appendix 1

Table 2 List of the typical grassland species recorded in the current and historic vegetation groups

Typical grassland species	No. of sites recorded		Current vegetation										Historic vegetation				
	Current vegetation	Historic vegetation	Miscanthus grassland			Understory of secondary woodland			Linear woodland edge along roads				Miscanthus grassland		Understory of pine woodland		
			C-G (n=10)	C-GW (n=6)	C-L (n=1)	C-GW (n=11)	C-L (n=2)	C-L (n=15)	C-GW (n=2)	H-W (n=1)	H-G (n=7)	C-G (n=2)	H-W (n=16)	H-G (n=2)	C-L (n=2)		
<i>Miscanthus sinensis</i>	42	29	10	6	1	11	2	11	1	0	0	7	2	16	2	2	
<i>Rubus parvifolius</i>	32	11	1	5	1	11	2	11	1	0	0	4	0	4	2	1	
<i>Potentilla freyniana</i>	26	17	0	4	1	9	2	8	1	1	1	4	0	11	2	0	
<i>Lеспедеза pilosa</i>	21	18	4	4	1	9	0	2	1	0	0	6	0	11	1	0	
<i>Sanguisorba officinalis</i>	24	13	1	6	1	9	1	4	2	0	0	4	0	9	0	0	
<i>Lеспедеза bicolor</i> f. <i>acutifolia</i>	23	11	0	3	1	9	1	8	1	0	0	2	0	9	0	0	
<i>Solidago virgaurea</i> var. <i>asiatica</i>	14	17	0	3	1	7	1	1	1	0	0	2	2	13	0	0	
<i>Arundinella hirta</i>	18	10	0	4	1	8	1	4	0	0	0	6	2	0	1	1	
<i>Pteridium aquilinum</i> var. <i>latiusculum</i>	13	11	0	5	1	3	2	2	0	0	0	6	0	4	1	0	
<i>Eupatorium chinense</i> var. <i>oppositifolium</i>	18	5	0	3	1	7	2	4	1	0	0	2	0	0	2	1	
<i>Aster scaber</i>	14	7	0	1	1	5	1	5	1	0	0	0	0	6	1	0	
<i>Ilex dentata</i>	18	0	2	0	0	5	2	8	1	0	0	0	0	0	0	0	
<i>Hydrocotyle ramiflora</i>	14	3	1	0	0	7	2	2	2	0	0	3	0	0	0	0	
<i>Indigofera</i> <i>pseudotinctoria</i>	2	15	0	1	0	0	0	1	0	0	0	4	2	7	1	1	

Table 2 (continued)

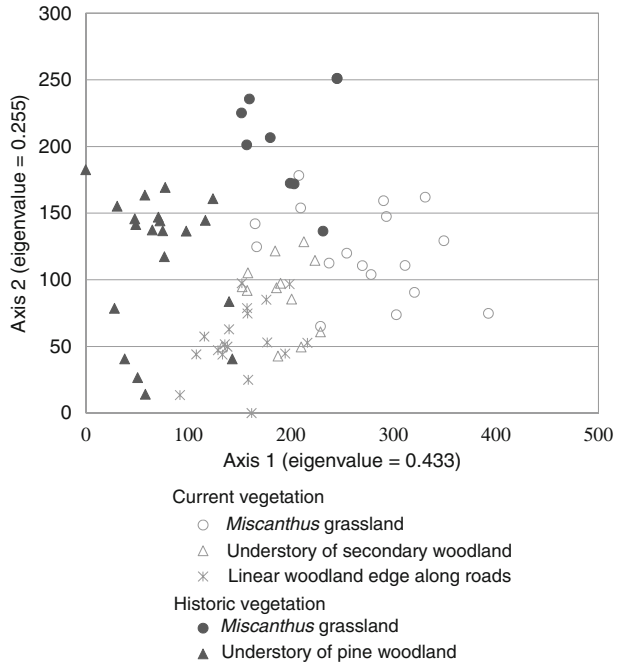
Typical grassland species	Current vegetation	No. of sites recorded	Current vegetation						Historic vegetation									
			Miscanthus grassland			Understory of secondary woodland			Linear woodland edge along roads			Miscanthus grassland			Understory of pine woodland			
			C-G (n=10)	C-GW (n=6)	C-L (n=1)	C-GW (n=11)	C-L (n=2)	C-L (n=15)	C-GW (n=2)	H-W (n=1)	H-G (n=7)	C-G (n=2)	H-W (n=16)	H-G (n=2)	C-L (n=2)			
<i>Lysimachia clethroides</i>	4	3	0	1	0	0	2	0	0	1	0	0	2	0	0	0	1	0
<i>Scilla sinensis</i>	5	2	1	0	0	0	4	0	0	0	0	0	2	0	0	0	0	0
<i>Hypericum erectum</i>	6	0	0	2	1	2	0	0	0	1	0	0	0	0	0	0	0	0
<i>Zoysia japonica</i>	4	2	0	2	0	0	1	0	0	1	0	0	0	0	2	0	0	0
<i>Viola grypoceras</i>	6	0	0	2	1	0	0	1	0	2	0	0	0	0	0	0	0	0
<i>Lilium auratum</i>	2	3	0	0	1	0	0	0	0	1	0	0	2	0	0	0	1	0
<i>Prunella vulgaris</i>	4	0	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0
ssp. <i>asiatica</i>																		
<i>Thesium chinense</i>	4	0	1	2	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Gentiana scabra</i>	0	4	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0
var. <i>buergeri</i>																		
<i>Inula salicina</i> var. <i>asiatica</i>	1	2	0	1	0	0	0	0	0	0	0	0	0	2	0	0	0	0
<i>Platycodon grandiflorum</i>	2	1	0	2	0	0	0	0	0	0	0	0	1	0	0	0	0	0
<i>Polygala japonica</i>	1	2	0	1	0	0	0	0	0	0	0	0	2	0	0	0	0	0
<i>Petasites japonicus</i>	1	2	0	1	0	0	0	0	0	0	0	0	2	0	0	0	0	0
<i>Eccolopus cotinifolius</i>	1	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0
<i>Agrimonia japonica</i>	2	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Viola mandshurica</i>	2	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Spiranthes sinensis</i> var. <i>amoena</i>	2	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0

Table 2 (continued)

	No. of sites recorded		Current vegetation					Historic vegetation							
	Current vegetation	Historic vegetation	<i>Miscanthus</i> grassland		Understory of secondary woodland		Linear woodland edge along roads		<i>Miscanthus</i> grassland		Understory of pine woodland				
			C-G (n=10)	C-GW (n=6)	C-L (n=1)	C-GW (n=11)	C-L (n=2)	C-L (n=15)	C-GW (n=2)	H-W (n=1)	H-G (n=7)	C-G (n=2)	H-W (n=16)	H-G (n=2)	C-L (n=2)
Typical grassland species	Current vegetation	Historic vegetation	C-G (n=10)	C-GW (n=6)	C-L (n=1)	C-GW (n=11)	C-L (n=2)	C-L (n=15)	C-GW (n=2)	H-W (n=1)	H-G (n=7)	C-G (n=2)	H-W (n=16)	H-G (n=2)	C-L (n=2)
<i>Achillea alpina</i> var. <i>longiligulata</i>	0	2	0	0	0	0	0	0	0	0	2	0	0	0	0
<i>Bupleurum scorzonerifolium</i> var. <i>stenophyllum</i>	0	2	0	0	0	0	0	0	0	0	2	0	0	0	0
<i>Calamagrostis epigetos</i>	2	0	0	1	0	1	0	0	0	0	0	0	0	0	0
<i>Dianthus superbus</i> var. <i>longicalycinus</i>	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Cynanchum paniculatum</i>	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0
<i>Arabis flagellosa</i>	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Sweria japonica</i>	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Iris ensata</i> var. <i>spontanea</i>	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
<i>Lespedeza virgata</i>	1	0	0	0	0	1	0	0	0	0	0	0	0	0	0

Appendix 2

Fig. 5 DCA ordination showing all the vegetation plots of the current and historic vegetation types



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