# Evaluation of techniques for restoration of natural forest community structure and dynamics

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# 博 士 学 位 論 文

## Evaluation of techniques for restoration of natural forest community structure and dynamics

## 群落の構造と動態を重視した自然林再生法の研究

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<Abstract>


## Abstract

Among the various methods of forest restoration in Japan, restoration of mid-successional, secondary forests, which have high biodiversity, could contribute to maintaining biological diversity in human-disturbed landscapes. In many restoration projects, the neighboring natural secondary forest is often chosen as the target vegetation. However, there is no established methodology for secondary forest restoration in Japan. Here, I compared tree species composition and stand structure between a restored site (10 years after planting) and the neighboring secondary forest designated as the target vegetation. I found that restoration contributed to increasing tree cover, but the vertical structure of the stand was poorly developed. Moreover, the species composition of the neighboring secondary forest had changed markedly in ten years, mainly due to pine-wilt disease. As a result, compositional similarity between the restored and target plots had not increased in 10 years, while structural similarity was lower than 10 years ago. I conclude that, although current forest restoration methods contribute to increasing vegetation cover, single plantings of native species may be insufficient to restore the natural community structure, especially because vegetation of the target forest is constantly changing. Subsequent adaptive management is needed to direct succession of restoration sites toward natural forest composition and structure.

To integrate human-disturbed hillslopes with the regional landscape, natural forest restoration has become an important objective of hillslope re-vegetation in Japan. At Kobe Municipal Sports Park (KMSP), seedlings of native species were planted in 1980 to restore semi-natural secondary forest (satoyama) in an urban setting. Here, I present 21 years of stand dynamics based on vegetation surveys conducted in 1992, 2000, and 2013 in two research plots (control and managed) at KMSP in relation to a reference forest to evaluate

<Abstract>

management effects and restoration success. Total basal area continued to increase in both the plots, but diameter-growth decreased in the control plot, whereas it continued to increase in the managed plot, which had been thinned by volunteers. In the control plot, which was planted at higher initial density than the managed plot, *Quercus phillyraeoides* (evergreen, mid-canopy tree) dominated the single-layered canopy and vertical development was delayed. In the managed plot, *Quercus serrata* (deciduous, canopy tree) dominated the upper canopy layer and evergreen broadleaved trees dominated the mid- to lower-canopy layers, resulting in a vertically well-developed canopy similar to the reference forest. The basal area of *Robinia pseudoacacia* decreased due to shading by evergreen trees, whereas that of *Nerium oleander*, an exotic species, had increased in the control plot. Ordination results indicated that vegetation of the control plot was diverging away from the reference forest, whereas thinning had directed the managed plot toward it. My results confirm that simultaneously planting seedlings of native species does not lead to natural forest stand structure. In the future, adaptive management, such as periodic thinning, removal of shade-tolerant, exotic species and enrichment planting of native species, will be needed to integrate forest restoration sites with the surrounding mid-successional, secondary forest.

To achieve natural forest restoration goals, I must evaluate objectively community development and ecosystem integrity. Here, I used non-metric multidimensional scaling (nMDS) ordination to evaluate a forest restoration project in Japan. The restoration objective was to achieve ecological continuity between restored forest and remnant, secondary forest. Assessment of photographs indicated visual continuity of the canopy was achieved 20 years after planting. However, nMDS ordination indicated species composition of the restored forest had changed very little over 22 years and was markedly different from

<Abstract>

that of the remnant forest. Stand structure of the restored forest was approaching, but diverging away from that of the remnant forest. I also compared community structure of the restoration site with two reference forests: natural secondary forest and late-seral forest, located nearby. Current species composition and stand structure of the remnant forest was outside the range of variability observed for natural secondary forest. This was attributed to lack of seed sources near the remnant forest, which likely led to an unnatural successional pathway. In contrast, chronological change in community structure of natural secondary forest was predictable, approaching that of late-seral forest. My results suggest that initial expectations for vegetation change in the restored forest have not been achieved and management is required to change the direction of forest succession. Furthermore, designation of an isolated remnant forest as the targeted community may have been inappropriate. When designating existing communities as restoration goals, I should recognize that, with forest succession, targeted communities will change and account for spatial and temporal variation in community structure. nMDS ordination is a powerful tool for evaluating both variation and chronological change in restored and target community structures.

## Contents

## Chapter 1 Introduction and background

Japan is a mountainous country and the main objective of hillslope re-vegetation is to increase plant cover quickly to stabilize the soil and prevent erosion (Yoshida 2007; Zuazo and Pleguezuelo 2008). Recently, restoration of the native vegetation has also become an important objective of re-vegetation projects. Regionally-grown seedlings and juvenile trees are planted in an effort to restore the native forest and integrate the restoration site with the regional landscape (e.g., Miyawaki 2004; Morimoto et al. 2006; Yamagawa et al. 2010; Yoshida 2007). It is uncertain, however, whether current re-vegetation techniques will produce mature, native forests (Oldfield et al. 2013). Most studies reporting results of natural forest restoration projects are within 5 years after planting (Oldfield et al. 2013). Due to logistic and budgetary constraints, subsequent adaptive management to direct succession of restored sites toward native forest composition and structure is rarely conducted (but see, MacKay et al. 2011; Nakamura et al. 2005). Thus, there is a need to accumulate case studies to compare and evaluate the effectiveness of various restoration techniques and to assess the need for adaptive management (Ruiz-Jaén and Aide 2006). In urban areas of Japan, mid-successional, secondary forests have high biodiversity (Hirayama et al. 2011). Therefore, restoration of secondary forests could contribute to maintaining biological diversity in human-disturbed landscapes. In many natural forest restoration projects, the neighboring secondary forest is often chosen as the target vegetation. However, specific methodology for restoring secondary forests in Japan is not yet established.

In chapter 2, I compared the stand structure and tree species composition between a restored stand (a disturbed hillslope planted in 2002) and the neighboring secondary forest, which was designated as the target vegetation, in suburban Kobe City. The objective of the restoration project was to integrate the disturbed hillslope with the neighboring secondary forest, which is abandoned satoyama (intensively managed, semi-natural secondary forests of rural Japan). To evaluate the success of the restoration project, I quantified how closely the species composition and stand structure of the restoration site resembled that of the target forest.

Recently, natural forest restoration has also become an important objective to realize integration of human-disturbed slopes with the regional landscape. Various methods have been proposed that use native seedlings and juvenile trees, which are planted simultaneously in an effort to restore natural forest vegetation (e.g., Miyawaki, 2004; Morimoto et al., 2006; Yamagawa et al., 2010; Yoshida, 2007). It is uncertain, however, whether such efforts will actually produce mature, native forests (Oldfield et al., 2013). Most studies monitoring the development of natural forest restoration projects report progress within ≤5 years after planting (Oldfield et al., 2013, their Table 1). In addition, due to logistic and budgetary constraints, most reforested sites remain unmanaged and subsequent adaptive management to direct succession toward native forest composition and structure is rarely conducted (but see, MacKay et al., 2011). Thus, there is a need to accumulate case studies to compare and evaluate the long-term outcome of various restoration projects and to assess the need for adaptive management (Ruiz-Jaén and Aide, 2006).

I am aware of only one study of forest restoration in Japan spanning more than 30 years in Expo Park in Osaka, where seedlings of late-successional species were planted in the 1970s to restore the native, warm-temperate evergreen-broadleaved forest. Thirty years later, the stand had become dense with a single-layered canopy dominated by late-successional, evergreen trees, and lower-canopy and understory species composition was poor (Nakamura et al., 2005). Thinning and gap-formation experiments were conducted to increase seedling regeneration in the understory and create vertically well-developed canopies that more closely resemble the structure and function of natural forests. The results of the EXPO Park reforestation project suggest that simultaneously planting seedlings of late-successional species does not lead to stand structures resembling natural forests and that adaptive management may be needed to direct succession of forest restoration sites toward natural forest composition and structure (Rebele and Lehman, 2002; Robinson and Handel, 2000; Tojima et al., 2004).

Recently, multiple criteria, including species diversity, vegetation structure, and ecosystem processes are used to evaluate restoration success (Ruiz-Jaén and Aide, 2005a). Vegetation structure is an important measure for predicting the future direction of succession after forest restoration, because structural complexity facilitates colonization by plant species other than those that were planted (Ruiz-Jaén and Aide, 2006). Vegetation structure is also correlated with diversity of other organisms (insects, birds, amphibians, etc.) and various ecosystem processes, such as seed dispersal and nutrient availability (Gamfeldt et al., 2013; Ishii et al.,2004; Ruiz-Jaén and Aide, 2005b). When promoting colonization after forest restoration, however, care must also be taken to prevent invasion of restoration sites by exotic species and escaped ornamentals (Ishii and Iwasaki, 2008; Sullivan et al., 2009).

In chapter 3, I report 21 years (1992–2013) of vegetation dynamics in a 33-year-old urban forest restoration site in Kobe Sports Park, Kobe City, Japan (hereafter KSP). This site is different from EXPO Park because part of the forest was intensively managed by civilian volunteers in an effort to recreate the structure and function of semi-natural secondary forests of rural Japan (satoyama), which was the dominant landscape before construction of KSP. In recent years, agricultural use of satoyama has ceased and many abandoned secondary forests are maturing and succeeding toward late-successional evergreen broadleaved forest (Azuma et al., 2014; Takeuchi et al., 2003; Yokohari and Amati, 2005) and forests surrounding KSP have similar land-use history. When the surrounding vegetation is in mid-succession, integration of forest restoration sites with the regional landscape becomes more difficult, because the restoration effort must pursue a “moving target” (Hotta and Ishii, 2015). At KSP, I analyzed the long-term vegetation dynamics following forest restoration using data from three vegetation surveys spanning 21 years. I compare composition and structure between control (unmanaged) and managed plots relative to a reference, semi-natural secondary forest to evaluate the success of adaptive management at this forest restoration site.

In Japan, urbanization often involves excavation of hillslopes and deforestation. As a result, developed land in urban areas of Japan is usually neighbored by artificially created, deforested hillslopes. An important objective of reforestation of such artificial hillslopes has been to prevent erosion and landslides (Nitta,1979, Iizuka and Kondo, 2010). In addition, because the topsoil is removed, many artificial hillslopes are not suitable for planting. Therefore, the main focus of conventional reforestation studies has been on soil improvement and plant establishment on degraded soils (Yamadera, 1986). In this regard, restoration success was evaluated using measures of vegetation amount, such as basal area growth or growth/survival rate of the planted trees. Recently, however, restoration of natural ecosystems has become an important objective to achieve integration of human-disturbed hillslopes with the native landscape (Matsuda et al., 2005). To evaluate whether natural forest restoration goals have been met, I must develop objective methods for evaluating ecosystem quality, such as degree of community development and ecosystem integrity.

Furthermore, in many restoration projects, the desired target vegetation is often unclear. In Japan, urban forests near human-inhabited areas are generally secondary forest with a long history of human use for obtaining firewood and other traditional forest resources. These forests were maintained at early-seral stages of secondary succession and are known as "satoyama". Satoyama is the traditional forest vegetation of Japan, which has constantly experienced intermediate disturbance, and has high biodiversity (Katoh et al. 2009; Morimoto 2011). There is social movement in Japan toward protection and conservation of the traditional satoyama landscape (Abe et al., 2004; Takeuchi 2010). Some researchers argue that satoyama should be the target community for urban forest restoration in Japan. Satoyama ecosystems, however, cannot be maintained without human management. Recently, many satoyama forests remain unmanaged due to decreasing demand for traditional forest resources (Fukamachi et al. 2001). Therefore, if satoyama is the desired goal for a restoration project, I must take into account that structure of the targeted plant community will change with time. This means that I must periodically monitor, not only whether the restored forest is succeeding toward the targeted community structure, but also how vegetation of the targeted community is changing as a result of natural succession (Hotta and Ishii, 2015). In secondary forest restoration projects, I must pursue a "moving target".

In chapter 4, I compared vegetation change over a 35-year-period in a natural forest restoration site in Kobe City, southwestern Japan. The restoration objective was to achieve ecological continuity with the remnant secondary forest (targeted community), which was unmanaged satoyama. My research objective was to develop an objective method for assessing whether initial restoration goals were achieved and to evaluate whether designation of the targeted community was appropriate. I employ a quantitative method for visualizing the variability and chronological change in community structure of the restoration site relative to the targeted community and a reference, natural forest community.

# Chapter 2: Evaluation of a natural forest restoration project in Kobe City, Japan ten years after planting

## Introduction

In this chapter, I compared the stand structure and tree species composition between a restored stand (a disturbed hillslope planted in 2002) and the neighboring secondary forest, which was designated as the target vegetation, in suburban Kobe City. The objective of the restoration project was to integrate the disturbed hillslope with the neighboring secondary forest, which is abandoned satoyama (intensively managed, semi-natural secondary forests of rural Japan). To evaluate the success of the restoration project, I quantified how closely the species composition and stand structure of the restoration site resembled that of the target forest.

## Materials and methods

1. Study site

The study was conducted in “ki-na no mori” Park, a forest park due to open in 2016 in northern Kobe City, Hyogo Prefecture in southwestern Japan (34°43’N, 135°05’E, 235 m ASL, Fig. 2-1). The study area is on a hillslope, which was excavated to obtain landfill material for the construction of Kobe Airport. In 2002−2003, regionally-grown seedlings of native species were planted on the hillslope at 15,000−20,000 trees ha$^{-1}$ densities in order to restore the natural vegetation. The restored area was divided into six main restoration zones. While some zones (e.g. along the road and trails) were designed as aesthetic areas mainly comprising flowering trees, in most zones the objective was restoration of natural secondary forest vegetation. Species planted in the natural forest restoration zone

(*Quercus serrata*, *Quercus accutissima*, *Pinus densiflora*, *Pourthiaea villosa*, *Cornus kousa*, *Ilex pedunculosa*, *Ligustrum obtusifolium*, *Vibrunum* spp, *Rhododendron* spp.) were chosen based on a vegetation survey conducted in the neighboring secondary forest, which is semi-natural secondary forest (satoyama), abandoned for ca. 40 years after being intensively managed for fire wood and organic soil to be used for agriculture.

Plot sizes for the original vegetation survey of the secondary forest and planting records of the restoration site were 10x10m. The original vegetation survey of the secondary forest was conducted in a single 10x10m plot. Using mapped records, I located and reestablished this plot. I also established one additional plot adjacent to it (hereafter: target plots, Fig. 2-2) to balance the number of plots with the restoration site. In the restoration site, I reestablished two of the original 10x10m plots in the natural forest restoration zone (hereafter: restored plots).

In 2012, I measured diameter at breast height (DBH, 1.3 m above ground level) and height of all woody plants taller than 1.3 m in the plots. I also obtained data on the original species composition of the target plot in 2002. For the restored plots, the original data included number and species of the planted seedlings, but no information was available on their sizes.

2. Data analysis

Because my plot replication was only two per site, statistical tests could not be performed. To evaluate restoration success, I pooled data from the two plots at each site and compared species composition between the restored and target plots using Chao's index of compositional similarity based on relative abundance (Chao et al. 2004):

$$J'_A = \frac{U_1\ U_2}{U_1\ + U_2 - U_1\ U_2}$$

Where, U1 and U2 are the relative abundance of common species in plot 1 and plot 2, respectively. I also calculated $J'_B$ based on relative basal area to assess similarity of stand structure (hereafter: structural similarity) between the two sites. In a study that analyzed compositional similarity among various forest types, Chao et al. (2008) calculated standard error (s.e.) estimates for J' using bootstrap resampling. Their maximum s.e. was 0.07. I used this as my conservative criterion for inferring statistically significant differences in $J'_A$ and $J'_B$.

**Results**

The initial planting density in the restored plots in 2002 was about 10% higher than the stand density in the original (2002) target plot (Table 2-1). After 10 years, stand density in the restored plots decreased, but remained higher than that of the target plots. At the time of planting, *R. reticulatum*, *Q. serrata*, *Cerasus jamasakura*, *Q. acutissima*, *Vibrunum wrightii*, and *P. densiflora*, *Cornus kousa*, and *Pourthiaea villosa* were the most abundant species. Of these, *Q. serrata* and *R. reticulatum* were also abundant in the original target plot and *P. densiflora* dominated the basal area. The other planted species, however, were either not found or occurred only in small numbers in the original target plot. The second most abundant species in the original target plot, *Clethra barbinervis*, was not planted in the restored plots. In 2012, *P. densiflora*, *Q. serrata*, and *Q. acutissima* (all planted) dominated the basal area of the restored plots. *R. reticulatum* had decreased markedly in abundance and *Callicarpa japonica* had become the most abundant shrub. Some new species, such as *Vibrunum dilatatum* and *Clethra barbinervis* had recruited into the restored plots. In the target plots, basal area of *P. densiflora* had decreased markedly and *Q. serrata* had become the dominant canopy tree. Abundance and basal area of evergreen species, namely *Eurya japonica* and *Ilex pedunculosa*, had increased.

As a result of these changes, species composition of the restored plots had become more similar to that of the original (2002) target plot in ten years (Table 2-2). This was reflected by the increase in $J'_A$ of the restored plots relative to the original target plot ($J'_A$ = 0.42 in 2002 to 0.51 in 2012). However, because the species composition of the target plot also changed, composition similarity between the restored and target plots in 2012 ($J'_A$ = 0.48) had not increased from that in 2002 ($J'_A$ = 0.42). In contrast, stand structure of the current (2012) restored plots was less similar to the current (2012) target plots ($J'_B$ = 0.61) than to the original target plot ($J'_B$ =0.73).

Maximum tree height in the restored plots was 6.7 m, whereas that of the target plots was 12.2 m. The canopy of the current (2012) restored plots was not stratified comprising suppressed individuals of canopy and mid-canopy species along with shrub species in the lowest canopy layer (Fig. 2-3). In the original (2002) target plot, there were numerous shrubs and saplings of mid-canopy trees in below 3.3 m. In 2012, *Q. serrata* and *P. densiflora* dominated the upper canopy and both evergreen and deciduous species were distributed across mid-canopy and shrub layers, resulting in vertical stratification.

**Discussion**

My results indicated that, 10 years after planting, stand density of the restored plots was in between that of the original (2002) and current (2012) target plots. Thus, in terms of quantity (vegetation cover), forest restoration was successful. In addition, compositional similarity of the restored plots relative to the original target plot had increased, indicating that the initial goal of secondary forest restoration was successful. However, because the vegetation of the target forest had changed, compositional similarity relative to the current target plots had not increased in 10 years. Furthermore, the stand structure of the restored plots did not resemble that of the target plots. Namely, canopy stratification observed in the target plots had not developed. Simultaneous planting at high density in the restored plots may have resulted in intense competition among seedlings as inferred from the presence of many suppressed individuals of canopy and mid-canopy species occurring with shrubs in the lower-canopy layer.

Normally, it takes more than 20 years for abandoned fields and bare lots in urban areas to reach the successional stage of woody dominance (e.g., Bornkamm 2007; e.g., Millard 2000; Prach et al. 2001). The aim of planting native seedlings and juvenile trees is to speed up this process (Miyawaki 2004; Yoshida 2007). Simultaneous planting, however, creates artificial cohorts, which are rarely observed in natural forests where, except after large-scale disturbances, the population structure tends to be multi-aged (e.g., Kominami et al. 2003). Furthermore, when the target vegetation is mid-successional forest, such as secondary forest, I can expect that the composition and structure of the target forest will continuously change as a result of natural disturbances and succession (Hiura 2001; Masaki et al. 1999). In warm-temperate regions of Japan, abandoned satoyama are succeeding toward late-successional evergreen broadleaved forest (Hirayama et al. 2011; Yokohari and Amati 2005) (Hirayama et al. 2011; Yokohari and Amati 2005). In addition, the recent spread of pine wilt disease has greatly altered the composition and structure of secondary forests in Japan (Itô et al. 2010; Mota and Vieira 2008). The target forest in this study was also in the process of successional recovery after pine wilt disease. This is reflected in the high mortality rate of *P. densiflora* and recruitment of shrubs and saplings of canopy trees in the lower canopy layer of the target plots, contributing to further development of canopy stratification by natural regeneration, which did not occur in the restored plots due to high stand density. I inferred that the decrease in structural similarity between the restored and target plots is the cumulative result of such differences in successional processes.

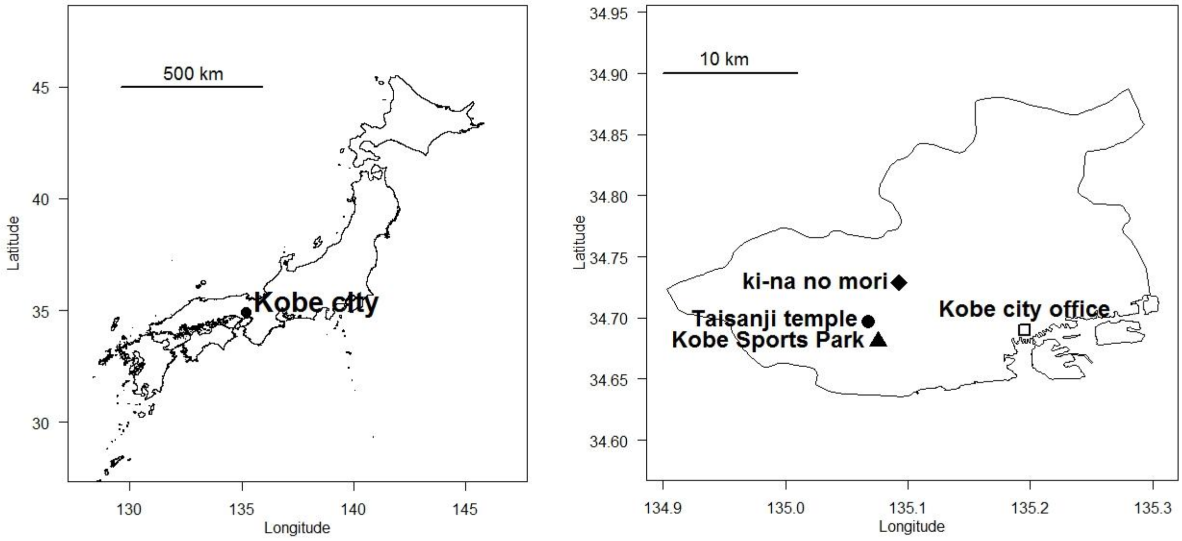



Fig. 2-1: The location of study sites. Kobe City is located in southwestern Japan. (maps drawn using shapefiles by http://www.gadm.org/country and R x64 3.3.1)

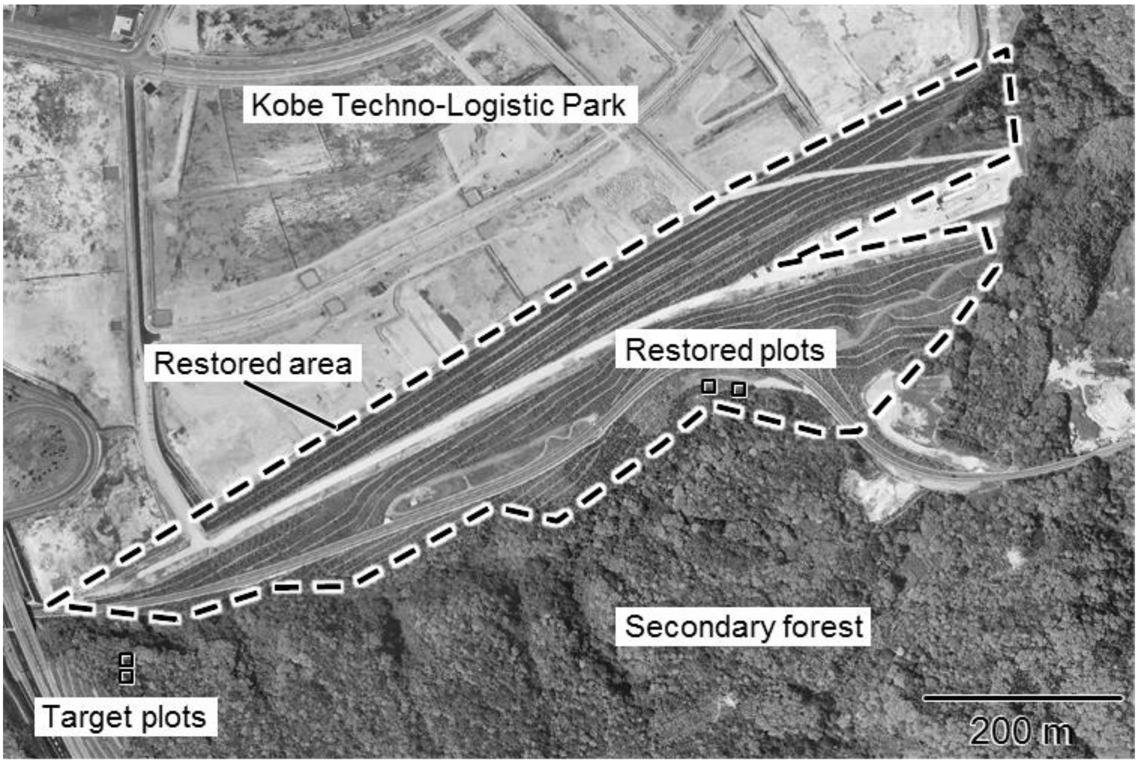



Fig. 2-2. Aerial photo taken in 2003 of the study site and research plots in Kobe City, Japan.

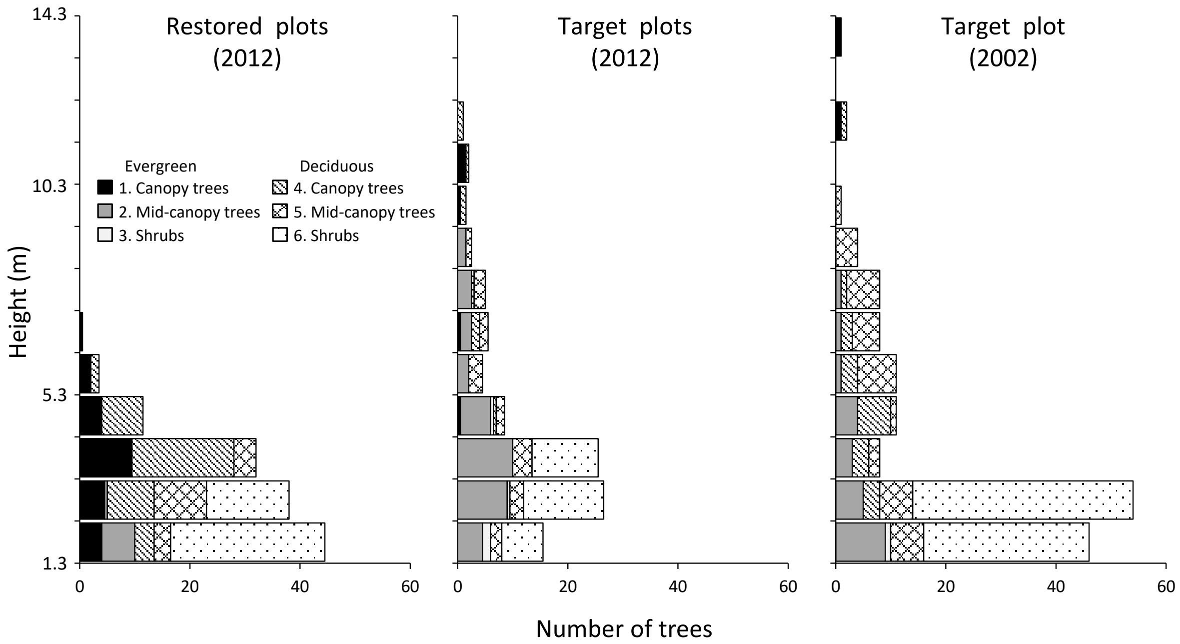



Fig. 2-3. Frequency distributions of tree heights of the restored and target plots in Kobe City, Japan. Pooled data from the two plots are shown for 2012.

Table 2-1. Species composition in the restored and target plots in Kobe City, Japan.

| | Restored plots | | | | Target plots | | | |
|---|---|---|---|---|---|---|---|---|
| | No. of trees (/ha) | | BA($m^2$/ha) | | No. of trees (/ha) | | BA($m^2$/ha) | |
| Species | 2002 | 2012 | 2002 | 2012 | 2002 | 2012 | 2002 | 2012 |
| *Rhododendron reticulatum* [6] | 2550.8 | 750.0 | - | 0.03 | 5800.0 | 3200.0 | 1.0 | 0.7 |
| *Quercus serrata* [4] | 2053.4 | 1950.0 | - | 1.4 | 1000.0 | 300.0 | 8.5 | 9.1 |
| *Cerasus jamasakura* [4] | 1604.2 | 200.0 | - | 0.04 | - | - | - | - |
| *Quercus acutissima* [4] | 1216.0 | 1400.0 | - | 2.0 | - | - | - | - |
| *Viburnum wrightii* [6] | 1179.6 | 250.0 | - | 0.02 | - | - | - | - |
| *Pinus densiflora* [1] | 1149.6 | 2450.0 | - | 5.9 | 200.0 | 150.0 | 22.2 | 5.7 |
| *Cornus kousa* [5] | 1012.1 | 800.0 | - | 0.2 | 100.0 | - | 0.01 | - |
| *Pourthiaea villosa* [5] | 1000.0 | 700.0 | - | 0.1 | 200.0 | 200.0 | 0.5 | 0.04 |
| *Quercus variabilis* [4] | 640.7 | 400.0 | - | 0.4 | - | - | - | - |
| *Callicarpa japonica* [6] | 640.7 | 1150.0 | - | 0.1 | - | - | - | - |
| *Eurya japonica* [2] | 359.3 | 250.0 | - | 0.01 | 1100.0 | 2600.0 | 0.2 | 1.8 |
| *Ilex pedunculosa* [2] | 215.6 | 200.0 | - | 0.02 | 1100.0 | 1100.0 | 2.0 | 4.0 |
| *Clethra barbinervis* [5] | - | 50.0 | - | 0.002 | 2700.0 | 1050.0 | 16.0 | 4.2 |
| *Abelia spathulata* [6] | - | - | - | - | 1000.0 | 150.0 | 0.1 | 0.01 |
| *Fraxinus sieboldiana* [4] | - | - | - | - | 900.0 | - | 0.9 | - |
| *Lyonia ovalifolia* [5] | - | - | - | - | 500.0 | 400.0 | 0.1 | 0.5 |
| *Viburnum dilatatum* [6] | - | 850.0 | - | 0.1 | - | - | - | - |
| *Morella rubra* [1] | - | - | - | - | - | 100.0 | - | 5.4 |
| Others | 3174.3 | 1650.0 | - | 0.1 | 800.0 | 550.0 | 0.2 | 0.7 |
| Total | 16796.3 | 13050.0 | - | 10.5 | 15400.0 | 9800.0 | 51.8 | 32.2 |

Species are listed in order of initial planting density (2002) in the restored plot and then by density in the original target plot (2002).

Numbers indicate leaf habit and canopy height at maturity of each species (1 evergreen canopy tree, 2 evergreen mid-canopy tree, 3 evergreen shrub, 4 deciduous canopy tree, 5 deciduous mid-canopy tree, 6 deciduous shrub).

Table 2-2. Chao's index of composition similarity based on relative abundance and basal area ($J'_A$ and $J'_B$, respectively) between the restored and target plots in a natural forest restoration site in Kobe City, Japan.

| $J'_b$ \ $J'_a$ | | 2002 | | 2012 | |
|---|---|---|---|---|---|
| | | Restored plots | Target Plot | Restored Plots | Target Plots |
| 2002 | Restored Plots | | 0.42 | 0.91 | 0.42 |
| | Target Plot | - | | 0.51 | 0.85 |
| 2012 | Restored Plots | - | 0.73 | | 0.48 |
| | Target Plots | - | 0.80 | 0.61 | |

Where there were two plots, data were pooled to calculate $J'_A$ and $J'_B$.

## Chapter 3: Twenty-one years of stand dynamics in a 33-year-old urban forest restoration site at Kobe Municipal Sports Park, Japan

### Introduction

In this chapter, I report 21 years (1992–2013) of vegetation dynamics in a 33-year-old urban forest restoration site in Kobe Municipal Sports Park, Kobe City, Japan (hereafter KMSP). This site is different from EXPO Park because part of the forest was intensively managed by civilian volunteers in an effort to recreate the structure and function of semi-natural secondary forests of rural Japan (satoyama), which was the dominant landscape before construction of KMSP. In recent years, agricultural use of satoyama has ceased and many abandoned secondary forests are maturing and succeeding toward late-successional evergreen broadleaved forest (Azuma et al. 2014; Takeuchi et al. 2003; Yokohari and Amati 2005) and forests surrounding KMSP have similar land-use history. When the surrounding vegetation is in mid-succession, integration of forest restoration sites with the regional landscape becomes more difficult, because the restoration effort must pursue a "moving target" (Hotta and Ishii 2015). At KMSP, I analyzed the long-term vegetation dynamics following forest restoration using data from three vegetation surveys spanning 21 years. I compare composition and structure between control (unmanaged) and managed plots relative to a reference, semi-natural secondary forest to evaluate the success of adaptive management at this forest restoration site.

## Materials and methods

1. Study site

KMSP is located in Suma Ward, Kobe City, Hyogo Prefecture in southwestern Japan (34.68◦N, 135.07◦E, 100 m ASL, Fig. 2-1). The inherent natural vegetation of this region is warm-temperate ever-green broadleaved forest (Miyawaki et al. 1984). Before KMSP was established, the area was semi-natural secondary forest (satoyama), abandoned for ca. 30 years after being intensively managed for firewood and organic soil to be used for agriculture. In 1980, after construction of KMSP, man-made cut slopes within the park were re-vegetated after adding more than 30 cm of topsoil. To restore the semi-natural secondary forest, saplings of native tree species, such as *Quercus serrata* Murray (deciduous), *Quercus glauca* Thunb., and *Quercus phillyraeoides* A. Gray (evergreen), were planted (Yoshidaet al. 2002). In 1992, two research plots, control (unmanaged) and managed, were established on a north-facing slope (mean inclination = 25.0◦) to investigate vegetation change after restoration (Fig. 3-1; Doi et al. 1993). The diameter at breast height (DBH, 1.3 m above ground level) and height of all woody plants taller than 1.3 m was measured in both the plots. Quantitative data on the number and species of trees planted in the plots were not available. Based on the total number of live and dead trees in the plots in 1992, Doi et al. (1993) estimated that the initial planting densities were ca. 0.64 and 0.22trees per m2in the control and managed plots, respectively. In 2002, a second survey was conducted (Yoshida et al. 2002), followed by a third survey in 2013. I interviewed former volunteers and park employees and found that the understory of the study area including the managed plot was thinned during 2006–2007 and that *Nerium oleander* L., an exotic species, was removed from the area between the managed and control plots in 2010. Quantitative data of the management practices, however, were not recorded.

2. Data analysis

Because my plots were not replicated, statistical tests could not be conducted. To evaluate restoration success, I compared the species composition of the restoration site with that of the reference forest using subjective Bray Curtis ordination analysis (Beals 1984) in R (vegan package, ver. 2.14.1, R Development Core Team). In this method, vegetation data of the restoration site are plotted in relation to selected reference points. The position and distance of the restoration site relative to the reference vegetation and the direction of change over time can be interpreted as restoration success (Ruiz-Jaén and Aide 2006). As the reference forest, I selected the mature semi-natural secondary forest at Taisanji Temple (34.68 N, 135.07◦E, 70–200 m ASL, Fig. 3-1), 2 km from the study site, where a vegetation survey was conducted in six 10 × 20 m plots2008 (Azuma et al. 2014). The land-use history of the reference forest is similar to that of the forests that existed before construction of KMSP, i.e., a secondary forest that was intensively used as satoyama until the 1950s, then abandoned (Iwasaki and Ishii 2005). Thus, this forest could be considered as the reference vegetation that was intended for the forest restoration project at KMSP.

## Results

Stand density decreased from 1992 to 2000 in both the plots (Fig. 3-2a). Between 2000 and 2013, stand density increased by 21%in the control plot, whereas in the managed plot, stand density decreased due to thinning. As a result, in 2013, stand density of the control plot (0.77 trees m$^{-2}$) was twice that of the managed plot (0.34 trees m$^{-2}$), which was similar to the reference forest (0.35 ± 0.04 trees m$^{-2}$). Total basal area was similar between the two plots in 1992 and 2000 (Fig. 3-2b). In 2013, total basal area of the control plot (36.38

$m^2ha^{-1}$) was greater than in the managed plot (27.53 $m^2ha^{-1}$), which was similar to the reference forest(22.45 ± 5.37 $m^2ha^{-1}$).In both the plots, DBH growth rate for 2000–2013 was lower than for 1992–2000, but the decline in DBH growth rate was more marked for the control plot than for the managed plot (Fig. 3-2c).A similar pattern was observed for height growth rate (Fig. 3-2d).Although mean DBH and tree height were similar between the plots($t = 1.13$, $P = 0.13$ and $t = 1.25$, $P = 0.11$, respectively) in 2013, maxi-mum DBH in the control plot (19.2 cm, *Quercus acutissima* Caruth)was nearly half that in the managed plot (34.3 cm, *Q. serrata*). Mean DBH of the reference forest in 2008 fell upon the growth trajectory of the managed plot. In the control plot, the relative basal area of upper-canopy trees (namely *Castanopsis sieboldii* Makino) decreased, while that of the mid-canopy tree, *Q. phillyraeoides*, increased during the study period (Fig. 3-3). In 2013, evergreen trees contributed 77% of the basal area of the control plot. In the managed plot, the relative basal area of the dominant species remained fairly constant during the study period, with the exception of *Q. serrata*, whose relative basal area increased from 16% in 1992 to 29% in 2013, which was similar to the value observed in the reference forest (27%). In 2013 evergreen trees contributed 61% of the basal area in the managed plot, which was similar to the value observed in the reference forest (60%).In the control plot, basal area of *Robinia pseudoacacia* and *N. oleander* were 0.80 and 0.93 $m^2ha^{-1}$(7 and 6%, Fig. 3-3), respectively in1992. Basal area of *R. pseudoacacia* had decreased to 0.08 $m^2ha^{-1}$ in2000 and to 0.003 $m^2ha^{-1}$ by 2013, whereas that of *N. oleander* had increased to 1.61 and 1.94 $m^2ha^{-1}$in 2000 and 2013, respectively. In the managed plot, basal area of *R. pseudoacacia* was 0.23 $m^2$ in1992. This had decreased to 0.02 $m^2ha^{-1}$ in 2000 and *R. pseudoacacia* did not exist in the managed plot by 2013. *N. oleander* was not found in the managed plot during the study period. Ordination results indicated that species composition of the control and managed plots were different from each other as well as from the reference forest (Fig. 3-4). From 1992 to 2000, species composition of the control plot approached that of the reference forest. From 2000 to 2013, however, it diverged away, while that of the managed plot was directed toward the reference forest. In 1992, frequency distributions of tree heights were unimodal in both the plots (Fig. 3-5). In 2000, there was some development of bimodality in the control plot, but *Q. phillyraeoides* and *Cleyera japonica* Tunb. dominated both the upper and lower canopy layers. In contrast, there was a clear distinction in structure and composition between upper-canopy (8.3–13.3 m) and lower-canopy (1.3–7.3 m) trees in the managed plot. In 2013, there was some recruitment of the shrub layer (1.3–2.3 m) in the control plot, but *Q. phillyraeoides* dominated the mid-canopy layer (4.3–8.3 m) and also occurred in all canopy layers below 10.3 m. In contrast, the canopy of the managed plot was vertically well-developed comprising shrub (<2.3 m), mid-canopy (2.3–7.3 m) and upper-canopy (8.3–15.3 m) layers, each comprising different species.

## Discussion

My results indicated that 33 years after planting, the forest restoration site had recovered to basal area similar to that of the reference forest, as well as other semi-natural secondary forests of this region (e.g., Hirayama et al. 2011; Itoh et al. 2010). Thus, in terms of quantity (vegetation cover), forest restoration was successful. Stand density in the control plot, however, was more than twice that of the managed plot and reference forest. This was mainly due to the higher initial planting density in the control plot (Doi et al.1993), indicating that the control plot needs to be thinned to create stand structure similar to natural forest.

I also found that growth rates were declining in the control plot and that the vegetation structure did not resemble that of the reference forest. Decreasing DBH growth in the control plot suggests that high initial planting density resulted in more intense competition and increased allocation to height growth relative to diameter growth of the planted trees. This is reflected in the high initial height growth rate observed in the control plot. Dominance by *Q. phillyraeoides* prevented vertical development of the canopy and understory establishment in the control plot, which could delay succession by preventing colonization by late-successional species. Although *Q. phillyraeoides* is a common species in mid- to late-successional forests of this region, when planted and grown at high densities such as in the control plot, it may delay development of the lower-canopy and understory layers by decreasing light availability. In the managed plot, thinning prevented dominance of evergreen species resulting in a vertically well-developed canopysimilar to that of the reference forest.

*R. pseudoacacia* is considered an invasive species in Japan (Junget al. 2009), as well as in Europe (Castro-Diaz et al. 2009) and parts of North America (Rice et al. 2004). It is a light-demanding, early successional species (Boring and Swank 1984) and I found that, after invading the restoration site, it was gradually shaded out as dominance of evergreen trees increased. In contrast, *N. oleander*continued to increase in the control plot, suggesting that active management may be needed to control this species.

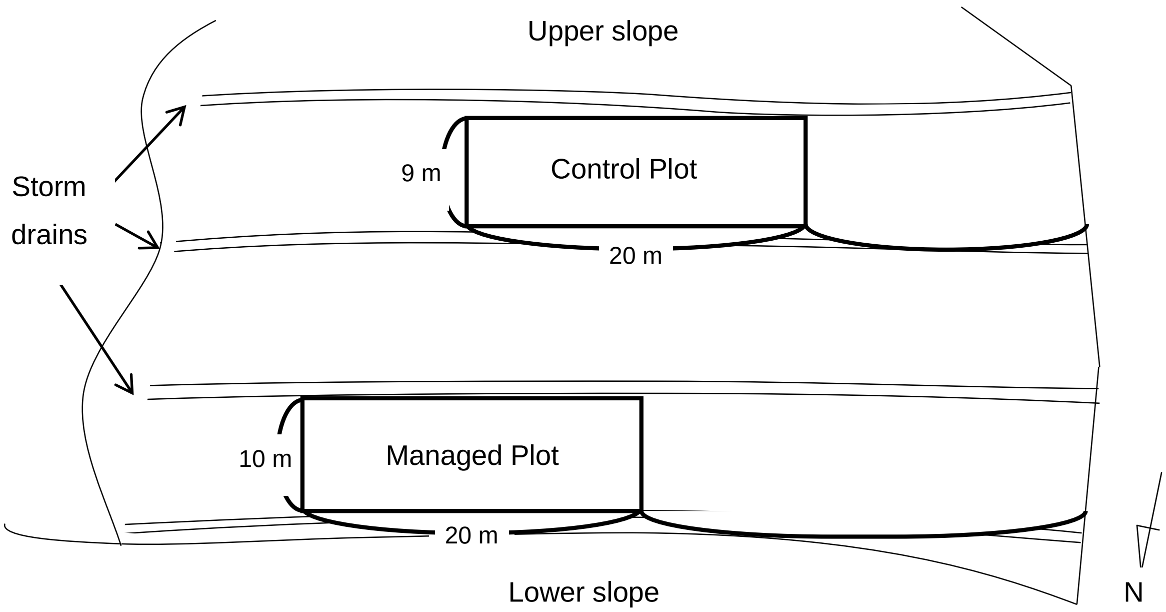



Fig. 3-1: Layout of the control and managed plots in the restoration site。.

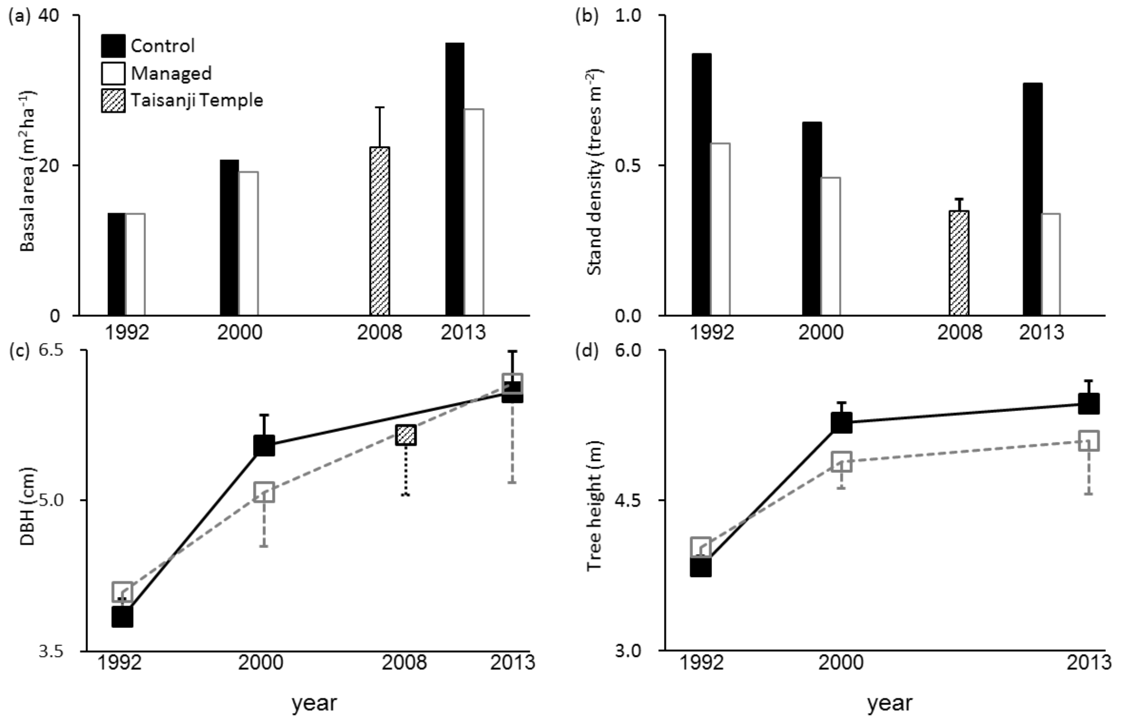



Fig. 3-2. Stand density, basal area, diameter at breast height (DBH), and height of trees in the control and managed plots in the restoration site and the reference forest. Basal area and stand density of the reference forest is the mean of six plots. Error bars indicate one standard deviation.

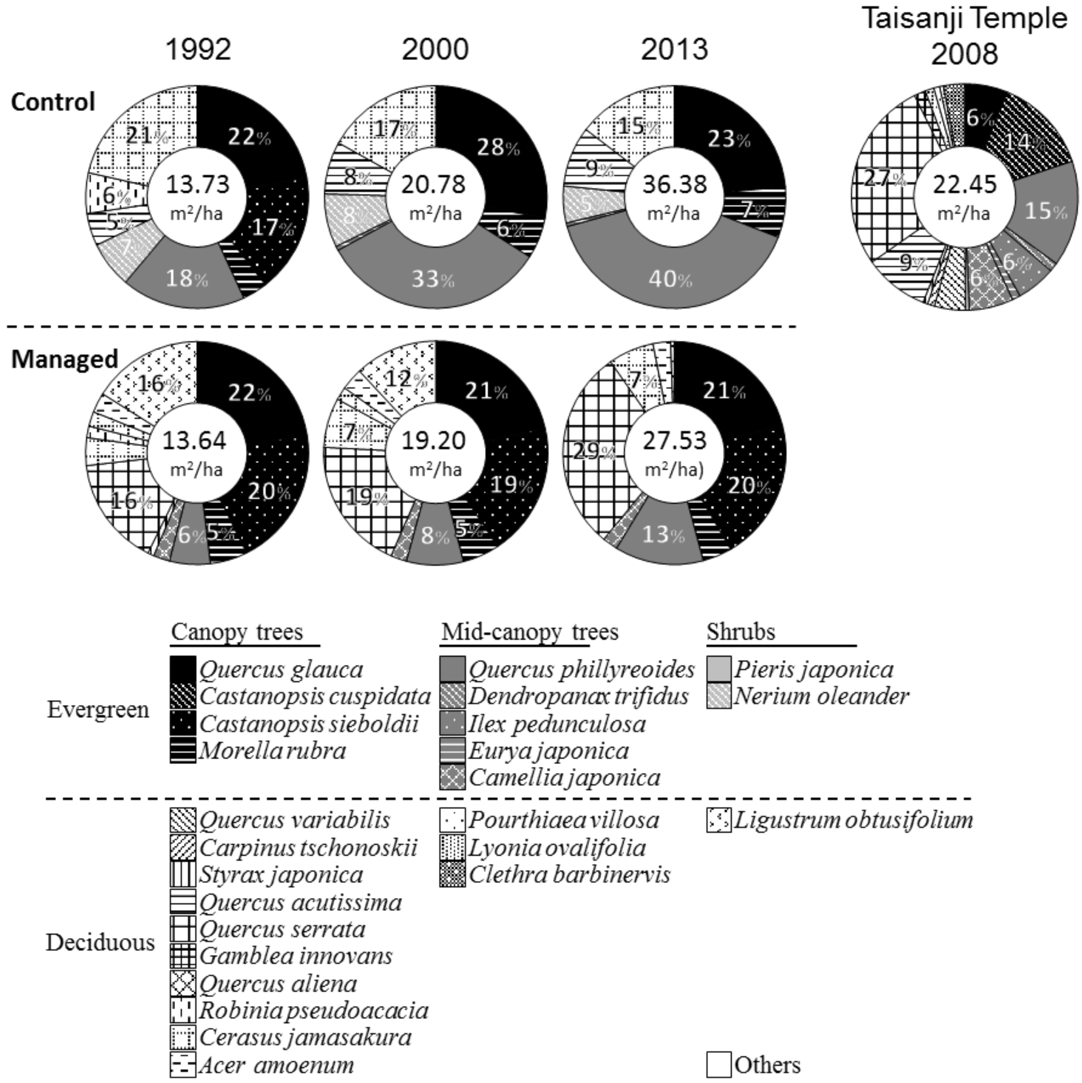



Fig. 3-3. Species composition according to relative basal area (%) in the control and managed plots in the restoration site and the reference forest.

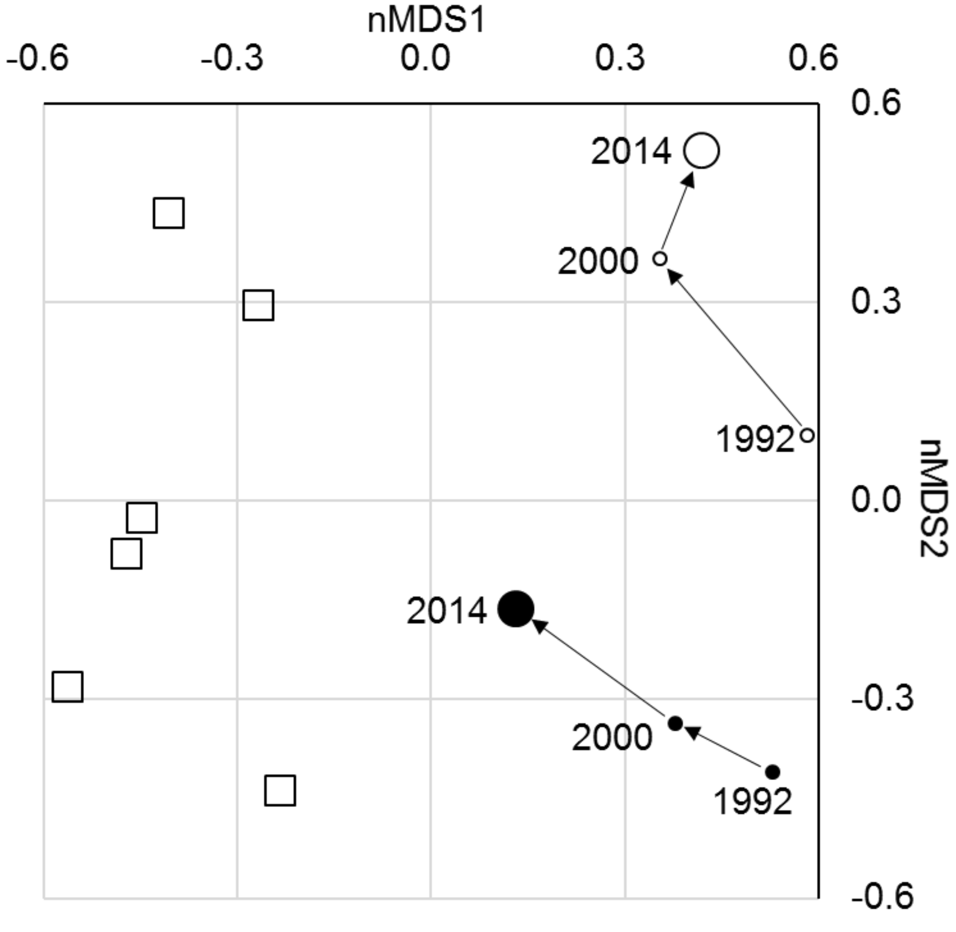



Fig. 3-4. Ordination results showing temporal change in species composition of the control (filled circles) and managed (open circles) plots in the restoration site along with the six plots established in the reference forest (open squares).

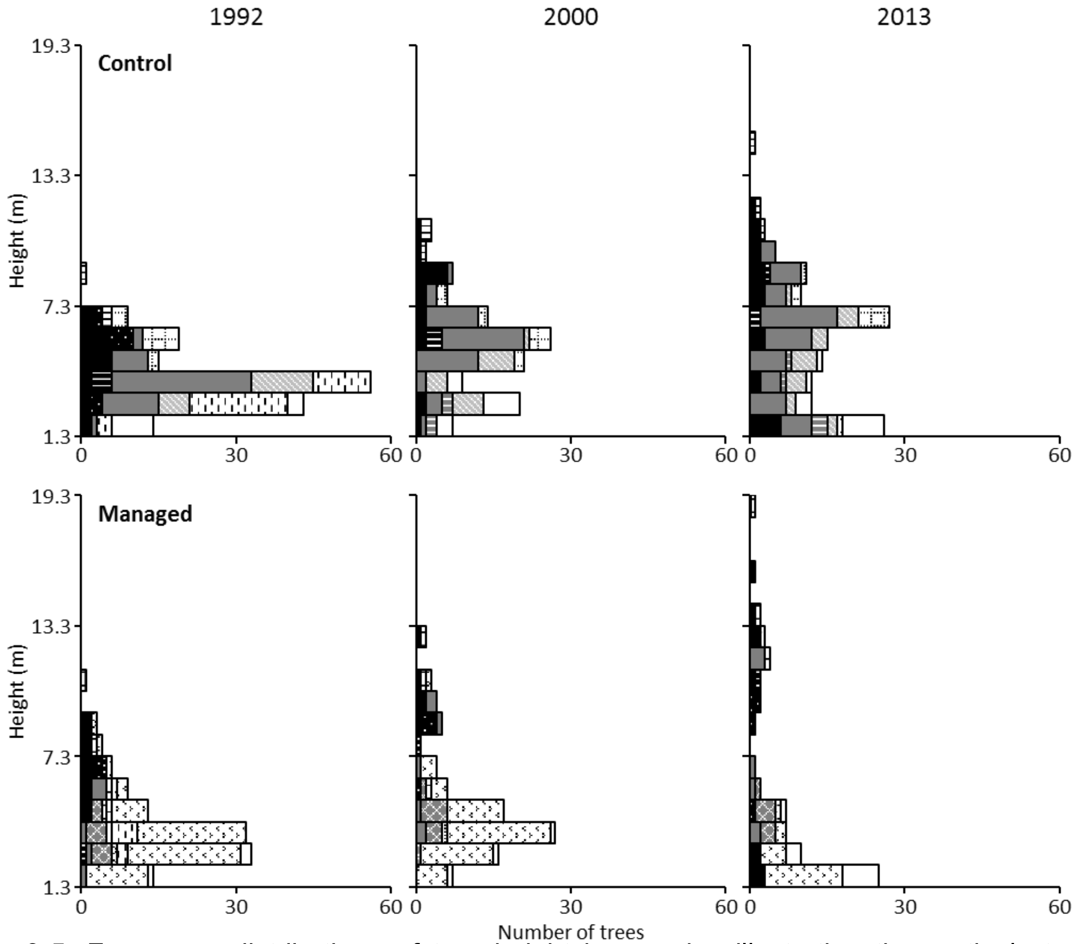



Fig. 3-5. Frequency distributions of tree height by species illustrating the vertical stand structure of the control and managed plots in the restoration site and reference forest. Colors and fill are as in Fig. 3-3.

# Chapter 4: Evaluating of a 35-year secondary forest restoration project in Kobe Sports Park, Japan

## Introduction

In this chapter, I compared vegetation change over a 35-year-period in a natural forest restoration site in Kobe City, southwestern Japan. The restoration objective was to achieve ecological continuity with the remnant secondary forest (targeted community), which was unmanaged satoyama. My research objective was to develop an objective method for assessing whether initial restoration goals were achieved and to evaluate whether designation of the targeted community was appropriate. I employ a quantitative method for visualizing the variability and chronological change in community structure of the restoration site relative to the targeted community and a reference, natural forest community.

## Methods

1. Study Sites

My study sites are in Kobe City, Hyogo Prefecture, southwestern Japan (Fig. 1). Because the forests in Kobe City are located near urban areas, they have been influenced by human impact (Miyawaki 1984). Many forests are former satoyama, maintained as short rotation (10–20 years) coppice forests for producing firewood and charcoal. Many of these secondary forests have been abandoned and remain unmanaged for as long as 50 years. Furthermore, in Kobe City, some forested mountains were excavated to construct artificial islands (e.g., the Port Island and the Rokko Island) or to develop residential areas. Many of these deforested hillslopes were replanted to prevent erosion and restore the natural ecosystem.

・ Kobe Sports Park

My first study site, Kobe Sports Park (KSP, 6.9 ha), is a large-scale forest restoration site. Kobe Sports Park is located in Suma and Tarumi Wards in Kobe City (Fig. 1, 34.68°N, 135.08°E, elevation 100 m). The park includes artificial hillslopes where saplings of native deciduous and evergreen broadleaved trees, such as *Quercus serrata*, *Quercus glauca*, and *Quercus phillyraeoides*, were planted after excavation in 1980 (Yoshida et al. 2002; Hotta et al. 2015). The restoration objective was to reestablish the natural vegetation and achieve ecological continuity with the remnant secondary forest dominated by deciduous and evergreen oaks (*Quercus* spp).

I established two plots (Plot A: 9 x 20 m and Plot B: 10 x 20 m) in the restored forest (hereafter: restored plots) and one plot (10 x 20 m) in the remnant forest adjacent to the restored forest, which was designated as the targeted community at the time of planting (hereafter: Target Plot). The distance between the two restored plots is 10 m, and the Target Plot is approximately 30 m from them. The two restored plots are located in areas of restored forest where originally, relative density of planted species differed. It was predicted that, eventually, the two restored plots and the remnant forest would all converge to similar late-seral community structure (Yoshida et al. 2002). See Hotta et al. (2015) for detailed description of the history of these plots.

・ Taisanji Temple

Due to the small area of remnant forest at KSP, I could only establish one Target Plot. In order to capture the variability of community structure in natural forests of this region, I established reference plots in secondary and late-seral forests at Taisanji Temple, approximately 2 km from KSP. In contrast to satoyama, which were heavily used by humans, forests in or near religious areas are less disturbed (Ishii et al. 2008). Taisanji is located in Nishi Ward, Kobe City (Fig. 2-1, 34.70°N, 135.07°E, elevation 70–200 m). The 16-ha forested area includes late-seral forest dominated by *Castanopsis cuspidata* and other evergreen broadleaved species. The oldest trees are about 100 years old. There is also secondary forest dominated by *Quercus serrata*. The secondary forest is adjacent to the local village and has similar history of human influence as the remnant secondary forest at KSP. I established 11 plots (10 x 20 m) in the late-seral forest and 6 plots (10 x 20 m) in the secondary forest (hereafter: late-seral plots and secondary-forest plots). I considered the forests at Taisanji as representing reference conditions for forests in this region because the late-seral forest has been completely undisturbed and the secondary forest has been unmanaged for nearly 50 years.

2. Data Collection

My objective was to quantitatively evaluate restoration success at KSP by comparing growth and dynamics of the restored plots to those of the Target Plot. I also wanted to evaluate whether selection of the target community was appropriate by comparing community structure of restored and target plots with those of the reference forest plots at Taisanji.

At KSP, the restored plots (Plot A and Plot B) were surveyed in 1992, 2000, and 2014 and the Target Plot was surveyed in 2015. At Taisanji, the late-seral plots were surveyed in 2003 and 2008 and the secondary-forest plots were surveyed in 2005, 2010, and 2015. In each survey, I identified all woody species (excluding lianas) and measured diameter at breast height (DBH, 1.3 m above ground level).

3. Data Analysis

To evaluate stand growth, I calculated total basal area of each plot for each survey year. To compare species composition and stand structure among plots, I calculated Bray-Curtis dissimilarity indices based on tree abundance and basal area (BA) data from each survey year using the vegan package in R (ver. 3.3.1, R Development Core Team). To visualize the relationships among plots, I used non-metric multidimensional scaling (nMDS; Kruskal 1964), which determines the non-parametric monotonic relationship between dissimilarities in the community-to-community matrix, and euclidean distances between communities to visualize the location of each community on a two-dimensional plane. Chronological changes in abundance- and BA-based dissimilarities were visualized using nMDS ordination to evaluate whether species composition and stand structure, respectively, of the restored plots were approaching those of the Target Plot and reference plots.

## Results

- Changes in tree density and basal area

Visual assessment of photographs taken at KSP indicated that in 1985, five years later after restoration, planted trees in the restored forest were much shorter than trees in the remnant forest (Fig. 4-1a). By 2000, 20 years later after planting, the restored and remnant forests were difficult to distinguish visually (Fig. 4-1b). The most current photograph taken in 2014 (Fig. 4-1c) shows canopy height of the restored forest is almost the same as the remnant forest and the border between the two is visually indistinguishable.

Although the two restored plots and Target Plot are in proximity to each other, tree density and species composition varied. Tree density in both restored plots had decreased after planting, but current densities were higher than that of the Target Plot (Fig. 4-2a). Although the initial planting densities were the same (ca. 10000 trees ha$^{-1}$), the current density in Plot A was about twice that in Plot B (Table 4-1a). Tree density in Plot A was lowest in 2000 and increased in 2014, while in Plot B, tree density decreased continuously. For each plot, the most abundant species were *Quercus phillyraeoides* (3100.0 trees ha$^{-1}$, 44.9%) in Plot A, *Ligustrum obtusifolium* (1150.0 trees ha$^{-1}$, 33.8%) in Plot B, and *Clethra barbinervis* (900 trees ha$^{-1}$, 40.0%), in Target Plot. Deciduous species were more abundant than evergreen species in all three plots.

Total basal area (BA) also varied between the two restored plots and Target Plot. In both restored plots, BA increased continuously after planting and current BA was nearly twice that of Target Plot (Fig. 4-2b). For each plot, species with largest BA were, *Q. phillyraeoides* (14.5 m$^2$ ha$^{-1}$, 39.7%) in Plot A, *Quercus serrata* (9.7 m$^2$ ha$^{-1}$, 33.2%) in Plot B, and *Cl. barbinervis* (5.3 m$^2$ ha$^{-1}$, 30.5%) in Target Plot (Table 4-1b). Evergreen species dominated the BA of both restored plots (70 and 58% in Plot A and B, respectively), whereas deciduous species contributed 78% of BA in Target Plot.

Compared to the restored plots at KSP, tree density and BA of reference forest plots at Taisanji were relatively stable during the study period (Fig 4-2). Tree density of the secondary-forest plots was similar to the current density of the restored plots, but higher than that of the Target Plot. Unlike the Target Plot, three evergreen species (*Eurya japonica*, *Camellia japonica*, and *Q. phillyraeoides*) were equally abundant in secondary-forest plots, while *Cam. japonica* was the single most abundant species in late-seral plots (Table 4-1a). BA of secondary-forest plots was similar to current BA of restored plots, but greater than in that of the Target Plot. BA of late-seral plots was similar to current BA of restored plots, but greater than that of the Target Plot. Unlike the Target Plot, the dominant species were *Q. serrata* (6.7 $m^2$, $ha^{-1}$, 23.3%) and *Cas. cuspidata* (24.5 $m^2$ $ha^{-1}$ (64.7%) in secondary-forest and late-seral plots, respectively (Table 4-1b).

· Comparison of species composition and stand structure

Both abundance- and BA-based dissimilarity indices between restored plots and the Target Plot were high, indicating species composition and stand structure were different (Table 4-2). Abundance-based indices were greater than 0.9, indicating that species composition of restored plots was markedly different from the Target Plot. BA-based indices of restored plots in relation to the Target Plot indicated that stand structure of Plot B was more similar to the Target Plot than that of Plot A.

The nMDS ordination based on abundance visually showed that species compositions of the two restored plots have changed very little over 22 years and are distinct from the Target Plot, as well as from each other (Fig. 4-3a). BA-based nMDS ordination showed that stand structure of the two restored plots were approaching, but diverging away from that of the Target Plot (Fig. 4-3b).

When analyzed together with the reference plots, nMDS ordination showed species composition and stand structure of the Target Plot were outside range of variability observed among 11 secondary-forest plots. The restored plots were also distinct from the reference forest plots, but with time, stand structures of both Plot A and Plot B approached those of secondary-forest plots, although Plot A diverted away in the most recent survey. On the other hand, species compositions and stand structure of most secondary-forest plots were approached those of late-seral plots.

## Discussion

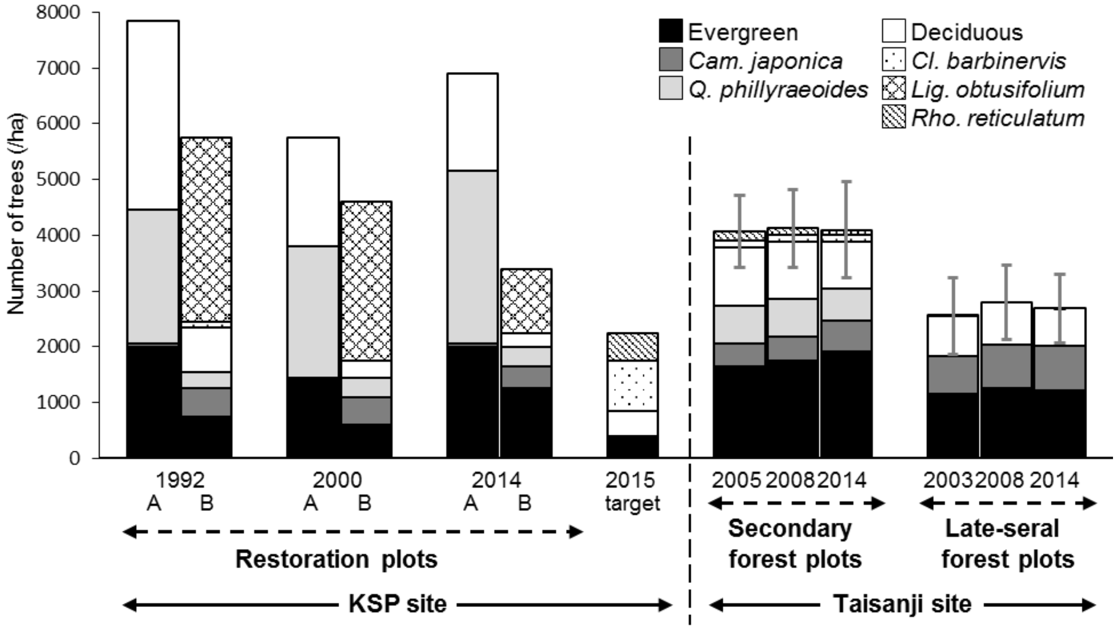
- Restoration success

Photographs showed that, it is possible to achieve visual continuity between restored and remnant forests in about 20 years after planting. This was because canopy height and basal area of the restored forest approached that of the remnant forest. However, my detailed analyses indicated species composition and stand structure of the restored forest was different from that of the remnant forest. In recent years, qualitative evaluation of community development and ecosystem integrity has become an important objective for forest restoration (Young 2000). In this regard, Bray-Curtis index is an objective method for quantifying dissimilarity of community structure between different forest types, as well as between successional stages (Lotter et al. 2014; Bruelheide et.al 2011), enabling objective evaluation of whether or not ecosystem quality has been restored. At KSP, I found that, in terms of species composition and stand structure, ecosystem continuity between the restored and remnant forests had not been achieved. Although it was initially predicted that vegetation of the restored forest would converge with the remnant forest to reach similar late-seral community structure, nMDS ordination showed that species composition and stand structure were markedly different between restored and remnant forests, as well as between the two restored forest plots. Furthermore, species composition of the restored forest had changed very little since planting. Because BA of the restored forest increased over time, its stand structure approached that of the remnant forest, but was diverting away from it. These results suggest that initial expectations for vegetation change in the restored forest have not been achieved and management may be required to change the direction of forest succession.

・ Target community designation

My analyses also showed that community structure of the remnant forest was outside the range of variability observed for the reference, natural forest, suggesting that designation of the target community may have been inappropriate. Tree density of the Target Plot was about half that of the secondary-forest plots and species composition was markedly different between the two. The direction of secondary succession can vary depending on species composition of the surrounding forest, which act as seed sources for secondary succession (Kepfer-Rojas et al. 2014; Arroyo-Rodriguez 2015). At Taisanji, the late-seral forest is an important seed source for the secondary forest. Azuma et al. (2014) showed that community structure of the secondary forest at Taisanji Temple is approaching that of the late-seral forest. The results of my nMDS ordination confirmed this, suggesting that future direction of secondary forest succession is predictable. In contrast, the remnant forest at KSP is surrounded by artificial vegetation and there are no natural forests nearby. This likely led to an unnatural successional pathway and contributed to the marked difference in species composition between the Target Plot and secondary-forest plots. My results suggest that if I designate natural, secondary forest as the target community for restoration, it is necessary to account for the range of spatial variation observed in the targeted community, as well as predict the future direction of secondary succession.


Fig. 4-1. Chronological change of the restored and remnant forests at Kobe Sports Park. Numbers indicate the year when photos were taken.



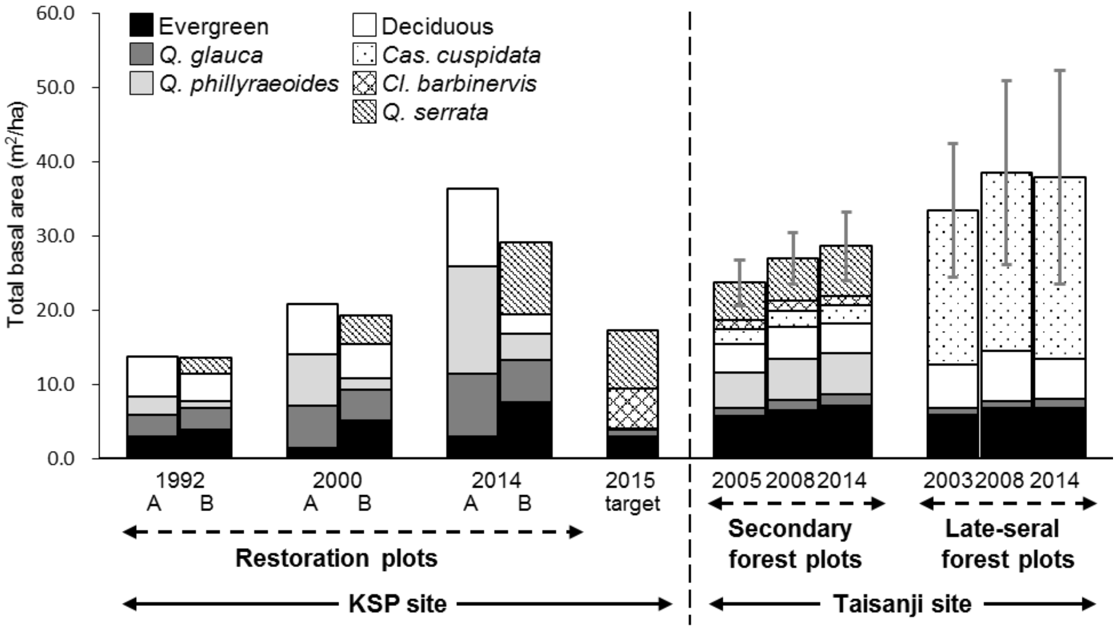




Fig.4-2. (a) tree population and (b) total BA transition. In Taisanji site, the mean of six (secondary forest) and 11 plots. Error bars indicate population standard deviation. The species which the tree population or the total BA are more than 20 percent of all specially are divided.

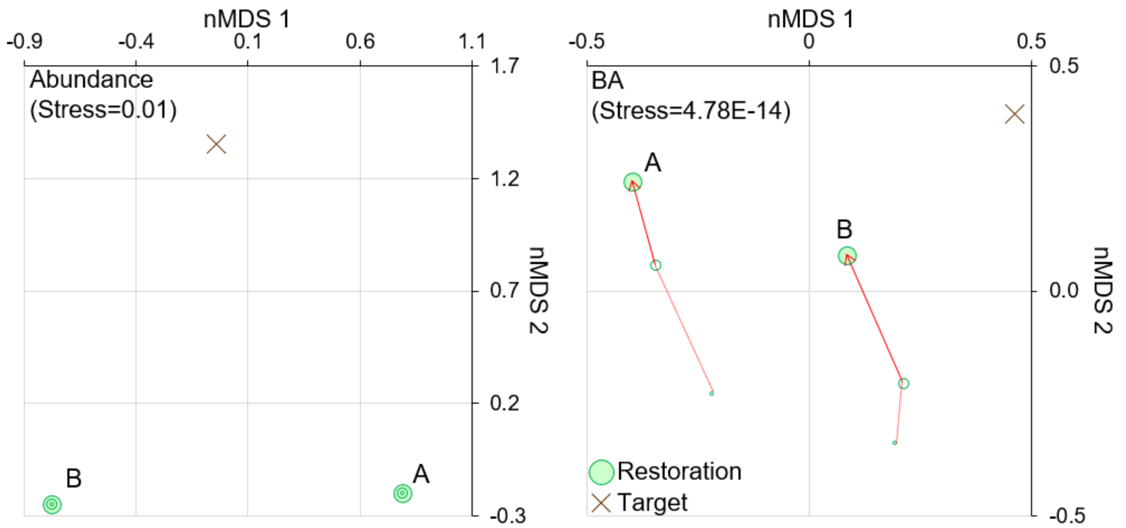



Fig. 4-3. Non-metric multidimensional scaling (nMDS) ordination of abundance- and basal-area (BA) based dissimilarity among plots in Kobe Sports Park (Table 2). Symbol size (small to large) reflects the chronological order of the surveys. Arrows indicate chronological change.

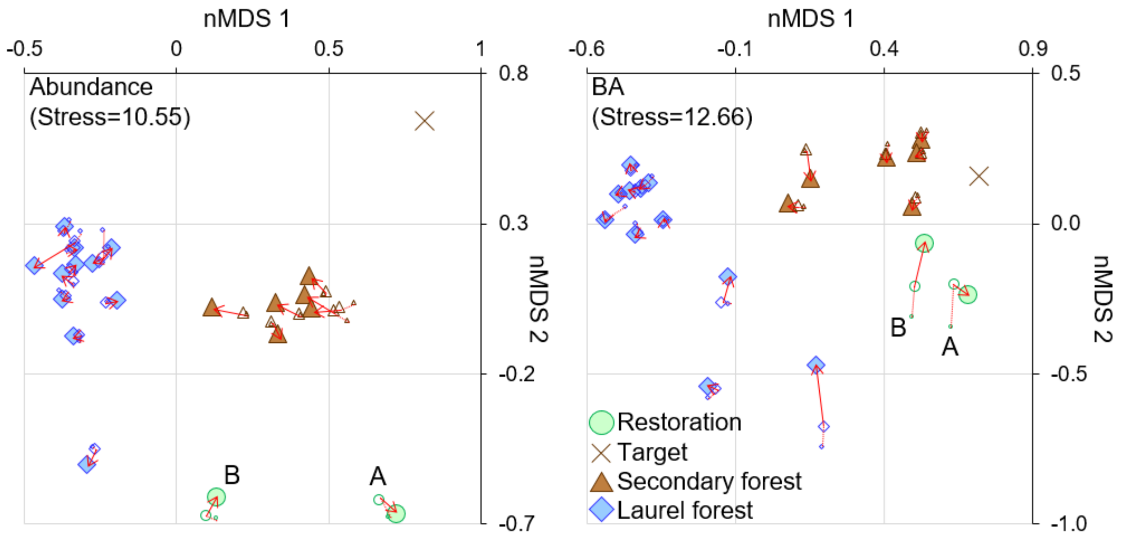



Fig. 4-4. Non-metric multidimensional scaling (nMDS) ordination of abundance- and basal-area (BA) based dissimilarity among plots in Kobe Sports Park and Taisanji. Symbol size (small to large) reflects the chronological order of the surveys. Arrows indicate chronological change.

Table 4-1: Current species composition (a: tree density, b: basal area) of the research plots. The “Others” include the species whose density and BA were less than 5% the plot total. Values in parentheses indicate percentages relative to the plot total.

**(a) Individual density (trees/ha)**

| Site | KSP | | | | | | Taisanji | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Species | Restoration A | | Restoration B | | Target | | Secondary forest [Average 6 plots] | | STDEV.P | Late-seral forest [Average 11 plots] | | STDEV.P |
| *Aphananthe aspera (Thunb.) Planch.* | - | - | - | - | - | - | - | - | - | 145.5 | (5.4) | 304.1 |
| *Aucuba japonica Thunb. var. japonica* | - | - | - | - | - | - | - | - | - | 245.5 | (9.1) | 163.0 |
| *Callicarpa mollis Siebold et Zucc.* | - | - | - | - | 300.0 | (13.3) | 8.3 | (0.2) | 18.6 | 13.6 | (0.5) | 30.8 |
| *Camellia japonica L.* | 50.0 | (0.7) | 400.0 | (11.8) | - | - | 566.7 | (13.8) | 265.6 | 795.5 | (29.6) | 317.3 |
| *Castanopsis cuspidata (Thunb.) Schottky* | - | - | - | - | - | - | 33.3 | (0.8) | 37.3 | 277.3 | (10.3) | 138.8 |
| *Castanopsis sieboldii (Makino) Hatus. ex T.Yamaz. et Mashiba* | - | - | 200.0 | (5.9) | - | - | - | - | - | - | - | - |
| *Cerasus jamasakura (Siebold ex Koidz.) H.Ohba* | 500.0 | (7.2) | 50.0 | (1.5) | - | - | - | - | - | - | - | - |
| *Cerasus leveilleana (Koehne) H.Ohba* | - | - | - | - | - | - | 50.0 | (1.2) | 50.0 | - | - | - |
| *Clethra barbinervis Siebold et Zucc.* | - | - | - | - | 900.0 | (40.0) | 125.0 | (3.1) | 62.9 | - | - | - |
| *Elaeagnus pungens Thunb.* | 100.0 | (1.4) | 200.0 | (5.9) | - | - | - | - | - | 4.5 | (0.2) | 14.4 |
| *Eurya japonica Thunb. var. japonica* | 250.0 | (3.6) | - | - | 200.0 | (8.9) | 683.3 | (16.7) | 292.5 | 140.9 | (5.2) | 90.0 |
| *Ilex pedunculosa Miq.* | - | - | - | - | 50.0 | (2.2) | 208.3 | (5.1) | 145.5 | 4.5 | (0.2) | 14.4 |
| *Ligustrum japonicum Thunb.* | - | - | - | - | - | - | 16.7 | (0.4) | 23.6 | 150.0 | (5.6) | 52.2 |
| *Ligustrum obtusifolium Siebold et Zucc.* | - | - | 1150.0 | (33.8) | - | - | - | - | - | - | - | - |
| *Lyonia ovalifolia (Wall.) Drude var. elliptica (Siebold et Zucc.) Hand.-Mazz.* | - | - | - | - | - | - | 233.3 | (5.7) | 207.5 | - | - | - |
| *Morella rubra Lour.* | 150.0 | (2.2) | 50.0 | (1.5) | 50.0 | (2.2) | - | - | - | - | - | - |
| *Nerium oleander L. var. indicum (Mill.) O.Deg. et Greenwell* | 950.0 | (13.8) | - | - | - | - | - | - | - | - | - | - |
| *Photinia glabra (Thunb.) Maxim.* | - | - | - | - | - | - | 350.0 | (8.6) | 258.2 | 86.4 | (3.2) | 118.9 |
| *Pittosporum tobira (Thunb.) W.T.Aiton* | 50.0 | (0.7) | 250.0 | (7.4) | - | - | - | - | - | - | - | - |
| *Platycarya strobilacea Siebold et Zucc.* | - | - | - | - | - | - | - | - | - | 13.6 | (0.5) | 30.8 |
| *Quercus acutissima Carruth.* | 150.0 | (2.2) | - | - | - | - | 8.3 | (0.2) | 18.6 | - | - | - |
| *Quercus glauca Thunb.* | 1100.0 | (15.9) | 450.0 | (13.2) | 100.0 | (4.4) | 308.3 | (7.5) | 224.4 | 186.4 | (6.9) | 318.4 |
| *Quercus phillyraeoides A.Gray* | 3100.0 | (44.9) | 350.0 | (10.3) | - | - | 558.3 | (13.6) | 190.2 | - | - | - |
| *Quercus serrata Murray* | - | - | 150.0 | (4.4) | 50.0 | (2.2) | 91.7 | (2.2) | 53.4 | - | - | - |
| *Rhaphiolepis indica (L.) Lindl. var. umbellata (Thunb.) H.Ohashi* | 350.0 | (5.1) | 50.0 | (1.5) | - | - | - | - | - | - | - | - |
| *Rhododendron reticulatum D.Don ex G.Don* | - | - | - | - | 500.0 | (22.2) | 83.3 | (2.0) | 68.7 | - | - | - |
| *Others (40 species) | 150.0 | (2.2) | 100.0 | (2.9) | 100.0 | (4.4) | 766.7 | (18.7) | 919.4 | 622.7 | (23.2) | 935.4 |
| Total | 6900.0 | (100.0) | 3400.0 | (100.0) | 2250.0 | (100.0) | 4091.7 | (100.0) | 862.4 | 2686.4 | (100.0) | 622.7 |

**(b) Total BA ($m^2$/ha)**

| Species | Restoration A | | Restoration B | | Target | | Secondary forest [Average 6 plots] | | STDEV.P | Late-seral forest [Average 11 plots] | | STDEV.P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Ap. aspera* | - | - | - | - | - | - | - | - | - | 0.3 | (0.8) | 0.6 |
| *Au. japonica* | - | - | - | - | - | - | - | - | - | 0.03 | (0.1) | 0.04 |
| *Cal. mollis* | - | - | - | - | 0.04 | (0.2) | 0.003 | (0.01) | - | 0.01 | (0.04) | 0.01 |
| *Cam. japonica* | 0.003 | (0.01) | 0.6 | (2.1) | - | - | 0.8 | (2.7) | 0.6 | 1.5 | (3.9) | 0.8 |
| *Cas. cuspidata* | - | - | - | - | - | - | 2.5 | (8.7) | 3.3 | 24.5 | (64.7) | 16.7 |
| *Cas. sieboldii* | - | - | 5.6 | (19.4) | - | - | - | - | - | - | - | - |
| *Cer. jamasakura* | 5.3 | (14.6) | 1.8 | (6.2) | - | - | - | - | - | - | - | - |
| *Cer. leveilleana* | - | - | - | - | - | - | 1.7 | (5.8) | 1.5 | - | - | - |
| *Cl. barbinervis* | - | - | - | - | 5.3 | (30.5) | 1.3 | (4.5) | 0.8 | - | - | - |
| *El. pungens* | 0.005 | (0.01) | 0.01 | (0.02) | - | - | - | - | - | 0.0004 | (0.001) | 0.001 |
| *Eu. japonica* | 0.2 | (0.5) | - | - | 0.6 | (3.5) | 0.4 | (1.5) | 0.2 | 0.1 | (0.2) | 0.1 |
| *I. pedunculosa* | - | - | - | - | 0.1 | (0.8) | 3.3 | (11.6) | 2.8 | 0.04 | (0.1) | 0.1 |
| *Lig. japonicum* | - | - | - | - | - | - | 0.01 | (0.1) | 0.03 | 0.3 | (0.7) | 0.2 |
| *Lig. obtusifolium* | - | - | 0.2 | (0.5) | - | - | - | - | - | - | - | - |
| *Ly. ovalifolia* | - | - | - | - | - | - | 0.9 | (3.0) | 0.7 | - | - | - |
| *Mo. rubra* | 2.7 | (7.3) | 1.2 | (4.0) | 2.2 | (12.5) | - | - | - | - | - | - |
| *Ne. oleander* | 1.9 | (5.3) | - | - | - | - | - | - | - | - | - | - |
| *Ph. glabra* | - | - | - | - | - | - | 0.4 | (1.3) | 0.3 | 0.1 | (0.2) | 0.1 |
| *Pi. tobira* | 0.01 | (0.03) | 0.1 | (0.3) | - | - | - | - | - | - | - | - |
| *Pl. strobilacea* | - | - | - | - | - | - | - | - | - | 2.3 | (6.0) | 5.4 |
| *Q. acutissima* | 3.2 | (8.8) | - | - | - | - | 0.01 | (0.04) | 0.03 | - | - | - |
| *Q. glauca* | 8.5 | (23.5) | 5.8 | (19.8) | 0.9 | (5.4) | 1.6 | (5.5) | 1.3 | 1.2 | (3.3) | 2.0 |
| *Q. phillyraeoides* | 14.5 | (39.7) | 3.5 | (12.1) | - | - | 5.5 | (19.3) | 2.4 | - | - | - |
| *Q. serrata* | - | - | 9.7 | (33.2) | 7.9 | (45.8) | 6.7 | (23.3) | 4.8 | - | - | - |
| *Rha. indica* | 0.1 | (0.2) | 0.001 | (0.00) | - | - | - | - | - | - | - | - |
| *Rho. reticulatum* | - | - | - | - | 0.2 | (1.2) | 0.05 | (0.2) | 0.04 | - | - | - |
| *Others (40 species) | 0.003 | (0.01) | 0.7 | (2.4) | 0.01 | (0.1) | 3.6 | (12.6) | 6.1 | 7.6 | (20.1) | 17.8 |
| Total | 36.4 | (100.0) | 29.1 | (100.0) | 17.3 | (100.0) | 28.6 | (100.0) | 4.6 | 37.9 | (100.0) | 14.4 |

***Other species**

| | | |
|---|---|---|
| Abelia serrata Siebold et Zucc. | Ficus erecta Thunb. var. erecta | Quercus salicina Blume |
| Abelia spathulata Siebold et Zucc. var. spathulata | Fraxinus sieboldiana Blume | Quercus variabilis Blume |
| Acer palmatum Thunb. | Gamblea innovans (Siebold et Zucc.) C.B.Shang, Lowry et Frodin | Rhododendron macrosepalum Maxim. |
| Aria japonica Decne. | Gardenia jasminoides Ellis | Rhododendron serpyllifolium (A.Gray) Miq. var. serpyllifolium |
| Callicarpa japonica Thunb. | Ilex chinensis Sims | Robinia pseudoacacia L. |
| Carpinus laxiflora (Siebold et Zucc.) Blume | Ilex crenata Thunb. | Sambucus racemosa L. subsp. sieboldiana (Miq.) H.Hara |
| Carpinus tschonoskii Maxim. | Ilex integra Thunb. | Symplocos prunifolia Siebold et Zucc. |
| Celtis sinensis Pers. | Ilex rotunda Thunb. | Ternstroemia gymnanthera (Wight et Arn.) Bedd. |
| Cerasus sp. | Litsea coreana H.Lév. | Vaccinium bracteatum Thunb. |
| Cinnamomum yabunikkei H.Ohba | Magnolia compressa Maxim. | Viburnum erosum Thunb. |
| Cleyera japonica Thunb. | Nandina domestica Thunb. | Viburnum wrightii Miq. |
| Daphniphyllum teijsmannii Zoll. ex Kurz | Osmanthus heterophyllus (G.Don) P.S.Green | Zanthoxylum ailanthoides Siebold et Zucc. |
| Dendropanax trifidus (Thunb.) Makino ex H.Hara | Pourthiaea villosa (Thunb.) Decne. var. villosa | Zelkova serrata (Thunb.) Makino |
| Distylium racemosum Siebold et Zucc. | | |

Table 4-2. Abundance and basal area (BA) based Bray-Curtis dissimilarity indices among plots in Kobe Sports Park for each survey year. Higher indices indicate less similar species composition.

| | | 1992 | | 2000 | | 2014 | | 2015 | data type |
|---|---|---|---|---|---|---|---|---|---|
| | | A | B | A | B | A | B | Target | |
| 1992 | A | | 0.47 | 0.38 | 0.52 | 0.58 | 0.53 | 0.91 | **BA** |
| | B | 0.78 | | 0.71 | 0.19 | 0.80 | 0.52 | 0.75 | |
| 2000 | A | 0.16 | 0.77 | | 0.65 | 0.28 | 0.51 | 0.88 | |
| | B | 0.83 | 0.14 | 0.84 | | 0.75 | 0.38 | 0.69 | |
| 2014 | A | 0.27 | 0.83 | 0.16 | 0.85 | | 0.62 | 0.88 | |
| | B | 0.80 | 0.41 | 0.79 | 0.35 | 0.78 | | 0.57 | |
| 2015 | Target | 0.93 | 0.91 | 0.91 | 0.94 | 0.92 | 0.93 | | |
| data type | | **Abundance** | | | | | | | |

## Chapter 5 General Discussion

Since the latter half of the 1990s, the greening projects which aim to restore the natural ecosystem have increased in Japan (Yoshida 2005; Yamadera 1986). Although these projects target natural forest or other types of natural vegetation, survey plots were set in only restored area (Yoshida 2005; Yamada 2008), In addition, follow-up surveys were usually carried out only for a short term (3 months to 3 years) after the restoration (Japan Road Association 2009). Conventionally, restoration success was assessed based only on quantitative measures (Japan Road Association 2009; Yoshida 2005) and qualitative assessment of ecosystem restoration was not made. Thus, new evaluation methods are necessary that can assess the level of achievement of ecosystem recovery (Burton 2014; Ruiz-Jaen and Aide 2005). Furthermore, the selection of target vegetation tends to be abstract and hypothetical, not concrete. In some cases, the target vegetation is not even specified (Hosogi 2011). The objective of this study, therefore was to establish a method to assess to what level natural ecosystem recovery was achieved and whether target vegetation selection was appropriate in a natural restoration project.

**In Chapter 2**, I set plots in both restored forest area, which has been ten years (in 2002) since planting nursery trees, and secondary forest area, which was surveyed in 2002 and designated as target vegetation for the restoration project. The species composition and vertical structure of the target area in 2012 was different from that in 2002 (Fig. 2-3). The compositional similarity between the restored and target plots in 2012 had not increased from that in 2002 (Table 2-2). I also showed that the composition and structure of the target vegetation is changing due to succession. The target forest was formerly used as satoyama, but has been abandoned for approximately 50 years. Moreover, it was affected by pine wilt. These social and pathological causes lead to vegetation change from the forest dominated by *Pinus densiflora* to the current forest dominated by low- or mid-height evergreen trees (Fig.2-3, Table 2-1). A reason why the restored vegetation did not approach the target vegetation may be because natural succession was not taken into consideration when designating the restoration goal ten years ago. Thus, I concluded that it is important to incorporate the concept of natural succession into restoration projects. Although I used only one type of similarity index (Chao index) to compare vegetation in this chapter, it would also be informative to compare it with other similarity indices to consider which is the most appropriate type of index.

**In Chapter 3,** I surveyed a restoration area, where thirty years have passed since planting in 1980, to clarify long-term dynamics of the restoration forest. I set two plots in restoration area, which were surveyed in 1992 and 2000. I assessed whether the restoration project was successful by comparing species composition and stand structure with nearby secondary forest, approximately 2 km from the restoration area. I also analyzed chronological change of the vegetation in the restoration area. Twenty years after restoration, the amount and size of trees in restoration forest were the same as those of secondary forest (Fig. 3-2, 3). In this chapter, I used Bray-Curtis index to quantify the relationship between the restoration area and the secondary forest. Moreover, I used the non-metric multidimensional scaling (nMDS) to visualize the relationships among the plots. The nMDS showed two results: a plot that approached the secondary forest and another plot that had approached the secondary forest but also diverted away from it (Fig. 3-4). The vertical structure also showed two results: an unstratified plot whose structure has hardly changed from the initial survey (for 21 years) and a plot having the same structure as the natural forests (Fig. 3-5). The reason why these differences occurred was not clear. It may be due to the proportion of evergreen trees, or whether management was done. Further investigation of the cause of these differences would greatly contribute to the establishment of management methods for natural restoration project.

**In Chapter 4,** I evaluated whether the selection of target forest type was appropriate in my research site. I quantified the relationship between the restoration forest site, the target forest area, which is remnant forest in secondary forest next to restoration forest, and the late-seral and secondary forest areas in the reference site, which was ca. 2 km from the restoration site. The visual appearance of the restoration forest area was similar to the remnant forest area (Fig.4-1). The species composition, however, was different between the restoration and the target forest area (Fig.4-2). The nMDS results showed vegetation of the restoration plots had diverted from that of the target plot (Fig. 4-3). The nMDS results also showed that vegetation of the target forest was discrete from the secondary forest in the reference site, even though it is only 2 km away (Fig. 4-4). Even within the same forest, vegetation had a wide range of variability. The nMDS results also indicated that, although the forest vegetation is changing, as I showed in Chapter 2, the direction may be predictable in some cases (Azuma et al. 2013). This suggests that, when setting restoration target, it is necessary to consider the range of variability of the targeted vegetation and predict the future direction of succession.

Conclusions

Using multi-variate criteria, I was able to assess, qualitatively, the success of two restoration projects in Kobe. Based on the results of my study, I conclude the following:

1) The target vegetation, the secondary forest in this study, is changing because of natural succession induced by social changes in human use of forests.
2) The canopy structure of restoration forests tends not to stratify. Although the restoration forest may achieve visual continuity with remnant forest (target forest), this does not necessarily reflect ecological continuity.
3) Similarity indices like Chao or Bray-Curtis can be used to quantify relationships among different vegetation types. Such data can be plotted using nMDS to visualize the relationships on a two-dimensional plane.

For future restoration projects, I propose the following:

<u>1. Clear designation of the target vegetation upon planning</u>

The goal of natural forest restoration is to re-establish the “natural” vegetation of the region. Most conventional projects, however, have no specific target vegetation. As a consequence, it is difficult to assess whether the restoration goal has been achieved. Thus, it is necessary to designate a clear target vegetation, not just a conceptual one. To do this, I need to comprehend the inherent range of target vegetation types and predict the dynamics of target vegetation.

<u>2. Promoting structural development of the restoration site</u>

In conventional restoration projects, planting is usually done only once, initially, at high density. The stand is expected to self-thin, but it has been found that this can hardly be expected. Thus, in order to promote development of stand structure, such as canopy stratification, it is necessary to plant multiple times and also conduct thinning management.

3. Re-designation of the target vegetation based on scientific inference

No matter how thoroughly surveying may be done when designating the target vegetation, it is conceivable that the target vegetation will change in an unexpected direction, or the vegetation of the restoration area may change toward different directions. In such case, it is necessary to manage vegetation of the restoration site to direct it toward the target, or re-designate the target.

A restoration project usually has 3 stages: planning, construction and management. The findings of this study suggest that it may not be suitable to divide natural restoration projects into three discrete stages. I propose that construction and management stages need to be repeated in long-term restoration projects until the desirable vegetation is reached.

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# 要旨

山がちな地形を持つ日本では宅地開発に際して，人工的な斜面，いわゆるのり面が生じる．のり面は土壌が安定しておらず，土砂の流出などを抑えるため土留めなど土木工事が行われる．その際に緑化を行い，植物体による土壌の安定が図られてきた．日本における緑化技術はこのように土木的な防災を目的として発展してきた．加えて開発行為には森林の伐採が伴うことがほとんどである．近年，緑化には開発により破壊された生態系のミティゲーションの役割も求められるようになってきたため，緑化技術には土木的な観点だけではなく，生態的な観点をも取り入れる必要が高まってきた．従来は土壌基盤に植物体を活着させられれば緑化の成功を意味したため，量的な指標で緑化の成功を評価可能であったが，生態系の復元を評価するためには，群落の種構成や林分構造などといった緑の質を評価する必要が出てきた．しかし緑化に生態学的な観点が取り入れられるようになってからまだ年数が浅く，評価体制は整っていない．生態学的観点を取り入れ自然再生を目的とする緑化は緑化目標とする群落（目標林）が設定され，目標を達成するよう計画が立てられる．そこで本研究では，緑化による生態系復元を評価するため，緑化目標の達成度を質的に評価する指標を作ることを目的とした．

第１章では自然再生を目的とした緑化地における調査は大部分が短期調査であり順応的管理を行うための情報が不十分であること，評価体制が整っていないこと，二次林を復元する技術が確立されていないことを指摘し，本研究の構成を概説した．

第２章では自然林再生を評価するため植栽後 10 年が経過した神戸市の緑化地における種組成および林分構造と，緑化目標とした二次林とを類似度指数（Chao index）を用い比較した．緑化地では木本種の本数密度が増加し，のり面被覆は成功したものの，林分の垂直構造が未発達であった．目標林はマツ枯れ発生後に植生が遷移し種組成が変化していた．そのため緑化地と目標林の種組成の類似度は植栽時と変わらず，林分構造の類似度は

<要旨>

植栽後 10 年が経過した現在の方が低かった．以上の結果から，在来種を一斉に植栽する現在の自然林回復緑化は，植生の量的回復には貢献できるものの，自然林の群落構造を再現するには不十分であると考えられた．緑化地において群落構造を早期に再現するためには目標林は変化するものであると認識し，遷移にあわせた継続的かつ順応的な管理が必要であると考えられた．

第３章では緑化地における群落構造の変化を把握した．1992・2000 年に追跡調査が行われた植栽後 33 年が経過した神戸市総合運動公園の緑化地で調査を行った．本緑化地は公園建設時に掘削され生じたのり面上の緑化地と掘削されずに残存していた森林との景観的連続性を実現することを目的として植栽が行われた．緑化地内に過去の調査と同一の２調査区を設置し，最初の調査から 21 年間の群落の変化を追った．その結果，緑の量を示す胸高断面積（BA）合計は年々増加し，2010 年の調査データのある近隣の太山寺二次林と同程度以上の BA 合計に達していることが分かった．しかし両者の種構成は大きく異なっていた．緑化地では太山寺二次林と比較して常緑樹の個体割合が高く，特にウバメガシ（Quercus glauca）の個体密度が高かった．森林は通常垂直的な階層構造が発達するが，緑化地では階層化が見られる調査区と，見られない調査区があった．緑化地は太山寺と比較して，緑の量（BA 合計）は同程度であったが，質（群落構造）は異なることが明らかになった．このことを定量的に示すため，緑化地と太山寺二次林の群落とを類似度指数（Bray-Curtis index）を用いて比較し，非計量多次元尺度構成法（non-metric multidimensional scaling：nMDS）を用い関係性を可視化した．緑化地と太山寺二次林は nMDS 上で距離が離れていた．階層化が見られた調査区は太山寺二次林に近づいている傾向があったが，階層化が見られなかった調査区は離れていく傾向があった．類似度指数と nMDS を用いた解析では，異なる群落を１つの二次元のグラフ上に示すことができるため，緑化目標が達成されたか否かを質的に評価する１つのツールとして有用であることが示された．

<要旨>

第４章では第３章と同一の緑化地に隣接する二次林内に調査区（目標林）を新たに設置した．1985・2000・2014 年に撮影された遠景写真から，景観的な連続性は植栽後 20 年ほどで達成されていたことが分かった．緑化地と目標林の群落とを nMDS を用いて比較したところ，緑化地群落は目標林と乖離しておりさらに年々異なる方向へと変化していることが明らかになった．目標林は 2015 年の調査結果のみであり，目標林自体の過去の群落動態は不明である．そこで太山寺二次林（調査年：2005・2010・2015）および照葉樹林（調査年：2003・2008・2015）を目標群落と仮定して解析を行った．その結果，二次林の群落構造には変異があることや二次林が照葉樹林へと遷移していることが明らかになった．これらの結果から，同一群落内でも種構成などには幅があり，既存林を目標林として設定する場合は空間的な変異を考慮する必要があることが示唆された．また，過去の植生変化のデータから既存林の遷移の方向がある予測可能であることが示された．nMDS は群落構造の空間的な変異と経時的な変化を同時に把握できる評価手法であることが示された．

第５章では以上の結果をまとめ，自然林再生を目的とする緑化地では，①目標林の種組成・群落構造の変異を把握した植栽計画，②目標林の動態を考慮し，将来的な遷移の方向予測にもとづいた植栽計画および順応的管理，③施工後の緑化地における目標達成度の客観的評価が必要であることを提言した．本研究で開発した，nMDS を用いた群落構造の解析方法は，群落間の構造の違いや動態を経時的に把握するために有効なツールである．

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