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Sasaki, Takeshi

Ishii, Hiroaki

Morimoto, Yukihiro

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Evaluating restoration success of a 40-year-old urban forest in reference to mature natural forest

Takeshi Sasaki[1, 2, 3], Hiroaki Ishii[2, *], Yukihiro Morimoto[3]

[1] Graduate School of Technology, Industrial and Social Sciences, Tokushima University, Tokushima 770-8506, Japan

[2] Graduate School of Agricultural Science, Kobe University, Kobe 657-8501, Japan

[3] Faculty of Bio-environmental Science, Kyoto Gakuen University, Kameoka, Kyoto 621-0022, Japan

*Author for correspondence

Phone/FAX: +81-78-803-5826

Email: hishii@alumni.washington.edu

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Abstract

We assessed whether forest restoration was successful in Expo '70 Commemorative Park in Osaka Prefecture, Japan, which was planted in the 1970s with native late-successional tree species. Detailed survey and analysis of species composition, stand vertical stratification, and forest dynamics, including comparison with a reference natural late-successional forest, were conducted. The restoration plots had grown to larger basal area compared with the reference plots, however, this was a consequence of very high densities of the overstory trees due to low self-thinning rate. Stand vertical structure of the restoration plots was biased toward overstory layers, causing high mortality of understory trees and shrubs. Because there are no mature forests near the restoration site that could act as a seed source, abundance and diversity of understory trees are likely to continue decreasing in the restoration plots, resulting in single-layered forest structure similar to those of monocultures and even-aged forests. Many seedlings of exotic species emerged in the restoration plots and this could lead to a plagiosere where exotic species dominate the vegetation inhibiting regeneration and growth of native species. Ordination analysis using different measures, basal area and abundance, showed apparently contradicting results, suggesting that multiple criteria are needed to evaluate forest restoration success. Our results indicate restoration of mature, late-successional forest cannot be achieved by simultaneous planting of native species. To sustain urban forests into the future, we must conduct long-term monitoring and

management referencing natural forest structure and dynamics.

Keywords: canopy stratification; late-successional forest; native species; reference site; restoration success; urban afforestation

Introduction

Globally, the proportion of the world’s population living in urban areas has reached 50% (United Nations, 2008), and it continues to increase. Urban forests play a significant role to improve urban environmental quality and human health (Nowak and Crane, 2002). Urban forests provide various ecological services, such as carbon sequestration (Nowak and Crane, 2002), climate regulation including heat island mitigation (Hardin and Jensen, 2007), aesthetic values (Tyrv□inen et al., 2003), and habitat for a variety of species (Alvey, 2006; Sandström et al., 2006). However, it is difficult to sustain forest ecosystem functions in harsh urban environments and this has led to degradation of urban forest quality in many cities (Yang et al., 2005; Ishii et al., 2010; La Paixa and Freedman, 2010; Jim, 2013; Ballantyne et al., 2014). Thus forest restoration projects have been conducted in urban areas around the world in an effort to sustain healthy urban forests (Morimoto et al., 2006; Ruiz-Jaén and Aide, 2006; Oldfield et al., 2013; Climate Summit, 2014).

As goals of the forest restoration, many projects pursue forests comprising native tree species indigenous to the region (Ruiz-Jaén and Aide, 2005b; Hotta et al., 2015; Almas and Conway, 2016; Gatica-Saavedra et al., 2017). These forests are expected to be self-sustaining and resilient to perturbation (Ruiz-Jaén and Aide, 2005a; Morimoto et al., 2006). However, it remains largely unknown whether such restorations will actually produce mature native forest (Oldfield et al., 2013). Despite the importance of long-term monitoring and management toward the goals, most studies monitoring the development of native urban forest restoration projects report progress within five years after planting (Ruiz-Jaén and Aide, 2005a; Oldfield et al., 2013). Very few studies have explored long-term forest dynamics by repeated monitoring after planting (but see Hotta et al., 2015).

In Japan, Environmental Protection Forests have been planted in urban and industrial areas since 1970s. These forests aimed to restore “potential natural vegetation” (*sensu* Miyawaki, 2004) of the region. In such forests, tree saplings, mainly composed of native late-successional tree species, are planted at high densities (20,000-120,000 trees/ha^2) on improved soil in order to achieve canopy closure within a short period (Nakashima et al., 1998; Miyawaki, 2004; Nakamura et al., 2005). The prevalence of this method was based on thinking that mature, climax forest, most of which had been lost by drastic urbanization in Japan during 1950-60s, should be conserved and restored (Morimoto et al., 2006). Initial assessment indicated success of this restoration method in terms of vegetation cover, as a result of rapid tree growth to the height of 9-11 m 15 years after planting (Murata and Komaki, 2001).

However, the concept of forest conservation and restoration has changed over time due to accumulation of new scientific knowledge. Recently, in addition to clearly defined quantitative measures, such as canopy height and vegetation cover, the importance of integrative measures of ecosystem quality, such as biodiversity and ecosystem services have been revealed (Mace, 2014; Mori, 2017). Intrinsically, late-successional natural forests are composed of various tree species and uneven-aged trees, and have more complex structure and higher productivity compared with monocultures and even-aged forests (Ishii et al., 2004). Structural complexity and tree-trait diversity increases vertical stratification and reduces niche overlap (Ishii et al., 2013; Dănescu et al., 2016), contributing to increasing biodiversity of a variety of organisms by generating various habitats (MacArthur and MacArthur, 1961; Brokaw and Lent, 1999; Ishii et al., 2004), and also enhances resilience to disturbances (Pretzsch, 2014).

Therefore, to evaluate the success of natural forest restoration, various parameters of forest structure should be measured, including not only aggregate measures such as forest height and vegetation cover, but also species composition and vertical structure (Rebele and Lehmann, 2002; Marziliano et al., 2013). Monitoring forest dynamics such as tree growth, mortality, and recruitment are also important for future ~~adaptive~~ management and strategies for long-term sustainability (Robinson and Handel, 2000; Morimoto et al., 2006; Oldfield et al., 2013). In addition, changes in forest structure need to be compared with that of reference sites to evaluate the level of restoration success (Hobbs and Norton, 1996; Ruiz-Jaén and Aide, 2006).

In this study, we assessed whether forest restoration was successful in Expo ’70 Commemorative Park in Osaka Prefecture, Japan. The urban forest within Expo ’70 Park (total park area = 130 ha) was planted in the 1970s and represents a pioneering example of the Environmental Protection Forests in Japan (Nakamura et al., 2005; Morimoto et al., 2006). In this site, long-term, repeated monitoring revealed that canopy closure was achieved by 30 years after planting, and stand volume growth could be promoted by improving soil drainage (Morimoto and Kobashi, 1985; Njoroge et al., 2000; Sasaki et al., 2007). However, detailed survey and analysis of the forest structure and dynamics, including comparison with reference forest, has not been conducted. The purpose of this study is to evaluate restoration success of this forest using various measures including species composition, stand vertical stratification, and forest dynamics and to compare these measures with those of a reference natural forest in the same vegetation zone.

Methods

Study sites and vegetation survey

The restoration site is Expo ’70 Commemorative Park (135°31–32’E, 34°47–48’N, 40~61m ASL), located in Suita City, Osaka Prefecture, Japan (Fig. 1a). The total area of the park is about 130 ha. This region is in the warm-temperate zone, where evergreen broad-leaved forest is regarded as the inherent natural vegetation (Miyawaki et al., 1984). The area was covered with semi-natural secondary forest until 1960s, followed by large-scale clearing for the World Exposition in 1970. After the Exposition, the site was mounded with imported local soils from the neighboring hills by filling and grading operations, and re-vegetation was conducted from 1972 to 1976. The forest restoration area, which occupies about 27 ha in the park, was designed to achieve climax forest, and saplings of various tree species, mainly composed of evergreen broad-leaved trees such as *Quercus glauca* Thunb. ex Murray, *Castanopsis cuspidata* (Thunb. ex Murray) Schottky, *Castanopsis sieboldii* (Makino) Hatusima ex Yamazaki et Mashiba, *Machilus thunbergii* Sieb. et Zucc., and *Cinnamomum camphora* (L.) Presl, *Camellia Japonica* L., and *Photinia glabra* (Thunb.) Maxim., were planted. These include canopy, sub-canopy, and shrub species, many of which are late-successional species native in this region (Miyawaki et al., 1984). Some species, however, are not precisely native. For example, it was revealed by later studies that *M. thunbergii* is native to regions nearer to the coast (Hattori, 1992), and *C. camphora* is native in regions further south than Expo Park (Tabata et al., 2004; Ishii et al., 2016). Along the circumference of the park, tree species which have high tolerance to air pollution were planted, including *Quercus phillyraeoides* A. Gray, *Pittosporum tobira* (Thunb. ex Murray) Aiton, and *Myrica rubra* Sieb. et Zucc., because the air pollution was severe in Japanese urban areas in

1970s. After the Expo, urbanization has progressed in the area surrounding the park. Presently, the distance from the nearest forest is about 2.5 km. Additional details on this park have been reported in previous studies (Morimoto et al., 2006; Sasaki et al., 2007, 2016).

In this study, 13 sample plots (restoration plots) within Expo Park were investigated (Fig. 1b). Among them, ten plots (E1-E10) were already established and investigated in the previous studies (Morimoto and Kobashi, 1985; Njoroge et al., 2000; Sasaki et al., 2007). For these plots, the earliest complete surveys used in this study were conducted in 1995. To increase the number of survey plots, we established three new plots (E11-E13) in 2006. The plot sizes ranged from 150 m^2 (10 m×15 m) to 375 m^2 (15 m×25 m) depending on tree size and density of the stands (Table 1). The first survey for this study was conducted in 2004 (E1-10) and 2006 (E11-13) when we labeled all trees greater than 1 cm in diameter at breast height (DBH, 1.3 m above ground level), and measured the DBH and tree height (*H*). In 2008, we re-measured the DBH and *H* of all labeled trees, and recorded new individuals greater than 1 cm in DBH. We conducted a third measurement of DBH and *H* in 2014, and measured all new individuals taller than 1.3 m regardless of DBH.

As the reference forest, we selected a mature, late-successional forest at Taisanji Temple (34°41’N, 135°04’E, 70–200 m ASL, Fig. 1a), Hyogo Prefecture, Japan, 43 km west of the Expo Park. This forest is dominated by *C. cuspidata*. The oldest tree is ca.95 years and the forest is estimated to be more than 100 years old (Azuma et al., 2014). This forest has been used as a place of Buddhist religious training and has had minimal human intervention, resulting in a highly natural forest with high species richness, representative of mature, late-successional forest in this region (Ishida et al., 1998).

Thus, this forest could be considered as a reference vegetation that was intended for the forest restoration in the Expo ’70 Park. Although using more than one reference sites is desirable (Ruiz-Jaén and Aide, 2005a), there were no other late-successional forests within a reasonable distance of Expo ’70 Park, because in the warm-temperate zone in Japan, vegetation has been strongly affected by human intervention, and very few natural forests remain (Kamada, 2005). In 2008, we measured DBH of all trees taller than 1.3 m in four 10 × 20 m plots (T1-T4, reference plots: Fig. 1b, Table 1), followed by a re-measurement of DBH and measurement of *H* in 2014. These data were compared with data from the first survey conducted in 2003 by Azuma et al., (2014). In all study plots, *H* was measured by a telescoping pole for trees shorter than 12 m and by an ultra-sound clinometer (Vertex III, Haglof, Sweden) for taller trees.

Data analysis

In this study, *C. sieboldii* and *C. cuspidata* were grouped together because these two species are genetically similar, known to hybridize, and difficult to distinguish morphologically (Yamada and Nishimura, 2000).

From the DBH data in 2014, we calculated basal area (BA, m^2/ha) for each tree species in each plot, and carried out a hierarchical cluster analysis using the vegan package in R (ver. 3.4.1, R Development Core Team). We used the Bray-Curtis dissimilarity index and Ward method to group plots based on BA composition. The Ward method hierarchically combines cases (plots in this study) into clusters to maximize within-group homogeneity and between-group heterogeneity. Hierarchical cluster analysis and Ward’s method is commonly used to classify ecological communities including urban forest vegetation (e.g., Steenberg et al. 2015; Nitoslawski et al. 2017).

For the restoration and reference sites respectively, the data within the same clusters were aggregated, and their vertical structures were compared by frequency distribution of tree heights. Differences among plots were compared using Wilcoxon rank test. By visually assessing the valleys and peaks in the frequency distributions, we defined trees taller than 8 m in 2004-2006 (restoration site) and 2003 (reference site) as overstory trees and analyzed the relationship between density and mean stem volume of overstory trees during the recent ca. 10 years. For the restoration plots dominated by *Q. phillyraeoides*, trees taller than 5 m in 2004 were defined as overstory trees because the tree heights were lower than for the other plots. The stem volume for each tree (V, m^3) was calculated as that of a cone (BA × H / 3).

We calculated rates of tree loss between surveys using to the following equation:

Tree loss rate = $1 - N_b / N_a$

Where N_a is number of trees at the initial survey and N_b is the number remaining at the following survey. We also compared species composition of newly recruited juvenile trees ($H \geq 1.3$ m) in 2014 among plots.

Within the restoration site, chronological changes in species composition were analyzed for plots classified into the same cluster as the reference plots using a non-metric multidimensional scaling (nMDS) ordination based on Bray-Curtis dissimilarity index in R (‘vegan’ package). nMDS is a distance-based ordination technique where the relationship among the restoration and reference plots are drawn on two-dimensional plane to display graphically the dissimilarities among plots based on species composition. Because distance between plots on the nMDS ordination plane represent their relative dissimilarities, position and distance of restoration plots relative to reference plots and the direction of change over time can be interpreted as restoration success (Ruiz-Jaén and Aide, 2006; Matthews and Spyreas 2010; Hiers et al. 2012). nMDS is suited for ecological analyses because it is nonparametric and can be used to relativize distance measures based on a wide variety of ecological data (McCune and Grace 2002). Ordination analysis was conducted for basal-area- and abundance-based species compositions of each plot using data of trees ≥ 1cm in DBH and differences among plots were tested with permutational ANOVA (999 permutations) using the 'adonis' package in R (Coleman-Derr et al., 2016).

Results

Species composition

Reference plots were divided into four groups according to cluster analysis based on basal area composition in 2014 (Fig. 2). Three restoration plots (E3, E9 and E13,) dominated by *C. cuspidata* (hereafter: Group 1 restoration plots) were grouped with the reference plots (T1-4). Within this group, E9 was more similar to the reference plots than E3 and E13. The other restoration plots were grouped according to their respective dominant canopy species (Table 2). Of these, plots dominated by *Q. phillyraeoides* (E1, E7 and E8) were more similar to the reference plots than those dominated by *C. camphora* (E6 and E11). Plots dominated by *Q. glauca* (E2, E4, E5, E10 and E12) were most distinct from the reference plots.

Vertical structure

Frequency distributions of tree height for each group of plots in 2014 are shown in Fig. 3. The restoration plots had larger basal area and higher density of overstory trees than the reference plots. Despite having common dominant species (*C. cuspidata*), vertical structures of Group 1 restoration plots and reference plots were different (W =13661, $P < 0.001$, and $W = 14580$, $P < 0.001$ for density and basal area, respectively). In reference plots understory trees (<3.3 m height) contributed 50% of the total number of trees with *Aucuba japonica* Thunb. and *Camellia japonica* contributing 25.9 and 31.8%, respectively of understory trees. In contrast, for Group 1 restoration plots, understory tree contribution was only 25.7% comprising species such as *C. cuspidata*, *Ligustrum japonica*, *Camellia japonica*, and *Photinia glabra*, each contributing 3.2-6.5% of understory trees. Understory tree contribution was also low (28.7%) for Group 4. There were many understory trees in Group 2 and 3 because the dominant species, *Q. phillyraeoides* and *C. camphora* are relatively shade intolerant and gaps had opened due to death of some overstory trees. However, 50.7 and 22.4% respectively, of understory trees in Groups 2 an 3 were *Ligustrum lucidum* Ait., an exotic species, while juveniles of the dominant overstory species contributed only 7.2 and 0% (*Q. phillyraeoides* and *C. camphora*, respectively for Group 2 and 3).

Density-volume relationship

Tree density and mean stem volume of overstory trees for the reference plots in 2003 were 413 trees/ha and 0.545 m^3 respectively. In the reference site, tree density had decreased with increasing stem volume along the self-thinning line from 2003 to 2014 (Fig. 4). Compared with the reference plots, all plot groups in the restoration site showed less density decrease relative to the increase in stem volume, indicating lower rate of self-thinning. Over-crowding was especially evident for Group 1 restoration plots (dominated by *C. cuspidata*), which had much higher density than the reference plots, despite similar overstory species composition.

Loss and recruitment of trees

Rates of tree loss for the main tree species in each plot group are shown in Table 3. In the restoration plots, loss rate of *Q. glauca* was the lowest, whereas that of understory trees and shrubs such as *P. tobira*, and *Ternstroemia gymnanther* (Wight et Arn.) were high. In the reference plots, loss rates of sub-canopy and shrubs such as *Cleyara japonica* and *Eurya japonica* Thunb. were lower compared with overstory treess such as *C. cuspidata*.

Table 4 shows numbers of newly recruited juvenile trees ($H \geq 1.3$ m) in 2014. In the restoration site, *L. lucidum* had the largest recruitment followed by *Q. glauca*. There were no seedlings of canopy dominants, *Q. phillyraeoides* and *C. camphora*. *Trachycarpus fortunei* Wendl., another exotic species, was newly recorded for 2014. In the reference site, all newly recruited trees were native species, and no exotic species were recorded.

nMDS Ordination analysis

The nMDS ordination plots indicated that, despite having common dominant species, *C. cuspidata*, compositions of the three restoration plots (E3, E9, E14) were distinct from the reference plots both in terms of basal area and abundance (Fig 5, $P < 0.05$ for all pair-wise comparisons in relation to reference plots). In agreement with results of cluster analysis, axis 1 score of the basal-area-based ordination for plot E9 in 2014 was closer to reference plots than E3 and E13 (Fig. 5a). E9, however, was distinct from the other restoration plots as well as from the reference plot along axis 2. Over time, basal-area composition of all three restoration plots had changed in the direction of the reference plots along axis 1, which may simply reflect increase in basal area of *C. cuspidata*. In contrast, abundance-based ordination indicated that composition of the restoration plots had diverged away from the reference plots (Fig. 5b).

Discussion

Our results indicated that, in just 40 years after planting, the forest restoration site had grown to larger basal area compared with the 100-year-old reference forest. Thus, it could be said that forest restoration was successful in terms of vegetation quantity. However, this was a consequence of very high densities of the overstory trees in the restoration plots due to low self-thinning rate. As a result, vertical structure was biased toward overstory layers and understory abundance and diversity of the restoration plots were low compared with the reference site. In a restoration site in Kobe City (44 km from Expo '70 Park), where trees were planted with high density, lack of management resulted in over-crowding of overstory trees similar to that of Expo '70 Park (Hotta et al., 2015).

While results of basal-area-based nMDS ordination indicated that species compositions of the restoration plots had changed in the direction of the reference plots, abundance-based ordination indicated the opposite (Fig. 5). Continuous increase in basal area of the dominant species, *C. cuspidata*, in the restoration plots likely contributed to the first result. However, many understory trees and shrubs had died and species richness had decreased in the restoration plots, resulting in apparent distinction from the reference plots in terms of abundance. This apparently contradicting result indicates that restoration success needs to be evaluated using multiple variables (Ruiz-

Jaén and Aide, 2006; Gatica-Saavedra et al., 2017).

Generally, the development of temperate natural forest after a disturbance can be classified into four stages: establishment, thinning, transition, and climax (Oliver, 1980; Peet and Christensen, 1987; Zipperer et al., 1997). The present state of the restoration plots appears to be at the thinning stage in which canopy closure and poor understory light condition inhibits recruitment of overstory tree species (Oliver, 1980; Ito et al., 2008). In natural forests, shade-tolerant species can establish and grow slowly under the canopy, resulting in species diversity and vertical stratification (Oliver, 1980) by niche differentiation along the vertical gradient of light availability (Ishii et al., 2013). In the restoration plots of the present study, despite planting various species, shade-tolerant understory trees and shrubs had decreased as a result of high mortality (Table 3). Recruitment can occur if there is mature forest nearby that can act as a seed source for shade-tolerant species (Azuma et al., 2014), whereas isolation from source forests hinders natural recruitment and decreases species diversity and community composition (Crouzeilles and Curran, 2016; Goosem et al., 2016). Because the rate of self-thinning of overstory trees is slow and there are no mature forests near Expo '70 Park, abundance and diversity of understory species are likely to continue decreasing in the restoration plots, resulting in single-layered forest structure similar to those of monocultures and even-aged forests. Our results suggest that planting various species simultaneously at high density may prevent niche separation and resource partitioning between overstory and understory species, resulting in forests with single-layered canopy structure, low productivity and biodiversity (Ishii et al., 2004, 2013; Pretzsch, 2014). Such forests are also vulnerable to disturbances (Felton et al., 2010; Pretzsch, 2014).

Management such as thinning of overstory trees is effective to prevent mortality of understory trees and revert vegetation succession of restoration forest toward mature, natural forest (Hotta et al. 2015). In the restoration site in Kobe City, thinning by volunteers achieved stand structure with vertically well-developed canopy, whereas in the stand without management the canopy became single-layered like in Expo ’70 Park (Hotta et al. 2015). Regeneration of the overstory trees is also important for long-term sustainability of the forest. At Expo ’70 Park, a gap creation experiment, mimicking gap dynamics of natural forests, was conducted in some parts of restored forest. In these experimental plots, many seedlings of various species emerged in the first year (Nakamura et al., 2005). However, as years passed after gap creation, only a few overstory species such as *Q. glauca* had survived, and invasion of exotic species such as *L. lucidum* was observed (Sasaki, unpublished).

Q. glauca, which had the second largest number of newly recruited trees in this study (Table 3), is a highly shade-tolerant species, and has been reported to increase markedly in isolated urban forests (Tabata et al., 2007). *L. lucidum* is a bird-dispersed exotic species that has been found to invade isolated urban forests and dominate over the native *L. japonicum* (Ishii and Iwasaki, 2008). It is shade-tolerant, able to establish in the understory of mature forests, but can grow quickly under the open canopy (Aragon and Groom 2003; Nesom 2009). In this study, many *L. lucidum* seedlings emerged in small gaps in the plots dominated by *Q. phillyraeoides* and *C. camphora*. *T. fortunei*, which was newly recorded in this site, is also a bird-dispersed exotic species. It is highly shade tolerant and has been found to spread rapidly in the understory of urban forests preventing regeneration of native species (Hagiwara, 1979; Ishii and Iwasaki, 2008; Ishii et al., 2016). These trends are typical of urban forests of warm-temperate Japan and could potentially lead to a plagiosere (*sensu* Tansley 1935) where exotic species dominate the vegetation inhibiting regeneration and growth of native species (e.g., Tojima et al., 2004). Therefore, monitoring and management to prevent invasion by exotic species is necessary to promote regeneration of native species and achieve the restoration goal of sustaining natural-forest-like conditions in urban forests.

Conclusions

This study revealed that, without additional management intervention, restoration of mature, late-successional forest cannot be achieved by simultaneous planting of native species. Although one-time planting of late-successional species is practiced in many restoration sites across Japan (Miyawaki 2004), such practices are unlikely to lead to natural forest stand structure because, in natural forest, regeneration of late-successional species occurs gradually over many years. Our results illustrate the importance of continuous management, such as thinning to mimic mortality of overstory trees and achieve multi-layered stand structures while also ensuring the availability of native seed sources for ensuring restoration success. Additional planting of native species may be necessary in highly developed areas without nearby natural forest that serve as potential seed sources. Furthermore, we showed that restoration success needs to be evaluated using multiple criteria. Comparison of stand structural features with a reference forest can be especially revealing to understand the chronological direction of vegetation change of the restored forest. Long-term monitoring and management referencing natural forest structure and dynamics are necessary to sustain urban forests into the future.

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Figure captions

Fig. 1. Location of the study sites: Expo '70 Commemorative Park (restoration site) and Taisanji Temple (reference forest), Japan (a) and the field-survey plots (b). Aerial photographs (taken in 2007) were downloaded from Geospatial Information Authority of Japan website.

Fig. 2. Cluster analysis indicated that the study plots can be divided into four groups according to their respective dominant species. Restoration plots dominated by *Castanopsis cuspidata* (E3, E13, E9) were grouped with the reference plots.

Fig. 3. Frequency distributions of abundance and basal area by tree height for each plot group in 2014. Restoration plots had higher tree density with more overstory trees and larger basal area than reference plots.

Fig. 4. Chronological change of the relationship between tree density and mean stem volume of overstory trees for the plot groups in restoration and reference sites. Diagonal line indicates the self-thinning line (slope = −3/2) according to Yoda et al. (1963), drawn through the data point of reference plots in 2003. Reference plots are following the self-thinning line, while tree density is increasing in restoration plots indicating lack of self-thinning.

Fig. 5. Nonmetric multidimensional scaling (nMDS) ordination plots for Bray-Curtis distances based on basal area (a) and abundance (b) of plots dominated by *C. cuspidata* (Group 1). Species composition of restoration plots are changing in the direction of reference plots along axis 1 of the basal-area-based ordination. However, abundance-based ordination indicated that restoration plots had diverged away from reference plots.

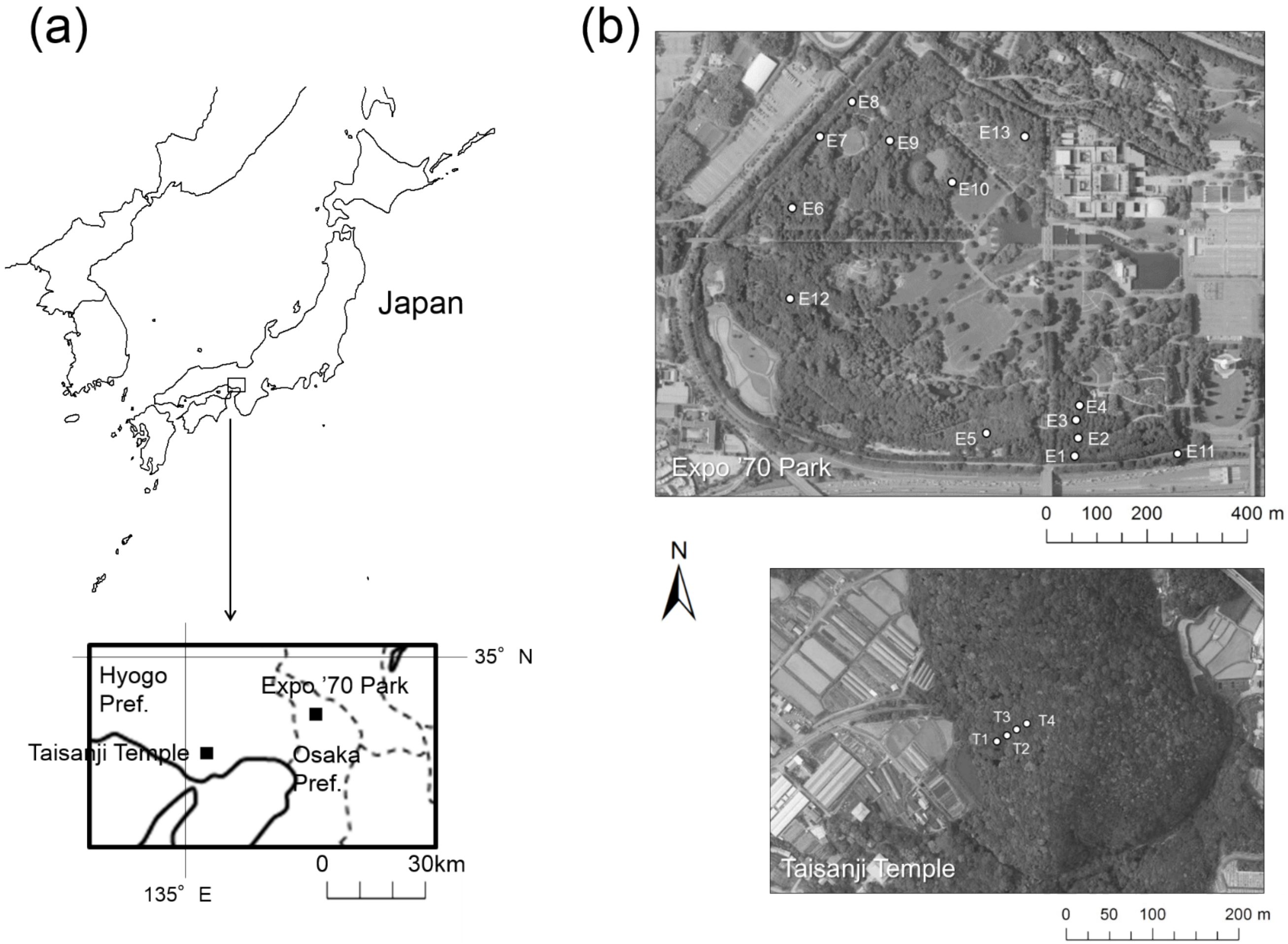
(a)
Japan
Hyogo Pref.
Expo '70 Park
Taisanji Temple
Osaka Pref.
35° N
135° E
0
30km
(b)
E1
E2
E3
E4
E5
E6
E7
E8
E9
E10
E11
E12
E13
Expo '70 Park
0 100 200 400 m
N
T1
T2
T3
T4
Taisanji Temple
0 50 100 200 m

Fig. 1.

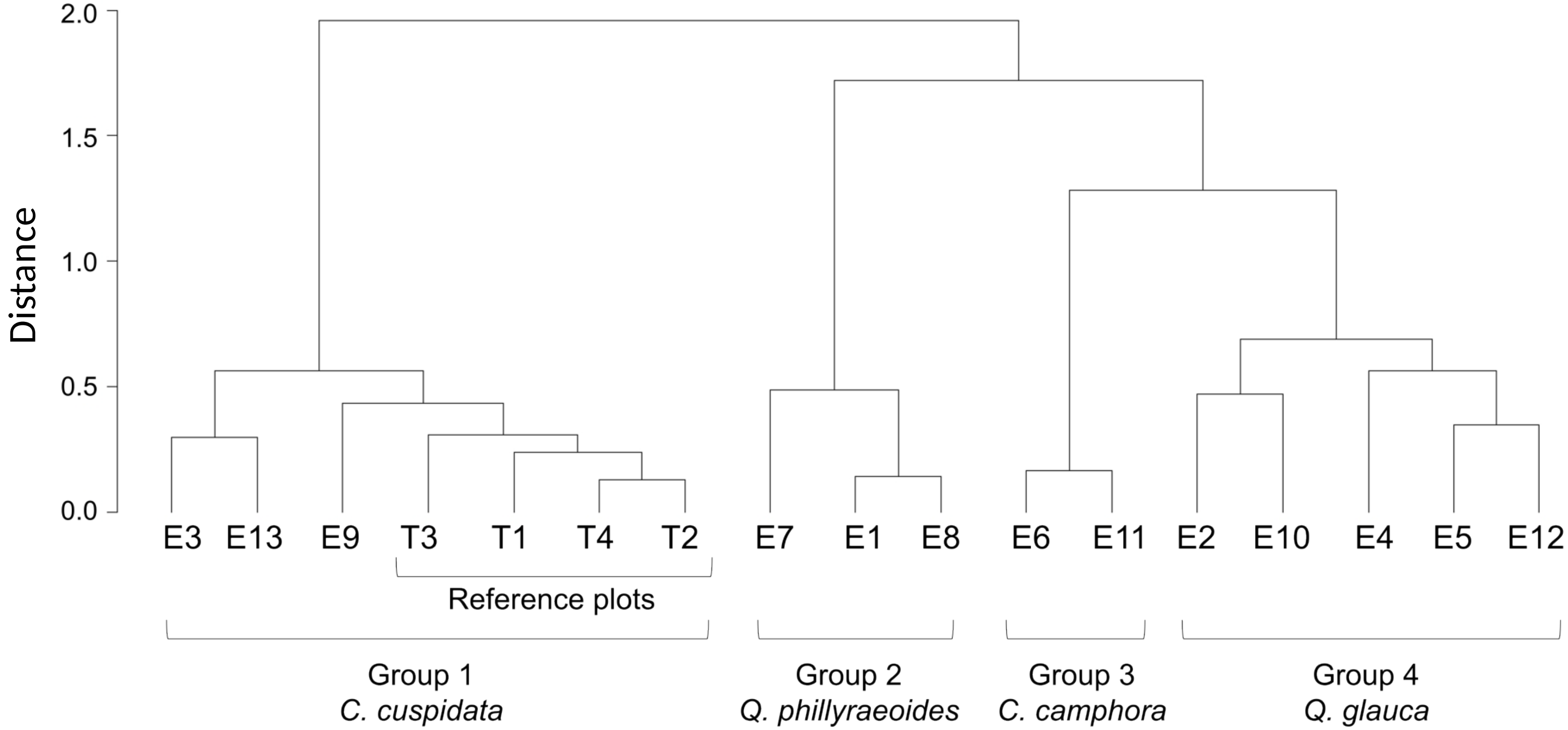
Distance
2.0
1.5
1.0
0.5
0.0
E3
E13
E9
T3
T1
T4
T2
E7
E1
E8
E6
E11
E2
E10
E4
E5
E12
Reference plots
Group 1
C. cuspidata
Group 2
Q. phillyraeoides
Group 3
C. camphora
Group 4
Q. glauca

Fig. 2.

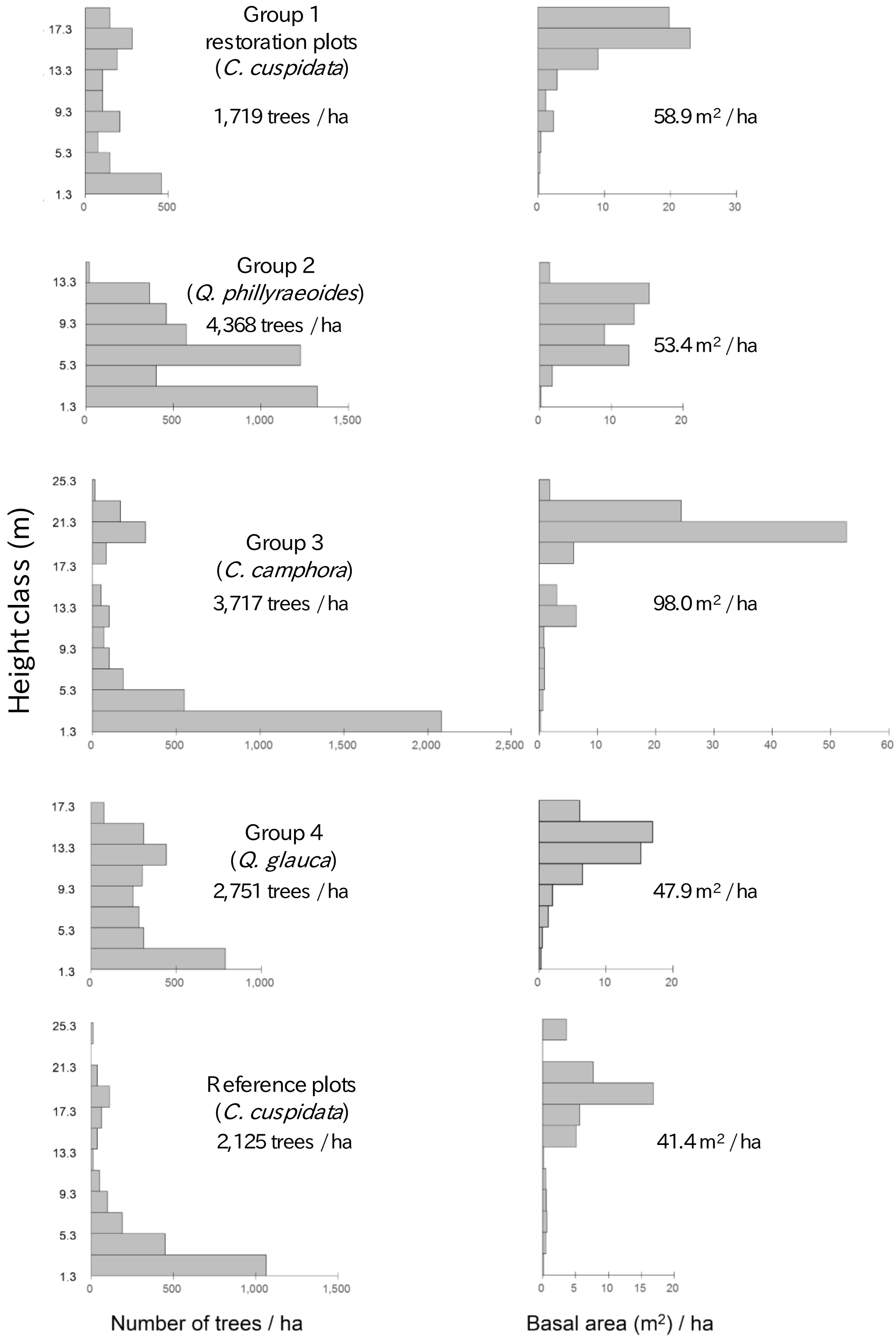
Height class (m)
Group 1
restoration plots
(C. cuspidata)
1,719 trees / ha
58.9 m2 / ha
Group 2
(Q. phillyraeoides)
4,368 trees / ha
53.4 m2 / ha
Group 3
(C. camphora)
3,717 trees / ha
98.0 m2 / ha
Group 4
(Q. glauca)
2,751 trees / ha
47.9 m2 / ha
Reference plots
(C. cuspidata)
2,125 trees / ha
41.4 m2 / ha
25.3
21.3
17.3
13.3
9.3
5.3
1.3
0
500
1,000
1,500
2,000
2,500
0
5
10
15
20
30
40
50
60
Number of trees / ha
Basal area (m2) / ha

Fig. 3.

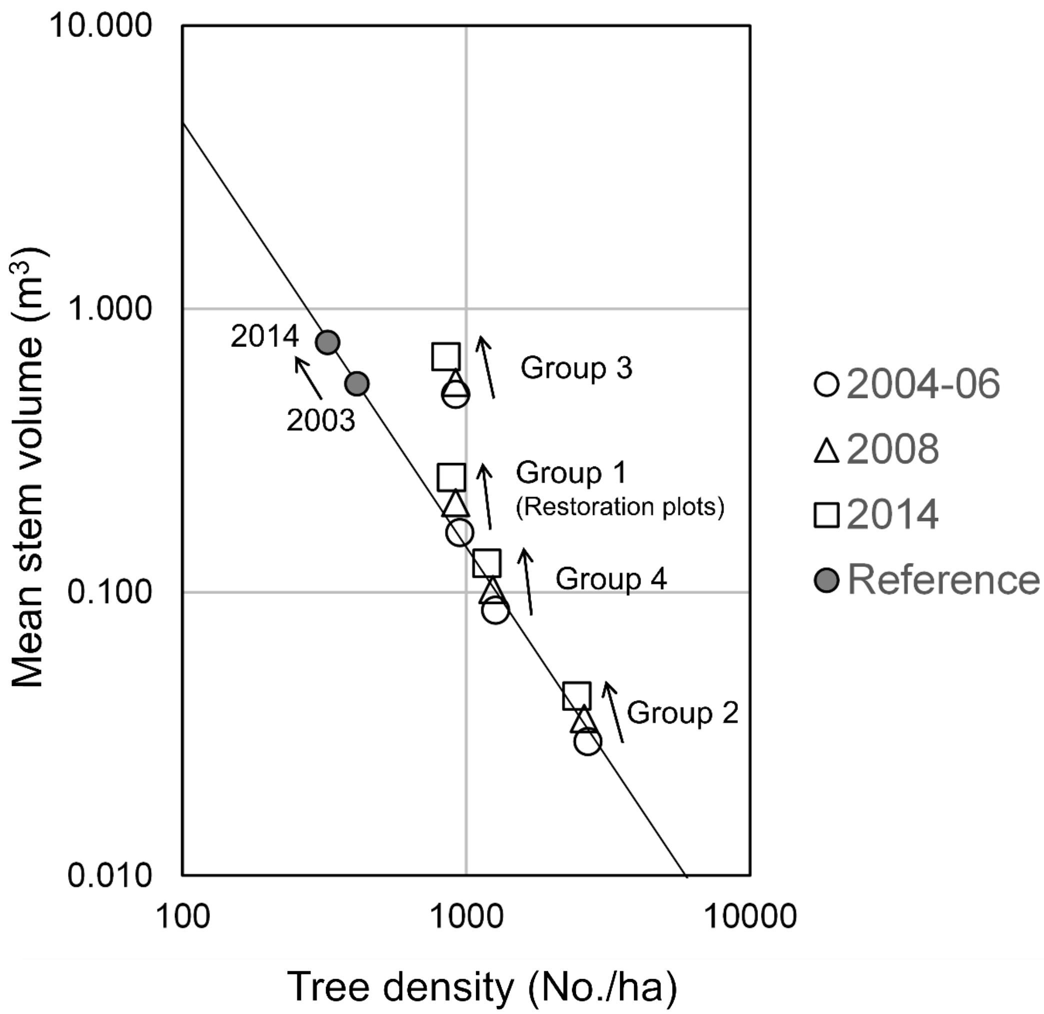
10.000
1.000
0.100
0.010
100
1000
10000
Mean stem volume (m³)
Tree density (No./ha)
2014
2003
Group 3
Group 1
(Restoration plots)
Group 4
Group 2
2004-06
2008
2014
Reference

Fig. 4.

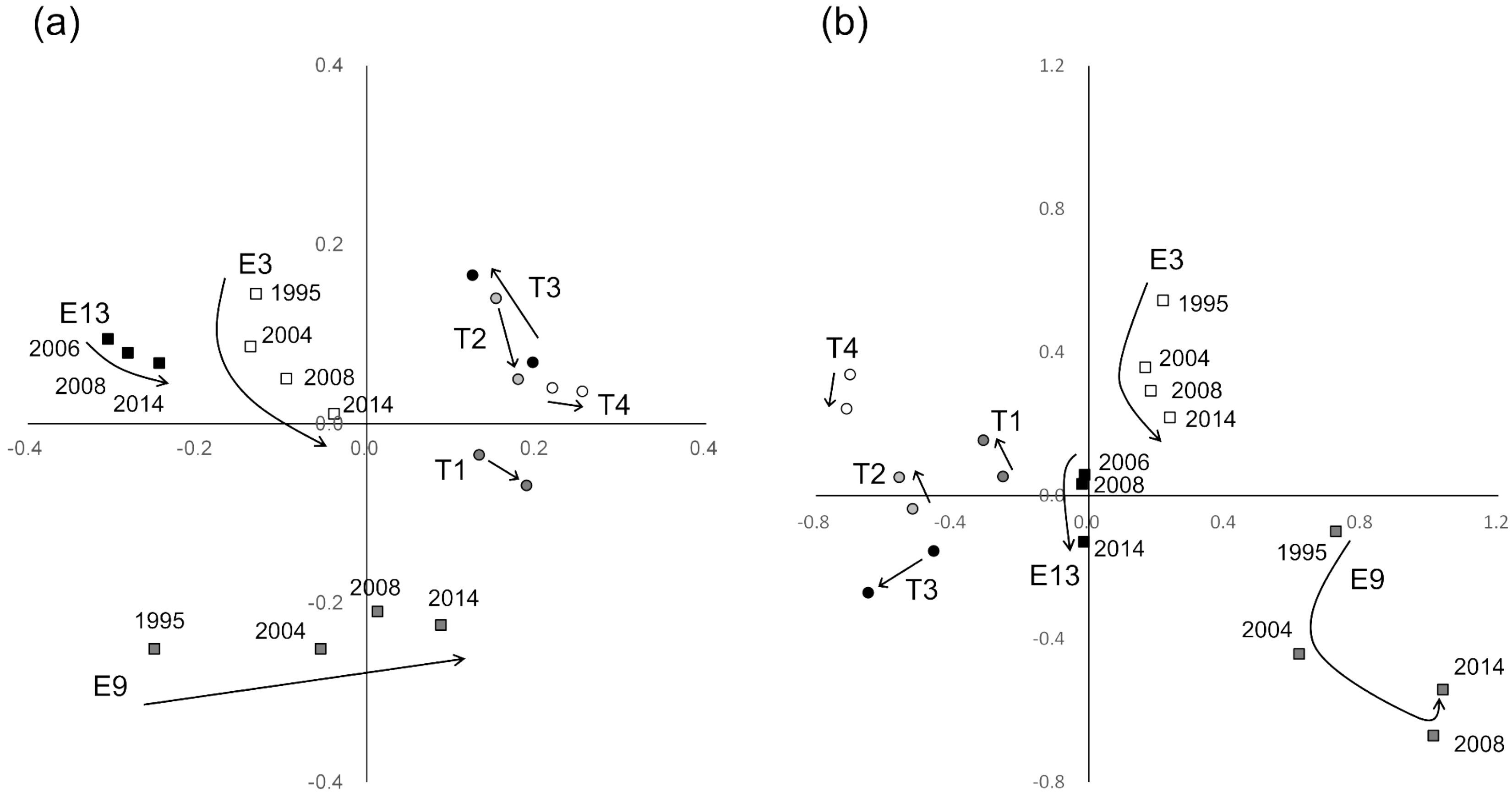
(a)
E13
2006
2008
2014
E3
1995
2004
2008
2014
T3
T2
T4
T1
1995
2004
2008
2014
E9
0.4
0.2
0.0
-0.2
-0.4
-0.4
-0.2
0.0
0.2
0.4
(b)
E3
1995
2004
2008
2014
T4
T1
T2
2006
2008
2014
E13
T3
1995
E9
2004
2014
2008
1.2
0.8
0.4
0.0
-0.4
-0.8
-0.8
-0.4
0.0
0.4
0.8
1.2

Fig. 5.

Table 1. Area and dominant species of the in the study plots.

Plot	Area (m^2)	Dominant tree species
Restoration site		
E1	150	*Quercus phillyraeoides*
E2	210	*Quercus glauca*, *Quercus serrata*, *Machilus thunbergii*
E3	225	*Castanopsis cuspidata*
E4	225	*Quercus glauca*
E5	225	*Quercus glauca*, *Cinnamomum camphora*
E6	225	*Cinnamomum camphora*
E7	150	*Quercus phillyraeoides*
E8	240	*Quercus phillyraeoides*
E9	225	*Castanopsis cuspidata*
E10	180	*Quercus glauca*, *Machilus thunbergii*
E11	375	*Cinnamomum camphora*
E12	225	*Quercus glauca*
E13	225	*Castanopsis cuspidata*
Reference site		
T1	200	*Castanopsis cuspidata*
T2	200	*Castanopsis cuspidata*
T3	200	*Castanopsis cuspidata*
T4	200	*Castanopsis cuspidata*

Table 2. Species composition of the four restoration plot groups and reference plots.

Species	Number of trees / ha	Basal area (m^2 / ha)
Group 1 restoration plots		
Castanopsis cuspidata	296	30.68
Machilus thunbergii	163	9.55
Celtis sinensis	89	4.43
Quercus glauca	104	4.08
Cinnamomum camphora	44	3.71
Ligustrum japonica	163	1.74
Camellia japonica	133	1.14
Photinia glabra	104	0.32
Others (19 species)	622	3.28
Total	1719	58.93
Group 2		
Quercus phillyraeoides	2452	47.91
Camellia japonica	747	2.65
Myrica rubra	38	1.32
Ternstroemia. gymnanther	115	0.67
Ligustrum lucidum	670	0.01
Others (12 species)	345	0.85
Total	4368	53.41
Group 3		
Cinnamomum camphora	633	86.44
Quercus glauca	617	3.93
Castanopsis cuspidata	83	3.78
Ligustrum japonica	33	1.03
Camellia japonica	183	0.59
Cinnamomum tenuifolium	167	0.54
Aphananthe aspera	783	0.38
Aucuba japonica	300	0.08
Ligustrum lucidum	483	0.05
Others (16 species)	433	1.17
Total	3717	97.99

Species	Number of trees / ha	Basal area (m^2 / ha)
Group 4		
Quercus glauca	789	21.55
Cinnamomum camphora	150	6.11
Machilus thunbergii	197	5.77
Quercus serrata	56	4.06
Castanopsis cuspidata	56	1.95
Ligustrum japonica	338	1.32
Quercus phillyraeoides	56	1.26
Lithocarpus edulis	103	1.16
Ulmus parvifolia	38	1.16
Pittosporum tobira	244	0.11
Others (26 species)	723	4.42
Total	2751	48.86
Reference plots		
Castanopsis cuspidata	237.5	35.02
Quercus glauca	25	1.60
Ilex rotunda	125	1.56
Camellia japonica	600	1.01
Ligustrum japonica	175	0.33
Cleyera japonica	175	0.25
Eurya japonica	150	0.12
Aucuba japonica	275	0.02
Others (12 species)	362.5	1.47
Total	2125	41.38

Table 3. Abundance of the main tree species in each survey year and their loss rates until 2014.

Restoration plots	Number of trees (/ha)		
	2004-2006	2014	Loss rate
Quercus phillyraeoides	561	508	0.095
Quercus glauca	303	288	0.050
Cinnamomum camphora	224	201	0.102
Camellia japonica	182	159	0.125
Ligustrum japonicum	178	159	0.106
Machilus thunbergii	106	95	0.107
Castanopsis cuspidata	91	76	0.167
Pittosporum tobira	68	34	0.500
Photinia glabra	61	49	0.188
Ternstroemia gymnanther	57	38	0.333
Others	531	421	0.207
Overstory trees	1608	1452	0.097
Understory trees	618	504	0.184
Shrubs	137	72	0.472
Evergreen	2211	1907	0.137
Deciduous	152	125	0.175
Total	2363	2029	0.141

Reference plots	Number of trees (/ha)		
	2003	2014	Loss rate
Camellia japonica	413	388	0.061
Castanopsis cuspidata	213	200	0.059
Ligustrum japonicum	188	175	0.067
Cleyera japonica	163	163	0.000
Eurya japonica	150	150	0.000
Ilex rotunda	100	88	0.125
Cinnamomum tenuifolium	63	38	0.400
Dendropanax trifidus	50	50	0.000
Others	238	163	0.316
Overstory trees	463	375	0.189
Understory trees	888	850	0.042
Shrubs	225	188	0.167
Evergreen	1538	1400	0.089
Deciduous	38	13	0.667
Total	1575	1413	0.103

All listed species, except Q. *phillyraeoides* and *C. camphora*, are late-successional, slow-growing, native evergreen trees. *Q. phillyraeoides* and *C. camphora* are shade-intolerant, fast-growing evergreen trees.

Table 4. Numbers of newly recruited juvenile trees (<1.3m height) in 2014.

Restoration plots

Species	Number of trees (/ha)
Ligustrum lucidum	227
Quercus glauca	143
Aphananthe aspera	140
Pittosporum tobira	77
Camellia japonica	52
Dendropanax trifidus	45
Aucuba japonica	42
Nandina domestica	31
Ligustrum japonicum	24
Ilex rotunda	21
Castanopsis cuspidata	21
Elaeagnus pungens	14
Cinnamomum tenuifolium	14
Elaeocarpus sylvestris var. *ellipticus*	10
Photinia glabra	10
Others (16 species)	66
Total	940

Reference plots

Species	Number of trees (/ha)
Aucuba japonica	188
Camellia japonica	150
Photinia glabra	50
Ilex rotunda	25
Litsea lancifolia	25
Cinnamomum tenuifolium	25
Castanopsis cuspidata	25
Ilex integra	13
Elaeagnus glabra	13
Quercus glauca	13
Sambucus sieboldiana var. *pinnatisecta*	13
Callicarpa japonica	13
Cleyera japonica	13
Total	563