Received: 31 July 2024 | Revised: 28 December 2024 | Accepted: 18 January 2025

DOI: 10.1111/cobi.70093

Conservation Biology

CONTRIBUTED PAPER

Relative effects of sacred forests and protected areas on forest conservation and structure in Japan

Jonathan Peereman[1,2] | Masahiro Aiba[3] | Teng-Chiu Lin[2]

[1]SPHERES Research Unit, University of Liège, Arlon, Belgium

[2]Department of Life Science, National Taiwan Normal University, Taipei, Taiwan

[3]Graduate School of Agricultural and Life Sciences, The University of Tokyo, Bunkyo-ku, Japan

Correspondence
Teng-Chiu Lin, Department of Life Science, National Taiwan Normal University, No. 88 Section 4, Ting-Chow Road, Taipei 11677, Taiwan. Email: tclin@ntnu.edu.tw

Article impact statement: Sacred forests may be integrated into conservation projects to bridge the gaps between legally protected areas in lowlands and urban areas.

Funding information
National Science and Technology Council, Grant/Award Numbers: 113-2740-M-003 -001 -, 113-2740-M-003 -002 -, 113-2114-M-003 -001 -

Abstract
Sacred forests, found on all inhabited continents, are globally recognized for their biodiversity and conservation value and their role in cultural landscapes. Assessments of the effects of forest sacredness have mostly focused on small regions and are based on diverse methods. We used geographic information provided by the public and datasets derived from remote sensing to locate and examine changes in shrine and temple forests across Japan. We compared the aboveground vegetation structure and annual rate of forest loss of these sacred forests with their surrounding nonsacred forests and with legally protected areas. We tested whether these differences were comparable between urban and rural areas and between shrine and temple forests. Based on a sample of 35,899 sacred forests, sacred forests had higher canopy height (mean [SD] = 15.5 m [0.02] vs. 15.2 m [0.01], t test, $p < 0.001$) than the surrounding nonsacred forests. An annual rate of sacred forest loss was 50% lower than other forests outside legally protected areas (0.07%/year and 0.13%/year, respectively) from 2000 to 2022. Sacred forests had forest loss rates comparable to strictly protected forests (0.05%/year for International Union for Conservation of Nature categories Ia–III), and sacredness and legal protections cumulatively reduced forest loss. The protection sacred forests offer was confirmed across urban and rural areas of Japan. Large-scale assessments of sacred forests' efficiency are now possible based on geographic information provided by the public. We found that sacred forests were as effective as strict legally protected areas at preventing forest loss and that shrine and temple forests were important features of lowland urban and rural landscapes of Japan, ranging from boreal to subtropical forests. These sites need to be further considered in national or even international conservation frameworks.

KEYWORDS
forest conservation, Japan, primary forest, protected area, remote sensing, sacred forest

INTRODUCTION

Sacred forests as conservation areas

Sacred forests provide nature experiences for people (Díaz et al., 2018) that have spiritual value (Rots, 2019; Tatay & Merino, 2023), as recognized in the framework of the Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services (Díaz et al., 2015). These spiritual contributions have tangible (place of worship, source of material) and intangible components (e.g., aesthetic, recreational, and inspirational values). Like other natural sites with important spiritual value (sacred natural sites [SNSs]) (Verschuuren et al., 2010), sacred forests are typically associated with cultural practices that may include restrictions, such as limited access and resource extraction. These restrictions may explain the differences in structure and biodiversity between sacred forests and their surrounding forests (Mgaya, 2023; Onyekwelu et al., 2022; Yang et al., 2021; Zeng, 2018). Sacred forests generally have less forest cover loss than other forests (Daye & Healey, 2015; Manabe et al., 2003; Onyekwelu et al., 2022; Sahle et al., 2021). Moreover, some sacred forests have higher biodiversity than nonsacred forests (Avtzis et al., 2018; Frascaroli et al., 2016; Irvine et al., 2019; Mgumia & Oba, 2003). Nevertheless, sacred forest management practices vary widely even within the same region (Marini Govigli et al., 2020). For example, management practices can include the introduction of new species (Barrow, 2010) or removal of invasive species (Ishii et al., 2016). Thus, higher biodiversity is not always higher in sacred forests (e.g., region and taxa dependent) (Sullivan et al., 2024) in comparison with nonsacred locations (Kamdem, 2010; Manabe et al., 2011), which likely leads to high variation in their conservation value across sites.

Current interest in and limitations of large-scale studies

SNSs, particularly sacred forests, are seen as an increasing source of nature experiences for people (Sahle et al., 2021) and an important conservation tool (Kovács et al., 2022; Melaku et al., 2023; Nakadai, 2023; Tatay & Merino, 2023; Verschuuren et al., 2021; Zannini et al., 2021) because they occur on all inhabited continents (Dudley et al., 2005; Wild & McLeod, 2008). Sacred forests offer alternative ways to maintain forest cover outside protected areas because their distribution is not limited to protected areas (Bhagwat & Rutte, 2006; Coggins & Chen, 2022). Also, their proximity to human communities often makes them rare natural remnants that host local biodiversity and provide other important ecosystem services (Mori et al., 2025), such as air quality regulation, and a range of cultural services besides the spiritual, including aesthetic features and recreation (Alohou et al., 2017; Daniel et al., 2012; Hashimoto et al., 2015; Ishii et al., 2010). The involvement of local communities in their management and the importance of Indigenous areas in biodiversity conservation further highlight the role of sacred forests in forest conservation (Garnett et al., 2018; Roux et al., 2022; Verschuuren et al., 2021). Nevertheless, most assessments of sacred forests' structure (height, age) and dynamics (forest surface area change) have been of a limited number of forest (Manabe et al., 2003; Onyekwelu et al., 2022; Yang et al., 2021; but see Anderson et al., 2005; Shen et al., 2015; Zannini et al., 2022). As a result, it is unclear how effective sacred forests are for regional-to-global scale forest conservation in comparison with legally protected areas.

The analysis of the effect of sacred forests on forest conservation at broad scales is constrained by the lack of georeferenced data on their locations. Well-known sacred forests under other types of protection (e.g., World Heritage) can be found in the World Database of Protected Areas (WDPA) (UNEP-WCMC & IUCN, 2023). However, current datasets do not represent the large number of sacred forests that occur outside protected areas (Samakov & Berkes, 2017; Techera, 2010). As a result, researchers have used commercial datasets or literature review to locate places of worship associated with sacred forests (Frascaroli et al., 2019; Manabe et al., 2011). Novel approaches to geographic information may allow the study of sacred forests across broad regions. Current efforts in Indigenous-led geographic information (Olson et al., 2016) and geographic information collected by volunteers (volunteered geographic information [VGI]) are improving the availability of land-use information (Spielman, 2014). Among freely accessible VGI datasets, OpenStreetMap (OSM) appears to be the most complete; it includes roads and building or surface type and land use (Neis & Zielstra, 2014; Vargas-Munoz et al., 2021).

Shrine and temple forests of Japan

Across different regions, Japan appears to be particularly suitable to an assessment of the effect of sacredness on forest conservation. First, Shinto and Buddhism are the dominant religions of Japan, and they are tightly linked to forested environments to which they associate spiritual and aesthetic values (Ishii et al., 2010). The importance of sacred forests in biodiversity and culture conservation in Japan has been recognized, and they are the focus of several site-based studies (Ishii et al., 2016; Nagaike, 2012; Sakamoto et al., 1989a, 1989b).

Second, individual sacred forests are small, but collectively they cover a large area of approximately 208,000 ha across Japan (Omura, 2004; Verschuuren et al., 2010), nearly 1% of the forested area of the country (Egusa et al., 2020). They are well represented across the islands, from 20°N to more than 45°N, and cover boreal forests to subtropical rain forests. They are associated with Shinto and Buddhist places of worship (i.e., shrines and temples, respectively) and other local religious sites (Chen, 2022; Ishii et al., 2005).

Third, the presence of the 2 religions associated with forested environments in the same region offers the opportunity to assess the effects of practices associated with different religions on forest dynamics. In Shinto, an animistic religion, trees around shrines are often regarded as the deities themselves, or at least essential in their rituals (Omura, 2004; Rots, 2015). Shinto sites tend to be more associated with natural features than Buddhist temples (Fujita & Kumagai, 2007). Although trees around Buddhist temples are important in creating the solemn atmosphere, their connection to the religion is weaker than in the case of Shinto (Ishii et al., 2010).

Finally, it is now possible to locate the forests with VGI data because there have been extensive contributions on shrine and temple locations. Although there is no official dataset on Japanese sacred forests, OSM contains precise data on the location and type of buildings in Japan. Since 2011, contributions to OSM have increased, identifying the location and land-use type across different prefectures (Herfort et al., 2023; Seto, 2022). In addition, sacred forests are widespread throughout Japan, indicating that the role of sacredness in forest protection is not limited to a particular forest type or a specific climate.

Japanese sacred forests are managed very differently (Hattori et al., 2010; Ishii et al., 2016; Rots, 2019). The age, status, and size of the sacred forests also vary greatly (Hattori et al., 2010; Ishii et al., 2010; Miura et al., 2001). The small size of some of these forests means they may be affected by edge effects (Murakami & Morimoto, 1999). Edge effects lead to lower tree cover (Dantas de Paula et al., 2016), simplified composition (Suzuki et al., 2013), and higher mortality in small relative to large forest stands (Murcia, 1995). In addition, the canopy height can be affected by sacred forest management and by the presence of *Cryptomeria japonica* D. Don, the tallest native tree species that is widely distributed across Japan (~13% of the land area [Matsumoto et al., 2006]). This species is extensively planted in the southwest and found growing naturally on all the main islands except Hokkaido (Miyajima, 1973; Takahara et al., 2023). Hence, the variation in these forests' management and characteristics (e.g., size, age) likely leads to variable growth conditions, but it is unclear how their height and cover dynamics compare with nearby nonsacred forests outside the bounds of protected areas.

We sought to assess the effects of forest sacredness on forest loss and structure (height) throughout Japan. To do so, we compared sacred forests with nearby nonsacred forests, as is commonly done in assessments of sacredness effects (see meta-analysis by Sullivan et al. [2024]). Our work is one of the first attempts to test the applicability of current VGI products for large-scale monitoring of SNSs. To this end, we used a recent OSM dataset for Japan to identify sacred forest locations to determine the change in forest cover in sacred forests relative to the change in forest cover in nonsacred forests surrounding sacred forests and in forests within protected areas; compare forest height between sacred forests and surrounding nonsacred forests; and assess how the differences in forest height and area loss between sacred forests and surrounding nonsacred forests vary between rural and urban areas and between shrine and temple forests.

METHODS

Studied area and sacred sites

We located the shrine and temple locations across Japan with OSM data from the geofabrik.de website (accessed 7 November 2023). We used the osm-filter Python script to filter all Shinto and Buddhist sites by narrowing the dataset to amenities registered as a place of worship with religion specified as Shinto or Buddhist. Other religions in Japan are not closely related to sacred forests (e.g., churches) or are not clearly identified in the OSM dataset.

Datasets

We used datasets related to vegetation, land use, and topography to characterize the sacred sites and the surrounding areas. The datasets were selected based on their availability for years close to 2023, fine spatial resolution (≤30 m for biotic parameters), and coverage of the entire studied latitudinal range. The vegetation height was characterized using the JAXA Earth Observation Center canopy height dataset (CH) 24.09 for 2016–2022 (Li et al., 2024). The CH is based on radar, LiDAR, and multispectral data and provides an estimation of the forest height (in meters) at a 10-m spatial resolution.

To estimate forest loss rates from 2000 to 2022, we used Global Forest Change (GFC) 1.10 (Hansen et al., 2013), which identifies forest location and cover (%) in 2000 and annual forest state up to 2022 at 30-m spatial resolution based on multispectral data. The mean annual forest loss rate (percent per year) was computed as the mean annual rate of change (Puyravaud, 2003) multiplied by −100.

We used JAXA High-Resolution Land-Use and Land-Cover 21.11 (LULC) map of Japan for 2020 (10-m spatial resolution) to identify the 5 forest types: deciduous broadleaf forests, deciduous needleleaf forests, evergreen broadleaf forests, evergreen needleleaf forests, and bamboo. The forest types do not provide information on the floristic composition because composition changes with latitude and elevation in Japan for each specific forest type (Numata et al., 1972). However, we identified *C. japonica* plantations based on the dataset from the Biodiversity Center of Japan (BCJ) and urban and rural areas with the Global Human Settlement Layers (GHSL) developed by JRC for the year 2020 (Schiavina et al., 2023). The GHSL has 7 different classes of urbanization at 1-km spatial resolution from low-density rural (<50 inhabitants/km) to urban center (> 1500 inhabitants/km). We regrouped these classes into 2 categories: urban (suburban or peri-urban to urban center) and rural (low density to rural cluster).

We identified protected areas across Japan from WDPA and BCJ datasets. Although the WDPA provides an extensive representation of current protected areas, the BCJ dataset identifies a few other protected locations. We differentiated between strictly and leniently protected areas in the WDPA database based on the IUCN management categories: Ia–III, strict level; IV–VI, lenient (Leberger et al., 2020). In Japan, categories Ia–III include wilderness areas, nature conservation areas, and some natural monuments and national parks. Categories IV–VI include habitat protection zones, national wildlife protection areas, quasi-national parks, and other national parks (Hiwasaki, 2005; Jones & Kobayashi, 2021). We used all protected areas to compare the effects of sacredness and legal protection, whereas only the strict and lenient categories were used to analyze the effect of protection strictness in comparison with sacredness (details in "Analyses"). Finally, site elevation was obtained from the digital elevation model NASADEM (NASA JPL, 2020). The CH, GFC, and GHSL datasets were extracted and resampled to 10-m resolution to match the LULC spatial resolution from Google Earth Engine (Gorelick et al., 2017).

Identification of sacred forests and their surrounding nonsacred forests

We exported the 2 types of data, point and polygon, based on shrine and temple locations derived from OSM. Based on a random sampling of 500 shrines and 500 temples and verification with Google Maps and Street View, we found that <4% of the points could have erroneous coordinates. To identify sacred forests, we demarcated a radius of 0.00055° (~55 m) around all shapes to delineate an area where the precincts' forest would be located. This radius also accounted for OSM product's slight location shift (Hecht et al., 2013). These locations are hereafter called sacred sites and sacred forests if their vegetation cover included forests according to the JAXA LULC. Our definitions of *sacred site* and *sacred forest* are based on the proximity of a Buddhist or Shinto place of worship and not on a precise boundary of the sacred site. Hence, around each sacred site, we then created a 0.003° (~300-m) buffer that was excluded from the analyses. This area could be a part of the sacred forest that was not properly delineated by the OSM data and our

55-m radius. Moreover, it represented the area with potential spillover effects; people may avoid disturbing forests close to the sacred sites. Finally, to identify nearby nonsacred forests, we delineated a 0.05° radius around the sacred sites (~4 km without the excluded spillover effect radius) where we identified the sections belonging to protected areas and from which we removed the other sacred sites and their radii. The choice of a 0.05° radius was based on the importance of comparing areas in the same landscape with similar climates. This section is hereafter called *surrounding area*, and its forested part is hereafter called *surrounding nonsacred forests*.

Processing

For each sacred site, we extracted data related to vegetation: mean canopy height and standard deviation, forest type or types, number of pixels belonging to each of the 5 forest types, forest extent, percentage of forest cover in year 2000, and mean annual forest loss from 2000 to 2022. We also identified urban and rural sites based on information in the GHSL dataset, the level of protection (none, lenient, or strict), and the mean elevation. We identified sacred forests and extracted information about their surrounding nonsacred forests (i.e., controls). Based on pixels in each surrounding nonsacred forest, we computed the mean canopy height and its standard deviation, number of pixels belonging to each of the 5 forest types, forest extent, percentage of forest cover in 2000, and mean annual forest loss. Prior to surrounding nonsacred forest data extraction, we masked areas in other sacred forests and their radii and identified the sections in protected areas. Lastly, we extracted the number of pixels belonging to each of the 5 forest types, forest extent, mean percent cover in 2000, mean annual forest loss from 2000 to 2022, and elevation in each protected area from the WDPA database and for the entire forested surface of Japan in 2020 according to JAXA LULC. No information on forest land use, besides plantations, was extracted. Thus, we did not address the effect of various land-use regulations. The main steps of this study are summarized in Figure 1.

Analyses

The analysis was carried out in R (R Core Team, 2022). We assessed the distribution of shrines and temples across urban and rural areas and their inclusion in protected areas. For sacred forests and surrounding nonsacred forests, we retrieved the percentage of each of the vegetation types identified based on the LULC dataset. A forest type was considered dominant if that type covered ≥75% of the forest. To assess the importance of sacred forests in old-growth evergreen broadleaf forest conservation, we used a subset of sacred forests dominated by evergreen broadleaf (i.e., primary forests) and compared proportions of forest types in their landscapes. We also compared the altitudinal range of the sacred forests with the altitudinal ranges of all forests of Japan and of all forests in protected areas.

We assessed the role of sacredness in forest height and forest cover loss by comparing sacred forests outside protected areas with the respective surrounding nonsacred forests outside protected areas. We compared the 2 locations for the mean canopy height, canopy height standard deviation, and the annual forest loss rates with paired t tests. We repeated the canopy height and annual forest loss rate comparisons based on areas without *C. japonica* plantations. In addition, we repeated the comparisons of sacred forests and surrounding nonsacred forests based on a subset of sites for which each forest pair shared the same dominant forest type to minimize the effect of this factor on our results.

We compared the mean annual forest loss rates in sacred forests dominated by different forest types outside protected areas. For sites with a dominant forest type, the 5 forest types were compared using Herberich et al.'s (2010) method for unbalanced group sizes with sandwich and multcomp R packages (Hothorn et al., 2008; Zeileis et al., 2020).

We analyzed the effects of general environmental pressures linked with religious identity on the differences between sacred forests and the surrounding nonsacred forests. First, we computed the difference in mean canopy height (dCH) and forest loss rate (dLoss) between the sacred forests and the surrounding nonsacred forests. We then compared dCH and dLoss between sacred forests close to shrines and those close to temples and between urban areas and rural areas with t tests. We repeated these comparisons for rural and urban subgroups for sacred forests dominated by different types of forest.

To compare the effectiveness of unprotected sacred forests in forest cover protection with the protected areas of Japan, we compared the mean annual rate of forest loss measured across these 2 types of locations with t tests. The IUCN management category is a predictor of the level of forest conservation (Leberger et al., 2020). We compared annual forest loss in protected areas with strict and lenient protection to assess the efficiency of unprotected sacred forests in protecting forest loss. To determine whether sacredness and legal protection had cumulative effects on forest area conservation, we compared the mean annual forest loss rate measured in sacred forests included in protected areas (i.e., protected sacred forests) with the rate measured in the surrounding nonsacred forests also in protected areas.

RESULTS

Distribution of Shinto and Buddhist sites

A total of 57,047 Shinto (32,145 shrines, 61% in cities) and Buddhist (24,902 temples, 76% in cities) sites were identified (Figure 2a). Not all places of worship that are sacred sites were associated with forests. We identified 22,988 shrines (72%) and 12,911 temples (52%) with forest vegetation (Figure 2c). Protected areas contained 5570 of the identified sacred forests (16%), 141 of which were under strict management (0.4%) and 5509 under lenient management (15%).

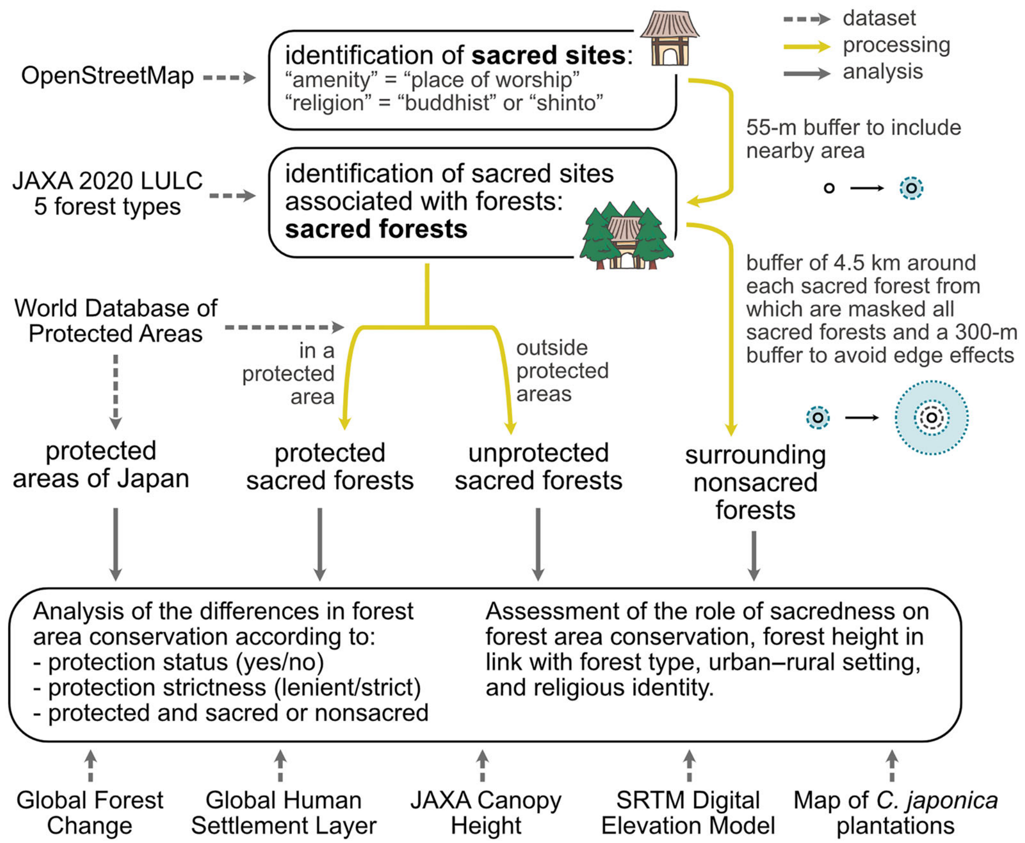

FIGURE 1 Steps used to assess the effects of forest sacredness on forest structure and conservation across Japan (JAXA LULC, Japanese Aerospace Exploration Agency Land-Use Land-Cover; SRTM, Shuttle Radar Topography Mission).

Representation of forest types and elevational range in sacred forests

The sacred forests contained various forest types (Figure 2d). Evergreen needleleaf forests were the most abundant (31.3%), followed closely by deciduous broadleaf (30.1%) and bamboo forests (29.9%). Deciduous needleleaf (2.1%) and evergreen broadleaf forests (6.6%) were not well represented. The most common forest types in sacred forests were different from the most common forest types in all nonsacred forests of Japan and from nonsacred forests in protected areas. In comparison with sacred forests, the entire forested area of Japan had a higher percentage of deciduous broadleaf forest (52.9%) and a lower percentage of bamboo forests (5.5%) (Appendix S1). The other 3 forest types were similarly represented in the country and in sacred forests (3.4%, 5.9%, and 32.3% for deciduous broadleaf, evergreen broadleaf, and evergreen needleleaf forests, respectively). In comparison with protected areas (Appendix S1), sacred forests contained more bamboo dominated forest (29.9% against 11.0% in protected nonsacred forest) and less deciduous broadleaf forests (30.1% against 46.4%). Finally, in comparison with sacred forests, surrounding nonsacred forests had more deciduous broadleaf forest (41.2%) (Appendix S1) and evergreen broadleaf forest (9.0%) but less deciduous needleleaf forest (1.1%) and bamboo (18.2%). The percentage of evergreen needleleaf forests was similar across sacred forests and their surrounding forests (30.5%). Sacred forests dominated by primary evergreen broadleaf forests were surrounded by nonsacred forests with different compositions (evergreen broadleaf, 45.6%; evergreen needleleaf, 19.4%; bamboo, 19.0%; deciduous broadleaf, 16.1%; deciduous needleleaf, 0.0%). Out of 35,899 pairs of sacred forests and surrounding nonsacred forests, 13,156 pairs had the same dominant forest type (Appendix S2).

Overall, sacred forests were generally located at lower elevation (mean [SE] = 127 m asl [1]) than the protected areas that included both terrestrial and marine-terrestrial areas (mean [SE] = 417 m asl [6] based on the mean of each protected area, $n = 4715$, Welch t test = −45.3, df = 4960.9, $p < 0.001$). Moreover, sacred forests covered mostly lowland forests of Japan because the average elevation of the country's forests is 480 m.

Effects of sacredness and vegetation type on forest height and area loss

Outside protected areas, sacred forests and their surrounding nonsacred forests had different structural characteristics (Figure 3). Across all areas, the mean canopy height in sacred forests (15.5 m [SE 0.02]) was significantly higher than that in surrounding nonsacred forests (15.2 m [0.01], paired t test = 10.231, df = 17,271, $p < 0.001$). However, the mean

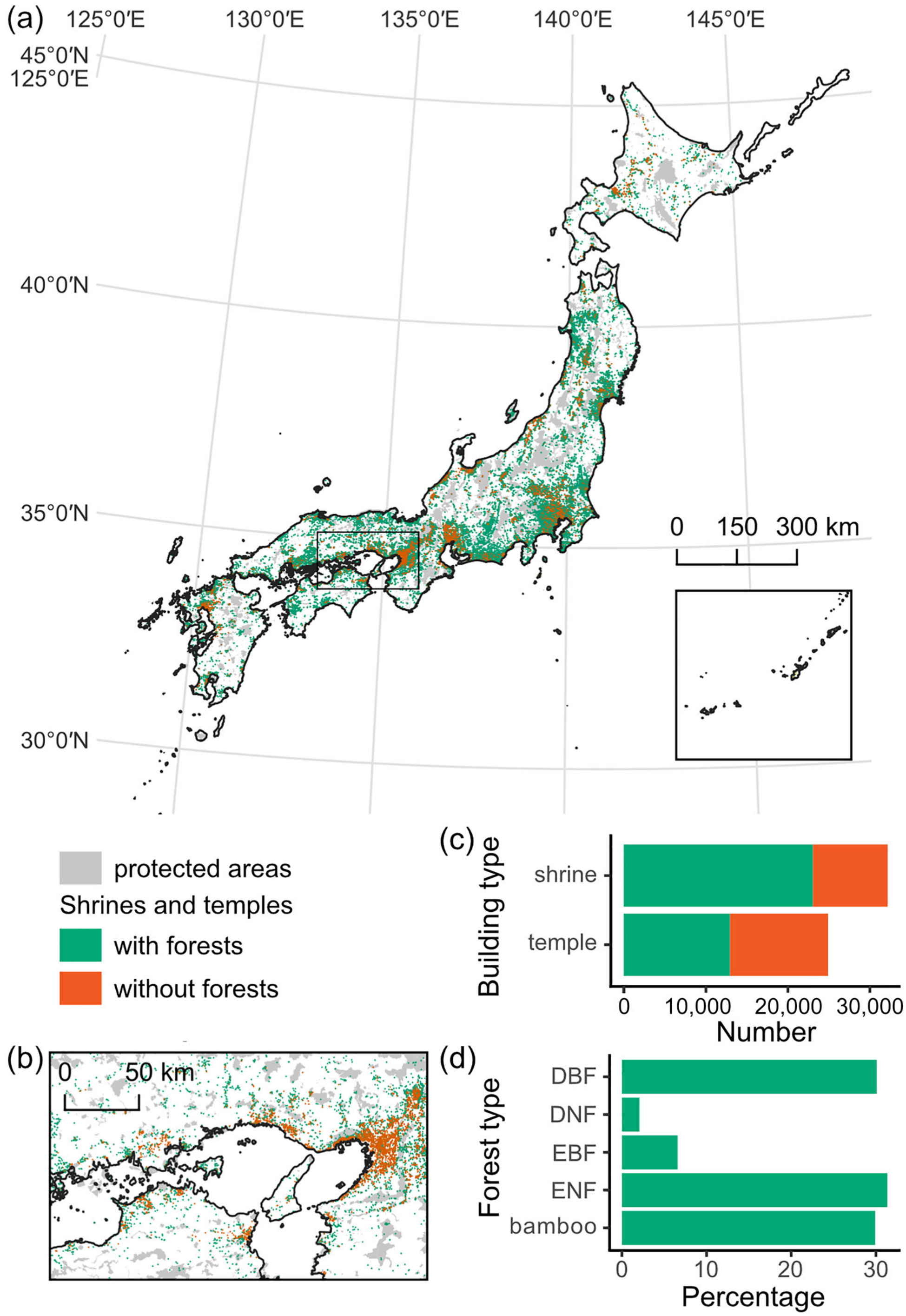

FIGURE 2 (a) Locations of shrines and temples across Japan of protected areas (gray) (UNEP-WCMC & IUCN, 2023) (green, sites associated with sacred forests), (b) detail of the spatial extent of places of worship in one region, (c) number of shrines and temples, and (d) percentage of the 5 vegetation types in sacred forests (DBF, deciduous broadleaf; DNF, deciduous needleleaf; EBF, evergreen broadleaf; ENF, evergreen needleleaf).

canopy height variation was lower in sacred forests (2.0 m [0.01]) than in surrounding nonsacred forests (3.1 m [0.01], $t = -132.8$, df = 17,079, $p < 0.001$). Similarly, sacred forests had higher mean canopy height than their surrounding nonsacred forests for areas without *C. japonica* plantations (15.1 m [0.02] and 14.6 m [0.01], respectively, $t = -28.859$, df = 16,817, $p < 0.001$). Finally, sacred forests had mean annual forest loss rates (0.07%/year) that were significantly lower than those in their surrounding nonsacred forests (0.13%/year, $t = -19.54$, df = 28,151, $p < 0.001$). These rates of loss were comparable to those in urban and rural areas (Figure 3) for areas without *C. japonica* plantations (0.07%/year and 0.10%/year, $t = -10.475$,

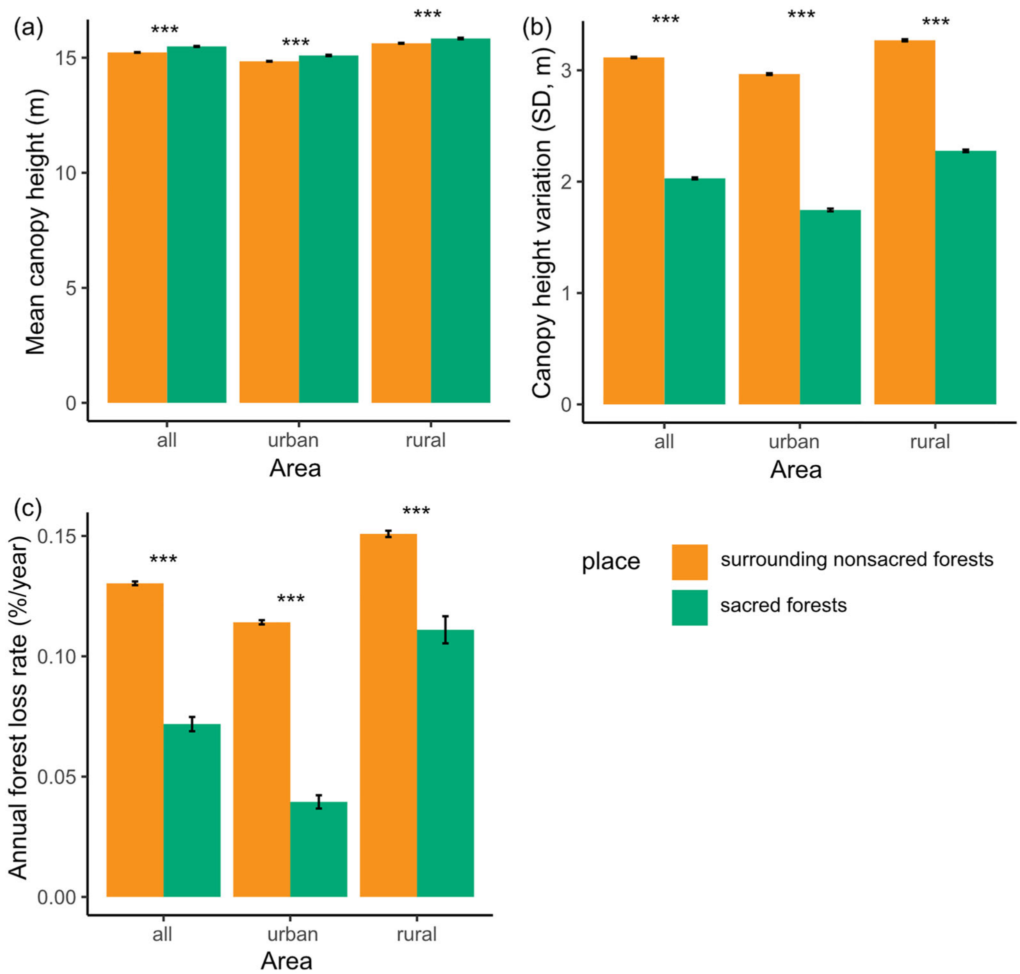

FIGURE 3 Mean (SE) (a) canopy height, (b) canopy height variation as standard deviation (SD), and (c) annual forest loss rate in sacred forests and in their unprotected, surrounding, nonsacred forests across Japan in urban and rural areas. The comparisons are based on paired *t* tests (***$p < 0.001$).

df = 27,700, $p < 0.001$) and for the subset of sacred forest and surrounding nonsacred forest pairs that shared the same dominant forest type (Appendix S2).

Annual forest loss rates were significantly different between sacred forests dominated by different forest types (Figure 4). Deciduous needleleaf-dominated ($n = 210$) and evergreen broadleaf-dominated ($n = 282$) sacred forests had the lowest annual forest loss rates (0%/year and 0.01%/year, respectively). However, the sacred forests dominated by either of the 3 other forest types showed loss rates 5 (bamboo, $n = 5485$, 0.06%/year; evergreen needleleaf, $n = 5286$, 0.07%/year) to 10 times higher than that of evergreen broadleaf sacred forests (for deciduous broadleaf, $n = 4952$, 0.09%/year).

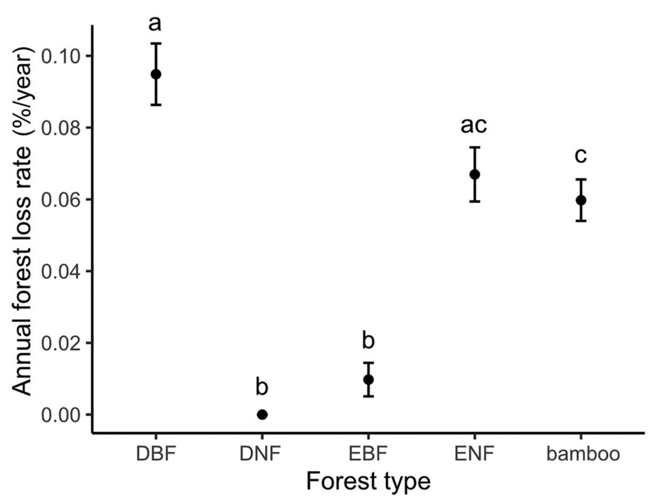

FIGURE 4 Mean (SE) annual forest loss rate from 2000 to 2022 of unprotected sacred forests dominated (75%) by deciduous broadleaf (DBF), deciduous needleleaf (DNF), evergreen broadleaf (EBF), evergreen needleleaf (ENF), and bamboo. Different letters indicate a significant difference based on a multiple comparison method for groups of unequal sizes (Herberich et al., 2010) (Appendix S3).

Effects of religious identity and urban–rural setting on forest characteristics

Religious identity was significant in characterizing canopy height differences between sacred forests and surrounding nonsacred forests outside protected areas. A smaller difference in

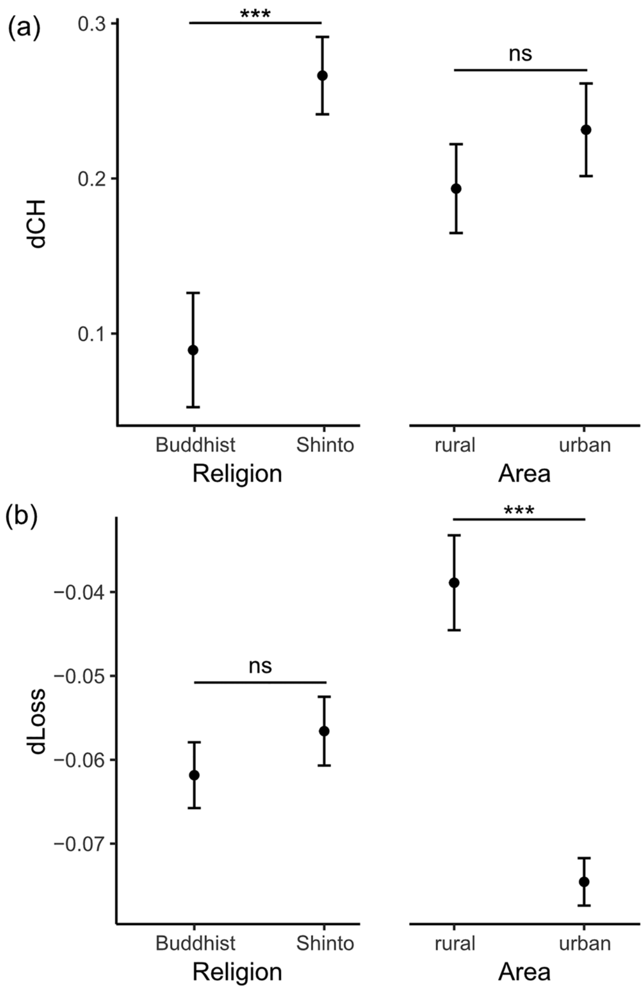

FIGURE 5 Mean (SE) difference in (a) canopy height (dCH, difference in mean canopy height in meters [sacred forest canopy height − nonsacred forest canopy height]) and (b) annual percent forest loss rate per year (dLoss) between sacred forests and surrounding nonsacred forests in Buddhist (temple) and Shinto (shrine) forests and in rural and urban sacred forests (***$p < 0.001$; ns, not significant [$p > 0.05$]).

canopy height was observed between temple sacred forests and surrounding nonsacred forests (dCH = 0.09 m) than between shrine sacred forests and surrounding nonsacred forests (dCH = 0.27 m, $t = 3.98$, df = 10,423, $p < 0.001$) (Figure 5a). When accounting for rural–urban areas and dominant forest types, the differences between temple and shrine forests for dCH were significant in broadleaf-dominated forests (urban and rural areas) and in rural needleleaf forests (Appendix S4). However, the difference in forest loss between sacred forests and surrounding nonsacred forests was similar in shrines and temples (dLoss = −0.06 for both religions, $t = 0.93$; df = 26,355; $p = 0.35$) (Figure 5b), even when accounting for urban (dLoss = −0.08 and −0.07, respectively, $t = -0.93$, df = 14,491, $p = 0.35$) and rural locations (dLoss = −0.04 and −0.05, respectively, $t = 1.16$, df = 10,503, $p = 0.25$).

The differences in canopy height between sacred forests and the surrounding nonsacred forests were not different between urban and rural areas (dCH $t = 0.93$, df = 17,082, $p = 0.35$) (Figure 5). Although mean annual forest loss was consistently lower in sacred forests across urban and rural areas, the contrast was greater for urban areas (dLoss = −0.07%/year) than for rural area (dLoss = −0.04%/year, $t = -5.64$, df = 18,909, $p < 0.001$).

Protection equivalence between sacred forests and strictly protected areas

Sacred forests outside protected areas ($n = 23{,}337$) had a lower rates of forest loss than nonsacred forests in protected areas (0.07%/year against 0.11%/year, Welch 2-sample t test = −7.01, df = 11,865, $p < 0.001$) (Figure 6a). Moreover, we found differences based on the considered IUCN management levels. Annual losses in unprotected sacred forests were similar to those in forests of protected areas under strict management ($n = 67$): 0.07%/year and 0.05%/year, respectively ($t = 1.42$, df = 70.60, $p = 0.16$) (Figure 6a). In contrast, protected areas under lenient management (0.11%/year, $n = 4599$) had significantly higher losses than sacred forests ($t = -7.23$, df = 11,221, $p < 0.001$). In protected areas, sacred forests had lower losses (0.06%/year, $n = 5570$) than the surrounding nonsacred forests (0.08%/year, paired t test = −3.38, df = 1242, $p < 0.001$) (Figure 6b). For comparison, we show the results for the pairs of sacred forests and surrounding nonsacred forests with the same dominant forest type in Appendix S2 and for all sacred sites (i.e., not just sacred forests) in Appendix S5.

DISCUSSION

Effects of sacredness on canopy height and forest area loss

Our results showed that sacred forests were generally taller than the surrounding forests even though their size increases their exposure to stressors. Omura (2004) reports a mean area of sacred forests of 6.1 ha. However, shrine and temple forests can be as small as 1 ha, and thus, they likely experience substantial edge effects (Murakami & Morimoto, 1999). Edge effects affect vegetation growth and biomass accumulation and composition (Harper et al., 2005; Reynolds et al., 2017) and increase wind exposure (Laurance & Curran, 2008). Nevertheless, our results suggest that management practices contributed to the differences in vegetation height (Oono et al., 2020) between sacred forests and nearby nonsacred forests. More precisely, some shrine and temple forests are old-growth natural forests, with giant trees (Manabe et al., 2003; Miura et al., 2001), and others have artificial origins (e.g., partially planted because some species are favored) and thus have different structure than the surrounding forests (Hattori et al., 2010; Ishii et al., 2010). As a result, the sacred forests in Japan are tall vegetation hotspots even though the height difference is small in comparison with other regions where the surrounding forests are intensively exploited (Alohou et al., 2017; Campbell, 2005; Oono et al., 2020).

The rate of loss of sacred forests (0.076%/year) was 50% of the rate in nearby forests (0.13%/year) and of other unprotected and nonsacred forests of Japan (0.16%/year based on

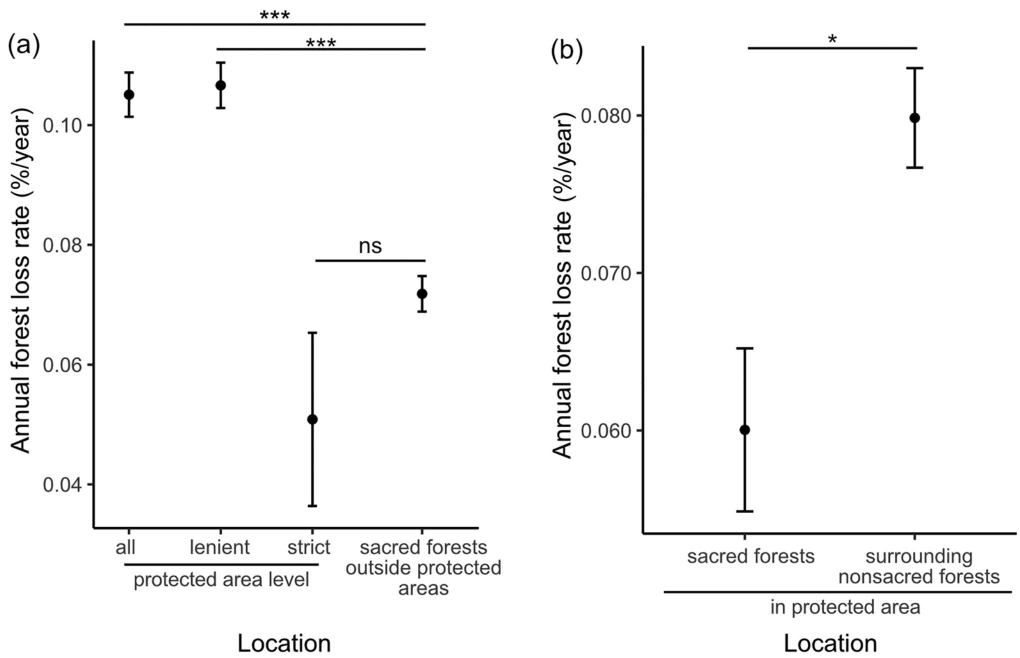

FIGURE 6 Comparisons of the mean (SE) annual forest loss rate in (a) sacred forests outside protected areas and forests in all protected areas based on management level (lenient, International Union for Conservation of Nature categories IV–VI; strict, categories Ia–III) and (b) sacred forests inside protected areas under strict and lenient management and the surrounding nonsacred forest contained in a protected area (***$p < 0.001$; *$p < 0.05$; ns, not significant [$p > 0.05$]).

our datasets), confirming that sacred forests are an effective protection against deforestation. Moreover, our findings were validated when we compared a subset of shrine and temple forests that shared the same dominant forest type with their surrounding nonsacred forests (Appendix S2). The forest loss rate we found was comparatively lower than those reported for sacred forests in other countries (although see Zannini et al. [2022] for Italy), such as in Ethiopia (0.39%/year) (Daye & Healey, 2015) and Guinea Republic (~3.8%/year) (Soumah et al., 2022). The high levels of conservation of sacred forests in Japan are likely explained by their recognized value for and by the community (Aoshima et al., 2018; Daniel et al., 2012; Kovács et al., 2022) and by the management of site owners and the community (Brondízio et al., 2021; Indrawan et al., 2014; Roux et al., 2022). Regulations on forest product extraction (e.g., logging [Ikebe, 2005]) may also have contributed to the differences in forest structure and dynamics between sacred forests and their surrounding forests because they are likely under different regulations. This information could not be included in our analyses, but it could help identify the main causes of forest losses where it is available. Future studies could include this information and additional data layers (e.g., topography, land-use history) to identify control forests to strengthen the results of these comparisons.

Association of lowland forest conservation with shrines and temples

The 2 times higher rate of forest loss in rural sacred forests than in the urban areas (Figures 3c & 5b) suggests that urban sacred forests had greater potential for conservation in comparison with their landscapes than rural sacred forests. These differences may not be related to different religious practices. The differences might be explained by varying pressures (e.g., resource extraction) and land conversion rates occurring in areas outside cities (DeFries et al., 2010) and by the strict laws regulating logging in urban areas (e.g., the Urban Green Space Conservation Act and the City Planning Act) (Yokohari et al., 2000). Moreover, the abandonment of traditional landscapes (i.e., satoyamas) (Indrawan et al., 2014) in peri-urban areas has led to the growth of secondary forests (Kobori & Primack, 2003). This likely contributed to forest conservation in areas defined as urban in our study, compared with rural areas.

Sacred forests can be an important protection feature of lowland environments. The sacred forests we detected were on average at lower elevations than the mean forest elevation across the country and across the protected areas. This is consistent with the distribution of the population toward lowland areas in Japan. Hence, sacred forests likely help maintain forest cover and biodiversity in lowland prone to development (Imanishi et al., 2007; Sakamoto et al., 1989a). Specifically, this effect is observed in the persistence of old-growth laurel-oak forests (Miura et al., 2001; Mizuuchi & Nakamura, 2021), as well as in increased plant (Nagaike, 2012; Tamura & Shimano, 2009), bird, and butterfly diversity (Kasada et al., 2017) in lowlands. Additionally, the resistance of urban shrine and temple forests to area loss makes them an important asset in urban biodiversity conservation (Manabe et al., 2007). Urban sacred forests can have specific microclimates with high humidity and lower temperatures that mitigate summer heat (Ishii et al., 2005). Shrine and temple forests are green islands that help connect the green infrastructure (Fujita, 2021; Ishii et al., 2010). Hence, sacred forests are an important component of urban sustainability (Blicharska et al., 2019).

Effectiveness of sacred forests in preventing forest loss

The protection associated with forest sacred value appeared to be as effective as strict legal protection to maintain forest cover because the sacred forests had rates of loss similar to those in protected areas in IUCN management categories Ia–III (Figure 6). Community-led conservation helps preserve forest cover outside legally protected areas (Porter-Bolland et al., 2012). We found that sacred forests were an additional and efficient alternative forest area conservation approach at the scale of a country. In rural landscapes outside the bounds of protected areas, sacred-forest-associated regulations, such as the conservation of trees around places of worship, can contribute to the maintenance of traditional landscapes (Indrawan et al., 2014) and forest cover. In addition, there are many more sacred forests than protected areas in Japan (35,899 vs. 4715, respectively, in our study) and likely most regions of the world (Deil et al., 2021), although they likely cover a smaller area than protected areas. They therefore provide significant and complementary environmental protection in geographical contexts that are not well represented in protected areas.

Sacred forests in protected areas had a lower forest loss rate than the protected surrounding forests, suggesting that multiple protection levels can increase a site's preservation, as reported by Liang et al. (2023) for Tanzania. In Japan, many shrines and temples are in large protected areas and World Heritage sites (Ishii et al., 2010). Legislation added to community rules and a site's owner management can lead to greater protection—through multiple restrictions—of these environments. Moreover, the protected areas containing these sacred forests possibly reduced the rate of forest loss at the edge of sacred forests and thus reduced vegetation stress (Hansen & DeFries, 2007).

Old-growth evergreen broadleaf forests are well conserved around shrines and temples (Kawata et al., 2023). These locations can be the last remnants of primary forests in some areas; we found that their surrounding nonsacred forests were mostly composed of other types of trees. This protection is likely to persist through time because we found an annual forest loss rate of 0.01%/year in sacred forests dominated by primary broadleaf evergreen trees outside protected areas (Figure 4). This effect explains the association between old-growth forests and sacred sites, such as the Mount Tatera in southwestern Japan (Miura et al., 2001).

Effect of religious practices on forest protection

Our findings reflect the observations of Manabe et al. (2011) in Kitakyushu, where shrines are generally surrounded by more forests (72% of shrines) and have higher diversity than the temples (52% of temples). Given this difference between Shinto and Buddhist practices, we expected the sacred forests around shrines might be lost at a slower rate and have a higher canopy height. However, we did not observe an effect of religious identity on forest loss, even when we focused on urban or rural areas to account for the biased distribution of Buddhist temples in urban areas. Hence, practices linked to both religions likely lead to similar effects on forest conservation. Nevertheless, Shinto sites are often associated with tall and old tree stands (Omura (2004), which may explain why the canopy height difference between sacred and surrounding nonsacred forests is greater in Shinto sites than in Buddhist sites (Figure 5a).

Recommendations for global forests conservation

In comparison with protected areas, sacred forests have the advantage of being close to regions with limited legal protections (e.g., lowlands, inhabited areas) and of having sustained local community involvement in their management (Marini Govigli et al., 2024; Ormsby & Bhagwat, 2010). Although large-scale studies on the effects of sacred sites on conservation have been carried out (Shen et al., 2015; Zannini et al., 2022), there is little information on their complementarity with existing protected areas. Hence, we recommend national-to-regional scale assessments of the effectiveness of the ecological protection associated with religious sites, including their identified nature experiences for people (Sahle et al., 2021). In addition, the assessment of religious activity and subsequent management effects on the various aspects of ecosystem conservation (e.g., limited extraction, forbidden access) is needed.

Legally protected areas are unevenly distributed. Joppa and Pfaff (2009) showed that the distribution of protected area is generally skewed toward higher elevations and areas far from urban centers, and Kamei and Nakagoshi (2006) found that the protected areas of Japan do not represent the full range of geographical contexts of forest communities. Hence, our results support the integration of sacred sites into national and global conservation projects (Bhagwat & Rutte, 2006) as beneficial for bridging the conservation gaps in lowland ecosystems and urban areas (see Frascaroli et al. [2019] for Italy). Moreover, our findings suggest that sacred forest protection is effective even in regions with low anthropogenic disturbance (Oono et al., 2020) and across a wide latitudinal and elevational range, as well as diverse forest types. However, the effectiveness of sacred forests likely varies between regions and practices; therefore, their integration in conservation projects may not be equally significant everywhere. Hence, we recommend that the effectiveness of sacred forests be assessed using a standardized methodology to ensure comparability (Verschuuren et al., 2021). Integrating locations with proven conservation value could help address the challenge of integrating many forests with low conservation potential (e.g., those highly disturbed or close to existing protected areas). However, the integration of sites with varying management status (e.g., different restrictions, religious practices) into existing conservation networks may remain a challenge if their management statuses do not align with the network's recommendations. Globally, the recognition of the conservation value of sacred forests will promote the inclusion of local communities in conservation initiatives to ensure their long-term existence (Dawson et al., 2021; Garnett et al., 2018; Maxwell et al., 2020).

Geospatial data and large-scale assessment of SNSs' conservation values

Our results demonstrate that VGI can be an effective tool with which to study sacred forests over large areas. Even though the dataset collected in our study (~23,722 ha) represents a fraction of the estimated 110,000 ha of sacred forest throughout Japan (Verschuuren et al., 2010), it contains over 30,000 sites, close to Omura's (2004) estimate of 34,000 sites. This discrepancy between our numbers and this author's numbers likely stems from our use of a conservative radius value to identify sacred forests around sacred sites identified in OSM. Finding or establishing comprehensive datasets on SNSs, including sacred forests, at a country scale is complex. For example, some sacred forests are in protected areas in Japan (Ishii et al., 2010), but their location inside these areas is unclear. In addition, sacred forests in legally protected areas represent a small share of sacred forests (~10% in our study). Hence, alternative approaches are necessary to create large datasets when dedicated datasets are lacking. Geographic information collected by volunteers, such as OSM, can be used to monitor SNS dynamics if the data are accurate (Spielman, 2014). Nevertheless, this approach does not identify the exact boundaries of sacred forests, and it is not applicable in every region. In fact, our approach is limited to sacred forests that are consistently associated with places of worship that are precisely located (i.e., a building or a landmark). Hence, the identification of distinct sacred forests is an approximation in complex sites covering large areas (e.g., Meiji Shrine). Besides this, the completeness of the VGI dataset is a critical factor in this application.

Although Japan's OSM data have above average completeness (Barrington-Leigh & Millard-Ball, 2017; Seto, 2022), the number of identified sacred sites in our study is less than half the number of corporations (140,000) assumed to be shrines or temples by the Japan National Tax Agency. Countries for which sacred forests could have a significant role in forest conservation (Boonman et al., 2024; Hounkpati et al., 2022) may have far from complete OSM data. Thus, our methodology is currently applicable in regions with enough VGI user contributions. Increasing OSM data collections could be done through campaigns (Neis & Zielstra, 2014) and collaboration with local communities and precinct managers to include additional information on precise sacred forest boundaries, cultural practices, and local management. Nevertheless, our results showed that OSM data allow the identification of locations of many sacred forests where VGI data are available and the importance of sacred forests as a tool for forest preservation.

ACKNOWLEDGMENTS

We acknowledge OpenStreetMap contributors for the expansion of the database used in this study. OpenStreetMap data is available under the Open Data Commons Open Database License at openstreetmap.org. This work is supported by National Science and Technology Council Grants No. 113-2740-M-003 -001 -, 113-2740-M-003 -002 -, and 113-2114-M-003 -001 -.

ORCID

Jonathan Peereman https://orcid.org/0000-0002-5128-3536
Teng-Chiu Lin https://orcid.org/0000-0003-1088-8771

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

How to cite this article: Peereman, J., Aiba, M., & Lin, T.-C. (2025). Relative effects of sacred forests and protected areas on forest conservation and structure in Japan. *Conservation Biology*, e70093. https://doi.org/10.1111/cobi.70093