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SPATIOTEMPORAL RISK OF SPECIES COLONIZATION OF PALO SANTO *(TRIPLARIS CUMINGIANA* FISCH, & C.A.MEY*)* IN THE QUEZON PROTECTED LANDSCAPE, PHILIPPINES

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ABSTRACT

Invasive Alien Plant Species threaten the sustainability of remaining forest ecosystems including those in Protected Areas. Unfortunately, there are only a few investigations on their colonization and dispersal over time, particularly most protected areas in the Philippines. This study aimed to assess the spatial extent of areas at risk of species colonization and dispersal of *T. cumingiana* that was introduced in the Quezon Protected Landscape over 40 years. A tree inventory following a combined belt transect and purposive sampling was made to determine occurrences of their natural regenerations. The spatial extent and areas at risk of species colonization were determined by modeling its species distribution using species occurrence and 9 environmental predictor variables via Random Forest Classification in a Geographic Information System. The species occurrences of *T. cumingiana* at different growth stages were used to project areas at risk to species colonization at different time slices. Results revealed a total of 996 individuals composed of 547 *seedlings*, 375 *saplings*, 54 *poles*, and 20 *standard* trees. Projections showed an increase in spatial extent and areas at risk to species colonization from 12.54 ha (1.29 %) in the 2000s to 39.28 ha (4.05 %) by the 2020s, with *distance to seed sources* and topography-related features, specifically *slope*, *aspect*, and *elevation,* among the most important variables determining its occurrence. These results indicated slow colonization of *T. cumingiana* in QPL which can provide foundations for understanding biology and impacts of this invasive species, as well as identifying its optimal management and control strategies for sustaining the ecology of the landscape.

Keywords: *invasive species, species distribution, exotic species, bioinvasion*

INTRODUCTION

Invasive alien plant species (IAPS) refer to plant species that have been displaced from their natural ecological range (EU, 2022; Vrabič-Brodnjak & Možina, 2022). The introduction and spread of IAPS outside their native region (Shah *et al*., 2020; Yadav *et al*., 2024) have threatened biodiversity, the delivery of ecosystem services (Pyšek *et al*., 2020), and the economy. IAPS can be introduced into an ecosystem from the import of exotic plants

for cultivation, botanical garden, as experimental plants, or accidentally via contamination of imports (Ricciardi, 2012; Tjitrosoedirdjo, 2005). Seed dispersal of successfully established IAPS enables species colonization by allowing the species to reach new sites with resources favoring regeneration, and escape pests and competition (Wright *et al*., 2008). This favors the species, leading to proliferation and bioinvasion. In the Philippines, some alien species of trees were introduced for reforestation in the 1910s, such as *Triplaris cumingiana* (Baguinon *et al*., 2005) have naturalized and exhibit invasive traits. Although some alien species pose no harm or detrimental effects (Paclibar & Tadiosa, 2019) and even raise as crops and ornamental plants (Tjitrosoedirdjo, 2005), some can exhibit rapid growth and create dense thickets, which can outcompete native vegetation and dominate the plant community over time.

The aggressive nature of IAPS poses a growing ecological threat, disrupting native ecosystems and biodiversity (Gazoulis *et al*., 2022; Yadav *et al*., 2024). It also alters ecosystem functions and properties (Vrabič-Brodnjak & Možina, 2022; Anjum *et al*., 2022) and overall ecological sustainability (Rai & Singh, 2020) while challenging traditional conservation efforts. IAPS are considered one of the major drivers of biodiversity loss and ecosystem degradation (Rai, 2022). Biological invasions by IAPS not only displace native vegetation (Gazoulis *et al*., 2022) but also modify soil cover, nutrient cycling, fire regimes, and hydrology (Vrabič-Brodnjak & Možina, 2022) making the site unsuitable for native species. IAPS may also occupy unfilled niches (Mathakutha *et al*., 2019) traditionally played by native species and cause disruption. For example, they may outcompete native plants, interrupting plant-pollinator interactions (Kovács-Hostyánszki *et al*., 2022).

IAPS are also threatening PAs, which play a key role in biodiversity conservation worldwide (Braun *et al*., 2016; Bhatta *et al*., 2020). The escape of ornamental plants from gardens and ornamental horticulture into the wild, including PAs, can be a major driver of alien plant invasions (van Kleunen *et al.,* 2018; Pyšek *et al.,* 2020; Vrabič-Brodnjak & Možina, 2022). In the Philippines, evidence on the impact of IAPS on local biota is rare and restricted to several anecdotal reports indicating that native species may be severely affected by competition, predation, habitat modification, and parasitism. This is attributed to 1) lack of knowledge about taxonomic identity of IAS and also lack of extensive and comprehensive technical information; 2) failure to realize the potential ecological damage to biodiversity and consequent economic losses and possible hazards to human health; 3) failure of implementation of laws on introduction of exotic species; and 4) unwillingness to interfere in the commerce and trade of exotic species (Joshi, 2006). Among the few studies in IAPS in the Philippines, Baguinon *et al*. (2011) reported a serious problem of quick bioinvasion in gap and open areas of Paper Mulberry (*Broussonetia papyrifera*) and Skyflower (*Thunbergia grandiflora*) while gradual and slow bioinvasion at the expense of secondary and primary forests of Big-leafed Mahogany (*Swietenia macrophylla*) and Palosanto (*Triplaris cumingiana*) in Mt. Making Forest Reserve. Unfortunately, the factors that determine the spatially uneven distribution of alien plant species (Essl *et al*., 2019), their modes of invasion, and impacts (Linders *et al*., 2019) are still poorly understood.

In 1980, *T. cumingiana* was planted along the road and buffer zone in QPL (Tadiosa, 2019) as enrichment planting and to improve the aesthetic of the site. This is considered a small tree, rarely medium, with a height reaching 20 m and a diameter up to 45cm. It is a prolific seeder with red fruits when ripe and greenish-white flowers suited for aesthetics and ornaments. Its seeds are dispersed by the wind, enabling it to spread rapidly and establish new populations. *T. cumingiana* has become naturalized and exhibits invasive traits and is considered IAPS today. It is widely known for its rapid growth and ability to create dense thickets, which can outcompete native vegetation and diminish biodiversity. The species is

especially problematic in forest environments. Its invasive nature endangers native plant species and has the potential to disrupt ecological processes in affected areas. Furthermore, bioinvasion of IAPS may impair the ability of the landscape to sustain its ecosystem services. Depending on the characteristics of invasive species, landscape composition and configuration may change, and these changes are crucial to conservation (Lee *et al*., 2019). But more importantly, IAPS may not effectively replace the role played by native tree species they displaced. This endangers not only the forest but also the riverine, agricultural, and other dependent downstream ecosystems over the whole landscape. Hence, understanding species colonization is crucial for sustaining overall landscape integrity.

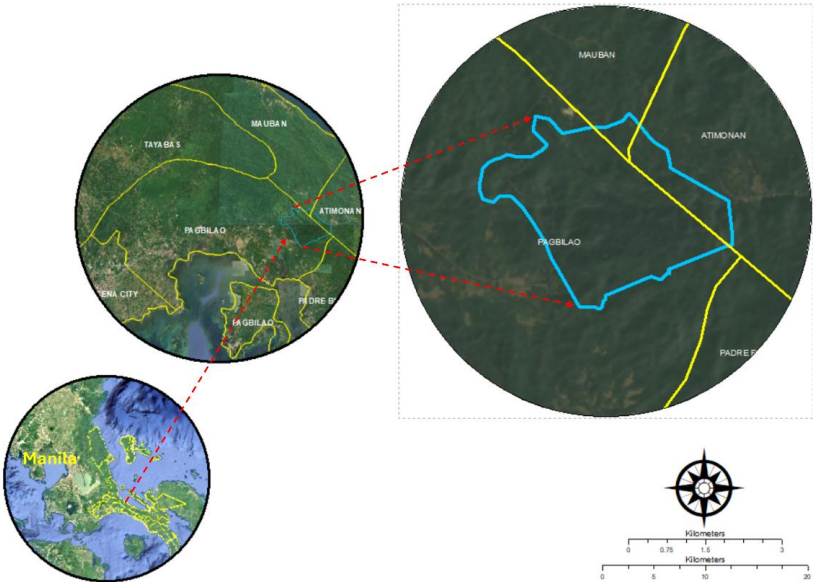
This study aimed to assess the dispersal, spatial extent, and areas in QPL that are at risk of species colonization by *T. cumingiana* by modeling its species distribution. This study hypothesized that the introduction of *T. cumingiana*, its seed dispersal by wind, and the 40-year period may have favored species invasion and increased areas at risk of colonization. These, in addition to the relative freedom from natural enemies, including pests and herbivores, may have ensured the successful establishment of the species. The study used the occurrence of the species and environmental predictor variables in modeling to determine factors that contribute to its occurrence. This can be valuable information for the effective management of potential IAPS and the ecological integrity of QPL, one of the remaining tropical rainforests at the southern end of the Sierra Madre Mountain Range.

MATERIALS AND METHODS

Study Site

QPL is a 983.08-hectare tropical rainforest (Paclibar & Tadiosa, 2019) located between 121°46'30" and 121°50' 00" East and 13°58'30" and 14°01'00" North in the southern Sierra Madre Mountain Range (Dagamac *et al*., 2015).

Fig. 1: Location of the study



The DENR, as cited by Paclibar & Tadiosa (2019) considers it a karst landscape and a very high priority for biodiversity conservation. It covers portions of communities in Silangang Malicboy, Pagbilao, Quezon, about a 45-minute drive from Lucena City and approximately 164 kilometers (102 miles) southeast of Metro Manila. Fig.1 shows the location of the study.

Modeling Spatial Distribution of *T. cumingiana*

Species distribution modeling generally requires the species' occurrence and environmental predictor variables to determine the probability of species occurrence over a given area. Areas with high probability of species occurrence were identified as suitable habitat. In this study, the risk of species colonization was evaluated based on a species distribution and habitat suitability model. That is, areas with likely occurrence or areas classified as suitable for *T. cumingiana* are pseudo data indicating spatial extent and areas at risk of species colonization. Hence, the increase in the suitability of an area for the species, the higher the risk of species colonization.

Inventory and Species occurrence of T. cumingiana. In 1980, 4 individuals of *T. cumingiana* were outplanted along the roadside, which serves as the mother trees. The inventory of *T. cumingiana* regenerants was made following a combined belt transect and purposive sampling. Based on the location of the mother tree, a 20-meter belt transect (10 meters on both sides of the transect line) was initially established for the inventory of its offspring or young natural regenerations at various stages of growth. The width of the transect is adjusted progressively to cover regenerants not covered in the 20-meter transect. The geographic locations of each regenerants were recorded and digitized in GIS software.

The regenerants were classified into different growth stages based on diameter. During the initial site visit, it was found that the diameter at breast height (DBH) of the 5 original mother trees ranged from 30 – 35 cm or roughly ~0.82 cm year^{-1} annual growth rate in 40 years. Based on QPL management officers' observation, reproductive maturity of *T. cumingiana* was estimated at 10 years (about 8 cm DBH). Based on these, regenerants were classified as *seedlings* (< 2 cm), *saplings* (2 to 6 cm), *poles* (7 to 15 cm), and *standards* (> 16 cm). These were then used as occurrence data for modeling the risk of species colonization.

The spatial distribution of regenerants relative to the mother trees was also assessed. Based on their geographic locations, the density was determined, and their distribution patterns were characterized by using Spatial Autocorrelation in GIS software.

Environmental Predictor Variables. There were 9 environmental predictor variables used in modeling the risk of species colonization. For climate variables, only *annual rainfall* and *mean daily temperatures* were used from 19 bioclimatic variables available at the Worldclim website (https://www.worldclim.org/) due to the multicollinearity among these variables. *Elevation*, *slope*, and *aspect* were the topographic features generated from the digital elevation model (DEM) accessed from the EarthExplorer website (https://earthexplorer.usgs.gov/) and used in this study. Proximity data included *distance to coast*, *distance to road*, and *distance to seed source* which were generated from Euclidean distances. *Wind speed* data were acquired from the Global Wind Atlas website (https://globalwindatlas.info/).

Species Distribution and Risk of Colonization Assessment. The risk of species colonization based on species distribution was modeled using the Random Forest algorithm in GIS software. This random forest regression (RFR) is a specific type of Random Forest Machine Learning that predicts continuous values (Liu *et al*., 2020 as cited by Islam *et al*., 2023) by creating ensembles using bootstrap aggregating (bagging). This develops multiple models and improve prediction (Breiman, 2001; Coulston *et al*., 2016) by averaging the results of all the individual decision trees (Islam *et al*., 2023). Random Forests are considered a powerful

machine learning algorithm for modeling species distribution (Zhang *et al*., 2019; Valavi *et al*., 2021). It can outperform other machine learning assessment tools even in undersampled areas (Mi *et al*., 2017). Modeling used *T. cumingiana* inventory/occurrence data and environmental variables describing the site conditions.

The species occurrences in the year 2000s, 2010s, and 2020s, representing 20, 30, and 40 years after the initial introduction of *T. cumingiana* in QPL, respectively, were used as projection scenarios. The species occurrences used for these scenarios were identified based on the DBH-based classification and its crude growth rate. For example, a 10 cm regenerant of *T. cumingaina* classified as *poles* must have grown in the area for at least 10 years, assuming ~0.82cm year^{-1} annual growth rate. These *poles* are added to the occurrences of *mother trees* and standards (>16 cm) and used as species occurrence data for projecting areas at risk 30 and 40 years after introducing *T. cumingiana* mother trees. Specifically, the species occurrences used for the projections at different time periods were as follows: 1) 2020s include all species occurrences; 2) 2010s occurrences of the *original mother trees* + *standards* + *poles*; and 2000s occurrences of the *original mother* + *standards*. All other environmental variables such as vegetation cover and climate were set at constant for all scenarios. In addition, absence data were formulated from traversed routes where no regenerants of *T. cumingiana* were observed. Absence points were created by buffering the routes by 20 m and converting them to 30 m fishnets, then using their centroids as absence points. These were then integrated with presence data.

Model Performance Validation. Random forest classification was made using 80 % of the presence-absence data as a training sample and 20 % for testing at 20 ensembles and compensation for sparse categories. Model performance was evaluated using median accuracy, F1-score, Matthews Correlation Coefficient (MCC), and sensitivity for the presence. Values greater than 0.7 for these parameters were considered acceptable model performance.

RESULTS

Regenerations of *T. cumingiana*

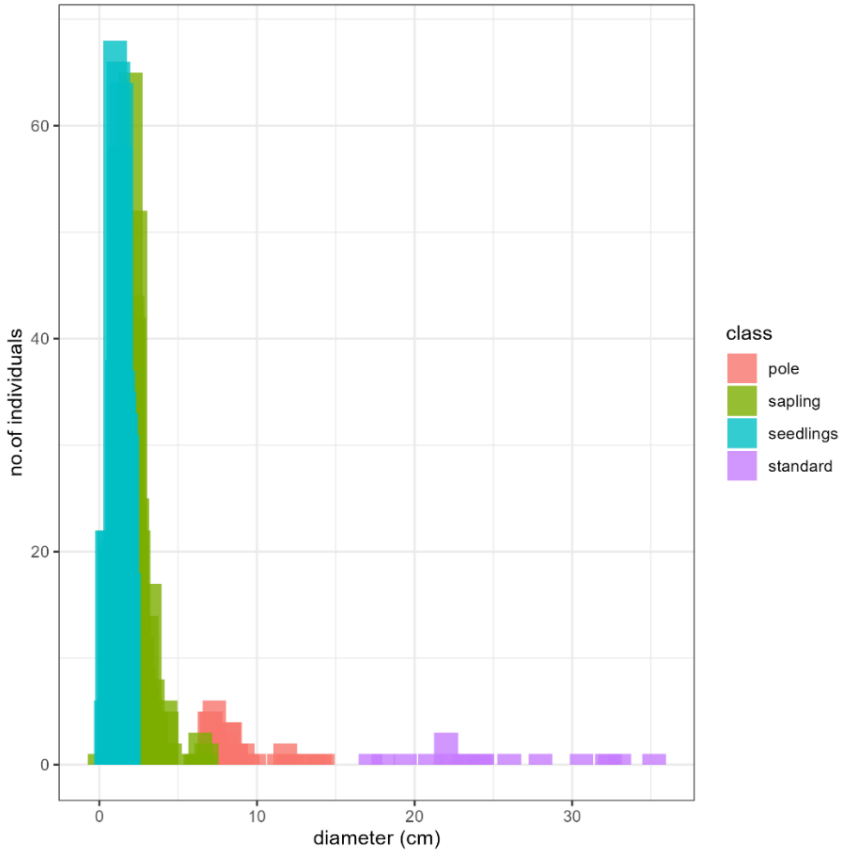
From 5 mother trees outplanted in 1980, a total of 991 regenerants composed of 547 seedlings, 375 saplings, 54 poles, and 15 standards were sampled from a total of 1.523 hectares of QPL. While the mean diameter of the mother trees was 32.68 cm over a 40-year period, the mean diameters of seedlings, saplings, poles, and standards were 1.24 cm, 2.62 cm, 8.73 cm, and 22.47 cm, respectively. Roughly, *T. cumingiana* grew by ~0.82 cm year^{-1} with reproductive maturity at about pole stage (about 8.17 cm in a 10-year period). Table 1 summarizes the inventory of *T. cumingiana* in QPL. Fig. 2 also shows the distribution of regenerants by diameter.

Table 1: Summary of inventory on *T. cumingiana.*

CLASS	NO. OF INDIVIDUALS	MEAN DISTANCE (m)*	DENSITY (hectare^{-1})
Seedlings	547	71.038^{a}	359.395
Saplings	375	59.280^{b}	246.386
Pole	54	39.163^{c}	35.480
Standard	15	45.450^{cd}	9.855

*Mean distances with different lower-case letters are different at α = 0.05.; * distance from mother trees*

Fig. 2: The distribution of regenerants of *T. cumingiana* by diameter.



Occurrence and Dispersal of *T. cumingiana*

Fig. 3 shows the geographic location of *T. cumingiana* regenerations in QPL. As observed, the mother trees of *T. cumingiana* were planted at an elevated portion of QPL where their seeds can be dispersed by the wind to lower elevations. Table 1 shows the mean distance of regenerants of *T. cumingiana* from the mother tree. As expected from anemochorous species, most seedlings are found beyond the drip zones of *T. cumingiana* canopy. Results also showed that the mean distance from the mother trees was 53.733 m for all stages of regeneration. The mean distance of all seedlings from the original mother tree was significantly higher ($F = 29.134$, $p < 0.001$) using Welch's ANOVA at 71.038 m than saplings, poles, and standard trees, a 56.30% increase in dispersal distance over 40 years. This indicates a slow and steady spread of *T. cumingiana* in QPL. The species occurrences used for the projections at different time periods were 996 for the 2020s (including all species occurrences), 74 occurrences for the 2010s (including *original mother trees + standards + poles)*, and 20 occurrences for the 2000s (including the *original mother + standards*).

Fig. 3: Shows the Geographic Location of the Palo santo tree (*T. cumingiana*) including its regeneration in various stages in the QPL.

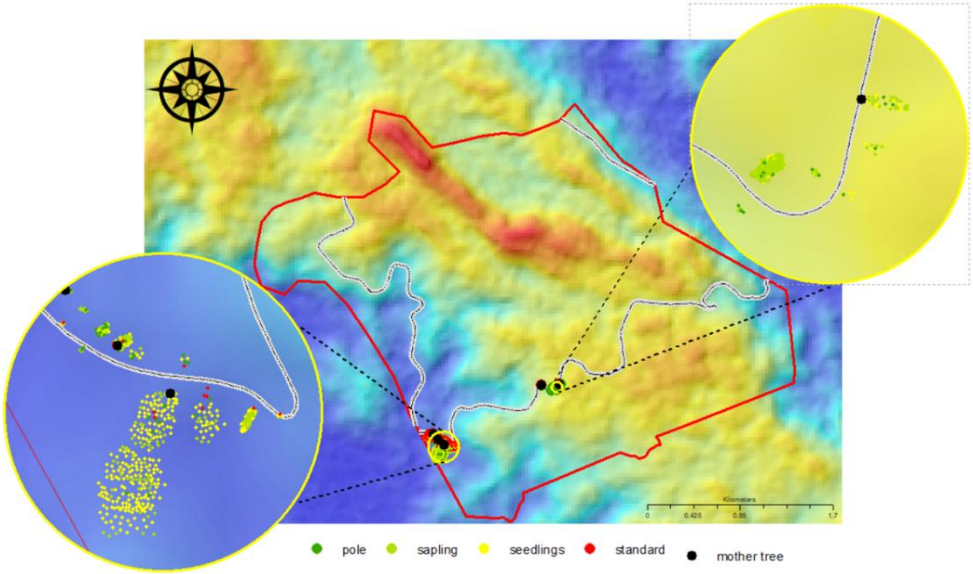


Fig. 4: The distribution of regenerants by diameter classes by distance to mother trees.

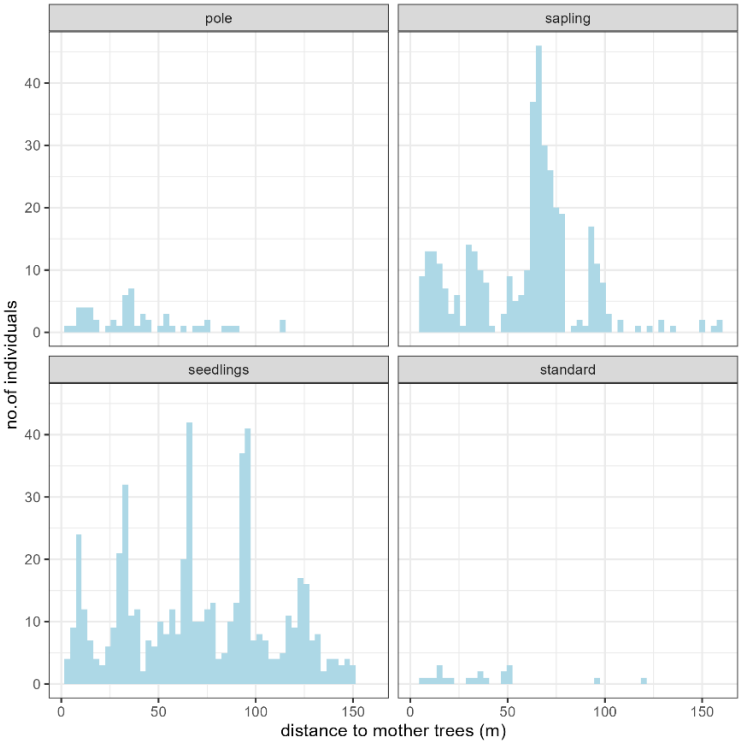
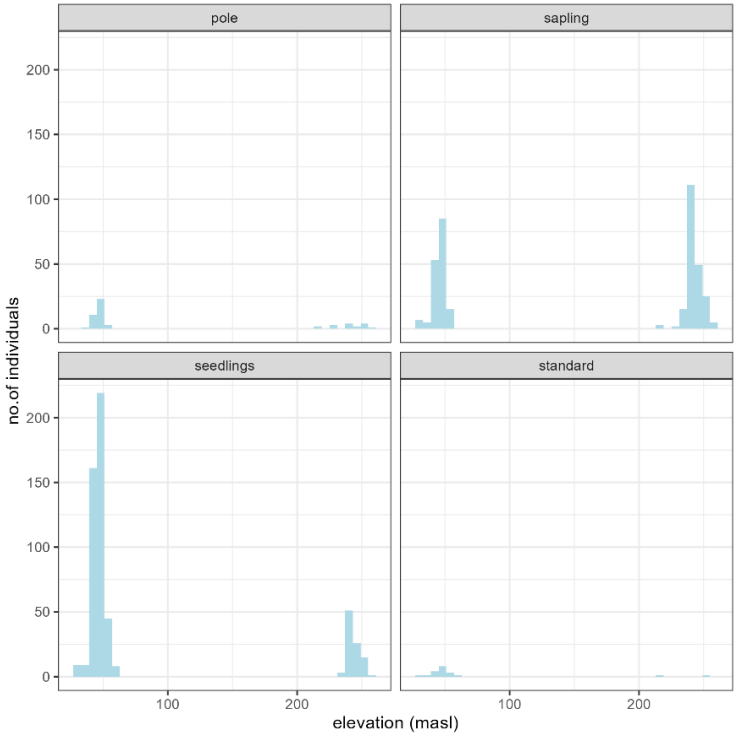


Fig. 4 also shows how different classes of regenerants were dispersed from the mother trees. The advanced stages of regenerants were closer to the mother trees than saplings and seedlings. This is indicative that not only the mother trees but also their advanced stages of regenerants may now be capable of reproduction and are providing propagules over far greater distances. This may confirm that *T. cumingiana* is slowly spreading in QPL.

Analysis also showed that regenerations of *T. cumingiana* tend to have a dispersed distribution about the mother tree at the tree level, although they cluster at the landscape level. Results showed that most of the seedlings, saplings, and poles are far from the standard trees which are the seed sources. Similarly, most of the seedlings were found in lower elevations relative to standard trees.

In addition, Fig. 5 shows the distribution of regeneration by elevation. Results show that there is an increasing number of regenerations at higher elevations. This can be problematic, especially when these regenerations reach reproductive age, since higher elevations are more exposed to wind and sunlight.

Fig. 5: The distribution of regenerants by elevation by diameter class.



Areas at Risk of Species Colonization

Fig. 6 shows the projected risk of colonization of Palo santo (*T. cumingiana*) in QPL. The modelling using Random Forest Classification shows that the projected areas suitable or at risk to species colonization of *T. cumingiana* increased from 1.29 % (12.54 hectares) in the year 2000s to 4.05 % (39.28 hectares) at present. Among the environmental predictor variables used, *distance to coast*, *distance to seed sources*, and topographic features specifically *slope*, *aspect*, and *elevation,* were consistently identified as among the top 5 important variables for the models. Tables 2 and 3 show the most important environmental

predictor variables and the model performances for the 2000s, 2010s, and 2020s projected risk of species colonization, respectively. These indicate the acceptability of the model.

Fig. 6: Changes in Spatial extent and Area at Risk of Species Colonization (*in red color*) in QPL for 2000s, 2010s, and 2020s.

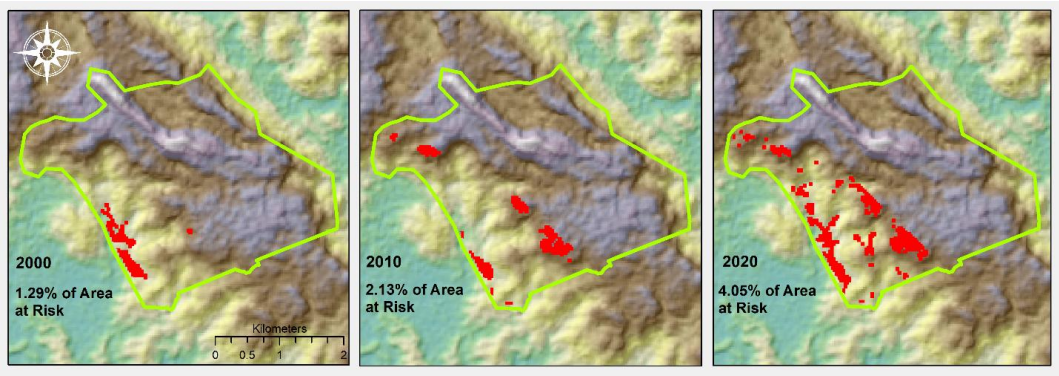


Table 2: The topmost important environmental predictor variables for the presence of *T. cumingiana* in 2000s, 2010s, and 2020s.

Environmental Predictor Variables	2000s Rank	2000s % Importance	2010s Rank	2010s % Importance	2020s Rank	2020s % Importance
distance to coast	1	17	1	16	1	14
distance to seed source	2	14	4	12	2	13
Aspect	4	12	3	12	4	12
Slope	---	---	5	11	3	13
annual rainfall	5	11	---	---	5	11
mean daily temperature	---	---	2	13	---	---
elevation	3	12	---	---	---	---

Table 3: Performance of the Models for Projected Risk of Colonization for the 2000s, 2010s, and 2020s.

YEAR	MEDIAN ACCURACY	F1-SCORE*	MCC*	SENSITIVITY*	RANDOM SEED
2000s	0.931	0.76	0.76	1.00	316877
2010s	0.948	0.87	0.84	0.94	470118
2020s	0.975	0.98	0.94	0.98	190221

**for species presence*

DISCUSSION

Natural Regeneration of *T. cumingiana* in QPL

The decline in the number of natural regenerations at different growth stages shown in Table 1 may be expected due to mortality associated with site factors and suitability, inherent capability of regenerants, and competition for basic needs. Since *T. cumingiana* was introduced in QPL, the site may be limiting its growth and development, and a long period may be needed for the species to adapt. Similarly, seed dispersal may be challenged as

suitable sites may be limited for introduced species. The decline is the number of individual representations at different growth stages resulting from seedling mortality, competition for sunlight, nutrients, water, and space during advanced stages. As trees grow, they maximize root length to compete for nutrients by preventing nutrient supplies from interacting with their neighbors (Craine *et al*., 2013). Plants also maximize height growth and crown development which positions their leaves above the canopy where they can maximize photosynthesis and reduce the growth of their competition by shading.

Spatial Distribution and Dispersal of *T. cumingiana*

The geographic location of *T. cumingiana* regenerations in QPL may have benefited from planting the mother trees at an elevated portion, where seeds can be dispersed by the wind to lower elevations. In addition, since these mother trees were planted along the road, they were relatively free from overtopping trees which can benefit their success in QPL while increasing the risk of species colonization. Most of the natural regenerations of *T. cumingiana* were found beyond canopy drip zones at a mean distance of 53.733 m. These results showed similarity with the reported tens to hundreds of meters seed dispersal by Sharpe *et al*. (2016) of wind-dispersed species like *T. cumingiana*. Furthermore, *seedlings* were significantly farther from the original mother trees than any other regenerations in other growth stages, indicating a slow and steady spread of *T. cumingiana* in QPL.

This gradual and slow spreading of *T. cumingiana* regeneration may favor species colonization of the area by reducing intra-specific competition for resources. As regenerants are dispersed further from one another, they are likely to encounter various environmental conditions and resource availability, which may facilitate greater individual growth and survival. Therefore, *seedlings*, *saplings*, and *poles* that reach the *standard* tree stage demonstrate greater resilience, adaptation, and competitive abilities. To attain maturity, they have overcome obstacles such as resource limitations, environmental stress, and biotic interactions. Their capacity to survive and grow to the standard tree stage indicates that they have advantageous characteristics and have successfully navigated earlier stages where mortality rates are higher. Later, these mature plants will provide propagules that contribute to spreading and species colonization of an area.

Younger regenerants also exhibited high densities with 359.395 ha^{-1} and 246.386 ha^{-1} for *seedlings* and *saplings*, respectively. This indicates high seed production, and the initial high density of younger regenerants showed prolific capacity to produce propagules that benefit species invasion and colonization (Gioria *et al*., 2023). *Poles* and *standards* trees showed a relatively low population, which can be due to limited propagule input (Lockwood *et al*., 2005; Gioria *et al*., 2023) from only 5 mother trees decades ago. This subsequent reduction from a high number of *seedlings* and *saplings* to a low abundance of *poles* and *standards* indicates increased competition for resources such as sunlight, water, nutrients, and spaces. As trees grow larger, competition for these resources intensifies, resulting in increased mortality and decreased tree density. As IAPS adapts to the site, high seedling density is also threatening since it may indicate that the invasive species proliferate and outcompete the native species (Antunes *et al*., 2016) and colonize an area. In PAs, this necessitates immediate action to safeguard the area's native biodiversity and ecological values (Gallardo *et al*., 2017).

Analysis also showed that natural regenerations of *T. cumingiana* tend to have a dispersed distribution about the mother tree at the tree level, although they cluster at the landscape level. Results showed that most of the *seedlings*, *saplings*, and *poles* are far from the *standard* trees which are the seed sources. Similarly, most of the seedlings were found in lower elevations relative to *standard* trees. This dispersed or scattered distribution is

expected from wind-dispersed species although wind patterns or local topography (Damschen *et al*., 2014) may result in clustering in seeds near mother trees at small scales (Seidler & Plotkin, 2006). The combination of dispersal mechanisms, environmental conditions, seed and seedling interactions, landscape features, and sporadic long-distance dispersal events creates a complex spatial pattern for the distribution of tree species in many ecosystems (Schupp *et al*., 2010).

Results also showed that there is an increasing number of regenerations at higher elevations upslope from the original mother trees. This can be problematic, especially when these regenerations reach reproductive age, since higher elevations are more exposed to wind and sunlight. These can favor the growth and reproduction, maximize seed dispersal to greater distances, and increase areas at risk to species colonization by *T cumingiana.*

Spatiotemporal Risk of Species Colonization

Projections showed that the areas at risk of species colonization of *T. cumingiana* increased 4.05 % (39.28 hectares) over the 40 years. The *distance to coast*, *distance to seed sources*, and topographic features specifically *slope*, *aspect*, and *elevation,* were consistently identified as among the top 5 most important environmental predictor variables for the models. Results showed a slow expansion of areas at risk to *T. cumingiana* colonization. This is relatively lower than the reported projected future expansion of suitable habitats of other IAPS such as *Mikania micrantha* (61–120 %), *Ageratina adenophora* (7–33 %), *Alternanthera philoxeroides* (12–74 %), and *Ambrosia artemisiifolia* (8–27 %) (Tu *et al*., 2021). The results also conform with the report of Baguionon *et al*. (2011) on slow bioinvasion of *T. cumingiana* in secondary and primary forests of Mt. Makiling Forest Reserve in the Philippines. This means that the introduction of species or their propagules onto new areas does not ensure successful colonization and invasion by exotic species. This exotic species must overcome barriers or filters to establishment, naturalization, production of localized self-sustaining populations, and eventually spread naturally before they are considered invasive (Pyšek *et al*., 2004; Shackleton *et al*., 2019; Gioria *et al*., 2023). These barriers and drivers can substantially differ among invasion stages (Dawson *et al*. 2009, 2013; Richardson & Pyšek 2012; Essl *et al*., 2019). The successful colonization and invasion by a plant species can be based not only on its inherent traits, such as high seed production but the extrinsic ecological characteristics of the recipient ecosystems defining its susceptibility to invasions (Gioria *et al*., 2023). Also, many exotic species may also experience long lag phases before they start to proliferate and spread (Aikio *et al*., 2010; Larkin, 2012; Osunkoya *et al*., 2021; Gioria *et al*., 2023). In this study, only the *distance to seed source* can be considered the inherent traits, while the other environmental predictor variables characterize the site condition and quality.

The *distance to coast* and topographic features were among the most important environmental predictor variables in predicting the occurrence of *T. cumingiana* in QPL. The *distance to coast* may indirectly influence plant species distribution through its effects on soil and climatic characteristics. The latter may include rainfall and temperature that were found to be more important variables describing the distribution of IAPS (Tu *et al*., 2021). Soil characteristics such as salinity (Shao *et al*., 2020) may vary from the coast to the inland and may limit plant growth and development. Paclibar & Tadiosa (2019) reported that most of the physicochemical factors are linked to the presence of invasive alien plant species in QPL and projected the boundary and buffer zones as the most suitable areas for IAPS.

Unlike Paclibar & Tadiosa (2019) and Tu *et al*. (2021), however, topographic features such as *slope*, *aspect*, and *elevation* were among the most important environmental predictor

variables that determine the presence of *T. cumingiana*. In previous studies, IAPS included herbs, shrubs, and vine species compared to small trees and wind-dispersed *T. cumingiana*. These topographic features may influence exposure to sunlight and wind which can favor its reproduction and seed dispersal. Initially, its mother trees were planted near the roadsides that are relatively exposed and have reduced overtopping crown and canopy of trees favoring its growth and reproduction. Tree canopy and distance to road, river, and settlement were identified by Bhatta *et al*. (2020) as the major factors affecting IAPS occurrence. These can be indicative of the exposure of an area.

In addition, climate variables seemed less influential on species occurrence. This conforms with Bates & Bertelsmeier (2021) suggesting it may not be biologically meaningful in understanding the true frequency of niche shifts. Accordingly, this fundamental niche is the set of environmental conditions under which the species can survive and reproduce. IAPS prefers a similar climate to its origin and has a limited climatic niche expansion (Liu *et al*., 2020) that often encounters biotic resistance (Gioria *et al*., 2023) and other barriers even in a suitable climate. This means the exotic plants need to cope with site suitability and interspecific challenges to naturalize and invade an area.

Implications on Management and Control of *T. cumingiana*

The invasion of exotic species into an ecosystem results from accidental escape to the wild or through deliberate introduction of the species for enrichment, rehabilitation, or in the case of *T. cumingiana* in QPL, aesthetic purposes. It appears that *T. cumingiana* is inherently capable of colonization with prolific seeding and wind-dispersed seeds. These are evident in the reported number of regenerants and their dispersal over the area, and seemingly early reproductive maturity. Fortunately, it appears that it has not significantly progressed in colonizing areas in QPL for over 40 years and 4.02 % of the area is projected to be at risk at present. The barriers that limit its dispersal and colonization in QPL may be evaluated and capitalized. Evidence, however, showed that regeneration is slowly reaching more exposed (via higher elevation) and suitable areas where they can maximize dispersal.

This slow pacing in species colonization of *T. cumingiana* implies less urgency compared to aggressive *B. papyrifera* and *T. grandiflora* reported by Baguinon *et al*. (2011) or the "Forest Killers" reported by Yadav *et al*. (2024). This pace provides time to understand the ecology of the species better, assess its ecological impacts, optimal management and control strategies, and/or its ecosystem services (Shackleton *et al*., 2019). Whether the species depends on allelopathic strategies to favor establishment (Thiébaut *et al*., 2019) or alters disturbance regimes or nutrient levels (Linders *et al*., 2019), this needs to be assessed not only to counter but to gauge its adverse effects.

For four decades, only about 70 individuals reached the pole stage or higher which may require a less intensive and less complex control strategy. Based on results, controlling these seed sources may lessen the areas at risk of species colonization. However, it is still early to conclude whether the current distribution and slow colonization of *T. cumingina* are proof of successful ecological resistance of QPL or just a lag time when it is initially gaining and in the process of overcoming barriers. Nonetheless, the species may disrupt the natural equilibrium and may not provide the same habitat, food, and shelter value for wildlife as native trees. Effective control during a successful invasion can be resource-intensive, complex, and challenging. Hence, efforts need to be made at this stage of *T. cumingiana* colonization.

CONCLUSION

The study assessed the risk of species colonization and dispersal of *T. cumingiana,* introduced to QPL in 1980, through inventory and modeling species distribution as pseudo-information for the risk of species colonization. The study hypothesized that the nature of seed dispersal, relative freedom from herbivory and pest over the 40-year period may have favored species invasion and increased areas at risk of colonization. In addition, the observed abundance and densities of seedlings and saplings indicate prolific seeding, and the seemingly early reproductive maturity favors species colonization. However, the mean distance from the mother tree and the projected areas at risk over the 40 years showed slow species colonization of *T. cumingiana* in QPL. The occurrence of the species may also be attributed to *distance to coast*, *distance to seed sources,* and topographic features of the protected area.

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CONFLICT OF INTEREST

The authors declare that they have no competing interests.

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