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Effects of long term canopy change on regulating ecosystem services in a tropical urban park

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Long-term management of urban parks is critical to maintaining urban sustainability. Regular examination of tree attributes is essential to maintain healthy tree growth, enabling them to deliver regulating ecosystem services (RES). Here, we examined the five-year changes, from 2019 to 2024, in RES and the monetary values provided by urban trees in Chulalongkorn University Centenary Park in Bangkok, Thailand. Using the i-Tree Eco model, we evaluated changes in canopy attributes and three RES: carbon sequestration, air purification, and stormwater runoff reduction. While the total monetary value of RES increased by 126% to 3,491 USD y^{-1} in 2024, 37% of the original trees were lost due to mortality and management practices, resulting in an annual monetary loss of 886 USD y^{-1}. Evergreen trees showed greater increases in canopy attributes and RES compared to deciduous trees. The study revealed that the mortality rate was more than double the default rate assumed in common forecasting models, primarily due to human management rather than natural causes. The findings point out the need to consider human intervention in urban forest management and emphasize that while urban parks have substantial potential for providing ecosystem services, improper management can significantly impair their long-term ecological and economic benefits.

Keywords Regulating ecosystem services, Monetary values, i-Tree eco, Urban parks, Temporal variation

In response to intensifying environmental challenges, cities worldwide are prioritizing the expansion and effective management of urban green spaces. Among these, urban parks—with their substantial tree canopies—play a critical role in providing essential ecosystem services such as air purification, climate regulation, and flood mitigation[1]. Diverse plant species within urban parks create homes for various flora and fauna, promoting biodiversity, which leads to ecological stability. Additionally, urban parks support social cohesiveness and well-being by offering leisure options and enhancing mental health for city dwellers[2]. However, these benefits can vary significantly over time due to dynamic changes in vegetation performance, in particular changes in canopy attributes influenced by urban development, climatic conditions, and vegetation management. Understanding how such long-term changes in canopy affect ecosystem services is crucial for sustainable urban planning, particularly in tropical cities where rapid urbanization interacts with sensitive ecological systems.

Informed decision-making and sustainable urban planning in urban parks depend on the assessment of ecosystem services and their monetary worth. Policymakers may make more targeted investments and wise land-use decisions by measuring the advantages that urban parks offer, such as air purification, climate regulation, and recreational activities[3]. Valuation of the services in monetary terms facilitates the presentation of their significance in economic terms, thus ensuring their consideration in budgetary and policy frameworks[4]. In fact, such analysis was demonstrated through the financial benefits from lower healthcare costs because of urban residents' better mental and physical health resulting from having access to well-maintained parks[5]. However, future growth in the monetary value of tree services will depend on changes in canopy attributes over time. For example, from 2010 to 2030 in the Bronx, NY, PM2.5 removal was projected to increase by 9.8% and 21.6% under high and low tree mortality scenarios, emphasizing the greater benefits of the policies ensuring the long-term survival of newly planted trees[6]. With these regards, ongoing monitoring of tree attributes is essential to

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maintain tree health and ensure that local management goals for ecosystem services and their monetary value are achieved.

Enhancing urban environmental quality and human well-being largely depends on the regulating services provided by urban parks, such as water management, air quality improvement, and climate control[7]. Because of variations in urbanization, environmental stressors, and vegetation maturity, these services differ dramatically from year to year. For example, as trees and vegetation in certain parks grow, their expanding canopies enhance ecosystem services like carbon dioxide capture, urban heat island reduction, and increased atmospheric moisture through transpiration[8]. Long-term and well-established vegetation also has more wide-ranging and complex root systems that improve soil stability and water penetration, thereby enhancing stormwater runoff and lowering the danger of urban flooding[9]. However, urban parks have experienced several environmental stressors, including pollution, climate change, and pest outbreaks, which can eventually limit their capacity to consistently provide these regulating functions. For instance, extended droughts or rising temperatures can stress plant species and impair their ability to perform regulating functions, such as temperature regulation and air purification[10]. Furthermore, depending on how successfully these green spaces are incorporated into the larger urban environment, changes in land use and urban infrastructure may increase or decrease their efficacy. Few studies have evaluated the long-term impacts of canopy changes on ecosystem services in tropical urban settings, particularly in Southeast Asia. To preserve and improve canopy health and guarantee the long-term viability of the regulating services that they offer, regular monitoring and adaptive management techniques are crucial.

To that end, this study aims to investigate how long-term changes of canopy attributes impacted the evaluation and monetary valuation of regulating ecosystem services (RES), including gross carbon sequestration, air purification, and runoff reduction, provided by an urban park. The study site was Chulalongkorn University Centenary Park (CU100 Park) in Bangkok, Thailand. Previously, the RES and associated monetary values were assessed using i-Tree Eco model with field data obtained during the growing season in 2019[11]. Here, we re-measured the canopy attributes of the same trees and other required field parameters and re-evaluated the RES and monetary values to examine their changes since the last investigation. Because urban parks are regularly managed, the analysis will offer information about how certain management techniques may influence the ecosystem services, which may or may not meet the expectations of park designers. Thus, results from this study would draw attention to the need for long-term and regular monitoring of ecosystem services to ensure optimal benefits from urban parks.

Results

Changes in tree population and attributes between 2019 and 2024

During the growing season of 2019, CU100 Park contained 694 planted trees, comprising 399 (57%) deciduous and 295 (43%) evergreen trees, representing 54 distinct species (Fig. 1). The park suffered a 37% loss by 2024, as 260 trees, including 15 species that were present in 2019, were no longer visible. These 15 species were rare in the park, with only 1 to 3 individuals per species recorded (see Supplementary Data, Table S1). However, 434 trees (39 species) persisted, continuing their growth into the growing season of 2024.

In this study, all analyses focused on the 434 surviving trees from 2019, which consisted of 53% (232 trees) deciduous and 47% (202 trees) evergreen species. Overall, all canopy attributes in 2024 significantly increased from those in 2019. The DBH had the greatest increment (59%; $P < 0.001$), followed by height (19%; $P < 0.001$) and LAI (9%; $P = 0.007$) (Table 1).

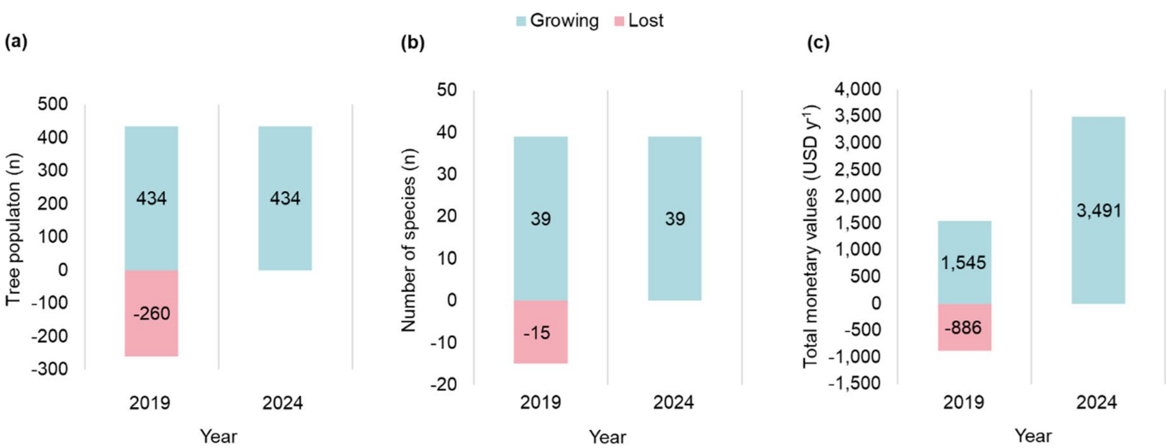

Fig. 1. Stacked bar graphs illustrate (**a**) tree population, (**b**) number of species, and (**c**) total monetary value (USD y^{-1}), which includes all regulating services considered in this study, associated with all trees in 2019 and 2024. The color scheme depicts two distinct conditions: growth (blue) and loss (red). Numbers in the bars indicate the values.

Parameters	Mean ± SD		P value
	2019	2024	
DBH (cm)	14.14 ± 3.96	22.42 ± 10.76	< 0.001
height (m)	7.45 ± 1.60	8.85 ± 2.68	< 0.001
LAI	2.06 ± 1.29	2.25 ± 1.32	0.007
GCS (kg y^{-1})	10.76 ± 6.79 (2.02 ± 1.28)	20.06 ± 13.67 (3.77 ± 2.57)	< 0.001 (< 0.001)
AP (kg y^{-1})	0.13 ± 0.16 (1.08 ± 1.34)	0.35 ± 0.49 (2.97 ± 4.16)	< 0.001 (< 0.001)
RR (m^3 y^{-1})	0.19 ± 0.24 (0.46 ± 0.57)	0.55 ± 0.77 (1.30 ± 1.82)	< 0.001 (< 0.001)
Total monetary values (USD y^{-1})	3.59 ± 2.31	8.04 ± 7.26	< 0.001

Table 1. Mean and one standard deviation of i-Tree eco's input and output parameters, averaged among the 434 trees growing from 2019 to 2024. P values for the statistical comparison at the 0.05 significance level (paired t-test) of the parameters between both years are also shown. DBH, LAI, GCS, AP, and RR denote diameter at breast height, leaf area index, gross carbon sequestration, air purification, and runoff reduction, respectively. Numbers in parentheses are the associated monetary values. The output parameters shown here are expressed as the values per tree.

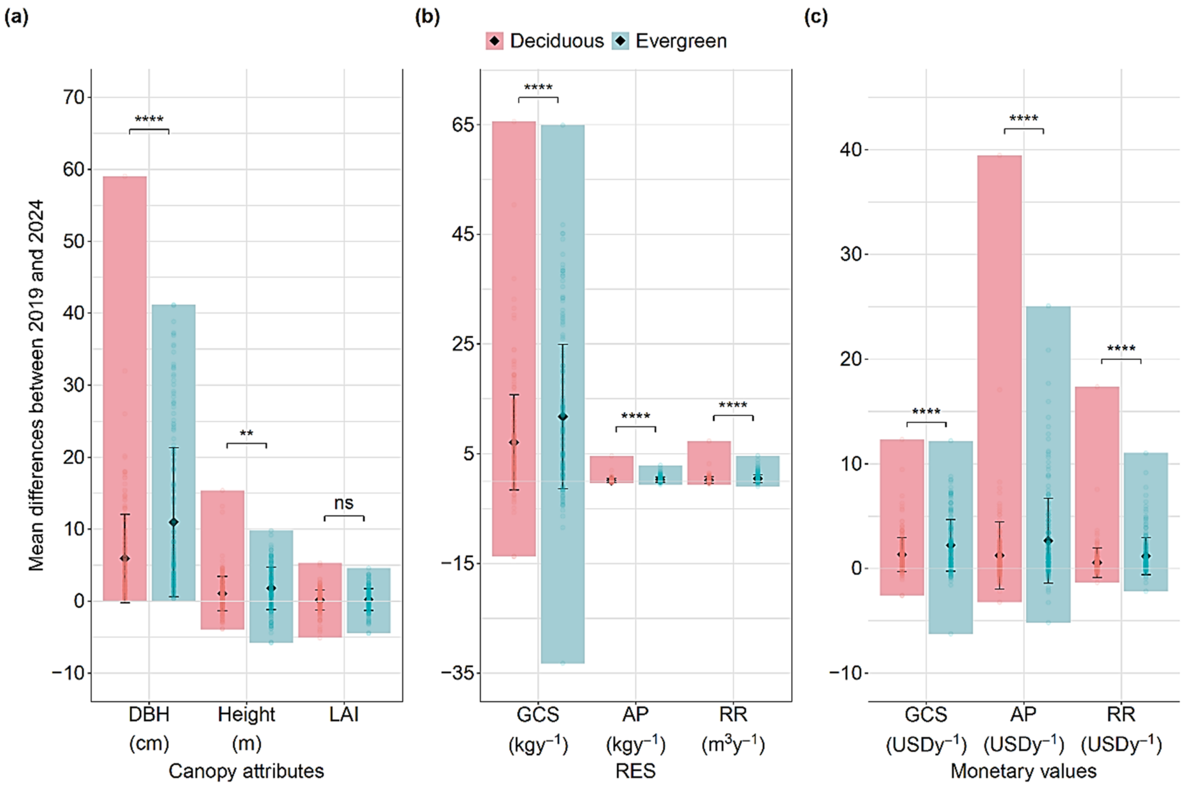

Fig. 2. Bar plots with the individual data points (colored) illustrate mean differences and error bars between 2019 and 2024 in canopy attributes with one standard deviation (black) (**a**), RES (**b**) and their monetary values (**c**) grouped by leaf phenology. P values are for the t-test comparison of the mean difference in parameters between phenological groups. DBH, LAI, GCS, AP, and RR denote diameter at breast height, leaf area index, gross carbon sequestration, air purification, and runoff reduction, respectively. Units are as follows: cm for ΔDBH, m for Δheight, no units for ΔLAI, kg y^{-1} for ΔGCS and ΔAP, m^3 y^{-1} for ΔRR, and USD y^{-1} for monetary values of each RES. Asterisks denote significant results for statistical comparison tests at a significant level of 0.05.

Both deciduous and evergreen trees showed significant increases in these attributes, with evergreen trees demonstrating greater mean changes than deciduous ones, except for LAI (Fig. 2a). For deciduous trees, the mean differences between 2019 and 2024 were 5.94 ± 6.15 cm (47%) in DBH, 1.05 ± 2.40 m (18%) in height, and 0.15 ± 1.39 (48%) in LAI. Evergreen trees, meanwhile, had average increases of 10.98 ± 10.36 cm (74%) in

DBH, 1.80 ± 2.95 m (29%) in height, and 0.23 ± 1.51 (51%) in LAI. However, 16 out of the 39 species exhibited declines in either height, LAI, or both, as indicated by the averages presented in Table 2. Among the dominant species, *Samanea saman*, *Dipterocarpus alatus*, *Afzelia xylocarpa*, *Tabebuia rosea*, *Bauhinia purpurea*, *Homalium tomentosum*, and *Dalbergia cochinchinensis* experienced average height increases of 4.57, 1.12, 1.06, 1.61, 1.82, 2.28, and 0.13 m, respectively. However, the heights of *Millingtonia hortensis* and *Hopea odorata* decreased by 0.17 and 0.01 m as mean values, respectively. For LAI, mean increases were observed in *Samanea saman*, *Dipterocarpus alatus*, *Afzelia xylocarpa*, *Tabebuia rosea*, *Bauhinia purpurea*, and *Homalium tomentosum*, while mean decreases occurred in *Millingtonia hortensis*, *Hopea odorata*, and *Dalbergia cochinchinensis*.

Impacts of tree population and attributes changes on regulating ecosystem services and associated monetary values

The total monetary value of RES for the 434 surviving trees increased from 1,545 USD y^{-1} in 2019 to 3,491 USD y^{-1} in 2024, marking a 126% rise (Fig. 1c). However, the loss of 260 trees resulted in an annual monetary loss of 886 USD y^{-1}, representing a 36% reduction from the total RES value in 2019. Considering a 126% increase in total monetary values, if the 260 lost trees had survived, they could have contributed to the total monetary value of 2,002 USD y^{-1} in 2024.

Among RES, runoff reduction exhibited the highest percentage increase (189%; $P < 0.001$), followed by air purification (169%; $P < 0.001$) and CO_2 sequestration (86%; $P < 0.001$) (Table 1). Similar to the variations in canopy attributes, evergreen trees showed significantly higher mean increases in RES compared to deciduous trees (Fig. 2b and c). On average, the RES (monetary value) provided by deciduous trees increased by 7.11 ± 8.67 kg y^{-1} $tree^{-1}$ (1.34 ± 1.63 USD y^{-1} $tree^{-1}$) for gross carbon sequestration, 0.15 ± 0.38 kg y^{-1} $tree^{-1}$ (1.24 ± 3.21 USD y^{-1} $tree^{-1}$) for air purification, and 0.23 ± 0.60 m^3 y^{-1} $tree^{-1}$ (0.55 ± 1.41 USD y^{-1} $tree^{-1}$) for runoff reduction. Evergreen trees exhibited mean increases of 11.8 ± 13.1 kg y^{-1} $tree^{-1}$ (2.22 ± 2.47 USD y^{-1} $tree^{-1}$), 0.31 ± 0.48 kg y^{-1} $tree^{-1}$ (2.65 ± 4.05 USD y^{-1} $tree^{-1}$), and 0.50 ± 0.75 m^3 y^{-1} $tree^{-1}$ (1.18 ± 1.78 USD y^{-1} $tree^{-1}$), respectively. Among the 39 species, four demonstrated a mean reduction in RES and their associated monetary values that are influenced by LAI (i.e., air purification and runoff reduction) due to decreases in their height and LAI, including *Syzygium cinereum*, *Pterocarpus indicus*, *Millettia leucantha*, and *Lagerstroemia floribunda* (Table 2). In particular, ninety-six trees (22%) exhibited diminished values in air purification and runoff reduction, or both (see Supplementary Data, Table S2).

Discussion

Most research on the temporal changes in RES in urban settings has focused on predicting RES over time in simulated scenarios using the i-Tree Eco application's Forecast model[6,12,13]. The model defaults to a 3% annual mortality rate for healthy trees, which equates to a 15% mortality rate over five years, based on the assumption that this is the natural rate of tree death[14]. However, our results based on the field measurements revealed that one-third of the tree population in CU100 Park perished, resulting in a mortality rate of 37% based on the five-year period.

Not only did natural factors, such as pests, influence the loss of healthy trees and their RES, but, during the past five years, the CU100 Park management itself also exerted impacts. Our observations revealed that a majority of trees in this park are often treated with topping cuts, which can lead to eventual death. According to a previous study, the topping cut can alter tree growth patterns by promoting the emergence of adventitious water sprouts and root suckers and reducing stem diameter growth[15]. At the leaf level, topping cuts were found to increase leaf area while reducing leaf mass per area, which may explain the higher incidence of dieback observed in topped branches[15]. Considering that trees in CU100 Park are regularly treated by this pruning method, the head cutting of the trees may have been the primary cause of the reductions in certain trees' height and LAI, even though the DBH of all trees had increased after five years.

A notable example was observed in 2024 when a topped tree exhibited complete dieback, resulting in zero RES and their monetary value (see Supplementary Data, Table S2). Thus, our findings emphasize that poor pruning practices not only diminish the current economic values of RES but also disrupt the long-term growth of canopy attributes, adversely affecting RES in the future. These findings also emphasize the importance of accounting for pruning methods when estimating urban tree mortality rates to improve the accuracy of RES and monetary value predictions.

From 2019 to 2024, the canopy attributes of evergreen trees increased more than those of deciduous trees. However, only a few dominant evergreen trees in CU100 Park, such as *Samanea saman*, influenced this growth. Due to its larger population and greater DBH, *Samanea saman* accounted for 42% and 49% increases in DBH and height among all CU100 Park trees (see Supplementary Data, Table S3). This effect explains the observed higher average increments of canopy characteristics, RES, and corresponding monetary values in evergreen than deciduous species in this study.

Despite these findings, assumptions in the i-Tree Eco model may lead to discrepancies in RES estimates between evergreen and deciduous species. For evergreen tree cover, the year-round LAI is considered constant at mid-summer, while LAI for deciduous trees is based on frost dates[16]. The model assumes the minimum LAI for deciduous trees to be zero during the winter. During a four-week leaf transitional period between seasons, a modified sigmoidal function estimates daily LAI values for the deciduous tree cover. For leaf biomass, the model assumes that deciduous trees only store carbon in their wood biomass and only calculate wood biomass as they drop their leaves annually[16]. Thus, when applying the model to determine RES by default, deciduous trees may give less RES than they actually do, resulting in inconsistencies in predicted RES and the contribution of evergreen and deciduous species to RES.

	Mean differences between 2019 and 2024									
	Canopy attributes			RES			Monetary values			
Species (n)	ΔDBH	Δheight	ΔLAI	ΔGCS	ΔAP	ΔRR	ΔGCS	ΔAP	ΔRR	ΔTotal
Evergreen species	**10.98**	**1.80**	**0.23**	**11.81**	**0.31**	**0.50**	**2.22**	**2.65**	**1.18**	**6.05**
Pterospermum acerifolium (2)	15.48	2.25	0.80	31.30	0.61	0.97	5.89	5.20	2.30	13.39
***Samanea saman* (65)**	23.39	4.57	0.16	16.84	0.69	1.12	3.17	5.92	2.64	11.73
Saraca thaipingensis (1)	4.82	4.00	1.50	7.80	0.40	0.63	1.47	3.40	1.49	6.36
Ficus racemosa (3)	9.40	2.07	1.00	4.14	0.43	0.68	0.78	3.64	1.61	6.03
Chukrasia tabularis (3)	9.48	1.40	-0.17	20.70	0.16	0.25	3.89	1.35	0.60	5.85
Syzygium cumini (4)	7.78	2.45	0.75	12.02	0.22	0.36	2.26	1.88	0.85	4.99
Fraxinus griffithii (1)	11.09	4.50	1.60	5.76	0.29	0.46	1.09	2.44	1.08	4.61
***Millingtonia hortensis* (39)**	4.96	-0.17	-0.42	19.91	0.06	0.10	3.74	0.52	0.24	4.50
Casearia grewiifolia (1)	5.31	-1.00	0.80	11.93	0.09	0.14	2.24	0.76	0.33	3.33
***Dipterocarpus alatus* (46)**	4.77	1.12	1.30	3.76	0.17	0.27	0.71	1.43	0.63	2.77
Mitrephora tomentosa (4)	3.89	0.60	0.95	1.78	0.18	0.29	0.34	1.57	0.70	2.61
Cinnamomum bejolghota (1)	5.91	-3.50	-0.70	1.36	0.16	0.26	0.25	1.40	0.62	2.27
Elaeocarpus grandiflorus (2)	4.43	0.50	0.15	4.40	0.11	0.18	0.82	0.95	0.43	2.20
Dolichandrone spathacea (1)	6.19	2.40	1.20	5.13	0.06	0.10	0.96	0.52	0.22	1.70
***Hopea odorata* (17)**	3.31	-0.01	-0.21	2.68	0.10	0.15	0.50	0.81	0.36	1.68
Melaleuca cajuputi (5)	6.26	-0.27	-2.56	6.73	0.01	0.03	1.26	0.13	0.07	1.46
Syzygium cinereum (5)	3.65	-1.92	-1.24	5.69	-0.01	-0.02	1.07	-0.12	-0.04	0.91
Saraca indica (2)	1.57	-0.90	-0.35	0.11	0.00	0.00	0.02	0.00	0.01	0.03
Deciduous species	**5.94**	**1.05**	**0.15**	**7.10**	**0.15**	**0.23**	**1.34**	**1.24**	**0.55**	**3.13**
Bombax ceiba (3)	34.66	13.67	2.97	25.07	2.34	3.73	4.71	19.98	8.80	33.49
Shirakiopsis indica (1)	17.87	4.50	2.20	65.59	0.26	0.41	12.33	2.22	0.97	15.52
Phyllanthus emblica (1)	8.78	4.30	1.30	17.15	0.51	0.81	3.22	4.33	1.91	9.46
Azadirachta indica (2)	9.95	-1.00	0.15	24.49	0.37	0.59	4.61	3.12	1.39	9.12
Barringtonia acutangula (2)	7.39	2.55	1.65	8.16	0.53	0.84	1.54	4.49	1.97	8.00
Artocarpus lacucha (4)	10.94	0.30	-1.60	5.56	0.41	0.66	1.05	3.49	1.55	6.08
Terminalia bellirica (6)	5.79	1.43	0.78	12.08	0.28	0.45	2.27	2.40	1.07	5.74
***Afzelia xylocarpa* (23)**	3.33	1.06	0.32	12.40	0.15	0.23	2.33	1.25	0.55	4.14
Crateva religiosa (4)	4.04	2.30	1.18	4.76	0.26	0.41	0.89	2.17	0.96	4.03
***Tabebuia rosea* (51)**	9.24	1.61	0.83	6.94	0.18	0.29	1.30	1.57	0.70	3.57
***Bauhinia purpurea* (12)**	9.11	1.82	0.12	12.67	0.06	0.10	2.38	0.54	0.25	3.17
Xylia xylocarpa (12)	5.08	0.02	-0.01	4.71	0.18	0.29	0.88	1.52	0.68	3.08
***Homalium tomentosum* (13)**	4.14	2.28	0.45	5.64	0.10	0.17	1.06	0.90	0.41	2.37
Terminalia chebula (7)	4.72	1.99	-0.33	4.89	0.09	0.14	0.92	0.77	0.34	2.03
Elaeocarpus hygrophilus (6)	4.50	-0.42	0.22	5.08	0.04	0.07	0.95	0.37	0.17	1.49
Pterocarpus indicus (1)	12.85	-0.10	-0.80	13.47	-0.09	-0.13	2.53	-0.79	-0.32	1.42
***Dalbergia cochinchinensis* (61)**	3.57	0.13	-0.32	2.83	0.05	0.08	0.53	0.39	0.19	1.11
Millettia leucantha (2)	6.02	-0.85	-0.85	22.18	-0.26	-0.39	4.17	-2.19	-0.92	1.07
Crateva adansonii (5)	2.08	-0.34	-0.72	5.03	0.00	0.01	0.94	0.03	0.01	0.99
Lagerstroemia floribunda (12)	2.28	0.08	-0.92	5.08	-0.03	-0.05	0.96	-0.28	-0.12	0.55
Terminalia alata (4)	1.41	-0.90	-0.70	2.12	0.00	0.01	0.40	0.05	0.02	0.47

Table 2. Mean per-tree differences in canopy attributes, regulating ecosystem services (RES), and associated monetary values between 2019 and 2024, by tree species, ranked in decreasing order of total monetary change. Values represent changes (Δ) calculated as 2024 minus 2019 and normalized by the sample size (n) of each species. Monetary values represent the economic valuation of each corresponding service, with ΔTotal indicating the summed change across all services. The color gradient indicates the magnitude of change, with blue representing the highest and red the lowest monetary values. Canopy attributes include diameter at breast height (DBH), tree height, and leaf area index (LAI). Regulating ecosystem services include gaseous pollutant removal (GCS), air pollution removal (AP), and rainfall interception (RR). Bold species names denote the nine dominant species. Orange and green shading indicate deciduous and evergreen species, respectively. Units are cm for ΔDBH, m for Δheight, unitless for ΔLAI, kg yr^{-1} for ΔGCS and ΔAP, m^3 yr^{-1} for ΔRR, and USD yr^{-1} for the monetary values of each RES.

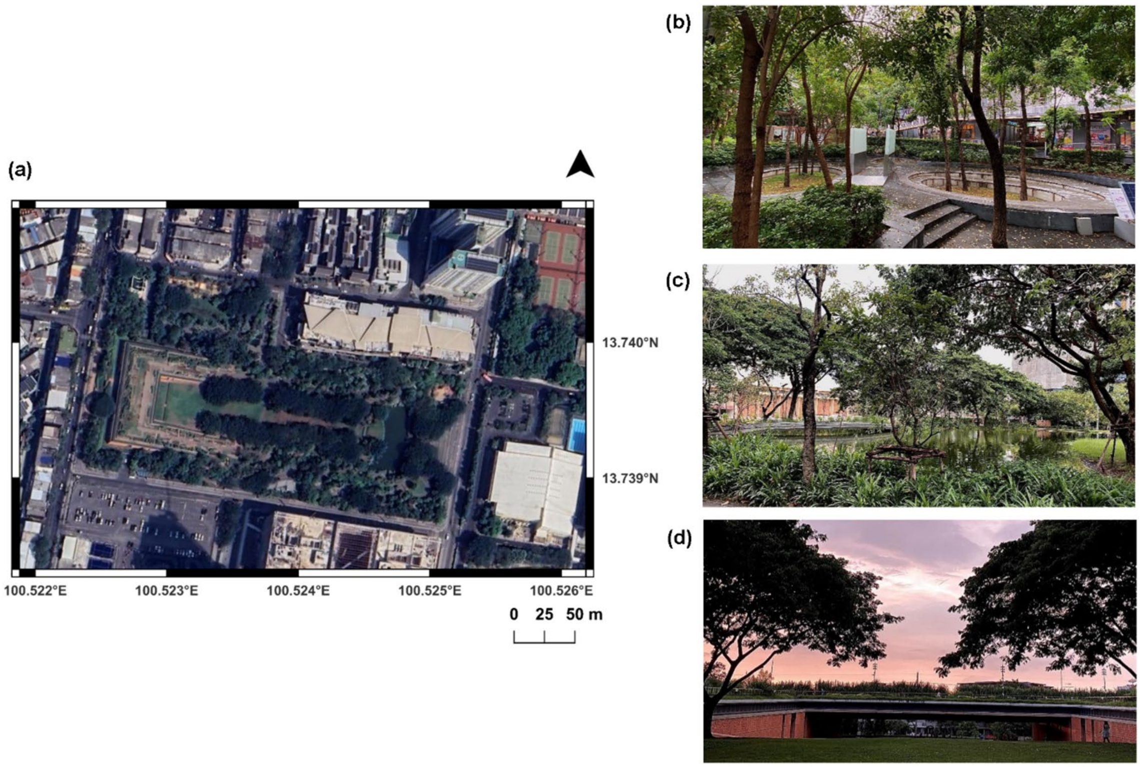

Fig. 3. Overview of CU100 Park (**a**) in Bangkok, Thailand, along with its various sections: (**b**) *Millingtonia hortensis* zone featuring a meditation pathway, (**c**) the water retention zone situated beneath the 3° tilted green roof, and (**d**) the upper green roof. The map (**a**) was created using QGIS (Quantum Geographical Information System) software version 3.40[20].

To address these inconsistencies, we recommend that future studies explore two critical aspects. First, investigate how factors beyond leaf phenology, such as tree size and species dominance, influence RES and associated monetary values in urban settings. Secondly, we recommend studying how seasonal changes in canopy attributes affect RES and their economic value estimates to understand how leaf phenology affects RES modeling in different seasons.

Methods

Study site

The study area is Chulalongkorn University Centenary Park (CU100 Park), a 4.48 ha park located in the center of Bangkok, Thailand (13°44′22″ N, 100°31′25″ E; Fig. 3). The site experiences a monsoonal tropical climate with a mean annual temperature of 28 °C and total annual rainfall of approximately 1600 mm based on data from 1990 to 2020[17]. To expand the amount of green space in the Bangkok metropolitan region, according to the Green Bangkok Project[18], the park was created in 2016, located in the heart of a business district and surrounded by roads and high-rise buildings to the south. With the installation of subterranean drainpipes that remove excessive water during flooding, the green roof was purposefully built with a 3° tilt to allow rainwater to drain into the retention pond below. Furthermore, the socio-economic benefits are addressed through the inclusion of communal spaces for residents to participate in, such as pathways for walking, a designated pathway for meditation, and research activities. Visitors generally frequented the park one to three times per week[19]. The staff of CU100 Park manages the park by watering trees daily with sprinklers and hoses on days without rain, applying chemical fertilizer, regularly controlling pests, trimming tree crowns, and head-cutting trees. The five common species found in 2019 were *Dalbergia cochinchinensis*, *Tabebuia rosea*, *Samanea saman*, *Millingtonia hortensis*, and *Dipterocarpus alatus*. The understory trees contributed 5.7% to the total tree population. The physical characteristics of the park have remained constant since 2019[11], including the impervious area and the sparse tree cover area.

Field measurements and environmental data

We initially examined the tree data from Yarnvudhi, et al.[11] 2019 analysis, as this study centers on contrasting RES estimates from 2024 with those from a previous period in 2019. Due to regular management of trees in the CU100 Park, some trees were removed and replaced by new ones. The procedure led to the survival of 63% of the total number of trees from 2019 until 2024. Because the 2019 measurements were conducted during the growing season period from September to October 2019, we repeated the measurements of the existing trees in a similar period from August to September 2024. We analyzed the importance of value index (IVI) to determine the dominant species in the site. The IVI value of tree species is calculated as the combination of relative density (relative density = $\frac{\text{Frequency of a species}}{\text{Total frequency of all species}} \times 100$) and relative dominance (relative dominance = $\frac{\text{Dominance of a species}}{\text{Total dominance of all species}} \times 100$)[21].

Field data were collected based on the protocols developed by the USDA Forest Service[22]. We conducted a complete inventory, surveying all trees in the park with DBH greater than 2.5 cm. Tree circumference was measured at 1.37 m above ground using a tape and then converted to diameter at breast height (DBH) based on the assumption of circular form of tree stems. Tree height was measured as the height from the ground to the top of the tree. Crown height was measured as the height difference between the live top of the tree and the crown base, and crown width was measured as the average width of the crown in the north-south and east-west directions. The tree height, crown height, and crown width were measured by a rangefinder (Nikon Forestry Pro, Nikon Inc., Tokyo, Japan). The Nikon Forestry Pro II Laser Rangefinder is factory-calibrated with no user calibration mode. However, due to the nuances in measuring the height of trees in the park, whose views tend to be obstructed, we typically measured from more than one direction to ensure the precision of the height measurements. Crown health was also recorded based on a visual assessment of the number of dead branches in a tree's crown and can be estimated as dieback and missing. The crown light exposure (CLE) measure for trees is determined by the number of sides and tops exposed to sunlight. For parameters assessed through visual inspection (crown missing, dieback, and CLE), the crew was trained using the USDA Crown-Condition Classification guidelines[23]. Each person was assigned to assess the same parameter throughout the measurement process to minimize bias from visual judgment. Detailed information regarding the protocol of field measurements is fully described in the i-Tree manual[16].

In the 2019 analysis[11], RES was estimated by the i-Tree Eco model using environmental data from a station, which was 17 km away from CU100 Park. Here, we reran the i-Tree Eco model using other closer environmental stations, thus allowing us to only focus on the differences in canopy attributes between the two years. Hourly meteorological data (e.g., temperature, relative humidity (RH), wind speed, and air pressure) was collected from the nearest National Climatic Data Center (NCDC ID: 484550 − 99999; 4.5 km east of the study area). Hourly concentrations of nitrogen dioxide (NO_2), ozone (O_3), particulate matter less than 10 microns (PM_{10}), particulate matter less than 2.5 microns ($PM_{2.5}$) (i-Tree Eco Station ID: 001; 2.5 km north of the site), and sulfur dioxide (SO_2) (i-Tree Eco Station ID: 002; 3 km west of the site area) were obtained from the nearest station serviced by the Pollution Control Department. To ensure comparability, we reviewed the meteorological and air pollution data for both years and found that the yearly average values for temperature, RH, $PM_{2.5}$, PM_{10}, CO, NO_2, O_3, and SO_2 remained within ± 10% between 2019 and 2024. Therefore, we concluded that the environmental and climatic conditions in 2024 were similar to those in 2019, so that all changes in ecosystem services and benefits could be attributed to canopy attributes.

Calculation of RES and their monetary values using i-Tree eco

For every species, the i-Tree Eco model estimates the monetary values associated with carbon sequestration, prevented runoff, and pollutant removal. Required input data comprises those obtained from the field measurements, including (1) plot information (% tree cover), (2) individual tree attributes (species, stem diameter, height, crown size, missing crown canopy and die-back, and crown light exposure), (3) weather data, and (4) air pollution data. To meet our main objective, which is the comparison between old and new RES, we followed the same calculation scheme as described in the previous work[11], albeit using new canopy attributes in 2024 as previously described.

The calculations of RES by the i-Tree Eco are summarized as follows. The i-Tree Eco model first estimates tree biomass and leaf area. The i-Tree Eco estimates the leaf area in three ways: (1) To find the leaf area of open-grown trees (those with a CLE of 4 or 5), the model uses regression equations based on crown height, crown width, and shading factor shown by Eq. (1)[24]; (2) For light-limited trees (those with a CLE of 0 or 1), i-Tree Eco uses Eq. (2) to find the leaf area of forests, which is based on the Beer-Lambert Law; and (3) For trees with crown light exposure = 2–3, it finds the average of the leaf area from the open-grown (crown light exposure = 4–5) and closed canopy equations (crown light exposure = 0–1). i-Tree Eco adjusts the leaf areas by a percentage of crown dieback to arrive at the final leaf area. The model then uses the final leaf area to calculate LAI, which can be calculated as follows: (1) LAI_p: total population leaf area divided by total study area size, and (2) LAI_c: leaf area standardized per unit tree cover: $LAI_c = LAI_p$/percentage tree cover.

$$\ln Y = -4.3309 + 0.2942H + 0.7312D + 5.7217S + (-0.0148)C \tag{1}$$

where Y is leaf area (m^2), H is crown height (m), D is average crown diameter (m), S is the average shading factor for the individual species (percent light intensity intercepted by foliated tree crowns)[16], and C is based on the outer surface area of the tree crown ($\pi D(H + D)/2$). If the shading coefficients (percentage of light intensity intercepted by foliated tree crowns) that are used in the regression did not exist for an individual, then the coefficients that are specific to the species, genus, family, order, subclass, or class were used.

$$LA = \frac{\ln(1 - X_s)}{-k} \pi r^2 \tag{2}$$

where X_s = shading coefficient, k = 0.52 for conifers and 0.65 for hardwoods[16], and r = crown radius.

The model calculates leaf biomass by converting leaf area estimates using species-specific measurements of leaf dry weight (g) per unit leaf area (m^2)[25,26]. Each measured tree's total dry-weight biomass is calculated using allometric equations from the literature[25,27]. If no biomass equation is found for an individual species, the average results from equations of the same genus or the next phylogenetic level are used as they are available. Equations that predict aboveground biomass are converted to whole tree biomass based on a root-to-shoot ratio of 0.26[28]. Since deciduous trees lose their leaves throughout the year, i-Tree Eco only accounts for tree biomass for those trees.

After estimating the biomass, i-Tree Eco then calculates carbon storage by multiplying a factor of 0.5 (average proportion of carbon in plant tissue)[29] by the total dry biomass. By using growth rates specific to each species, the model predicts the carbon that will be stored for the next year. The model then contrasts the carbon storage in the current year (year 0) with the carbon storage in the next year (year 1) to estimate the annual carbon sequestration. Estimates for air pollution removal are calculated using a dry deposition model with leaf area, pollution concentration, and meteorological data[30]. For avoided runoff, based on leaf and bark area and local hourly weather data, the model estimates hourly rain interception, evaporation from leaf surfaces, potential evapotranspiration, transpiration, and avoided runoff values[26,31,32]. Lastly, i-Tree Eco evaluates the monetary values of those RES based on the U.S. EPA regulation, including social cost for carbon sequestration, health value for air purification, and the U.S. national average cost of stormwater control and treatment for runoff reduction[33–35]. Detailed information about the calculation of those RES and associated monetary values was also fully described in the i-Tree manual[16]. Although the model offers reliable estimates, its results are influenced by the quality of local input data and the assumptions embedded in allometric equations and pollutant removal coefficients. We would perform RES estimations and their monetary values using i-Tree Eco version v6.0.35.

Statistical analysis

We examined the changes in the three canopy attributes of the trees in the park over time, as well as the monetary values associated with the three RES they provide. The parameters included DBH, height, LAI, gross carbon sequestration, air purification, runoff reduction, and associated monetary values. The differences in these parameters between 2019 and 2024 were presented as mean ± standard deviation (SD). We then used a paired t-test with a significance level of 0.05 to test for the significance of the difference between two years. We also performed the paired t-test to analyze the changes for both deciduous and evergreen trees and further investigated the differences in these parameters in all 39 tree species. All statistical analyses were conducted in R 4.2.2[36].

Data availability

The datasets collected and analyzed in the current study are available from the corresponding author upon reasonable request.

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Author contributions
N. Kasikam participated in conceptualization, formal analysis, investigation, methodology, writing the original draft, and revising the manuscript. P. Tor-ngern participated in conceptualization and provided supervision in analysis and interpretation, writing, reviewing, and editing. A. Yarnvudhi, N. Leksungnoen, and T. Näsholm contributed to reviewing and commenting on drafts and the final version of the manuscript.

Declarations

Competing interests
The authors declare no competing interests.

Additional information
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