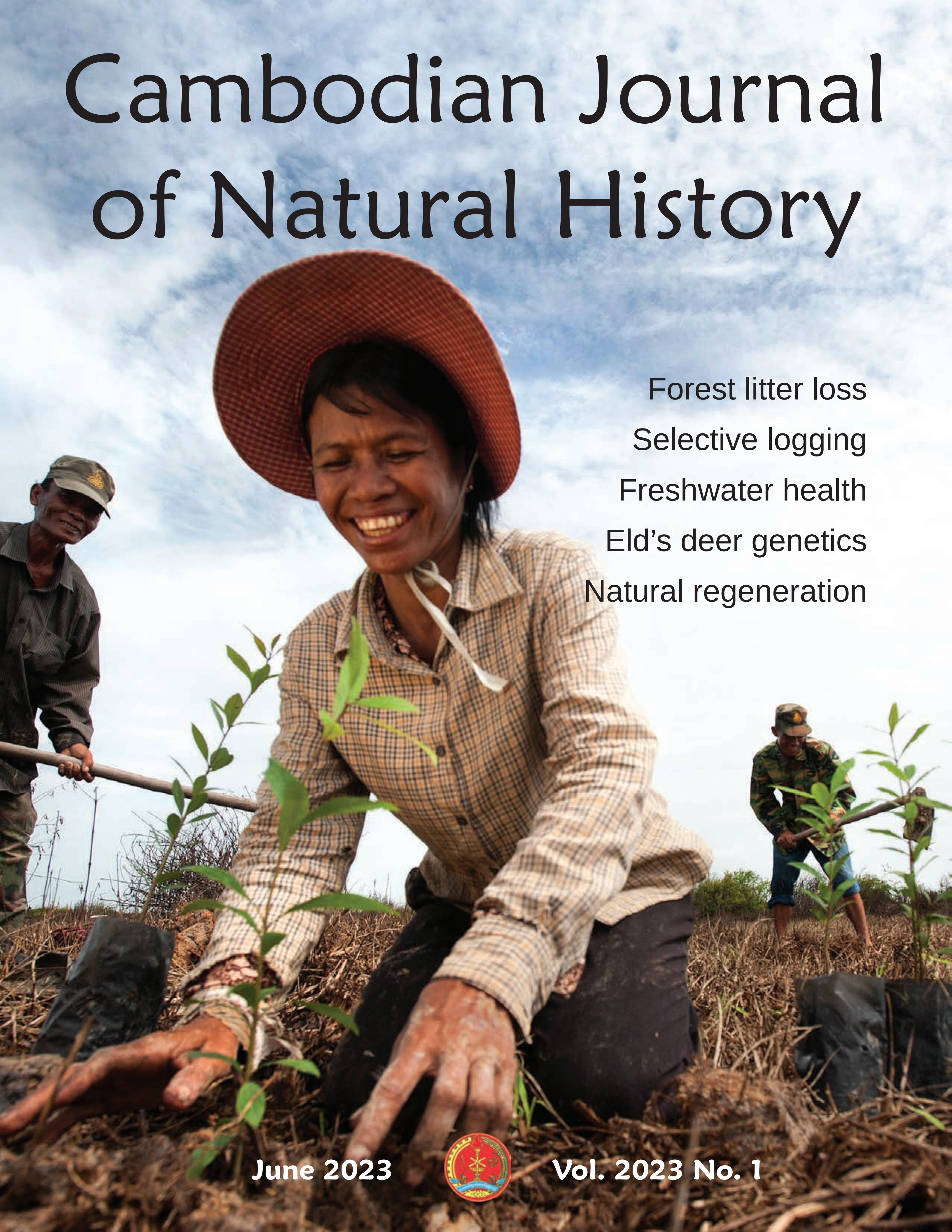
Cambodian Journal of Natural History



Forest litter loss

Selective logging

Freshwater health

Eld's deer genetics

Natural regeneration

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Cover image: Villagers and local authorities planting seedlings of *Barringtonia* spp. as part of reforestation efforts in Siem Reap Province (© Jeremy Holden).

Short Communication

Litter loss in Cambodian evergreen forests is mainly caused by soil macrofauna feeding

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Deforestation and forest degradation are ongoing issues in Cambodia (MoE *et al*., 2020). In lowland dry evergreen forests in Kampong Thom Province, many large-diameter trees of Dipterocarpaceae species e.g., *Dipterocarpus costatus* C.F.Gaertn. (*chhoeuteal bankouy* in Khmer) and *Anisoptera costata* Korth. (*phdiek*), have been felled (Ito *et al*., 2010). Selective logging results in microsites with different degrees of disturbance. We examined the weight loss of leaf litter to verify whether the decomposition rate of leaf litter differs among microsites at selective logging sites.

Our study was conducted in lowland dry evergreen forests in Kampong Thom Province in central Cambodia (Fig. 1). These forests developed on sandy alluvial plains, where soils are deep (Ito *et al*., 2021). Mean annual precipitation and temperature are 1,625.8 mm and 27 °C (Kabeya *et al*., 2021), respectively. The monthly average temperature range is 24–29 °C (Chann *et al*., 2011). The seasonal tropical climate, which is governed by monsoons, has been described in detail elsewhere (Ito *et al*., 2021; Kabeya *et al*., 2021).

Experiments were conducted on 12 plots located within a rectangle of approximately 2 km (east–west) × 4 km (north–south). Each plot was set up on the site of a dipterocarp tree logging area that was cut during the 2007–2008 dry season. The plots were established in June 2018; thus, approximately ten years had passed from logging to plot establishment. Each plot comprised five subplots: control, stump, timber, unused trunk and crown (Fig. 2). The control was selected in a direction opposite to that of logging and in an area with little human disturbance. In the stump subplot, sawing operations were conducted after felling. The lumber was removed from the forest, leaving a large amount of sawdust at the site (Fig. 3). The sawdust had disappeared from the forest by the time of the survey, although decaying wood blocks remained from the sawmilling operations (Fig. 4). Logging roads for lumber removal often passed near the stump and timber subplots (Fig. 5). The quantity of litter on the forest floor was examined near locations where

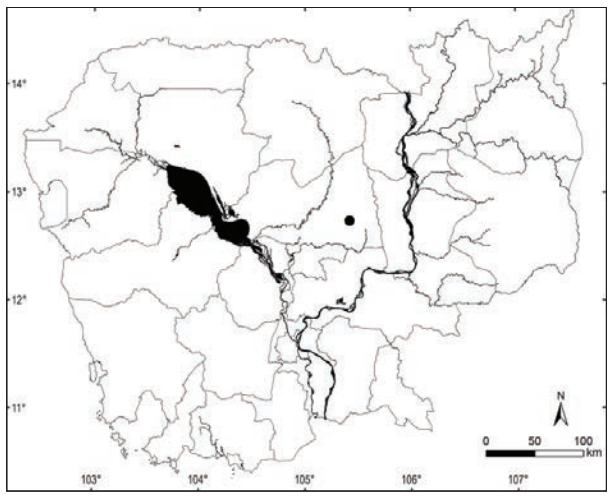

Fig. 1 Location of the study site (solid circle) in Cambodia.

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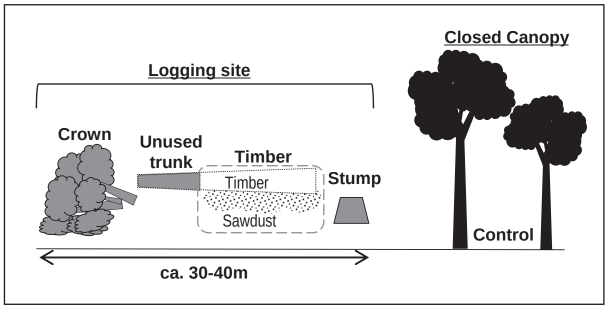

Fig. 2 Schematic diagram of the five subplots: control, stump, timber, unused trunk, and crown. Timber and sawdust may have been deposited in the timber plot during logging operations, but were not present during the survey.

Fig. 3 A) Sawing operations and B) sawdust deposited at the logging site.

Fig. 4 Study site ten years after logging. No sawdust remained, although decaying wood blocks from the sawmilling operation were still present.

Fig. 5 Logging roads near the stump and timber subplots were maintained even ten years after logging.

leaf litter bags (described below) were placed. Organic matter was collected from within a 50 × 50 cm frame and dry weights were measured. The survey dates were 6–13 December 2018 and 7–10 December 2019, depending on the survey plot. The total dry weight of fallen leaves and dry branches did not differ among subplots, whereas the dry weights of bark and wood blocks were significantly higher in the timber subplot than in the unused trunk and control subplots (Table 1).

Leaf litter bags used for leaf decomposition experiments were made of vinylon cheesecloth (polyvinyl alcohol fiber, #500; UNITIKA, Japan), with an opening size of 1.0 mm. Each litter bag was stapled with 0.2 mm thick cold-resistant vinyl numbered tape and packed with fallen dipterocarp leaves weighing approximately 5 g (air-dry weight). Air-dry weight was converted to oven-dry weight using a preliminary sample. Leaves were identified as *D. costatus*, *A. costata*, *Vatica odorata* and *Hopea recopei*. Further details on leaf litter collection are provided in Table 2 and Fig. 6a.

Collected leaves were sorted into leaves and bags and their oven-dry weights were measured. Collection bags installed for the dry season (2.5 months, Table 1) remained white until the time of collection, whereas those collected after the rainy season (at 9.5 and 12 months) had microscopic amounts of soil trapped in the gaps between the fibers (Fig. 6b). Consequently, some bags increased in weight during the collection period (Fig. 7).

Unexpectedly, nearly all bags were damaged (605 of 610) and many were partially lost (Figs 7, 8, 9a). The underside of bags in contact with the ground were more prone to extensive damage and loss (Figs 8, 9b) and when a large part of a bag was lost, its upper and lower sides were often equally damaged (Fig. 8). Number tapes were frequently also partially lost; some remaining number tapes had semi-circular gouges with a radius of approximately 2 mm, which were presumed to be bite marks (Fig. 9c). Thus, we infer that soil macrofauna with a bite width of approximately 2 mm had foraged on the litter bags and surrounding leaves. We also observed sand corridors or clusters at or near the undersides of the damaged nets (Fig. 9d). These observations may imply that the bag damage and loss were caused by wood- and/or litter-eating termites.

During the field surveys, we did not observe termites or any other soil animals foraging on litter bags installed at the site. However, termite mounds were found, especially near stumps (Fig. 9e) and termites were observed feeding on timber in the same forest as the study site (Fig. 9f). Termites are major ecosystem engineers responsible for decomposing wood and leaf litter (Abe *et al*., 2000; Ohkuma, 2003). Termites have been reported in the seasonal tropics of the Indochina Peninsula (Harris, 1968), including in some ecological studies of forest (Vietnam: Vu *et al*., 2007; Thailand: Takematsu

Table 1 Dry weight (Mg ha^{-1}) of litter on the forest floor according to study subplots. Data are means ± standard deviation (range). Different letters among columns indicate significant differences according to analysis of variance (ANOVA: p<0.05).

Item	Control	Stump	Timber	Unused Trunk	Crown	p
Total	4.2 ± 2.2 (1.8–9.2)	4.6 ± 3.6 (1.8–15.4)	6.8 ± 6.7 (1.7–23.9)	3.5 ± 1.3 (1.7–6.1)	3.6 ± 1.0 (2.0–5.3)	0.1361
Leaves	2.0 ± 0.8 (1.0–3.5)	1.4 ± 0.6 (0.7–2.8)	1.6 ± 1.7 (0.6–6.8)	1.4 ± 0.6 (0.9–2.5)	1.5 ± 0.7 (0.7–3.3)	0.6879
Branches	1.5 ± 1.5 (0.2–5.5)	0.7 ± 0.5 (0.1–1.9)	0.8 ± 0.5 (0.2–2.0)	1.7 ± 1.5 (0.5–5.1)	1.1 ± 0.8 (0.1–2.9)	0.1037
Bark & wood blocks	0.4 ± 0.7^{b} (0.0–1.8)	2.4 ± 4.0^{ab} (0.0–14.0)	4.4 ± 6.9^{a} (0.0–22.3)	0.4 ± 0.8^{b} (0.0–2.1)	0.9 ± 1.0^{ab} (0.0–3.3)	0.0244

Table 2 Duration of leaf litter bag installation.

Duration (months)	Installation Date	Collection Date	Period (days)	Season	n
2.5	13 December 2018	2 March 2019	79–80	Rainy	10
9.5	2–3 March 2019	7–11 December 2019	280–284	Rainy, briefly dry	300
12	2–3 March 2019	26–27 February 2020	360–362	Rainy and dry	300

Fig. 6 Leaf decomposition experiments using leaf litter bags. A) Installation and B) collection of leaf litter bags at the same site.

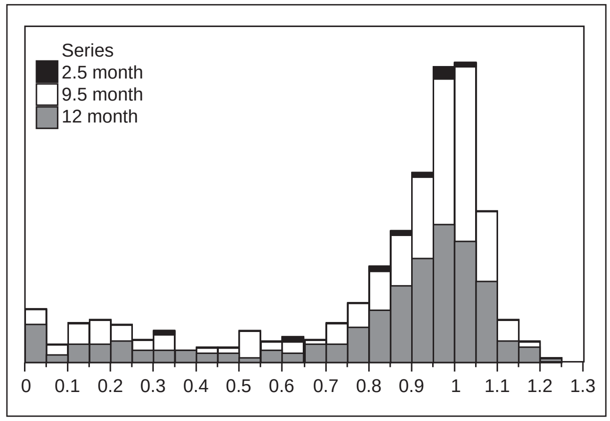

Fig. 7 Histogram of weight ratios of litter bags before and after installation (after/before).

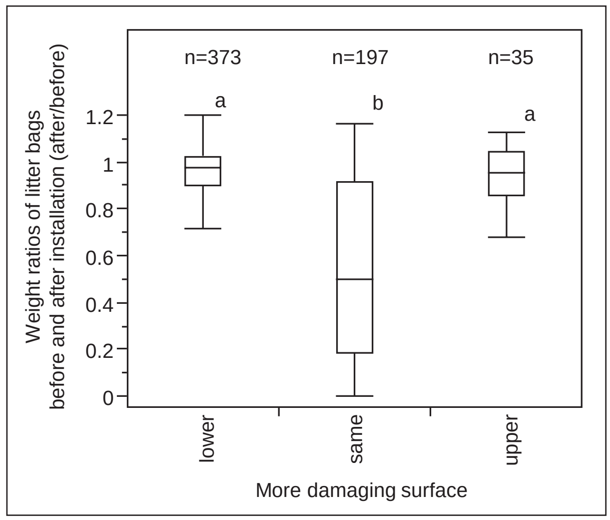

Fig. 8 Relationship between litter bag damage and installation position, and between weight ratios before and after installation. Plots show the median, 25% and 75% quartiles, and range of the data. Different letters among columns indicate significant differences according to analysis of variance (ANOVA: $p<0.05$).

& Vongkaluang, 2012). Termite research in Cambodia is primarily concerned with damage to buildings (e.g., Megna & Liotta, 2015). Only a few studies have examined the ecology of forest termites, either as food consumed by jackals (Kamler *et al*., 2021) or in relation to a new beetle species (Maruyama, 2012). Although not conducted in our study, comparison of termite species composition, population density, and ecological traits between the logging site and surrounding undisturbed sites would contribute to clarifying the roles of termites in Cambodian forest ecology. Further research on termite ecology in Cambodian forests is needed.

The residual weight ratio of leaves i.e., the ratio of leaf litter weight in bags after installation compared to before installation, was clearly related to bag disappearance (Fig. 10). As the degree of bag residuals decreased, the degree of leaf litter residuals decreased exponentially. Thus, we infer that wood/litter-eating termites entered the bags through holes made by chewing and preferentially foraged on the trapped leaves. This suggests that litter loss in Cambodian evergreen forests is mainly caused by soil macrofauna feeding, but it is also possible that soil organisms with body sizes that can pass through a mesh size of 1 mm (microfauna and mesofauna) could be contributing to the reduction in leaf weight. However, it is not possible to quantify the relative contributions of micro- vs. meso- vs. macrofauna to leaf weight reduction with our data. This should be clarified through leaf litter

Fig. 9 A) Damaged litter bags and number tapes at study site, B) Underside of a severely damaged bag that was in contact with ground surface, C) Semi-circular gouge in a damaged number tape, D) Soil clusters, likely derived from soil animals, were found on ground surface after litter bags were collected, E) Termite mounds near stumps, F) Termites feeding on rotten timber.

decomposition tests with graded mesh sizes using robust materials that are not damaged by soil animals.

We developed generalized linear models to predict the weight ratios of litter bags and leaves before and after installation using subplots and installation period as independent variables and plot as a random effect. All statistical analyses were performed using JMP v10.0 statistical software (SAS Institute Inc., USA). Due to the limited number of samples collected at 2.5 months, we used only the data from samples collected at 9.5 and 12 months. Subplots significantly predicted the weight ratios of litter bags (p<0.0001), whereas installation period had no significant effect. Timber subplots showed

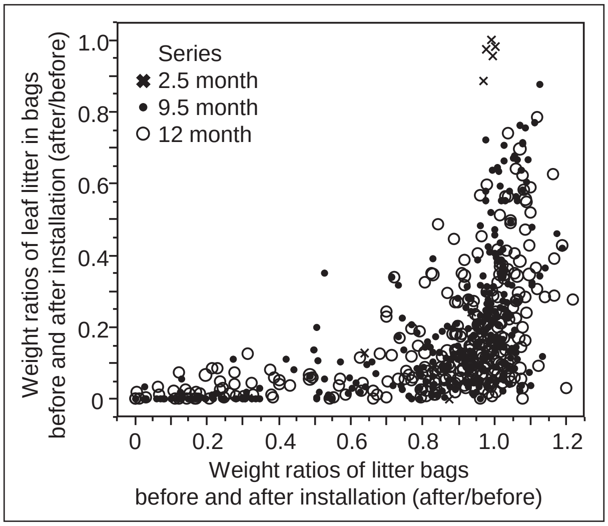

Fig. 10 Relationship between residual weight ratios of litter bags and leaves in bags.

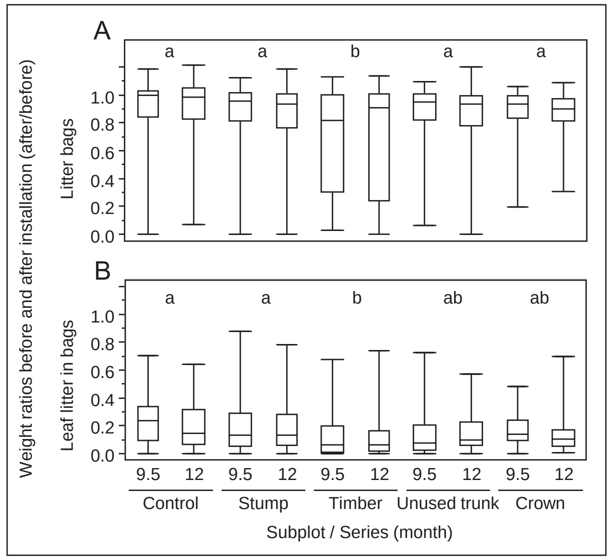

Fig. 11 Residual weight ratios of A) litter bags and B) leaves in bags according to subplot and sampling duration. Plots show the median, 25% and 75% quartiles, and range of the data. Different letters among columns indicate significant differences according to analysis of variance (generalized linear model: $p<0.05$).

significantly lower ratios than the other subplots (Fig. 11a). Subplots significantly predicted the weight ratios of leaves (p=0.0003) and installation period was also significant (p=0.0423). The timber subplots had significantly lower ratios than the control and stump subplots (Fig. 11b). Weight ratios were significantly lower for leaves collected at 12 months (least-squares mean=0.16) than for those collected at 9.5 months (0.19). Significantly greater bag and leaf sample losses in the timber subplots may have been caused by termite density. Wood-feeding termites were found to be more abundant in recently logged areas in Amazonia (Azevedo *et al*., 2021). It is possible that the large amount of sawdust left at the timber subplot in the past attracted termites, leading to higher termite density at that plot.

There were no differences in litter accumulation such as fallen leaves and branches on the forest floor between subplots (Table 1). Variability in the degree of leaf litter loss from the bags was lower among subplots than among bags (Fig. 11). The scale of the logging site was several tens of meters (Fig. 2). The distance from some termite nests to feeding areas was on a similar or greater scale (Abe, 1979; Hoare & Jones, 1998), which suggests that termites move through the forest floor to forage for fallen leaves. Although logging of large-diameter trees causes significant heterogeneity in the forest floor environment, the material cycle originating from plant litter supplied to the forest floor by termites or other leaf litter foragers migrating across the forest floor is likely to be relatively uniform. Selective logging may have a lower long-term impact on nutrient cycling and decomposition than other anthropogenic disturbances (Azevedo *et al*., 2021). Our results are consistent with this finding.

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Genetic diversity of Eld's deer *Rucervus eldii siamensis* populations captive-bred at Phnom Tamao Wildlife Rescue Centre, Takeo, Cambodia

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មូលន័យសង្ខេប

វិបុលកម្ម (ការបង្កាត់ពូជ) ជាយុទ្ធសាស្ត្រអភិរក្សត្រូវបានស្នើឡើងជាទូទៅសម្រាប់ប្រភេទដែលមានប៉ូពុយឡាស្យុងធ្លាក់ចុះខ្លាំងពេក ដើម្បីឲ្យមាននានាភាពនៃសេនេទិចក្នុងព្រៃ តួយ៉ាងដូចជាប្រភេទរងគ្រោះសត្វរមាំង ក្នុងប្រភេទរង *Rucervus eldii siamensis*។ កម្មវិធីទាំងនេះទាមទារការគ្រប់គ្រង និងការថែរក្សាយ៉ាងទូលំទូលាយនូវភាពចម្រុះសេនេទិចខ្ពស់ក្នុងប៉ូពុយឡាស្យុងដែលបង្ខាំងក្នុងដែនកំណត់។ មជ្ឈមណ្ឌលសង្គ្រោះសត្វព្រៃភ្នំតាម៉ៅ និងព្រៃឈើព័ទ្ធជុំវិញក្នុងខេត្តតាកែវនៃប្រទេសកម្ពុជា ជាកន្លែងបង្ខាំង និងប្រលែងប៉ូពុយឡាស្យុងសត្វរមាំងបង្កាត់ ដែលជាកូនចៅរបស់សត្វរមាំងពីរក្បាលបានរឹបអូសពីការជួញដូរសត្វព្រៃខុសច្បាប់នាកំឡុងឆ្នាំ២០០១។ ពេលដែលប៉ូពុយឡាស្យុងបានកើនឡើង វាត្រូវបានបំបែកជាពីរហ្វូងដែលដាក់នៅដាច់ដោយឡែកពីគ្នារហូតដល់ក្រុមតូចៗត្រូវបានប្រលែងនៅឆ្នាំ២០១៨។ យើងបានធ្វើការវាយតម្លៃទៅលើកម្រិតនៃការបង្កាត់ជិត និងភាពសម្បូរបែបនៃសេនេទិចរបស់ប៉ូពុយឡាស្យុង (សត្វ២៧ ក្បាល) ដោយការធ្វើតេស្តមើលភាពចម្រុះនៃសេណូមរបស់វាឡើងវិញ (re-sequencing) លើសំណាកឈាមសត្វរមាំងចំនួនប្រាំបីក្បាល។ កម្រិតទាបនៃភាពចម្រុះសេនេទិចត្រូវបានរកឃើញក្នុងសត្វទាំងនេះ។ អត្រាអេតេរ៉ូស៊ីកូតមធ្យម និងភាពចម្រុះនៃនុយក្លេអូទីត (π) គឺ $5.538 \times 10^{-6} \pm 1.854 \times 10^{-6}$ និង 5.475×10^{-5}។ តម្លៃមេគុណនៃការបង្កាត់ជិតជាមធ្យម (F_{ROH}) គឺ 0.026 ± 0.060 ហើយឯកត្តៈភាគច្រើនជាប់សាច់ញាតិជំនាន់ទី២។ កំណត់ត្រាប្រជាសាស្ត្រសត្វរមាំងពីខែមករា ឆ្នាំ២០០៩ ដល់ខែមេសា ឆ្នាំ២០២២ ចំពោះប៉ូពុយឡាស្យុសត្វដែលបង្ខាំងទុកបានបង្ហាញថាអត្រាស្លាប់នៃកូនទើបនឹងកើតជាមធ្យមគឺ26.85% ដែលមានន័យថាប៉ូពុយឡាស្យុងបច្ចុប្បន្នមិនរងគ្រោះខ្លាំងពីការបង្កាត់ជិតទេ។ លទ្ធផលនៃការសិក្សាបង្ហាញថា ហ្វូងសត្វបង្ខាំង និងហ្វូងសត្វពាក់កណ្ដាលព្រៃ (semi-wild) ដែលបានប្រលែងនៅមជ្ឈមណ្ឌលសង្គ្រោះសត្វព្រៃភ្នំតាម៉ៅអាចជាប្រភពប៉ូពុយឡាស្យុងសមស្របសម្រាប់ការបង្កើនចំនួនសត្វរមាំងឡើងវិញនាពេលអនាគត បើទោះបីជាពួកវាអាចនឹងទទួលបានអត្ថប្រយោជន៍ពីការបន្ថែមឯកត្តៈថ្មីដើម្បីការពារការថយចុះនូវនានាភាពនៃសេនេទិចក៏ដោយ។

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Abstract

Captive-breeding is a commonly proposed conservation strategy for species whose populations have become too small to be genetically viable in the wild, such as the Endangered Eld's deer subspecies *Rucervus eldii siamensis*. These programmes require extensive management and maintenance of high genetic diversity within the captive population. The Phnom Tamao Wildlife Rescue Centre and surrounding forests in Takeo, Cambodia, are home to a captive and a released breeding population of Eld's deer that are descended from two founders confiscated from the illegal wildlife trade in 2001. As the captive population grew, it was separated into two herds which remained isolated until small groups were released in 2018. We assessed the level of inbreeding and genetic diversity of the population (27 animals) by re-sequencing genome wide variants using blood samples from eight captive individuals. A low level of genetic diversity was found in the animals. The average heterozygosity rate and nucleotide diversity (π) were $5.538 \times 10^{-6} \pm 1.854 \times 10^{-6}$ and 5.475×10^{-5}, respectively. The mean inbreeding coefficient (F_{ROH}) was 0.026 ± 0.060, and most individuals were in second degree kinship. Demographic records from January 2009 to April 2022 for the captive population revealed average neonatal mortality was 26.85%, which suggests the population does not currently suffer striking hazards from inbreeding. Our results suggest that the captive and released semi-wild herds at the centre may be a suitable source population for future reintroductions, although they would benefit from the addition of new individuals to protect against genetic erosion.

Keywords Captive-breeding, inbreeding, kinship coefficient, reintroduction, whole-genome sequencing.

Introduction

Captive-breeding has become a well-established strategy in conservation plans to safeguard against species extinction (Wilson & Stanley Price, 1994; IUCN/SSC, 2008; Bowkett, 2009; Leus, 2011). The ultimate aim of these programmes is to create genetically diverse and resilient captive source populations for future reintroduction into native habitats (Beck *et al*., 1994; IUCN/SCC, 2013; Ralls & Ballou, 2013). Reintroduction can re-establish species in areas where they have been extirpated and provide new genetic lineages to small wild subpopulations suffering inbreeding depression and Allee effects after prolonged isolation (Deredec & Courchamp, 2007; IUCN/SCC, 2013). Captive-breeding programmes for reintroduction complement *in situ* conservation initiatives, which can be as equally resource intensive (Ralls & Ballou, 1992). These programmes require complicated collaborations amongst various stakeholders, decades of captive management (to mitigate the original causes of wild population decline before release becomes feasible and maintain a captive source population post-release), and prolonged monitoring of the reintroduced population (Wilson & Stanley Price, 1994; Spalton *et al*., 1999; Ralls & Ballou, 2013). Despite the challenges, captive-breeding and reintroduction initiatives have led to re-establishment of extirpated species including golden lion tamarins *Leontopithecus rosalia* (Kierulff *et al*., 2012); California condors *Gymnogyps californianus* (Toone & Wallace, 1994), Arabian oryx *Oryx leucoryx* (Spalton *et al*., 1999); Père David's deer *Elaphurus davidianus* (Jiang *et al*., 1973) and Persian fallow deer *Dama dama mesopotamica* (Bar-David *et al*., 2005).

Maintaining genetic diversity in captive populations has long been a goal of breeding programmes (El Alqamy *et al*., 2012; Ralls & Ballou, 1992, 2013; Chen *et al*., 2019). To mitigate inbreeding depression, loss of genetic variability and minimize adaptation to captivity, these programmes aim to start with as large and diverse a number of founders as realistically possible and limit the number of generations in captivity (McPhee, 2004; Frankham, 2008; Robert, 2009; Purohit *et al*., 2021). To support the reintroduction success of individuals and encourage appropriate behaviours for survival, programmes attempt to provide captive environments that mimic wild habitats (Bremner-Harrison *et al*., 2004; McPhee, 2004). In practice, this can be challenging when captive populations are descended from a small number of founders, when few individuals remain in the wild for capture or when the pressures on wild populations causing declines limit the options for release (Ahmad Zafir *et al*., 2011; Ralls & Ballou, 2013). Inbreeding depression has been commonly associated with lower fecundity and increased neonatal mortality within populations (Ralls *et al*., 1979; Ralls & Ballou, 1986; Lacy *et al*., 1993). However, species-specific studies have found less definitive links between highly inbred populations suffering lower juvenile survivorship (Kalinowski & Hedrick, 2001; Zeng *et al*., 2013). Even if high levels of inbreeding do not affect a population in such a way, it may leave them more vulnerable to successive stochastic events, an important consideration for reintroduction programmes based on a limited number of founders (Thévenon & Couvet, 2002).

Southeast Asia supports a large number of threatened species that have experienced population declines due to habitat loss and illegal hunting (Sodhi *et al*., 2010; Gray *et al*., 2018). Eld's deer *Rucervus eldii* is an Endangered tropical cervid that historically occurred across the region but has suffered population fragmentation and declines throughout its range (Gray *et al*., 2015). Three subspecies are traditionally recognised, *R. e. eldii*, *R. e. thamin* and *R. e. siamensis* (Gray *et al*., 2015), although individuals from Hainan Island (China) are now considered to be a fourth subspecies, *R. e. hainanus* (Wong *et al*., 2021) and are recognized as such in this study. *Rucervus e. siamensis* has been extirpated from Vietnam and Thailand, although a reintroduction programme has begun in the latter (Wong *et al*., 2018). Wild populations remain in Laos and Cambodia and Cambodia is considered to be the last stronghold for the subspecies despite suffering a 90% population reduction in the early 2000's (Gray *et al*., 2015). Less than 400 individuals are currently estimated to remain in fragmented populations across nine protected areas in the country, which are suspected to be declining (Ladd *et al*., 2022).

Genetic studies of the four subspecies have found that *hainanus* and *eldii* have likely suffered bottleneck effects in wild and captive populations (Balakrishnan *et al*., 2003; Pang *et al*., 2003; Angoma & Hussain, 2013; Zheng *et al*., unpubl. data). At the time of testing, Balakrishnan *et al*. (2003) concluded that populations of *thamin* and *siamensis* remained genetically diverse. Since wild populations face a risk of genetic erosion, captive populations may provide fresh genetic lineages that can contribute to genetic recovery (Theodorou & Couvet, 2004; Hedrick & Fredrickson, 2008). Efforts are underway to breed and reintroduce Eld's deer subspecies in their native ranges, including *eldii* in India (Singh & Dookia, 2017), *siamensis* and *thamin* in Thailand (Wong *et al*., 2018) and *hainanus* in China (Wong *et al*., 2021). As suitable natural habitats still remain in Cambodia, similar actions for *siamensis* have been recommended in the country (Gray *et al*., 2015; Ladd *et al*., 2022). At present, few captive populations of *siamensis* are recorded in facilities in Thailand, France and the USA. Additionally, the genetic purity of the subspecies is unknown and could be hybridized with *thamin* (Hartley, M. pers. comm.). As such, one pure population of *siamensis* may remain at present, namely breeding herds in natural enclosures or semi-wild animals in the forests surrounding the Phnom Tamao Wildlife Rescue Centre (PTWRC or centre) in Takeo Province, Cambodia.

As of May 2022, populations of Eld's deer at the PTWRC comprised 15 individuals in captivity and an estimated 40 free-roaming animals that were released from enclosures or born in the forests surrounding the site. The population began with two unrelated individuals which were rescued as fawns from the illegal wildlife trade in 2001. Although the herd was not managed for captive-breeding and reintroduction, it was split into two sub-herds as it grew and these remained separate until they were mixed into small groups for release efforts in 2018. In 2017, blood samples were taken by Kadoorie Farm Botanic Garden to study the taxonomic status of *hainanus* in relation to *siamensis* using low coverage next-generation re-sequencing data. The level of inbreeding within the PTWRC's captive herd was examined by re-sequencing genome wide variants of these individuals. This paper documents the genetic indices of the Phnom Tamao population in terms of inbreeding and genetic diversity. At the time of testing, no *siamensis* had been released, but all deer released into the forest since then are descendants of this herd. Details of captive-care and release protocols are described. Metrics of population health are included to argue the viability of the Phnom Tamao herds as a source population for future reintroduction initiatives in Cambodia or genetic exchanges with other fragmented populations.

Methods

Study site

The PTWRC was established in 1995 by the Cambodian Forestry Administration and has been supported technically and financially by Wildlife Alliance (WA) since 2001. The centre is set within a large area of regenerated deciduous dipterocarp forest which covered 2,025 ha in July 2022. Enclosures are spread across 400+ ha of this area, which was enclosed by a chain link fence in 2016. The centre and surrounding forests are protected by a community anti-poaching unit, which patrols the forest to confiscate hunting equipment, apprehend offenders and remove snares, traps and dogs. The site is a safe location and now holds healthy populations of reintroduced wild boar *Sus scrofa*, red muntjac *Muntiacus muntjak*, sambar deer *Rusa unicolor* and Eld's deer (WA, unpubl. data).

Captive-care

The two captive Eld's deer sub-herds are currently housed in outdoor enclosures measuring 60 m x 60 m. These are located in a public area where animals are on display for visitors. Both enclosures include a pool and an additional roofed section measuring 10 m x 10 m. The first enclosure was built in 2004 after the two original arrivals had two fawns. The second was built in 2009 and joined to the first with a gate, to accommodate the growing herds and create two sub-herds which remained separate until 2018. Originally constructed around a section of forest and filled with natural vegetation that provided good cover for fawns, these areas have since become over-grazed and so grasses no longer regenerate.

A third enclosure was built in a remote area of forest in the PTWRC as a pre-release enclosure in 2018. This is made of two separate sections measuring 60 m x 60 m each, connected by a central holding area which measures 10 m x 10 m. These is not accessible to the public and contain natural vegetation to encourage wild behaviours and disassociation with humans. Groups moved into the pre-release enclosure were formed by mixing deer from the two sub-herds, with one stag and at least two hinds included in each group.

The herds in the display and pre-release enclosures are left to select their own mates and breed at will. The offspring are mother-raised and hand-rearing only occurs when fawns have been neglected and are visibly weakened. All of the herds are breeding, including the smaller groups moved to the pre-release enclosure for acclimatization.

Release

Small groups of deer were first released from the pre-release enclosure and subsequently from the original enclosures within the centre. Deer were released in groups of two to four animals. Single stags were released when there was an imbalance in sexes within the herd to reduce pressure and fighting. When numbers increased sufficiently, the enclosure door was opened and selected deer were allowed to leave at will. Supplemental food was provided twice daily (potatoes and bananas in the morning and water greens in the evening) in the three locations where the semi-wild herds were consistently observed. This was done to encourage the deer to remain in the area and enable *ad hoc* monitoring of their health, movements and births. Not all deer returned for the supplementary feeding, although wilder populations in remoter forest areas of Phnom Tamao have been seen occasionally.

Genetic sample collection

Deer were sedated by veterinary staff at the centre by administering Xylazine (0.25 mg/kg) which was injected intramuscularly with a dart gun, and were revived with Atepamezole (0.250.5mg/kg) which was injected intravenously. To reduce the possibility of capture myopathy and stress on the herds (totalling 27 animals), the easiest adults to catch in the display enclosures that met our requirements were targeted. 100–200 µl of blood was collected from each of these (with five males and five females sampled) and kept in Eppendorf tubes containing 500µl of 95–100% ethanol. The samples were stored on site in a freezer (at -20 °C) prior to transport in a cooler box to laboratory facilities in Phnom Penh.

Whole-genome re-sequencing

Total genomic DNA was extracted using the QIAamp DNA Mini Kit (Qiagen, Germany) following the manufacturer's protocol. DNA quality was quantified with a NanoDrop ND-2000 (Thermo Fisher Scientific, USA). Qualified DNA was sequenced on the Illumina HiSeq Xten platform (Illumina, USA) with PE150 using standard library preparation protocols and an insert size of 350 bp. Raw data from these individuals was processed by removing low-quality bases with Phred-quality scores < 20 and adapter sequences. The sequencing and filtering were performed by default pipelines by the Beijing Genomics Institute company. Clean data was aligned to the draft genome assembly of red deer *Cervus elaphus* (GCA_910594005.1) using the Burrows-Wheeler Aligner (v0.5.17) with default parameters (Li & Durbin, 2009). Sequence depth and coverage were obtained using Bamdst (https://github.com/shiquan/bamdst). After PCR duplicate removal using Picard v1.91 (Broad Institute, 2019), genetic variants as single nucleotide polymorphisms (SNPs) were called using GATK v4.0.2 (McKenna *et al*., 2010). We filtered the low-quality SNPs with sequencing depths lower than three, base-missing rates higher than 10% and minimum quality values lower than 20 using VCFtools v0.1.14 (Danecek *et al*., 2011) and removed the sites on the sex chromosomes. We also removed the SNPs that deviated significantly from the Hardy-Weinberg expectation (p<0.001). As spurious clustering may generate during the process of population structure analysis due to the background linkage disequilibrium (LD), we filtered the LD-based SNPs (--indep-pairwise 50 10 0.2) using Plink v1.91 (Purcell *et al*., 2007) to calculate kinship coefficients.

Genetic diversity analysis

To evaluate levels of genetic diversity, we estimated the genome-wide heterozygosity for all sequenced individuals using filtering SNPs without LD pruning. Genome-wide heterozygosity is calculated as the total number of heterozygotes divided by genome effective length. The number of heterozygotes and the nucleotide diversity (π) were calculated using VCFtools v0.1.14 (Danecek *et al*., 2011) based on a sliding window approach (window size: 50 kb). We also estimated the level of inbreeding through inbreeding coefficients (F_{ROH}). First, the runs of homozygosity (ROH) segments were identified using Plink v1.91 (Purcell *et al*., 2007) with adjusted parameters (--homozyg-window-snp 50 --homozyg-snp 50 --homozyg-window-missing 3 --homozyg-kb 100 --homozyg-density 1000) based on the filtering SNPs without LD pruning (Meyermans *et al*., 2020). Following this, F_{ROH} was computed via the ratio of the total length of ROH to genome effective length in the individual's genome (Mcquillan *et al*., 2008). The genomic effective length of each individual was defined as the reference genome length multiplied by the coverage of genome alignment.

We calculated pairwise kinship values among all re-sequenced individuals using KING v2.1.3 (Manichaikul *et al*., 2010) based on the unlinked SNPs. This method estimates the kinship coefficient by accurately calculating the genetic distance between a pair of individuals as a function of their allele frequencies. Based on Manichaikul *et al*. (2010), a pair of individuals (a dyad) possess a full-sibling or parent-offspring relationship (first degree relative) when their kinship coefficient values range between 0.177 and 0.354 and to a half-sibling relationship (second degree relative) if these values fall between 0.088 and 0.177. A negative kinship coefficient value or a value lower than 0.044 means individuals are not closely related genetically.

Demographic analysis

Manual records were kept at the centre during its early years and tabulated in Excel from 2009 onwards. The records for 2009–2022 analysed in this paper include two sets of monthly stock lists (one of which is organized by enclosure and the other by species) and a report of all arrivals to and departures from the centre each month. The latter includes new arrivals of rescued animals, animals that are born or die and animals that are released from the centre. As hundreds of rescued animal of dozens of species move through the centre each month, records are organized in terms of gross numbers of species and not by individual animals.

In analysis, annual numbers of births, deaths and releases of captive Eld's deer were compiled based on data from the monthly reports on arrivals and departures and checked against both stock lists. All fatalities of captive animals in the same month as a birth were considered to be of fawns unless noted otherwise in the reports. The percentage of neonatal death was calculated each year and averaged between January 2009 and April 2022. The relative birth rate was calculated by dividing the total number of recorded births in a year by the total number of captive animals recorded in the December stock lists of that year, except for 2022 which was calculated from April. As inconsistencies between the stock lists and reports on arrivals and departures suggested births and deaths were under-reported, a minimum count per year was used.

No systematic census has been undertaken as yet of captive-bred Eld's deer released into the forest surrounding the PTWRC and offspring subsequently born. In some cases however, species stocklists included notes on new births of semi-wild fawns within the released groups that returned for supplementary food.

Results

Genetic diversity

Among the ten deer sampled, samples from two individuals (05A & 09A) yielded a low volume of highly degraded DNA and so were not used to construct a whole genome library (Table 1). The eight remaining individuals were successfully sequenced. We obtained an average number of 51,171,950 mapped reads and an average mapping rate of 97.37%. This gave an average sequencing depth of 2.5×. We obtained 48,505 autosomal SNPs and 3,086 unlinked autosomal SNPs after filtering for further analysis.

We obtained an average heterozygosity site of 15,955 ± 5,343 (mean ± SD, the same as below), accounting for a heterozygosity of $5.538 \times 10^{-6} \pm 1.854 \times 10^{-6}$ (Table 1). The mean nucleotide diversity (π) of the sample individuals was 5.475×10^{-5}. Values of F_{ROH} ranged from 0.00013 to 0.177, with a mean F_{ROH} of 0.026 ± 0.060.

Genetic relatedness

From the eight individuals sequenced, we obtained a matrix of pairwise kinship coefficients among 28 dyads with a minimum allele frequency of ≥0.05. The number of SNPs for each pair was 2274. The largest value of kinship was 0.133 and the lowest value was -0.066, and most dyads were second degree relative pairs (Table 2).

Captive management

Five subgroups of Eld's deer were released between 2018 and 2021. As of August 2022, herds observed appear to be adapting well, with no known deaths and at least five births recorded in the forest in December 2020 and December 2021. Wild births and deaths are under-recorded, as observations occur *ad hoc* when the herds

Table 1 Summary of blood samples, DNA concentration, sequencing coverage and depth, homozygotes (HOZ), heterozygotes (HEZ) and heterozygosity rate (HR) of Eld's deer *Rucervus eldii siamensis* studied at the Phnom Tamao Wildlife Rescue Centre, Takeo, Cambodia.

Sample Codes	♀ / ♂	DNA Concentration (ng/μL)	Total Mass (μg)	No. of Mapped Reads	Mapping Rate	Depth	Cover (%)	HOZ	HEZ	HR ($\times10^{-6}$)	Inbreeding Coefficient (F_{ROH})
01A	♀	29.4	0.735	41,409,279	0.9759	2.5291	99.83	28936	17459	6.059	0.00296
02A	♂	19.5	0.4875	37,006,729	0.9709	2.2981	99.82	28855	17540	6.087	0.00323
03A	♂	18.7	0.4675	9,160,690	0.9741	2.5179	99.70	43447	2948	1.023	0.17516
04A	♂	23	0.575	40,677,926	0.9740	2.5266	99.83	26913	19482	6.761	0.00230
05A	♀	0.8	0.0184	/	/	/	/	/	/	/	/
06A	♂	16.2	0.405	40,191,601	0.9724	2.5178	99.82	27915	18480	6.414	0.177
07A	♀	34.8	0.87	40,192,742	0.9727	2.5203	99.79	28300	18095	6.282	0.00782
08A	♀	23.2	0.58	40,717,672	0.9765	2.5354	99.83	28863	17532	6.084	0.00806
09A	♀	0.7	0.0161	/	/	/	/	/	/	/	/
10A	♂	51	1.275	37,677474	0.9731	2.5351	99.77	30289	16106	5.592	0.00013

Table 2 Relative kindship values calculated between pairs of Eld's deer *Rucervus eldii siamensis* sampled at the Phnom Tamao Wildlife Rescue Centre, Takeo, Cambodia.

ID1	ID2	Kinship	ID1	ID2	Kinship
06A	10A	0.0271	03A	06A	0.1653
01A	03A	-0.3521	03A	08A	0.1777
04A	08A	0.1234	01A	07A	0.1148
08A	10A	-0.0281	03A	10A	0.2304
01A	02A	0.0832	03A	04A	-0.3026
01A	08A	0.0985	02A	03A	-0.4122
06A	08A	0.1062	07A	10A	0.0167
04A	07A	0.1273	03A	07A	-0.3662
04A	10A	0.0230	06A	07A	0.1019
07A	08A	0.1058	02A	04A	0.1023
01A	10A	-0.0192	02A	08A	0.0697
01A	06A	0.0910	02A	10A	-0.0375
01A	04A	0.1023	02A	07A	0.0811
04A	06A	0.1332	02A	06A	0.0781

come for supplementary food. The only intervention in the semi-wild herd to date occurred in January 2022, when a juvenile with a broken leg was recaptured for treatment and released back into the forest in April 2022. The average percentage of neonatal death in captivity was 26.85% (SD ± 21.58) (Table 3, Fig. 1). The herd has grown from two unrelated individuals (which arrived at the centre in 2001) to 15 captive animals and 19 released into the forest, giving a minimum of 36 deer (Table 3, Fig. 2). Based on our observations however, we estimate the semi-wild herds have actually grown to between 30 and 40 individuals.

Table 3 Demographics of captive Eld's deer *Rucervus eldii siamensis* from January 2009 to April 2022 at the Phnom Tamao Wildlife Rescue Centre, Takeo, Cambodia.

Year	No. of Births	Relative Birth Rate (%)	No. of Fawn Deaths	Neonatal Death (%)	No. of Deaths	No. of Releases	No. of Animals in Captivity[1]
2009	1	16.7	0	0	0	0	6
2010	2	28.6	0	0	0	0	7
2011	3	33.3	1	33.3	1	0	9
2012	4	36.4	2	50.0	2	0	11
2013	4	33.3	2	50.0	2	0	12
2014	3	17.7	0	0	0	0	17
2015	6	27.3	3	50.0	3	0	22
2016	5	20.0	1	20.0	1	0	25
2017	5	18.5	2	40.0	3	0	27
2018	7	26.9	1	14.3	3	3	26
2019	4	14.8	1	25.0	2	0	27
2020	5	27.8	3	60.0	3	10	18
2021	3	18.8	1	33.3	1	6	16
2022	2	13.3	0	0	0	0	15

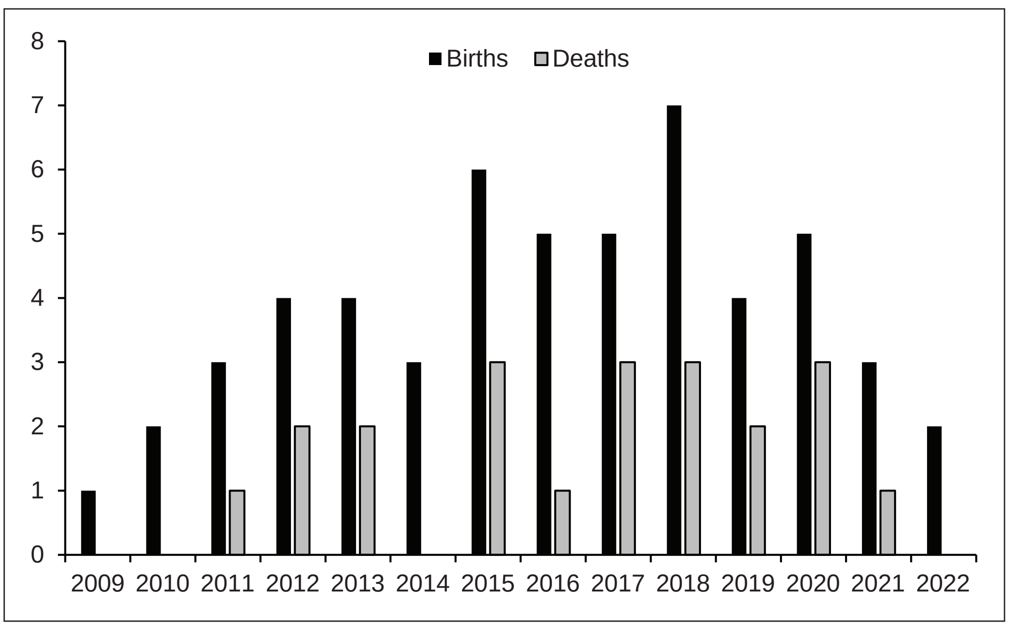

Fig. 1 Births and deaths of captive Eld's deer *Rucervus eldii siamensis* from January 2009 to April 2022 at the Phnom Tamao Wildlife Rescue Centre, Takeo, Cambodia.

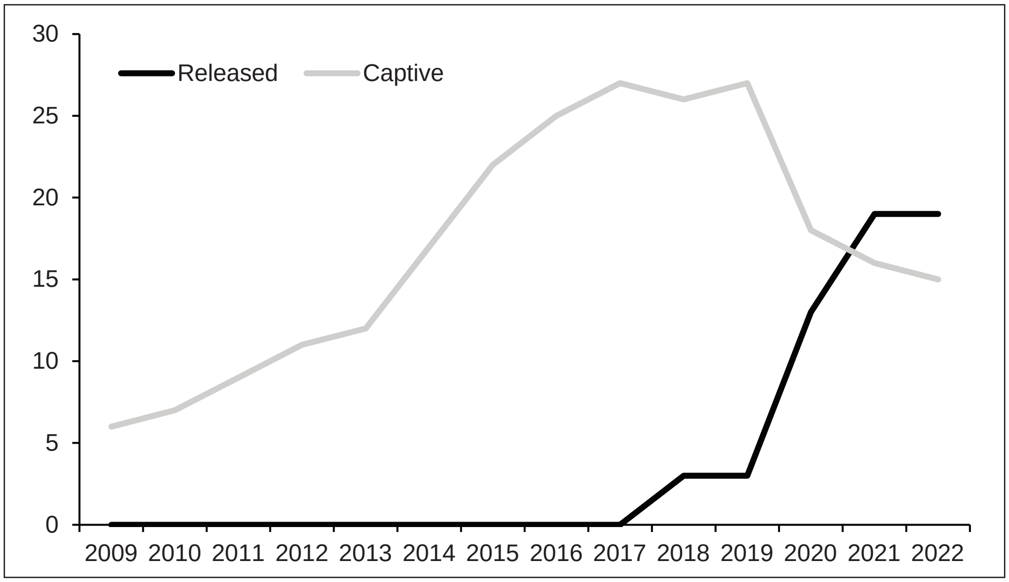

Fig. 2 Total number of Eld's deer *Rucervus eldii siamensis* in captivity and released into the surrounding forest from January 2009 to April 2022 at the Phnom Tamao Wildlife Rescue Centre, Takeo, Cambodia.

Discussion

As of August 2022, the captive herds in the PTWRC and semi-wild herds released into the surrounding forests appear to form a healthy and viable breeding population. These may represent the only increasing subpopulation of Eld's deer in Cambodia (Ladd *et al*., 2022). As the PTWRC population originated from just two founders and most dyads in our analysis were second degree relatives, there is a risk of inbreeding depression. Our genetic analysis of eight individuals showed levels of inbreeding were not particularly high (F_{ROH} 0.026). This result is encouraging, although it may not be representative of the entire population. Sample collection from more individuals within the herds is recommended to increase the accuracy of the results. Future testing of other *siamensis* populations (captive or wild) for comparative purposes would inform future management of the Phnom Tamao population.

An extremely low level of genetic diversity i.e., heterozygosity and nucleotide diversity, was observed during the study. The value of genome-wide heterozygosity (5.538×10^{-6}) was lower than most Endangered species, such as crested ibis *Nipponia nippon* (430×10^{-6}) (Li *et al*., 2014), mountain gorilla *Gorilla beringei* (640×10^{-6}) (Xue *et al*., 2015) and island fox *Urocyon littoralis* ($1.33 \pm 30 \times 10^{-6}$) (Robinson *et al*., 2018). As genetic diversity determines the adaptive potential of a species to environmental change, it plays a key role in their long-term survival (Booy *et al*., 2000; Supple & Shapiro, 2018). While strong deleterious mutations can be removed through genetic purifying to mitigate inbreeding depression in small populations, moderate deleterious mutations can still accumulate during prolonged bottlenecks (Xie *et al*., 2022), reducing population viability. As such, estimation of deleterious mutations using high-quality sequencing data should be undertaken to illuminate the effects of inbreeding and subsequent extinction risk to the study population.

High neonatal deaths and decreased fecundity in adults of various species, including Eld's deer, have long been attributed to inbreeding depression (Prescott, 1897; Ralls *et al*., 1979; Thévenon & Couvet, 2002). The continued breeding of the free-roaming and captive deer with relatively high infant survivorship suggests the PWTRC population does not suffer severe negative effects of inbreeding. Since 2009, the average percentage of neonatal deaths among the captive herds has been 26.85%, although this figure must be taken as a minimum count as records are incomplete. The survivorship of infants in the semi-wild herds is unknown although the herds continue to grow. Although the forests surrounding the centre contain predators such as jackals *Canis aureus*, the semi-wild deer herds are not subject to the same level of predation as wild herds, so infant survivorship could be higher compared to these (Linnell *et al*., 1995; Jarnemo, 2004). However, Dion *et al*. (2020) found similar rates of neonatal mortality in populations of white-tailed deer *Odocoileus virginianus* in areas with and without natural predators. Nevertheless, levels of infant mortality in the captive population at Phnom Tamao are still substantially smaller than the mortality rate of up to 90% recorded for captive Eld's deer in France, where only 21% of offspring made it to 18 months (Prescott, 1987). High neonatal mortality and reduced fecundity in captive Eld's in Thailand has also long been a concern, although the exact rates are not available for comparison (Siriaroonrat, 2003). The survival of fawns and relative stability of the birth rates at PTWRC over the 12 years analysed could be partly due to allowing the deer to select their mates, which is recommended as a way of increasing reproductive output (Asa *et al*., 2011; Martin-Wintle *et al*., 2018). Although free mate selection can increase offspring survival, there is a risk of decreased genetic diversity as an uneven distribution of individuals will breed (Haig *et al*., 1990; Gooley *et al*., 2018). However, several studies of captive and reintroduced populations of ungulates with inbreeding coefficients above 0.2 have found low genetic variability did not affect herd demographics, breeding rates or neonate mortality (Kalinowski & Hedrick, 2001; Sternicki *et al*., 2003; Zeng *et al*., 2013; Moreno *et al*., 2020). As such, the benefits of allowing free mate choice among animals kept in their natural social configurations may outweigh the risks of reduced genetic diversity.

To safeguard against genetic erosion within the captive herd, it would be beneficial to introduce new and unrelated individuals of pure *siamensis* from wild populations that continue to decline. While it would be preferable to protect habitats of threatened species and ensure connectivity between fragmented populations, should the wild population decline dramatically, it may not be able to recover naturally without intervention (Phumanee *et al*., 2020). As with all subspecies of Eld's deer, the population of *siamensis* is fragmented in the wild and therefore risks losing genetic variability, leaving it more susceptible to environmental stochasticity (Song, 1996; Thévenon & Couvet, 2002; Angom & Hussain, 2013). Should a species breed faster in captivity, Tenhumberg *et al*. (2004) recommended capturing an entire wild population containing less than 20 females. However, it would be more realistic to exchange a few stags between the fragmented wild and captive populations, provided adequate protection can be ensured for the former. Depending on the relative genetics of these, this could support genetic rescue in both (Theodorou & Couvet, 2004). This has been undertaken successfully with captive Mexican wolves *Canis lupus baileyi*, whereby one population descended from three founders was mixed with unrelated individuals from two separate lineages each originating from two founders, increasing genetic diversity among the three populations (Hedrick & Fredrickson, 2008). The addition of new lineages to the captive populations of Eld's Deer at the PTWRC should begin as soon as possible to safeguard their genetic diversity until such time as reintroduction becomes a responsible option. Many captive-breeding initiatives begin too late to acquire sufficient founders to ensure a genetically diverse and sustainable population, thus risking the success of the overall programme (Ahmad Zafir *et al*., 2011; Ralls & Ballou, 2013).

Captive-breeding programmes must consider that captive environments can select for adaptations that are inappropriate for survival in the wild (Bremner-Harrison *et al*., 2004; McPhee, 2004; Frankham, 2005). Continued release of Eld's deer in the forests of Phnom Tamao may mitigate the effect of captivity on the genome, allowing herds to live and breed within a safe setting. Exposure to a wild environment has been shown to increase the survival of offspring of released animals (Evans *et al*., 2014). This has also been achieved through enclosures that mimic wild habitats (Beck *et al*., 1991; Frankham, 2008). The Eld's deer enclosures at the centre are located in the on-display section and have lost much of their vegetation due to over-grazing. However, fawns are still born despite the lack of undergrowth, and once deer were moved into the remote forest enclosure, they became wary of humans (Marx, N. pers. obs.). Some of the deer released from the centre display tolerance for people, but continue to forage naturally. The survival of released individuals and their offspring could be partly due to the minimal occurrence of natural predators and hunting in the forest. Should release become an option in future, only animals expressing predator avoidance and sufficient fear of humans should be selected. Further, candidates for release should be monitored to ensure they exhibit appropriate behaviours in acclimatization enclosures prior to release (IUCN/SCC, 2013).

There are many complimentary actions that can be taken to conserve wild populations of *R. e. siamensis*. The herds at PWTRC should not be overlooked as a potential source of animals that could be used in future to re-establish the subspecies within its historical range. Successful programmes exist that originated from a few founders, such as one for black-footed ferrets *Mustela nigripes* that was based on ten individuals, though only five were successfully breeding (Ralls & Ballou, 2013). Further, Moreno *et al*. (2020) found that rates of births and infant survival were similar between reintroduced and captive groups of Cuvier's gazelles *Gazella cuvieri* with high inbreeding coefficients, despite the released animals experiencing greater stress in the wild. As such, we believe it would be incorrect to dismiss the Eld's deer herds at the PWTRC as a source for future reintroduction efforts due to the small number of animals the population descended from.

The forests surrounding Phnom Tamao are an undervalued resource for conserving certain species. These include sambars, which were also bred in captivity and released at the centre. Although these are currently listed as Vulnerable, captive populations will be increasingly important to survival of the species as the wild populations continue to decline throughout their range (Gray *et al*., 2012; Timmins *et al*., 2015). The forests of Phnom Tamao are now a breeding ground for this and other species, some of which could be translocated to other forests once these sites are known to be safe. If properly managed following IUCN guidelines, with animals translocated to carefully selected sites, sambar or Eld's deer from PTWRC may be used to repopulate other forests.

Given that forests and wildlife in Cambodia are declining throughout the country, the PTWRC and other captive populations could ensure the survival of some species. Successful captive-breeding and monitored release efforts for other species, such as the reintroduction of captive-born pileated gibbons *Hylobates pileatus* to the Angkor Archaeological Park (Leroux *et al*., 2019) and release of binturongs *Arctictis binturong* into Tatai Wildlife Sanctuary (Marx & Roth, 2014), demonstrate that this approach works. Despite a small sample size, our study suggests that herds of Eld's deer at Phnom Tamao have been managed in a way that will allow similar releases in future provided appropriate areas of safe habitat can be found. Provided the centre is professionally managed and the surrounding forests are protected, Phnom Tamao can contribute greatly to conservation in Cambodia by rehabilitating, captive-breeding and releasing rescued animals to bolster dwindling populations in the wild.

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Selective cutting of large-diameter trees in a lowland evergreen forest in central Cambodia

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មូលន័យសង្ខេប

រូបភាពថតលើផ្ទៃដីលំអិតពីរបៀប និងពេលវេលាដែលព្រៃឈើត្រូវបានសឹករិចរិលអាចផ្តល់ព័ត៌មានជាមូលដ្ឋានគ្រឹះសម្រាប់ការស្តារព្រៃឈើឡើងវិញនាពេលអនាគត។ ការកាប់ដើមឈើបែបជម្រើសដែលមានអង្កត់ផ្ចិតធំគឺជាដំណាក់កាលចាប់ផ្តើមនៃការរិចរិលព្រៃមានដើមឈើធំ។ យើងបានកត់ត្រាការកាប់ព្រៃឈើក្នុងតំបន់ទំនាបព្រៃស្រោងស្ងួតក្នុងខេត្តកំពង់ធំ ភាគកណ្តាលនៃប្រទេសកម្ពុជា។ ការចុះប្រមូលទិន្នន័យនៅទីវាលត្រូវបានធ្វើឡើងដើម្បីកត់ត្រាប្រភេទ និងទំហំដើមឈើដែលបានកាប់នៅកន្លែងបានកំណត់ដោយរូបភាពពីផ្កាយរណប។ ការប្រៀបធៀបរូបភាពផ្កាយរណប ALOS/PRISM ពីរដែលមានកម្រិតភាពច្បាស់ 2.5m ថតនៅខែវិច្ឆិកាឆ្នាំ២០០៦ និងខែមីនា ឆ្នាំ២០០៨ បានបង្ហាញអោយឃើញថា ៥០១ ទីតាំងមានការបាត់បង់គំរបព្រៃយ៉ាងធំក្នុងទំហំផ្ទៃដីសិក្សា ១៤.៩៣ គម²។ ការសិក្សារបស់យើងបានបង្ហាញថាទីតាំងចំនួន ១០១ ក្នុងចំណោម ១០៣ទីតាំងដែលបានត្រួតពិនិត្យឃើញថាមានការបាត់បង់ដើមឈើ ដោយមានភស្តុតាងនៃស្នាមកាប់ចំនួន៨៨ ទីតាំងក្នុងចំណោម ១០១ទីតាំង។ ដង់ស៊ីតេនៃការកាប់ឈើដែលបានប៉ាន់ប្រមាណពីឆ្នាំ២០០៦ ដល់ឆ្នាំ២០០៨ (ឧទាហរណ៍ ចំនួនពីររដូវប្រាំង) មានយ៉ាងតិច ២៨.៧ ដើម/គម²។ ដើមឈើអំបូរឈើទាល (dipterocarps) ចំនួនពីរ ប្រភេទដែលបានកាប់គឺ *Dipterocarpus costatus* (ឈើទាលបង្គួយ) និង *Anisoptera costata* (ផ្ដៀក) មានអង្កត់ផ្ចិតជាមធ្យម ៨៤ សង់ទីម៉ែត្រ និងកម្ពស់ ១៣០ សង់ទីម៉ែត្រ។ ការកាប់បំផ្លាញធ្វើឡើងដោយក្រុមជាច្រើនដែលមានចម្ងាយ ១.៣ គីឡូម៉ែត្រពីផ្លូវធំ។ ព័ត៌មានដង់ស៊ីតេនៃការកាប់ និងទំហំដើមឈើដែលបានកាប់រួមចំណែកក្នុងការគណនាពីការសឹករិចរិលព្រៃឈើនៅប្រទេសកម្ពុជាកន្លងមក។

Abstract

Detailed ground-based records of how and when forests have been degraded can provide fundamental information for future restoration efforts. Selective cutting of large-diameter trees represents the initial stage of high-storage forest degradation. We recorded such cutting in a lowland, dry evergreen forest in Kampong Thom Province in central Cambodia. Field surveys were performed to record the species and sizes of cut trees in locations identified with satellite imagery. Comparison of two ALOS/PRISM satellite images with a 2.5m resolution taken in November 2006 and March 2008 revealed 501 sites with large canopy loss within a 14.93 km^2 study area. Our field surveys revealed that 101 of 103

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sites checked had experienced tree mortality, with mortality due to cutting evident at 88 of the 101 sites. The estimated cutting density from 2006 to 2008 (i.e., over two dry seasons) was at least 28.7 trees/km^2. The two species of dipterocarps cut, *Dipterocarpus costatus* (*chhoeuteal bankouy*) and *Anisoptera costata* (*phdiek*), had a median diameter of 84 cm at a height of 130 cm. The cutting was carried out by several groups within 1.3 km of the main road. This information on the cutting density and the size of the cut trees contributes to quantifying past forest degradation in Cambodia.

Keywords *Anisoptera costata*, conservation, crown gap, *Dipterocarpus costatus*, forest degradation, selective cutting.

Introduction

Deforestation and forest degradation in Cambodia are being addressed, but remain challenging. The causes of historical deforestation and forest degradation under different political regimes in Cambodia were summarized by Kim *et al*. (2005) and Tsujino *et al*. (2019). Until 2002, when a moratorium on logging was declared, the commercial forestry concession system introduced in 1993 formed the mainstay of forest management in Cambodia (Ty *et al*., 2009; Sasaki *et al*., 2013). Under the 2002 Forestry Law, commercial logging has been largely prohibited, except when plantations (for example, of rubber trees) are approved (Ty *et al*., 2011). Broadhead & Izquierdo (2010) observed that the logging moratorium resulted in the closure of mills and reductions in illegal logging, but also shifted the focus of illegal logging from commercial to small-scale operators, from few players to many, and from export to domestic markets. Deforestation between 2002 and 2010 (or 2016) was related to rapid population and economic growth nationally (Michinaka *et al*., 2013; Tsujino *et al*., 2019). In 2023, forest crimes continue, despite continued crackdowns by the authorities (Khmer Times, 2023).

Reliable forest statistics are essential when drafting guidelines for sustainable forest use. Detailed documentation of when and how forests have been subjected to anthropogenic disturbances, based on field surveys, is also important for improving forest management and planning forest restoration. Forest degradation due to illegal selective logging (hereafter cutting) has been recognized, but at least in the government report submitted in 2009, no information was available to assess forest degradation in Cambodia on a nationwide basis (Ty *et al*., 2009). Unsustainable fuel wood collection, one of the main drivers of forest degradation, has been well studied and its impacts quantified (Top *et al*., 2004a, 2004b, 2004c; San *et al*., 2012; Ito & Tith, 2020). However, there is somewhat less information on forest degradation caused by cutting of large-diameter trees for timber. A recent analysis of the impact of selective logging/cutting on forest ecosystems found that studies were biased geographically, with in Southeast Asia, including Cambodia, being poorly studied (Hari Poudyal *et al*., 2018). Langner *et al*. (2018) used differential indicators from satellite images to detect canopy disturbance due to selective logging in evergreen forests across Cambodia. In addition to using very high-resolution imagery, they conducted a field survey in Kampong Thom Province to confirm the accuracy of satellite detection. However, finer details regarding forest canopy disturbance due to selective logging have yet to be reported.

The purpose of our study was to document the early stages of forest degradation, starting from selective cutting in relatively well-preserved high-storage evergreen forests. For this purpose, we conducted a field survey to record the details of selective cutting of large-diameter trees that occurred under a logging moratorium in Kampong Thom Province, central Cambodia. The aspects of the cutting activity and site disturbance were documented by surveying areas of significant canopy loss in evergreen forests detected by comparing satellite images from two time periods.

Methods

Study site

Our study site was located in Sandan District, Kampong Thom Province (12°43'–12°48' N, 105°27'–105°30' E, thus about 8.75 by 6.25 km in area; Fig. 1). The seasonal tropical climate is governed by monsoons and November through April are dry. Mean annual temperature is 27°C and mean annual rainfall (± SD) is 1,542 (± 248) mm (2000–2010; NIS, 2012).

The study site was previously a concession forest. A license for the concession was issued in July 1995 to Grand Atlantic Timber International Co. Ltd., which was subsequently cancelled in June 2002 (World Bank, 2005a, 2005b; Kurashima *et al*., 2013). During our study period (2006–2009), the site existed as a cancelled concession forest. A circular area of land with a radius of about 10 km, including the study site, was designated as a protection forest. The date of its designation is unknown, but

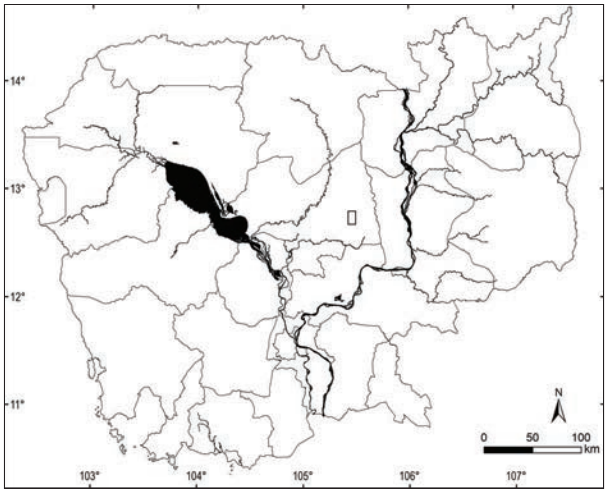

Fig. 1 Location of the study site (rectangle) in Cambodia.

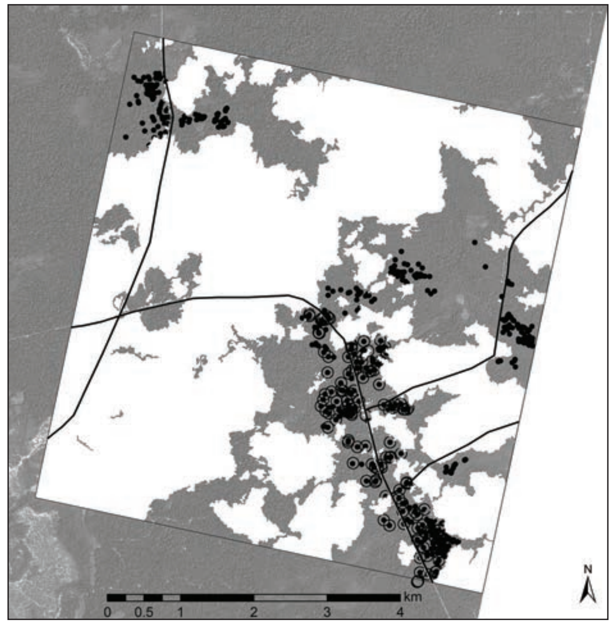

Fig. 2 The distributions of image-detected crown gaps and field-surveyed cutting sites. The closed black circles and the empty circles indicate crown gap sites detected on images and sites of tree mortality found during the field survey, respectively. Within the almost-square area of interest, the black lines indicate major roadways whereas the white areas indicate cloud-affected image areas that were not examined. The ALOS/PRISM images used for visual identification are shown in the background.

the site was included on a 2008 map of forest land uses in Cambodia (Sasaki *et al*., 2013).

Forests at the study site are classified as lowland dry evergreen forest. Evergreen forests in Cambodia are categorized into four subtypes based on their elevation (lowland or sub-montane) and moist or dry climate, using the 2007 classification system of the Ministry of Environment (cited in Brun, 2013). As mentioned, this subdivision is based on elevation, with 650 m used as the boundary between lowland and sub-montane types, as well as other bioclimatic criteria that differentiate the humid coastal ranges (moist, annual precipitation ca. >2000 mm), lower-humidity inland forests (dry) and hinterlands (Brun, 2013). Lowland dry evergreen forest, one of the four evergreen forest subtypes, is included in dry evergreen forest in the classification system of the Cambodian Forestry Administration (FA, 2011). The distribution of evergreen forests in Cambodia and the environmental conditions associated with their classification are described by Ito *et al*. (2021).

Members of the Dipterocarpaceae typically dominate in lowland dry evergreen forest (Rundel, 1999; Tani *et al*., 2007; Ito *et al*., 2021). Two tall dipterocarp species, *Dipterocarpus costatus* C. F. Gaertn. (*chhoeuteal bankouy* in Khmer) and *Anisoptera costata* Korth. (*phdiek*), dominate the upper canopy layer of forests at the study site (Pooma, 2002). The study site features a sandy soil, Haplic Acrisol (Alumic, Profondic) (Toriyama *et al*., 2007, 2008), characteristic of gently undulating, sandy alluvial plains.

Detection of potential cutting sites by remote sensing

We obtained advanced land observing satellite panchromatic remote-sensing instrument stereo mapping (ALOS/PRISM) images (with a 2.5 m spatial resolution and a revisit cycle of 46 days) taken on 27 November 2006 and 1 May 2008, thus 1.25 years apart. The 2006 images include many clouds, whereas the 2008 images are clear. The total area explored was 14.93 km2, excluding the cloud-affected areas in the first image set. We employed object-based classification (Walter, 2004) to identify cutting of emergent trees with large crowns (>22.5 m in diameter, thus occupying 9 x 9 pixels on the ALOS/PRISM images). Selective cutting of such large trees can be identified via crown disappearance and the presence of large crown gaps. In the pre-felling images, the crowns of large-diameter trees fill the upper layer of the forest and the images are bright. After cutting, the canopy disappears and the images are dark. We performed principal component analysis (PCA) of both of the image sets to enhance the altered areas (Awaya & Tanaka, 2003) and identify the disappearance of large tree crowns. Large negative values of the second component indicated new crown gaps. In total, 501 potential cutting sites were visually identified (Fig. 2) and a subset of these was selected for field surveys.

Ground truthing survey

Our field survey was undertaken from 31 October 2008 to 4 November 2008, although preliminary surveys were also conducted on 31 August 2008 and 5 March and 30 November 2009. We visited potential cutting sites detected in ALOS/PRISM images. We recorded the location, size, tree-stump species, cause of death (cutting, natural mortality or collision with a felled tree) and the fall direction of felled trunks. Stump diameter was measured at breast height (1.3 m) (DBH) and for stumps <1.3 m in height, DBH was estimated using regression equations based on the maximum height and diameter and the diameter at ground level of the stump (Ito *et al*., 2010). Stump height was measured only when stumps were <1.3 m in height. For taller stumps, the height classes were estimated using photographs taken at 10 cm increments. The total area covered in the field was estimated from tracks recorded in a GPS 60CSx (Garmin, USA).

We also visually checked for cut trees or dead trees not detected on ALOS/PRISM images within 20 m on either side of paths taken between points of interest and investigations similar to those described above were undertaken when stumps were found. We estimated when such trees had been cut or had died, thus before or after the initial satellite imagery (before November 2006 or after December 2006 respectively) based on the freshness of a cut surface, the extent of resin exudation and the amount of bark on the sides of a stump. The conditions of stumps detected on ALOS/PRISM images were used as references for the decision criteria. The date of creation of industrial refuse left on stumps also served as a reference. Cutting largely occurred during the dry season (November to March), which was the agricultural off-season at that time. Therefore, stump ages were in one-year increments and thus (in our view) easy to determine. However, we concede that such judgments are subjective and therefore somewhat uncertain. Further, for trees cut during the 2007–2008 dry season, it was near-impossible to determine whether cutting occurred between the initial and final imagery (between December 2006 and February 2008) or after the final imagery (post-March 2008). All statistical analyses employed JMP software v10.0 (SAS Institute Inc., USA).

Results

Observations at sites of crown gaps

Our field surveys covered 0.34 km^2 and we visited 103 sites of crown gaps observed on images and confirmed that 101 were of these genuine gaps; trees had died. The trees were healthy at the remaining two sites. Of the 101 dead trees, 87.1% (n=88) had been cut, 9.9% (n=10) had died of natural causes and 3.0% (n=3) had died as a result of collision with a felled tree. In one location where a tree had died of natural causes, a large tree had not died, but a clump of smaller trees had. Thus, the percentage of large-diameter trees cut relative to the number of crown gaps on images was 88/103 (85.4%). We therefore estimate that 428 of the 501 canopy gaps detected in images of the 14.93 km^2 study area reflected cutting, with a cutting density per unit area from 2006 to 2008 (during two dry seasons) of at least 28.7 trees/km^2. Since our satellite imagery did not detect all crown gaps, this will naturally be an underestimate.

Spatial distribution of putative cutting sites

It was evident that the crown gaps (or cutting sites) were intense in small areas near main roads (Fig. 2) and many were within 50 m of a main road (Fig. 3a). As noted above, 85.4% of the crown gaps identified in satellite imagery were estimated to be sites where large-diameter trees were cut. The number of these sites decreased as the distance from a main road increased and cutting generally ceased at 1.2–1.3 km (Fig. 3a). The detection density (as in the number of detections divided by the area analysed: Fig. 3b) reduced less with distance from roads than the number of detections (Fig. 3c). The detection density within 50 m of a main road was 100 sites/km^2, whereas the average for other classes (>50 m & ≤1.3 km) was 30.8 sites/km^2. This detection density was multiplied by the percentage of large-diameter trees cut relative to the number of crown gaps on images (85.4%) to give an estimated cutting density of 85.4 trees/km^2 within 50 m of a main road and 26.3 trees/km^2 for >50 m and ≤1.3 km from a main road.

Species and sizes of trees cut

We recorded 222 stumps of cut/naturally fallen trees during the field survey, which included the cloud-covered area of the satellite images. Of these, 191 were cut trees, five had been killed by adjacent cutting and 26 had died naturally. We identified 190 of the cut trees to species in the field and found 113 (59.5%) were *D. costatus* and 56 (29.5%) were *A. costata*.

The DBH frequencies of the dipterocarps cut are shown in Fig. 4. Their medians and interquartile ranges were 84.0 cm and 75.7–97.7 cm. Mean ± SD DBH (range) were 86.4 ± 16.2 (56.0–131.7) cm and 90.1 ± 17.2 (55.6–150.6) cm for *D. costatus* and *A. costata*, respectively. Both species were cut before November 2006 (white histograms: Fig. 4a, 4b) or after December 2006 (black histograms: Fig. 4a, 4b). The size frequency distribution

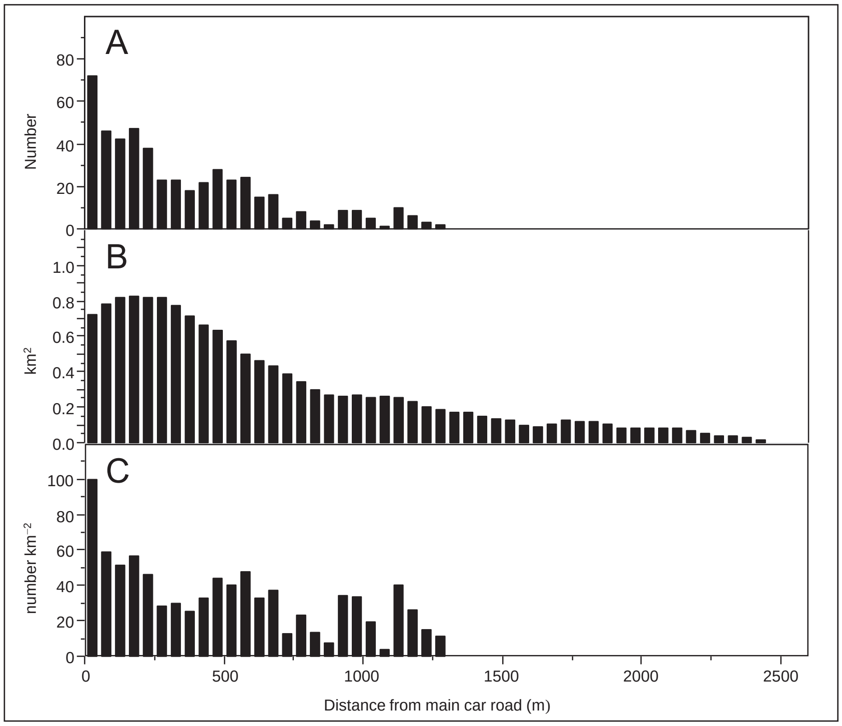

Fig. 3 Frequency distributions by linear distance from a main road. A) Number of crown gap sites evident on images, B) Analysis area, omitting areas hidden by clouds, C) Detection density.

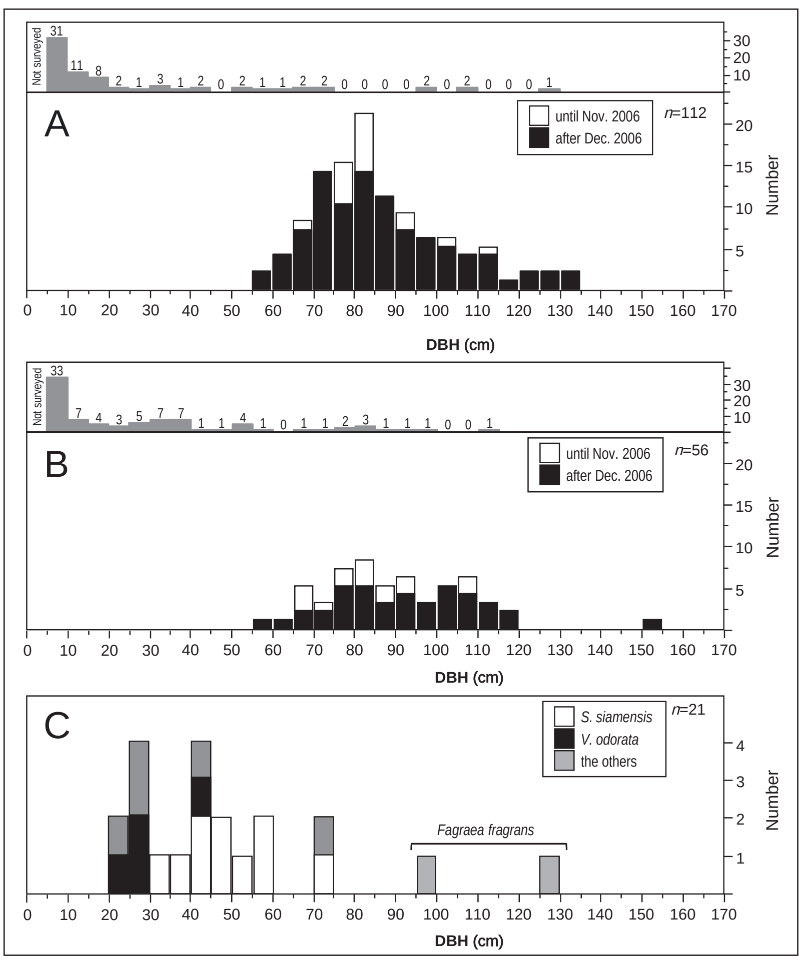

Fig. 4 Frequency distributions of diameter at breast height (DBH) for trees cut: A) *Dipterocarpus costatus*, B) *Anisoptera costata*, C) Other species. In A and B, the white and black columns are data obtained before and after the initial imaging (before November 2006 and after December 2006) respectively. The gray bars at the top of the figure show the frequency distribution of DBH for each species population from a tree census in the surrounding area, with numbers above the columns indicating the number of confirmed individuals (Ito *et al.*, 2023). In C, the white and black columns refer to *Sindora siamensis* and *Vatica odorata*, respectively. A DBH datum was missing for one stump of *D. costatus*.

of both tree species obtained from tree censuses of the surrounding area is also shown elsewhere (Fig. 4a, 4b: Ito *et al*., 2023).

Other species cut (*n*=21) included *Sindora siamensis* Teysm. ex Miq. (Fabaceae, *ko koh*, *n*=10), *Vatica odorata* (Griff.) Symington (Dipterocarpaceae, *chromas*, *n*=4), *Fagraea fragrans* Roxb. (Loganiaceae, *tatrao*, *n*=2), among others. Such species commonly had smaller DBHs, apart from *F. fragrans* (Fig. 4c). Most of the *S. siamensis* stumps were old (before November 2006), although a few were more recent.

Naturally dead trees

As noted above, 26 trees that had died naturally were found during the ground survey, including three *D. costatus* and eight *A. costata*. Mean DBH ± SDs (ranges) of trees that died naturally were 118.3 ± 14.6 (77.0–163.0) cm and 86.3 ± 8.9 (54.1–133.8) cm for *D. costatus* and *A. costata* respectively. Nominal logistic analysis was used to predict whether mortality was anthropogenic or natural using tree species as the predictor variable. This revealed significantly higher natural mortality of *A. costata* ($\chi2$=6.94, *p*=0.014, *df*=1, *n*=180) than other species (odds ratio 5.53, *p*=0.0084).

The other 15 trees that died naturally included *Lophopetalum duperreanum* Pierre (Celastraceae, *proloup*, *n*=3), *Vatica odorata* (Dipterocarpaceae, *chromas*, *n*=2), *Tristaniopsis merguensis* (Griff.) Paul G.Wilson & J.T.Waterh. (Myrtaceae, *mdenh meas*, *n*=2), *Madhuca* sp. (Sapotaceae, *srokum*, *n*=2), Anacardiaceae (*n*=1), and unknown species (*n*=5). The mean DBH ± SD (range) of the trees that had died naturally was 51.9 ± 18.8 (15.0–76.0) cm (*n*=14, with 1 missing datum).

Cutting and processing operations and ground disturbance in the site

Chainsaws were used for cutting. Most stumps were cut between 60–110 cm in height, with a value of approximately 80 cm most frequent for both dipterocarp species (Fig. 5). A significant number of trees were cut higher. Of trees cut higher than 120 cm, 42% were *D. costatus* and 16% *A. costata*. The former were often cut higher to avoid lower deep scars on burns made to collect resin (Fig. 6), although there were many cases where felled trees had burn scars. Neighbouring stumps were often of similar height (data not shown). Some trees were left to rot. We found at least seven such trees. Trees found to be hollow may be abandoned (Fig. 7).

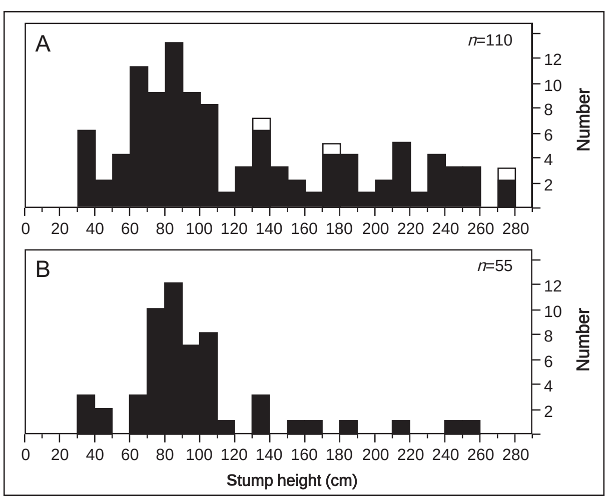

Fig. 5 Frequency distributions of the stump heights of A) *Dipterocarpus costatus*, B) *Anisoptera costata*. The white columns in A indicate that the figures are underestimates.

Fig. 6 Stumps of *Dipterocarpus costatus* with burn scars created to collect resin. A) Lower cut position, B) Higher cut position.

Felled trees were sawn on site. Sometimes, the logs were removed after the bark had been removed with an axe (Fig. 8a, 8b) and sometimes, the timber was processed into planks (Fig. 8c). Chain saw chips were observed near stumps when the cuts were very fresh (Fig. 8d). Fine sawdust disappeared rapidly, whereas bark and wood remained (Fig. 8e). One to two years after cutting, only wood blocks and bark remained on the forest floor (Fig. 8f).

No road was sufficiently wide to admit large vehicles. Former sawmill sites were often converted to cutting roads (Fig. 9a), and surrounding juvenile trees cut, especially if the road was extensively used (Fig. 9b).

Fig. 7 A felled tree with hollows left to rot at the study site.

Discussion

Forest degradation due to loss of large-diameter trees from 2006 to 2008

Our study provides key information on forest degradation due to selective cutting in the study area from 2006 to 2008. The most fundamental figure in this study is the DBH frequencies for cut dipterocarps (Fig. 4). To quantify forest degradation in terms of its carbon emissions, it is necessary to estimate the amount of biomass loss over time in a given area, for which a combination of field inventories and remote sensing is necessary (Bustamante *et al*., 2016; Gao *et al*., 2020). Our ground-based information, combined with satellite imagery, contributes to quantifying forest degradation more reliably and suggests that a cutting density of 28.7 trees/km^2 occurred over two dry seasons in our study area.

Information on the sizes of trees cut also indicates the progression of forest degradation. Comparisons of the DBH frequencies of cut dipterocarps and natural population suggests that the loggers cut trees from the largest diameter class (Fig. 4a, 4b). There was no difference in tree size in terms of the time of cutting (before November 2006 or after December 2006). However, cut stumps with a DBH <65 cm were only found after December 2006, implying that large-diameter trees had been depleted and smaller individuals subsequently targeted.

Shifts in the species targeted for cutting may reflect the progression of forest degradation. We found that *D. costatus* and *A. costata* were primary targets for cutting (Fig. 4). Both species are straight dipterocarps which attain 30–35 m in height (Toyama *et al*., 2013) and are thus priority targets for cutting (Kim Phat *et al*., 2002). This situation persisted from at least November 2006 to November 2009 i.e., from the time the first image was acquired to the time the last ground survey was conducted. This indicates that forest degradation in the study area was at a stage where the most useful large-diameter tree species were selectively cut, which likely corresponds to an initial stage of forest degradation. Aside from the two dipterocarp species, stumps of *S. siamensis* were most commonly found. Most of these were old (before November 2006), but some stumps were more recent, implying that the species might have been targeted anew. Although the number of observations is small (n=10), these records likely reflect changes in the species selected as the primary targets were depleted. Excessive selective logging often results in different vegetation, which is a typical sign of forest degradation (Thompson *et al*., 2013).

The amount of wastage in forest harvesting processes and its impact on carbon sequestration potential has been little addressed in studies on the influence of selective logging (Hari Poudyal *et al*., 2018). We quantified stump height and DBH for trees cut at our study site. Stump heights varied considerably (Fig. 5), possibly due to cutting practices and tree species (see below). Tree stumps are the main component of deadwood mass, one of the four carbon pools (above-ground and below-ground biomass, deadwood and litter) (Kiyono *et al*., 2010, 2011, 2017, 2018). These data provide baseline information for considering greenhouse gas emissions under REDD+ (Reducing Emissions from Deforestation and forest Degradation) projects.

Identifying crown gaps is a standard method for detecting forest degradation (Mitchell *et al*., 2017). The resolution of our ALOS/PRISM images (2.5 m) was sufficient to detect emergent tree crown gaps ca. 30 m in diameter. As such, archived ALOS/PRISM images may be effective to detect the early stages of former forest

Fig. 8 Processing of harvested trees and forest-floor remnants. A) Bark stripped off by an axe on the forest floor (1 November 2008), B) Logs on main road (31 August 2008), C) Lumber milled into boards (5 March 2009), D) Fine sawdust after chainsaw use at a recent cutting site (26 June 2018), E) Over time, fine sawdust disappears and bark becomes more prominent (5 March 2009), F) Wood blocks and bark at one to two years after cutting (1 November 2008).

degradation due to selective cutting in Cambodian evergreen forests. While earlier satellite images have certain quality limitations, these data are nonetheless valuable. However, this is the maximum density of large-diameter tree loss that could be estimated. To apply the coefficient for converting crown gap density to cutting density we obtained (85.4%) to other areas, the assumption that natural mortality of large-diameter trees is comparable to the study site would have to be met. Integrated assessment of forest degradation across an entire region requires regionally robust and consistent approaches, and locally developed approaches that assess degradation levels in small target areas on a case-by-case basis are considered inadequate (Miettinen *et al*., 2014). As information on forest degradation due to selective logging is scarce for Cambodia (Hari Poudyal *et al*., 2018) however, it is also necessary to continue to build on findings obtained from locally developed approaches.

Cutting operations under the logging moratorium

Under the logging moratorium, illegal logging (cutting) operations shifted from commercial to small-scale operators, from few players to many, and from export to domestic markets (Broadhead & Izquierdo, 2010). Our observations in this study support this view. The most frequent cutting height was approximately 80 cm, implying that the operator was standing on the ground when using the chainsaw. However, a significant number of trees were cut higher (Fig. 5) and sometimes over 2 m. Neighbouring stumps were often of similar height (data not shown), implying that different cutting teams favoured particular heights. *Dipterocarpus costatus* was often cut higher to avoid lower deep scars on burns made to collect resin (Fig. 6), although there were many cases where trees felled had burn scars. This variation can be explained by the practices of individual cutting groups (personal communications from local residents). This suggests that cuttings in this area during our study period may have been carried out by several groups with different cutting practices, although the reasons underlying the choice of felling height remain unclear.

The development of roads is a key factor in promoting forest degradation (Walker *et al*., 2013; Lapola *et al*., 2023). Tree cutting, particularly of tall trees, is often intense near main roads. We found intensive cutting of large-diameter trees in small areas near main roads (Fig. 2). Cutting intensity, as indicated by crown gaps, was extremely high within 50 m of the main road (Fig. 3c) where there, all large trees would have been cut (85.4 trees/km^2). Conversely, the number of crown gaps decreased as the distance from a main road increased and generally ceased by 1.2–1.3 km (Fig. 3a). This finding is consistent with cases reported in Borneo, where logging activity can extend up to 1 km or more in flat terrain (Bryan *et al*., 2013). This could imply that the labour required for transporting cut timber offsite on the flatlands was similar. However, reductions in our detection density in terms of the distance from roads were lower (Fig. 3c). Thus, cutting intensity was relatively uniform within the reaches of the cutting groups, where a significant number of (though not all) large trees were cut (26.3 trees/km^2). Former forest degradation could be more reliably assessed using information on the road network, logging equipment and transportation machinery at a given point in time, and our study provides the basic information for this purpose.

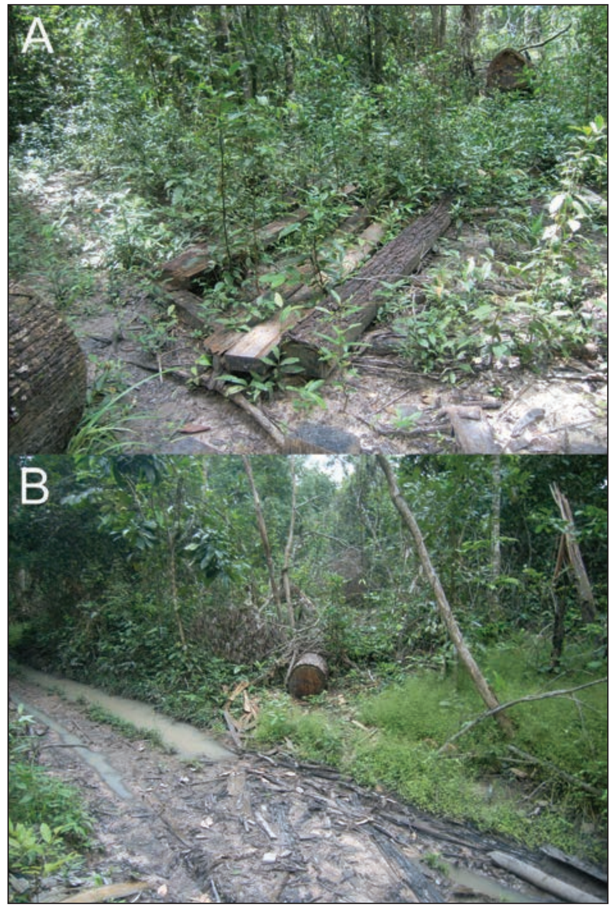

Fig. 9 Cutting roads at study site. A) Former sawmill access roads converted to cutting road, B) Felled juvenile trees adjacent to heavily-used cutting road.

Some cutting teams lacked the ability to foresee decay (Fig. 7) and many valuable trees and genetic resources were wasted as a result. This suggests that the rapid entry of unskilled small-scale operators into cutting may have resulted in unsustainable selective cutting. A cultural shift from timber mining to successful common-pool resource management has been proposed to halt poor logging practices (Putz *et al*., 2000; Zimmerman & Kormos, 2012). Deforestation in Cambodia after 2002 has been linked to population growth, gross agricultural production and large-scale plantation development (Michinaka *et al*., 2013). The relatively high cutting pressure evident at our study site from 2007 to 2009 was probably attributable to a population boom after the civil war and availability of chainsaws. It would have been considerably more difficult to foster a sustainable culture, for example, consensus on sustainable forest management, under rapid development pressure.

It has been noted that unauthorized logging may have led to the export of illegal timber (GIATOC, 2021). Kim *et al*. (2006) assumed that all individuals of *D. costatus* and *A. costata*, the two main forest resource species in Cambodia, were processed into veneer wood, although the exact percentages of processing uses was unknown. Use of dipterocarp trees for construction and plywood manufacturing has also been reported (Barney, 2005; GIATOC, 2021). The demand for wood for domestic housing construction in Cambodia, related to war and population growth, has been substantial (Kim *et al*., 2005), even into the 2000s (Broadhead & Izquierdo, 2010). According to local informants, the large-diameter tree cutting observed in our study area was for house construction. If true, alternative sources of timber supply are needed to halt the practice (Ty *et al*., 2011) and development of a stable and affordable supply of sustainable building materials would contribute to relieving the pressure on local forest resources.

Implications for forest conservation and restoration

Our detection density for crown gaps provides an important value for forest restoration: 100 sites km^{-2} in the closest class to a main road (Fig. 3c). Our field survey confirmed the mortality of individual trees in almost all (98%, 101/103) of the crown gaps detected. Since the selective cutting was exploitative, this estimate can be considered the natural population density of large-diameter dipterocarp trees in the study area. This figure would be useful for future restoration efforts in indicating the structure and biomass of the former forest.

The size of naturally dead trees of *A. costata* (86.3 ± 8.9 (54.1–133.8) cm) appeared to be smaller than those of *D. costatus* (118.3 ± 14.6 (77.0–163.0) cm), although the size distribution in the natural population did not differ greatly (Fig. 4a, 4b). Additionally, the quantity of naturally dead stumps of *A. costata* was significant. The reason for this is unknown, but it could have been a situation where natural mortality of *A. costata* was evident. Many tropical forest studies have shown that mortality rates in logged and unlogged forests are similar, except for the high mortality rate of trees damaged during logging (Sist & Nguyen-The, 2002; Yamada *et al*., 2013; Stas *et al*., 2023). However, disturbance-sensitive species, even in the absence of incidental damage during logging, may decline as a result of exposure of residual trees to constraining conditions such as increased wind stress or windthrow (Figueira *et al*., 2008; Garcia-Florez *et al*., 2017). Should *A. costata* be a disturbance-sensitive species, it could experience increased mortality due to surrounding forest degradation. In this case, it should be given special consideration in efforts to conserve local tree populations.

In conclusion, our study characterises the early stages of forest degradation in lowland evergreen forests due to intensive cutting of large-diameter dipterocarp trees. Under the logging moratorium, cutting activity was performed by several groups with different logging practices and covered an area up to about 1.3 km from the main road. The information we provide can help quantify forest degradation and is valuable for the REDD+ framework and forest restoration efforts. Steady accumulation of current and past insights on many aspects of forest information is essential to improve forest management in Cambodia.

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Freshwater health index assessment of the Tonle Sap basin

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មូលន័យសង្ខេប

បឹងទន្លេសាប និងអាងទន្លេគ្របដណ្តប់ស្ទើរតែពាក់កណ្តាលនៃផ្ទៃដីប្រទេសកម្ពុជា ដែលផ្ទុកនូវធនធានជលផលទឹកសាបដ៏ធំបំផុតមួយនៅលើពិភពលោក និងសម្បូរទៅដោយជីវចម្រុះ។ ខណៈដែលបឹងទន្លេសាប និងទំនាក់ទំនងរបស់វាជាមួយទន្លេមេគង្គត្រូវបានគេសិក្សាយ៉ាងទូលំទូលាយ រហូតមកដល់បច្ចុប្បន្ននេះសុខភាពទឹកសាបរបស់អាងទន្លេនេះត្រូវបានគេមើលរំលង។ យើងបានប្រើសន្ទស្សន៍សុខភាពទឹកសាបដើម្បីធ្វើការវិនិច្ឆ័យស្ថានភាពអាងទន្លេនៅខែធ្នូ ឆ្នាំ២០២១។ ភាពរឹងមាំនៃប្រព័ន្ធអេកូឡូស៊ីទទួលបានពិន្ទុ៤១/១០០ ឯសេវាកម្មប្រព័ន្ធអេកូឡូស៊ីទទួលបានពិន្ទុ៧៥/១០០ ខណៈដែលអភិបាលកិច្ច និងភាគីពាក់ព័ន្ធទទួលបានពិន្ទុ៥៨។ ដូចគ្នានឹងការវាយតម្លៃសុខភាពទឹកសាបនៅផ្នែកផ្សេងៗនៃទន្លេមេគង្គ អាងទន្លេសាបផ្តល់នូវសេវាកម្មប្រព័ន្ធអេកូឡូស៊ីដ៏មានតម្លៃ។ ប៉ុន្តែសមាសភាពរបស់ប្រព័ន្ធជីវរូបបាតទឹករបស់វាត្រូវបានថយចុះ។ បណ្តាញទន្លេដែលបែកខ្ញែកគ្នាយ៉ាងខ្លាំងនៃអាង និងចំនួនខ្ពស់នៃប្រភេទរងគ្រោះ (ជាពិសេសត្រី) គំរាមកំហែងដល់អនាគតនៃជលផលដ៏សំខាន់របស់បឹង ដែលក្នុងនោះទទួលបានពិន្ទុមធ្យម៦០។ ភាគីពាក់ព័ន្ធបានវាយតម្លៃប្រព័ន្ធអភិបាលកិច្ចទឹក និងកម្រិតនៃការចូលរួមរបស់ភាគីពាក់ព័ន្ធថាមានកម្រិតមធ្យម។ ខណៈដែលជម្លោះនៅមានកម្រិតទាប ការអភិវឌ្ឍន៍ហេដ្ឋារចនាសម្ព័ន្ធប្រព័ន្ធធារាសាស្ត្រទំនងជានឹងប៉ះពាល់ដល់លំហូរទៅកាន់បឹង និងជះឥទ្ធិពលដល់ជលផល។ ភាគីពាក់ព័ន្ធត្រូវបានបែងចែកសារៈសំខាន់លើការផ្គត់ផ្គង់ទឹកដែលអាចជឿទុកចិត្តបាន និងធនធានជលផល ដែលបង្ហាញពីសក្តានុពលនៃចំនុចអាចកើតមានជម្លោះនាពេលអនាគត។ យើងបានកំណត់អត្តសញ្ញាណកង្វះទិន្នន័យ បង្ហាញពីថាមវន្តសង្គមសំខាន់ៗ ផ្តល់ឲ្យភាគីពាក់ព័ន្ធនូវទស្សនៈវិស័យទូលំទូលាយអំពីអាងទន្លេ និងបានគូសបញ្ជាក់ពីសារៈសំខាន់នៃបរិស្ថានដែលមានសុខភាពល្អសម្រាប់អនាគតនៃប្រព័ន្ធធនធានធម្មជាតិដ៏មានសារៈសំខាន់របស់ប្រទេសកម្ពុជា។

Abstract

The Tonle Sap Lake and River basin cover almost half of Cambodia's land surface, contain one of the world's largest inland fisheries and are rich in biodiversity. While the Tonle Sap Lake and its relationship to the Mekong River is well studied, until now the freshwater health of the basin has been overlooked. We used a freshwater health index to diagnose the basin's condition as of December 2021. Ecosystem vitality and ecosystem services scored 41 and 75, respectively, out of a possible 100, while governance and stakeholders scored 58. Consistent with freshwater health assessments in other parts of the Mekong, the Tonle Sap basin provides valuable ecosystem services. But components of its underpinning biophysical system are degraded. The basin's highly fragmented river network and high numbers of threatened species (particularly fish) threaten the future of the lake's vital fishery, which received a moderate score

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of 60. Stakeholders rated the water governance system and degree of stakeholder engagement as moderate. While the degree of conflict was low, the development of irrigation infrastructure will likely impact flows to the lake, thus affecting the fishery. That stakeholders were divided over the importance placed on the provision of a reliable water supply versus the fishery, indicates a potential future point of conflict. We identified data deficiencies, revealed important social dynamics, provided stakeholders with a basin wide perspective, and highlighted the importance of a healthy environment for the future of Cambodia's most important natural resource system.

Keywords Ecosystem services, ecosystem vitality, freshwater health index, governance, Tonle Sap basin.

Introduction

The Tonle Sap Lake is central to Cambodia's cultural identity and economy. The lake is a UNESCO Biosphere Reserve and central to what is arguably the world's largest and most productive inland fishery, the lower Mekong basin (Baran *et al*., 2013). The Tonle Sap basin provides 80% of the protein intake of Cambodian people (Hortle, 2007), supports extensive rice production (Mahood *et al*., 2020) and is home to 1.7 million people (Salmivaara *et al*., 2016). The Tonle Sap Lake and its flooded forests provide habitat for ca. 300 species of fish (Baran *et al*., 2013) and numerous threatened birds and mammals (Campbell *et al*., 2006). The Tonle Sap River connects the lake to the Mekong River and its unique pattern of seasonally reversing flow drives much of the lake's productivity. While the Tonle Sap Lake and its relationship with the Mekong River has been extensively studied (Uk *et al*., 2018), its basin has been largely overlooked. Ninety-five percent of this 87,940 km^2 basin lies within Cambodia (with 5% in Thailand) and comprises 46% of Cambodia's land surface. The basin contains extensive areas of forest and rice fields, the settlements of Battambang and Siem Reap, and the World Heritage listed Angkor Archaeological Park. The Tonle Sap Lake and river continue to be degraded due to overfishing (KC *et al*., 2017; Ngor *et al*., 2018b), climate change (Daly *et al*., 2020), hydropower dam development, irrigation, sand mining (Chua *et al*., 2022) and deforestation (Lohani *et al*., 2020; Chen *et al*., 2022).

We conducted a freshwater health index (FHI) assessment to gain a holistic understanding of the health of the Tonle Sap basin and inform management of this vital part of Cambodia and wider Mekong system. This is the second FHI assessment undertaken in the Lower Mekong basin, as a previous study assessed the transboundary Sesan, Srepok and Sekong River basin (Souter *et al*., 2020). The FHI is a nested, quantitative indicator system that assesses three interrelated components of freshwater health: ecosystem vitality, the health of freshwater ecosystems; ecosystem services, water-associated provisioning, regulating & cultural services; and stakeholders & governance, the people who have an interest in or influence over freshwater ecosystems and the rules, regulations and institutions that regulate the way in which stakeholders engage with freshwater ecosystems (Vollmer *et al*., 2018). The FHI aggregates data and knowledge from the social and natural sciences under a social-ecological framework to characterize the health of freshwater systems on a scale of 0–100. The process of undertaking an FHI assessment assists stakeholders in understanding freshwater ecosystem dynamics, how these are manipulated to affect water-related services and how the governance regime manages these dynamics. We use the results of the FHI assessment to make a series of recommendations to improve the freshwater health of the Tonle Sap basin.

Methods

Indicator calculation

We calculated scores for all major indicators within the three components of the FHI: ecosystem vitality, ecosystem services and governance & stakeholders. These scores were derived from assessments of 28 sub-indicators which were calculated using standard and modified methods. Several sub-indicators—groundwater storage depletion, sediment regulation and recreation—were not calculated due to a lack of data. Standard methods were calculated using the FHI toolbox (Shaad & Alt, 2020) and are described in the FHI user manual (FHI, 2021). Readers should consult the user manual for full details of the methodology, as modifications only are described hereafter. The assessment used the most current data available up to 31 December 2021.

Common data sets

We developed two new datasets which were used to calculate several FHI indicators. The Tonle Sap basin network is a combination of the '*Level 7 HydroBasins*' classification (Lehner & Grill, 2013) and the '*Major Flood Extent of the Tonle Sap Lake and Mekong Flooding*' map

(MLMUPC, 2011). This comprises 33 sub-basins which include the extent of permanent water of the Tonle Sap Lake and the inundated floodplain of the Tonle Sap Lake and Sen River as discrete sub-basins. The remaining 31 are river sub-basins, some of which were combined to remove very small basins. We derived the Tonle Sap River network from the HydroSHEDS 15 arc-second resolution drainage direction map (Lehner *et al*., 2006).

Ecosystem vitality

Water quantity was assessed as the deviation from natural flow regime (DvNF) metric as we did not assess groundwater storage depletion due to a lack of data. We calculated DvNF for pre- and post-regulation Tonle Sap Lake and river levels from three sites (Fig. 1, Table 1) using data obtained from the Mekong River Commission (MRC). The pre- and post- periods were summarized into a single average year by calculating the mean lake/river level for each month. We calculated the metric using the formula described in FHI (2021). The basin wide DvNF score was the geometric mean of DvNF scores from the three locations.

Water quality is the geometric mean of six surface water quality sub-indicators: total suspended solids (TSS), total phosphorous (TP), total nitrogen (TN), total nitrate and nitrite (NO_2 & NO_3), Chemical Oxygen Demand (COD) and pH. We used the ecosystem service indicator method 2 (Shaad *et al*., 2022) to calculate a water quality index score for each parameter. A total of 1,249 monthly or bimonthly samples collected between August 1993 and December 2017 from six MRC monitoring stations (Backprea, Kampong Chnang, Kampong Luong, Phnom Krom, Phnom Penh Port and Prek Kdam: Fig. 1) were available and employed in analysis. We assessed water quality data for the last five years of sampling (2013–2017) against the lower Mekong basin protection of aquatic ecosystem thresholds for TP (< 0.13 mg/L) and NO_2 & NO_3 (0.5 mg/L) and protection of human health for COD (5 mg/L) (Ly & Larsen, 2016). The lowland river threshold was used for TN (<1.6 mg/L) (Hart *et al*., 1999). We established monthly TSS thresholds by calcu-

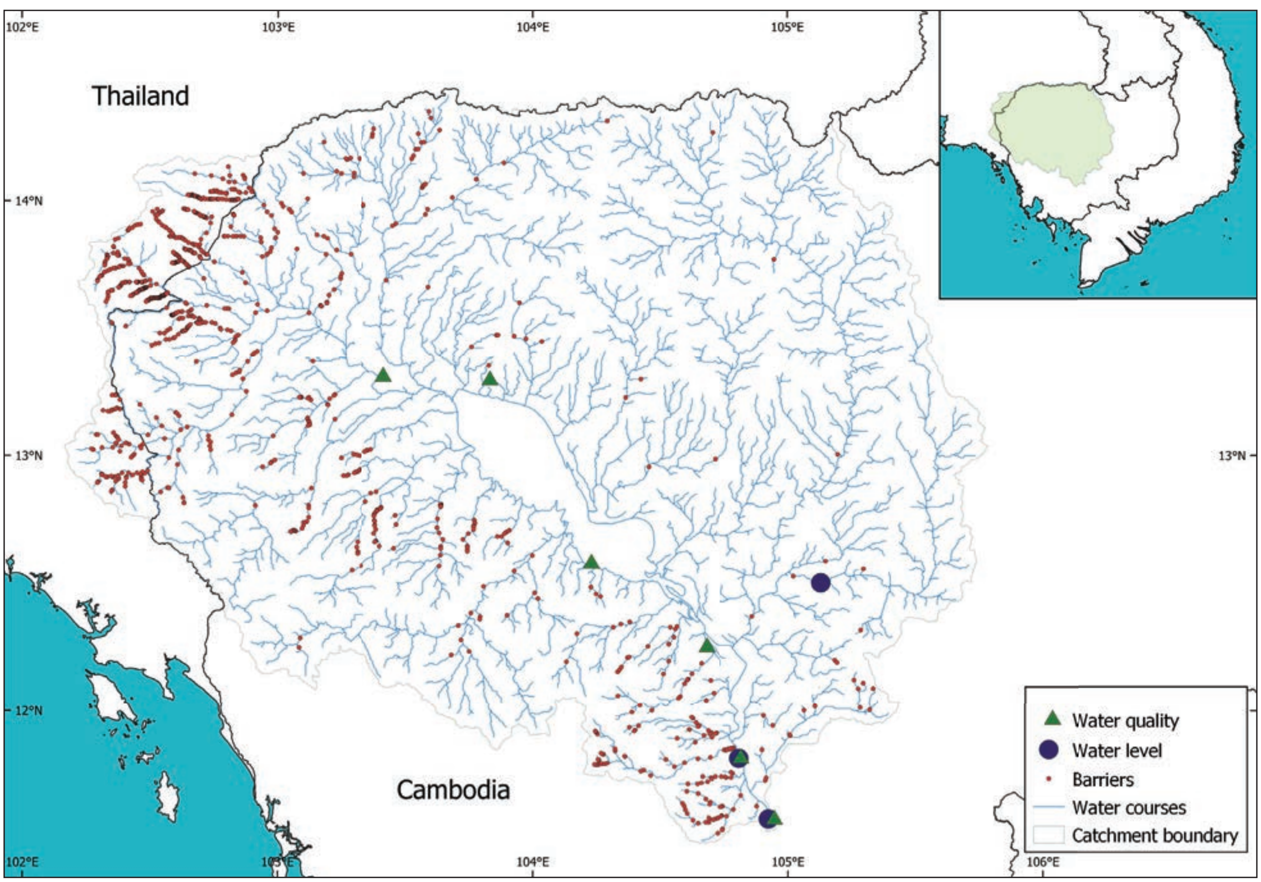

Fig. 1 Tonle Sap basin showing river height gauging stations, water quality monitoring stations and dams/barriers.

Table 1 Tonle Sap Lake and River level gauging stations used to calculate deviation from natural flow regime.

Gauge Location (No.)	Latitude (°N)	Longitude (°E)	Pre Regulation Period	Post Regulation Period
Prek Kdam (020102)	11.811	104.807	1 January 1960 – 31 December 1990	1 January 2014 – 31 December 2018
Kampong Thmar (620101)	12.501	105.1308	1 January 1962 – 30 April 1970; 19 May 1981 – 31 December 1990	1 January 2014 – 31 December 2018
Phnom Penh Port (020101)	11.573	104.923	1 January 1960 – 31 December 1990	1 January 2014 – 31 December 2018

lating the minimum and maximum TSS values for each calendar month from the earliest date of sampling to December 2012. We used these values as baseline thresholds, against which we assessed monthly data from January 2013 to December 2017.

Drainage basin condition is the geometric mean of three sub-indicators: bank modification, flow connectivity and land cover naturalness.

Bank modification measures the extent of unmodified river channel in the basin. Calculated in the FHI Toolbox, we used a 200 m buffer around the Tonle Sap River network and land cover data from the Cambodian Ministry of Environment's '*Cambodian Forest Cover 2016*' shapefile (MoE, unpubl. data). We used this dataset as it was of higher resolution than regional land cover datasets, though it did not include the 5% of the basin located in Thailand. Land cover types were assigned scores at an expert workshop held in Phnom Penh on 6 May 2022 based on the following criteria: degree of naturalness, degree of human management of the water cycle to maintain this land cover, degree of pollution emissions and vegetation characteristics (Table 2).

Flow connectivity assesses the disruption caused by dams and other barriers. Following Shaad *et al*. (2018), we used the unmodified HydroSHEDS 15 arc-second resolution drainage direction map and designated the junction of the Tonle Sap and Mekong Rivers as the basin outlet. We identified 1,076 dams and barriers using MRC (2015), ODC (2015), WLE Mekong (2016), local knowledge and examination of Google Earth satellite imagery. Most barriers were identified using Google Earth and ranged from numerous small earthen embankments—most abundant in Thailand—to large weirs and dams. We could only confirm the presence of fish passage for three barriers (Table 3). While the effectiveness of these three passages is unknown, we assigned a passability value of 0.75 to each for analysis. Many of the smaller barriers are likely to be inundated at high water levels allowing fish to pass but as we could not confirm this, all other barriers received a passability score of 0.

The dendritic connectivity index (DCI) evaluates both the loss of connectivity between the Tonle Sap basin and the Mekong River (DCId), which affects migratory fish, and between the various segments created within the basin due to the dams (DCIp), which also affects non-migratory fish. We calculated a combined index (cDCI) weighted by the proportion of migratory vs. non-migratory fish complied from eleven sources (Annex 1). Of the 356 species recorded, 191 were classified as white or migratory fish and 75 as resident grey or blackfish. Of the remainder, 24 were estuarine and 63 could not be classified on the information provided and were not included in our analysis. We set the percentage of migratory vs. non migratory fish at 72% and 28% respectively.

Land cover naturalness is a proxy indicator for the degree to which a river basin's natural systems regulate pollution, flooding, erosion and changes to infiltration and run-off. We calculated land cover naturalness using the FHI toolbox and by deriving land cover data from the Cambodian MoE's '*Cambodian Forest Cover 2016*' shapefile (MoE, unpubl. data) and the naturalness scores in Table 2.

Biodiversity signifies ecosystem health, with declining populations of native species, increasing numbers of threatened species and increasing populations of invasive and nuisance species indicating deteriorating conditions or ecosystem degradation. We used the FHI toolbox to calculate the biodiversity indicator and its two sub-indicators: species of concern, and invasive and nuisance species. Species of concern comprises three components. The first component comprises the proportion of threatened freshwater species, which was calculated using spatial data from IUCN (2019) for amphibians, terrestrial mammals, reptiles and the freshwater polygon groups for fish, molluscs, plants, odonates, shrimps, crayfish and crabs, and bird data from Birdlife International

Table 2 Land Use/Land Cover types, naturalness score and dengue exposure.

Raster Category	Raster ID	Naturalness Score	Dengue Exposure	Description	Degree of Naturalness; Water Cycle Modification; Vegetation
Evergreen forest	1	100	0	Trees maintaining their leaves all year	Natural and semi-natural; None; Native
Semi-evergreen forest	2	100	0	Mixed evergreen and deciduous trees	
Deciduous forest	3	100	0	Mixed dry deciduous and dry dipterocarp forest	
Flooded forest	9	100	0.5	Tonle Sap Lake forests and shrublands	
Water	22	100	0	Open fresh water	
Grassland	17	100	0.25	Grasslands	
Wood shrub	5	60	0	Evergreen and deciduous woodland < 5 metres high	Cultural assisted sys-tem; Low; Mixed, high diversity
Forest regrowth	10	60	0	Naturally regenerated forest previously impacted by logging, agricultural land use, and human induced fire, etc.	
Rubber plantation	8	50	0.5	Existing rubber plantations	Transformed system; Low to Moderate; Permanent cover with atypical species
Tree plantation	14	50	0.5	Introduced trees (e.g., eucalyptus, cashew etc.)	
Paddy field	15	50	0.5	Rice paddy field	Transformed system; Low to Moderate; Seasonal cover with atypical species
Crop land	16	30	0.5	Arable and tillage land, agroforestry systems under the tree plantation and forest thresholds.	Transformed system; Moderate to High; Seasonal cover with atypical species
Village	19	10	1	Houses and gardens	Completely artificial; Moderate to high; Sparse to no cover
Rock	20	10	0.25	Naturally exposed rocks or mines, quarries and gravel pits.	
Sand	21	10	0.25	Thin soil or sand, dry salt flats, beaches, sand dunes.	
Built-up area	18	1	1	Buildings and construction	Completely artificial; High; Sparse to no cover

(2019). We included all listed aquatic species except those classified as possibly extant, due to a lack of confirmed records. The second component, change in the number of species of concern, was not calculated as this is the first basin assessment, whereas the third component, average population trend, was calculated using unpublished nest count data comparing 2017 and 2021 numbers for seven species of colonial nesting birds at Prek Toal (MoE, unpubl. data).

The number of invasive and nuisance species in the Tonle Sap basin was determined through a literature review (primarily van Zalinge (2006)) and information from regional experts who assessed the degree of invasiveness of introduced species present within the basin.

Ecosystem services

The ecosystem services metric comprises three major indicators: provisioning, regulation & support and cultural &

Table 3 Barriers in the Tonle Sap Lake basin with known fish passage.

Barrier Name	Latitude (°N)	Longitude (°E)
Sala Taaun	13.141	103.221
Stung Pursat Weir	12.487	103.809
Stung Pursat Weir	12.333	103.702

aesthetic. Unless otherwise stated, we employed method 2 of Shaad *et al*. (2022) to calculate each sub-indicator for provisioning and regulation & support using data that measured either one or more of, a spatial (F_1) measure of the system's ability to provide the ecosystem service, a temporal (F_2) measure of how frequently the system fails to provide the ecosystem service, and the magnitude (F_3) of deviation from the threshold value.

Provisioning is the geometric mean of two sub-indicators: water supply reliability relative to demand and biomass for consumption. We calculated water supply reliability relative to demand as the geometric mean of monthly average sustainable irrigation areas (%) across Cambodia from the 2020 scenario in MRC (2018: section 6.1.1, Table 6–8, p. 43). Biomass for consumption was calculated from 2,951 daily fish catch monitoring records collected from four community fisheries areas (Anlong Reang and Ou Ta Prok in Pursat Province; and Doun Sdaeung and Pov Veuy Senchey in Kampong Thom Province) on the Tonle Sap Lake from 1 January 2015 to 31 August 2019. One fisherman in each community recorded the total weight of fish they caught each day. As fishers considered a daily catch of 1.5 kg of fish to be the minimum required to meet their daily subsistence needs, we set this value as the threshold. In the event of fishers catching no fish we calculated the metric using a nominal weight of 0.01 kg, otherwise the equations were intractable.

Regulation & Support is the geometric mean of three sub-indicators: deviation of water quality metrics from benchmarks, flood regulation and exposure to water-associated diseases. We calculated deviation of water quality metrics from benchmarks using 20 surface water quality parameters: TSS, TP, TN, pH, electrical conductivity (EC), dissolved oxygen (DO), COD, total NO_2 & NO_3, ammonia (NH_3), ammonium (NH_4), calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), alkalinity, chloride (Cl), sulphate (SO_4), Ca/Mg, Na/Cl, Na/K, Ca/SO_4. Samples were obtained at the same time and from the same sites as those assessed for the ecosystem vitality water quality indicator. Lower Mekong basin benchmark values for protection of human health were used to assess pH (6–9), DO (4 mg/L), COD (5 mg/L), NO_2 & NO_3 (5 mg/L) and NH_3 (0.5 mg/L). We adopted an agricultural value for EC of 700 mS/m (Ly & Larsen, 2016) and the lowland rivers threshold for TN (<1.6 mg/L) (Hart *et al*., 1999). Water quality data for 2013–2017 were compared against these benchmarks. For the other parameters, we established monthly minimum and maximum TSS thresholds for the ecosystem vitality water quality indicator (Souter *et al*., 2020). Five of these parameters (TOTP, Mg, Cl, $NO_2 + NO_3$ and NH_4) recorded zero values, which were converted to 0.01 to enable the indicator to be calculated.

We calculated flood regulation from the Prek Kdam (No. 020102) gauging station on the Tonle Sap River which had both flood and flood warning levels (10 m & 9.5 m respectively) using the water level time series data from 1 January 2014 to 31 December 2018.

We calculated exposure to water-associated diseases as dengue fever exposure using the water associated disease index (WADI: Dickin *et al*. 2013) exposure indicator. We developed the index following Souter *et al.* (2020) using four datasets (Table 4) analysed using the Google Earth Engine to derive exposure values for each month of 2021 and land cover exposure values (Table 2). We set an exposure value of 0.25 and the final indicator score was the geometric mean of the 12 monthly values.

Table 4 Dengue WADI exposure indicator components and data analysed in Google Earth Engine.

Component	Dengue WADI Factor	Data Source
Climate	Maximum temperature; Precipitation	Wan *et al*. (2021); Funk *et al*. (2015)
Land environment	Land cover	'*Cambodia Forest Cover 2016*' shapefile (MoE, unpubl. data)
Human environment	Population density	CIESIN (2018)

Cultural & aesthetic ecosystem services were calculated using the conservation/cultural heritage sites sub-indicator. We calculated this using a protected areas map derived from two sources (ODC, 2016; IUCN & UNEP-WCMC, 2017) per Souter *et al.* (2020). We did not calculate the recreation sub-indicator because although there are tourism operations on the Tonle Sap Lake, evaluation required a dedicated survey for which we did not have the resources.

Governance & stakeholders

The governance & stakeholders metric comprises four major indicators: enabling environment, stakeholder engagement, vision & adaptive governance and effectiveness, which include 12 sub-indicators. We assessed the governance & stakeholders metric via an online questionnaire in English (Annex 2) which asked stakeholders to rate their level of agreement with 54 statements using a standard five-point Likert scale (e.g., Vollmer *et al*. 2021). Survey responses were anonymous and 15 people with specific knowledge of the Tonle Sap and its governance system completed the survey. While their perceptions were their own, they were employed by government agencies, non-government organizations and academic institutions.

Indicator weighting

To ensure that our aggregated values for ecosystem services and governance & stakeholders reflected stakeholder preferences, we asked the stakeholders that completed the governance & stakeholders survey to also complete the FHI indicator weighting exercise. Respondents used a swing weighting approach (Edwards & Barron, 1994), which was conducted online in English following Souter *et al*. (2020). Their individual weights were aggregated by arithmetic mean, while their level of consensus or agreement on the weights was calculated based on Shannon α and β entropy (Goepel, 2013).

Results

Our Tonle Sap FHI assessment gave three scores: Ecosystem vitality=41, ecosystem services=72 and governance & stakeholders=58 (Fig. 2).

Ecosystem vitality

Water quantity scored 66 with the two lake sites, Prek Kdam (68) and Kampong Thmar (67), scoring slightly higher than Phnom Penh Port (64) which is located at the junction of the Tonle Sap and Mekong Rivers.

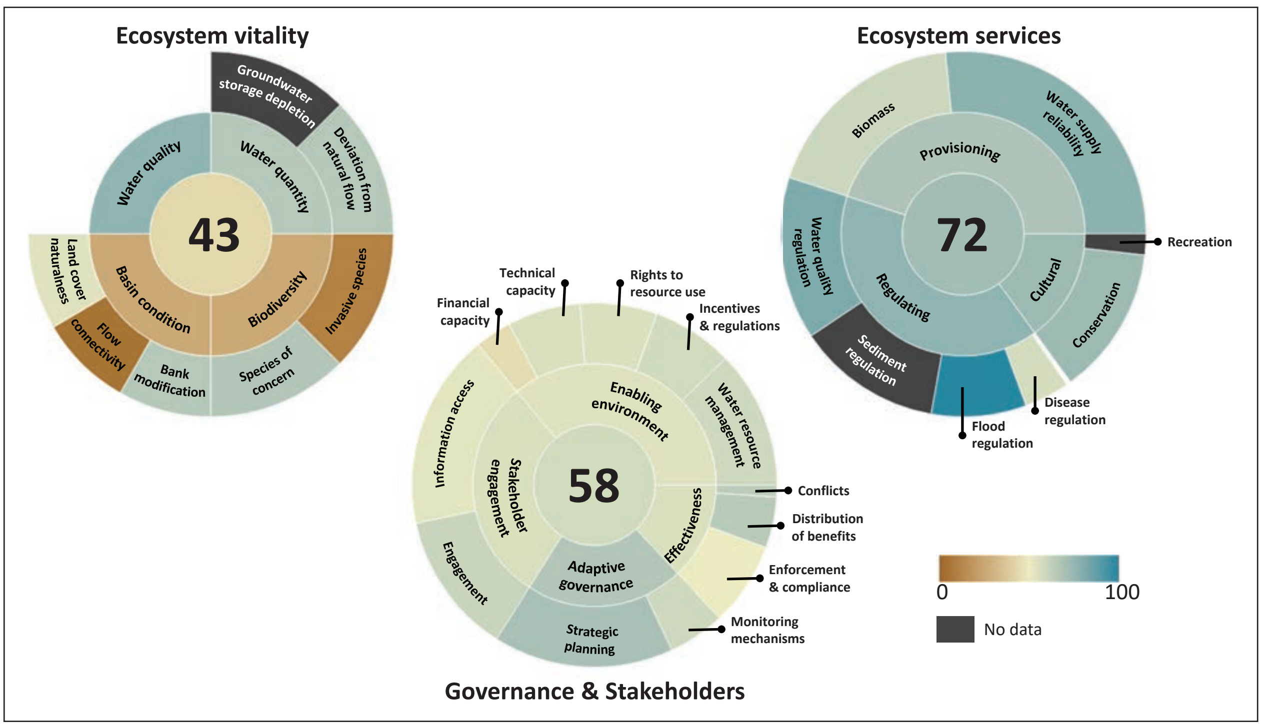

Fig. 2 Summary FHI scores for the Tonle Sap basin, as of December 2021. The size of each wedge reflects its relative weight in determining aggregated indicator (or component) scores.

Water quality scored 77 and comprised high scores for all parameters except for COD and TP which had moderate and low scores respectively (Table 5).

Drainage basin condition scored 22, which comprised scores for bank modification (66), flow connectivity (3) and land cover naturalness (54). For bank modification, the highest proportion of water course banks ran through rice fields (rubber and tree plantations were minor components), followed by natural vegetation (Table 6). For land cover, rice fields were the predominant land type followed by natural vegetation (Table 6).

Biodiversity scored 26. Species of concern scored 67 as we recorded 72 threatened species out of a total of 1,057. Most threatened and Critically Endangered species were fish (Actinopterygii), which were also the most species-rich group (Table 7). Nest counts for three of the seven bird species monitored at Prek Toal declined between 2017 and 2021: milky stork *Mycteria cinerea* (-60%), painted stork *M. leucocephala* (-6%) and Asian openbill *Anastomus oscitans* (-9%). Nest counts for the remaining four species increased: greater adjutant *Leptoptilos dubius* (+21%), lesser adjutant *L. javanicus* (+24%), spot-billed pelican *Pelecanus philippensis* (+25%) and oriental darter *Anhinga melanogaster* (+6%).

The invasive and nuisance species indicator scored 10 as we identified nine highly invasive species (Table 8). Van Zalinge (2006) recorded 21 exotic species from the Tonle Sap Biosphere Reserve and surrounds. Of these, we identified 16 species as having either high or moderate impact or abundance in the Tonle Sap Basin. Twelve of these were plants, three were fish and one was an invertebrate.

Table 5 Individual F_1, F_3 and ecosystem service indicator (ESI) scores for water quality parameters in 2013–2017.

Parameter	F_1	F_3	ESI
Total nitrogen	100	1.0	90.2
Total phosphorus	100	28.2	46.9
pH	83.3	0.4	94.6
Chemical oxygen demand	100	9.2	69.7
Total nitrate & nitrite	75	4.95	80.7
Total suspended solids	66.7	3.75	84.2

Table 6 Bank modification (BM) and land cover naturalness (LCN) proportions for the Tonle Sap basin.

Degree of Naturalness / Vegetation	% Modification	
	BM	LCN
Natural and semi-natural: Native	33.8	18.9
Cultural assisted system: Mixed high diversity	6.8	4.9
Transformed system: Permanent cover with atypical species / rice paddy	40.5	55.1
Transformed system: Seasonal cover with atypical species	16.6	13.6
Completely artificial: sparse cover with grass	2.1	5.5
Completely artificial: none	0.2	1.9

Ecosystem services

Ecosystem services scored 75 and comprised scores of 70 for provisioning (weighting of 0.45), 82 for regulation & support (weighting 0.40) and 73 for cultural & aesthetic (weighting 0.15). The weightings achieved a high degree of consensus at 68%.

Provisioning (70) comprised indicators for water supply relative to demand which had a score of 78 (weighting 0.59) and biomass for consumption, which had a score of 60 (weighting 0.41). Fishers at all four sites failed to catch sufficient fish for their daily subsistence on at least one occasion (F_1=100). They also fell below the threshold on 51 occasions (F_3=16). The fisher in Pov Veuy Senchey caught no fish on three occasions between 14–16 April 2017. There was a low degree of consensus (27%) in the weightings between the two sub-indicators, highlighting that stakeholder opinions were polarized i.e., strongly favouring either water supply or biomass.

Regulation & support scored 82 and comprised three indicators for which the weightings received a high degree of consensus (80%). While we could not assess sediment regulation, the weighting this received from stakeholders (0.32) suggests they viewed it as an important service worthy of further investigation.

Deviation of water quality metrics from benchmarks scored 81 (weighting 0.36). Individual indicators ranged from 42 for dissolved oxygen to 100 for electrical conductivity and ammonia (Table 9). Eleven parameters had an F_1 score of 100, which meant that they exceeded threshold values at every site at least once. However, for these parameters, the F_3 values were much lower, showing that threshold breaches were generally infrequent and of a small magnitude.

Table 7 Number of IUCN Red List species present in the Tonle Sap basin categorized by higher taxonomic groups. Status: CR=Critically Endangered, EN=Endangered, VU=Vulnerable, NT=Near Threatened, LC=Least Concern, DD=Data Deficient.

Status	Actinopterygii	Aves	Reptilia	Amphibia	Mollusca	Mammalia	Odonata	Plantae	Decapoda	Total
CR	6	2	3					1		12
EN	5	6	2	1	2	1		1	1	19
VU	15	3	6	6	13	3	2	2	1	51
NT	11	15		3	20			1	1	51
LC	315	76	90	38	77		142	122	30	923
DD		1								1
Total	352	102	101	48	112	4	144	127	33	1,056

Table 8 Invasive species present in the Tonle Sap basin, based on the risk of species being invasive in the Tonle Sap Biosphere Reserve according to van Zalinge (2006). Invasive score was assigned by regional experts: 1=High impact/abundance, 2=Moderate impact/abundance.

Common name	Binomial	Family	Invasive Risk	Invasive Score
Giant mimosa	*Mimosa pigra*	Fabaceae	Major threat	1
Giant sensitive mimosa	*Mimosa invisa*	Fabaceae	Uncertain	1
Sensitive mimosa	*Mimosa pudica*	Fabaceae	Uncertain	2
Water lettuce	*Pistia stratiotes*	Araceae	Low/abundant	2
Candlebush	*Senna alata*	Fabaceae	Low	1
Seedbox	*Ludwigia hyssopifolia*	Onagraceae	Low	1
Para grass	*Urochloa mutica*	Poaceae	Uncertain / common	2
Hippo grass / creeping paddy weed	*Echinochloa stagnina*	Poaceae	Uncertain / locally dominant	2
Cutgrass	*Leersia hexandra*	Poaceae	Uncertain / locally dominant	2
Water hyacinth	*Eichhornia crassipes*	Pontederiaceae	Invasive	1
Guinea grass	*Megathyrsus maximis*	Poaceae	-	1
Apple snail	*Pomacea* spp.	Pilidae	High	1
Red-bellied pacu	*Piaractus brachypomus*	Characidae	Substantial	1
Nile tilapia	*Oreochromis niloticus*	Cichlidae	Uncertain	2
African catfish	*Clarias gariepinus*	Clariidae	Uncertain	1

Flood regulation scored 100 (weighting 0.21) as the highest river level recorded during the assessment period (9.03 m) was less than either the flood or flood warning levels. Water associated diseases scored 57 (weighting 0.10). Exposure to dengue fever was highest in August (ecosystem service indicator [ESI] score of 40) and lowest in May (ESI score 83).

Cultural & aesthetic scored 73 (weighting 0.15). One hundred and eighty-nine kilometres of river bordered protected areas and 4,220 km of river were contained within protected areas. This gave a percentage of river length protected score of 28. Stakeholders gave conservation of cultural heritage a weighting of 0.88 compared to 0.12 for recreation (not assessed) with a high degree of consensus at 95%.

Table 9 Deviation of water quality metrics from benchmarks (DyWO) F_1, F_3 and indicator scores for 21 water quality parameters for 2013–2017.

Water Quality Parameter	F_1	F_3	DyWO Score
Total suspended solids	50	4.3	85
Total phosphorous	66.7	2.9	86
Total nitrogen	100	1.0	90
pH	83.3	0.4	95
Electrical conductivity	0	0	100
Dissolved oxygen	100	34.0	42
Chemical oxygen demand	100	9.2	70
Total nitrate and nitrite	83.3	2.8	85
Ammonia	0	0	100
Ammonium	50	0.3	96
Calcium	100	0.8	91
Magnesium	100	4.8	78
Sodium	100	4.8	78
Potassium	100	0.5	93
Alkalinity	100	0.9	91
Chloride	100	18.2	57
Sulphate	83.3	0.7	92
Calcium/Magnesium	83.3	3.1	84
Sodium/Chloride	100	15.3	61
Sodium/Potassium	83.3	2.2	87
Calcium/Sulphate	100	0.5	93

Table 10 Summary of weighted scores for governance & stakeholder indicators.

Governance & Stakeholders Major- (bold) & Sub- Indicators with Stakeholder Weightings [ ] and Consensus (%)	Weighted Score
Aggregate score	58
Enabling environment [0.36] – 79%	**55**
Water resources management [0.34]	59
Rights to resource use [0.19]	55
Incentives & Regulations [0.20]	58
Technical capacity [0.18]	56
Financial capacity [0.09]	45
Stakeholder engagement [0.30] – 14%	**56**
Information & knowledge [0.58]	53
Engagement in decision-making [0.42]	61
Vision & Adaptive governance [0.21] – 47%	**67**
Monitoring mechanisms [0.24]	59
Comprehensive planning & Adaptive management [0.76]	70
Effectiveness [0.13] – 94%	**56**
Enforcement & compliance [0.57]	50
Distribution of benefits from ecosystem services [0.35]	65
Water-related conflict [0.08]	64

Governance & stakeholders

Governance & stakeholders scored 58 (Table 10). Vision and adaptive governance received the highest major indicator score (67), within which comprehensive planning & adaptive management had the highest sub-indicator score (70) and received the highest weight. Among the other major indicators, enabling environment scored 55, whereas stakeholder engagement and effectiveness both scored 56 (Table 10). Among the sub indicators, financial capacity scored the lowest at 45.

Stakeholder preferences ranged widely for the four major indicators, with enabling environment rated as most important and effectiveness as the least important (Table 10). There was a high degree of consensus in weightings for the major indicators (72%). Weightings within the sub-indicators also varied widely as did the levels of consensus. Although consensus for enabling environment was high (79%), the highest degree of variance within individual questions were those regarding the quality and clarity of rules for handling wastewater and fisheries.

Discussion

Our December 2021 FHI assessment revealed that while the Tonle Sap basin's environment was stressed (with an ecosystem vitality score of 41), it provided ecosystem services (score of 72), although not to the full extent required. This was within a functioning but somewhat variable governance and management system (governance & stakeholders score of 58). Thus, without improvement, the stressed environment may not be able to support the current level of ecosystem service provision in the future. Whether the current system of governance and stakeholder engagement can respond to the need to improve environmental conditions is uncertain.

The low score for ecosystem vitality was due to low scores for drainage basin condition and biodiversity. The land cover naturalness score was influenced by over half the basin having being transformed, largely to grow rice. The assessment's lowest score of three, for flow connectivity, was due to the large number of barriers within the basin. Our assessment identified many more barriers than were previously known from the Tonle Sap basin (see Baran *et al*., 2007). Most barriers were found in Thailand, where long sections of stream appear to have been modified to form small continuous irrigation ponds. Furthermore, recent large-scale irrigation development was recorded in Cambodia. The construction of three fish passes—while acknowledging and attempting to solve the connectivity problem—had no effect on the overall score.

Our application of the flow connectivity index did not account for the full complexity of the Tonle Sap's fish biodiversity. Seventy-two percent of the lakes fish are migratory and many move between the lake and its tributaries. Ideally, we would assess connectivity separately for each tributary, but without tributary specific species lists or population sizes this was not possible. Large numbers of fish migrate long distances from the Tonle Sap Lake up the Mekong River where barriers are also a significant issue (Souter *et al*., 2020). These were also not assessed and neither was the influence of the Dai fishery in the lower reaches of the Tonle Sap River, which blocks fish passage through harvest, although its impact is debated (Grenouillet *et al*., 2021). While a more in-depth assessment of connectivity within the Tonle Sap system is warranted, the large number of barriers we identified will impact on its fish fauna, most of which is migratory. Further threats to the basin's fish fauna were revealed by the biodiversity assessment, as most threatened and critically endangered species were fish. There were also numerous invasive and nuisance species.

Our higher deviation in natural flow regime scores from the lake compared to the Tonle Sap/Mekong River junction suggests that local basin inflow has a moderating effect on larger changes in flow from the Mekong. However, the growth of irrigation and hydropower development within the Tonle Sap Basin—particularly over the last decade—could alter this moderating effect, causing a further departure in the lakes' flow regime from natural conditions.

Our ecosystem vitality score revealed a stressed environment, signs of which are appearing in the provision of ecosystem services. The stress on the basin's biodiverse fish fauna is concurrent with the reduced ability of the lake to provide local fishers with a subsistence level catch. In calculating water supply relative to demand, we relied upon a whole of Cambodia estimate, calculated using a simplified method, rather than Tonle Sap basin specific data. The increase in irrigation development within the Tonle Sap basin suggests that this current high level of service provision is likely to decline in future as water is captured and used for irrigation at the expense of other uses, such as providing flow to sustain the Tonle Sap fishery. While stakeholders weighted water supply relative to demand as being of higher importance than biomass for consumption, there was a low degree of consensus, indicating the potential for future conflict. This highlights the need to accurately assess water consumption and demand within the basin.

Regulation & support received the highest ecosystem service score and all of its components measured scored highly. Deviation of water quality metrics from benchmarks received the highest score for sub-indicators, although point and non-point sources of pollution have been documented around floating villages and the Tonle Sap River (Ung *et al*., 2019; Shivakoti & Pham, 2020; Sor *et al*., 2021). Our flood regulation indicator must be viewed with low confidence as it was calculated from only a single site. Sediment regulation was viewed as the second most important regulation & support service but could not be calculated due to a lack of data. The productivity of the Tonle Sap Lake is driven by high levels of sediment inflow from the Mekong River. This highlights the difficulty of assessing the freshwater health of the Tonle Sap basin in isolation from the rest of the Mekong basin.

One of the most common comments regarding the management system for the Tonle Sap was that while numerous plans and policies had been developed, implementation had been limited due to a lack of resources. This was supported by our survey results with comprehensive planning & adaptive management receiving the highest sub-indicator score, whereas financial capacity and enforcement & compliance received the lowest (Table 10). Jurisdictional overlap between government departments and moves to decentralize power to the provinces—which lack technical capacity—were also seen to hinder effective management. The assertions that information to support decision making is often lacking and that stakeholder consultation could be improved were also supported by our survey results.

Our FHI results can guide management of the Tonle Sap basin. First, adequate capacity and resources are required for the implementation of existing management policies and plans. While examining existing plans and policies was beyond the scope of our study, we recommend priority be given to those that address ecosystem vitality, primarily: restoring natural vegetation cover, threatened species conservation, managing the impact of invasive species and improving fish passage. To preserve and increase the supply of ecosystem services, we recommend improved fisheries management and ensuring that irrigation development does not negatively impact the fishery, primarily through reductions in flow to the Tonle Sap Lake. The localized impact of point source water quality pollution and extensive development of the catchment in Thailand also deserves further investigation.

In conclusion, the poor environmental condition of the Tonle Sap basin and areas of stress in delivering ecosystem services is concerning given their importance for Cambodia. Increased irrigation development and consequent future water diversion has the potential for conflict. While the governance system for basin was partially functioning, it needs to be improved to meet the challenges posed by increased development and poor environmental health. Several important areas where data are lacking include water use and sediment supply. The complex relationship between the Tonle Sap Lake, its local basin and the wider Mekong basin presented challenges in undertaking this assessment.

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Annex 1 Fish species of the Tonle Sap basin

The presence of fish species in the Tonle Sap basin were compiled from ten sources (Lim *et al*., 1999; Lamberts, 2001; Chan *et al*., 2008, 2017; Hartmann *et al*., 2013; Kong *et al*., 2017; Ainsley *et al*., 2018; Marsden *et al*., 2018; Ngor *et al*., 2018a; Pool *et al*., 2019). Information in these references regarding migratory life history and that in Baran *et al*. (2014) were used to develop a consensus position on whether fish are Migratory (M), Non-migratory (NM), Estuarine (E), or could not be classified. Mr=Marine, W=White, G=Grey, B=Black, O=Opportunist, P=Present but no migratory information provided. Figures in Baran *et al*. (2014) indicate the level of evidence for the species being migratory, 1 being the lowest 3 the highest.

Species	**Baran *et al.* (2014)**	**Lim *et al.* (1999)**	**Lamberts (2001)**	**Chan *et al.* (2008)**	**Hartmann *et al.* (2013)**	**Kong *et al.* (2017)**	**Chan *et al.* (2017)**	**Marsden *et al.* (2018)**	**Ngor *et al.* (2018a)**	**Ainsley *et al.* (2018)**	**Pool *et al.* (2019)**	**Consensus**
Aaptosyax grypus	2								W			M
Acanthocobitis sp.									B			NM
Acanthopsis spp.									B			NM
Acanthopsis sp. 1				W				P				M
Acanthopsis sp. 5				W				P				M
Acanthopsoides delphax	2			W					W			M
Acanthopsoides gracilentus	P			W					W			M

Annex 1 Cont'd

Species	**Baran *et al.* (2014)**	**Lim *et al.* (1999)**	**Lamberts (2001)**	**Chan *et al.* (2008)**	**Hartmann *et al.* (2013)**	**Kong *et al.* (2017)**	**Chan *et al.* (2017)**	**Marsden *et al.* (2018)**	**Ngor *et al.* (2018a)**	**Ainsley *et al.* (2018)**	**Pool *et al.* (2019)**	**Consensus**
Acanthopsoides hapalias					P							-
Achiroides leucorhynchos	P			W			W		W			M
Albulichthys albuloides	P	P		W			W		W			M
Ambastaia sidthimunki									W			M
Amblypharyngodon chulabhornae										P		-
Amblyrhynchichthys truncatus	2	P		W			W		W			M
Amblyrhynchichthys sp.								P				-
Anabas testudineus	P	P	B	B		NM	B	P	B	P	NM	NM
Anguilla marmorata	3			W					W			M
Arius caelatus				W								M
Arius maculatus				Mr					E			E
Arius sona				Mr								E
Aris stromi		P										-
Arius venosus									E			E
Arius sp.							W					M
Aulopareia janetae									E			E
Bagarius bagarius	P			W					W			M
Balitora meridionalis									W			M
Bagarius suchus				W					W			M
Bagarius yarrelli	2			W								M
Bagrichthys macracanthus								P				-
Bagrichthys obscurus				W					W			M
Bagrius sp.							W					M
Balitoropsis zollingeri									W			M
Bangana behri	2			W								M
Bangana sp.				W					W			M
Barbodes altus		P	W									M
Barbodes aurotaeniatus										P		-
Barbodes rhombeus									G			NM
Barbonymus altus	P			W				P	G			NM
Barbonymus gonionotus	P	P		W				P	W		M	M
Barbonymus schwanenfeldi	P			W					G			M
Belodontichthys dinema		P										-
Belodontichthys truncatus	P			W			W		W			M
Betta prima					P							-
Boesemania microlepis	P	P		W			G		G			NM

Annex 1 Cont'd

Species	**Baran *et al.* (2014)**	**Lim *et al.* (1999)**	**Lamberts (2001)**	**Chan *et al.* (2008)**	**Hartmann *et al.* (2013)**	**Kong *et al.* (2017)**	**Chan *et al.* (2017)**	**Marsden *et al.* (2018)**	**Ngor *et al.* (2018a)**	**Ainsley *et al.* (2018)**	**Pool *et al.* (2019)**	**Consensus**
Botia caudipunctata								P				-
Botia eos								P				-
Botia helodes		P		W								M
Botia modesta				W				P				M
Botia morleti		P		W								M
Botia sidthimunki				W								M
Botia sp. cf. *beauforti*				W								M
Botia sp. cf. *lecontei*		P		W								M
Botia sp.							W					M
Brachirus harmandi	2			W				P				M
Brachirus orientalis	P			W				P				M
Brachirus panoides									E			E
Butis amboinensis									E			E
Catlocarpio siamensis	2	P		W			W		W			M
Channa gachua	P			B	P				B	P		NM
Channa grandinosa				B								NM
Channa lucius	P	P		B			B	P	B			NM
Channa marulioides	P			B					B			NM
Channa melasoma				B								NM
Channa micropeltes	P	P	B	B					B			NM
Channa striata	P	P	B	B	P	NM		P	B	P		NM
Chelonodon fluviatilis		P										-
Chelonodon nigroviridis		P										-
Chitala blanci	2	P		W					W			M
Chitala lopis	P			W					W			M
Chitala ornata	P	P		W			G		W			M
Cirrhinus cirrhosus	P			W					W			M
Cirrhinus jullieni	1			W				P	W			M
Cirrhinus microlepis	2	P	W	W			W		W			M
Cirrhinus molitorella	3	P							W			M
Cirrhinus prosemion				W								M
Clarias batrachus	P	P	B	B	P				B	P		NM
Clarias gariepinus	P			B				P				NM
Clarias macrocephalus	P	P		B				P	B	P		NM
Clarias meladerma	P								B			NM
Clarias nieuhofii									B			NM

Annex 1 Cont'd

Species	Baran *et al.* (2014)	Lim *et al.* (1999)	Lamberts (2001)	Chan *et al.* (2008)	Hartmann *et al.* (2013)	Kong *et al.* (2017)	Chan *et al.* (2017)	Marsden *et al.* (2018)	Ngor *et al.* (2018a)	Ainsley *et al.* (2018)	Pool *et al.* (2019)	Consensus
Clarias sp.									B			NM
Clupeichthys aesarnensis	P	P		W								M
Clupeichthys goniognathus		P		W								M
Clupeichthys sp.											NM	NM
Clupeoides borneensis		P		W							M	M
Clupisoma longianalis									W			M
Clupisoma sinensis	2			W								M
Coilia lindmani	P	P							E		NM	NM
Coilia macrognathos	P	P							E			NM
Coilia sp.							E					E
Corica laciniata	P			W								M
Cosmochilus harmandi	3	P		W			W		W			M
Crossocheilus atrilimes	2			W					W			M
Crossocheilus reticulatus	2							P	W			M
Cyclocheilichthys apogon	1							P		P	M	M
Cyclocheilichthys armatus	2	P				NM			G	P		M
Cyclocheilichthys enoplos	1	P		W		M			W		M	M
Cyclocheilichthys furcatus	3			W								M
Cyclocheilichthys lagleri		P								P		-
Cyclocheilichthys repasson	P			W				P	G			M
Cyclocheilichthys tapiensis				W								M
Cyclocheilos furcatus									W			M
Cynoglossus cynoglossus		P										-
Cynoglossus feldmanni	P	P		W								M
Cynoglossus microlepis	2			W					E			M
Cyprinus carpio	P	P		W					W			M
Dangila cf. *cuvieri*		P										-
Dangila kuhli		P										-
Dangila lineata		P										-
Dangila spilopleura		P	O									E
Danio albolineatus					P							-
Dasyatis laosensis	2			W								M
Datnioides polota	P								W			M
Datnioides undecimradiatus	P			W					W			M
Dermogenys siamensis					P					P		-
Discherodontus ashmeadi	P								W			M

Annex 1 Cont'd

Species	**Baran *et al.* (2014)**	**Lim *et al.* (1999)**	**Lamberts (2001)**	**Chan *et al.* (2008)**	**Hartmann *et al.* (2013)**	**Kong *et al.* (2017)**	**Chan *et al.* (2017)**	**Marsden *et al.* (2018)**	**Ngor *et al.* (2018a)**	**Ainsley *et al.* (2018)**	**Pool *et al.* (2019)**	**Consensus**
Discherodontus parvus									W			M
Devario leptos	P								W			M
Ellochelon vaigiensis				E								M
Esomus longimanus								P		P		-
Esomus metallicus	P								B	P		NM
Esomus sp.											NM	NM
Euryglossa harmandi		P										-
Euryglossa orientalis		P										-
Euryglossa panoides		P										-
Gambusia affinis				B					B			NM
Garra cambodgiensis	P							P				-
Garra fasciacauda	1			W				P	W			M
Glossogobius aureus	P	P							E			NM
Glossogobius giuris									E			E
Glyptothorax fuscus	P								W			M
Glyptothorax laosensis	P								W			M
Gobiidae ksan									B			NM
Gymnothorax tile									E			E
Gyrinocheilus aymonieri	P	P						P				-
Gyrinocheilus pennocki	2			W				P	W		NM	M
Hampala dispar	P	P		W				P	W			M
Hampala macrolepidota	P	P	W	W				P	W		NM	M
Helicophagus waandersi	2			W					W			M
Hemiarius stormii				Mr					W			M
Hemibagrus filamentus	2						W		W			M
Hemibagrus spilopterus	P			W				P	W			M
Hemibagrus wycki	2			W				P	W			M
Hemibagrus wyckioides	1			W					W			M
Hemipimelodus borneensis				Mr								E
Hemipimelodus intermedius				Mr								E
Hemisilurus mekongensis	2			W								M
Hemimyzon pengi									W			M
Henicorhynchus caudimaculatus		P										-
Henicorhynchus cryptopogon		P										-
Henicorhynchus lobatus	3		O	W		M			W	P	M	M
Henicorhynchus siamensis	3	P		W		M		P	W	P	M	M

Annex 1 Cont'd

Species	Baran *et al.* (2014)	Lim *et al.* (1999)	Lamberts (2001)	Chan *et al.* (2008)	Hartmann *et al.* (2013)	Kong *et al.* (2017)	Chan *et al.* (2017)	Marsden *et al.* (2018)	Ngor *et al.* (2018a)	Ainsley *et al.* (2018)	Pool *et al.* (2019)	Consensus
Heterobagrus bocourti				W				P				M
Hypophthalmichthys molitrix	P			W					W			M
Hypophthalmichthys nobilis	P			W								M
Hyporhamphus limbatus	P	P		E				P			M	M
Hypsibarbus lagleri	3			W		M			W			M
Hypsibarbus malcolmi	3			W					W			M
Hypsibarbus pierrei	1							P				M
Hypsibarbus suvattii	P								W			M
Hypsibarbus vernayi				W					W			M
Hypsibarbus wetmorei	2			W				P	W			M
Kryptopterus cryptopterus	P	P		W					W		NM	M
Kryptopterus geminus											NM	NM
Labeo chrysophekadion	P			W		M	G	P	W		M	M
Labeo dyocheilus	P			W					W			M
Labeo rohita	P			W					W			M
Labiobarbus leptocheilus	1								W		M	M
Labiobarbus lineatus	2			W		M			W			M
Labiobarbus siamensis	1			W		M	W	P	W	P	M	M
Laides longibarbis	P			W								M
Laubuka lankensis										P		-
Laubuka laubuca									G			NM
Lepidocephalichthys hasselti					P					P		-
Leptobarbus hoeveni	2	P	W	W					G		M	M
Leptobarbus rubripinna							W					M
Lobocheilos melanotaenia	2	P		W				P	W			M
Luciosoma bleekeri	2			W				P	W		NM	M
Luciosoma setigerum	P	P										-
Lycothrissa crocodilus	P	P		E					E		NM	NM
Macrochirichthys macrochirus	P	P		W			W		G			M
Macrognathus circumcinctus	P			W					B			-
Macrognathus maculatus	P	P										-
Macrognathus siamensis	P	P		W					B	P		-
Macrognathus taeniagaster	P	P										-
Mastacembelus armatus	P	P		W				P	E			-
Mastacembelus erythrotaenia	P	P							W			M
Mastacembelus favus	P	P										-

Annex 1 Cont'd

Species	**Baran *et al.* (2014)**	**Lim *et al.* (1999)**	**Lamberts (2001)**	**Chan *et al.* (2008)**	**Hartmann *et al.* (2013)**	**Kong *et al.* (2017)**	**Chan *et al.* (2017)**	**Marsden *et al.* (2018)**	**Ngor *et al.* (2018a)**	**Ainsley *et al.* (2018)**	**Pool *et al.* (2019)**	**Consensus**
Mastacembelus sp.							B					NM
Megalops cyprinoides				B					E			NM
Mekongina erythrospila	3			W								M
Micronema apogon		P	W	W								M
Micronema bleekeri		P		W				P				M
Micronema cheveyi				W				P	W			M
Micronema hexapterus									W			M
Micronema micronema		P										-
Micronema sp.							W					M
Misgurnus anguillicaudatus	P								W			M
Monopterus albus				B				P	B	P		NM
Monotrete barbatus				E								E
Monotrete cambodgiensis								P				-
Morulius chryosphekadion		P	W									M
Mugil cephalus				E								E
Mystacoleucus obtusirostris									W			M
Mystus albolineatus	P	P		W		NM		P	G			NM
Mystus atrifasciatus	P	P		W					G	P		NM
Mystus bocourti	P					NM			G			NM
Mystus filamentus		P										-
Mystus multiradiatus	P	P		W					G	P	M	M
Mystus mysticetus	P	P		W				P	G	P	M	M
Mystus nemurus	P	P										-
Mystus singaringan	P	P		W		NM			G			NM
Mysus wicki		P										-
Mysus wickioides		P										-
Mystus wolffii	P	P		W								M
Mystus sp.			B									NM
Neolissochilus blanci				W					W			M
Nemacheilus pallidus					P					P		-
Nemapteryx nenga									W			M
Netuma thalassinus				Mr								E
Notopterus notopterus	P	P	W	W		NM		P	G	P	NM	-
Ompok bimaculatus	P	P		W				P	G		NM	NM
Ompok eugeneiatus		P								P		-
Ompok hypophthalmus		P		W					G			-

Annex 1 Cont'd

Species	Baran *et al.* (2014)	Lim *et al.* (1999)	Lamberts (2001)	Chan *et al.* (2008)	Hartmann *et al.* (2013)	Kong *et al.* (2017)	Chan *et al.* (2017)	Marsden *et al.* (2018)	Ngor *et al.* (2018a)	Ainsley *et al.* (2018)	Pool *et al.* (2019)	Consensus
Ompok siluroides										P		-
Ompok urbaini											NM	NM
Onychostoma fusiforme									W			M
Onychostoma gerlachi									W			M
Ophisternon bengalense				B				P				NM
Opsarius koratensis	P	P										-
Opsarius pulchellus	P	P										-
Oreochromis niloticus	P			B								NM
Osphronemus exodon	P			B					B			NM
Osphronemus goramy	P			B					B			NM
Osteochilus hasselti		P		W				P			M	M
Osteochilus lineata											NM	NM
Osteochilus lini	P			W				P	W	P		M
Osteochilus melanopleura	P		W	W				P	W			M
Osteochilus microcephalus	1			W				P	W			M
Osteochilus schlegeli	1	P		W					G			M
Osteochilus vittatus	P					NM			W	P	M	M
Osteochilus waandersii	3			W				P	W			M
Osteogeneiosus militaris				Mr					E			E
Oxyeleotris marmorata	P	P		W				P	W	P		M
Oxygaster anomalura										P		-
Oxygaster pointoni										P		-
Pangasianodon gigas	2			W								M
Pangasianodon hypophthalmus	2	P		W			W		W			M
Pangasius bocourti	3			W					W			M
Pangasius conchophilus	2			W					W			M
Pangasius djambal	1			W					W			M
Pangasius krempfi	3			W					W			M
Pangasius larnaudiei	2	P		W			W		W		M	M
Pangasius macronema	2			W					W		M	M
Pangasius mekongensis	1			W								M
Pangasius micronemus				W								M
Pangasius pleurotaenia				W				P				M
Pangasius polyuranodon	3			W					W			M
Pangasius san-itwongsei				W								M
Pangasius siamensis		P		W								M

Annex 1 Cont'd

Species	Baran *et al.* (2014)	Lim *et al.* (1999)	Lamberts (2001)	Chan *et al.* (2008)	Hartmann *et al.* (2013)	Kong *et al.* (2017)	Chan *et al.* (2017)	Marsden *et al.* (2018)	Ngor *et al.* (2018a)	Ainsley *et al.* (2018)	Pool *et al.* (2019)	Consensus
Pangasius sp.							W		W			M
Parachela maculicauda	P	P				NM			G	P		NM
Parachela oxygastroides										P	NM	NM
Parachela siamensis	P								G	P		NM
Paralaubuca barroni	P							P				-
Paralaubuca harmandi	1			W								M
Paralaubuca riveroi	1							P	G			M
Paralaubuca typus	2	P		W		M		P	W		NM	M
Parambassis apogonoides	P	P					G		G	P	NM	NM
Parambassis siamensis	P								G		NM	NM
Parambassis wolffi	P	P		W		NM	G		G		NM	NM
Pao cambodgiensis									G			NM
Pao cochinchinensis									G			NM
Pao leiurus									E			E
Periophthalmodon septemradiatus									B			NM
Phalacronotus apogon	1								W			M
Phalacronotus bleekeri	2								W			M
Phalacronotus kryptopterus											NM	NM
Phalacronotus micronemus	P								W			M
Piaractus brachypomus									B			NM
Plotosus canius				E								E
Polynemus borneensis		P										-
Polynemus dubius	P								E			E
Polynemus longipectoralis		P		E								E
Polynemus melanochir									E			E
Polynemus multifilis							E		E			E
Poropuntius deauratus	P			W					W			M
Poropuntius normani	P							P				-
Pristolepis fasciata	P	P		W		NM		P	B	P	NM	NM
Probarbus jullieni	2	P		W			W		W			M
Probarbus labeamajor	3			W					W			M
Pseudolais pleurotaenia	2								W			M
Pseudomystus siamensis	P			W				P	W			M
Puntioplites bulu	2	P							W			M
Puntioplites falcifer	3			W		NM			W			M
Puntioplites proctozysron	P	P		W				P	W			M

Annex 1 Cont'd

Species	**Baran *et al.* (2014)**	**Lim *et al.* (1999)**	**Lamberts (2001)**	**Chan *et al.* (2008)**	**Hartmann *et al.* (2013)**	**Kong *et al.* (2017)**	**Chan *et al.* (2017)**	**Marsden *et al.* (2018)**	**Ngor *et al.* (2018a)**	**Ainsley *et al.* (2018)**	**Pool *et al.* (2019)**	**Consensus**
Puntius brevis	P	P							B	P	NM	NM
Puntius masyai		P										-
Puntius orphoides	P			W				P				M
Puntius rhombeus	P			W	P							M
Puntigrus partipentazona										P		-
Raiamas guttatus	1			W					W			M
Rasbora atridorsalis	P							P				-
Rasbora aurotaenia	1	P								P	NM	NM
Rasbora borapetensis	P								G	P		NM
Rasbora daniconius		P							G			NM
Rasbora dorsinotata				W								M
Rasbora hobelmani	P	P						P				-
Rasbora myersi								P				-
Rasbora pausisquamis		P										-
Rasbora paviana	P	P		W	P			P		P		M
Rasbora tornieri	P	P				NM		P	G			NM
Rasbora trilineata	P								G	P		NM
Rasbora sp.				W								M
Rasbosoma spilocerca									G	P		NM
Scaphognathops bandanensis	1			W					W			M
Scaphognathops stejnegeri				W								M
Scatophagus argus				W								M
Schistura aramis									W			M
Schistura athos									W			M
Schistura crabro									W			M
Schistura daubentoni	P								W			M
Schistura latifasciata									W			M
Scleropages formosus									B			NM
Setipinna melanchir		P					E					E
Syncrossus beauforti	3								W			M
Syncrossus helodes	2								W			M
Systomus orphoides		P								P		-
Systomus rubripinnis							G		W			M
Tenualosa thibau-deaui	3	P		W			E		W			M
Tenualosa toli	3	P		W					E			E
Tetraodon sp.											NM	NM

Annex 1 Cont'd

Species	Baran *et al.* (2014)	Lim *et al.* (1999)	Lamberts (2001)	Chan *et al.* (2008)	Hartmann *et al.* (2013)	Kong *et al.* (2017)	Chan *et al.* (2017)	Marsden *et al.* (2018)	Ngor *et al.* (2018a)	Ainsley *et al.* (2018)	Pool *et al.* (2019)	Consensus
Thynnichthys thynnoides	2	P	O	W		M		P	G	P	M	M
Tor laterivittatus	P			W					W			M
Tor sinensis	2			W					W			M
Tor tambroides	2			W					W			M
Toxotes chatareus	P	P					E					-
Toxotes microlepis	P	P		W					E			M
Trichogaster microlepis	P	P		B								NM
Trichogaster pectoralis	P	P		B								NM
Trichogaster trichopterus	P			B								NM
Trichogaster sp.							B					NM
Trichopodus microlepis						NM	B		B	P	NM	NM
Trichopodus pectoralis							B		B			NM
Trichopodus trichopterus									B	P	NM	NM
Trichopsis pumila										P		-
Trichopsis vittata	P				P					P		-
Trichopterus microlepis											NM	NM
Wallago attu	P	P	W	W			W	P	W			M
Wallago leerii	3	P		W								M
Xenentodon cancila	P	P		W		M			W			M
Xenentodon canciloides	P	P										-
Xenentodon sp.										P	NM	NM
Yasuhikotakia caudipunctata	P								W			M
Yasuhikotakia lecontei	P								W			M
Yasuhikotakia modesta	2								W			M

Annex 2 Governance & stakeholders survey, Tonle Sap freshwater health index

The freshwater health index is an analytical tool developed by Conservation International and partners to promote freshwater security and the sustainable management of freshwater ecosystems. It provides a comprehensive assessment of freshwater ecosystems along three dimensions—ecosystem vitality, ecosystem services, and governance—with a goal of linking science, policy, and practice.

This survey is designed to gather information for the governance assessment and aims to understand the views of different stakeholders from the Tonle Sap River basin on the coordinating mechanisms, participatory processes, governance effectiveness, and long-term planning within the region. Your response to the survey and all questions is voluntary. Your valuable advice will provide a basis for a comprehensive assessment of the

Annex 2 Cont'd

state of the current governance system within the Tonle Sap basin. Your answers will remain anonymous, but we ask for your opinions (not the views of your institution) as well as basic identifiers of your country and affiliation. The information collected will only be used for research purposes, and personal data will be kept confidential. Thank you for your cooperation and help.

Current affiliation

Government; NGO; Research/academia; Industry; Other

Unless otherwise stated all questions are assessed according to the following criteria:

Based on your own knowledge of the current situation, please evaluate the degree to which the following functions are being fulfilled throughout the basin. Provide a rating between 1 and 5 following the criteria below. Please skip any items which you do not feel qualified to answer.

Rating	Criteria
1	Strongly disagree with statement
2	Disagree with statement
3	Neutral
4	Agree with statement
5	Strongly agree with statement

Water resource management (1 of 12)

Integrated water resources management is a guiding framework for coordinating both development and management of all resources within a basin, to maximize welfare without compromising ecological sustainability. In some cases a single agency, such as a river basin authority, is responsible for coordinating and overseeing these functions; the questions below focus on the specific functions as managed within your jurisdiction (e.g. transnational, national or provincial) regardless of whether they are all carried out by the same agency.

A) Implementation of existing water resource development and management policies are well coordinated. For example: if there is catchment organization or commission, how effective is it in coordinating the different agencies, levels of government (e.g., national, provincial, local), and private interests when establishing integrated development plans for the catchment?

B) Infrastructure such as dams, reservoirs, and treatment plants are centrally managed or coordinated. For example: dam operators communicating the timing and volume of reservoir releases, or assessing cumulative impacts of dams.

C) There is (adequate) financial contribution towards water resources management. For example: cost-sharing for common projects, or collecting user fees/taxes.

D) Ecosystems conservation priorities are developed and actions implemented. For example: protecting forested watersheds, maintaining wetland/river connectivity, or developing an aquatic species biodiversity action plan.

E) Dispute resolution mechanisms are used to settle potential conflicts between districts or stakeholders within the catchment. For example: negotiations mediated by the provincial government to reach consensus.

Rights to resource use (2 of 12)

Clear and enforceable rules are recognized as a requirement for the efficient use of scarce resources, and as a means of resolving conflicts. These rules encompass various uses and users of water, and can be both formal (i.e., legislated by a government body) or informal rules administered by communities.

A) Rules for allocating water among different sectors (e.g., municipal, industrial, agricultural) are clear and transparent. For example: prioritizing water according to use, or limits on the timing and amount of water that can be withdrawn.

B) Rules for allocating water among administrative boundaries (e.g., cities, provinces, countries) are clear and transparent. For example: prioritizing water according to use, or limits on the timing and amount of water that can be withdrawn.

C) Rules for groundwater abstraction are clear and transparent. For example: guidelines regarding the depth of wells, or amount of water that can be withdrawn within a certain time period.

D) Rules for wastewater handling and water pollution are clear and transparent. For example: guidelines regarding the discharge of wastewater (e.g. pollutant concentrations, volume, temperature, time of release) into water bodies.

E) Rules for managing land use (including aquaculture) to safeguard water resources are clear and transparent. For example: guidelines regarding soil management practices, the amount of forested land in watersheds, or the volume of runoff allowed for a given plot of land.

F) Rules for freshwater fisheries are clear and transparent. For example: guidelines on catch limits, protected species, or fishing methods.

Annex 2 Cont'd

Incentives and regulations (3 of 12)

Various management tools, from conventional regulations to market-based instruments can be applied within a governance system. Having a variety of tools offers opportunities to increase the efficiency of interventions (e.g., cost per unit outcome) or lead to a more equitable distribution of benefits.

A) Environmental and social impact assessments for all major water projects, regardless of funding source, are carried out prior to decisions being taken. For example: environmental impact assessment (EIA) that is submitted to a government body for evaluation.

B) There are financial incentives for environmental stewardship. For example: mechanisms for providing payments for watershed services provided by upstream stakeholders (e.g., farmers, forest managers, local governments).

C) There are market-based exchange schemes. For example: tradeable water rights, wetland mitigation banking, pollutant trading, inter-basin transfer schemes or REDD+ initiatives.

D) There are honorary recognition programs in water resources management. For example: publishing lists of industries with good environmental performance, or awards for local governments practicing good water stewardship.

E) There is a land use zoning policy that is designed to support water management. For example: requirements for riparian buffers, floodplain development, or forested catchment zones.

Technical capacity (4 of 12)

Lack of local capacity is often cited as an impediment to a variety of issues in resource management. Here we are referring to people employed in areas of water resource management, service delivery, monitoring and enforcement, and related research, but excluding international consultants.

A) There is an adequate number of staff (including local consultants) to fulfil necessary functions. For example: backlogs (work waiting to be done) in a particular agency, or open positions remaining vacant due to lack of candidates.

B) Staff have sufficient expertise to fulfil necessary functions. For example: hydrologists to evaluate a proposed dam, or fisheries ecologists to assess fish stocks.

C) There are opportunities for professional training and certification on water resources management. For example: financial support or time allocated for continuing education courses related to improving technical skills.

Financial capacity (5 of 12)

Water resource development and management is often under-financed, particularly for services that do not generate revenue, such as ecosystem protection. Although financial capacity can be measured directly as a function of existing allocations relative to estimated budget needs, qualitative information is also useful in providing insights and identifying priorities.

A) There is sufficient investment in water supply development. For example: financial resources for building and maintaining reservoirs or irrigation systems.

B) There is sufficient investment in service delivery systems. For example: financial resources for building and maintaining water distribution networks (i.e. piped supply) or household wells.

C) There is sufficient investment in wastewater handling and treatment. For example: financial resources for building and maintaining community toilets, or treatment systems to process waste water.

D) There is sufficient investment in ecosystem conservation and rehabilitation. For example: financial resources for protecting wetlands to mitigate flood risk, remediating impaired streams, or rehabilitating fish stocks.

E) There is sufficient investment in monitoring and enforcement. For example: financial resources for evaluating EIAs, collecting environmental data, inspecting facilities, and enforcing regulations.

Information access and knowledge (6 of 12)

Sound water governance requires information on a range of topics and from many sources. Even in cases where data and information are abundant, if they are not made accessible (across agencies, with citizens, etc.) then they are less likely to aid in wise decision making.

A) Information is accessible to interested stakeholders. For example: reports made freely available through a website, or data available upon request to the agency with the information.

B) Information meets expected quality standards, in terms of frequency, level of detail, and subjects of interest to stakeholders. For example: time series data on streamflow, water levels, or water quality for specific locations within the basin.

Annex 2 Cont'd

C) Information is transparently sourced. For example: methods used to collect data are documented, or authors (source) of these data are clearly identified.

D) All available, sound and relevant information is routinely applied in decision-making. For example: modifying an infrastructure project based on EIA results, or adjusting fisheries management guidelines based on fish catch data.

Engagement in decision-making processes (7 of 12)

Stakeholder engagement encompasses the process by which any person or group with an interest in a water-related topic can be involved in decision-making and implementation. It is associated with improved information transfer, better targeted and more equitable plans and policies, improved transparency and accountability, and reduced conflict.

A) All relevant stakeholders have been identified and notified when considering major decisions. For example: mapping and notifying stakeholders affected by a proposed water supply infrastructure project (e.g. construction of a water supply dam).

B) Stakeholders (men and women) are able to provide comments prior to major decisions being taken. For example: consultation meetings or an information gathering period where stakeholders may provide input regarding a policy or project.

C) Representatives from catchment district and other actors meet regularly to exchange information and, when appropriate, take decisions. For example: steering committee or other political meetings convened by the Cambodian government, workshops convened by a provincial agency or NGO.

D) Decisions are made based on stakeholders' participation. For example: processes for reaching joint agreements among a group of stakeholders prior to approval of a major policy or project, or projects being revised subsequent to stakeholder feedback.

Enforcement and compliance (8 of 12)

In many societies, there is a gap between laws and their actual enforcement, reflecting either insufficient capacity or a lack of accountability. Enforcement and compliance can be ensured through fines, incentives, or social pressure, but weak enforcement leads to poor management and a lack of confidence in the system.

A) Surface water abstraction guidelines are enforced. For example: industries restricted from withdrawing more than a specified amount of surface water, or farmers sanctioned for withdrawals during the dry season.

B) Groundwater abstraction guidelines are enforced. For example: farmers or industries restricted from pumping more than a specified amount of groundwater.

C) Flow requirement guidelines are enforced. For example: dam operators meeting the expectations of downstream water users, to meet environmental flows, human water needs, and/or flood protection.

D) Water quality guidelines are enforced. For example: industries and communities complying with requirements related to pollutant discharges, or non-negotiable fines are levied on violators.

E) Land use guidelines are enforced. For example: environmentally sensitive zones (e.g., catchment forests and wetlands) being protected from development or degradation.

Distribution of benefits from ecosystem services (9 of 12)

Equity is an important issue in water resource management, most closely associated with access to safe water and sanitation. Here we extend the concept to include all benefits from ecosystem services in the basin (water and sanitation, fisheries, flood mitigation, water quality maintenance, disease regulation, and cultural services).

A) Low income (rural) communities benefit from ecosystem services. For example: poor households' access to improved water supply sources at a reasonable cost, protection from inland flood risks, or rural compared to urban populations' benefits.

B) Local communities benefit from ecosystem services. For example: exercising customary rights related to water, including for consumptive as well as cultural uses, or maintaining traditional fisheries.

C) Women and girls benefit from ecosystem services. For example: amount of time collecting water for households, or provision of toilets for females.

D) Resource-dependent communities benefit from ecosystem services. For example: fishermen and smallholder farmers' incomes compared to other economic sectors.

E) All districts and stakeholders share in the benefits from ecosystem services. For example: water for irrigation, water for industry, and tourism.

Annex 2 Cont'd

Water-related conflict (10 of 12)

Tensions among stakeholders are expected when there is competition for scarce resources such as water. An effective governance system should prevent tensions from escalating into conflicts, here defined as a difference that prevents agreement, and therefore delays or undermines a decision taken with the basin.

A) There are frequent conflicts due to overlapping decision making powers (e.g., between national governments in transboundary systems, provincial and national government, or between agencies). For example: disputes between the local environmental bureau and a national ministry about authority within a floodplain, or between agencies in managing agricultural pollution.

B) There are frequent conflicts about water rights allocation. For example: disputes about how water is allocated between two municipalities, or between agricultural and industrial users.

C) There are frequent conflicts about access to water resources. For example: disputes about having access to safe water and sanitation, or the costs of such access.

D) There are frequent conflicts regarding the placing of infrastructure. For example: disputes about reservoir development and resettlement plans for residents and land owners, or downstream impacts to fisheries or water users.

E) There are frequent conflicts over water quality and other negative downstream impacts. For example: disputes between upstream and downstream stakeholders about dry season flows or pollution concentrations.

Monitoring and learning mechanisms (11 of 12)

Policy and planning decisions about water resources management are ideally based on sound data and information, which must be collected on a regular basis. Monitoring entails costs and so data collection should be based on needs and assessed relative to resource constraints, where a comparatively wealthy basin might invest in higher spatial and temporal coverage of information.

Provide a rating between 1 and 5 following the criteria below.

Rating	Criteria
1	Data are very poorly monitored, or not monitored at all
2	Data are poorly monitored
3	Data are acceptably monitored
4	Data are well monitored
5	Data are very well monitored

A) Overall standard of water quantity monitoring. For example: streamflow being regularly measured, estimated, or modeled in the basin.

B) Overall standard of water quality monitoring. For example: water quality samples taken from water bodies and measured, or water quality being modeled based on data related to discharge of pollutants.

C) Overall standard of biological and ecological monitoring. For example: surveillance undertaken to assess aquatic species (e.g., harvested, threatened, invasive) populations or communities (e.g. macroinvertebrates).

D) Overall standard of monitoring access to, and use of, water. For example: household surveys administered to estimate the coverage of access to improved water and sanitation sources, or estimates of farmers' groundwater extraction.

Strategic planning and adaptive governance (12 of 12)

Comprehensive planning is the process of developing goals and objectives concerning water quantity and quality, surface and groundwater use, land use change, river basin ecology, and multiple stakeholders' needs. Adaptive management refers to the ability to handle changes, unintended consequences, or surprises to the water resource system through updating planning and processes using new information

A) A shared vision is established and used to set objectives and guide future development. For example: goals for improvement are jointly established by multiple stakeholders, or a process is in place for developing local water plans that inform higher-level (provincial or national) plans.

B) There are strategic planning mechanisms. For example: basin-specific spatial plans or management plans that guide investments and policy, or climate change adaptation plans.

C) There is an adaptive management framework that is effectively applied. For example: updating plans to reflect new knowledge or changing economic development priorities, or to address issues such as climate change.

Spatially heterogeneous natural regeneration of tall evergreen dipterocarps, a target of selective logging

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មូលន័យសង្ខេប

ការកាប់បំផ្លាញ និងការសឹករិចរិលព្រៃឈើនៅប្រទេសកម្ពុជាកំពុងត្រូវបានដោះស្រាយ ប៉ុន្តែនៅតែជាបញ្ហាប្រឈម។ ព័ត៌មានលំអិតអំពីដើមឈើមិនទាន់ពេញវ័យនៃប្រភេទដែលបានកាប់ នឹងមានសារៈប្រយោជន៍ចំពោះការលើកកម្ពស់ការដំណុះឡើងវិញនូវប្រភេទរុក្ខជាតិ និងការស្តារព្រៃឈើឡើងវិញ។ យើងបានផ្តោតលើពពួកឈើទាលមានកម្ពស់ខ្ពស់ចំនួនពីរប្រភេទគឺ *Anisoptera costata* (ផ្ដៀក) និង *Dipterocarpus costatus* (ឈើទាល បង្គួយ) ដោយបានធ្វើជំរឿនដើម្បីកំណត់របាយទំហំរបស់ពួកវានៅភាគកណ្តាលនៃប្រទេសកម្ពុជា។ សមមាត្រទាប (ប្រភេទ *A. costata* ៣៩.៣% និងប្រភេទ *D. costatus* ៤៣.១%) នៃដើមឈើមានអង្កត់ផ្ចិតតូចបំផុត (៥ - ១០ សង់ទីម៉ែត្រ) បានបង្ហាញពីដំណុះឡើងវិញមិនគ្រប់គ្រាន់នៃប្រភេទដើមឈើទាំងពីរ ទោះបីជាគេឃើញវត្តមានរបស់ដើមឈើមេក៏ដោយ។ លក្ខខណ្ឌសំណើមនៅទីតាំងសិក្សារបស់យើង ត្រង់កន្លែងមានដើមឈើមិនទាន់ពេញវ័យគឺដុះច្រើនលើសលុប បញ្ជាក់ថាដើមឈើតូចៗលូតលាស់លើដីសើមច្រើនជាងដើមឈើពេញវ័យ។ លទ្ធផលនៃការសិក្សារបស់យើងបង្ហាញពីភាពចាំបាច់នៃការកំណត់អត្តសញ្ញាណព្រៃឈើ រួមទាំងទីតាំងសមរម្យសម្រាប់ជ្រើសរើសដើមឈើដើម្បីប្រឈមនឹងបម្រែបម្រួលអាកាសធាតុ។

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Abstract

Deforestation and forest degradation in Cambodia are being addressed, but remain challenging. Detailed ground-based insights on juvenile trees of logged species will be useful for efforts to promote species regeneration and forest restoration. We focused on two species of tall dipterocarps, *Anisoptera costata* (*phdiek*) and *Dipterocarpus costatus* (*chhoeuteal bankouy*), undertook a tree census to determine their size distributions in central Cambodia. The small proportion (39.3% & 43.1% for *A. costata* & *D. costatus*, respectively) of individuals in the smallest diameter class (5–10 cm) suggested inadequate regeneration of both species, despite the presence of mother trees. The moist site conditions of our study site, where juvenile trees are exceptionally abundant, imply that small trees thrive on wetter soils than mature individuals. Our findings demonstrate the need to identify forests including suitable sites for tree recruitment in the face of continuing climate change.

Keywords *Anisoptera costata*, conservation, *Dipterocarpus costatus*, forest regeneration, seedling, topography.

Introduction

Despite efforts to limit deforestation and forest degradation in Cambodia (MoE *et al*., 2020), large-diameter trees of Dipterocarpaceae species such as *Dipterocarpus costatus* C.F.Gaertn. (*Chhoeuteal Bankouy* in Khmer) and *Anisoptera costata* Korth. (*Phdiek*) have been extensively felled in lowland dry evergreen forests in Kampong Thom Province (Ito *et al*., 2010). Various initiatives have been considered to strengthen forest conservation in Cambodia, including proposals for systematic conservation plans (Strange *et al*., 2007), accumulated findings based on seed supply demand for plantations (Norn & Sobon, 2014) and identifying framework tree species (Sobon *et al*., 2017). Although forest structure information for each forest-type has been obtained from ground-based studies in-country (Kimphat *et al*., 2000, 2002; Kao & Iida, 2006; Ouk, 2006; Pin *et al*., 2013; Toyama *et al*., 2015; Chheng *et al*., 2016; Ito *et al*., 2017, 2022), understanding of forest dynamics in Cambodia is far from complete. To the best of our knowledge, there is no information on whether or not lowland dry evergreen forests undergoing exhaustive selective logging are stocked with sufficient numbers of young trees to allow them to recover from forest degradation. In other words, it has yet to be determined whether forest resources can recover from the anthropogenic disturbance of exhaustive selective logging without human assistance. In this study, we analysed the results of a tree census undertaken before selective logging at the site was intensified to determine the size distribution of two dipterocarp species in the study area.

Methods

Our study was conducted in lowland dry evergreen forests in Kampong Thom Province in central Cambodia. The forests typically develop on gently undulating, sandy alluvial plains with deep soils (Ito *et al*., 2021). The study site features a sandy soil, Haplic Acrisol (Alumic, Profondic) within the World Reference Base system (Toriyama *et al*., 2007, 2008). Mean annual precipitation in the study area is 1,625.8 mm (Kabeya *et al*., 2021). The monthly average temperature ranges from 24°C to 29°C, with a mean of 27°C (Chann *et al*., 2011). The seasonal tropical climate is governed by monsoons and described in detail elsewhere (Ito *et al*., 2021; Kabeya *et al*., 2021).

We used data from past tree censuses undertaken in 15 plots established within a rectangular area measuring 6 km east-west and 13 km north-south (centered on 12.72° N, 105.47° E). The location of the plots is shown in a digital surface model in Fig. 1 and almost all plots were located on the gentle hill tops of undulating terrain within the catchment area of the Chinit River. The plots measured 30 × 40 m (n=13), 30 × 80 m (n=1, plot no. 05) and 200 × 200 m (n=1, plot no. 09) (Table 1). Plot no. 05 was the most well-preserved forest plot in the area and was first surveyed in 2003.

We employ data from plot measurements undertaken in 2011. To estimate the reduction in biomass accumulation associated with forest degradation, 13 plots (30 × 40 m) were established to include forests with a similar species composition to plot no. 05, but with different degrees of human-induced degradation. Plot no. 09 was established as a reference forest plot for a meteorological observation tower. In accordance with observation protocols for forest hydrology, the tower was located in a transitional section between lower and higher ground in the centre of the plot. As our objective was to investigate relationships between hydrological data and forest structure, the plot spanned 100 m in all cardinal directions from the tower (i.e., 200 m × 200 m in total), which is generally considered to be the range of influence for hydrological data. In each plot, we divided the plot area

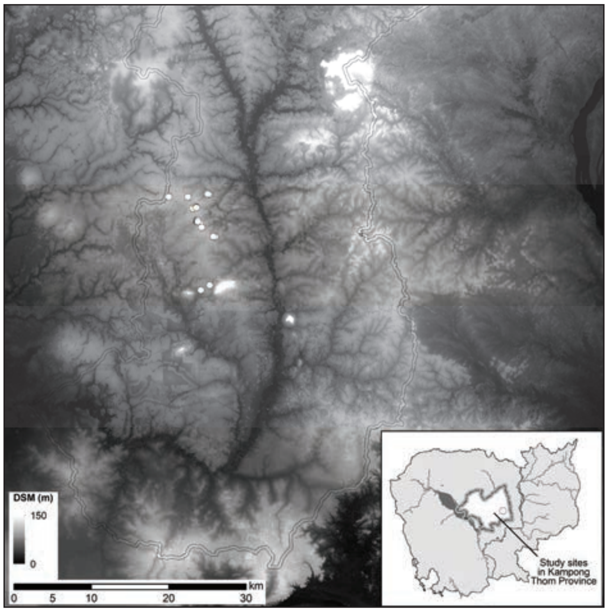

Fig. 1 Digital surface model of the study sites (white circles) in Kampong Thom Province, Cambodia. The background gradient is the digital surface model provided by JAXA (ALOS World 3D–30m). Double lines indicate the catchment area of the Chinit River.

into sub-quadrants measuring 10 × 10 m and the girth of all standing woody stems with a diameter at breast height (DBH, defined as 1.3 m above ground) ≥ 5 cm was measured to the nearest 1 mm. For trees with buttress roots at a height of 1.3 m, the measurement was made just above their protrusions. Species were categorized as *D. costatus*, *A. costata*, or others.

As the plots were located in lowland dry evergreen forests, they were dominated by evergreen tree species, with a predominance of *Vatica odorata* (Griff.) Symington (Dipterocarpaceae) (*Chromas*), *Diospyros venosa* Wall. ex A.DC. (Ebenaceae) (*Angkot Khmao*), *D. undulata* Well. ex. G. Don. var. *cratericalyx* (Craib) Bakh. (Ebenaceae) (*Chhoeu Phleung*), *Melodorum fruticosum* Lour. (Annonaceae) (Romdoul), *Sindora siamensis* Teysm. ex Miq. (Fabaceae) (*KoKoh*), *Syzygium* spp. and *Memecylon* spp. Plot no. 09 was located in a transitional zone between lower and higher ground, the former being a swamp dominated by *Myristica iners* Blume (Myristicaceae), which has aerial roots and is thus able to grow in waterlogged sites (Theilade *et al*., 2011). Stand structure parameters derived from the tree census are provided in Table 1.

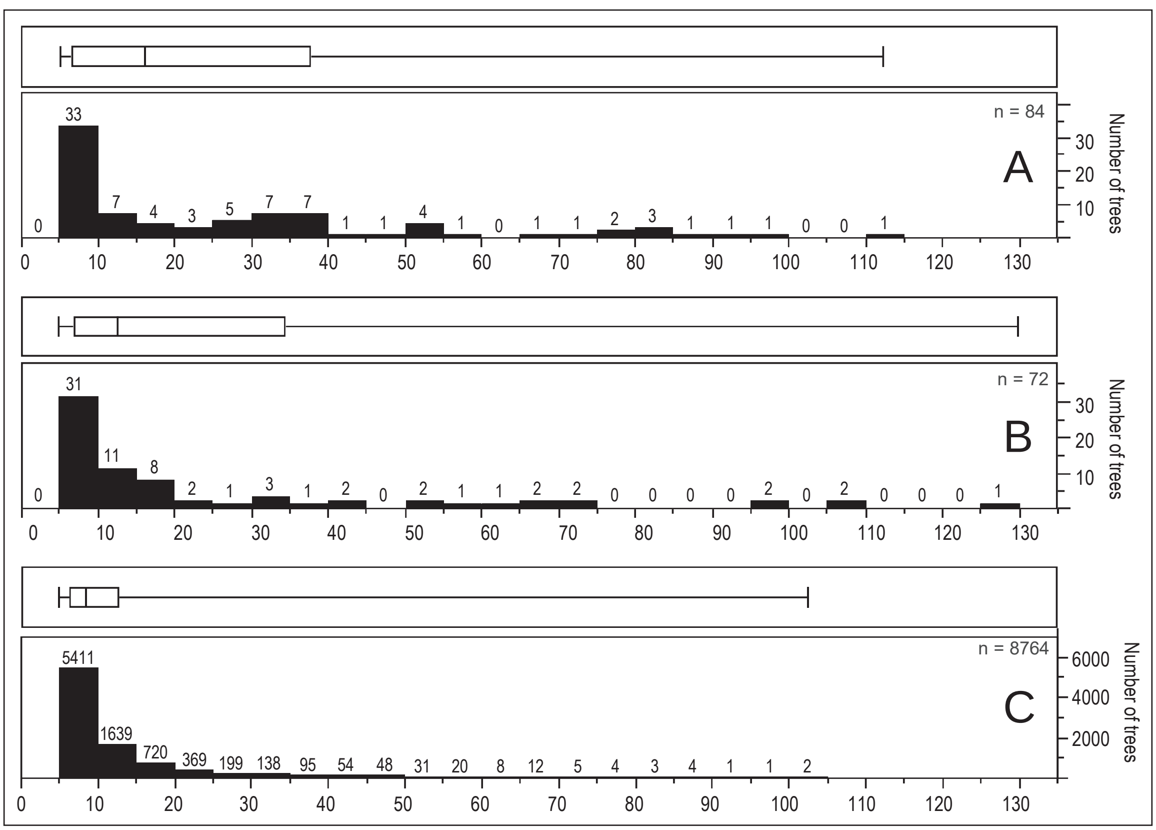

Fig. 2 Size (DBH, cm) distribution of A) *Anisoptera costata*, B) *Dipterocarpus costatus*, C) other species.

Table 1 Stand structure parameters for plots in study area. Values for basal area, stem density and maximum stem diameter are given as average (± SD) (range) for all species in each 10 × 10 m sub-quadrant within each plot.

No.	Location[1]	Alt (m)[2]	Size (m)	Census Year	Basal Area (m^2 /100 m^{-2})	Stem Density (stems /100 m^{-2})	Maximum Stem Diameter (cm)
1	548292, 1412088	91–93 (102–106)	30×40	2010	0..26±0.10 (0.10–0.48)	19.5±5.9 (8–29)	27.1±9.2 (17.4–49.1)
2	550755, 1412086	100 (101–105)	30×40	2010	0.31±0.28 (0.08–1.07)	12.1±2.3 (8–17)	39.2±24.7 (14.6–109.1)
3	553334, 1412455	81 (95–97)	30×40	2010	0.29±0.15 (0.13–0.61)	16.4±4.7 (8–24)	32.2±16.7 (15.4–77.6)
4	553506, 1412400	81 (95–97)	30×40	2010	0.29±0.06 (0.18–0.39)	18.8±3.9 (13–25)	32.9±6.7 (23.1–45.4)
5	551425, 1410577	101 (113–121)	30×80	2011	0.42±0.35 (0.11–1.48)	18.2±5.1 (8–26)	44.2±32.3 (15.7–129.8)
6	551791, 1410814	102 (115–120)	30×40	2010	0.38±0.32 (0.07–1.26)	13.6±3.1 (9–19)	42.0±23.1 (13.2–99.9)
7	551868, 1410718	101 (107–116)	30×40	2010	0.29±0.24 (0.03–0.91)	13.5±6.2 (4–22)	37.1±21.6 (10.6–87.9)
8	551871, 1410671	101 (111–116)	30×40	2010	0.34±0.20 (0.09–0.73)	17.8±4.9 (10–25)	37.4±16.5 (17.9–71.5)
9	551851, 1408841	85–90 (89–105)	200×200	2015 (2017, 2020)[3]	0.28±0.21 (0.01–1.08)	14.8±5.5 (3–33)	36.0±19.0 (7.5–112.3)
10	552294, 1408635	91 (105–108)	30×40	2010	0.39±0.27 (0.11–0.91)	15.0±3.4 (10–22)	43.1±18.3 (23.5–80.8)
11	552537, 1408086	91 (107–111)	30×40	2010	0.32±0.18 (0.09–0.58)	17.2±4.2 (10–22)	40.6±20.9 (17.2–79.9)
12	554068, 1406888	86 (97–101)	30×40	2010	0.24±0.17 (0.07–0.61)	13.0±2.9 (9–18)	32.0±15.7 (14.1–59.1)
13	554288, 1406844	84–85 (87–94)	30×40	2010	0.27±0.13 (0.08–0.50)	20.3±5.7 (13–29)	42.0±23.1 (13.2–99.9)
14	553592, 1400648	74 (88–94)	30×40	2010	0.23±0.10 (0.08–0.40)	12.2±3.9 (8–19)	34.6±8.3 (18.8–50.4)
15	552453, 1400014	70–71 (77–79)	30×40	2010	0.15±0.07 (0.05–0.27)	25.6±8.8 (11–36)	16.1±6.2 (9.8–33.4)

[1] Southwest corner of the plot (WGS 1984, UTM Zone 48N). [2] DTM (DSM). Altitude based on digital terrain model (DTM) with 50 m resolution and digital surface model (DSM) of ALOS World 3D–30m, ver. 3.1 with approximately 30 m resolution (https://www.eorc.jaxa.jp/ALOS/en/dataset/aw3d_e.htm). [3] For plot no. 09, juvenile tree data from tree censuses in 2017 and 2020 are also shown in Fig. 6, but are not included in the size distributions given in Figs 2 & 3.

Results

The size (DBH) distribution of the two dipterocarp species (*A. costata*, *D. costatus*) and other species was examined using the combined data from all plots (Fig. 2). When all tree species were combined, the size distribution was L-shaped, with a significant proportion (61.4% of 8,920 trees) belonging to the smallest diameter class (5–10 cm). A rather small proportion of individuals in the smallest diameter class (5–10 cm) were found for two dipterocarp species, namely 39.3% of 84 trees for *A. costata* and 43.1% of 72 trees for *D. costatus* (Fig. 2a, 2b). Moreover, comparison of plot-by-plot size distributions for the two dipterocarp species revealed that L-shaped size distribution was only found in plot no. 09 (Fig. 3). The total population density of the two dipterocarp species in plot no. 09 was relatively low (9.3–10.0 tree ha^{-1}). Nonetheless, a considerable proportion of the smallest-diameter individuals of

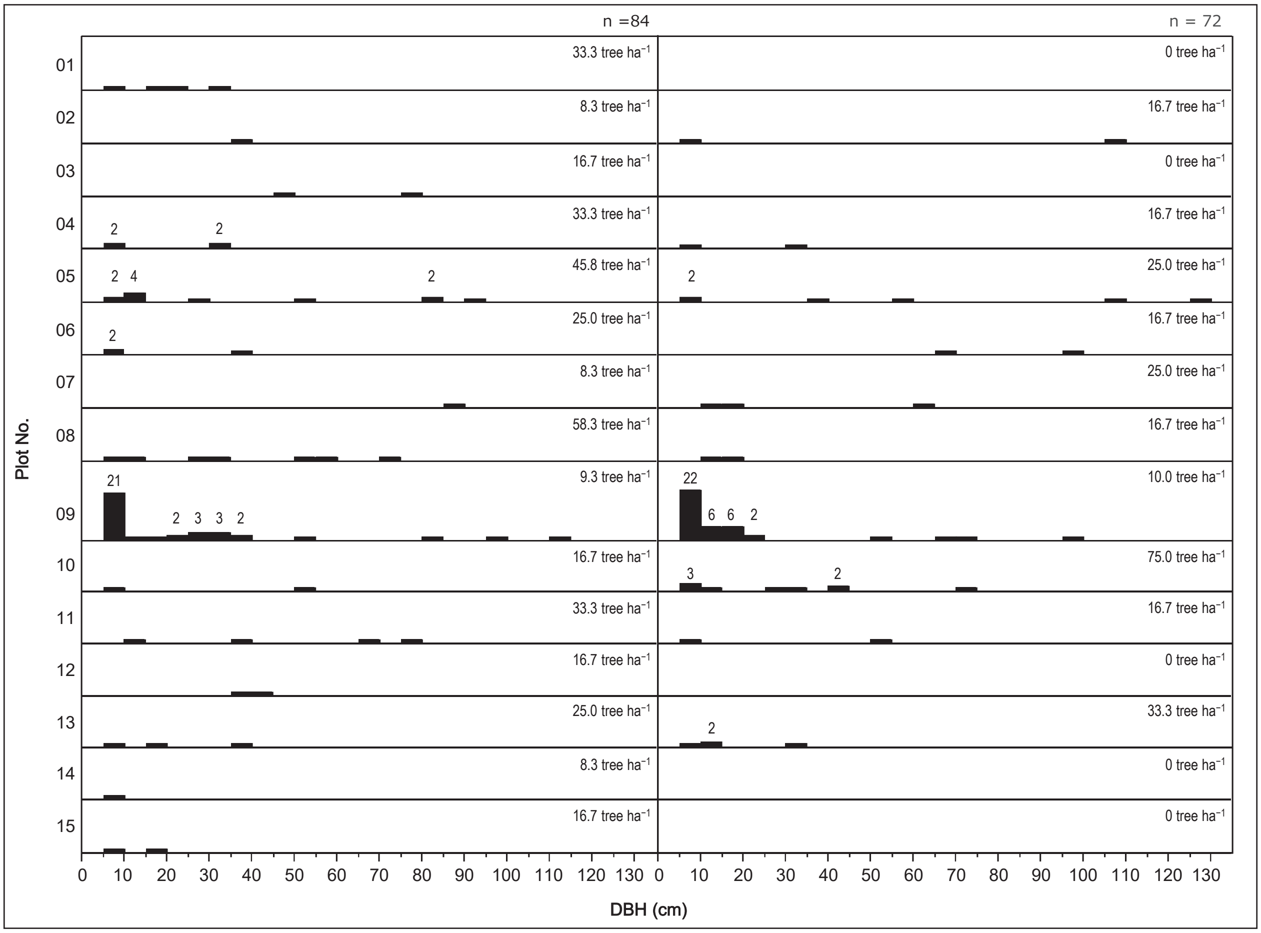

Fig. 3 Size (DBH) distribution of *Anisoptera costata* (left) and *Dipterocarpus costatus* (right) by plot. Figures above the columns indicate the number of individuals. Columns without a number indicate one individual per class and tree densities (trees ha^{-1}) for each plot are included in the figure.

these species occurred in plot no. 09 (56.8% of 37 trees for *A. costata* and 55.0% of 40 trees for *D. costatus*), whereas the proportion in our remaining 14 plots was about half of this (25.5% of 47 trees for *A. costata* and 28.1% of 32 trees for *D. costatus*).

Our survey plots spanned a narrow elevation range, between 70 and 102 m (Table 1). Almost all of the plots were located on gentle hill tops in undulating terrain (Fig. 4a), with the exception of plot no. 09 (tower plot, Fig. 4b). A previous field survey (Ohnuki *et al*., 2022) showed that the ground surface of the plot no. 09 was higher in the northeast and lower in the southwest (Fig. 5). In other words, the plot no. 09 was located in a transitional section between higher ground (hilltops) and lower ground and contained areas of water accumulation. Soil thickness became shallower from northeast to southwest across the topographic conversion area (Fig. 5a). Soil thickness is directly related to the moisture content of the soil surface layer during the dry season (Fig. 5b; Ohnuki *et al*., 2022).

Examination of the spatial distribution of the two tall dipterocarps in plot no. 09 showed that both species were absent from the swampland in the central area where *M. iners*, a large tree, predominated (Fig. 6). The area where the aerial roots were distributed was swampy and had water on the surface even during the dry season (Fig. 6, field observation). The distribution of *M. iners* reflected the distribution of soil moisture conditions determined in this plot (Fig. 5b; Ohnuki *et al*., 2022). Analysis of the spatial distribution of the two dipterocarps species by diameter class showed that large-diameter trees (DBH > 50 cm) occurred in the northeastern portion of the plot and thus at a topographically higher location (Fig. 6). Medium-diameter trees (DBH 20–40 cm) were also present in the northeastern portion of the plot as well as the southwestern portion (for *A. costata*), but were absent

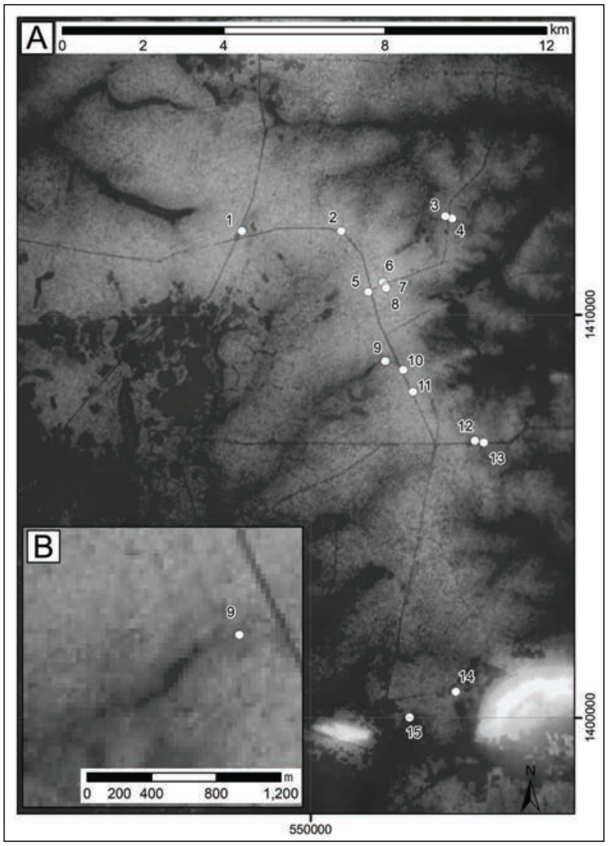

Fig. 4 Microtopography in A) study area, B) plot no. 09. White circles indicate southwest corner of the 15 survey plots. Background gradient is a digital elevation model (50 m resolution) overlapped by the digital surface model provided by JAXA (ALOS World 3D–30m), in which white indicates higher terrain.

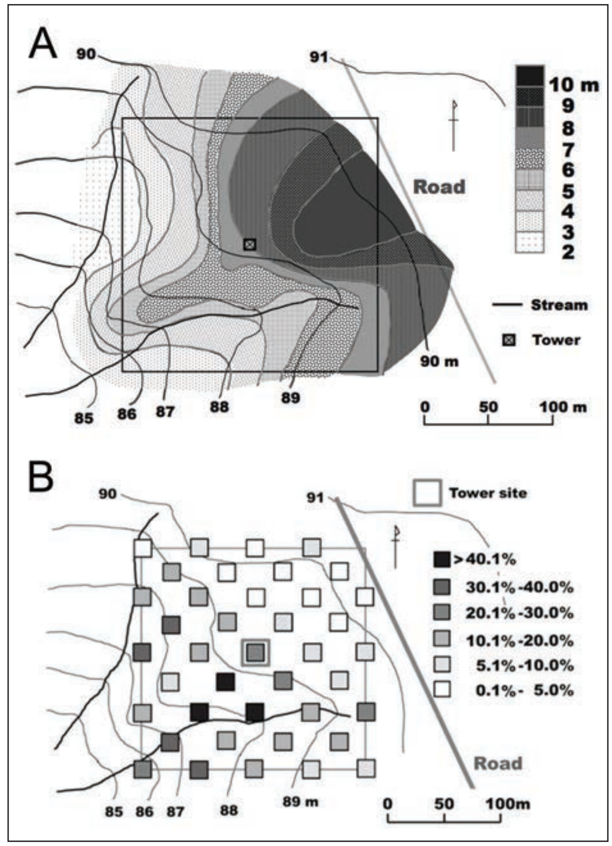

Fig. 5 A) Soil thickness and B) soil water content during the late dry season in plot no. 09. Redrawn from Ohnuki *et al*. (2022).

from the swampy area. Among the small-diameter trees, those with a DBH of 10–20 cm occurred in the marginal area of the swampy forest, but mostly in the northeastern upland area. The smallest individuals (DBH 5–10 cm) grew closer to the swamp than larger individuals. In particular, *A. costata* occurred at higher density in the southwest portion (Fig. 6a).

Discussion

Despite the presence of mother trees, regeneration in the 14 plots other than plot no. 09 seemed inadequate (Fig. 3). Selective logging has led to a sharp decline both in the number of flowering plants and in the population density of the two tall dipterocarp species in the lowland dry evergreen forests of Kampong Thom Province (Ito *et al*., 2023). Reduction in population density has a serious impact on genetic diversity and pollination efficacy (Ghazoul *et al*., 1998; Tani *et al*., 2009, 2012; Duc *et al*., 2016). It is therefore unlikely that useful dipterocarp timber will be available again in the lowland dry evergreen forests. However, in the process from reproduction to the establishment and survival of juvenile and immature trees, the most limiting stage for regeneration remains to be identified. This information is critical for implementation of proactive measures to ensure sustainable forest use.

The spatial distribution of the two dipterocarps species in plot no. 09 differed by diameter class (Fig. 6). This trend differs from the pattern for *Shorea curtisii* (Dipterocarpaceae) seedlings in hill dipterocarp forest in Peninsular Malaysia, where suitable microtopography for seedling establishment was the same as those for mother trees, both being abundant on ridges (Yagihashi *et al*., 2010). Seeds and seedlings most susceptible to pathogen attack and mortality are those closest to the mother tree (Gilbert, 2002), which may explain the discrepancy in the distribution of large vs. small dipterocarp individuals in our plot no. 09. Furthermore, there may have been a change in soil moisture conditions since the time when the current large-diameter trees first established.

A comparison of the site conditions of plot no. 09, which was exceptionally abundant in juvenile trees of the two dipterocarps, with those of the other plots may provide insights for effective forest management. Plot no. 09 was located in the transitional zone between lowland dry evergreen forest (in the northeast) and wet evergreen forest (in the southwest). The density of trees in plot 09 (9.3–10.0 tree ha^{-1}; Fig. 3) was only slightly higher than the threshold density of 9 trees/ha at which the fertility of *S. siamensis* (Dipterocarpaceae) rapidly declines (Ghazoul *et al*., 1998). However, the calculation which provided the relatively low value for tree density in plot no. 09 included the swamp area which was characterized by *M. iners*. The spatial distribution of the two dipterocarps overlapped little with that of *M. iners* and the swamp area is probably unsuitable for dipterocarp growth (Fig. 6). Conversely, small dipterocarps tended to be more abundant than larger individuals in the vicinity of the swamp, where the soil moisture content was high even during the dry season (Fig. 5). These observations suggest that small trees may thrive better than mature individuals in areas with a higher soil moisture content. This hypothesis is consistent with the lower density of small trees in the other 14 plots, which were located in the higher portions of our study area.

We also need to consider the origin of the seed supply for small trees in the southwestern portion of plot no. 09. There are three possible seed sources: large trees in the northeast, small- to medium-sized trees in the southwest and large trees that existed in the southwest until recently. The distance from a large tree (>50 cm DBH) in the northeast to the group of small trees in the southwest was approximately 130–150 m (Fig. 6). While Dipterocarpaceae species are characterized by winged seeds, their dispersal distance is usually less than 30 m (Tamari & Jacalne, 1984; Ghazoul, 2016) because they rely on wind-assisted gyration. Estimation of the seed dispersal distances of other Dipterocarpaceae species using genetic analysis has shown that the majority of these occur within 50 m of the mother tree and that dispersal beyond 100 m is both species-limited and under-represented (Takeuchi *et al*., 2004). Water dispersal could have compensated for the limited distance achieved by wind dispersal. Seeds could have been dispersed by flood waters after water stagnated in swamp areas and then established in the southwest. Water dispersal of Dipterocarpaceae seeds has been observed in *D. oblongifolius* (Ridley, 1905) and the dispersal distance of seeds may be extended via animals; in *A. costata*, there appears to be dispersal by

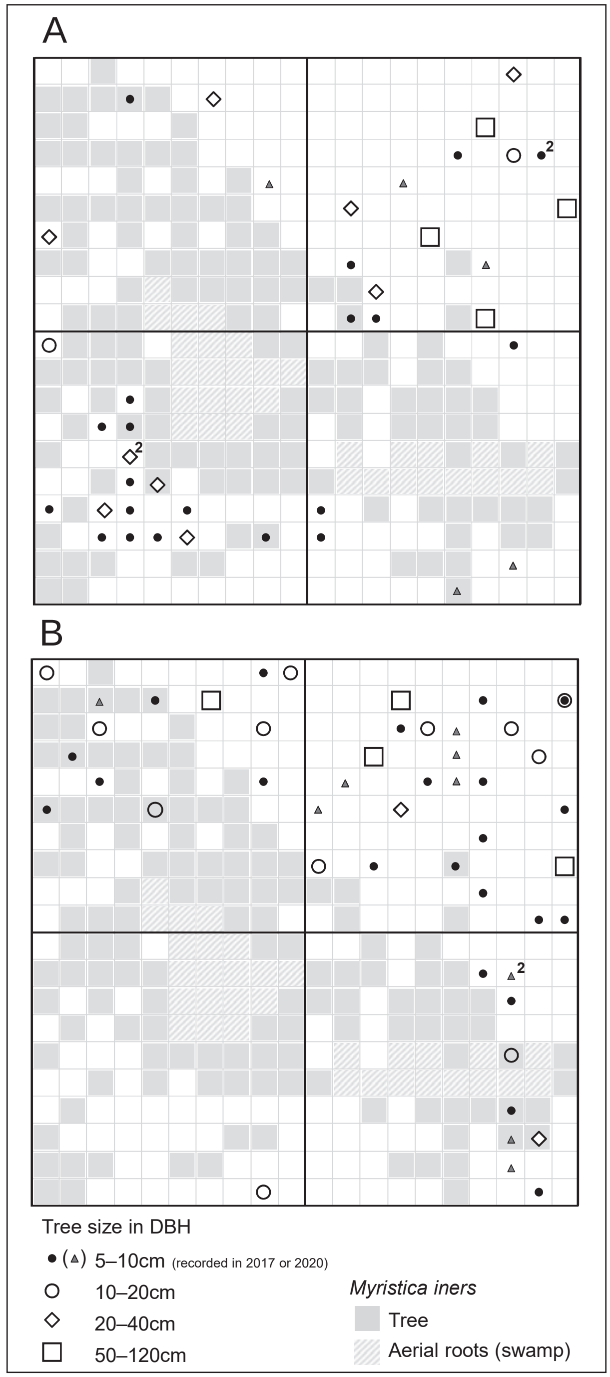

Fig. 6 Spatial distribution of A) *Anisoptera costata* and B) *Dipterocarpus costatus* in plot no. 09. Symbols indicate tree size (DBH) of both species. Data are based on tree census in 2015, although data for smallest individuals (DBH 5–10 cm) recorded during tree censuses in 2017 and 2020 are also shown. All symbols represent one individual, apart from the annotated symbol where two individuals were present. Gray and gray-shaded cells indicate the location of *Myristica iners* and the swampy areas where aerial roots of *M. iners* were present, respectively.

birds, squirrels and bats (Nguyen *et al*., 2021). Regarding the second possibility (i.e., small- to medium-sized trees in the southwest), we have no information on the fruiting initiation size for *A. costata* or *D. costatus,* although DBH values of 20–30 cm accounted for 90% of tree reproduction for other Dipterocarpaceae species in Cambodia (Ito *et al*., 2016). Therefore, it is possible that medium-sized individuals (DBH 20–40 cm) around the young trees in the southwestern part of the plot produced seeds. Leaves and flowers of *A. costata* were collected using litter traps in the northeastern portion of the plot, but only small amounts of leaves were trapped in the southwestern portion (Ito, unpubl. data). In the case of *D. costatus*, flower bud scales and seeds were collected in the northeastern portion of the plot, but only leaf bud scales were trapped in the southwestern portion (Ito, unpubl. data). Although not fully investigated, fruiting has yet to be confirmed in the southwestern population. Regarding the third possibility (i.e., large trees that existed in the southwest until recently), our field surveys did not find any large-diameter stumps or naturally dead trees in the southwestern 1-ha area. However, because dead stumps of large-diameter trees tend to be covered by termite mounds (Ito & Tith, 2023), the possibility that large-diameter trees were once present cannot be completely ruled out. As discussed below, it is necessary to clarify the actual condition of seed dispersal for forest management that encompasses both mother trees and suitable areas for the growth of juvenile trees.

Despite a seasonal tropical climate, whereby little rain falls for half the year (Kabeya *et al*., 2021), plants in lowland dry evergreen forests in Cambodia have access to abundant groundwater (Araki *et al*., 2008; Ohnuki *et al*., 2008b; Toriyama *et al*., 2011) via their deep root systems (Tanaka *et al*., 2004; Ohnuki *et al*., 2008a), which facilitates year-round foliage retention. Dry-season water availability is determined by the vertical distance to the groundwater table. The relative position of trees in undulating terrain is closely related to water availability during the dry season (Ohnuki *et al*., 2022) and being dependent on soil moisture at shallower depths, juvenile trees are more vulnerable to drought than larger trees. Climate-driven changes in drought in Southeast Asia may alter tree recruitment rates (Nguyen *et al*., 2019). Our study suggests a shift in suitable recruitment sites, or at least those related to moisture availability, to areas lower than those which allowed the establishment of presently large trees. While registration of new forest reserves to protect mother trees is clearly important, our findings demonstrate the need to identify forests including suitable sites for tree recruitment in the face of continuing climate change.

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Recent Master's Theses

This section presents the abstracts of research theses produced by Royal University of Phnom Penh graduates recently awarded the degree of Masters of Science in Biodiversity Conservation. The abstracts have been edited for English and brevity.

An investigation of the impacts of air pollution on bird species richness and abundance in urban habitats in Phnom Penh city

CHIM Samhaiy

មូលន័យសង្ខេប

សត្វស្លាបគឺជាធនធានធម្មជាតិដ៏មានតម្លៃដែលផ្តល់សារៈប្រយោជន៍ជាច្រើនដល់ប្រព័ន្ធអេកូឡូស៊ី ព្រមទាំងរួមចំណែកដល់សង្គមតាមរយៈមធ្យោបាយជាច្រើន។ ដោយសារការបំពុលខ្យល់អាចគំរាមកំហែងដល់សត្វស្លាបនៅក្នុងតំបន់ទីក្រុង ប៉ុន្តែមិនទាន់មានការសិក្សានៅប្រទេសកម្ពុជានៅឡើយ ដូច្នេះគោលបំណងនៃការសិក្សារបស់ខ្ញុំ គឺអង្កេតទំនាក់ទំនងរវាងគុណភាពខ្យល់ និងភាពសម្បូរបែបនៃសត្វស្លាបនៅទីតាំងចំនួនពីរក្នុងរាជធានីភ្នំពេញ។ ខ្ញុំបានប្រមូលទិន្នន័យនៅសាកលវិទ្យាល័យភូមិន្ទភ្នំពេញ និងសាកលវិទ្យាល័យភូមិន្ទកសិកម្ម ចាប់ពីថ្ងៃទី២១ ខែកុម្ភៈ ដល់ថ្ងៃទី២១ ខែឧសភា ឆ្នាំ២០២២។ ទិន្នន័យគុណភាពខ្យល់បូកបញ្ចូលទាំងកម្រិតភាគល្អិត (PM 2.5) ស្ពាន់ធ័រឌីអុកស៊ីត(SO_2) អាសូតឌីអុកស៊ីត(NO_2) និងអូហ្សូន(O_3) ទទួលបានពីស្ថានីយ៍ត្រួតពិនិត្យនៅក្បែរទីតាំងសិក្សាទាំងពីរ។ ចំណែកការរាប់តាមចំណុច (Point count) ត្រូវបានប្រើដើម្បីរាប់ចំនួន និងកំណត់អត្តសញ្ញាណរបស់ប្រភេទសត្វស្លាបនៅក្នុងតំបន់នីមួយៗ។ ជាលទ្ធផល មានវត្តមានសត្វស្លាបចំនួន១១,១៩០ក្បាល ត្រូវនឹង ៣៩ប្រភេទ ត្រូវបានរកឃើញក្នុងបរិវេណសាកលវិទ្យាល័យភូមិន្ទភ្នំពេញ និងសត្វស្លាបចំនួន៧,១៩៩ក្បាល ត្រូវនឹង ៥៦ប្រភេទ ត្រូវបានរកឃើញនៅក្នុងបរិវេណសាកល-វិទ្យាល័យភូមិន្ទកសិកម្ម។ Non-parametric tests ត្រូវបានប្រើប្រាស់ដើម្បីស្វែងរកភាពខុសគ្នារវាងទីតាំងសិក្សាទាំងពីរ ខណៈដែល logistic regression ត្រូវបានប្រើដើម្បីរកទំនាក់ទំនងរវាងចំនួនប្រភេទ និងចំនួនឯកត្តៈនៃប្រភេទ នីមួយៗជាមួយនឹងគុណភាពខ្យល់ ដោយរួមបញ្ចូលទាំងការប្រើតម្លៃលក្ខណៈវិនិច្ឆ័យព័ត៌មាន Akaike (AIC, Akaike information criterion) ជាជម្រើសគំរូ។ ទិន្នន័យទាំងនេះបង្ហាញថា ចំនួនប្រភេទ និងភាពសម្បូរបែបនៃសត្វស្លាបមានច្រើនជាងគឺនៅសាកលវិទ្យាល័យភូមិន្ទកសិកម្ម ចំណែកឯចំនួនឯកត្តៈនៃប្រភេទនីមួយៗមានចំនួនច្រើនជាងគឺនៅសាកលវិទ្យាល័យភូមិន្ទភ្នំពេញ។ កម្រិតភាគល្អិត (PM 2.5) ស្ពាន់ធ័រឌីអុកស៊ីត(SO_2) អាសូតឌីអុកស៊ីត(NO_2) និងអូហ្សូន(O_3) មានកម្រិតខ្ពស់ជាងគឺនៅសាកលវិទ្យាល័យភូមិន្ទកសិកម្ម បើទោះបីជាកម្រិតដែលបានកត់ត្រានៅទីតាំងទាំងពីរទាបជាងស្តង់ដាររបស់រដ្ឋាភិបាលក៏ដោយ។ នៅពេលដែលទិន្នន័យត្រូវបានបញ្ចូលគ្នា គំរូម៉ូឌែលដ៏ល្អបំផុតបានបង្ហាញថាស្ពាន់ធ័រឌីអុកស៊ីត(SO_2) មានទំនាក់ទំនងអវិជ្ជមានជាមួយចំនួនប្រភេទ និងចំនួនឯកត្តៈនៃប្រភេទ តែគំរូម៉ូឌែលល្អបំផុតនៅសាកលវិទ្យាល័យភូមិន្ទភ្នំពេញតែមួយគត់បានបង្ហាញថាចំនួនប្រភេទមានទំនាក់ទំនងអវិជ្ជមានជាមួយនឹងអូហ្សូន(O_3) ឯគំរូម៉ូឌែលល្អបំផុតនៅសាកលវិទ្យាល័យភូមិន្ទកសិកម្មបានបង្ហាញថាអូហ្សូន(O_3) មានទំនាក់ទំនងអវិជ្ជមានជាមួយនឹងចំនួនឯកត្តៈនៃប្រភេទនីមួយៗ។

Abstract

Birds are a valuable natural resource as they provide a variety of important ecosystem services and contribute to society in many ways. As air pollution may threaten birds in urban areas but remains unstudied in Cambodia, the aim of my study was to investigate relationships between air quality and the species richness and abundance of birds at two sites in Phnom Penh. I conducted sampling at the Royal University of Phnom Penh (RUPP) and the Royal University of Agriculture (RUA) from February to May 2022. Air quality data including levels of particular matter 2.5 (PM2.5), sulphur dioxide (SO_2), nitrogen dioxide (NO_2) and ozone (O_3) were obtained from nearby monitoring stations, whereas point counts were used to register the number and identity of bird species in each area. This resulted in 11,190 birds representing 34 species being recorded at RUPP and 7,144 birds belonging to 46 species being recorded at RUA. Non-parametric tests were used to test for differences between the two sites, whereas logistic regression was employed to explore associations between species richness and abundance and air quality, including the use of Akaike information criterion values for model selection. These revealed that bird species richness and diversity were significantly greater at RUA, whereas their abundance was significantly greater at RUPP. Levels of PM2.5, SO_2, NO_2 and O_3 were significantly greater at RUA, although the levels recorded at both sites were lower than government standards. When the data were pooled, the best model suggested that SO_2 was negatively associated with bird species richness and abundance, although the best model for RUPP alone suggested that species richness was negatively correlated with O_3 and the best model for RUA suggested O_3 was negatively correlated with abundance.

Ecological characteristics and temporal dynamics of mosquito species in Mondulkiri Province, Cambodia

DOEURK Bros

មូលន័យសង្ខេប

សត្វមូសជាង ២៩០ ប្រភេទត្រូវបានកត់ត្រានៅក្នុងប្រទេសកម្ពុជា ដោយរួមបញ្ចូលទាំងប្រភេទមូសមួយចំនួនដែលមានសារៈសំខាន់ក្នុងផ្នែកវេជ្ជសាស្ត្រ និងបសុសាស្ត្រ។ ការផ្លាស់ប្តូរនៃការប្រើប្រាស់ដីដូចជាការកាប់បំផ្លាញព្រៃឈើអាចប៉ះពាល់ដល់ភាពសម្បូរបែប និងអេកូឡូស៊ីសត្វមូស ប៉ុន្តែមានការយល់ដឹងតិចតួចណាស់នៅប្រទេសកម្ពុជា។ ដូចនេះយើងបានសិក្សាស្វែងយល់អំពីលក្ខណៈនៃភាពសម្បូរបែប ចំនួនឯកត្តៈ និងអកប្បកិរិយានៃការខាំរបស់សត្វមូសនៅដប់ទីតាំង ក្នុងស្រុកកោះញែក ខេត្តមណ្ឌលគិរី។ យើងបានប្រមូលសត្វមូសរៀងរាល់មួយខែម្តង រយៈពេល១១ខែ នៅឆ្នាំ២០២០ ពីបីទីតាំងនៅក្នុងភូមិ និងប្រាំពីរទីតាំងពីព្រៃនៅក្នុងតំបន់។ អន្ទាក់ស្បៃមុងពីរជាន់ ដោយប្រើមនុស្សជាផ្ទាក់ត្រូវបានដាក់នៅទីតាំងចំនួនប្រាំបួន ខណៈដែលអន្ទាក់ដូចគ្នាដែលប្រើមនុស្សជាផ្ទាក់ និងអន្ទាក់មួយទៀត ដែលប្រើសត្វគោធ្វើជាផ្ទាក់ត្រូវបានដាក់នៅទីតាំងទីដប់។ ជាលទ្ធផលសត្វមូសសរុបចំនួន ៥៤,៦៨០ ក្បាល ត្រូវនឹង ១១៩ ប្រភេទ និង ១៦ ពួកត្រូវបានកត់ត្រា។ សត្វ មូសចំនួនបីប្រភេទរួមមាន *Culex vishnui*.g *Aedes albopictus* និង *Anopheles dirus* មាន ចំនួន៣៥.៤% នៃសំណាកសត្វមូសទាំងអស់ដែលបានប្រមូលពីប្រាំបួនទីតាំងដែលបានប្រើមនុស្សជាផ្ទាក់។ ប្រភេទទាំងនេះមាន សារៈសំខាន់ចំពោះសុខភាពសាធារណៈ ដោយសារពួកវាជាភ្នាក់ងារចម្លងវីរុសរលាកខួរក្បាល វីរុសគ្រុនឈាម និងប៉ារ៉ាសិត គ្រុនចាញ់ជាដើម។ លក្ខណៈនៃការខាំប្រចាំថ្ងៃរបស់មូសមានភាពខុសគ្នាតាមបែបស្ថិតិ បើទោះបីជាសត្វមូសទាំងបីប្រភេទនេះមានវត្តមានទាំងពេលថ្ងៃ និងពេលយប់ក៏ដោយ នេះមានន័យថាពួកវាអាចចម្លងជម្ងឺទៅកាន់មនុស្សបានគ្រប់ពេលវេលា។ លទ្ធផលនៃការសិក្សានេះបង្ហាញថា ភាពសម្បូរបែបនៃប្រភេទមូសនៅក្នុងភូមិ ទាបជាងប្រភេទដែលនៅក្នុងព្រៃ។ វត្តមាននៃសត្វមូសជាច្រើនប្រភេទមានសមាមាត្រប្រហាក់ប្រហែលគ្នា រវាងអន្ទាក់ដែលប្រើមនុស្សធ្វើជាផ្ទាក់ និងអន្ទាក់ដែលប្រើសត្វគោធ្វើជាផ្ទាក់។ ដោយសារសត្វមូសមានវត្តមានទាំងនៅក្នុងភូមិ និងនៅក្នុងតំបន់ព្រៃ ពួកវាអាចជាភ្នាក់ងារដែលមានសក្តានុពលក្នុងការចម្លងជម្ងឺ។ លទ្ធផលនៃការសិក្សារបស់យើងបានបង្ហាញថា តំបន់នេះជាកន្លែងដែលសម្បូរជម្ងឺគ្រុនចាញ់ និងអាចបង្កអោយមានហានិភ័យនៃការចម្លងជម្ងឺដែលបង្កឡើងដោយវីរុស(Arbovirus) ពីសត្វទៅមនុស្ស។ លទ្ធផលនេះក៏បានបង្ហាញថាភាពលើសលុបនៃសត្វមូសប្រភេទ *Cx. vishnui*.g និង *Ae. albopictus* អាចមានទំនាក់ទំនងនឹងការរំខានរបស់មនុស្ស។

Abstract

Over 290 mosquito species are known in Cambodia, including some species which have medical and veterinary importance. As land use changes such as deforestation may affect the diversity and ecology of mosquitoes but is poorly understood in Cambodia, we explored patterns in the diversity, relative abundance and biting behaviour of mosquitoes at ten sites in Kaoh Nheaek district, Mondulkiri Province. To achieve this, we collected mosquitoes each month for 11 months at three sites in villages and seven sites in forest areas within the district in 2020. Double net traps baited with human odour were used at nine sites, whereas this and the same trap baited with cow odour were employed at the tenth site. A total of 54,680 mosquitoes representing 119 species formed in 16 genera were collected. Three species, *Culex vishnui.g*, *Aedes albopictus* and *Anopheles dirus*, accounted for 35.4% of mosquitoes collected at the nine sites baited with human odour alone. These taxa are important for public health, being vectors for Japanese encephalitis virus, dengue virus and malaria parasites, respectively. Their daily biting patterns also differed significantly, although all three were observed during the day and at night which means they could infect humans at any time. Our results indicate that mosquito diversity is lower in villages compared to forest areas. Many species occurred in similar proportions between traps baited with human and cow odours and since these occurred in villages as well as forests, they could potentially act as bridge vectors. Our results suggest that the study area, which remains a hotspot for malaria, could pose a risk of arbovirus transmission from animals to humans. They also suggest that the dominant species, *Cx. vishnui.g* and *Ae. albopictus*, may be associated with human disturbance.

Assessment of macroinvertebrate biodiversity in Phnom Nam Lyr, Mondulkiri Province

MEI Sophea

មូលន័យសង្ខេប

ក្រុមសត្វឥតឆ្អឹងកង (Macroinvertebrates) គឺជាសមាសភាពសំខាន់នៃប្រព័ន្ធអេកូឡូស៊ីទឹកសាបដែលត្រូវបានយកមកប្រើជាប្រភេទសូចនាករសម្រាប់ចង្អុលបង្ហាញការវាស់វែងគុណភាពទឹក។ ទោះបីមានការសិក្សាស្រាវជ្រាវថ្មីៗក៏ដោយ ក៏ក្រុមសត្វឥតឆ្អឹងកងនេះនៅតែមានការសិក្សាតិចតួចនៅក្នុងតំបន់ការពារក្នុងប្រទេសកម្ពុជា។ គោលបំណងនៃការសិក្សារបស់ខ្ញុំ គឺធ្វើការអង្កេតទៅលើបណ្តុំសត្វឥតឆ្អឹងកងនៅក្នុងដែនជម្រកសត្វព្រៃភ្នំណាមលៀរ (ខេត្តមណ្ឌលគិរី) និងធ្វើការស្វែងរកពីទំនាក់ទំនងក្រុមសត្វឥតឆ្អឹងកងជាមួយអថេរភាពនៃមជ្ឈដ្ឋានរស់នៅរបស់វា (កត្តារូប និងកត្តាគីមី)។ ដើម្បីសម្រេចគោលបំណងនៃការសិក្សានេះ ខ្ញុំបានប្រើស្នៀដង (Hand-nets) និងបង្កើតជម្រកសិប្បនិម្មិតដើម្បីប្រមូលសំណាកសត្វឥតឆ្អឹងកងនៅទីតាំងជាច្រើនក្នុងស្ទឹងចំនួន៣ ចាប់ពីខែមីនាដល់ខែមេសា ឆ្នាំ២០២២។ កត្តារូប និងគីមីត្រូវបានវាស់នៅទីតាំងប្រមូលសំណាកនីមួយៗដោយប្រើ handheld probes និង standard protocols ដើម្បីវាយតំលៃទីជម្រក។ ជាលទ្ធផល សត្វឥតឆ្អឹងកងសរុបចំនួន ២១១៨ក្បាល ត្រូវបានប្រមូល (សត្វល្អិតចំនួន១៦៦៥ ខ្យងចំនួន២៥២ និងក្រុមក្រុសតាសេចំនួន២០១) តំណាងឲ្យ ៦២អំបូរ និង១៥លំដាប់។ ប្រភេទក្នុងលំដាប់ Ephemeroptera ត្រូវបានរកឃើញច្រើនជាងគេដោយមានចំនួន៦២៩ក្បាល (ភាគច្រើនជាប្រភេទក្នុងអំបូរ Caenidae) ចំណែកឯលំដាប់ Lepidoptera បានរកឃើញតែ២ក្បាល ដែលជាចំនួនតិចជាងគេនៅក្នុងការសិក្សានេះ។ ឆ្លងកាត់ការវិភាគសត្វឥតឆ្អឹងកងពីទីតាំងសិក្សាទាំងអស់ ចំនួនឯកត្តៈមានជាមធ្យម ១១៨±៣៨ក្បាល ចំនួនប្រភេទជាមធ្យម ២១±៥ និងចំនួនមធ្យមសន្ទស្សន៍នៃភាពសម្បូរបែបមាន (Shannon diversity value) ២.៥±០.៥។ គ្មានភាពខុសគ្នាគួរអោយកត់សម្គាល់ត្រូវបានរកឃើញរវាងស្ទឹងទាំង៣ ក្នុងប្រៀបធៀបពីការវាស់វែងតម្លៃមធ្យមចំនួនឯកត្តៈ ចំនួនប្រភេទ និងចំនួនសន្ទស្សន៍នៃភាពសម្បូរបែប។ ចំនួនឯកត្តៈនៃសត្វឥតឆ្អឹងកងមានទំនាក់ទំនងវិជ្ជមានជាមួយកម្រិតអុកស៊ីនក្នុងទឹក និងមានទំនាក់ទំនងអវិជ្ជមានជាមួយសីតុណ្ហភាពទឹក ចំណែកឯសន្ទស្សន៍នៃភាពសម្បូរបែបមានទំនាក់ទំនងវិជ្ជមានជាមួយទីជម្រកមានគម្របព្រៃឬស្សី តែមានទំនាក់ទំនងអវិជ្ជមានជាមួយទំហំថ្ម (>២៥៥ មម) ដែលមាននៅក្នុងបាតស្ទឹង។ ការរកឃើញរបស់ខ្ញុំ ផ្តល់ជាមូលដ្ឋានមានប្រយោជន៍សម្រាប់ការសិក្សាក្រោយៗ និងសម្រាប់ធ្វើការតាមដានពីការប្រែប្រួលនៃសត្វឥតឆ្អឹងកងដែលរស់នៅក្នុងដែនជម្រកសត្វព្រៃភ្នំណាមលៀរ។

Abstract

Macroinvertebrates are a key component of freshwater ecosystems and are often used as indicator species in water quality assessments. Despite recent research, they remain poorly documented in protected areas in Cambodia. The aim of my study was to investigate macroinvertebrate assemblages in Phnom Nam Lyr Wildlife Sanctuary (Mondulkiri Province) and explore their relationships to selected environmental variables (physical and chemical). To achieve this, I used hand-nets and artificial substrate traps to collect samples of macroinvertebrates at multiple locations on three streams within the wildlife sanctuary in March and April 2022. Environmental variables were measured at each loca-

tion using handheld probes and standard protocols for habitat assessments. This resulted in the collection of 2,118 macroinvertebrates (1,665 insects, 252 molluscs and 201 crustaceans) representing 62 families and 15 orders. Members of the Ephemeroptera were the most common taxa found (particularly members of the Caenidae) with 629 individuals, whereas lepidopteran taxa were among the rarest, with only two individuals. Across all sampling sites, the mean abundance of macroinvertebrates was 118 ± 38 individuals, whereas mean species richness was 21 ± 5 species and the mean Shannon diversity value was 2.5 ± 0.5. No significant differences were found between the streams in terms of these measures. Macroinvertebrate abundance was positively associated with dissolved oxygen levels and negatively associated with water temperature, whereas Shannon diversity values were positively associated with the proportion of bamboo present and negatively associated with the percentage cover of large stones (>255 mm) within the stream substrates. My findings provide a useful baseline for future studies and monitoring of macroinvertebrates inhabiting Phnom Nam Lyr Wildlife Sanctuary.

Bat and small mammal diversity in the Phnom Tamao forest, Takeo Province, Cambodia

PHANN Sam Ath

មូលន័យសង្ខេប

ប្រចៀវ និងថនិកសត្វតូចៗផ្សេងទៀតដើរតួនាទីយ៉ាងសំខាន់ក្នុងការរក្សាតុល្យភាពប្រព័ន្ធអេកូឡូស៊ីព្រៃឈើ តាមរយៈការធ្វើដំណើរលំអងរុក្ខជាតិ ការពង្រាយគ្រាប់រុក្ខជាតិ និងការគ្រប់គ្រងពួកអាត្រូប៉ូត។ គោលបំណងនៃការសិក្សារបស់ខ្ញុំគឺដើម្បីអង្កេតមើលអំពីចំនួនប្រភេទ ចំនួនឯកត្តៈ និងភាពសម្បូរបែបនៃសត្វប្រចៀវនិងថនិកសត្វតូចៗ ដែលមានវត្តមាននៅតំបន់ព្រៃក្នុងឧទ្យានសួនសត្វនិងមជ្ឈមណ្ឌលសង្គ្រោះសត្វព្រៃភ្នំតាម៉ៅ នៅក្នុងខេត្តតាកែវ ព្រមទាំងធ្វើការប្រៀបធៀបបណ្តុំនៃប្រភេទទាំងនោះរវាងជម្រកក្នុងព្រៃនិងជម្រកគែមព្រៃ។ ក្នុងការសិក្សានេះខ្ញុំបានប្រមូលសំណាកសត្វប្រចៀវនិងថនិកសត្វតូចៗពីជម្រកខាងក្នុងព្រៃចំនួន០៦ទីតាំង និងជម្រកគែមព្រៃចំនួន០៦ទីតាំង នៅតំបន់ព្រៃភ្នំតាម៉ៅ ដោយការសិក្សានេះត្រូវបានធ្វើឡើងកាលពីខែមេសា ឆ្នាំ២០២២ តាមរយៈការប្រើប្រាស់អន្ទាក់សំណាញ់ អន្ទាក់រាំង និងអន្ទាក់ទ្រុង។ ប្រចៀវសរុបចំនួន១១៩ក្បាល ត្រូវជាចំនួន០៥ប្រភេទ និងថនិកសត្វតូចចំនួន២៥ក្បាល ត្រូវជា០៨ប្រភេទ ត្រូវបានកត់ត្រាចេញពីការសិក្សានេះ។ ក្នុងចំណោមសត្វប្រចៀវទាំងអស់ ប្រភេទប្រចៀវស៊ីផ្លែឈើ *Cynopterus sphinx* គឺជាប្រភេទដែលចាប់បានច្រើនជាងគេមានចំនួនដល់ទៅ១០១ក្បាល ហើយបន្ទាប់មកមានប្រភេទ *Megaderma spasma* (០៩ក្បាល) *Rhinolophus acuminatus* (០៦ក្បាល) *Kerivoula picta* (០២ក្បាល) និង *K.* cf. *hardwickii* (០១ក្បាល)។ ចំពោះថនិកសត្វតូចវិញ ប្រភេទកន្សើក *Tupaia belangeri* គឺជាប្រភេទដែលមានចំនួនឯកត្តៈច្រើនជាងគេ (ចំនួន០៩ក្បាល) បន្ទាប់មកមានប្រភេទកង្ហិចកម្ពុជា *Tamiops rodolphii* (០៥ក្បាល) កណ្តុរស្រុក *Rattus tanezumi* (០៣ក្បាល) កណ្តុរព្រៃ *Maxomys surifer* (០៣ក្បាល) កំប្រុកពណ៌ *Callosciurus finlaysonii* (០២ក្បាល) ហើយប្រភេទកណ្តុរស្រែ *R. argentiventer* កណ្តុរសញ្ញាសី *R. exulans,* និងប្រភេទកំប្រុកដី *Menetes berdmorei* មានតែ០១ក្បាលក្នុងមួយប្រភេទ។ របាយនៃសត្វប្រចៀវនិងថនិកសត្វតូចៗនៅតំបន់ព្រៃភ្នំតាម៉ៅមិនមានភាពដូចគ្នាទេ។ ខណៈដែលមិនមានភាពខុសគ្នាក្នុងស្ថិតិនៃភាពសម្បូរបែបរបស់ក្រុមសត្វទាំងនេះរវាងទីតាំងជម្រកខាងក្នុងព្រៃ និងទីតាំងជម្រកគែមព្រៃ ប៉ុន្តែសមាសភាពរបស់ពួកវាមានភាពខុសគ្នា ដោយលទ្ធផលបានបង្ហាញថា ទីតាំងគែមព្រៃមានចំនួនប្រភេទ និងចំនួនឯកត្តៈទាបជាងទីតាំងខាងក្នុងព្រៃ។ ភាពខុសគ្នានៃទីជម្រកដើរតួនាទីសំខាន់សម្រាប់ទ្រទ្រង់រចនាសម្ព័ន្ធ និងសមាសភាពនៃបណ្តុំសត្វប្រចៀវ និងថនិកសត្វតូចៗ ដែលមានវត្តមាននៅតំបន់ព្រៃភ្នំតាម៉ៅ។

Abstract

Bats and other non-volant small mammals play important roles in regulating forest ecosystems through plant pollination, seed dispersal and arthropod suppression. The aim of my study was to investigate the species richness, abundance and diversity of bats and small mammals inhabiting forest areas of Phnom Tamao Zoological Park and Wildlife Rescue Centre, Takeo Province and compare assemblages of these taxa between forest interior and forest edge habitats. To

achieve this, I sampled bats and small mammals at six forest interior and six forest edge locations at the site in April 2022 using mist nets, harp traps and cage traps. A total of 119 bats representing five species and 24 small mammals representing eight species were recorded. Among the bats recorded, *Cynopterus sphinx* was the most common species with 101 individuals, followed by *Megaderma spasma* (nine), *Rhinolophus acuminatus* (six), *Kerivoula picta* (two) and *K.* cf. *hardwickii* (one). Among the other small mammals, *Tupaia belangeri* was the most abundant species with nine individuals, followed by *Tamiops rodolphii* (four), *Rattus tanezumi* (three), *Maxomys surifer* (three), *Callosciurus finlaysonii* (two) and *R. argentiventer*, *R. exulans* and *Menetes berdmorei*, each with one individual apiece. The distribution of bats and small mammals in the forests of Phnom Tamao was not homogenous. While there was no significant difference in the diversity of these groups between forest edge and forest interior habitats, their composition did differ, with lower abundance and species richness observed in forest edge habitats. Habitat differences play an important role in structuring the structure and composition of bat and non-volant small mammal assemblages in the forests of Phnom Tamao.

Assessing spatial and seasonal patterns of fish diversity of artisanal fisheries in the Sre Ambel River system

YON Tony

មូលន័យសង្ខេប

នេសាទលក្ខណៈគ្រួសាររួមចំណែកយ៉ាងសំខាន់ក្នុងការផ្គត់ផ្គង់តម្រូវការប្រូតេអ៊ីនដល់ប្រជាជនកម្ពុជា។ ប៉ុន្តែការគ្រប់គ្រងរបស់រាជរដ្ឋាភិបាលលើវិស័យជលផលនៅមានកម្រិត និងមានលក្ខណៈបែបបុរាណ ដែលនាំឲ្យការគ្រប់គ្រងការនេសាទលក្ខណៈគ្រួសារមិនសូវបានល្អ និងអាចឈានដល់ការនេសាទហួសកម្រិត។ ការរៀបចំសហគ្រប់គ្រង ដែលរដ្ឋាភិបាលផ្ទេរសិទ្ធិនៃការគ្រប់គ្រងមួយចំនួនដល់សហគមន៍មូលដ្ឋានជាដំណោះស្រាយមួយ ប៉ុន្តែត្រូវបានរាំងស្ទះដោយកង្វះទិន្នន័យស្តង់ដារ ដែលមានសារៈសំខាន់សម្រាប់ការគ្រប់គ្រង។ និក្ខេបបទរបស់ខ្ញុំពិពណ៌នាពីការប្រើកម្មវិធី citizen science ដោយជ្រើសរើសប្រជានេសាទចំនួន ១៥ នាក់ ពីភូមិចំនួនបួន ដែលបានបង្កើតដើម្បីសហគមន៍នេសាទនៅព្រែកស្រែអំបិលភាគខាងត្បូងប្រទេសកម្ពុជា។ គោលបំណងនៃការសិក្សានេះគឺដើម្បីកំណត់បំរែបំរួលនៃការប្រមូលផលនេសាទនៅប្រព័ន្ធព្រែក និងដើម្បីប្រៀបធៀបការប្រមូលផលក្នុងអំឡុងពេលនៃការសិក្សា។ ជាលទ្ធផលនៃការសិក្សាជាងមួយឆ្នាំ ត្រីចំនួន ៧០,៧៧៨ក្បាល ត្រូវនឹង ១៥១ ប្រភេទ ៦០ អំបូរ និង ២៧ លំដាប់ ត្រូវបានកត់ត្រា។ ឧបករណ៍នេសាទច្រើនជាងដប់ប្រភេទត្រូវបានធ្វើកំណត់ត្រា។ ទោះបីជាសំណាញ់មងត្រូវបានគេប្រើប្រាស់ច្រើន ជាងគេក៏ដោយក៏ទិន្នន័យនៃឧបករណ៍នេសាទនេះត្រូវបានប្រើដើម្បីធ្វើការវិភាគប្រៀបធៀប។ ប៉ែកខាងលើនៃព្រែកកំពង់សោមមានប្រភេទត្រីច្រើនជាងគេ បន្ទាប់មកគឺតំបន់ពាមនៃព្រែកស្រែអំបិល។ ប៉ែកកណ្តាលនៃព្រែកកំពង់សោម និងភាគខាងកើតនៃព្រែកគង់ មានប្រភេទត្រីតិចជាងគេ។ អត្រាការចាប់ត្រីនៅតំបន់ទាំងនេះមានលក្ខណៈប្រហាក់ប្រហែលគ្នា លើកលែងប៉ែកកណ្តាលនៃព្រែកកំពង់សោម ដែលមានភាគរយតិចជាងគេ។ ទោះបីជាការនេសាទមានភាពសកម្មខ្លាំងនៅរដូវវស្សា តែភាពសម្បូរបែបនៃប្រភេទត្រី និងអត្រាចាប់បានមានច្រើននៅរដូវប្រាំង។ ទិន្នន័យទាំងនេះអាចត្រូវបានប្រើដើម្បីវាយតម្លៃការឆ្លើយតបទៅនឹងសកម្មភាពជលផលនាពេលអនាគតដែលជាវិធីសាស្រ្តសម្រាប់សម្របការគ្រប់គ្រង។

Abstract

Small-scale fisheries provide a significant portion of the protein required by many people in Cambodia. However, government resources for fisheries management are limited and traditionally dedicated to more important fisheries, leaving small-scale fisheries largely under-managed and susceptible to overfishing. Co-management arrangements, whereby government agencies transfer some authority for management to local communities, represent one solution but are hindered by a lack of standardized data collection, which is necessary for informed management. My thesis describes a citizen science initiative involving 15 fishers from four villages which was developed for a community fishery in the Sre Ambel River system in southern Cambodia. My study objectives were to determine variation in harvests across the river system and to compare harvests over time. Over one year of monitoring, data from 70,778 fish

representing 151 species arranged in 60 families and 27 orders were recorded. More than ten types of fishing gear were documented, although as gillnets were most commonly used, these data were employed for comparative analyses. The upper reaches of the Kampong Som River were most species rich, followed by the estuarine-influenced mainstem of the Sre Ambel River. The middle reaches of the Kampong Som River and the eastern Kaong River were less speciose by comparison. Catch rates were similar between these, except for the middle reaches of the Kampong Som River, which were significantly lower. Although fishing effort was significantly greater during the wet season, species richness and catch rates were greater in the dry season. These data may be used to evaluate the response of the fishery to future actions, allowing for an adaptable approach to management.

Instructions for Authors

Purpose and Scope

The *Cambodian Journal of Natural History* (ISSN 2226–969X) is an open access, peer-review journal published biannually by the Centre for Biodiversity Conservation at the Royal University of Phnom Penh. The Centre for Biodiversity Conservation is a non-profit making unit, dedicated to training Cambodian biologists and the study and conservation of Cambodia's biodiversity.

The *Cambodian Journal of Natural History* publishes original work by:

- Cambodian or foreign scientists on any aspect of Cambodian natural history, including fauna, flora, habitats, management policy and use of natural resources.
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Title: A succinct description of the work, in no more than 20 words.

Abstract: (Full papers only). This should describe, in no more than 250 words, the aims, methods, major findings and conclusions. The abstract should be informative and intelligible without reference to the text, and should not contain any references or undefined abbreviations. Cambodian authors are strongly encouraged to submit a Khmer translation of the English abstract.

Keywords: (Full papers only). Up to eight pertinent words, in alphabetical order.

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Berzins, B. (1973) Some rotifers from Cambodia. *Hydrobiologia,* **41,** 453–459.

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Tanaka S. & Ohtaka A. (2010) Freshwater Cladocera (Crustacea, Branchiopoda) in Lake Tonle Sap and its adjacent waters in Cambodia. *Limnology,* **11,** 171–178.

Books and chapters:

Khou E.H. (2010) *A Field Guide to the Rattans of Cambodia.* WWF Greater Mekong Cambodia Country Programme, Phnom Penh, Cambodia.

MacArthur, R.H. & Wilson, E.O. (1967) *The Theory of Island Biogeography*. Princeton University Press, Princeton, USA.

Rawson, B. (2010) The status of Cambodia's primates. In *Conservation of Primates in Indochina* (eds T. Nadler, B. Rawson & Van N.T.), pp. 17–25. Frankfurt Zoological Society, Frankfurt, Germany, and Conservation International, Hanoi, Vietnam.

Reports:

Lic V., Sun H., Hing C. & Dioli, M. (1995) *A Brief Field Visit to Mondolkiri Province to Collect Data on Kouprey* (Bos sauveli), *Rare Wildlife and for Field Training.* Unpublished report to Canada Fund and IUCN, Phnom Penh, Cambodia.

Theses:

Yeang D. (2010) *Tenure rights and benefit sharing arrangements for REDD: a case study of two REDD pilot projects in Cambodia*. MSc thesis, Wageningen University, Wageningen, The Netherlands.

Websites:

IUCN (2010) *2010 IUCN Red List of Threatened Species.* Http://www.redlist.org [accessed 1 December 2010].

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Cambodian Journal of Natural History

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Errata: Two images were incorrectly attributed in the caption for Fig. 2 of Chartier & Kosterin (2022). Fig. 2E should have been attributed to D. Jump, whereas Fig. 2F should have been attributed to E. Smith. Consequently, "F - used with kind permission of author" should have read "E - used with kind permission of the author".

Chartier, G. & Kosterin, O.E. (2022) An annotated checklist of the butterflies (Lepidoptera: Papilionoidea) of Cambodia. *Cambodian Journal of Natural History*, **2022**, 99–126.

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