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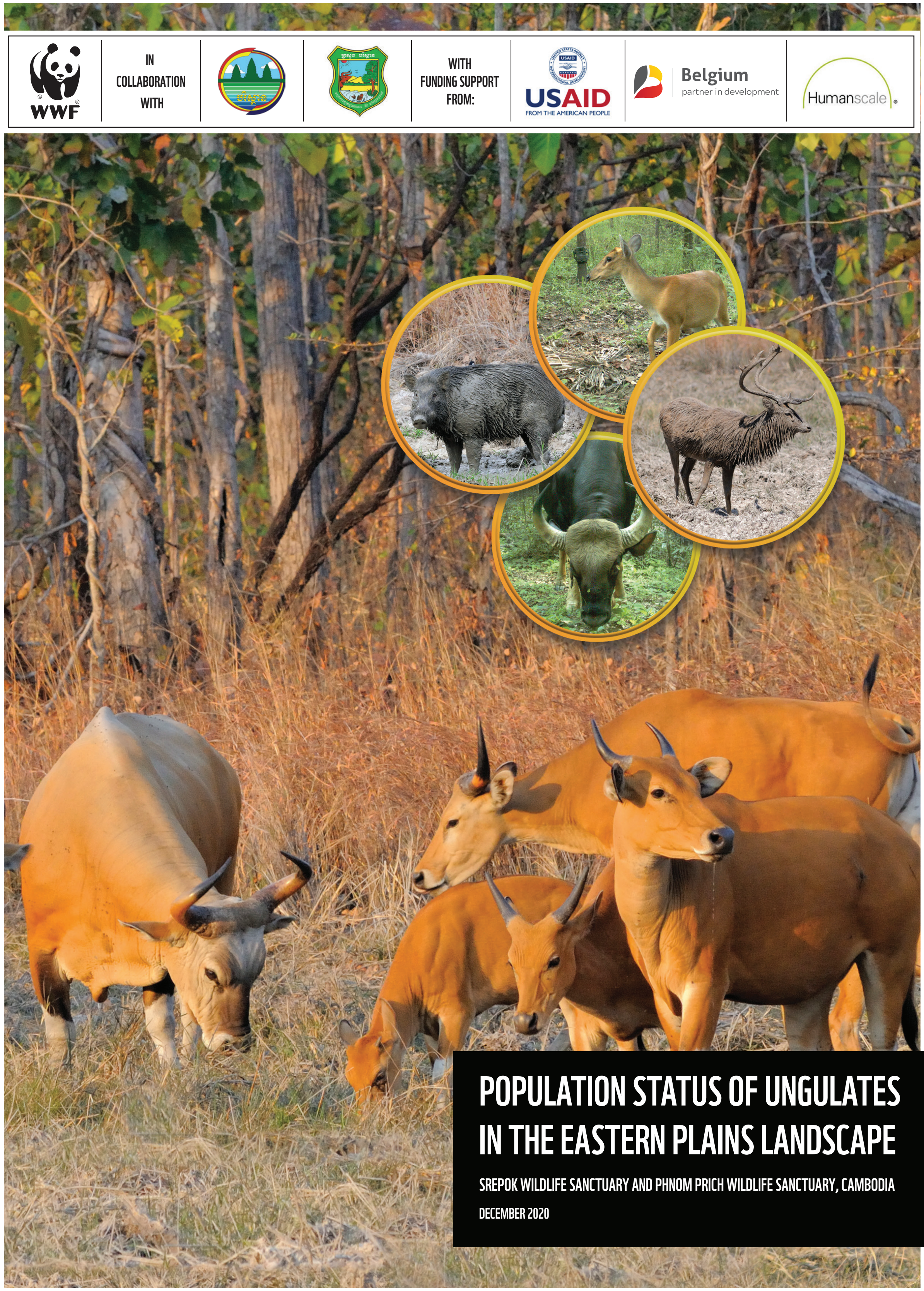

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POPULATION STATUS OF UNGULATES IN THE EASTERN PLAINS LANDSCAPE

SREPOK WILDLIFE SANCTUARY AND PHNOM PRICH WILDLIFE SANCTUARY, CAMBODIA

DECEMBER 2020

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Suggested citation: Groenenberg, M., Crouthers, R. & Yoganand, K. (2020) Population Status of Ungulates in the Eastern Plains Landscape. Srepok Wildlife Sanctuary and Phnom Prich Wildlife Sanctuary, Cambodia. Technical Report. WWF-Cambodia, Phnom Penh, Cambodia.

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Publication design and layout by: Von Sreyrat and Ravy Sophearoth
Photo on front cover: © Fletcher & Baylis / GDANCP / WWF-Cambodia
Photo on back cover: © Gordon Congdon / WWF-Cambodia

Acknowledgement summary

This long-term ungulate monitoring programme was made possible with funding from WWF-Belgium, WWF-Germany, WWF-Sweden, WWF-Switzerland, WWF-US, USAID, and Humanscale. Surveys in former Mondulkiri Protected Forest (2010/11/14) were conducted with the permission and support of the Forestry Administration of the Ministry of Agriculture Forestry and Fisheries. Surveys in SWS (2016/18/20) and in PPWS were conducted with permission and support of the Ministry of Environment.

This monitoring programme, the survey designs, implementation, and analysis were conducted by Thomas Gray (2010-11), Rachel Crouthers (2014-2018), and Milou Groenenberg (2020). None of this research and long term data would have been possible without their hard-work and dedication and the research officers and research rangers. A warm thanks to all those to contribute with technical advice, and for providing logistical and administrative support. Detailed acknowledgements are included in page 59.

***Errata:** calculation errors in Table 4 of page 39 and associated text in Section 3.3 have been corrected in this updated version of the report, dated September 2021

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សេចក្តីសង្ខេប

តំបន់ព្រៃទេសភាពខ្ពង់រាបភាគខាងកើត (EPL) នៃប្រទេសកម្ពុជា និងភាគកណ្តាលនៃប្រទេសវៀតណាម គឺជាតំបន់ព្រៃល្បោះដ៏ធំមួយដែលមាននៅសេសសល់នៅក្នុងតំបន់ដែលកំពុងរងការគម្រាមកំហែងយ៉ាងធ្ងន់ធ្ងរ។ តំបន់នេះទ្រទ្រង់ប្រភេទសត្វកំពុងរងគ្រោះថ្នាក់ជាច្រើននៅក្នុងសកលលោករួមមាន សត្វទន្សោង(Bos javanicus) ដែលមានចំនួនច្រើនជាងគេ ប្រភេទសត្វត្រយ៉ង និងសត្វត្មាតដែលជាប្រភេទរងគ្រោះថ្នាក់ជិតផុតពូជបំផុត ដំរីអាស៊ី(Elephas maximus) ដែលមានចំនួនច្រើនជាងគេនៅក្នុងប្រទេសកម្ពុជា សត្វក្រពើភ្នំ (Crocodylus siamensis) ដែលមានចំនួនចុងក្រោយនៅក្នុងប្រទេស និងសត្វខ្លារខិន (Panthera pardus delacouri) ដែលមានចំនួនសេសសល់នៅក្នុងតំបន់ឥណ្ឌូចិនតែមួយគត់។

ការតាមដានពពួកសត្វក្រចកជើងចំពាមនៅក្នុង តំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើតក្នុងរយៈពេលមួយទសវត្សរ៍ និងការប៉ាន់ប្រមាណចំនួនសត្វព្រៃ គឺមានសារៈសំខាន់ណាស់ក្នុងការវាយតម្លៃពីផលប៉ះពាល់នៃកត្តាគំរាមកំហែងទៅលើសត្វព្រៃ គាំទ្រនូវការសម្រេចចិត្តដោយផ្អែកទៅលើមូលដ្ឋាននៃការអភិរក្ស និងប្រសិទ្ធភាពនៃការគ្រប់គ្រងតំបន់ការពារ។ នៅក្នុងដែនជម្រកសត្វព្រៃភ្នំព្រេច និងស្រែពក(PPWS និង SWS) ដែលមានទីតាំងស្ថិតនៅចំកណ្តាលនៃតំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើតបានអនុវត្តកម្មវិធីតាមដានពពួកសត្វមានក្រចកជើងចំពាមក្នុងកំឡុងពេលមួយទសវត្សរ៍ (២០១០- ២០២០) ដែលលទ្ធផលនឹងបង្ហាញនៅក្នុងរបាយការណ៍នេះ។ ដោយប្រើវិធីសាស្រ្តដើរបន្ទាត់ត្រង់ (Distance-sampling) យើងអាចវាយតម្លៃទៅលើបច្ចុប្បន្នភាពនៃចំនួនប៉ាន់ប្រមាណសរុប និងបម្រែបម្រួលចំនួននៃសត្វទន្សោង(Bos javanicus) ឈ្លូស(Muntiacus vaginalis) និងជ្រូកព្រៃ(Sus scrofa) ដែលសុទ្ធសឹងជាប្រភេទរំពារសម្រាប់ពពួកមំសាសីនៅក្នុងតំបន់ទេសភាពនេះ។

លទ្ធផល

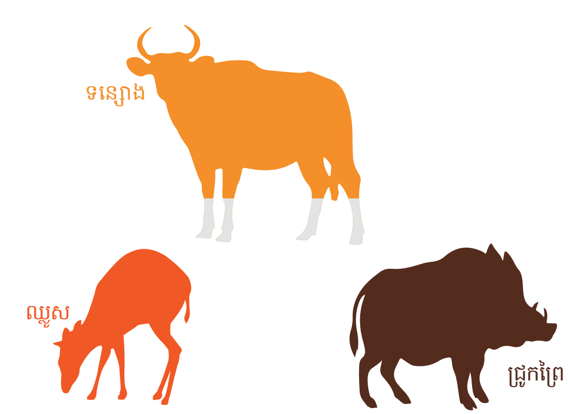
តាមការសិក្សារវាងឆ្នាំ២០១០-១១ បានប៉ាន់ប្រមាណថាតំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើតជាទីជម្រកនៃសត្វទន្សោងច្រើនជាងគេក្នុងពិភពលោក ដែលមានចំប្រមាណ១,៩១១ក្បាល [មានតម្លៃចន្លោះ ៨៧០- ២,៩៥២ក្បាល(កម្រិតជឿជាក់ ៩៥%)] នៅក្នុងដែនជម្រកសត្វព្រៃស្រែពក និងមានចំនួនប្រមាណ១,១០២ក្បាល [មានតម្លៃចន្លោះពី៦០២-២០១៨ក្បាល(កម្រិតជឿជាក់៩៥%)] នៅក្នុងដែនជម្រកសត្វព្រៃភ្នំព្រេច។ នៅក្នុងឆ្នាំ២០២០ ចំនួនសត្វទន្សោង៧២% បានធ្លាក់ចុះ បើធៀបទៅនឹងចំនួនប៉ាន់ប្រមាណនៅឆ្នាំ២០១០-២០១១។ ចំនួនប៉ាន់ប្រមាណដង់ស៊ីតេនៃសត្វឈ្លូសក៏ធ្លាក់ចុះដែរនៅឆ្នាំ២០២០នេះ បើប្រៀបធៀបទៅនឹងឆ្នាំ២០១០-២០១១។ មានតែសត្វជ្រូកព្រៃតែប៉ុណ្ណោះដែលចំនួនដង់ស៊ីតេរបស់វាមិនធ្លាក់ចុះ ប៉ុន្តែបានបង្ហាញពីការបម្រែបម្រួលនៃចំនួនរបស់វាទៅតាមពេលវេលាសម្រាប់ដែនជម្រកសត្វព្រៃទាំងពីរ។ នៅឆ្នាំ២០២០ដង់ស៊ីតេនៃ ជ្រូកព្រៃទាបជាងដង់ស៊ីតេនៃតម្លៃមូលដ្ឋាននៅដែនជម្រកសត្វព្រៃស្រែពក ប៉ុន្តែមានតម្លៃប្រហាក់ប្រហែលគ្នានៅក្នុង ដែនជម្រកសត្វព្រៃភ្នំព្រេច (តារាងទី១)។

អត្រានៃការអង្កេតប្រភេទសត្វមានក្រចកចំពាមដូចជារមាំង (Rucervus eldii) ខ្ទីង (Bos gaurus) និងប្រើស (Rusa unicolor) មានតម្លៃទាបណាស់ (ចាប់ពីសូន្យរហូតដល់៥ក្បាលក្នុងមួយការអង្កេត) ដែលចំនួនតិចតួចនេះកំពុងរងការគម្រាមកំហែងយ៉ាងខ្លាំង។

តារាងទី១–ដង់ស៊ីតេ និងចំនួនសរុប (±កម្រិតលំអៀង) ចំពោះប្រភេទសត្វមានក្រចកចំពាមបីប្រភេទនៅក្នុងដែនជម្រកសត្វព្រៃស្រែពក និងភ្នំព្រេចបានប៉ាន់ប្រមាណពីការសិក្សាដោយប្រើវិធីដើរបន្ទាត់ត្រង់(ផ្ទៃដី-ដង់ស៊ីតេដែលបានប្រើសម្រាប់សិក្សាឆ្នាំ២០១០-១១ បានប៉ាន់ប្រមាណដង់ស៊ីតេនៃតំបន់ស្នូលនៅក្នុងដែនជម្រកសត្វព្រៃស្រែពក) និងការបម្រែបម្រួលដង់ស៊ីតេរវាងទិន្នន័យមូលដ្ឋាន និងទិន្នន័យទទួលបានចុងក្រោយដែលមានកម្រិត (p-value)។

ប្រភេទ	តំបន់	ឆ្នាំ	ដង់ស៊ីតេ/គម² ± កម្រិតលំអៀង	ដង់ស៊ីតេនៃកម្រិតជឿជាក់៩៥%	បំរែបំរួលដង់ស៊ីតេ២០២០ធៀបនឹង២០១០-១១	ចំនួនសរុប± កម្រិតលំអៀង	ចំនួនសរុបនៃកម្រិតជឿជាក់៩៥%
ទន្សោង	ស្រែពក	2010-11	1.10 ± 0.31	0.50-1.72	-80.61% (p=0.003)	1911 ± 531	870-2952
	ស្រែពក	2020	0.21 ± 0.10	0.09-0.53		371 ± 177	150 - 918
	ភ្នំព្រេច	2010-11	0.66 ± 0.20	0.36-1.21	-55.99% (p=0.069)	1102 ± 341	602-2018
	ភ្នំព្រេច	2020	0.29 ± 0.14	0.11-0.74		485 ± 238	191 - 1230
ឈ្លូស	ស្រែពក	2010-11	2.56 ± 0.24	2.09-3.06	-67.99% (p=8.000E-10)	4453 ± 424	3623-5283
	ស្រែពក	2020	0.82 ± 0.16	0.55-1.22		1425 ± 284	962-2112
	ភ្នំព្រេច	2010-11	1.52 ± 0.29	1.04-2.22	-24.3% (p=0.127)	2543 ± 478	1744-3708
	ភ្នំព្រេច	2020	1.15 ± 0.15	0.89-1.50		1925 ± 256	1480-2504
ជ្រូកព្រៃ	ស្រែពក	2010-11	1.91 ± 0.42	1.08-2.77	-34.56% (p=0.092)	3314 ± 738	1398-4330
	ស្រែពក	2020	1.25 ± 0.26	0.84-1.87		2169 ± 443	1451-3240
	ភ្នំព្រេច	2010-11	0.95 ± 0.29	0.53-1.71	15.9% (p=0.347)	1595 ± 479	891-2855
	ភ្នំព្រេច	2020	1.11 ± 0.26	0.70-1.75		1848 ± 431	1170-2919

ដង់ស៊ីតេនៃពពួកសត្វក្រចកចំពាមនៅក្នុងដែនជម្រកសត្វព្រៃស្រែពក និងភ្នំព្រេចមានតម្លៃទាបខ្លាំងធៀបទៅនឹងដង់ស៊ីតេនៅក្នុងតំបន់អាស៊ីខាងត្បូងដែលបង្ហាញថាចំនួនរបស់វាកំពុងធ្លាក់ចុះខ្លាំងនៅតំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើត។ ការវាយតម្លៃទៅលើជីវម៉ាស់នៃចំណីរបស់សត្វខ្លាធំគឺ១១៥.៧គ.ក្រ/គ.ម²នៅក្នុងឆ្នាំ២០២០ គឺតិចជាង៥%នៃការវាយតម្លៃជីវម៉ាស់នៃចំណីខ្លាធំនៅក្នុងតំបន់អាស៊ីអាគ្នេយ៍ (ដែនជម្រកសត្វព្រៃ ហួខាខែង នៃប្រទេសថៃ (Simcharoen et al., 2014))។



ការកើនឡើងនៃកត្តាគំរាមកំហែង

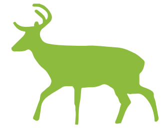
ប្រវត្តិសាស្ត្រនៃតំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើត ជាតំបន់មួយសំបូរទៅដោយពពួកសត្វក្រចកចំពាមធំៗជាច្រើនដែលទ្រទ្រង់ទៅដល់ពពួកមំសាសីនៅក្នុងតំបន់។ ដោយសារមានជម្លោះប្រដាប់អាវុធជាច្រើនទសវត្សរ៍នៅក្នុងប្រទេសភ្ជាប់ជាមួយនឹងការបរបាញ់ខុសច្បាប់បានបណ្តាលឲ្យមានការបាត់បង់ថនិកសត្វធំៗនៅក្នុងដែនជម្រកមួយចំនួនរបស់ពួកវា (Loucks et al., 2009) និងបណ្តាលឲ្យផុតពូជប្រភេទមួយចំនួនដូចជា គោព្រៃ (Bos sauveli) និងសត្វខ្លាធំ (Panthera tigris)ពីព្រៃធម្មជាតិជាដើម។ នាបច្ចុប្បន្ននេះវិបត្តិនៃការបរបាញ់ និងការដាក់អន្ទាក់ កើតឡើងពាសពេញក្នុងតំបន់ឥណ្ឌូចិន។ រីឯនៅតំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើតនៃប្រទេសកម្ពុជា បាននិងកំពុងទទួលរងសម្ពាធយ៉ាងខ្លាំងដោយសារការបរបាញ់នៅក្នុងមួយទសវត្សរ៍ចុងក្រោយនេះ។ នៅក្នុងកំឡុងពេលនៃការតាមដានចន្លោះពីឆ្នាំ២០១០-២០២០ អត្រានៃការប្រមូលអន្ទាក់មានការកើនឡើងពី០.០៤ /១០០គមនៅឆ្នាំ២០១០ ទៅ ៦.៤៦/១០០គមនៅឆ្នាំ២០២០ ដែលត្រូវបានប្រមូលដោយក្រុមល្បាតនៃមន្ត្រីឧទ្យានុរក្សដោយថ្មើជើង និងម៉ូតូ។ បច្ចុប្បន្ននេះមានការកើនឡើងនៃវិធីសាស្ត្រនៃការដាក់អន្ទាក់ជាច្រើនរួមមានការដាក់អន្ទាក់ដោយអង្គប់ និងអន្ទាក់ដោយខ្សែភ្លើងអគ្គិសនីដើម្បីសម្លាប់សត្វ។

ការកើនឡើងនៃការបរបាញ់សត្វព្រៃ គឺជាកត្តាចម្បងបណ្តាលមកពីការជួញដូរសត្វព្រៃ និងការកើនឡើងតម្រូវការសាច់សត្វព្រៃ បំណែកផ្សេងៗនៃសត្វព្រៃ និងផលិតផលផលិតពីសត្វព្រៃនៅក្នុងប្រទេសកម្ពុជា និងការនាំចេញទៅប្រទេសវៀតណាម។ ក្រៅពីការបរបាញ់ កត្តាគំរាមកំហែងផ្សេងទៀតដល់ពពួកសត្វក្រចកចំពាមនេះរួមមាន ការរុករានដីព្រៃ ការបំប្លែងដីព្រៃនិងភាពរិចរិលនៃព្រៃឈើ ការជួញដូរឈើខុសច្បាប់ ការរីកចម្រើននៃសេដ្ឋកិច្ចសង្គម និងការកើនឡើងនៃគមនាគមន៍ចូលក្នុងតំបន់ព្រៃដាច់ស្រយាលដោយសារការកសាងផ្លូវថ្នល់។

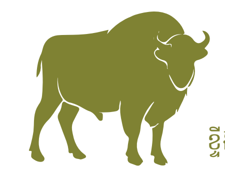
ការសន្និដ្ឋាន


សត្វរមាំង

• សត្វរមាំង ប្រើស និងខ្ទីងមានវត្តមាននៅសេសសល់តិចតួច និងបែងខ្ញែកពីគ្នា។ ការធ្លាក់ចុះយ៉ាងខ្លាំងនៃចំនួនសត្វទន្សោង និងឈ្លូស គឺជាការព្រួយបារម្ភយ៉ាងខ្លាំងដោយសារប្រភេទទាំងនេះដើរតួយ៉ាងសំខាន់នៅក្នុងប្រព័ន្ធអេកូឡូស៊ីព្រៃរបោះ។


ប្រើស

• បច្ចុប្បន្ននេះ ជីវម៉ាស់ចំណីរបស់សត្វខ្លាធំនៅក្នុងតំបន់ព្រៃទេសភាពខ្ពង់រាបខាងកើតគឺមានកម្រិតទាប និងត្រូវការជាចាំបាច់នូវ "ការគ្មានការបរបាញ់" នៅក្នុងរយៈពេលមួយទសវត្សរ៍ ដើម្បីស្តារនូវចំនួននៃចំណីសត្វខ្លានៅក្នុងកម្រិតមួយអាចប្រៀបធៀបជាមួយនឹងតំបន់នៅក្នុងប្រទេសថៃដែលជាប្រភពមានចំនួនសត្វខ្លាធំគ្រប់គ្រាន់។


ខ្ទីង

• កត្តាគំរាមកំហែងដោយសារការបរបាញ់ ជាពិសេសទម្រង់នៃការដាក់អន្ទាក់នានានៅក្នុងដែនជម្រកសត្វព្រៃទាំងពីរ គឺបង្ហាញថាការគ្រប់គ្រងតំបន់ការពារ និងការអនុវត្តច្បាប់បច្ចុប្បន្ននៅមានកម្រិតនៅឡើយ។

• តម្រូវការជាបន្ទាន់នូវផែនការយុទ្ធសាស្ត្រសម្រាប់ការអភិរក្សយ៉ាងតឹងរឹង និងកម្មវិធីស្តារពពួកសត្វក្រចកចំពាមដែលជាក់លាក់មួយ ដើម្បីបញ្ចៀសនូវការធ្លាក់ចុះនៃប្រភេទទាំងនេះ។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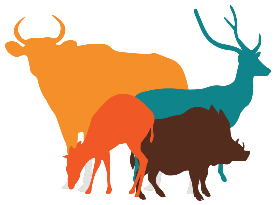
អនុសាសន៍សម្រាប់ការគ្រប់គ្រង



• វត្តមានរបស់សត្វរមាំងនិងទន្សោងនៅក្នុងតំបន់ទេសភាពខ្ពង់រាបខាងកើត គឺជាតម្លៃអភិរក្សដ៏ពិសេសវិសាលមួយដោយសារស្ថានភាពកំពុងរងគ្រោះថ្នាក់របស់ពួកវា ក៏ដូចជាសារៈសំខាន់សម្រាប់ជាប្រភេទរំពាររបស់ខ្លាធំនៅពេលយើងធ្វើការស្តារពូជនិងព្រលែងទៅក្នុងធម្មជាតិវិញ។ ដើម្បីបញ្ចៀសការផុតពូជនៃប្រភេទទាំងនេះដែលជាគន្លងនៃប្រភេទគោព្រៃ និងខ្លាធំ យើងត្រូវការយន្តការគ្រប់គ្រងយ៉ាងមានប្រសិទ្ធភាពជាបន្ទាន់ដូចជា៖

• ការបង្កើតកម្មវិធីស្តារពូជពពួកប្រភេទសត្វមានក្រចកជំពាម និងការជ្រើសរើសតំបន់ស្តារសត្វព្រៃឡើងវិញ ‘Wildlife Recovery Zone’ (WRZ) នៅក្នុងតំបន់ការពារ និងរួមបញ្ចូលទាំងការបង្កាត់ពូជផងដែរ។ ការស្តារពូជពពួកសត្វមានក្រចកជំពាមនៅក្នុងតំបន់ WRZs គួរតែការបង្កើនចំនួនមេបាបង្កាត់ពូជ ការត្រួតពិនិត្យជំងឺ ការបំប៉នចំណីអាហារ តម្រូវការទឹក និងការវាស់វែងពីការគ្រប់គ្រងនៃទីជម្រកនានា។

• ការទប់ស្កាត់ពីការបរបាញ់សត្វព្រៃ ដោយការបង្កើនការល្បាត ការបំពាក់បំប៉នឲ្យបានគ្រប់គ្រាន់ដល់មន្ត្រីឧទ្យានុរក្សនៅក្នុងដែនជម្រកសត្វព្រៃទាំងពីរ, ភ្ជាប់ជាមួយការបង្កើតសហគមន៍ល្បាត ប្រសិទ្ធភាពល្បាតនិងបំពាក់ឧបករណ៍បច្ចេកវិទ្យា ដូចជាប្រព័ន្ធត្រួត-ពិនិត្យដោយយន្តហោះដ្រូន។ WRZs ត្រូវការការពារនៅក្នុងកម្រិតខ្ពស់រួមមានប្រព័ន្ធសញ្ញាឲ្យដឹងជាមុននិងការល្បាតជុំវិញជាញឹកញាប់។ ការល្បាតទាំងនេះត្រូវធ្វើឡើងដោយការអង្កេតដោយយកចិត្តទុកដាក់ និងការផ្តន្ទាទោសជាដើម។ ប្រសិទ្ធភាពនៃការការពារ WRZs និងដែនជម្រកត្រូវការប្រភពនៃធនធានច្បាស់លាស់។

• ការពង្រឹងការអនុវត្តច្បាប់ទៅលើបទល្មើសការជួញដូរសត្វព្រៃនៅក្នុងបណ្តាលខេត្តប៉ែកខ្ពង់រាបខាងកើតនិងជាពិសេសនៅតាមបណ្តោយព្រំដែនដែលប្រើសម្រាប់ការផ្គត់ផ្គង់ និងតម្រូវការពីប្រទេសវៀតណាម។

• ការសិក្សាពីអ្នកប្រើប្រាស់ និងតម្រូវការចំពោះសាច់សត្វព្រៃ និងផលិផលសត្វព្រៃ សម្រាប់ការបង្កើតនូវផែនការយុទ្ធសាស្រ្តដ៏មានប្រសិទ្ធភាពសម្រាប់ផ្លាស់ប្តូរសង្គម និងអាកប្បកិរិយាក្នុងគោលបំណងកាត់បន្ថយការបរិសាច់សត្វព្រៃក្នុងរយៈពេលវែង។

• បន្តការសិក្សាពីបំរែបំរួលចំនួនសត្វនិងការប៉ាន់ប្រមាណចំនួនពពួកសត្វមានក្រចកជំពាមដែលជាប្រភេទសម្រាប់ការតាមដានទាំងនៅក្នុងនិងក្រៅ WRZs ។ បន្ថែមពីនេះការតាមដានអំពីកត្តាគម្រាមកំហែងសំខាន់ៗ និងប្រសិទ្ធភាពនៃយន្តការការអភិរក្ស គឺមានសារៈសំខាន់ក្នុងផ្តល់ព័ត៌មានទាន់ពេលវេលា និងការគ្រប់គ្រង។

ការផ្តល់អនុសាសន៍សម្រាប់ការគ្រប់គ្រង និងផ្តល់ដំណោះស្រាយ គឺមានសារៈសំខាន់ក្នុងការស្តារព្រៃដ៏ធំធេងនៅក្នុង តំបន់ទេសភាពខ្ពង់រាបខាងកើតដែលធ្លាប់មានប្រភេទថនិកសត្វជាច្រើនប្រភេទ។ ធនធានទាំងនេះនឹងផ្តល់មកវិញនូវសេដ្ឋកិច្ចសង្គមនិរន្តរភាពសម្រាប់សហគមន៍មូលដ្ឋាន អ្នកពាក់ព័ន្ធ និងរដ្ឋាភិបាលថ្នាក់ក្រោមជាតិនិងថ្នាក់ជាតិ។ តំបន់ទេសភាពដ៏វិសេសវិសាលនេះបាននឹងកំពុងជួបកត្តាគំរាមកំហែងនៅក្នុងកម្រិតមួយត្រូវការការពារ ស្តារឡើងវិញជាបន្ទាន់ និងជាអាទិភាព។

Herd of banteng © Fletcher & Baylis / WWF-Cambodia

EXECUTIVE SUMMARY

The Eastern Plains Landscape (EPL) in Eastern Cambodia and central Viet Nam comprises one of the largest remaining expanses of lowland Deciduous Dipterocarp Forest, a type of forest that is severely threatened. The EPL harbours globally significant populations of endangered wildlife, including the largest population of wild banteng (Bos javanicus) in its native range, critically endangered ibis and vulture species, Cambodia's largest population of Asian elephants (Elephas maximus), one of the last Siamese crocodile populations (Crocodylus siamensis), and the only remaining Indochinese leopard population (Panthera pardus delacouri) in Indochina.

A Decade of Ungulate Monitoring in the Eastern Plains Landscape
Robust monitoring of wildlife populations is critical to assess the impacts of threats to wildlife, to support evidence-based conservation decision-making, and for effective protected area management. In the Phnom Prich and Srepok Wildlife Sanctuaries (PPWS and SWS, respectively), located in the heart of the EPL, a long-term ungulate monitoring programme is being implemented, the results of the first ten-year period (2010-2020) of which are presented in this report. Using a standardized distance-sampling statistical framework, this programme assessed the population status and trends for banteng, northern red muntjac (Muntiacus vaginalis) hereinafter referred to as red muntjac, and Eurasian wild pig (Sus scrofa), all are important prey species for the diverse assemblage of large carnivores in the landscape.

Key Results

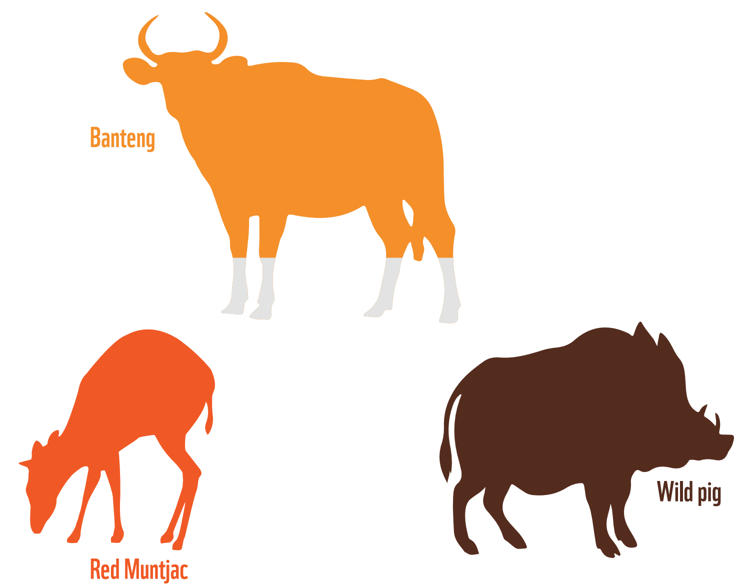
The 2010-11 baseline survey revealed that the EPL contained the largest global population of the endangered banteng in its native range: with an estimate of 1,911 individuals (95% CI: 870-2952) in SWS and 1,102 individuals (602-2,018) in PPWS. In 2020, this globally significant population has dramatically declined by about 72% compared to the 2010-11 population. The population density of the relatively resilient red muntjac has also declined in 2020 compared to the 2010-11 baseline. Wild pig is the only ungulate for which population density did not decline, but showed a fluctuating trend over time in both sites. In 2020, population density of the wild pig was lower than the baseline in SWS but similar in PPWS (Table 1).

Encounter rates for other key ungulates: Eld's deer (Rucervus eldii), gaur (Bos gaurus), and sambar (Rusa unicolor), were extremely low (ranging from zero to five encounters per survey), suggesting that only small populations of these species, all of which are globally threatened, remain.

Table 1– Density and population size, along with Standard Error (±SE) for three ungulate species in Srepok Wildlife Sanctuary and Phnom Prich Wildlife Sanctuary estimated from distance-sampling based line transect surveys (area-weighted density was used for 2010-11 to obtain density for the core zones of SWS), and the changes in density between the baseline and the latest survey year with their significance level (p-value).

Species	PA Name	Year	Density of individuals/km² ±SE	Density 95% CI range	Density change in 2020 compared to 2010-11	Population size ± SE	Population size 95% CI range
Banteng	SWS core	2010-11	1.10 ± 0.31	0.50-1.72		1911 ± 531	870-2952
	SWS core	2020	0.21 ± 0.10	0.09-0.53	-80.61% (p=0.003)	371 ± 177	150 - 918
	PPWS	2010-11	0.66 ± 0.20	0.36-1.21		1102 ± 341	602-2018
	PPWS	2020	0.29 ± 0.14	0.11-0.74	-55.99% (p=0.069)	485 ± 238	191 - 1230
Red muntjac	SWS core	2010-11	2.56 ± 0.24	2.09-3.06		4453 ± 424	3623-5283
	SWS core	2020	0.82 ± 0.16	0.55-1.22	-67.99% (p=8.000E-10)	1425 ± 284	962-2112
	PPWS	2010-11	1.52 ± 0.29	1.04-2.22		2543 ± 478	1744-3708
	PPWS	2020	1.15 ± 0.15	0.89-1.50	-24.3% (p=0.127)	1925 ± 256	1480-2504
Wild pig	SWS core	2010-11	1.91 ± 0.42	1.08-2.77		3314 ± 738	1398-4330
	SWS core	2020	1.25 ± 0.26	0.84-1.87	-34.56% (p=0.092)	2169 ± 443	1451-3240
	PPWS	2010-11	0.95 ± 0.29	0.53-1.71		1595 ± 479	891-2855
	PPWS	2020	1.11 ± 0.26	0.70-1.75	15.9% (p=0.347)	1848 ± 431	1170-2919

The ungulate densities in SWS and PPWS are much lower compared to ecologically similar landscapes in South Asia, suggesting that the ungulate populations in the EPL are currently severely depleted. The estimated tiger prey biomass of 115.7kg/km2 in 2020 is less than 5% of prey biomass density in a comparable tiger site in Southeast Asia (Huai Kha Khaeng Wildlife Sanctuary in Thailand; (Simcharoen et al., 2014)).



Increasing Trend in Threats

Historically, the EPL contained vast aggregations of large ungulate species that supported a variety of large carnivores. Decades of armed civil conflict paired with illegal hunting led to significant reductions of most large mammal populations across their range (Loucks et al., 2009) and resulted in the likely extinction of species such as the kouprey (Bos sauveli) and local extirpation of the tiger (Panthera tigris). More recently, an unprecedented poaching and snaring crisis is sweeping through Indochina. In the EPL of Cambodia, a rapid acceleration and intensification of poaching has been observed over the past decade. During the monitoring period (2010-2020), the rate at which ranger patrols on foot and motorcycle encountered and removed snares and other such lethal wildlife traps increased over a hundredfold from 0.04 traps per 100 km patrolled in 2010 to 6.46 in 2020. Increasingly devastating poaching methods such as blanket snaring, jaw traps, and live electrified wires to kill animals have been observed regularly in recent years.

This trend of increased poaching is primarily fuelled by the illegal trade in wildlife and growing demand for wild meat and other wildlife parts and products from within Cambodia and from across the border in Viet Nam. The detrimental effect of poaching on ungulates is further exacerbated by the impacts of illegal forest encroachment, and forest conversion and degradation, driven primarily by the illegal timber trade, rapid socio-economic growth, and increased access to these once remote forests due to road development.

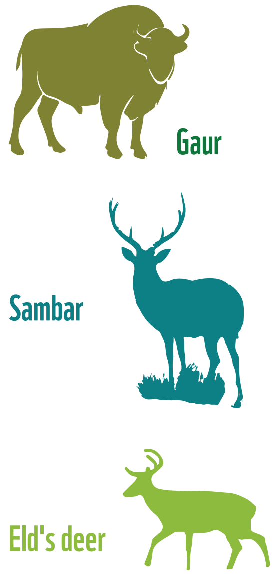
Conclusions

- The precarious existence of Eld's deer, sambar deer and gaur as small, spatially isolated populations, and the steep declines in banteng and red muntjac populations are highly concerning, as these species play vital ecological roles in the dry forest ecosystem.

- Currently, tiger-prey biomass in the EPL is very low and it would take over a decade of ideal, no-poaching conditions for the prey populations to recover naturally to the level found in a comparable site in Thailand that is currently sustaining a source tiger population.

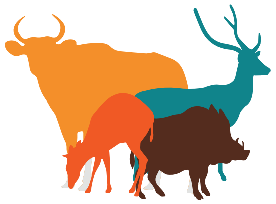
- The poaching threat, particularly in the form of snaring, in the two protected area sites in the EPL has grown by two orders of magnitude during this monitoring period, indicating that the current level of protected area management and law enforcement efforts are no longer sufficient to tackle the threats effectively.

- More intensive conservation strategies and a comprehensive ungulate recovery programme are urgently required to reverse the population trends.




Majestic male eld's deer at waterhole in Srepok Wildlife Sanctuary © Fletcher & Baylis / WWF-Cambodia

Management Recommendations



● The Eld's deer and banteng populations in the EPL are of irreplaceable conservation value due to their endangered status, and they would be important prey species for tigers when reintroduced. To prevent these species from following the same extinction path as the kouprey and the tiger in the EPL, it is crucial that effective management interventions are made immediately, including by:

● Initiating an ungulate recovery programme, concentrated in carefully selected 'Wildlife Recovery Zone' (WRZ) within the PAs, and including an integrated captive breeding component. Ungulate recovery in the WRZs, should be complemented by restocking, along with disease control, food supplementation, surface water provision and other intensive habitat management measures.

● Strengthening protection from poaching by increasing patrolling efforts, putting in place adequate ranger strength in the two PAs alongside employing community rangers, in conjunction with increasing patrolling effectiveness with state-of-the art technology such as anti-poaching drone alert systems and artificial intelligence based analytics. The WRZs require particularly high levels of protection, including fencing with alarm systems and frequent perimeter patrols to stop poachers. These patrolling efforts must be combined with careful investigations, prosecutions and convictions of poachers. Effective protection of the WRZs and the larger wildlife sanctuary area requires dedicated and long-term commitment of resources.

● Strengthening law enforcement actions on illegal wildlife trade (that drives poaching in the EPL) in the provinces around the EPL and in particular, along the border areas being used to supply the demand coming from Viet Nam.

● Consumer research on demand for wild meat and other wildlife parts should inform the development of effective social and behaviour change strategies aimed at longer-term demand reduction.

● Continuing the research on population dynamics and regular estimations of ungulate populations are essential for monitoring species recovery both inside and outside of the WRZs. In addition, more robust monitoring of key threats and the effectiveness of conservation interventions are crucial to inform timely adaptive management.

These management recommendations and proposed interventions are critical for the restoration of the large and relatively intact forests of the EPL with the historical diverse assemblage of large mammal populations. This in turn will provide the natural capital for sustainable socio-economic opportunities for local communities, other stakeholders, and the provincial and national governments. This unique natural landscape, facing severe threats to its integrity, is at a tipping point, and comprehensive protection of forests and restoration of extirpated wildlife - a 'rewilding of the landscape,' is an urgent priority.

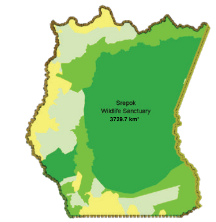
Phnom Prich Wildlife Sanctuary © Nick Cox / WWF-Cambodia

LIST OF ACRONYMS AND ABBREVIATION

AICc: Akaike Information Criteria corrected for small sample size
CDS: Conventional Distance Sampling
CI: Confidence Interval
DDF: Deciduous Dipterocarp Forest
EF: Evergreen Forest
ELC: Economic Land Concessions
EPL: Eastern Plains Landscape
GAM: Generalized Additive Model
GDANCP: Department of Nature Conservation and Protection of the Environment
HKKWS: Huai Kha Khaeng Wildlife Sanctuary in Thailand
WRZ: Wildlife Recovery Zone
IUCN: International Union for Conservation of Nature
KSWS: Keo Seima Wildlife Sanctuary
MCDS: Multiple-Covariate Distance Sampling
MDF: Mixed Deciduous Forest
MIST: Management Information System
MoE: Ministry of Environment
MPF: Mondulkiri Protected Forest (currently known as Srepok Wildlife Sanctuary)
PA: Protected Area
PPWS: Phnom Prich Wildlife Sanctuary
RGC: Royal Government of Cambodia
SE: Standard Error
SEF: Semi-evergreen Forest
SMART: Spatial Monitoring and Reporting Tool
SWS Core: An area of 1,736Km^2 within SWS where line transect surveys were conducted (see figure 2)
SWS Inner Core: A sub-area in the centre of SWS-Core comprising 460km^2 that was surveyed in 2011 and 2014 (see figure 2)
SWS Outer Core: A sub-area within SWS-Core comprising 1,276km^2 that together with SWS inner core (460km^2) forms the entire SWS-Core (1,736km^2) (see figure 2).
SWS: Srepok Wildlife Sanctuary (formerly known as Mondulkiri Protected Forest)
USAID: United States Agency for International Development.
WWF: World Wide Fund for Nature (known as World Wildlife Fund in the U.S.A.)
YDNP: Yok Don National Park

A waterhole in Srepok Wildlife Sanctuary © Arnulf Koehncke / WWF-Cambodia

1. INTRODUCTION

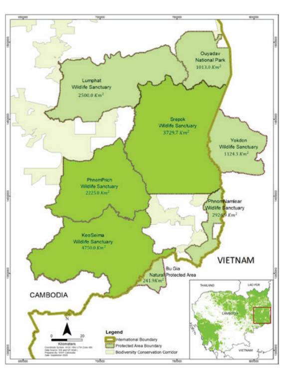


Zonation map of Srepok Wildlife Sanctuary

The transboundary Eastern Plains Landscape (EPL), a protected area (PA) complex in Eastern Cambodia, extending to Viet Nam, is comprised of approximately 14,000km2 of mosaic habitat which forms one of the largest lowland Deciduous Dipterocarp Forest (DDF) remaining on earth (Wohlfart et al., 2014). While tropical dry forests are considered the most threatened and least effectively protected major forest type globally (Janzen, 1988; Hoekstra et al., 2004), a relatively large area (~23,000km²/~35%) of the Cambodian DDF remains relatively intact and lies within the boundaries of officially protected areas (Pin et al., 2013; Wohlfart et al., 2014). These vast forests are located in a major hotspot for evolutionary history and biodiversity: 'the Indochina bioregion', with exceptionally high levels of species richness and endemism (Jenkins et al., 2013; de Bruyn et al., 2014). The EPL supports globally significant populations of critically endangered birdlife including the giant ibis (Thaumatibis gigantea) and white-shouldered ibis (Pseudibis davisoni) (Gray, Pollard, et al., 2014; Loveridge & Ty, 2015), one of the last Siamese crocodile populations remaining globally (Han et al., 2015), Cambodia's largest population of Asian elephants (Gray, Vidya, et al., 2014), the largest global population of wild banteng in its native range (Gray, Prum, et al., 2012) and the only remaining population of Indochinese leopards in Indochina (Rostro-García et al., 2016; Rasphone et al., 2019).

Zonation map of PhnomPrich Wildlife Sanctuary

The Phnom Prich Wildlife Sanctuary (PPWS) and the Srepok Wildlife Sanctuary (SWS) are located in the centre of the EPL and fall under the management of the Department of Nature Conservation and Protection of the Ministry of Environment (GDANCP/MoE) of the Royal Government of Cambodia (RGC), with technical support provided by WWF-Cambodia since 1998. The PPWS, a PA that covers 2,225km², was established by royal decree in 1993. The SWS, spanning 3,730km2, was first designated as the Mondulkiri Protected Forest (MPF) by the Ministry of Forestry, Fisheries and Agriculture in 2002, and later renamed the Srepok Wildlife Sanctuary by the GDANCP in 2016. As part of Cambodia's commitment to the global goal to doubling the wild tiger populations by the year 2022, endorsed at the 2010 International Tiger Forum in St. Petersburg (Global Tiger Initiative, 2010), the Forestry Administration produced the Cambodia Tiger Action Plan (Forestry Administration, 2011) which identified the EPL as the first priority site for tiger reintroduction.



Maps of Eastern Plains Landscape (EPL)

Historically, the north and eastern Cambodian forests were renowned gameland containing vast aggregations of large ungulates that supported a variety of large carnivores (Wharton, 1966). Decades of armed civil conflict, followed by economic growth, led to significant reductions in large-bodied mammal populations (Loucks et al., 2009), as well as the extinction of species such as the kouprey and local extirpation of tigers (Panthera tigris) (Corlett, 2010; Gray, Rattanak, et al., 2012; O'Kelly et al., 2012). Nonetheless, until a little over a decade ago, the EPL still contained some of the most intact assemblages of megafauna in the region (Gray, Prum, et al., 2012; Gray, Vidya, et al., 2014).

However, the rapid acceleration and intensification of threats has placed increasing pressure on these remaining populations (Schipper et al., 2008; Hughes, 2017). The unprecedented poaching and snaring crisis across Southeast Asia, is fuelled by an increasing urban demand for wildlife products, particularly meat (Gray et al., 2018; Scheffers et al., 2019; Heinrich et al., 2020). In the EPL, this demand originates both from within Cambodia as well as from across the border in Viet Nam (Belecky & Gray, 2020). Habitat conversion, habitat degradation, infrastructure development, porous legal frameworks and deficient law enforcement further compromised species persistence in the region (Brooks et al., 1999; Barrett et al., 2001; Sodhi et al., 2004; Gibson et al., 2011; Neef et al., 2013).

Robust monitoring of wildlife populations is critical to accurately assess the impacts of these threats on wildlife, and to support evidence-based conservation decision-making and effective PA management (Sutherland et al., 2004; Pullin & Knight, 2009; Lindenmayer et al., 2012). Due to the foresight and investment of WWF to set up a long-term monitoring programme for target species in the EPL, data now exists to allow for measurements of population change. WWF-Cambodia established a monitoring baseline for ungulate species for the SWS and PPWS in 2010-11 using line-transect based distance-sampling methods (Buckland et al. 1993; Gray, Phan et al., 2012). Ungulates were selected as target species due to their important ecological roles in the dry forest ecosystem and their essential role in tiger reintroduction in the landscape. Ungulates are key prey species for medium-sized and large carnivores and for scavengers (Hayward et al., 2012; Phearun Sum & Loveridge, 2016; Wolf & Ripple, 2016; Rostro-García et al., 2018; Kamler et al., 2020), and sufficiently high prey density is a critical condition for carnivore persistence in a landscape (Ramakrishnan et al., 1999; Rasphone et al., 2019).

Ungulates are also important seed dispersers (Corlett, 1998; Brodie, 2007; Brodie et al., 2009; Sridhara et al., 2016), and large herbivores have an important influence on vegetation composition and on habitat features such as salt licks and waterholes which in turn are critical for the survival of other endangered and critically endangered wildlife (Ahrestani et al., 2016; King et al., 2016; Eames et al., 2018; Pin et al., 2018).

The ungulate surveys were repeated in 2014, 2016, 2018, and 2020, yielding one of the longest-term and most robust ungulate monitoring datasets currently available in mainland Southeast Asia.

This report aims to:

- Present the results of the decade long ungulate monitoring programme in the EPL, with a particular focus on population status and trends of banteng, red muntjac, and wild pig;

- Discuss the conservation implications of the results including for species conservation and recovery, and tiger reintroduction to the EPL.

A recovery of the EPL to its historical potential with rich and abundant wildlife and large intact forests that support human wellbeing and economy is possible, but transformative conservation action is urgently and immediately required.

A community research ranger working side by side with a government ranger in biodiversity research © Ravy Sophearoth / WWF-Cambodia

2. METHODS

2.1 Study Area

The study was conducted in the SWS (3,729.71 Km², formerly the MPF) and the PPWS (2,225.00Km²). These two adjacent PAs are located within the centre of the EPL that consists of eight contiguous PAs. Six PAs are located in Eastern Cambodia and are connected to two PAs located in Viet Nam (see figure 1). DDF, dominated by Shorea obtusa and Dipterocarpus tuberculatus trees (Tani et al., 2007; Pin et al., 2013), makes up the majority of forest cover in the PPWS and SWS, with the remaining forest cover consisting of a mosaic of Mixed Deciduous Forest (MDF) and Semi-evergreen/Evergreen Forest (SEGF/EF) patches, interspersed with smaller patches of bamboo forest and riverine forest. The SEG/EF forest patches are slightly more abundant in the PPWS (23% and 5%, respectively) compared to the SWS (9% and 1%, respectively) (Gray, Pollard, et al., 2014). The climate is characterized by a distinct dry season (Dec-May) and a monsoon-influenced wet season (June-November).

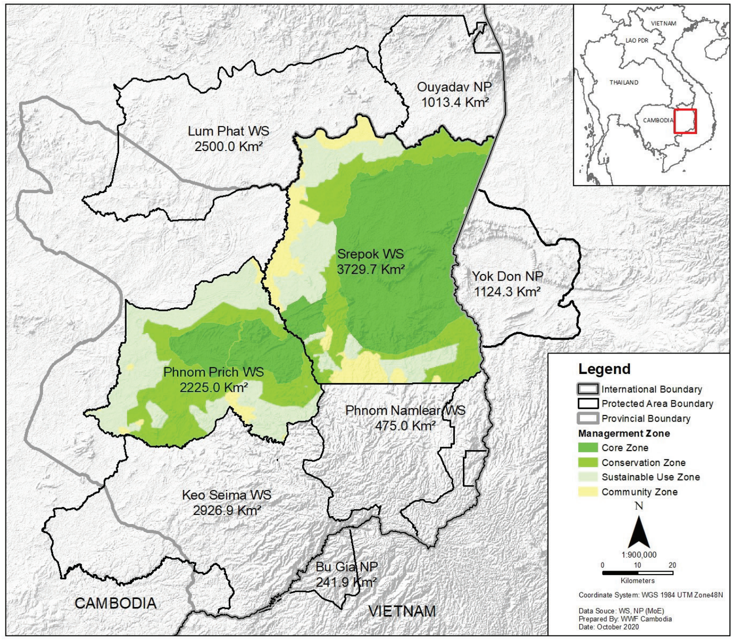


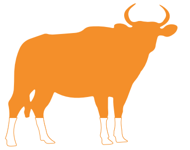
Figure 1- The Eastern Plains Landscape (EPL) PA complex includes the Srepok Wildlife Sanctuary and the Phnom Prich Wildlife Sanctuary, four other PAs in Cambodia, and two PAs in Viet Nam.

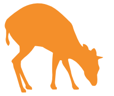
2.2 Study Species

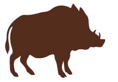
Six ungulate species were the focus of this study: Eld's deer, sambar deer, gaur, banteng, red muntjac, and wild pig. Sufficient encounters were recorded for the latter three species to allow for population size estimation. The banteng is an endangered wild cattle species (Gardner et al., 2016) that is relatively understudied across mainland Southeast Asia and whose ecological attributes may vary significantly across its' range.

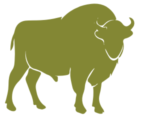
The banteng appears especially well adapted to dry and open habitats including dry dipterocarp forests, but is also known to utilise mixed deciduous and evergreen forest types in landscapes containing a mosaic of vegetation types such as the EPL (Prayurasiddhi, 1997; Steinmetz, 2004; Channa & Gray, 2010). It has a prefere nce for lower elevations (Jornburom et al., 2020). Banteng is a typical roughage feeder with a predominantly graminoid diet (Ahrestani et al., 2016). Prayurasiddhi (1997) found that mean home range size in the wet and dry season were similar (30.0±8.5km² and 26.4km², respectively) but that seasonal shifts of home range locations occur, possibly following fire and water availability patterns (Channa & Gray, 2010). Banteng usually occur in small herds (2-10 individuals), although larger herds have been observed, or as solitary males, and herd composition changes over seasons (Nowak, 1999; Steinmetz, 2004; Rahman et al., 2019). The life history of the banteng is relatively slow compared to the smaller red muntjac and wild pig. Banteng reach sexual maturity at 2-4 years old, with a gestation period of 9.5-10 months usually producing a single calf (Copland, 1974; Choquent, 1993; Rahman et al., 2019).

Whilst the red muntjac is listed as least concern in the IUCN Red List, many populations are declining in Southeast Asia (Timmins et al., 2016). This species can be considered a habitat generalist as it occurs in a wide variety of forest types, grasslands, and croplands (Laidlaw, 2000; Bali et al., 2007; Rahman et al., 2017). They have small (<1km2) home ranges and show a relatively high level of site fidelity (Odden & Wegge, 2007). Red muntjac are mixed feeders that forage on browse and grasses and select for leaf and fruit materials with low fibre content (Brodie, 2007; Ahrestani et al., 2016; Sridhara et al., 2016). They are largely solitary although breeding pairs and females with offspring have been found to associate (Barrette, 1977; Odden & Wegge, 2007). Red muntjac have a relatively fast life history, reaching the age of first fawning at 14-18 months and having a gestation period of 6-7 month in captivity (Pudjirahaju et al., 2015; Faruk Miazi et al., 2016).

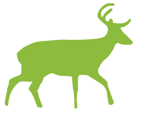

Banteng


Red Muntjac


Wild pig


Gaur


Sambar


Eld's deer

The wild pig is also listed as least concern in the IUCN Red List and is widespread and common across most of Europe and Asia (Keuling & Leus, 2019). It is a true habitat generalist and considered a resilient species with a high level of tolerance to disturbance (Allwin & Swaminathan, 2016; Sridhara et al., 2016; Phumanee et al., 2020). Wild pigs have home ranges of 6-34 km², they are omnivorous with a high degree of dietary flexibility, although their diet often includes fruits, and wild pigs may travel significant distances to follow periodic mast fruiting events (Caley, 1997; Curran & Leighton, 2000; Ballari & Barrios-García, 2014; Sridhara et al., 2016). Wild pigs are gregarious, although solitary males occur, and live in herds that vary in size according to season and locality. Wild pigs have a seasonal but high reproductive output which varies across geographic regions and with available resources (Choquenot et al., 1996; Bieber & Ruf, 2005; Keuling & Leus, 2019).

2.3 Survey Design

Line transect surveys using a distance-sampling statistical framework (Buckland et al. 1993) was the method used in the dry seasons of 2010 and 2011 to estimate ungulate population densities, which then formed the monitoring baseline for ungulates in the EPL (Gray, Phan, et al., 2012). Subsequently, line transect surveys were conducted every two years starting in 2014, with comparatively large survey efforts in 2016, 2018, and 2020 (table 2).

Table 2 - Survey effort and related details of ungulate population monitoring conducted using distance-sampling based line transect surveys in the Phnom Prich Wildlife Sanctuary and the Srepok Wildlife Sanctuary. Data from the 2010 and 2011 surveys were combined to produce a single estimate for each Wildlife Sanctuary and for each ungulate species, this will be referenced as a single "2010-11" survey in the remainder of the documents

PA Name	Area covered (km²)	Year	Survey dates	# Transects	Mean transect length (range) in km	Mean # surveys/ transect (range)	Total Km surveyed	Lead Researcher for WWF
PPWS	1,670	2010	21/03 – 08/06	33	2.3 (1.4-2.7)	2.0 (2-2)	155	Thomas Gray
PPWS	1,670	2011	02/01 – 26/05	34	3.4 (2.5-3.7)	4.1 (2-8)	467	Thomas Gray
PPWS	1,670	2014	10/02 – 30/05	34	3.0 (2.3-3.4)	5.5 (4-8)	560	Rachel Crouthers
PPWS	1,670	2016	16/01 – 28/05	56	3.0 (2.8-3.2)	6.2 (3-9)	1044	Rachel Crouthers
PPWS	1,670	2018	01/01 – 29/06	56	3.0 (2.5-3.0)	6.8 (6-9)	1130	Rachel Crouthers
PPWS	1,670	2020	10/01 – 15/06	58	3.0 (2.5-3.1)	8.0 (8-8)	1390	Milou Groenenberg
SWS	1,736	2010	03/01 – 12/06	38	3.1 (2.2-3.9)	2.3 (1-5)	273	Thomas Gray
SWS	460	2011	06/02 – 29/05	38	2.8 (1.2-3.4)	3.9 (2-5)	415	Thomas Gray
SWS	460	2014	30/01 – 29/05	38	2.9 (1.2-3.5)	5.5 (4-8)	603	Rachel Crouthers
SWS	1,736	2016	27/01 – 30/05	58	2.9 (2.3-3.2)	6.4 (2-10)	1096	Rachel Crouthers
SWS	1,736	2018	23/01 - 30/06	58	2.9 (2.2-3.3)	6.5 (6-8)	1109	Rachel Crouthers
SWS	1,736	2020	10/01 – 11/06	58	3.0 (2.2-3.2)	8.0 (8-8)	1371	Milou Groenenberg

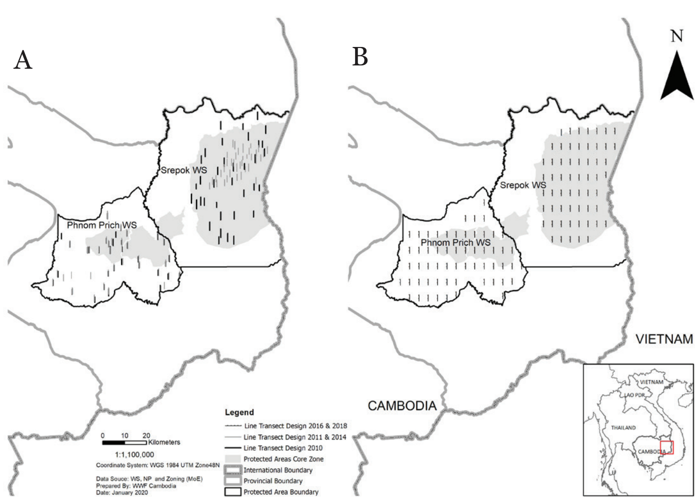
Data from the 2010 and 2011 surveys were combined to produce a single estimate for each Wildlife Sanctuary and for each ungulate species, this will be referenced as a single "2010-11" survey in the remainder of the documents

The baseline survey in 2010-11 followed a sampling design of vertical transects with a random starting point using the survey design engine in DISTANCE 6.0 (Thomas et al., 2010). This design incorporated a minimum distance of 500m between transect lines, distributed across the study area, totalling 40 transect lines in the PPWS, and 40 transects in the SWS (figure 2). Transect lines situated within Economic Land Concessions (ELCs), rubber plantations, villages, or agricultural concessions were excluded and therefore only 33-34 transects in the PPWS and 38 transects in the SWS were surveyed in the 2010-11 and 2014 surveys. The survey area in the PPWS covered 1,670km² and excluded a 2km buffer around village centres and a 500m buffer along permanent rivers (Gray, Phan, et al., 2012). Transect lines that overlapped or were less than 500m apart were excluded from the 2014 surveys. The design remained the same over the 2010 and 2011 survey years with the exception that the average transect length was extended by approximately 1.1km in 2011 (3.4) compared to 2010 (2.3) (table 2). In the SWS, a core area of 1,736km², excluding 500m buffer zones along the main rivers to allow safe and continuous access to the transect lines, was surveyed in 2010. In 2011, a sub-area within this core area consisting of 460km² was surveyed, with the aim of producing key tiger prey species density estimates in an area that was at the time considered 'likely to be designated as an inviolate tiger recovery zone' (Gray, Phan et al., 2012).

Mr. Nurth Thurt, community research ranger, using a range finder to measure the distance to an animal © Ravy Sophearoth / WWF-Cambodia

Species-specific encounter rates in the 2010 and 2011 surveys were combined in the data analysis, thus improving the global detection function for the PPWS and SWS. Area-specific analyses were performed in three survey areas: 1) the PPWS; 2) the SWS outer core (1,276km^2); and 3) the SWS inner core (460km^2). These area classifications should not be confused with the recently officially designated core zones of the SWS zonation plan (RGC, 2019). In 2014, the same design as in 2011 was used (figure 2A). In 2016, the survey design was modified from a random design to a systematic random design, with a random starting point with parallel positioning of lines in a grid-based system with 4km spacing using DISTANCE 6.2 (figure 2B). The rationale behind this change was to permit a representative sampling effort across the entirety of both survey areas, covering the wide diversity of habitat types with random probability, whilst simultaneously reducing the likelihood of autocorrelation and whilst gathering data on presence of additional species in both the PAs. In addition, the survey effort was increased as a response to low encounter rates and to obtain sufficient detections to produce ungulate density estimates. The number of transects increased from 34 to 56 in the PPWS, and 38 to 58 in the SWS, with a higher frequency at which each transect line was surveyed (mean of 6.2 in the PPWS, and 6.4 in the SWS), resulting in an increased survey effort (1,044km in the PPWS, and 1,096km in the SWS; table 2). In 2018 and 2020, the same transect lines were surveyed with two additional lines in PPWS in 2020.

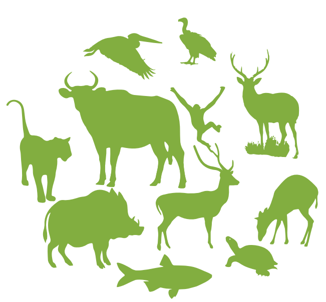
Figure 2 - The locations of transects used for the baseline surveys in 2010-2011, also used in 2014 (A), and the locations of transects used in 2016, 2018, 2020 (B).



Mrs. Sacorn Sochea, one of the only two local women to serve as community research rangers in the Eastern Plains Landscape. © Ravy Sophearoth / WWF-Cambodia

2.4 Field Methods

Distance sampling comprises a suite of methods in which animal density or abundance is estimated from a sample of perpendicular distances to detected animals or animal clusters (Buckland et al., 2001). This method accounts for imperfect detectability and assumes a declining probability of detection with increasing distance between the animal and the observer, for example due to reduced visibility by obscuring vegetation. In the EPL surveys two observers walked each transect line slowly and quietly in the early morning (06:00- 10:00) and late afternoon prior to sunset (15:00-18:30). Upon observing any ungulate, observers recorded the species, distance to the centre of the group (measured with a laser range finder), and the compass bearing. In addition, surveyors recorded the number of individuals (cluster size) and sex-age classes, where possible. Trip specifics such as date, time, and bearing of line transects were also recorded along with covariates such as observer name, weather conditions, and habitat type. Prior to commencing the surveys, all transects were marked with red spray paint, thus ensuring consistency and repeatability throughout the survey period. In addition, all team members underwent rigorous training to minimize the level of observer variability.



Georeferenced data on snaring and other lethal traps in both PAs were obtained from the Management Information System (MIST: http://www.ecostats.com/MIST) in 2010-2012 and the Spatial Monitoring and Reporting Tool (SMART: https://smartconservationtools.org/) reports between 2012 and 2018. Patrolled distances were obtained from the geolocations (GPS waypoints and track logs) of PA ranger team leaders, they include both foot and motorcycle patrols. The MIST and SMART data collection protocols include the recording of all observed illegal activities, their type, nature of the offense, offender details (e.g. nationality, number of people involved, occupation, sex), information on equipment or materials used (e.g. vehicle type, materials used for hunting or logging), action taken by the patrol team (e.g. confiscation, warning letter, arrest), and details of any illegal products if encountered (e.g. volume of wood, species of wildlife).

2.5 Analytical Methods

Analytical methods followed a standardized distance sampling approach based on Buckland, et al. (1993, 2001). Prior to modelling, all data was checked for anomalies, errors removed, and right truncated to prevent the inclusion of additional adjustment terms and to improve model fit (Buckland, et al., 2001; Thomas et al., 2010).

The datasets were analysed using Conventional Distance Sampling (CDS) in software DISTANCE 6.0 (Thomas et al., 2010) for the 2010-2011 surveys, in DISTANCE 6.2 for the 2014 surveys, in DISTANCE 7.2 for the 2016 and 2018 surveys, and in R with the Distance package version 1.0.1. (Miller et al., 2019) for the 2020 survey. Analysis performed using DISTANCE and R calculated ungulate encounter rates, detection probability, cluster size, and abundance (population size and density) estimates. These estimates were independently calculated for each species.

For the 2010-11 data analysis, ungulate densities were estimated for the species having a minimum of 50 encounters. Encounter data per species from all survey areas and both survey years was pooled to create a single global detection function that was fitted per species and used to calculate landscape-wide and area-specific densities for the three individual survey areas (the PPWS, the SWS outer core, and the SWS inner core). For the 2014 data analysis, species specific encounter data from the PPWS and SWS were pooled together for the 2010-11 and 2014 surveys due to the low number of encounters. Species specific global detection functions were derived and area specific densities per survey year were produced for both the SWS and PPWS. Species specific encounters for red muntjac (2016, 2018, and 2020) and wild pig (2016, 2018) were pooled across both PAs for each distinct survey year, and PA specific densities were produced. In 2020, wild pig encounters in PPWS were used for an area-specific detection functions, whilst for SWS encounters for 2018 and 2020 were used for a pooled detection function to ensure a sufficiently large sample size. Due to the lower number of banteng in encounters recorded within a single survey year, global detection functions were derived using detection data from the particular survey year and the previous survey year(s). Thus the 2016 global detection function incorporated encounter data from the 2014 and 2016 surveys, the 2018 global detection function incorporated encounter data from the 2016 and 2018 surveys, and the 2020 global detection function incorporated encounter data from the 2016, 2018, and 2020 surveys. These global detection functions were subsequently used to estimate area specific densities per year for the PPWS and SWS. This approach assumes similar detection probability across survey years (see discussion).

Expected cluster size was estimated through a regression of 'log cluster size' against the estimated probability of detection. Whenever the regression equation was non-significant due to size bias (Drummer & McDonald, 1987), the mean observed cluster size was used instead (Buckland, et al., 2001; Gray, Phan, et al., 2012; Gray, Prum, et al., 2012).

The best model for estimating group and individual species densities was selected based on Akaike Information Criteria corrected for small sample size (AICc) (Buckland et al., 2015a). Cramer von Mises Goodness-of-Fit scores and Kolmogorov-Smirnov tests, as well as a visual assessment of fit to the models were also considered during analysis stages (Buckland et al., 2015b, p. 5).

To allow for comparability of population density estimates in SWS, the area-weighted average density and its variance was calculated for the SWS core for 2010-11 following stratified analysis as per Buckland et al. (2001, p. 89 - 91; further details in Annex 1). Statistical significance of changes in mean densities between years or survey areas was calculated with a one-tailed Z-test (Buckland, et al., 2001).

To understand the implications of our results for a potential future tiger reintroduction to the EPL, biomass density of the tiger prey species were estimated. The biomass densities were estimated based on the earliest (2010-11) and most recent (2020) estimated densities of the banteng and wild pig and on body weights extracted from Simcharoen et al. (2018) which account for sex-age related biases in predation by tigers (Karanth & Sunquist, 1995). Red muntjac was not included in these calculations as this species generally makes up a negligible proportion of tiger diets, both in terms of relative biomass as well as total animals killed (Kumaraguru et al., 2011; Hayward et al., 2012; Simcharoen et al., 2018). Wild cattle species, including banteng and gaur, are considered important tiger prey species that make up an significant proportion of dietary intake (Andheria et al., 2007; Hayward et al., 2012; Kumaraguru et al., 2011; Simcharoen et al., 2018). Wild pig, weighing <60kg, is not necessarily a preferred prey species but has been identified as an important prey item for some sites (Hayward et al., 2012; Mukherjee & Sen Sarkar, 2013; Chestin et al., 2017), whilst only making up a small proportion of tiger diet in others (Kumaraguru et al., 2011; Simcharoen et al., 2018). Due to their higher relative abundance, it is reasonable to assume that wild pig would be an important tiger prey item in the EPL (Steinmetz et al., 2020).

Multi-year data from the MIST and SMART database of encountered snares, corrected for patrol effort, was used as an indicator for the intensity of snaring in PPWS and SWS. The patrolled areas overlap almost entirely with the line transect survey area, although in SWS some patrols also occurred outside the core zone not covered by the line transects.

Herd of wild pig in Phnom Prich Wildlife Sanctuary © Fletcher & Baylis / WWF-Cambodia

3. RESULTS

3.1 Encounter Rates

Encounter rate details for all ungulate species are available in **Annex 2.** Despite a significant increase in survey effort between the baseline year (2010-11) and subsequent years, the total number of encounters of ungulates in the PPWS and SWS combined were relatively similar in 2010-11 (325), 2016 (382), 2018 (329), and 2020 (315). The encounter rate (defined as the number of encounters per 10 km of transect walked) in the PPWS and SWS combined for all ungulate species declined between the baseline and the most recent survey. Encounter rates for three ungulate species were extremely low in all the survey years: Eld's deer (range: 0-5); gaur (range: 0-5); and sambar (range: 1-5), and were insufficient to estimate population densities.

3.2 Population Density and Size

Annex 3 contains detailed information on the density and population estimates for the three species with sufficient encounters: banteng, red muntjac, and wild pig. Table 3 and Figures 3-5 show the trends in population density of banteng, red muntjac and wild pig, respectively, over the monitoring period.

Table 3 – Estimated mean density along with Standard Error (±SE) and 95% Confidence Intervals (CI) for three ungulate species in the Srepok Wildlife Sanctuary and the Phnom Prich Wildlife Sanctuary based on distance-sampling based line transect surveys (area-weighted density was used for 2010-11 to obtain density for the core zone of the SWS). Also shown are changes in density between survey years with their significance level (p-value). P-values<0.05 are marked in blue.

Species	PA Name	Year	Density of individuals/km² (±SE)	Density 95% CI range	Density change compared to 2010-11	Density change compared to	Density change compared to 2016	Density change compared to 2018
Banteng	SWS core	2010-11	1.10 ± 0.31	0.50-1.72	-	NA	-	-
	SWS inner core	2014	2.28 ± 0.53	1.44-3.61	NA	NA	NA	NA
	SWS core	2016	0.39 ± 0.13	0.20-0.75	-64.62% (p=0.016)	NA	-	-
	SWS core	2018	0.25 ± 0.12	0.10-0.61	-77.32% (p=0.005)	NA	-35.91% (p=0.212)	-
	SWS core	2020	0.21 ± 0.10	0.09-0.53	-80.61% (p=0.003)	NA	-45.2% (p=0.145)	-14.49% (p=0.407)
	PPWS	2010-11	0.66 ± 0.20	0.36-1.21	-	-	-	-
	PPWS	2014	1.27 ± 0.38	0.70-2.29	92.69% (p=0.079)	-	-	-
	PPWS	2016	1.39 ± 0.29	0.93-2.09	110.69% (p=0.019)	9.34% (p=0.40	-	-
	PPWS	2018	0.35 ± 0.14	0.17-0.76	-46.37% (p=0.108)	-72.17% (p=0.(	-74.54% (p=0.001)	-
	PPWS	2020	0.29 ± 0.14	0.11-0.74	-55.99% (p=0.069)	-77.16% (p=0.(	-79.11% (p=3.109E-04)	-17.95% (p=0.375)
Red muntjac	SWS core	2010-11	2.56 ± 0.24	2.09-3.06	-	NA	-	-
	SWS inner core	2014	2.09 ± 0.34	1.50-2.90	NA	NA	NA	NA
	SWS core	2016	1.94 ± 0.28	1.46-2.59	-24.17% (p=0.048)	NA	-	-
	SWS core	2018	1.22 ± 0.17	0.92-1.60	-52.61% (p=0.000)	NA	-37.51% (p=0.013)	-
	SWS core	2020	0.82 ± 0.16	0.55-1.22	-67.99% (p=8.000E-10)	NA	-57.79% (p=2.813E-04)	-32.45% (p=0.046)
	PPWS	2010-11	1.52 ± 0.29	1.04-2.22	-	-	-	-
	PPWS	2014	1.69± 0.26	1.23-2.31	10.72% (p=0.337)	-	-	-
	PPWS	2016	2.04 ± 0.28	1.55-2.69	34.03% (p=0.100)	21.05% (p=0.1	-	-
	PPWS	2018	1.17 ± 0.15	0.90-1.50	-23.5% (p=0.134)	-30.91% (p=0.(	-42.93% (p=0.003)	-
	PPWS	2020	1.15 ± 0.15	0.89-1.50	-24.3% (p=0.127)	-31.63% (p=0.(	-43.52% (p=0.003)	-1.04% (p=0.478)
Wild pig	SWS core	2010-11	1.91 ± 0.42	1.08-2.77	-	NA	-	-
	SWS inner core	2014	6.52 ± 1.95	3.64-11.68	NA	NA	NA	NA
	SWS core	2016	2.94 ± 0.61	1.96-4.42	54.25% (p=0.082)	NA	-	-
	SWS core	2018	1.90 ± 0.33	1.35-2.67	-0.4% (p=0.494)	NA	-35.43% (p=0.066)	-
	SWS core	2020	1.25 ± 0.26	0.84-1.87	-34.56% (p=0.092)	NA	-57.57% (p=0.005)	-34.3% (p=0.059)
	PPWS	2010-11	0.95 ± 0.29	0.53-1.71	-	-	-	-
	PPWS	2014	2.21 ± 0.63	1.26-3.88	131.85% (p=0.035)	-	-	-
	PPWS	2016	1.06 ± 0.17	0.77-1.46	10.87% (p=0.378)	-52.18% (p=0.(	-	-
	PPWS	2018	2.44 ± 0.46	1.68-3.54	155.45% (p=0.003)	10.18% (p=0.3	130.4% (p=0.003)	-
	PPWS	2020	1.11 ± 0.26	0.70-1.75	15.9% (p=0.347)	-50.01% (p=0.(	4.54% (p=0.438)	-54.63% (p=0.006)

*2014 estimates in the SWS were for the inner core area only and are therefore not directly comparable to the estimates in other years which covered the entire core area, these are indicated as "NA".

An aerial view of Phnom Prich Wildlife Sanctuary © Chhit Samarth / WWF-Cambodia

Estimated banteng density (Figure 3) was initially higher in SWS than in PPWS (although not statistically significantly at the 5% significance level, due to high variance in the estimates). However, in 2016, density estimates are significantly lower in SWS than in PPWS (p=0.001), and no significant difference was seen in 2018 and 2020.

In the SWS-Core, the banteng mean density estimate showed a consistent declining trend. In 2016, the density estimate declined by 64.62% from the 2010-11 baseline density estimate; in 2018 it indicated a further decline compared to the baseline (77.32%); and finally in 2020 the decline compared to the baseline was 80.61 (Table 3). In PPWS, the mean estimated banteng density increased between the 2010-11 baseline and the two subsequent surveys, with the increase from 2010-11 to 2016 being statistically significant. However, in 2018, banteng density saw a steep and significant decline by 74.54% and 72.17%, compared to the 2016 and 2014 surveys, respectively. The decline appeared to continue between 2018 and 2020 albeit at a lower rate (17.95%) and the change was not statistically significant.

The current estimated population size of banteng is 371 individuals (95% CI: 150 - 918) in SWS core and 485 individuals (95%CI: 191 – 1,230) in PPWS compared to the 2010-11 baseline estimates of 1,911 individuals (95%CI: 870-2,952) in SWS core and 1,102 individuals (95%CI: 602-2018) in PPWS.

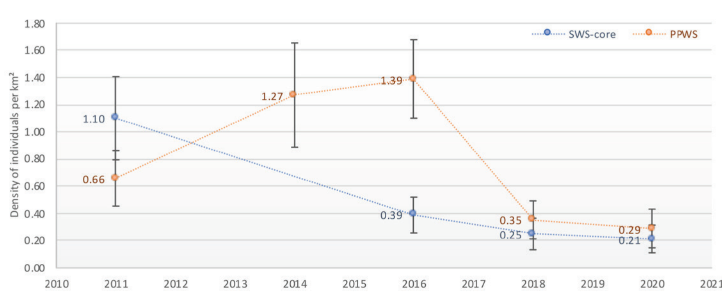


Figure 3 - Banteng population density estimates (±SE) in Srepok Wildlife Sanctuary (blue circles) and Phnom Prich Wildlife Sanctuary (orange circles) based on distance-sampling based line transect surveys (area-weighted density was used for 2010-11 to obtain density for the core zone of SWS).

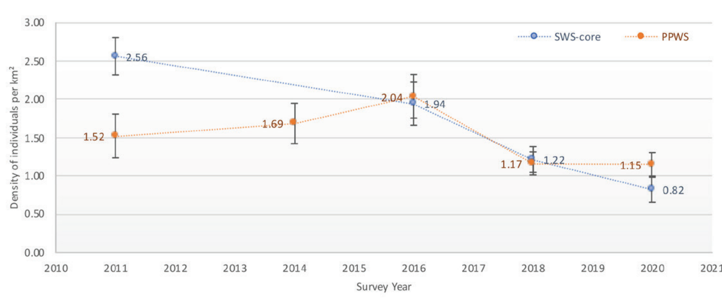
Herd of banteng captured by camera trap © GDANCP / WWF-Cambodia

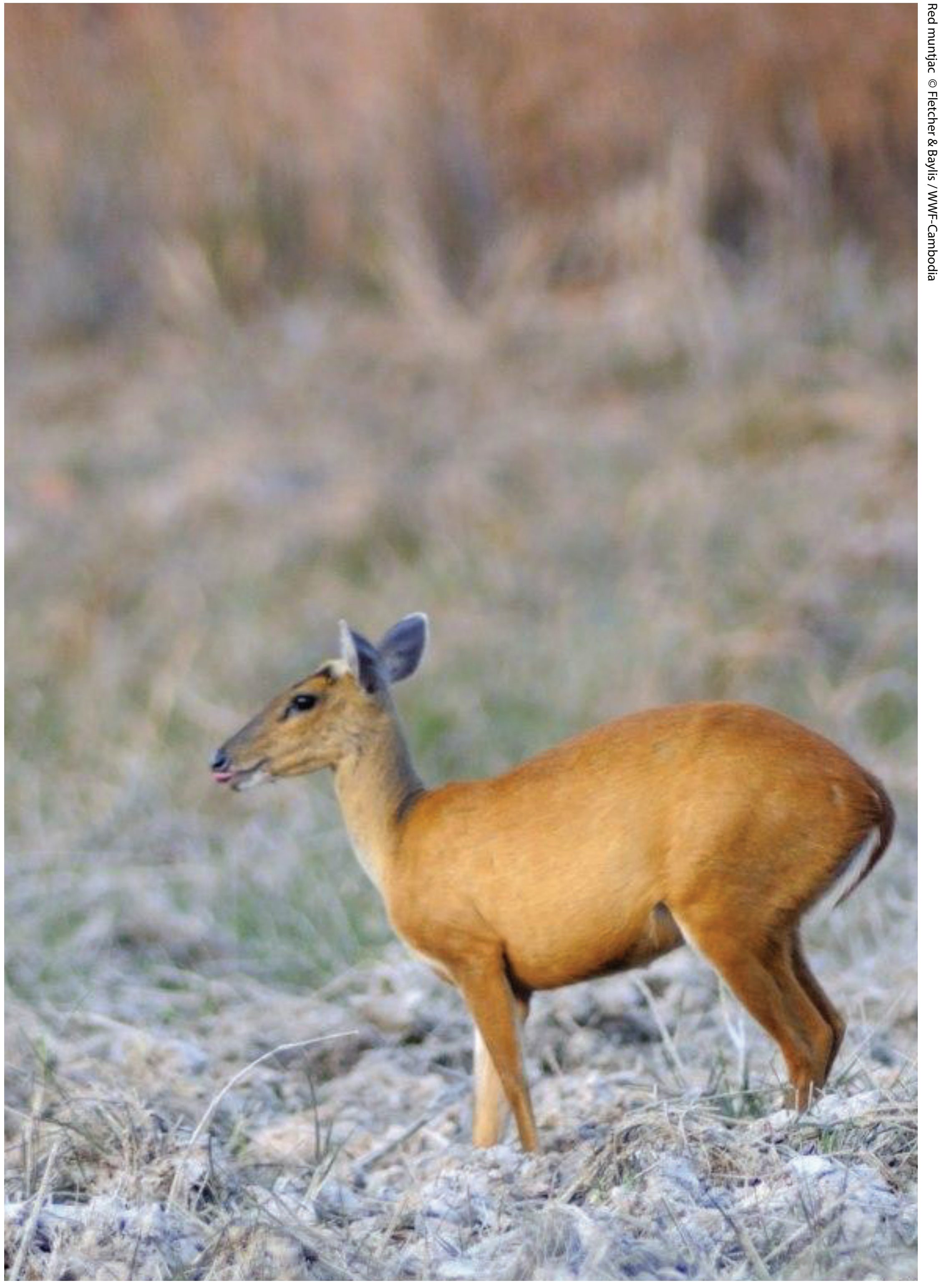
Red muntjac density (Figure 4) was 40.62% higher in the SWS compared to the PPWS during the baseline survey. However, since 2016, red muntjac density estimates remained similar in the two PAs until 2020 when the estimate in PPWS exceeded the estimate in SWS (although not statistically significant).

Red muntjac density in SWS-Core declined significantly from the 2010-11 baseline survey to the 2016 survey (24.17%), the 2018 survey (52.61%), and the 2020 survey (67.99%) (Table 3). In PPWS, red muntjac density followed a similar trend as the banteng: the population appeared to be slightly increasing between the 2010-11 baseline and the two subsequent survey years (albeit not significantly), but declined by 42.93% in 2018 compared to 2016, after which it appeared to remain stable. Overall, the decrease in red muntjac density between 2020 and 2010-11 baseline was 24.30% in PPWS (although not statistically significant).

The current red muntjac population is estimated at 1,425 individuals (95%CI: 962-2,112) in SWS core and 1,925 individuals (95%CI: 1,480-2,504) in PPWS, compared to the 2010-11 baseline estimates of 4,453 individuals (95%CI: 3,623-5,283) in SWS core and 2,543 individuals (95%CI: 1,744-3,708) in PPWS.

Figure 4- Red Muntjac population density estimates (±SE) in Srepok Wildlife Sanctuary (blue circles) and Phnom Prich Wildlife Sanctuary (orange circles) based on distance-sampling based line transect surveys (area-weighted density was used for 2010-11 to obtain density for the core zones of SWS).





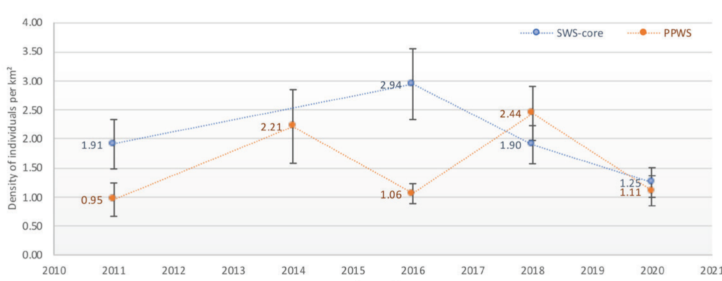
Red muntjac © Fletcher & Baylis / WWF-Cambodia

Wild pig density was initially lower in PPWS compared to SWS and this difference was significant in 2016 (64.40%, p=0.001). However, in 2018 wild pig density became higher in the PPWS compared to the SWS, although not statistically significant, and in 2020 the estimates were at similar levels (Figure 5).

Significant fluctuations in wild pig density estimates for both PAs occurred over the years (Figure 5). However, when comparing the 'dips' and 'peaks' of these fluctuations, no statistically significant difference ($p < 0.05$) was evident. Results suggest that wild pig density in SWS-Core remains at a similar level from the 2010-11 baseline and the most recent survey in 2020 (Table 3). Densities appear to increase between 2010-11 and 2016, and then decline further in 2018 and 2020. In PPWS, wild pig densities significantly increased between the baseline survey (2010-11) and the subsequent survey (2014). However, densities in 2016 decreased by 52.18% from 2014 estimates, followed by a significant increase of 130.4% in 2018 compared to 2016, and again a decrease of 54.63% in 2020 compared to 2018.

The current estimated population size of wild pig is 2,169 individuals (95% CI: 1,451-3,240) in SWS core and 1,848 individuals (1,170-2,919) in PPWS, compared to the baseline in 2010-11 estimates of 3,314 individuals (95% CI: 1,868-4,760) in SWS core and 1,595 individuals (95%CI: 891-2,855) in PPWS.

Figure 5. Wild Pig population density estimates (±SE) in Srepok Wildlife Sanctuary (blue circles) and Phnom Prich Wildlife Sanctuary (orange circles) based on distance-sampling based line transect surveys (area-weighted density was used for 2010-11 to obtain density for the core zones of SWS).



Wild pig © Fletcher & Baylis / WWF-Cambodia

3.3 Prey Biomass

The tiger prey biomass density (wild pig and banteng combined1) is currently slightly higher in PPWS (124.3Kg/Km2), compared to SWS (107.5 Kg/Km2), whereas in 2010-11 the tiger prey biomass density was lower in PPWS (224.7 Kg/Km2), compared to SWS (386.6 Kg/Km2) (Table 4). On average, the area-weighted mean prey biomass density decreased by 62.3% over the past decade, and the proportional representation of wild pig total biomass increased from 18.3% to 43.0% in SWS, and from 15.7% to 33.0% in PPWS.

Table 4- Biomass density of tiger prey species in Srepok Wildlife Sanctuary and Phnom Prich Wildlife Sanctuary based on the most recent (2020) density estimates compared to the estimates in 2010-11. average body weights of banteng (287kg), and wild pig (37kg) are extracted from Simcharoen et al., (2018).

		2010-11		2020		
PA Name	Species	Mean Prey Biomass of individuals/km²	Proportion of total Prey Biomass	Mean Prey Biomass of individuals/km²	Proportion of total Prey Biomass	Biomass density change in 2020 compared to 2010-11
SWS	Banteng	316,0	81,7%	61,3	57,0%	-80,6%
	Wild pig	70,6	18,3%	46,2	43,0%	-34,6%
	Total Prey	386,6		107,5		-72,2%
PPWS	Banteng	189,4	84,3%	83,3	67,0%	-56,0%
	Wild pig	35,3	15,7%	41,0	33,0%	15,9%
	Total Prey	224,7		124,3		-44,7%
Area-weighted	Banteng	253,9	82,6%	72,1	62,3%	-71,6%
	Wild pig	53,3	17,4%	43,6	37,7%	-18,2%
	Total Prey	307,2		115,7		-62,3%

3.4 Poaching in the PPWS and SWS

The detection rate of lethal wildlife traps, including snares (defined as nooses made of twisted cables, metal wires, or ropes, with one end anchored to the ground or tied to a plant or pole), demonstrated a steep increasing trend of poaching effort between 2010 and 2016 (Figure 6). Results from both PAs highlight a peak in trap detection rates in 2016, followed by subsequent annual fluctuations in drops and peaks in the period 2017-2019. On average the detection rate and absolute number of lethal traps are much lower in the first half of the decade compared to the second half.

The number of lethal traps removed, corrected for patrol effort, were similar across both PAs until 2015, whereas in 2016-2018 more traps were recorded and removed per unit patrolled distance in PPWS compared to SWS. Absolute snare numbers rose faster in SWS compared to PPWS, and with the exception of 2017, there are consistently more snares encountered in SWS compared to PPWS (mean difference 141%, range 42%-283%).

1. *Based on the characteristic of preferred tiger prey and observed species composition of tiger diets (Avinandan et al., 2008; Hayward et al., 2012; Simcharoen et al., 2014), it cannot be expected that wild pig would form the majority of tiger prey biomass, and thus the table likely represents an overestimate of ecologically plausible prey availability.*

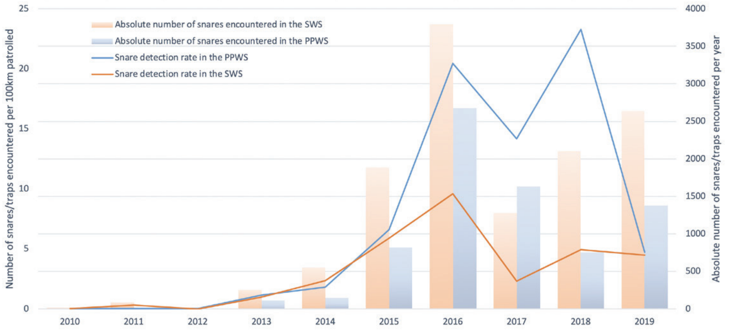


Figure 6- Absolute numbers of snares or such lethal traps encountered and removed on annual basis (bars) and traps removed per 100Km patrolled on foot and motorcycle (lines) by protected area rangers in Phnom Prich Wildlife Sanctuary (blue) and Srepok Wildlife Sanctuary (orange)between 2010 and 2019, based on WWF-Cambodia's MIST and SMART data. For PPWS, there exists a data gap in 2017 due to a temporary absence of SMART reporting in this year.

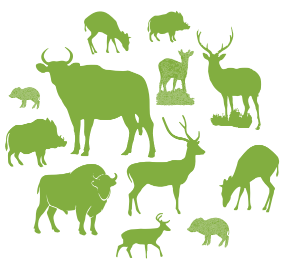
On average, 1,786 and 1,075 lethal traps have been removed per year between 2013 and 2019 in SWS and PPWS, respectively, totalling 20,026 traps over this time period for both PAs combined. In addition, electrified wires used to kill animals have been recorded and removed in the SWS since 2013, and in the PPWS since 2017. Between 2013 and 2019, 3,716 meters of electrified wire were removed from the SWS, and in 2017 to 2019, 1,421 meters from PPWS (Annex 4).

Srepok river © Merril Halley / WWF-Cambodia

4. DISCUSSION

4.1 Ungulate Population Status and Trends

Current ungulate densities in the SWS and PPWS are at very low levels, considering the potential carrying capacity of the landscape compared to similar landscapes in South Asia (Annex 5). Current ungulates densities are similar or slightly higher when compared with two other PAs within the EPL for which density figures are available: the Keo Seima Wildlife Sanctuary (KSWS) and the Yok Don National Park (YDNP) (see figure 1 and Annex 5).



Results comparisons with the most recent available surveys of neighbouring PAs (2018/19) indicate that the red muntjac densities are significantly higher in the PPWS and the SWS compared to the YDNP and the KSWS, and wild pig densities are significantly higher in the PPWS, the SWS, and the YDNP compared to the KSWS (Annex 5). Banteng densities in both the SWS and PPWS are significantly higher than in the KSWS. The most recent banteng population estimate for the KSWS (in 2018) is 13 individuals (95%CI: 2.4-73), compared to 433 (95%CI: 178-1,055) in the SWS and 591 (95%CI: 276-1,264) in the PPWS in the same year. Similar to the PPWS and SWS, banteng population estimates for the KSWS appear to have decreased from 2014 (492 individuals (95%CI: 225-1,075)) to 2018 (13 (95% CI 2.4 - 73)) (WCS, 2019, unpublished results). However, these results have wide confidence intervals and limited statistical robustness, due to low encounter rates per survey year. Consequently, data pooling to create a global detection function incorporated all encounters over six survey years between 2010-2018 (WCS, 2019, unpublished results). In the YDNP, there were no visual observations of banteng during their 2018/19 surveys, although signs are still sporadically encountered during patrols (WWF-Viet Nam, 2019). Consequently, mean estimated density comparisons will be limited to two ungulate species (red muntjac and wild pig) for the YDNP. These comparisons highlight the importance of the PPWS and the SWS within the larger landscape for banteng conservation.

Only a few baseline estimates and peer reviewed studies on ungulate densities exist for Southeast Asia, due to the relatively high level of resources (human and financial) required to conduct robust distance-based line transect surveys and the dense rainforest habitat across much of the tiger range in this region. In general, ungulate densities in South Asia are much higher compared to Southeast Asia (Annex 5). Recent surveys in Nepal indicate that the total ungulate density in at least three different tiger reserves is of an order of magnitude higher than those observed in the PPWS and SWS (DNPWC & DFSC, 2018).

Similar high densities are observed in Indian tiger reserves, where figures are generally higher than 30 individuals/km2 (Karanth & Nichols, 2002) and can even exceed 100 individuals/km^2 in some areas (see for example Sankar et al., 2010).

Seasonal dry forest types, comprising the majority of forest cover in the EPL, are considered to have high carrying capacity for large herbivore species due to high levels of food availability (Eisenberg & Seidensticker, 1976; Smith et al., 2011). Indeed, the semi-dry deciduous forests of Ranthambore National Park and Gir National Park in India have estimated ungulate densities of 75 and 52.8 individuals/km^2 respectively (Khan et al., 1996; Bagchi et al., 2003; Jathanna et al., 2003), suggesting that ungulate populations have been severely depleted in the EPL.

Low encounter rates of gaur, sambar, and Eld's deer during our line transect surveys may suggest that only small isolated populations remain. This is supported by other research findings in the area including: low camera trapping rates of large deer and gaur (<20 individuals captured per 100 trap nights, J. Kamler May 2019 pers. comm. and Gray, 2011, 2012); low numbers of direct and indirect sightings of these species during other research surveys (R. Crouthers, July 2019 pers. comm.) and law enforcement patrols (SMART data 2010-2019); low volumes of Eld's deer dung samples collected during a targeted non-invasive Eld's deer DNA survey (R. Crouthers, July 2019, pers. comm.); and the absence of these species in leopard diet, determined by scat analysis using genetics (Rostro-García et al., 2018).

Historical and current poaching of sambar and Eld's deer combined with their low recovery rates are considered the primary causes of decline in Cambodia (Weiler, 2003; Loucks et al., 2009). In addition, it has been suggested that larger sexual dimorphic deer species may be less resilient to and/or slower to recover from male-targeted trophy hunting due to the negative reproductive and infant survivability consequences of skewed sex-ratios (Steinmetz et al., 2010; Gray, Phan, et al., 2012). The apparent low number of both sambar and Eld's deer across the two PAs are of international concern, as it appears that no large intact populations of both these species remain in the Indochinese region (Weiler, 2003; Timmins et al., 2015). Furthermore, the globally endangered Eld's deer only exists within a few spatially fragmented subpopulations in Laos, Cambodia, India, and Hainan in China and is therefore of global conservation concern (Gray et al., 2015). Thus it is crucial that direct management intervention steps are taken to protect and restore this globally important population.

Gaur may have a lower detection probability compared to banteng because: they prefer and forage longer in mixed deciduous and semi-evergreen forest types than in DDF forest types, where visibility is lower due to denser undergrowth (Prayurasiddhi, 1997; Gray, 2011); and more importantly they are less abundant and thus fewer individuals are available for detection (Royle & Nichols, 2003; Gray, 2012). Whilst gaur are generalist feeders that use a diversity of habitat types and elevations (Sankar et al., 2013), banteng is a DDF habitat selector with a higher tolerance for lowland dry conditions (Prayurasiddhi, 1997; Steinmetz, 2004; Gray, 2012), thus PPWS and SWS provide an optimal banteng habitat. The relative rarity of denser forest may result in naturally lower densities of gaur in the PPWS and SWS, this would be consistent with low gaur densities in other areas that consists of similar habitat coverage and composition, such as the Xe Pian Protected Area in Laos (Steinmetz, 2004). Nonetheless, the extremely low number of recorded encounters, which were five or less (Annex 2), suggest that only a few small spatially fragmented groups may remain. If the species would have been at a carrying capacity population level based upon habitat availability, a higher number of encounters and observations of large groups would be expected. This notion is further supported by results from other research studies (see above) and suggest that the resident gaur population size may be beyond the possibility of natural recovery.

Overall, our results suggest that populations of both wild cattle species (gaur and banteng) that occur in the EPL are severely depleted by poaching. The significant and steep declines of banteng densities in both the SWS and PPWS highlight the vulnerability of this species to current threats. The current banteng population constitutes of only 28.4% of the estimated population size in 2010-11 (Annex 3). Although red muntjac densities were higher in the PPWS and the SWS compared to the neighbouring KSWS and YDNP in 2018, the ongoing significant and steep population decline(s) observed over the past two years, particularly in SWS, may soon eliminate that difference. As the rate of population decline can be expected to decrease with declining population size, more sophisticated modelling approaches are required to discern if the decreasing decline rate in the period between 2018 and 2020 compared to the period between 2016 and 2018, is attributable to management interventions or not. Small clumped populations can be expected to have lower capture rates in snares compared to larger, more widely and uniformly distributed populations.

In contrast, our survey results demonstrate that the wild pig population trends strongly fluctuated over the years. Fluctuation in wild pig population estimates is a typical pattern observed for this species, which exploits pulsed resources (Bieber & Ruf, 2005; Cutini et al., 2013). Wild pig population size, reproduction, and habitat use is generally dependent on patchy spatial-temporal food distribution and can rely on super-annual mast fruiting events, including those of the Dipterocarpaceae sp. trees (Curran & Leighton, 2000). However, based on wild pig annual growth rates in areas without hunting pressures (see for example Steinmetz et al., 2010), we would have expected a greater increase in average density and population size estimates across the PPWS and SWS if hunting levels would have been lower, especially considering the low predation pressure. The relatively stable wild pig population demonstrates their relative resilience to hunting, as also found in other studies (e.g. Allwin & Swaminathan, 2016; Phumanee et al., 2020)

Research team scouting target site near a dry riverbed © Ravy Sophearoth / WWF-Cambodia

4.2 Prey Biomass for Tiger Reintroduction

The current estimated mean tiger prey biomass of 107.5kg/km² in the SWS, of 124.3kg/km² in the PPWS, and the area-weighted mean of 115.7kg/km² for both PAs combined, demonstrate dramatic declines compared to the estimates in 2010-11 (table 4).

The decline in prey biomass density is relatively larger than the decline in prey population density because the proportional representation of the larger banteng relative to the smaller wild pig has decreased in the past decade. The relatively high proportion of wild pig of total prey biomass (43.0% in SWS and 33.0% in PPWS) also indicates that the estimated tiger prey biomass in 2020 most likely represents an overestimate of ecologically realistic prey availability because this proportion is much higher than commonly observed in tiger prey diets (Avinandan et al., 2008; Hayward et al., 2012; Simcharoen et al., 2014). When inferring predator carrying capacity from available prey biomass, the exclusive use of preferred prey items yielded more robust results (Hayward et al., 2007; Kiffner et al., 2009), and tigers tend to prefer larger ungulate species weighing between 60 and 250kg (Hayward et al., 2012). Although the high relative abundance of wild pig could render this species an important prey item, it would still be expected that the vast majority of prey would be constituted by preferred prey species, including large cervids and younger bovids (Karanth & Sunquist, 1995; Avinandan et al., 2008; Simcharoen et al., 2014). Banteng would form a major tiger diet component in the EPL, in addition to sambar and similar sized deer, which have been identified as a preferred tiger prey species (Andheria et al., 2007; Kumaraguru et al., 2011; Hayward et al., 2012; Zhang et al., 2013; Simcharoen et al., 2018). Previous research suggests that declines of such important prey species have contributed to declines in tiger populations, even whilst leopards have continued to persist (Ramakrishnan et al., 1999). Therefore, the observed declines in banteng and the extremely low encounter rates of large deer species suggest a serious limitation in natural prey availability.

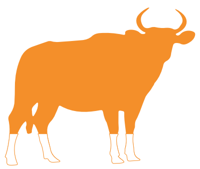
If prey biomass density for the EPL were to be modelled on the ecologically similar Huai Kha Khaeng Wildlife Sanctuary (HKKWS) in Thailand, which stands at 2,511kg/km2 (Simcharoen et al., 2014), the current prey biomass densities in the SWS and PPWS only reaches 4.3% and 4.9%, respectively, of this target. To estimate the unaided population recovery potential of the two currently viable prey species (banteng and wild pig) in the PPWS and SWS, we applied the intrinsic population growth suggested by Steinmetz et al. (2010) for gaur (0.31) and wild pig (0.14) for a context in which poaching ceased, and found that it would take eleven years to increase from the current biomass density to a level similar to that observed in the HKKWS (2,652 kg/km2).

However, a detailed Population Viability Analysis would be required to account for complex and confounding factors influencing recovery, for example density-dependent behavioural changes, reproductive rates, natural predation and human threat levels (Brook et al., 2000; Beissinger & McCullough, 2002). Tiger reintroduction in the EPL, therefore, relies on 'restocking', i.e. supplementing wild ungulate populations by releasing animals that are captive-bred or caught from elsewhere in the wild. This strategy can have additional conservation benefits for the resident prey populations and the ecosystem as a whole if managed appropriately (Moreno et al., 2004; Champagnon et al., 2012). Ungulate supplementation measures have proven successful in other countries, such as for gaur in Central India (Sankar et al., 2013), and for Eld's deer in China (Pan et al., 2014).

The draft Cambodia Tiger Reintroduction Plan proposes a stepwise approach where an intensively managed area is secured and fenced for prey recovery and an initial release of 3 tigers when prey numbers exceed 1,500 adults, primarily comprised of banteng, Eld's deer, sambar, and wild pig (possibly supplemented by domestic Asian water buffalo (Bubalus bubalis). After the initial release, prey populations would continue to increase and, given the vast areas of relative intact DDF and MDF, the carrying capacity for these species would be high enough to accommodate restocking and recovery of 46 tigers in the SWS over a period of 15 years (Harihar et al., 2018), and to potentially hold 385 tigers in the entire EPL over the longer term (Gray et al., 2017). Independent of the importance of a potential future tiger reintroduction, the recovery of these threatened ungulate species and the enhanced protection of their habitats, constitute critical conservation objectives in their own right.

An aerial view of Srepok river © Thou Sothean / WWF-Cambodia

4.3 Declines in the Largest Remaining Banteng Population



Banteng

The ongoing decline in the banteng population in the EPL is particularly concerning as the landscape supports the largest global population of this endangered species in its' native range, giving this population an irreplaceable conservation value (Gray, Prum, et al., 2012; Gardner et al., 2016). Remaining banteng populations in Indonesia, Thailand, Malaysia (Sabah), Laos, Myanmar, and Vietnam are small (rarely exceeding 50 individuals), fragmented, and declining (Gardner et al., 2016). A significant population, consisting of approximately 6,000 pure-bred banteng (i.e. no introgression with other domesticated Bos species) currently occurs in Garig Gunak Barlu National Park, Australia (Bradshaw et al., 2006). However, this population is considered a different subspecies (Bos j. javanicus) from the mainland Southeast Asian subspecies (Bos j. birmanicus). In addition, there is a high likelihood that this population is genetically homogenous, as the initial source population consisted of only 20 domesticated individuals from Bali in 1849 (Hassanin & Ropiquet, 2007; Ropiquet et al., 2008; Gardner et al., 2016). Banteng have important ecosystem functions as key prey species for large carnivores and scavengers and as ecosystem engineers (Simcharoen et al., 2014; Phearun Sum & Loveridge, 2016; Rostro-García et al., 2018; Pin et al., 2018). With an estimated population size of 856 individuals, the EPL banteng population is still the largest remaining population in its native range and therefore, it is vital to reverse the current decline in order to conserve this globally significant population.

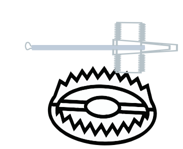
4.4 Poaching and Ungulate Declines

Current ungulate density levels are far below the intrinsic carrying capacity of DDF and this is likely caused by high historical hunting levels further exacerbated by an observed increase in snaring and poaching across the landscape in recent years. Large mammals with slower life histories, including the banteng, can be expected to be among the first species to respond to such pressures (Scheffers et al., 2019). However, the declines in red muntjac cause additional concern, as this species is generally considered to be a resilient generalist species with relatively high recovery rates from disturbances; thus any decline would indicate intense levels of poaching (Steinmetz et al., 2010; Timmins et al., 2016). The red muntjac plays an important role in the EPL ecosystem as a preferred prey species of the remnant Indochinese leopard population, a sub-species recently uplisted to critically endangered status (Rostro-García et al., 2019), as well as of other carnivore species such as the endangered dhole (Cuon alpinus) (Brodie, 2007; Brodie et al., 2009; Rostro-García et al., 2018; Kamler et al., 2020).The observed population declines of banteng and red muntjac are most likely a result of the poaching and snaring crisis that is currently sweeping through Southeast Asia (Belecky, & Gray, 2020; Gray et al., 2018; Tilker et al., 2019), fuelled by wild meat consumption and the illegal trade in wildlife and wildlife parts, which disproportionately targets large-bodied mammals (Sodhi et al., 2004; Schipper et al., 2008; Ripple et al., 2015; Scheffers et al., 2019). In Southeast Asia, wire/rope snares that are likely to trap most ground living animals are predominant, and they are cheap to construct and can be placed with little investment and planning (Belecky, & Gray, 2020).

Evidence from SMART data supports the idea that poaching is a rapidly intensifying threat in the EPL, as the numbers of snares encountered during patrols increased significantly over the last decade. Furthermore, particularly damaging hunting techniques, such as blanket snaring and live electrified wire snaring, have an additional negative impact and have been observed in the SWS since 2014, and in the PPWS since 2017. Marescot et al. (2019) used multispecies dynamic site-occupancy models for EPL SMART data from 2013-2016, to demonstrate steady and dramatic declines in the proportion of spatial coverage by wildlife species matched by a substantial increase of spatial coverage by poachers.

It should be noted that SMART records of snares are not an exact index for the actual snare density on the ground as the detection rate will vary with patrolling strategy and coverage, as well as skills and motivation of the patrolling team (Keane et al., 2008; Gavin et al., 2010; O'Kelly et al., 2018). For example, the apparent snaring peak in 2016 can be explained by an intensive targeted snare sweeping strategy implemented by the MoE and WWF in that year, whereas the apparent decline in 2017 may be explained by a subsequent increased focus on illegal logging compared to snaring. Nonetheless, it is evident that snaring forms a real threat to ground-dwelling mammals in the EPL, especially considering that the reported number of snares is a vast underrepresentation of actual snares in the landscape as only a small proportion of PAs are patrolled and 67-80% of snares may remain undetected (O'Kelly et al., 2018; Belecky, & Gray, 2020; Ibbett et al., 2020).

Armed civil conflict paired with illegal hunting



Illegal Poaching & Snaring Crisis

Although the number of snares per patrol effort is higher in the PPWS than in the SWS according to SMART data, the absolute numbers of encountered snares in the PPWS is lower compared to the SWS in almost all years (Annex 4). The earlier declines in banteng and red muntjac observed in the SWS compared to the PPWS, may be explained by the higher number of snares and the earlier intensification of poaching in the SWS compared to the PPWS. The SWS is located on an international border, and has a flatter, more open, and accessible terrain that may have enabled higher poaching influx. Anecdotal evidence suggests that poaching with firearms may also more prevalent in SWS than PPWS (J.P. Lourens, June 2020, pers. comm.). Infrastructure developments in Mondulkiri province, such as the paving in 2016 of National Route 76 which cuts through the SWS further increased accessibility to this formerly remote area, have inadvertently led to increased habitat loss and fragmentation, as well as ease of access for poachers and illegal loggers, not to mention ease of transportation of illegal wildlife products, timber and NTFPs (Clements et al., 2014; WWF-Cambodia, 2017). Tree-felling driven by cross-border illegal timber trade peaked in the PPWS after the SWS for these same reasons, with the expansion of ELCs, sawmills, and timber extractions rising rapidly between 2012-2016 in the PPWS (WWF-Cambodia, 2016; EIA, 2018). Between 2008 and 2016, almost 50,000 hectares of land located inside and along the boundaries of the PPWS was assigned to ELCs (PD-MAFF, 2019). It has been suggested that these ELCs acted as gateways for logging (targeted at high-value timber) and illegal timber movements (EIA, 2018). In addition, forest conversion and infrastructural developments within ELC boundaries may have further intensified levels of human disturbance. Anecdotal reports suggest that illegal hunting and logging intensified within and near ELC boundaries, which in turn resulted in wildlife losing previous areas of their habitat. The combination of these anthropogenic factors has likely forced wildlife into the less disturbed forested areas inside PA boundaries (R. Crouthers, April 2020, pers. comm.). Logging has been demonstrated to accelerate poaching, both because of opportunistic wild meat consumption and trade by loggers, but also because of improved access through logging road networks (Wilkie et al., 2000; Clements et al., 2014).

Recent missions by the National Committee for Prevention and Crackdown on National Resources of the RGC to combat illegal timber trade across the borders in the eastern part of Cambodia, including the EPL (ODC, 2019), may partially explain the reduced rate of decrease in banteng and red muntjac observed in the period between 2018 and 2020 compared to the period between 2016-2018. The cancellation of licenses for several large ELCs since 2016, as well as an increased number of boots on the ground through recruitment of community rangers from 2017 (SWS) and from 2018 (PPWS) onward, and improved patrolling strategies support artificial intelligence in 2019 (SWS) and 2020 (PPWS) may further explain this observation. However, further strengthening of law enforcement and protected area management will be required to fully bend the curve for these ungulate populations.

The relative stability of the wild pig population is perhaps unsurprising due to high fecundity levels and the adaptable and opportunistic diet of this resilient generalist species (Bieber & Ruf, 2005; Ballari & Barrios-García, 2014; Allwin & Swaminathan, 2016). Wild pigs have been documented to exhibit explosive population growth in similar ecosystems where natural predators have been removed and food availability is abundant (Ickes, 2001). Considering the high reproductive rate, and the extirpation of the pig's main predator (the tiger) and low abundance of other natural predators such as leopards and dhole (Gray, Rattanak, et al., 2012; Rostro-García et al., 2016, 2018), it could be expected that the population trend would be fluctuating but with an overall increase over an 8-year term. However, this is not the case in the EPL as the differences between 'dips' and 'peaks' are not statistically significant (although such comparison is complicated by the wide variance due to the highly variable cluster sizes and encounter rates). Given the declines in other ungulate species and the high proportion (~40%) of wild pig among all carcasses (deaths due to unnatural causes) encountered on SMART patrols 2010-2019, poaching is undoubtedly a contributing factor in limiting population growth.


Sunrise over Srepok Wildlife Sanctuary © WWF-Cambodia

4.5 Limitations of the Estimation Methods

Distance sampling is a widely applied robust methodology for estimating ungulate population sizes (Seber, 1992; Karanth & Nichols, 2002). The method accounts for imperfect detectability, and when backed by adequate survey design, data quality control, rigorous training and thorough execution of protocol, the key assumptions of the method can be met, leading to reliable population estimates (Buckland, et al., 2001).

There is one key design assumption and three key model assumptions of the distance sampling method (Buckland, et al., 2001):

1.) Animals are distributed independently of the transect lines; this assumption holds in this study as a suitably randomised survey design was used, and thus animals are expected to be located at any given distance from the line with equal likelihood.

2.) Objects on the transects are detected with certainty; this assumption holds in this study as medium-large sized ungulates are generally available for detection and relatively easily detected. Bias caused by missed detections on the line is generally of greater concern for species that can be hidden from sight, for example submerged cetaceans or animals in burrows, or when species are more easily missed, for example smaller well-camou flaged animals (Buckland et al., 2001).

3.) Distance measurements are exact; this assumption holds in this study as highly accurate distance measurements are obtained using laser range finders and all observers are extensively trained in the appropriate use of this technical equipment.

4.) Objects are detected at their initial location; this assumption is met when observers record the original location of the animal prior to any evasive movement and when the movement of animals independent of the observer is slow relative to the observer. In 2010, the data indicated responsive movements to observers, moving away from the line prior to detection which could have led to an underestimation in population estimates (Gray et al., 2011). Improved training techniques and rigorous data quality control from 2011 onwards reduced this constraint. Potential bias caused by independent animal movements that are proportionally faster than observer speed can lead to overestimations in population size, although bias caused by responsive movement is generally greater (Glennie et al., 2015). Ungulate species may be considered slower moving due to foraging, resting, and vigilance behaviours (Wong et al., 2019), for example Prayurasiddhi et al. (1997) estimated a mean length of daily movement by banteng of 2.5±0.3km, which is less than the ~3km transect length that is normally traversed by observers in less than 2 hours. In addition, for species with movement constrained to lie within smaller home ranges, such as the red muntjac (Sukmasuang, 2001), the independent movement bias is further reduced (Buckland et al. 2001). Finally, all observations of animals overtaking observers were excluded from the datasets to avoid double-counting animals within the same unit of sampling effort (Buckland et al., 2001).

Given that the most critical assumptions of distance sampling are met in this study, any other minor assumptions are generally of little consequence (Buckland et al., 2001). The analytical variance of the population estimates comprises of three components: (1) uncertainty in the estimates of the detection function parameters, (2) variability in encounter rate between transects, and (3) variability in the cluster size of encountered animals (Buckland et al., 2001, Fewster et al., 2009; Thomas et al., 2010). Commonly the largest component of the variance is in encounter rate variability, as was the case for banteng and red muntjac encounters in this study for all years. Spatially-explicit modelling approaches could be considered as they can explain this type of between-transect variation and therefore may reduce variance in estimates considerably (Miller et al., 2013). For wild pigs, uncertainty in density estimation was caused, approximately equally, by variation in encounter rates and in cluster size for most years. Future studies may consider estimating cluster densities rather than individual densities for this species to compensate for the high variability of cluster size within and between survey years.

For reliable estimation of detection functions there should be a minimum number of ~60 encounters of individuals or clusters of animals (Buckland, et al., 2001). To meet this critical requirement, encounter and distance data from the PPWS and SWS and/or from subsequent survey years were combined to create a pooled detection function (see methods section). Consequently, density estimates can then be calculated by stratum (in this case by PA), although bias could occur if detectability differs greatly between the PAs and/or years (Buckland et al., 2015b). The PPWS has a slightly hillier terrain and a larger proportion of denser forest types compared to the SWS (Gray, Pollard, et al., 2014), and these conditions could lead to lower detectability due to visual obstructions. An overestimation of detection probability would lead to an underestimation of population size and density. To validate the assumption that detection probability in the PPWS and SWS is similar, comparisons between detection probabilities were made between transects grouped per proportion of habitat type and forest cover.

No major differences were found, although small sample size and lack of high resolution habitat data would have impacted these results (R. Crouthers, November 2019, pers. comm.).

Pooling data for detection functions across years may be problematic if detection probability changes over time, as may be expected if vegetation characteristics change or if high and increasing hunting pressure leads to antipredator behavioural adaptations and/or changes in activity patterns (Croes et al., 2007; Solberg et al., 2010; Fragoso et al., 2016). For example, if an increase in gun hunting between subsequent survey years would have resulted in reduced detectability due to wary behaviour, using a global detection function may result in slightly inflated detection probability leading to a possible underestimation in the population estimates. However, there have been no major forest clearances in the line transect areas in the study period, natural predation pressure has actually declined, and hunting avoidance behaviour and nocturnal/crepuscular movement patterns have been predominant for at least 10 years (R. Crouthers, July 2019, pers. comm.). In addition, animals have been heavily hunted for decades in the EPL (Loucks et al., 2009), SMART records of gun hunting incidents are at similar levels in the study period, and behavioural and avoidance adaptations will have little effect against the more recent snaring threat (Gray et al., 2018).

In 2020, Multiple-Covariate Distance Sampling (MCDS) models that model the scale parameter of the detection function as a function of covariates were tested (Buckland, 2004). Including the year or area as a covariate did not improve model fit for any species, further confirming the validity of using pooled detection functions in this study. Spatial models for distance sampling data provide an alternative approach to explore the relation between environmental covariates and animal abundance (Buckland, 2004; Miller et al., 2013). For example, the detection functions created for the CDS/MCDS analysis can also be used in a density surface modelling approach (Buckland, 2004; Hedley & Buckland, 2004) including Generalized Additive Models (GAM) (Wood, 2017). The benefit of these types of models is that they can predict spatial distribution patterns over a smaller subset of the surveyed area without relying on stratification as in CDS, or over a larger or different area then originally sampled (Miller et al., 2013), and that these can be visualised on maps as a powerful communication tool to inform management. GAM can also be used to develop smooth functions to assess temporal trends (Buckland, 2004). As these methods are rapidly evolving and becoming more readily available to field practitioners, they open up an interesting avenue for future exploration of this study's dataset.

An aerial view of Phnom Prich Wildlife Sanctuary © Thou Sothean / WWF-Cambodia

4.6 Management Recommendations

The steady decline in the globally most significant population of the endangered banteng and of the comparatively resilient red muntjac are highly concerning and appear to be reflective of an intensifying poaching crisis. The EPL is following a commonly observed defaunation trajectory where larger and rarer species disappear first followed by progressively smaller and more common species and eventually leading to an 'empty forest', with profound negative impact on ecosystem services and human wellbeing (Redford, 1992; Belecky, & Gray, 2020). To reverse this worrying trend, and to rewild the 'Serengeti of Asia', intensive conservation actions are urgently required.

Current patrolling effort is inadequate in terms of spatial coverage and time allocation (Marescot et al., 2019). Furthermore, the current number of rangers per square kilometre (0.82/100km2) is lower than the optimal ranger density of 2-10/100km2 (Henson et al., 2016) and far below those observed in tiger reserves in Nepal (12.5-107.29 /100km^2) and India (12.5-127.44/100km2) (Walston et al., 2010; Harihar et al., 2018). Rangers can be more effectively deployed if smaller areas within the PAs are designated as WRZs with particularly high levels of protection, including fencing, with alarm systems and frequent perimeter patrols for maintenance and to stop poachers. These zones could act as 'safe havens' for wildlife recovery and augmentation. To strengthen the patrolling and enforcement of PA law and to foster cooperation and support for biodiversity conservation, community members, including members of Community Protected Areas, should be fully involved in protection efforts. Social and Environmental Impact Assessments and Environmental and Social Safeguards will guide responsible implementation of the WRZs. Effective protection of the WRZs and the larger wildlife sanctuary areas requires dedicated and long-term commitment of resources.

However, even increasing patrolling effort and patrolling effectiveness through infrastructure, through patrol team supplementation with community members, through implementation of state-of-the art technology, and through high quality training standards, will not be fully effective in reducing poaching pressures in the EPL, unless it is combined with careful investigations, effective prosecution and convictions of the poachers (O'Kelly et al., 2018; Gray et al., 2018; Marescot et al., 2019). Simultaneously, strengthening law enforcement actions on illegal wildlife trade (that drives the poaching in the EPL) in the provinces around the EPL, and in particular, along the border areas being used to supply the demand coming from Viet Nam, will be essential. In addition, in-depth consumer research into local and regional wild meat consumption and use of wildlife parts should inform the development of legislative change, and effective behaviour change strategies and targeted outreach aimed at consumer demand reduction (Drury, 2011; Challender & MacMillan, 2014; Steinmetz et al., 2014).

To prevent the rare and rapidly disappearing ungulate species in the EPL from following the extinction path of the tiger, the kouprey, and the wild water buffalo in Cambodia, a comprehensive ungulate recovery programme should be initiated urgently. The extremely low encounter rates with gaur, Eld's deer and sambar, indicate that natural recovery of these species may no longer be an option even in the unlikely scenario where poaching is completely eliminated. For example, Phumanee et al. (2020) demonstrated worryingly low population recovery rates for red muntjac, sambar deer, and gaur in two PAs in western Thailand despite a tripled patrolling effort and demonstrably low poaching levels in the five-year study period and attribute this to Allee effects that occur when density levels have dropped below a critical threshold (Allee et al., 1949). Therefore, the proposed WRZ should also include restocking (i.e. augmentation) and an integrated captive breeding component that incorporates assisted reproduction where needed. Recovery and breeding in the WRZs should be supported through disease control, food supplementation, surface water provision and other intensive habitat management such as the creation of artificial waterholes and salt licks. The WRZs will allow for the recovery of source populations to eventually help rewild the entire landscape and provide the enabling conditions for the initial release of tigers in the next 5 – 10 years.

Continuing the research on population dynamics and the regular estimations of ungulate populations are essential for monitoring prey recovery both inside and outside the WRZs. In addition, more robust monitoring of key threats and the effectiveness of conservation interventions are required to inform timely adaptive management.

These management recommendations and proposed interventions will be critical to ensure the survival of two of Asia's most threatened species: banteng and Eld's deer, and to avoid a rapid wildlife depletion trajectory towards becoming an empty forest. They are also required if the prestigious tiger reintroduction programme, planned by the Royal Government of Cambodia, is to become a reality. The large and relatively intact forests of the EPL can be restored with the historical diverse assemblage of large mammal populations that, in turn, will provide natural capital for sustainable socio-economic opportunities for local communities, other stakeholders, and the provincial and national governments. This natural landscape, facing severe threats to its integrity, is at a tipping point, and comprehensive protection of forests and restoration of wildlife - a 'rewilding of the landscape' - is an urgent priority.

ACKNOWLEDGEMENTS

This long-term ungulate monitoring programme was made possible with funding from WWF-Belgium, WWF-Germany, WWF-Sweden, WWF-Switzerland, WWF-US, USAID, and Humanscale. Surveys in the former Mondulkiri Protected Forest (2010/11/14) were conducted with the permission and support of the Forestry Administration of the Ministry of Agriculture, Forestry and Fisheries, in particular from His Excellency Cheng Kimsun, Men Phymean, Song Keang, Chheat Sophal and Keo Sopheak and Long Boung. Surveys in the SWS (2016/18/20) and in the PPWS were conducted with permission and support of the Ministry of Environment and support from His Excellency Chay Samith, Sanrang Dy Vicheth, Keo Sopheak, Han Sakhan, Dim Bunthoeun, Prum Sovanna, and Pet Phaktra.

This monitoring programme and initial design was established by Thomas Gray, who also performed all the ungulate density and other analysis for the 2010-11 data. Subsequent survey designs were developed by Rachel Crouthers and the implementation and analyses were conducted by Rachel Crouthers (2014-2018), and Milou Groenenberg (2020). None of this research and long-term data would have been possible without their hard work and dedication, as well as that of research officers Khaev Oudom (2010-11), Ing Seangnirithy (2010-11), Pin Chanrattana (2010-2014), Prum Sovanna (2010-2016), In Visattha (2016), Chhit Vichea PDoE (2016), Mang Tukla (2018), Kim Marin (2020), Tropin Bun (2020), Sothea Bun (2020), and counterpart Chan Sereivatanak (2020. A huge thank you to all the following staff who have acted as field team leaders: Lean Nhor (2010-2018); Van Sanny (2010-2016) ; Sary Tre, Mai Samorn, Son Nin (2010-2020); Chan Touy (2010-2016); Thea Chanthorn, Mony Samart, Mai Kosal, Hong Dae, and Chhok Kroeurl (2016-2020, SWS), and Soli Mech, and Ien Khve (2020). Last but not least, we would like to thank all of the field researchers, which due to the large survey team size are listed in Annex 6.

A warm thanks to Craig Bruce, Pin Chanrattana, Nick Cox, Keith Metzner, Amy Malling, Pem, Maly, Michelle Owen, Seng Teak, Robert Solar, Thibault Ledecq, Chhit Sam Ath, Mark Dubois, Moul Phath, Mark Drew, Phurba Lhendup, and Phalla Mey for providing logistical and administrative support, and to Hannah O'Kelly, Bivash Pandav, Abishek Harihar, Paul Doggerty, Shannon Barber-Meyer and Barney Long for assisting in the initial study design, establishing survey objectives and for providing statistical advice. Finally, we would like to thank Thomas Gray, James Burton, Robert Steinmetz, Julie Louvrier, and Andrew Tilker for their technical reviews of this support.

Trunks of trees of the Lagerstroemia species © WWF-Cambodia

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ANNEX 1: FORMULAS FOR AREA-WEIGHTED DENSITY CALCULATIONS

The area-weighted mean density and its' variance was calculated following stratified analysis as per Buckland et al. (2001c p. 89 - 91):

$$D = \frac{\sum A_v D_v}{A}$$

Where D is the area-weighted mean density, A is the total survey area, and v is the indication of a geographic stratum, so that Av is the stratum-specific survey area and D_V is the stratum-specific survey density (i.e. SWS inner core and SWS outer core).

With Variance calculated with the following formula:

$$var\,(D) = D^2 * \left\{ \frac{var\,(M)}{M^2} + \frac{var\,[(h(0)]}{[h(0)]^2} \right\}$$

Where: $$M = \frac{\sum A_v M_v}{A}$$

And: $$M_v = (n_v * S_v)/K_v$$

And: $$var\,(M) = \frac{\sum A^2_v * var\,(Mv)}{A^2}$$

With: $$var\,(M_v) = M^2_v * \left\{ \frac{var\,(n_v)}{n^2_v} + \frac{var\,(s_v)}{s^2_v} \right\}$$

Where n is the sample size, s the mean cluster size, and k the number or replicate transect lines.

Confidence intervals were calculated using the following formulas extracted from S.T. Buckland et al. (2001c p. 76 - 78):

$$D \pm z_\alpha * \sqrt{var\,(D)}$$

Where z_α is the upper α point of the N(0,1) distribution (z_α is $z_{0.025}$ = 1.96 for a 95% confidence interval).

ANNEX 2: UNGULATE ENCOUNTER RATES

Species Name	IUCN Status*	2010/11 # Encounters	2010/11 Encounter rate	2014** # Encounters	2014** Encounter rate	2016 # Encounters	2016 Encounter rate	2018 # Encounters	2018 Encounter rate	2020 # Encounters	2020 Encounter rate	Percentage change from 2010-11 to 2020
Banteng	EN	63	0.480	44	0.38	47	0.220	20	0.089	27	0.098	-79.6%
Eld's deer	EN	2	0.020	5	0.04	1	0.005	1	0.004	0	n/a	-
Gaur	VU	3	0.020	5	0.04	0	n/a	1	0.004	1	0.004	-81.9%
Sambar	VU	5	0.040	1	0.01	5	0.023	4	0.018	3	0.011	-72.8%
Wild Pig	LC	54	0.410	58	0.50	109	0.509	130	0.581	106	0.384	-6.3%
Red Muntjac	LC	198	1.510	115	0.99	220	1.028	173	0.773	178	0.645	-57.3%
All ungulates combined		325		228		382		329		315		

*IUCN Red List status, 2020 (https://www.iucnredlist.org/)

**PPWS & SWS inner core

Number of clusters (i.e. individuals or groups of animals with a well-defined location for the group centre) and encounter rate of clusters (clusters per 10km surveyed) for Phnom Prich Wildlife Sanctuary and Srepok Wildlife Sanctuary combined, for each distance-sampling based line transect survey (prior to truncation).

ANNEX 3: DENSITY AND POPULATION ESTIMATES

stimates of individual densities and population sizes, detection probability, and mean cluster sizes with standard error (SE) and 95% confidence interval (CI), for banteng, red muntjac, and wild pig in Phnom Prich Wildlife Sanctuary and Srepok Wildlife Sanctuary.

Species	PA Name	Year	Area covered (km²)	Density of individuals/km² (±SE)	Density 95% CI range	Population size ± SE	Population size 95% CI range	Density & Abundance %CV	Detection probability (P) pooled ± SE	Detection probability (P) pooled 95% CI	Mean Cluster size ± SE
Banteng	SWS outer core	1276	2010	0.81 ± 0.31	0.39-1.72	1039 ± 399	493-2190	38.37	0.61 ± 0.06	0.51-0.73	5.10 ± 0.58
	SWS inner core	460	2011	1.90 ± 0.37	1.30-2.77	872 ± 169	596-1276	19.44	0.61 ± 0.06	0.51-0.73	5.10 ± 0.58
	SWS core*	1736	2010-11	1.10 ± 0.31	0.50-1.72	1911 ± 531	870-2952	27.80	-	-	-
	SWS inner core	460	2014	2.28 ± 0.53	1.44-3.61	1048 ± 246	662-1659	23.45	0.45 ± 0.05	0.35-0.57	4.99 ± 0.41
	SWS core	1736	2016	0.39 ± 0.13	0.20-0.75	676 ± 228	351-1301	33.75	0.46 ± 0.04	0.39-0.54	4.34 ± 0.36
	SWS core	1736	2018	0.25 ± 0.12	0.10-0.61	433 ± 201	178-1055	46.41	0.54 ± 0.03	0.48-0.61	3.38 ± 0.89
	SWS core	1736	2020	0.21 ± 0.10	0.09-0.53	371 ± 177	150 - 918	47.80	0.79 ± 0.06	0.67-0.90	3.40 ± 0.81
	PPWS	1670	2010-11	0.66 ± 0.20	0.36-1.21	1102 ± 341	602-2018	30.94	0.61 ± 0.06	0.51-0.73	5.10 ± 0.58
	PPWS	1670	2014	1.27 ± 0.38	0.70-2.29	2123 ± 639	1177-3832	30.11	0.45 ± 0.05	0.35-0.57	4.99 ± 0.41
	PPWS	1670	2016	1.39 ± 0.29	0.93-2.09	2322 ± 481	1546-3485	20.71	0.46 ± 0.04	0.39-0.54	4.34 ± 0.36
	PPWS	1670	2018	0.35 ± 0.14	0.17-0.76	591 ± 234	276-1264	39.54	0.54 ± 0.03	0.48-0.61	3.25 ± 0.57
	PPWS	1670	2020	0.29 ± 0.14	0.11-0.74	485 ± 238	191 - 1230	49.18	0.79 ± 0.06	0.67-0.90	2.94 ± 0.81
Red Muntjac	SWS outer core	1276	2010	2.47 ± 0.46	1.69-3.60	3150 ± 590	2159-4594	18.74	0.57 ± 0.02	0.52-0.62	1.08 ± 0.03
	SWS inner core	460	2011	2.83 ± 0.35	2.22-3.62	1303 ± 159	1021-1663	12.24	0.57 ± 0.02	0.52-0.62	1.08 ± 0.03
	SWS core*	1736	2010-11	2.56 ± 0.24	2.09-3.06	4453 ± 424	3623-5283	9.51	-	-	-
	SWS inner core	460	2014	2.09 ± 0.34	1.50-2.90	961 ± 158	692-1333	16.39	0.33 ± 0.02	0.32-0.39	1.03 ± 0.02
	SWS core	1736	2016	1.94 ± 0.28	1.46-2.59	3376 ± 489	2537-4493	14.49	0.40 ± 0.03	0.34-0.47	1.06 ± 0.03
	SWS core	1736	2018	1.22 ± 0.17	0.92-1.60	2210 ± 292	1605-2774	13.83	0.46 ± 0.03	0.41-0.51	1.18 ± 0.06
	SWS core	1736	2020	0.82 ± 0.16	0.55-1.22	1425 ± 284	962-2112	19.91	0.56 ± 0.03	0.49-0.63	1.08 ± 0.02
	PPWS	1670	2010-11	1.52 ± 0.29	1.04-2.22	2543 ± 478	1744-3708	18.80	0.57 ± 0.02	0.52-0.62	1.14 ± 0.02
	PPWS	1670	2014	1.69± 0.26	1.23-2.31	2816 ± 441	2057-3854	15.65	0.33 ± 0.02	0.32-0.39	1.06 ± 0.02
	PPWS	1670	2016	2.04 ± 0.28	1.55-2.69	3409 ± 476	2589-4488	13.95	0.40 ± 0.03	0.34-0.47	1.05 ± 0.02
	PPWS	1670	2018	1.17 ± 0.15	0.90-1.50	1945 ± 251	1507-2512	12.91	0.46 ± 0.03	0.41-0.51	1.11 ± 0.05
	PPWS	1670	2020	1.15 ± 0.15	0.89-1.50	1925 ± 256	1480-2504	13.29	0.56 ± 0.03	0.49-0.63	1.08 ± 0.03
Wild pig	SWS outer core	1276	2010	1.93 ± 0.56	1.10-3.39	2460 ± 716	1398-4330	29.10	0.50 ± 0.05	0.40-0.62	5.25 ± 0.88
	SWS inner core	460	2011	1.86 ± 0.52	1.08-3.20	854 ± 239	496-1470	27.97	0.50 ± 0.05	0.40-0.62	5.25 ± 0.88
	SWS core*	1736	2010-11	1.91 ± 0.42	1.08-2.77	3314 ± 738	1868-4760	22.26	-	-	-
	SWS inner core	460	2014	6.52 ± 1.95	3.64-11.68	2998 ± 898	1673-5372	29.95	0.30 ± 0.02	0.26-0.36	7.51 ± 1.36
	SWS core	1736	2016	2.94 ± 0.61	1.96-4.42	5112 ± 1060	3405-7674	20.73	0.53 ± 0.02	0.49-0.58	3.77 ± 0.57
	SWS core	1736	2018	1.90 ± 0.33	1.35-2.67	3301 ± 572	2439-4638	17.33	0.45 ± 0.03	0.40-0.51	3.23 ± 0.35
	SWS core	1736	2020	1.25 ± 0.26	0.84-1.87	2169 ± 443	1451-3240	20.43	0.57 ± 0.05	0.46-0.67	2.25 ± 0.22
	PPWS	1670	2010-11	0.95 ± 0.29	0.53-1.71	1595 ± 479	891-2855	30.03	0.50 ± 0.05	0.40-0.62	5.25 ± 0.88
	PPWS	1670	2014	2.21 ± 0.63	1.26-3.88	3698 ± 1060	2109-6482	28.66	0.30 ± 0.02	0.26-0.36	3.61 ± 0.59
	PPWS	1670	2016	1.06 ± 0.17	0.77-1.46	1863 ± 301	1354-2564	16.18	0.53 ± 0.02	0.49-0.58	1.47 ± 0.13
	PPWS	1670	2018	2.44 ± 0.46	1.68-3.54	4074 ± 775	2807-5914	19.02	0.45 ± 0.03	0.40-0.51	2.81 ± 0.34
	PPWS	1670	2020	1.11 ± 0.26	0.70-1.75	1848 ± 431	1170-2919	23.33	070 ± 0.09	0.53-0.88	2.07 ± 0.26

*(area-weighted mean)

ANNEX 4: MIST AND SMART PATROLLING DATA OF PPWS AND SWS IN 2010-2019

Length of electrified wire snares and number of lethal traps including snares encountered and removed by SMART patrolling teams in PPWS and SWS, distance covered during patrols and the number of lethal traps corrected for distance patrolled (number of lethal traps encountered and removed per 100Km patrolled), based on MIST and SMART data in the period 2010-2019.

Year	SWS				PPWS				SWS & PPWS Combined			
	Electrified wire traps (meters)	Lethal trap no.	Distance patrolled (in Km)	Lethal traps/100Km patrolled	Electrified wire traps (meters)	Lethal trap no.	Distance patrolled (in Km)	Lethal traps/100Km patrolled	Electrified wire traps (meters)	Lethal trap no.	Distance patrolled (in Km)	Lethal traps/100Km patrolled
2010		13	31004	0.00		7	15480	0.05	0	20	46485	0.04
2011		86	27964	0.31		4	19551	0.02	0	90	47515	0.19
2012		0	4207	0.00		4	16161	0.02	0	4	20368	0.02
2013	1000	254	25538	0.99		115	9946	1.16	1000	369	35484	1.04
2014	800	556	23789	2.34		145	8078	1.79	800	701	31867	2.20
2015	401	1883	32020	5.88		818	12398	6.60	401	2701	44418	6.08
2016	15	3794	39487	9.61	1021	2678	13083	20.47	1036	6472	52570	12.31
2017	1100	1276	54646	2.34	200	1632	11527	14.16	1300	2908	66173	4.39
2018	300	2105	42809	4.92	0	754	3237*	23.29	300	2859	46046	6.21
2019	100	2634	58977	4.47	200	1382	29359	4.71	300	4016	62214	6.46

*For PPWS, there exists as data gap in 2017 due to a temporary absence of SMART reporting in this year, therefore the patrolled distance appears comparatively low

ANNEX 5: DENSITY ESTIMATES OF UNGULATE (TIGER PREY) SPECIES FROM DISTANCE-SAMPLING BASED LINE TRANSECT SURVEYS IN SOUTH AND SOUTHEAST ASIAN TIGER RANGE COUNTRIES

Region	Country	Location	Year	Mouse deer *Tragulus sp.*	Muntjac/barking deer *Muntiacus sp.*	Wild Pig *Sus scrofa*	Niligiri tahr *Nilgiritragus hylocrius*	Chinkara *Gazella bennettii*	Sambar deer *Rusa unicolor*	Hog deer *Hyelaphus porcinus*	Chital/Spotted deer *Axis axis*	Chowsingha *Tetracerus quadricornis*	Nilgai *Boselaphus tragocamelus*	Banteng *Bos javanicus*	Wild water buffalo *Bubalus arnee*	Gaur *Bos gaurus*	Combined tiger prey density
South Asia	Nepal	Parsa National Park	2017-2018	*	*	4.89±1.2	*	*	2.2±0.6	*	8.82±3.6	*	*	*	*	*	22.02±3.8
South Asia	Nepal	Chitwan National Park	2017-2018	*	3.84±1.19	3.8±0.89	*	*	9.96±2.04	13.4±3.4	43.85±8.2	*	*	*	*	*	70.7±7.49
South Asia	Nepal	Banke National Park	2017-2018	*	*	*	*	*	*	*	*	*	*	*	*	*	8.1±1.6
South Asia	Nepal	Bardia National Park	2017-2018	*	*	2.04±0.57	*	*	1.48±0.32	*	56.44±5.75	*	*	*	*	*	77.51±6.56
South Asia	Nepal	Shuklaphanta National Park	2017-2018	*	*	9.03±2.15	*	*	*	10.14±3.33	48.8±6.6	*	*	*	*	*	68.04±6.95*
South Asia	Bhutan	Jigme Singye Wangchuck National Park	2005-2006	*	2.17±0.36	3.68±1.39	*	*	0.98±0.25	*	*	*	*	*	*	*	7.4
South Asia	India	Rajai National Park	2006-2007	*	*	1.9±1.3	*	*	14.6±3.6	*	49.9±13	*	2.4±2.4	*	*	*	68.8
South Asia	India	Pench National Park	1998-1999	*	*	2.59	*	*	6.09	*	80.75	0.29	0.43	*	*	0.34	90.3
South Asia	India	Panna Tiger Reserve	2006	*	*	*	*	*	11.89±2.95	*	16.31±6.36	*	16.13±5.33	*	*	*	42.44±8.4
South Asia	India	Anamalai Tiger Reserve	2004	0.18	0.28	20.61	13.67	*	6.54	*	20.54	*	*	*	*	12.34	72.1**
South Asia	India	Sariska Tiger Reserve	2008-2009	*	*	15.4±4.4	*	*	26.2±4.9	*	46.7±9.5	*	19.5±3.3	*	*	*	107.8
South Asia	India	Kalakad-Mudanthurai Tiger Reserve	2006 & 2010	*	*	1.3±0.71	*	*	7.0±1.5	*	*	*	*	*	*	3.6±1.5	11.9±3.7
South Asia	India	Manas National Park	2008	*	2.06±1.42	2.75±2.47	*	*	3.95±2.51	4.59±2.54	*	*	*	*	22.88±11.63	5.79±3.26	42.02
South Asia	India	Similipal Tiger Reserve	2012	0.6±0.2	1.6±3.2	3.2±0.7	*	*	2.8±1.8	*	5.0±2.2	*	*	*	*	*	4.9±0.6****
South Asia	India	Ranthambhore National Park	2000-2001	*	*	9.77	*	5.62	17.15	*	31	*	11.36	*	*	*	*
South Asia	India	Bhadra Tiger Reserve	2000	*	3.48	*	*	*	0.81	*	1.6	*	*	*	*	0.64	*
South-East Asia	Cambodia	Keo Seima Wildlife Sanctuary	2018	*	0.66±0.13	0.76±0.25	*	*	*	*	*	*	*	0.01±0.01	*	0.05±0.04	*
South-East Asia	Cambodia	Phnom Prich Wildlife Sanctuary	2018	*	1.2 ±0.2	2.4 ±0.5	*	*	*	*	*	*	*	0.4 ±0.1	*	*	*
South-East Asia	Cambodia	Srepok Wildlife Sanctuary	2018	*	1.2 ±0.2	1.9 ±0.3	*	*	*	*	*	*	*	0.2 ±0.1	*	*	*
South-East Asia	Vietnam	Yok Don National Park	2018-2019	*	0.74±0.11	2.81± 0.71	*	*	*	*	*	*	*	*	*	*	*
South-East Asia	Indonesia	Bukit Barisan Selatan National Park, Sumatra	1999	2.74	4.44	4.6	*	*	0.62	*	*	*	*	*	*	*	*

*Includes langur
**Includes elephant, and black-naped hare
***Pre- and post monsoon densities were provided to compare seasonality, pre-monsoon densities are used here for comparability with the EPL field season
****Includes primates (rhesus macaque and common langur)
*****Analysis for gaur and bateng were based on very low encounter rates, with detection functions built over multiple years

Ungulate density estimates (±SE) based on distance-sampling based line transect surveys, sources: Nepal all parks (DNPWC & DFSC, 2018), Bhutan: Jigme Singye Wangchuck National (Wang, 2010), India: Rajai National Park (Harihar et al., 2009), Nandhaur Region (Mann et al., 2013)Pench (Biswas & Sankar, 2002), Panna Tiger Reserve (Gopal et al., 2010), Anamalai Tiger Reserve, (Kumaraguru et al., 2011), Sariska Tiger Reserve (Sankar et al., 2010), Kalakad-Mudanthurai Tiger Reserve (Ramesh et al., 2012), Manas National Park (Goswami & Ganesh, 2014), Similipal Tiger Reserve (Nayak et al., 2014), Ranthambhore National Park (Bagchi et al., 2003), Bhadra Tiger Reserve (Jathanna et al., 2003), Western Terai Arc Landscape, India ((Harihar et al., 2014)Cambodia: Keo Seima Wildlife Sanctuary (WCS 2019, unpublished results), Viet Nam: Yok Don National Park (WWF Viet Nam, 2019), Indonesia: Bukit Barisan Selatan National Park, Sumatra (O'Brien et al., 2003).

ANNEX 6: LIST OF FIELD ASSISTANTS ON LINE TRANSECT TEAMS IN 2014, 2016, 2018, AND 2020

Names of field assistants 2014			Names of field assistants 2016			Names of field assistants 2018			Names of field assistants 2020		
1	Art Richard	អាត រីឆាត	1	Blorch Rai	ប្លូច រ៉ៃ	1	Blorch Rai	ប្លូច រ៉ៃ	1	Blorch Rai	ប្លូច រ៉ៃ
2	Bo Mean	បូ មាន	2	Chan thuy	ចាន់ ធុយ	2	Chan thuy	ចាន់ ធុយ	2	Chan Khamsay	ចាន់ ខាំសាយ
3	Chan thuy	ចាន់ ធុយ	3	Chea Minea	ជា មិនា	3	Chhok Kroeurl	ឆក់ គ្រើល	3	Chhok Kroeurl	ឆក់ គ្រើល
4	Chhok Kroeurl	ឆក់ គ្រើល	4	Chhok Kroeurl	ឆក់ គ្រើល	4	Hong De	ហុង ដេ	4	Chran Deim	ច្រាន ឌីម
5	Eng Ratha	អេង រដ្ឋា	5	Chras Prim	ច្រាស ព្រឹម	5	Ien Khve	អៀន ខ្វេរ	5	Dem Blev	ឌឹម ប្លែវ
6	Hong Dae	ហុង ដែ	6	Dom Dorn	ដុំ ដន	6	Klev Srouv	ក្លេវ ស្រូវ	6	Den Khunny	ឌិន ឃុននី
7	Hong De	ហុង ដេ	7	Hong De	ហុង ដេ	7	Len Nhor	ឡេន ញ៉	7	Hong De	ហុង ដេ
8	Keurb Yan	គើប យ៉ាន	8	In Veasna	អ៊ិន វាសនា	8	Loun Noun	លួន នួន	8	Horn Sela	ហ៊ន សិលា
9	Kha Phorlla	ខា ផល្លា	9	Klev Srouv	ក្លេវ ស្រូវ	9	Mai Samorn	ម៉ៃ សាម៉ន	9	Ien Khve	អៀន ខ្វេរ
10	Khaim Nhum	ខាម ញុម	10	Leh Nim	ឡេ និម	10	May Kosal	ម៉ៃ កុសល	10	Kheom Sovanbottra	ខឹម សុវណ្ណបុត្រា
11	Len Nhor	ឡេន ញ៉	11	Len Nhor	ឡេន ញ៉	11	Mhes Puoeung	មេស ពឿង	11	Klev Srouv	ក្លេវ ស្រូវ
12	Loun Noun	លួន នួន	12	Loun Noun	លួន នួន	12	Muny Samath	មុន្នី សំអាត	12	Mai Samorn	ម៉ៃ សាម៉ន
13	Mai Samorn	ម៉ៃ សាម៉ន	13	Mai Samorn	ម៉ៃ សាម៉ន	13	Nhem Bu So	ញឹម ប៊ូសូ	13	May Kosal	ម៉ៃ កុសល
14	May Kosal	ម៉ៃ កុសល	14	May Kosal	ម៉ៃ កុសល	14	Nuenh Punh	នួញ ពុញ	14	Muny Samath	មុន្នី សំអាត
15	Muny Samath	មុន្នី សំអាត	15	Muny Samath	មុន្នី សំអាត	15	Nurt Thurt	នឹត ថឹត	15	Nurt Thurt	នឹត ថឹត
16	Neurv Tiep	និវ ទាប	16	Ngim Tib	ង៉ឹម ទីប	16	Pheak Chea	ភាក់ ជា	16	Nusuk Chanthon	នូ សុខចាន់ថន
17	Nun Sokleng	ណាន់ សុខឡេង	17	Nurt Thurt	នឹត ថឹត	17	Roeng Pyunh	រឹង ព្យុង	17	Pheak Chea	ភាក់ ជា
18	Sary Tre	សារី ត្រែ	18	Phin Sreng	ភិន ស្រេង	18	Sary Tre	សារី ត្រែ	18	Roeng Pyunh	រឹង ព្យុង
19	Sorm Sareun	សុំ សារឿន	19	Roeng Pyunh	រឿង ព្យុង	19	Savorn Chan	សាវ៉ន ចាន់	19	Sacorn Sochea	សាខន សុខជា
20	Sun Nin	ស៊ិន និន	20	Sary Tre	សារី ត្រែ	20	Sin Sothea	ស៊ិន សុធា	20	Sambath Sukh	សម្បត្តិ សុខ
21	Thea Chanthorn	ធា ចន្ធន	21	Soli Mech	សូលី មិច	21	Soli Mech	សូលី មិច	21	Sary Tre	សារី ត្រែ
22	Then Thol	ថេន ថុល	22	Sor Phakchea	សរ ភាក់ជា	22	Sophoeun Oeun	សុភឿន អឿន	22	Soli Mech	សូលី មិច
23	Vann Sani	វណ្ណ សានី	23	Sorn Kheem	សន ឃីម	23	Sran Chek	ស្រាន់ ចែក	23	Sophoeun Oeun	សុភឿន អឿន
			24	Sran Chek	ស្រាន់ ចែក	24	Sran Khvet	ស្រាន់ ខ្វែត	24	Sran Chek	ស្រាន់ ចែក
			25	Sran Khvet	ស្រាន់ ខ្វែត	25	Sun Nin (Sin Nin)	ស៊ិន និន	25	Sran Khvet	ស្រាន់ ខ្វែត
			26	Sun Nin (Sin Nin)	ស៊ិន និន	26	Ten Kak	តេង កាក	26	Sun Nin (Sin Nin)	ស៊ិន និន
			27	Thea Chanthorn	ធា ចន្ធន	27	Thea Chanthorn	ធា ចន្ធន	27	Ten Kak	តេង កាក
			28	Then Thol	ថេន ថុល	28	Thin Sophan	ធីន សុផាន់	28	Thea Chanthorn	ធា ចន្ធន
			29	Tien Sothy	តៀន សុធី	29	Trerb Poum	ត្រើប ពូម	29	Theng Ting	ថេង តឹង
			30	Toung Kyt	ត្អង គីត	30	Vann Sani	វណ្ណ សានី	30	Thin Sophan	ធីន សុផាន់
			31	Trerb Poum	ត្រើប ពូម				31	Thol Virak	ថុល វីរៈ
			32	Vann Sani	វណ្ណ សានី				32	Tonh Prich	តញ់ ព្រិច
			33	Ven Saomony	វេន សៅមុន្នី				33	Trerb Poum	ត្រើប ពូម
									34	Veasna Phea	វាសនា ភា
									35	Vorn Mom	វន មុំ
									36	Yon Sopha	យន់ សុផា

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