A multiscale analysis and quantification of human impacts on Sumatran tiger (*Panthera tigris sumatrae*) habitat in Riau, Sumatra

Erin Elizabeth Poor

Dissertation submitted to the faculty of the Virginia Polytechnic Institute and State University in partial fulfillment of the requirements for the degree of

Doctor of Philosophy
In
Geospatial and Environmental Analysis

Marcella J. Kelly, Committee Chair
Emmanuel A. Frimpong
Yang Shao
Sunarto
Lisette P. Waits

July 31, 2018
Blacksburg, Virginia

Keywords: deforestation, fragmentation, habitat loss, human impacts, land cover change, protected areas

A multiscale analysis and quantification of human impacts on Sumatran tiger (*Panthera tigris sumatrae*) habitat in Riau, Sumatra

Erin Elizabeth Poor

ABSTRACT

Worldwide, we are losing biodiversity at unprecedented rates, and due to deforestation, degradation and poaching, Southeast Asian wildlife is facing extreme threats. Indonesia recently eclipsed Brazil in having the world's highest deforestation rate, largely due to the rise of the palm oil industry. Indonesia contains multiple biodiversity hotspots and endangered species such as the Sumatran tiger (*Panthera tigris sumatrae*). While Riau Province, Sumatra, produces approximately 20% of the world's palm oil, tigers still inhabit parts of Riau, though their habitat and prey are understudied. Thus, in this research, I aim to assess and quantify how tiger habitat has changed, how it will continue to change, and provide recommendations on how to improve the landscape for tigers. I create the first accuracy-assessed land cover maps of Riau, and then predict land cover change from 2016 – 2050. Using this newly created land cover map, I assess whether Tesso Nilo National Park, Bukit Tigapuluh National Park, and Rimbang Baling Wildlife Reserve are effective at preventing deforestation. Next, I examine human impacts within Tesso Nilo specifically, due to its suitability for oil palm and its potential as a stepping stone for wildlife movement from the western, mountains to the eastern peatlands of Sumatra. Finally, I examine impacts of human presence within Rimbang Baling on felid-prey relationships. I predict that by 2050, over 60% of natural forest in Riau will be lost, and all protected areas only confer low levels of protection. I determined that Tesso Nilo National Park has nearly 2500 km of roads within it and no areas within the park are untouched by humans. Wildlife detections were low near the boundary of Rimbang Baling and there was evidence of humans negatively impacting mousedeer (*Tragulus spp*) behavior. I suggest focusing on securing the habitat within Rimbang Baling and Bukit Tigapuluh to ensure habitat for dispersing tigers from the western mountains, in addition to, and perhaps before focusing on restoring Tesso Nilo and creating wildlife corridors. While tiger recovery in Riau will be difficult, with education, dedication, persistence and intelligent planning, tigers may be able to persist in this unique ecosystem in the long-term.

A multiscale analysis and quantification of human impacts on Sumatran tiger (*Panthera tigris sumatrae*) habitat in Riau, Sumatra

Erin Elizabeth Poor

GENERAL AUDIENCE ABSTRACT

Worldwide, we are losing plants and animals at unprecedented rates, and due to deforestation, degradation and poaching, Southeast Asian wildlife is facing extreme threats. Indonesia recently passed Brazil in having the world's highest deforestation rate, largely due to the rise of the palm oil industry. Indonesia has a wide diversity of plants and animals, including endangered species such as the Sumatran tiger. While Riau Province, Sumatra, produces approximately 20% of the world's palm oil, tigers still inhabit parts of Riau, though their habitat and prey are understudied. Thus, in this research, I aim to assess how tiger habitat has changed, how it will continue to change, and provide recommendations to improve the landscape for tigers. I create the first land cover maps of Riau that have been verified with field data, and then predict land cover change from 2016 – 2050. Using this land cover map, I assess whether Tesso Nilo National Park, Bukit Tigapuluh National Park, and Rimbang Baling Wildlife Reserve are effective at preventing deforestation. Next, I examine human impacts within Tesso Nilo, due to its soil characteristics making it suitable for oil palm and its potential as habitat for wildlife movement from the western, mountains to the eastern swamps of Sumatra. Finally, I examine impacts of humans within Rimbang Baling on wild cat-prey relationships. I predict that by 2050, over 60% of forest in Riau will be lost, and all protected areas only provide slight protection. I determined that Tesso Nilo has nearly 2500 km of roads within it and no areas within the park are untouched by humans. Wildlife sightings were low near the boundary of Rimbang Baling and there was evidence of humans negatively impacting tiger prey behavior. I suggest focusing on securing the forests within Rimbang Baling and Bukit Tigapuluh to ensure forest for tigers that may come from the western mountains, in addition to, and before focusing on restoring Tesso Nilo and creating wildlife movement areas. While tiger recovery in Riau will be difficult, with education, dedication, persistence and intelligent planning, tigers may be able to persist in this unique ecosystem in the long-term.

Dedication

For Annabelle Kong for inspiring and motivating me in the toughest times,

Grandma McGill, for encouraging my love of nature,

And to Bill and Carla for their love.

Acknowledgements

To say this took a village would be an understatement. Marcella Kelly provided support and humor when things got rough and was patient and understanding when things were not going as planned (which wasn't much). Her advising has allowed me to grow as a scientist and as a human being. My committee provided guidance and encouragement and pushed me beyond what I thought I could achieve.

My generous funders who made this project possible include: the National Science Foundation, U.S. Student Fulbright program, the Philadelphia Zoo, the Cleveland Zoo, the Riverbanks Zoo, IdeaWild, Virginia Tech Graduate School, the Wild Felid Association, the World Wildlife Fund, and many private donors. Barney Long was instrumental in assisting with WWF funding, providing advice in Indonesia and lots of encouragement. Your help and encouragement is so appreciated!!

I don't know where to start in my thanks to those in Indonesia who have helped me. I thank the World Wildlife Fund-Indonesia, Universitas Gadjah Mada, The Eijkman Institute, Balai Konservasi Sumber Daya Alam, Kementerian Kehutanan dan Linkungan Hidup and AMINEF for their patience and support in logistics and research.

The field teams at WWF-Pekanbaru were amazing. Their knowledge and dedication to conservation are awe inspiring. I especially want to thank Tugio (aka Bapak Egi), Zulfahmi, Efendy Pajaitan, Antika Fardhilla, Doni Susanto, Ucok Rahmad Adi for their help above and beyond what they were asked. Other team members who were no less helpful were: Yogi Alro, Asep Firman, Ata, Romi Chandra, Erizal, Febrian, Yansen Gautam, Hermanto, Jerry Irawan, Jaka, Kusdianto, Wiwid Prayoga, Heri Priatna, Syabendri, Ravel Wibowo and Zuhri. This research belongs to all of you. It would have been impossible without you. Dengan senang hati, I thank you.

At the Pekanbaru office, Febri Anggrian Widodo, Wishnu Sukmantoro,Virta Ishanul Mustika Jati, Rafselia Novalina, Elmadia Achmad were extremely helpful and supportive with logistics, research and laughter. At UGM, Pak Muhammad Ali Imron, was generous with his conversation, support, advice and insights. He edited many of the manuscripts and is an advisor I would recommend to anyone. Thank you.

Although our genetics work is not included here, I also thank Jen Adams at University of Idaho, Ibu Hera, Isabella Apriyani, Jessica Wiludjaja, Jessica Saragih and Bertha for assistance in the lab, their kind smiles, and their advice. Stay tuned for genetics results!

Of course, conducting research in Sumatra would have been completely impossible without my language skills. For that, I have Wisma Bahasa and their amazing teachers in Yogyakarta to thank. I can't stress the amount of patience and humor they must have to deal with all of us bules. In Yogya, I would be remiss without thanking Ibu and Bapak Heru for the unending hospitality and help with permits. They kept a clean roof over my head, (too much) delicious food in my stomach and made me feel cared for in my first three months in a new country.

Elizabeth Ballare, Martha Walters, Lauren Yapp, Alison Purnell and Laura Kemp were my found family in Indonesia. They have seen me at my worst, consoled me when my project (mostly) failed, hugged me when I was scared, ate expired cheese with me and found ways to keep the wine flowing. I don’t have words to tell you how thankful I am that I found you all. I love you all – no matter the time or distance. Alison, John, Catherine and Ken Purnell opened their home to me and made me feel like family on numerous occasions which was an amazing feeling after having been in Sumatra without ‘home’ for months. You made life abroad immensely easier and more enjoyable. I wish I could repay you for your friendship and hospitality. Come visit! I miss you!

Back in Blacksburg, I would like to thank the faculty of the Department of Fish and Wildlife Conservation, Dana Keith and Lynn Hayes. I also need to thank everyone in the WHAPA lab including: Anne Hilborn, Asia Murphy, Zach Farris, Lindsey Rich, Dana Morin, Kanchan Thapa, Chris Rowe, Robert Alonso, Dave McNitt, Bernardo Mesa, Christine Proctor, and Claudia Wultsch. Tom McNamara provided humor and of course, delicious breaks from research! They all gave me friendship, advice, scientific knowledge and encouragement. Claudia and Zach gave me invaluable help for genetics and occupancy modeling, respectively, and Robert kept me company and taught me some of the basics of trapping in Indonesia. And although our plans for trapping didn’t come to fruition, I also need to thank Dairen Simpson for his kind words and encouragement. Thank you!

Throughout this process, my family’s unconditional love for me has been tested and proven time and again. Nothing about this process was easy, but none of them gave up on me. My mom and dad, Chuck and Jenny Poor came to visit me in Indonesia, just when I needed it. Seeing the joy on their faces when they saw an orangutan in the wild made everything worth it. My husband, David Clayman, also visited me and brought trapping gear with him. He has fed me and taken care of me when I could not. He has stood by me while I pursue my insane dreams and for this, I am beyond grateful. He is patient and he understands me. I can’t say how thankful I am to have a partner like this. My fur babies, Emmy and Cat, have provided endless stress relief. My aunt and cousins, Cathy, Nick and Alex Ovenhouse, my sister and brother-in-law, Allison and Matthew Haraminac, my parents-in-law, Lillian and Roger Clayman, and sister-in-law, Rebecca Clayman have also all been supportive, excited for me, loving, and patient with me through this process. Your love means so much and I thank you.

For anyone I have forgotten to mention by name, you do not mean any less – I am just very tired. I am still thankful for you. This work was supported not only by those mentioned but also by countless administrators, secretaries and support staff, as well as everyone I met in Indonesia who passed on a smile, knowledge, information, a handshake, a new word or any kind of encouragement. I also need to thank Andy Cohen and the ladies of the OC, NYC, Atlanta, Potomac, Beverly Hills and New Jersey for helping me procrastinate, laugh, cry and be thankful beyond words that I am in no other field than wildlife conservation.

Throughout this process, I have seen myself as a facilitator – all of you who have made this research possible are the ones deserving this degree. I have been carried like a piece of rice by a colony of biting ants to reach my destination. I am infinitely grateful for you all. Thank you.

Table of Contents

List of Figures

Chapter 1

Chapter 2

Chapter 3

Chapter 4

Chapter 5

List of Tables

Chapter 2

Chapter 3

Chapter 4

Chapter 5

Attribution

Contributors are listed in authorship order. In all cases, I am first and corresponding author and I have collected the data with aid from field teams, analyzed the data, and written the bulk of each manuscript.

Chapter 2

Yang Shao aided in conception, data analysis and editing. Marcella J. Kelly provided funding support, chapter editing, and scientific guidance.

Chapter 3

Emmanuel Frimpong contributed to conceptualization, data analysis and editing. Muhammad Ali Imron edited the chapter. Marcella J. Kelly provided chapter editing, guidance, and funding support.

Chapter 4

Virta I.M. Jati contributed to data preparation and analysis. Muhammad Ali Imron contributed to conceptualization and editing of the manuscript. Marcella J. Kelly conceptualized, edited, and aided in funding for this manuscript.

Chapter 5

Marcella J. Kelly aided in conceptualization, data interpretation, manuscript editing, and funding.

Chapter 1

Introduction

Globally, wildlife is being lost at unprecedented rates (Pimm et al. 2014) due to habitat loss and hunting. Southeast Asia has one of the highest deforestation rates in the world and a significant amount of biodiversity could be lost by 2100 (Sodhi et al. 2009; Margono et al. 2014). Because mammals may be more sensitive to forest disturbance than other taxa (Sodhi et al. 2009), they are likely be the most affected by an increase in development, with estimates of a loss of 21-48% of Southeast Asian mammals by 2100 (Brooks et al. 1999). Species that are wide-ranging and that exist in low density like many carnivores are often the first to go extinct when habitat is fragmented by development (Beier 1993; Pimm and Clark 1996; MacNally and Bennett 1997).

Carnivore guilds found in Southeast Asia are higher in diversity than on other continents, yet many Asian carnivore species now occur at population sizes too small to fulfill their past ecological functions (Dalerum et al. 2009). As roads and railroads continue to bisect habitat, dams flood habitat, and mines and towns fragment forests, the overall decrease in habitat and the increased distance among habitat patches will lead to reduced carnivore presence (Crooks 2002; Mortelliti and Boitani 2008). Although carnivore presence often correlates with prey abundance, below a certain patch size threshold, use of habitat may completely stop regardless of prey abundance (Mortelliti and Boitani 2008). In a comparison of a protected area with many small (<100 ha) patches to one with fewer large (>400 ha) habitat patches in Thailand, Pattanavibool and Dearden (2002) found the protected area with large patches still contained large mammals that had been extirpated from the more fragmented protected area. Preserving connectivity among isolated patches of habitat during the early stages of degradation is one of the most important factors in conserving endangered carnivore species such as the tiger (Carroll et al.

2004).

Threats to tigers (*Panthera tigris*) are growing more rapidly than the rate at which accurate and reliable scientific information is being collected (Ranganathan et al. 2008). Only 7% of the original tiger range remains (Dinerstein et al. 2007) and the recent global population estimate of tigers is at an all-time low. The decline in the tiger population is a multi-faceted, multi-scale, far-reaching problem, affecting nearly every country in the world through activities such as trade in tiger parts, tiger consumption, tiger poaching, illegal logging and conversion of tiger habitat, consumption of agricultural products grown on such converted land, negligent and corrupt law enforcement and/or insufficient funds to support law enforcement.

However, at the International Tiger Conservation Forum in 2010, the governments of the 13 tiger range countries and conservation scientists across multiple continents acknowledged that the seemingly imminent loss of tigers across Asia would result in a loss of healthy ecosystems and thus agreed to make a concerted effort to double the number of wild tigers by 2022. With their “Declaration on Tiger Conservation”, the above parties called on the entire international community to work towards the goal of tiger recovery. With a population estimate of 300-400 (Tilson and Traylor-Holzer 1994, Soehartono et al. 2007) and an IUCN status of Critically Endangered (Linkie et al. 2008), the wild Sumatran tiger (*P.t. sumatrae*) is the most endangered of all tiger subspecies and its persistence is tenuous. Since most tiger habitat is unprotected (Chundawat et al. 1999; Wikramanayake et al. 2004) and there is a good chance more habitat will be lost, populations are likely to become further fragmented into genetically isolated sub-populations, making connectivity among habitat patches, and understanding human impacts on tiger habitat and prey availability, critical to overall species’ survival.

Besides conserving tigers for the sake of conservation, or their significance to many cultures, there are at least three reasons to be concerned about the recently low and declining estimates of the wild tiger population. First, tigers are a wide-ranging species and hence often seen as an umbrella species, meaning that their conservation could ensure the protection of other species within their wide range across diverse habitats. Second, there is evidence to suggest that carnivores regulate the density of herbivores and thus affect vegetation structure and other ecosystem processes (Terborgh 1994; 2001; Ripple and Beschta 2012). If this is true of tigers, their continual decline could lead to a decline in overall ecosystem health across the tiger range (Estes et al. 2011; Soule et al. 2003). Finally, preliminary studies suggest that native tiger habitat contains a greater amount of carbon and can filter more rain water for neighboring human populations than areas converted to agriculture (WWF, unpublished data), making their habitats crucial to providing the ecosystem services needed for the growing human population.

Agricultural conversion of tiger habitat has been a growing threat to Sumatran tigers over the past few decades, and Riau Province, Sumatra, produces about 20% of the global palm oil supply. From 1990-2005, at least 56% of oil palm expansion replaced forest (Koh and Wilcove, 2008). Oil palm was first planted in Sumatra in 1911 (Corley and Tinker, 2003) and there are now approximately 6.1 million ha of oil palm in Indonesia (FAO 2006). While tigers have been seen in oil palm, overall, oil palm plantations support fewer than 50% of vertebrate species as primary forests (Danielsen et al. 2009), have lower species richness than disturbed forests, and support fewer species than other types of agriculture (Fitzherbert et al. 2008). Acacia plantations have been found to hold higher beetle richness, a possible indicator of overall biodiversity, and have a species composition more similar to natural forest than oil palm plantations (Chung et al. 2000). Loss of species diversity in oil palm plantations may be due to a loss in structural complexity and plant species richness that occurs when plantations are productive (Chung et al.

2000, Glor et al. 2001, Aratrakorn et al. 2006).

Although Riau currently has relatively low tiger density in Sumatra (Figure 2; Sunarto et al. 2013), it also contains much of the remaining lowland rainforest on Sumatra and has one of the highest human population growth rates in Indonesia (Badan Pusat Statistik, 2011). Human-tiger conflict was one of the major factors that contributed to the extinction of the Javan and Bali tigers (Hoogerwerf 1970, Seidensticker 1987) and the Sumatran tiger may follow if human use and modification of tiger habitat is not understood and better managed. Conversion of natural forest to palm oil plantations often leads to new roads and access to forested areas, which can then lead to increased poaching. Tilson et al. (2010) estimate that 253 Sumatran tigers were poached from 1998-2002. With a population of <400, killing a few tigers can have a significant demographic effect on small populations (Kenney et al. 1995). About 73 tigers were poached in Riau between 1990-2000, of which 42% were taken from outside parks and approximately 58% were taken from within parks (Tilson et al. 2010).

This research in this dissertation takes place in the Tesso Nilo-Bukit Tigapuluh Conservation Landscape, which includes Tesso Nilo National Park, Bukit Tigapuluh National Park, Bukit Batabuh Protection Forest, Kerumutan Wildlife Reserve, and Bukit Rimbang Bukit Baling Wildlife Reserve (Figure 1). These protected areas contain some of the last remaining lowland tropical forest in Sumatra, and are touted as a stronghold for Sumatran tigers. However, there are no officially protected corridors, and the areas between these protected areas are mixed use and agricultural areas including acacia and oil palm plantations. Both Tesso Nilo National Park and Rimbang Baling Wildlife Reserve are categorized as Class III (of five) Tiger Conservation Landscapes (TCL) meaning they have habitat to support some tigers but also have moderate-high levels of threat and minimal conservation investment (Sanderson et al. 2006).

There possibility of maintaining viable tiger populations exists within these areas if they are connected to Bukit Tigapuluh National Park (BTNP), a Class I TCL, which may still have a stable tiger population (Sanderson et al. 2006). Due to its rugged and difficult to access landscape, BTNP still has habitat that may support at least 100 tigers, with evidence of breeding and minimal to moderate threat levels (Sanderson et al. 2006).

As one of the least studied and most diverse tropical areas in the world (Ripple et al. 2016; Myers et al. 2000), central Sumatra stands to lose important endemic flora and fauna – including the Critically Endangered Sumatran tiger – from vast amounts of deforestation with little acknowledgement. With this research, I attempt to heighten awareness of the plight of such species, and quantify impacts that humans are having on this ecosystem, especially given the dearth in wildlife data from the area, partly due to extreme difficulties with research permitting. I use a multiscale, interdisciplinary approach to quantify and predict land cover change (Chapter 2), determine whether and where protected areas in this region are providing protection to wildlife (Chapter 3), call attention to the amount of disturbance within one valuable lowland ecosystem (Chapter 4), and provide a first assessment of the impacts of humans on predator-prey interactions (Chapter 5). I use this information to provide recommendations on tiger corridor placement, where to increase enforcement around protected areas, and how to improve this landscape for tigers. In the long term, I hope this information will aid in maintaining and ultimately increasing tiger populations resulting in an overall healthier ecosystem for all in Riau, Sumatra.

References

Aratrakorn S, Thunhikorn S, Donald, PF. 2006. Changes in bird communities following conversion of lowland forest to oil palm and rubber plantations in southern Thailand. Bird Conservation International, **16**(1): 71-82.

Badan Pusat Statistik. 2011. Statistik Indonesia (Statistical Yearbook of Indonesia). Badan Pusat Statistik Republik Indonesia.

Beier P, Majka DR, Spencer WD. 2008. Forks in the roads: choices in procedures for designing wild land linkages. Conservation Biology, **22**(4): 836-851.

Brooks TM, Pimm SL, Kapos V, Ravilious CA. 1999. Threat from deforestation to montane and lowland birds and mammals in insular south-east Asia. Journal of Animal Ecology, **68**: 1061–1078.

Carroll C, Noss RF, Pacquet PC, Schumaker NH. 2004. Extinction debt of protected areas ‘ in developing landscapes. *Conservation Biology*, **18**:1110-1120.

Chundawat RS, Gogate N, Johnsingh AJT. 1999. Tigers in Panna: preliminary results from an Indian tropical dry forest. In Ridging the Tiger: Tiger Conservation in Human-dominated Landscapes (eds J. Seidensticker, S. Christie and P. Jackson), pp. 123-129. Cambridge University Press, Cambridge, UK.

Chung AYC, Eggleton P, Speight MR, Hammond PM, Chey VK. 2000. The diversity of beetle assemblages in different habitat types in Sabah, Malaysia. Bulletin of Entomological Research, **90**(6): 475-496.

Corley RHV, Tinker PB. 2003. The Oil Palm, Blackwell Science, Oxford.

Crooks KR. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology, **16**:488–502

Dalerum F, Cameron EZ, Kunkel K, Somers MJ. 2009. Diversity and depletions in continental carnivore guilds. Implications for prioritizing global carnivore research. Biology Letters, **5**: 35-38.

Dinerstein E, Loucks C, Wikramanayake E, Ginsberg J, Sanderson E, Seidensticker J, Forrest J, Bryta G, Heydlauff A, Klenzendorf S, Leimgruber P, Mills J, O’Brien TG, Shrestha M, Simons R, Songer M. 2007. The Fate of Wild Tigers. Bioscience, **57**(6): 508–514.

Estes JA, Terborgh J, Brasheres JS, Power ME, Berger J, Bond WJ, Carpenter SR, Essington TE, Holt RD, Jackson JBD, Marquis RJ, Oksanen L, Oksanen T, Pain RT, Pikitch EK, Ripple WJ, Sandin SA, Scheffer M, Schoener TW, Shurin JB, Sinclair ARE, Soule ME, Virtanen R, Wardle DA. 2011. Trophic downgrading of planet earth. Science, **333**: 301-306.

Ferreras P, Gaona P, Palomares F, Delibes M. 2001. Restore habitat or reduce mortality? Implications from a population viability analysis of the Iberian lynx. Animal Conservation, **4**: 265–274.

Fitzherbert EB, Struebig MJ, Morel A, Danielsen F, Bruhl CA, Donald PF, Phalen B. 2008. How will oil palm expansion affect biodiversity? Trends in Ecology and Evolution, **23:** 538-545.

Food and Agriculture Organization of the United Nations. 2006. Global Forest Resources Assessment 2005, FAO.

Glor RE, Flecker AS, Benard MF, Power AG. 2001. Lizard diversity and agricultural disturbance in a Caribbean forest landscape. Biodiversity and Conservation, **10**(5): 711-723.

Hoogerwerf A. 1970. Ujung Kulon: The Land of the Last Javan Rhinoceros. E. J. Brill, Leiden, The Netherlands, 512 pp.

Kenney JS, Smith JLD, Starfield AM, McDougal C. W. 1995. The long-term effects of tiger poaching on population viability. Conservation Biology, **9**: 1127–1133.

Koh LP, Wilcove DS. 2008. Is oil palm agriculture really destroying tropical biodiversity? Conservation Letters, **1**(2): 60–64.

Linkie M, Smith RJ, Zhu Y, Marty, DJ, Suedmeyer B, Pramono J, Leader-Williams N. 2008. Evaluating biodiversity conservation around a large Sumatran protected area. Conservation Biology, **22**: 683–690.

MacNally R, Bennett AF. 1997. Species-specific prediction of the impact of habitat fragmentation: local extinction of birds in the box-ironbark forests of central Victoria, Australia. Biological Conservation, **82**: 147–155.

Mortelliti A, Boitani L. 2007. Interaction of food resources and landscape structure in determining the probability of patch use by carnivores in fragmented landscapes. Landscape Ecology, **23**(3): 285–298.

Nyhus PJ, Tilson R. 2004. Characterizing human-tiger conflict in Sumatra, Indonesia: implications for conservation. Oryx, **38**(1): 68-74.

Pattanavibool A, Dearden P. 2002. Fragmentation and wildlife in montane evergreen forests, northern Thailand. Biological Conservation, **107**: 155-164.

Pimm SL, Jenkins CN, Abell R, Brooks TM, Gittleman JL, Joppa LN, Raven PH, Roberts CM, Sexton JO. 2014 The biodiversity of species and their rates of extinction, distribution, and protection. Science, **344**:1246752

Pimm SA, Clark TW. 1996. Marking sense of the policy process for carnivore conservation. Conservation Biology, **10**: 1036–1045.

Ranganathan J, Chan KMA, Karanth KU, Smith JLD. 2008. Where can tigers persist in the future? A landscape-scale, density based population model for the Indian subcontinent. Biological Conservation, **141**: 67-77.

Ripple WJ, Abernethy K, Betts MG, Chapron G, Dirzo R, Galetti M, Levi T, Lindsey PA, Macdonald DW, Machovina B, Newsome TM, Peres CA, Wallach AD, Wolf C, Young H. 2016. Bushmeat hunting and extinction risk to the world’s mammals. Royal Society of Open Science, **3**: 1–16.

Ripple WJ, Beschta RL. 2012. Large predators limit herbivore densities in northern forest ecosystems. European Journal of Wildlife Research, **58**: 733-742

Seidensticker J. 1987. Bearing Witness: Observations on the extinction of *Panthera tigris balica* and *Panthera tigris sondaica*. In Tigers of the World: The Biology, Biopolitics, Management, and Conservation of an Endangered Species (eds R.L. Tilson & U.S. Seal), pp. 1–8. Noyes Publications, Park Ridge, USA.

Sodhi NS, Lee TM, Koh LP, Brook BW. 2009. A meta-analysis of the impact of anthropogenic forest disturbance on Southeast Asia’s biotas. Biotropica, **41**: 103-109.

Soehartono T, Wibisono HT, Sunarto, Martyr D, Susilo HD, Maddox T, Priatna, D. 2007. Strategies and Conservation Action Plan of Sumatran Tigers (Panthera tigris sumatrae) 2007– 2017. Directorate General of PHKA, Ministry of Forestry, 24 pp. (in bahasa Indonesia).

Soule ME, Estes JE, Berger J, del Rio CM. 2003. Ecological effectiveness: conservation goals for interactive species. Conservation Biology, **17**: 1238-1250

Sunarto, 2011. Ecology and restoration of Sumatran tigers in forest and plantation landscapes. Virginia Tech.

Sunarto, Kelly M, Parakkasi K, Klenzendorf S, Septayuda E, Kurniawan H. 2012. Tigers need cover: multi-scale occupancy study of the big cat in Sumatran forest and plantation landscapes. PLoS ONE, **7**(1): e30859.

Sunarto, Kelly MJ, Klenzendorf S, Vaughn M, Zulfahmi, Hutajulu MB, Parakkasi K. 2013. Threatened predator on the equator: multi-point abundance estimates of the tiger *Panthera tigris* in central Sumatra. Oryx, **47**(2): 211-220.

Terborgh J, Wright SJ. 1994. Effects of mammalian herbivores on plant recruitment in two neotrpical forests. Ecology, **75**(6): 1829-1833.

Terborgh J, Lopez LV, Nunez P, RaoM, Shahabuddin G, Orihuela G, Riveros M, Ascanio R, Adler G, Lambert TD, Balbas L. 2001. Ecological meltdown in predator-free forest fragments. Science, **294**(5548): 1923-1926.

Tilson R, Nyhus P, Sriyanto, Rubianto A. 2010. Poaching and poisoning of tigers in Sumatra for the domestic market. In: Tilson R., Nyhus PJ, eds. Tigers of the World: The Science, Politics, and Conservation of Panthera tigris, 2nd edn. Academic Press, Burlington, MA, USA, pp. 101–12.

Tilso R, Traylor-Holzer K. 1994. Estimating poaching and removal rates of tigers in Sumatra. In (R. Tilson, K. Soemarna, W. Ramono, S. Lusli, K. Traylor-Holzer and U. Seal, eds.) Sumatran Tiger Population and Habitat Viability Analysis Report, pp. 75-76. IUCN/SSC Conservation Specialist Group. Apple Valley, Minnesota.

Wikramanayake E, McKnightM, Dinerstein E, Joshi A,Gurung B, Smith D. 2004. Designing a conservation landscape for tigers in human-dominated environments. Conservation Biology, **18**: 839–844.

Figures

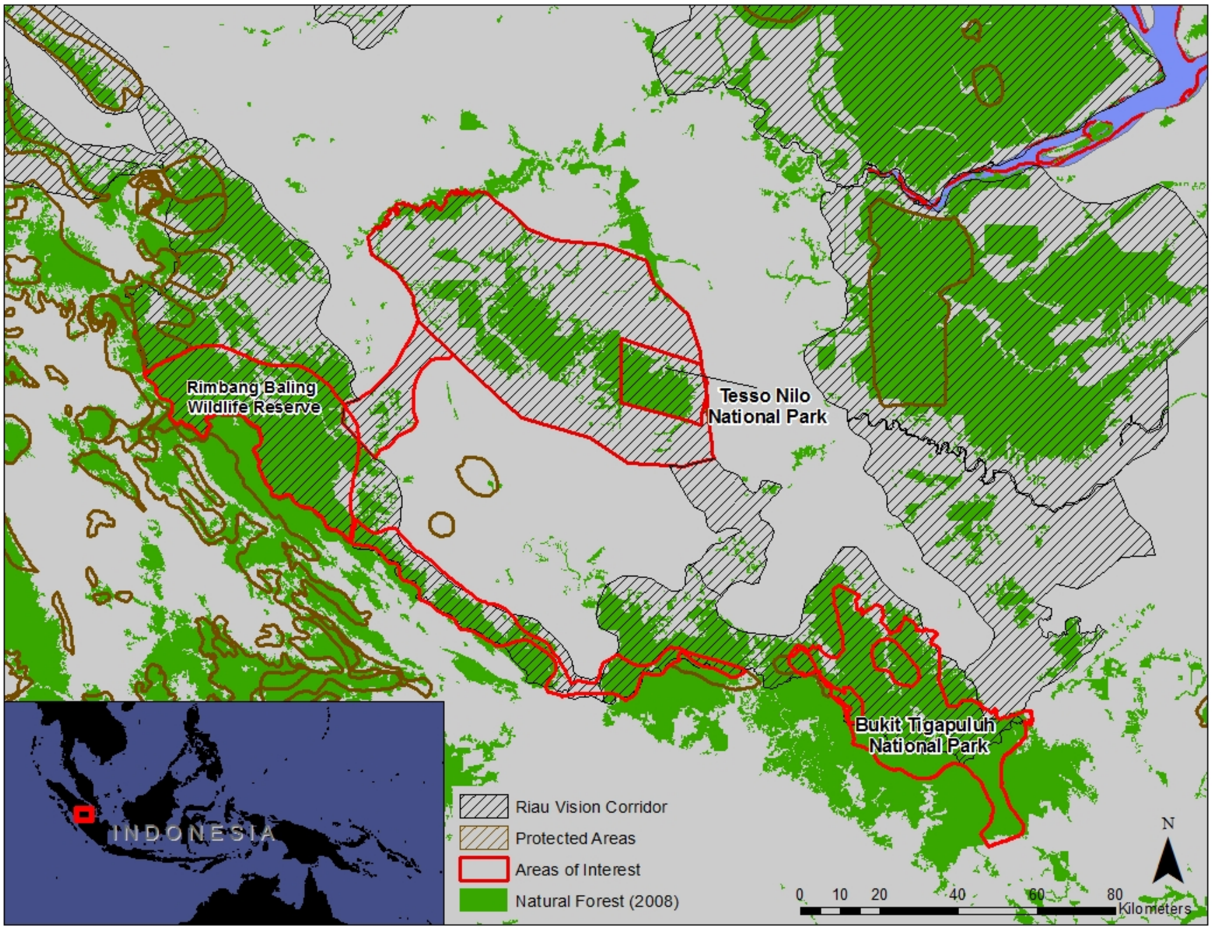

Figure 1. Area of interest in central Riau province, Sumatra, Indonesia.

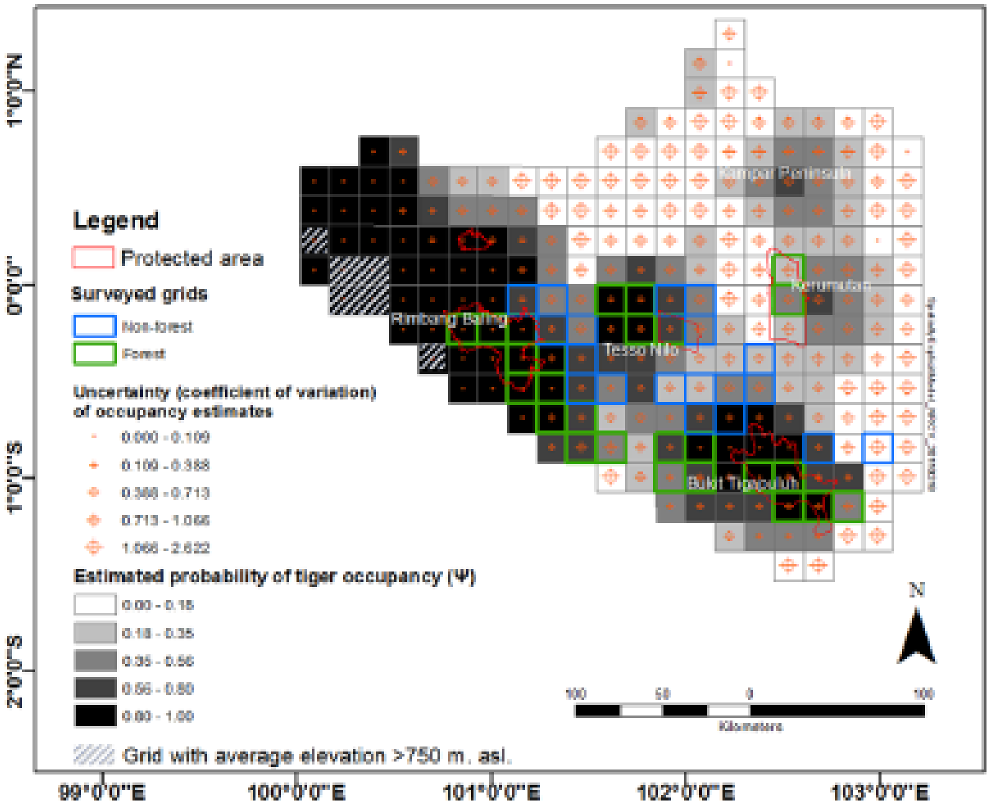

Figure 2. Map of probability of tiger occupancy in the central Sumatra landscape. This map is constructed from the best occupancy model developed based on the landscape-scale survey in 17×17 km grid cells representing forest and other major landcover types.

(With permission from Sunarto et al. 2011)

Chapter 2

Mapping and predicting forest loss in a Sumatran tiger landscape from 2002 – 2050

Erin E. Poor, Yang Shao, Marcella J. Kelly

Abstract

Riau Province in central Sumatra, with its peatland, lowland, and montane forest habitats, was once a stronghold for Sumatran tiger (*Panthera tigris sumatrae)* populations. Today, Riau may have one of the highest deforestation rates in the world and wildlife populations are dwindling, with natural forest now comprising approximately only 18% of the province, mostly contained within protected areas. Agriculture (acacia, rubber, and oil palm) makes up the majority of Riau's land cover and deforestation for the creation of new plantations is rampant. Natural forest and tigers still remain in Bukit Tigapuluh National Park and Rimbang Baling Wildlife Reserve, which remain connected to tiger populations in montane forest on the western edge of Sumatra. In this study, using freely available Landsat imagery and a maximum likelihood classification algorithm, we create land cover maps for central Sumatra from 2002 – 2016. We then use current land cover, elevation, and slope variables to predict changes from forest to plantation from 2016 – 2050 at five year intervals using a multilayer perceptron neural network. Finally, we compare connectivity based on a 100 km distance threshold (based on potential tiger dispersal) across the landscape and across years. Land cover maps had 80-90% accuracy, and we predict forest in Tesso Nilo and the western edge of the study area to be lost by 2050 given current rates of deforestation. Our connectivity analysis shows that Tesso Nilo and the area between Rimbang Baling and Bukit Tigapuluh are important components for maintaining connectivity throughout the study area. Focusing conservation and rehabilitation efforts on forests close to plantations in flat areas, including Tesso Nilo, is necessary to maintain forests and increase connectivity in

Riau to ensure future habitat connectivity for survival of tigers and Sumatra's other diverse endemic species.

Keywords: connectivity, deforestation, land cover map, land cover prediction, palm oil

Introduction

Contraction and modification of natural habitat by human development is one of the main factors driving small isolated wildlife populations to extinction (Ferreras et al. 2001). Because mammals may be more sensitive to forest disturbance than other taxa (Sodhi et al. 2009), they are likely be the most affected by an increase in development, with estimated losses of 21-48% of Southeast Asian mammals by 2100 (Brooks et al. 1999). Species that are wide-ranging and that exist in low density like many carnivores are often the first to go extinct when habitat is fragmented by development (Beier 1993; Pimm & Clark 1996; MacNally & Bennett 1997).

Carnivore guilds found in Southeast Asia are higher in diversity than on other continents, yet many Asian carnivore species now occur at population sizes too small to fulfill their past ecological functions (Dalerum et al. 2009). As roads and railroads continue to bisect habitat, dams flood habitat, and mines and towns fragment habitats, the overall decrease in habitat and the increased distance among habitat patches will lead to reduced carnivore presence (Crooks 2002; Mortelliti & Boitani 2008). Although carnivore presence often correlates with prey abundance, below a certain patch size threshold, use of habitat may completely stop regardless of prey abundance (Mortelliti & Boitani 2008). In a comparison of a protected area with many small (<100 ha) patches to one with fewer large (>400 ha) habitat patches in Thailand, Pattanavibool and Dearden (2002) found the protected area with large patches still contained large mammals that had been extirpated from the more fragmented protected area. Preserving connectivity among isolated patches of habitat during the early stages of degradation is one of the most important factors in conserving endangered carnivore species such as the tiger (*Panthera tigris)* (Carroll et al. 2004).

Sumatra holds all of Indonesia's remaining tigers in approximately 88,000 km^2 (Sanderson et al. 2006) but agricultural conversion of tiger habitat has been a growing threat to Sumatran tigers (*P. t. sumatrae*) over the past few decades. In comparison with other tiger habitats across South East Asia, tiger habitat in central Sumatra experienced one of the greatest deforestation rates since 2000 (Joshi et al. 2016). Oil palm was first planted in Sumatra in 1911 (Corley & Tinker, 2003) and there are now approximately 6.1 million ha of oil palm in Indonesia (FAO 2006). From 1990-2005, at least 56% of oil palm expansion replaced forest (Koh & Wilcove, 2008). While tigers have been seen in oil palm, overall, oil palm plantations support fewer than 50% of vertebrate species as primary forests (Danielsen et al. 2009), have lower species richness than disturbed forests, and support fewer species than other types of agriculture (Fitzherbert et al. 2008). Loss of species diversity in oil palm plantations may be due to a loss in structural complexity and plant species richness that occurs when plantations are productive (Chung et al. 2000; Glor et al. 2001; Aratrakorn et al. 2006).

In Indonesia, like many developing countries, funds for habitat protection and enforcement are lacking, and anecdotal evidence and regional national land cover data show a decrease in forest. Tigers have been observed in oil palm plantations, but Sunarto et al. (2012), found that tigers were more likely to use forest than any other land cover type, followed by acacia, oil palm, rubber, and mixed agriculture in their study. Furthermore, Yaap et al. (2016) showed that a wide diversity of mammals use forest remnants outside of national parks or core forest areas, but species richness increased when in larger patches or closer to larger forest blocks. In addition, when compared to forest remnants >2 km away from core forest, tiger, clouded leopard (*Neofelis diardi*), and leopard cat (*Prionailurus bengalensis*) were all only found in remnant patches within 1 km of core forest, underscoring the importance of accessible natural habitat (Yaap et al. 2016).

While land cover maps have been created for Sumatra and the greater South East Asia region (Gaveau et al. 2009; JAXA 2010), these maps have been relatively low in resolution (e.g., 800 m and 250 m). In order to more accurately assess loss of tiger habitat in Riau province, we created finer scale land cover maps using Landsat 5, 7, and 8 imagery for 2002, 2010, 2013/14, and 2016, to build on Gaveau et al. (2009) and track changes in potential tiger habitat since the rise of oil palm plantations in the early 2000's. In order to completely assess habitat connectivity, both structural and functional connectivity should be quantified. Structural connectivity may be described as the structure of the habitat with respect to, but independent of, species, while functional connectivity describes the behavioral response of the species or animal to the habitat (Tischendorf and Fahrig 2000). Although equally or perhaps more important than structural connectivity, we were unable to assess functional connectivity in the current study. As a first step to identifying habitat connectivity for tigers as a whole, here we quantify structural connectivity and predict future habitat connectivity in Riau, using natural forest as a proxy for habitat.

Methods

Study Area

Riau Province is in central Sumatra (Figure 1), bordered on the west by the Barisan mountain range and West Sumatra province, and on the east by peatlands and the South China Sea. Riau's climate is classified in the Koppen-Geiger system as Af, tropical. Average temperature is 27° C while average rainfall is 2696 mm per year. The network of protected areas in Riau is centered by Tesso Nilo National Park, lowland tropical rainforest. Bukit Tigapuluh National Park is to the southeast of Tesso Nilo and Rimbang Baling Wildlife Reserve is southwest of Tesso Nilo. Both Bukit Tigapuluh and Rimbang Baling are primarily comprised of montane rainforest and may provide connections from the mountainous forests of the Barisan range along western Sumatra to Tesso Nilo and Kerumutan Wildlife Reserve, mostly peast swamp forest, on the eastern side of Riau. Within the Sundaland biodiversity hotspot (Myers 2000), this area still contains endangered and endemic species such as Sumatran tiger, Malayan tapir (*Tapirus indicus*), Sumatran elephant (*Elephas maximus sumatranus*), Sunda clouded leopard (*Neofelis diardi*), and Sunda pangolin (*Manis javanica*).

Data

We searched for cloud-free Landsat imagery of our study area in the USGS GLOVIS website. Three Landsat scenes were needed to cover our study site. For 2002, we used one image per scene, all from the 2002 dry season. Due to frequent cloud cover in the tropics and smoke cover from slash and burn agricultural practices in Sumatra during the end of the dry season (May – October), images from 2010 were used for the Rimbang Baling and Tesso Nilo scenes, while 2009 and 2011 images were used for the Bukit Tigapuluh scene for the second time step. Similarly, for the third time step, 2013/2014, we used images from 2014 for Tesso Nilo and from 2013 for Bukit Tigapuluh and Rimbang Baling. For the last time step, we used multiple 2016 images to form the Rimbang Baling and Bukit Tigapuluh datasets (Table 1). Therefore, we created land cover maps for four times steps, which varied depending on availability of cloud-free data: 2002, 2009/2010/2011, 2013/2014 and 2016. Land cover was created at 30 m resolution. We created distance to open land and distance to plantation variables using these land cover maps at 30 m resolution. Elevation and slope were derived from ASTER GDEM V2 2011 data (JAXA 2011) at 30 m resolution.

Land Cover Mapping

For image preparation, we atmospherically corrected to top of atmosphere reflectance using the Radiometric Calibration tool in ENVI software package. These reflectance bands were then put into the ENVI Fmask tool (Zhu et al. 2015) to identify cloud and shadow. We classified each image separately using a maximum likelihood algorithm. To improve classification accuracy we incorporated a DEM as additional input to classify the 2016 Bukit Tigapuluh scene.

We conducted accuracy assessments for 2013/2014 and 2016 image classifications obtained from ground surveys in March 2015 – July 2016. Because teams were surveying for felid scat and signs in forested areas, ground truth points were biased for forest land cover. Although we were able to collect more plantation ground truth points when surveying roads on motorbike, field teams did not feel confident they could safely enter plantations away from roads without being questioned or instigating conflict from plantation workers. After observing low accuracy within the oil palm and bare classes, we shifted the oil palm points 90 m to the west, accounting for collecting ground truth data along roads in plantations and for the low/open ground cover often found along roads in plantations. Since field work began in 2015, field ground truth data were unavailable for 2002, 2009/2010/2011 and 2013/2014. We digitized ground truth points using ArcMap 10.4 for forest, plantations, and open/bare land using visual interpretation of the 2014 image and field knowledge. Using 380-970 points (Table 1) as a reference for all 2013/2014 and 2016 images, we generated error matrices (Supplementary Material). We were unable to assess accuracy for our 2002 and 2009/2010 images due to lack of ground truth data and lack of familiarity with the landscape at this time. Our image classification methods were the same across years, and hence we assume similar accuracy levels from the 2002 and 2009/2010 imagery.

Land Cover Prediction

We used IDRISI’s TerrSet Land Change Modeler (LCM) (Eastman 2009) to model land cover change. LCM allows modeling of non-linear relationships between predictor and response variables through its multi-layer perceptron neural network algorithm. Additional advantages of LCM include its easy user interface and its multiple accuracy assessment and validation tools (Pontius et al. 2008; Mas et al. 2014). The LCM has also been used to describe changes in tiger habitat elsewhere (Areendran et al. 2017), and has performed well in the tropics (Koi & Murayama 2010; Fuller et al. 2011; Perez-Vega et al. 2012).

To calibrate the land cover change model, we used the 2002 and 2013 land cover maps. Land cover classes included forest, water, plantation or non-forest vegetation, and open/bare land. We created a deforestation sub-model, to model the transitions of forest to plantation. Predictors were included based on Cramer’s V (Table 2) and their potential to impact change on the landscape. Cramer’s V is a measure of correlation between two variables, ranging from 0 (no correlation) to 1 (identical variables). Our chosen variables included distance to open areas, distance to plantation, elevation, slope, distance to major roads, and distance to forest. Distance variables were natural log transformed. The land cover variables were selected as dynamic variables that change with changing land cover. Due to the lack of enforcement of protected areas, the high human habitation within the parks and the high human activity in and around parks, we do not include protection status as a variable and we assume that the rate of change inside parks is similar to that outside of officially protected areas. We modeled roads as static variables. Given past trends in Indonesia, and the length of time new infrastructure projects take to complete or even initiate due to bureaucracy, land tenure conflicts, and funding issues, we assume that no new major infrastructure (save for the possibility of the Trans-Sumatra Toll

Road, which will largely overlap current roads) will be built within the near future.

We used a multi-layer perceptron neural network (MLPNN) to model transition potentials (the simulation portion of our model) with 10,000 cells (50% training and 50% testing) per land cover class and 10,000 iterations. An MLPNN is an assumption-free, machine-learning algorithm used to model non-linear relationships through multiple non-linear algorithms and generalize these relationships with novel data (Gardner & Dorling 1998). We modeled forest change from 2013-2016 using the modeled transition potentials from 2002-2013. To validate the model, we compared the 2016 predicted land cover to the 2016 land cover created using Landsat imagery. We validated the model using the receiver operating characteristic (ROC) curve, comparing actual 2013-2016 forest to plantation change with the predicted map of 2016 plantations within the previously forested area (Pontius & Schneider 2001). We then created land cover maps for every five-year interval from 2020 – 2050, with one recalculation stage for each interval. Although the rate of deforestation may not remain constant over time, our model assumes that the rate of change does remain constant. We recognize the rate of change is likely to vary, and as with any predictive modeling, our model uncertainty will be higher for the predictions into the more distant future than it will be for the predictions of the near future. These assumptions are drawbacks of land change modeling and prediction, and we emphasize here that we are predicting land cover under a business-as-usual scenario. See Mas et al. (2014) for a visual depiction of this complete process.

Habitat Connectivity

The simplest method to measure connectivity among habitat patches is to use a Euclidean distance measure (Moilanen & Hanski 2001), and that is what we use here to describe structural connectivity in this landscape, in the absence of species data. Due to the resolution of our land cover data, we assume ‘habitat’ is forested area. Habitat connectivity measures may be calculated by treating the landscape as a graph of nodes (habitat patches) and links (paths, or distances between habitat patches) (Urban & Keitt 2001). In identifying specific patches important for maintaining habitat connectivity, we first identified forest patches greater than 0.5 km^2 to increase processing speeds. We then used Conefor Sensinode 2.6 (Saura & Torne 2009) to calculate the betweenness index (BC) (Bodin & Saura 2010) and the integral index of connectivity (IIC) (Pascual-Hortal & Saura 2006) for the actual 2016 landscape and the 2050 predicted landscape to measure the predicted change in structural connectivity of this landscape over time. The improved betweennesss index (BC(IIC)) is a measure of node connectivity, and measures the number links in a path passing through a respective patch while taking the patch’s area into account (Bodin & Saura 2010). Shorter paths indicate higher connectivity and are given a higher weight. A patch with a high BC(IIC) can be considered better connected than a patch with a low BC(IIC) measurement. While many habitat connectivity metrics are not sensitive to important changes that impact connectivity negatively, the IIC takes patch area, landscape area, and path distances into account, making it sensitive to fragmentation (Pascual-Hortal & Saura 2006). This metric can also be used as a general measure of habitat connectivity; an increase in IIC indicates an increase in connectivity, whereas a decrease indicates a connectivity decline landscape-wide (Pascual-Hortal & Saura 2006). Because our aim is to determine whether forest is still connected in this landscape despite human modification with respect to tigers, we used a 100 km distance threshold, assuming this is the approximate maximum distance tigers can disperse in this landscape (Smith 1993; Wang et al. 2015). With this assumption, forest patches within 100 km of each other are considered connected for tigers, and more distant patches may be connected to each other through a network of intermediate patches if these intermediate patches are located within the 100 km potential dispersal distance with respect to one another, i.e. a single link between two patches can be 100 km, maximum.

Results

Land Cover Mapping

Our land cover classification accuracies gathered from ground truth and digitized validation points ranged from 81.05% to 92.06%. Land cover mapping in 2016 for Bukit Tigapuluh proved challenging, requiring the use of three Landsat images and elevation data to achieve accuracies in the 80%'s (Table 1). From 2002 – 2016, 34.55% of forest has been lost in our study area (Figure 2).

Land Cover Prediction

The MLPNN transition potential model final accuracy rate gathered from validation procedures was 71.75%. Slope was the most influential variable in the model, while distance to forest was least influential. The area under the curve (AUC) for predicted new plantation from forested areas was 68%. The model over-predicted forest loss within Tesso Nilo in comparison with our Landsat-based land cover maps (Figure 3), with 58.19% of forest predicted to be lost from 2016 - 2050. Small remnant patches of forest seem to remain in Tesso Nilo through 2040, and a small fragment remains through 2050. Our model also under-predicts the amount of forest in Kerumutan, which, despite being surrounded by acacia plantations, does still contain natural peat forests. The most significant losses in forest are predicted to be in the northeast corner of the study area, near Kerumutan, an area rich in peat, and in the forest remnants between Bukit Tigapuluh and Rimbang Baling. In 2050, the models predict forest will remain in Rimbang Baling and Bukit Tigapuluh, presumably due to higher elevation and steeper slopes (Figure 3).

Habitat Connectivity

Overall, habitat connectivity decreased from 2016 – 2050. In both time periods, all habitat was connected given the 100 km threshold distance (Figure 4). However, the IIC, a relative measure of habitat connectivity, decreased by nearly 92% (Figure 4). In 2016, the remaining forest of Tesso Nilo and the forest corridor between Rimbang Baling and Bukit Tigapuluh had the highest BC(IIC), indicating their importance for maintaining connectivity between forest across the landscape. However, in 2050, the forest of Tesso Nilo is lost along with the forest northeast of it, and the remaining patches between Rimbang Baling and Bukit Tigapuluh have the highest BC(IIC) within the landscape.

Discussion

The land cover maps that we created as part of this study provide a novel, fine-scale analysis of central Riau's changing land cover since the expansion in palm oil plantations. While we believe that the land cover maps we created from ground truth data are the first accuracy assessed set of maps dating from 2002 – 2016, there is also room for improvement. Validation procedures for land cover change predictions are still debated (Pontius et al. 2004), and typical statistical validation procedures such as k-fold cross validation are insufficient due to the possibility of spatial and land cover class quantity errors (Pontius et al. 2004; Pontius et al. 2008). Improved validation techniques could alter our results. Additionally, distinguishing between natural forest, oil palm plantation, and acacia plantation proves challenging, but was improved with the use of all Landsat bands and in some cases, elevation data. Budgetary restrictions forced us to use freely available, lower resolution Landsat imagery, but our methods are easily repeatable for use in other parts of Sumatra, free of imagery costs, by those wishing to replicate this study. The other challenge we faced when creating the land cover maps included extensive cloud or smoke cover. We attempted to use scenes from the dry season, where cloud cover was less significant, but in some years (2013/2014, 2016), smoke from slash-and-burn land clearing covered significant areas of Landsat scenes. To remedy this, future modelers may incorporate radar data, which can be used regardless of cloud or smoke cover.

Our models predict less forest in 2016 than our mapped forest, and they predict a rapid decrease in forest of Kerumutan, to the northeast of Tesso Nilo. This may be due to the inaccuracies of the input land cover map. Distinguishing between acacia and natural forest proved difficult, and we believe our maps may have slightly overestimated the amount of acacia near Kerumutan. While there is a substantial amount of acacia forest in this area, future transitions of additional forest may be slow due to the alteration and draining of the land in this particular area that is required before a first planting. The over-prediction of forest loss may also be attributed to a possibly higher deforestation rate from 2002-2013 (used for model calibration) than observed from 2013-2016 (model validation). Future work could focus on incorporating socio-economic drivers of land cover change such as the price of palm oil and land tenure regulation enforcement efforts, and could include a sensitivity analysis by adjusting deforestation rates during calibration to compare potential changes to model outputs.

Although all patches are connected under the 100 km distance threshold, (used as a ‘best-case’ maximum movement distance for tigers), this assumes that wildlife moves from one habitat patch in a straight line to the next closest as they move about the landscape. However, in reality, this may not be true and wildlife may not move directly from one patch to the next closest patch, thus making the distance or effort to travel between forest patches greater. In these cases, the entire landscape may not be ‘connected’, as our connectivity indices show, under the 100 km distance threshold chosen for this landscape, and wildlife may struggle to move from forest patch to forest patch. While identifying changes in forest structural connectivity is an important first step, identifying functional connectivity of endangered species in this landscape is possibly even more important in informing conservation decisions. Our future work will focus on combining species data with this spatial data analysis to better inform conservation and management by identifying current, and creating future, corridors to enhance Sumatran tiger and other endangered and endemic felid populations.

Despite some minor inaccuracies in our model, it is still clear that if current land clearing practices continue in Riau, we stand to lose a significant amount of forest cover, which could negatively impact critically endangered and endemic wildlife that still exists in this highly modified landscape. This straightforward analysis highlights the need for immediate conservation interventions. Tesso Nilo has already lost >50% of its natural forest since 2002, and our results could be used as a worst-case scenario of forest loss, assuming the current deforestation laws in Indonesia become better enforced and deforestation slows in the future.

Generally, our model accuracies are relatively high, and, since these are the first accuracy-assessed land cover maps and the first land cover prediction maps created for Riau, we believe they can provide useful guidance to land cover management and valuable insights for areas most vulnerable to forest loss. Our models indicate that clearing for plantations is most likely to happen in flat, lowland areas near areas that are already plantations. The remaining forest within Tesso Nilo meets these prerequisites, which, when combined with its importance in maintaining landscape-wide connectivity, as a potential stepping-stone for wildlife moving from the western edge of Riau to the peatlands of the northeast, makes it a critically important patch of forest to protect. We also recommend focusing efforts on the remaining forest patches between Rimbang Baling and Bukit Tigapuluh to maintain north-south connectivity between these two mountainous protected areas that are likely to persist into the future.

Given the amount of deforestation that has already occurred within this landscape, we stress the potential role that reforestation and restoration could play in this landscape. Average forest patch size in 2016 of patches > 0.5 km^2 was just above 11 km^2. With a home range requirement of around 100 km^2 (Sunarto et al. 2012), tigers in our study area are likely already facing a habitat deficit, further supported by an observed increase in wildlife conflict in this and neighboring provinces. Tesso Nilo has already lost more than half of its forested area, and it currently is not large enough by itself to maintain one tiger, let alone a tiger population. Tigers occasionally are reported by villagers in this area and continued human population growth could lead to conflict echoing that plaguing the resident Sumatran elephant population. Restoring some areas to a forested state would provide additional habitat and potentially could mitigate or decrease future conflicts. We recommend restoring Tesso Nilo to forest, though we also recognize the social and political challenges that would accompany any restoration efforts.

Tracking deforestation and identifying areas for mitigation is extremely important throughout the tiger range, but this is just one piece of the puzzle in achieving the 'Tx2' goal of doubling the wild tiger population by 2022, put forth by the St. Petersburg Declaration in 2010. Many tiger landscapes are also experiencing high and/or increased poaching and hunting levels or pressure from more organized poaching syndicates targeting tigers or prey (Risdianto et al. 2016). If these large international, social, and legal issues are not addressed, conservation of habitat is futile. While there are countless scientists and non-governmental organizations working towards tiger conservation and Tx2, sustaining and increasing tiger populations by acting on conservation recommendations remains the responsibility of local and national governments. We hope this work highlights the urgency of the situation of forest loss in Riau and that it better informs those working on the ground as to where best to focus conservation efforts.

References

Aratrakorn S, Thunhikorn S, Donald PF. 2006. Changes in bird communities following conversion of lowland forest to oil palm and rubber plantations in southern Thailand. Bird Conservation International, **16**(1): 71–82.

Areendran G, Raj K, Mazumdar S, Sharma A. 2017. Land use and land cover change analysis for Kosi River wildlife corridor in Terai Arc Landscape of Northern India: Implications for future management. Tropical Ecology, **58**(1): 139–149.

Beier P. 1993. Determining minimum habitat areas and habitat corridors for cougars. Conservation Biology, **7**: 94–108.

Bodin O, Saura S. 2010. Ranking individual habitat patches as connectivity providers: integrating network analysis and patch removal experiments. Ecological Modeling, **221:** 2393–2405.

Brooks TM, Pimm SL, Kapos V, Ravilious CA. 1999. Threat from deforestation to montane and lowland birds and mammals in insular south-east Asia. Journal of Animal Ecology, **68**: 1061–1078.

Carroll C, Noss RF, Pacquet PC, Schumaker NH. 2004. Extinction debt of protected areas in developing landscapes. Conservation Biology, **18**:1110–1120.

Chung AYC, Eggleton P, Speight MR, Hammond PM, Chey VK. 2000. The diversity of beetle assemblages in different habitat types in Sabah, Malaysia. Bulletin of Entomological Research, **90**(6): 475–496.

Corley, RHV,Tinker, PB. 2003. The Oil Palm. Wiley-Blackwell.

Crooks KR. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology, **16**: 488–502.

Dalerum F, Cameron EZ, Kunkel K, Somers, MJ. 2009. Diversity and depletions in continental carnivore guilds. Implications for prioritizing global carnivore research. Biology Letters, **5**: 35–38.

Danielsen F, Beukema H, Burgess ND, Parish F, Bruhl CA, Donald PF, Murdiyarso D, Phalan B, Reijnders L, Struebig M, et al. 2009. Biofuel plantations on forested lands: double jeopardy for biodiversity. Conservation Biology, **23**(2): 348–358.

Eastman, J.R., 2012. IDRISI Selva (Worcester, MA: Clark University).

Ferreras P, Gaona P, Palomares F, Delibes, M. 2001. Restore habitat or reduce mortality? Implications from a population viability analysis of the Iberian lynx. Animal Conservation, **4**: 65–274.

Fitzherbert EB, Struebig MJ, Morel A, Danielsen F, Bruhl CA, Donald PF, Phalen B. 2008. How will oil palm expansion affect biodiversity? Trends in Ecology and Evolution, **23**: 538–545.

Food and Agriculture Organization of the United Nations. 2006. Global Forest Resources Assessment 2005, FAO.

Fuller DO, Hardiono M, Meijaard E. 2011. Deforestation projections for carbon-rich peat swamp forest of Central Kalimantan, Indonesia. Environmental Management, **48**: 436–447.

Gardner MW, Dorling SR. 1998. Artificial neural networks (the multilayer perceptron) – a review of applications in the atmospheric sciences. Atmospheric Environment, **32**: 2627-2636.

Gaveau DLA, Epting J, Lyne O, Linkie M, Kumara I, Kanninen M. 2009. Evaluating whether protected areas reduce tropical deforestation in Sumatra. Journal of Biogeography, **26**: 2165–2175.

Glor RE, Flecker AS, Benard MF, Power AG. 2001. Lizard diversity and agricultural disturbance in a Caribbean forest landscape. Biodiversity and Conservation, **10**(5): 711–723.

Joshi AR, Dinerstein E, Wikramanayake E, Anderson ML, Olson D, Jones BS, Seidensticker J, Lumpkin S, Hansen MC, Sizer NC et al. 2016. Tracking changes and preventing loss in critical tiger habitat. Science Advances, **2**(e1501675): 1–8.

Koh LP, Wilcove DS. 2008. Is oil palm agriculture really destroying tropical biodiversity? Conservation Letters, **1**(2): 60–64.

Koi DD, Murayama Y. 2010. Forecasting areas vulnerable to forest conversion in the Tam Dao National Park Region, Vietnam. Remote Sensing, **2**(5): 1249–1272.

MacNally R, Bennett AF. 1997. Species-specific prediction of the impact of habitat fragmentation: local extinction of birds in the box-ironbark forests of central Victoria, Australia. Biological Conservation, **82**: 147–155.

Mas J, Kolb M, Paegelow M, Teresa M, Olmedo C, Houet T. 2014. Inductive pattern-based land use / cover change models : A comparison of four software packages. Environmental Modelling & Software, **51**: 94–111.

Moilanen A, Hanski I. 2001. On the use of connectivity measures in spatial ecology. Oikos, **95**: 147–151.

Mortelliti A, Boitani L. 2007. Interaction of food resources and landscape structure in determining the probability of patch use by carnivores in fragmented landscapes. Landscape Ecology, **23**(3): 285–298.

Pascual-Hortal L, Saura S. 2006. Comparison and development of new graph-based landscape connectivity indices: towards the prioritization of habitat patches and corridors. Landscape Ecology, **21**: 959-967.

Pattanavibool A, Dearden P. 2002. Fragmentation and wildlife in montane evergreen forests, northern Thailand. Biological Conservation, **107**: 155–164

Perez-Vega A, Mas J-F, Ligmann-Zielinska A. 2012. Comparing two approaches to land use/cover change modelling and their implications for the assessment of biodiversity loss in a deciduous tropical forest. Environmental Modeling & Software, **29**(1): 11–23.

Pimm SA, Clark TW. 1996. Marking sense of the policy process for carnivore conservation. Conservation Biology, **10**: 1036–1045.

Pontius Jr. RG, Boersma W, Castella JC, Clarke K, de Nijs R, Dietzel C, Duan Z, Fotsing E, Goldstein N, Kok K, et al. 2008. Comparing the input, output, and validation maps for several models of land change. Annals of Regional Science, **42**(1): 11–37.

Pontius Jr. RC, Schneider LC. 2001. Land-cover change model validation by an ROC method for the Ipswich watershed, Massachusetts, USA. Agriculture Ecosystems & Environment, **85**: 239-248.

Risdianto D, Martyr DJ, Nugraha RT, Harihar A, Wibisono HT, Haidir IA, Macdonald DW, D'Cruze N, Linkie M. 2016. Examining the shifting patterns of poaching from a long-term law enforcement intervention in Sumatra. Biological Conservation, **204**: 306-312.

Sanderson EJ, Forrest J, Loucks C, Ginsberg J, Dinerstein E, Seidensticker J, Leimgruber P, Songer M, Heydlauff A, O'Brien T, et al. 2006. Setting priorities for the conservation and recovery of wild tigers: 2005-2015. WCS, WWF, Smithsonian and NFWF-STF.

Santika T, Meijaard E, Budiharta S, Law EA, Kusworo A, Hutabarat JA, Indrawan TP, Struebig M, Raharjo S Huda I, et al. 2017. Community forest management in Indonesia: Avoided deforestation in the context of anthropogenic and climate complexities. Global Environmental Change, **46**: 60-71.

Saura S, Torne J. 2009. Conefor Sensinode 2.2 : A software package for quantifying the importance of habitat patches for landscape connectivity. Environmental Modelling & Software, **24**: 135–139.

Smith JLD. 1993. The role of dispersal in structuring the Chitwan tiger population. Behavior, **124**(3/4): 165–195.

Sodhi NS, Lee TM, Koh LP, Brook BW. 2009. A meta-analysis of the impact of anthropogenic forest disturbance on Southeast Asia's biotas. Biotropica, **41**: 103–109.

Sunarto, Kelly M, Parakkasi K, Klenzendorf S, Septayuda E, Kurniawan H. 2012. Tigers need cover: multi-scale occupancy study of the big cat in Sumatran forest and plantation landscapes. PLoS ONE, **7**(1): e30859.

Tischendorf L, Fahrig L. 2000. How should we measure landscape connectivity? Landscape Ecology, **15**: 633–641.

Urban DL, Keitt T. 2001. Landscape connectivity: a graph-theoretic perspective. Ecology, **82**: 1205-1218.

Wang T, Feng L, Mou P, Wu J, Smith JLD, Xiau W, Yang H, Dou H, Zhao X, Cheng Y, et al. 2015. Amur tigers and leopards returning to China: direct evidence and a landscape conservation plan. Landscape Ecology, **31**(3): 491–503.

Yaap B, Magrach A, Clements GR, Mcclure CJW, Paoli GD, Laurance WF. 2016. Large Mammal Use of Linear Remnant Forests in an Industrial Pulpwood Plantation in Sumatra, Indonesia. Tropical Conservation Science, **9**(4): 1-13.

Zhu Z, Wang S, Woodcock CE. 2015. Improvement and expansion of the Fmask algorithm: cloud, cloud shadow, and snow detection for Landsats 4-7, 8, and Sentinel 2 images. Remote Sensing of Environment, **159**: 269–277.

Tables

Table 1. Year of Landsat image used for land cover mapping for each of the three study areas, as well as the accuracy and kappa statistics for 2014 (we were unable to accuracy assess the two earliest time periods). The number of ground truth (GT) points and the percent of ground truth points collected from field work are also reported. If points were not collected from field work, they were digitized using Landsat imagery.

Protected Area	Accuracy 2014	Kappa 2014	% True GT (Total)	Accuracy 2016	Kappa 2016	% True GT (Total)
Tesso Nilo *(2002, 2010, 2014, 2016)*	92.06	0.8986	0% (380)	81.05	0.6992	25% (970)
Rimbang Baling *(2002, 2010, 2013, 2016(2))*	81.59	0.7696	0% (668)	84.38; 82.34	0.8113; 0.7844	19% (576); 23% (467)
Bukit Tigapuluh *(2002, 2009/2011, 2013, 2016(3))*	84.04	0.7809	0% (589)	81.07; 81.24; 86.07	0.7647; 0.7603; 0.8113	34% (693); 35% (673); 47% (499)

Table 2. Overall Cramer's V values for all variables in the transition potential land cover change prediction model, used to predict forest to plantation transitions from 2016 – 2050 in central Sumatra.

Variable	Overall Cramer’s V
Distance to 2002 open/bare land (ln)	0.2196
Elevation	0.2095
Distance to 2002 plantations (ln)	0.2056
Distance to major roads (ln)	0.1895
Slope	0.1847
Distance to forest (ln)	0.2039

Figures

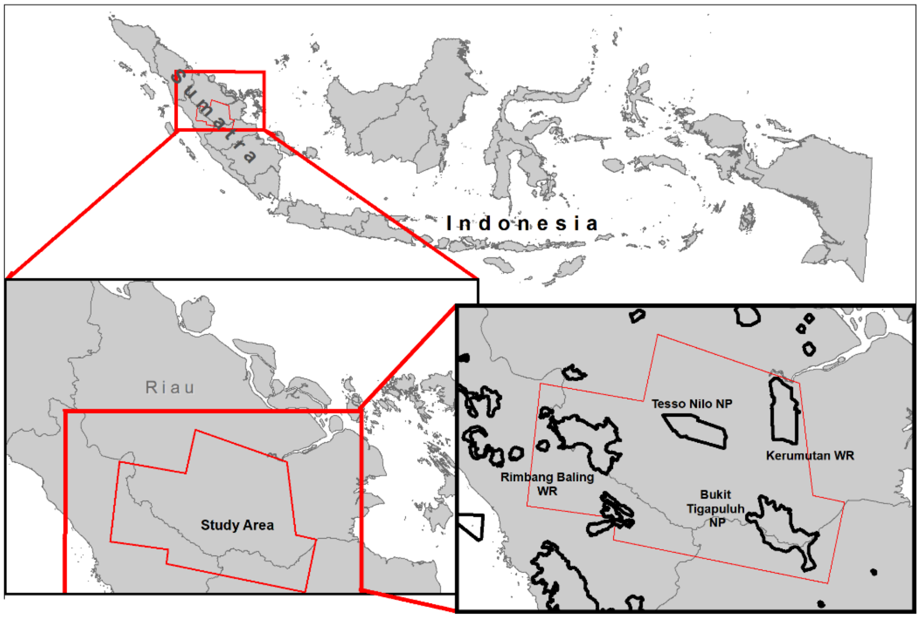

Figure 1. Location of study area. Study area, which encompasses Tesso Nilo National Park, Rimbang Baling Wildlife Reserve and Bukit Tigapuluh National Park and protected areas of interest within central Sumatra, Indonesia.

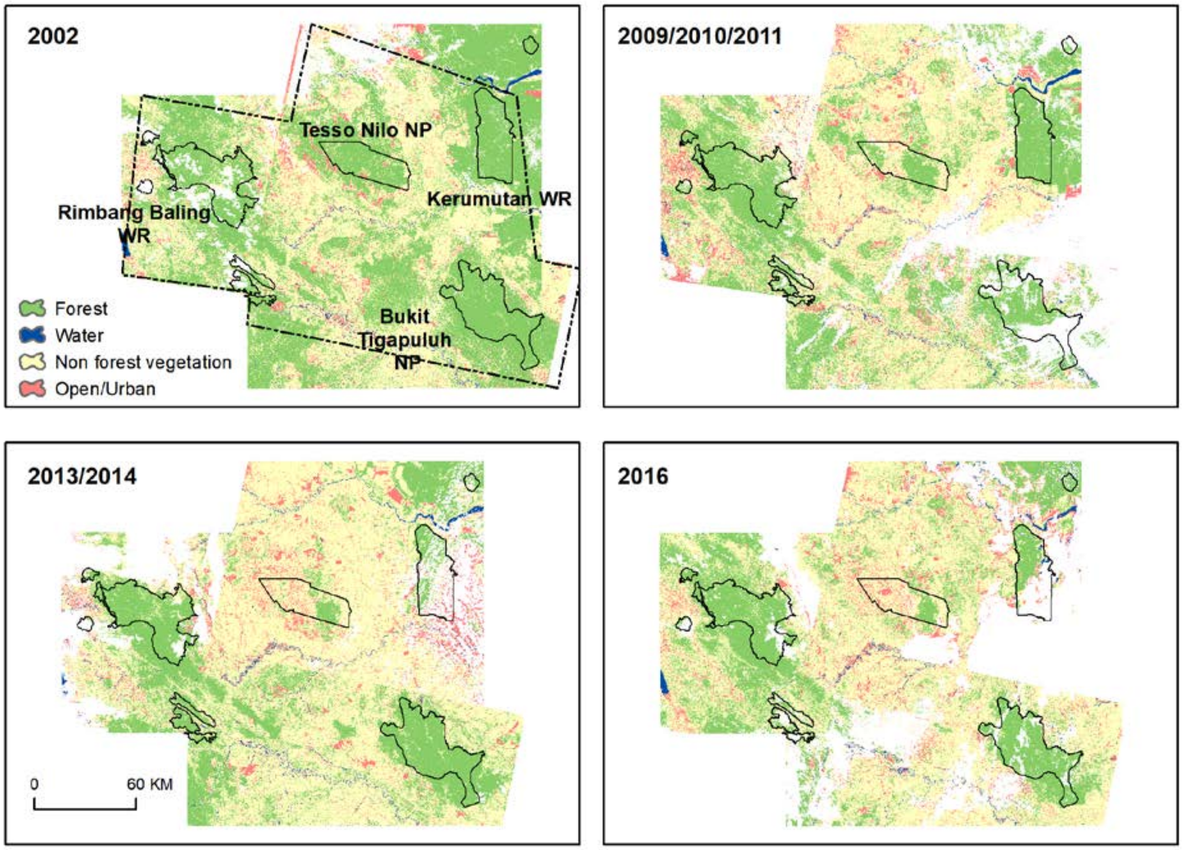

Figure 2. Land cover maps. Land cover for the larger central Sumatra study area (dotted black line, top left) and focal protected areas, Tesso Nilo National Park, Kerumutan Wildlife Reserve, Bukit Tigapuluh National Park and Rimbang Baling Wildlife Reserve from 2002 – 2016, created using three Landsat scenes, all of which have >80% accuracy. White areas within the study area indicate cloud cover.

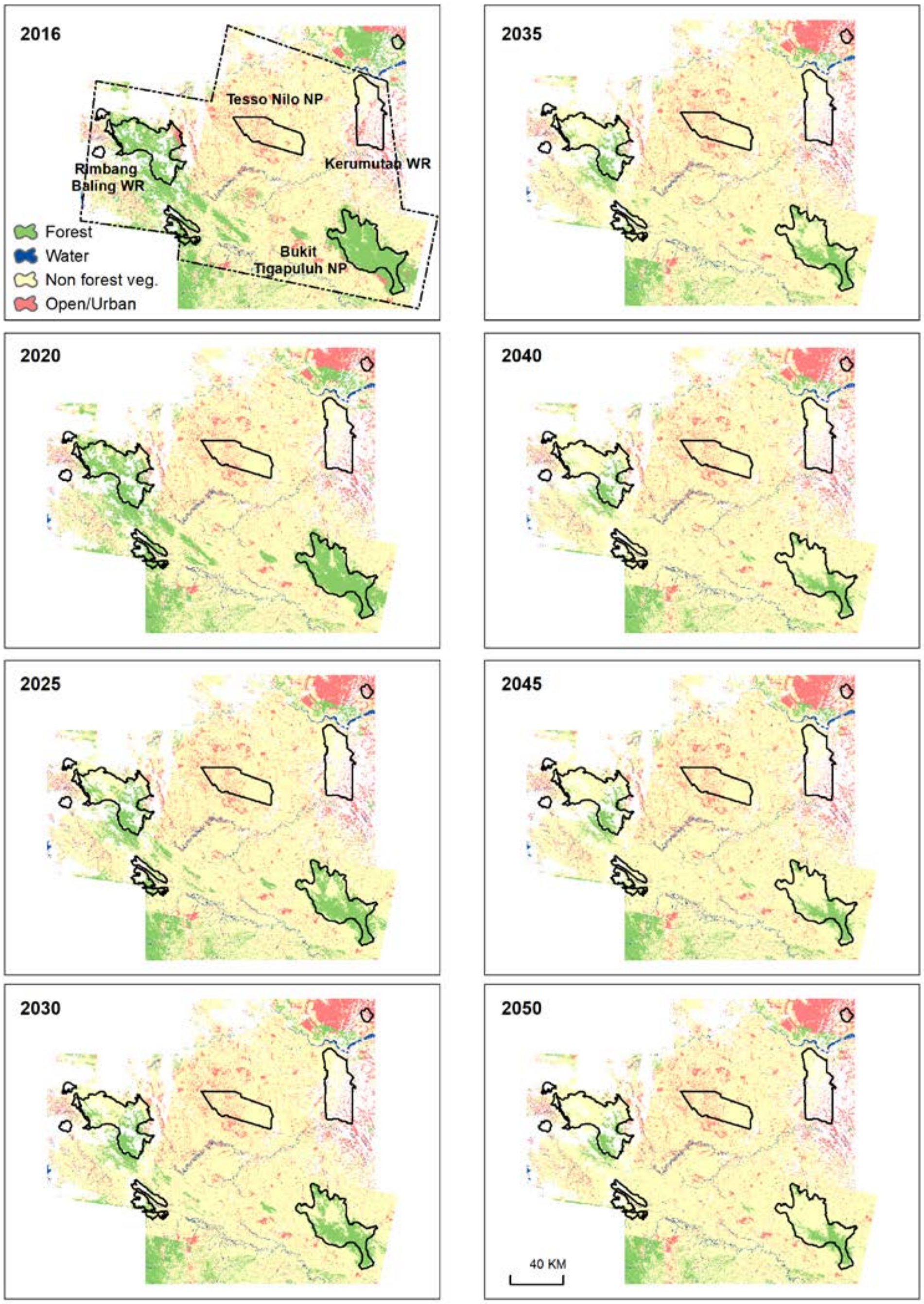

Figure 3. Land cover predictions. Predictions created using a multilayer perceptron neural network for Tesso Nilo National Park, Kerumutan Wildlife Reserve, Bukit Tigapuluh National Park and Rimbang Baling Wildlife Reserve in Riau, Sumatra, with 2013-2016 used as a validation time period. White areas within the study area (top left; dotted line) are clouds.

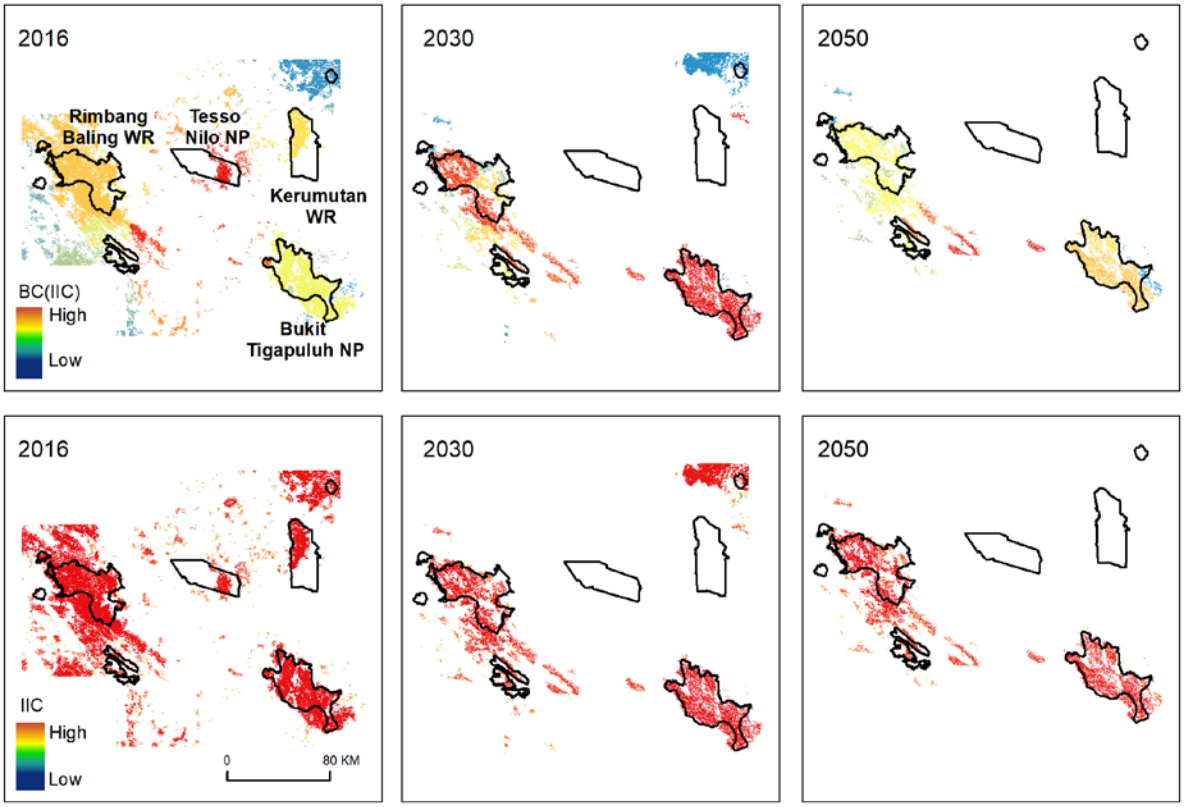

Figure 4. Landscape connectivity. Two measures of connectivity, betweenness connectivity index (BC(IIC)) (top row) and the integral index of connectivity (IIC), where high values indicate high connectivity, for forest patches < 0.5 km^2 across central Sumatra and a study area including focal protected areas Tesso Nilo National Park, Kerumutan Wildlife Reserve, Bukit Tigapuluh National Park and Rimbang Baling Wildlife Reserve, from 2016 (actual land cover) through 2050 (predicted).

Supplementary Material

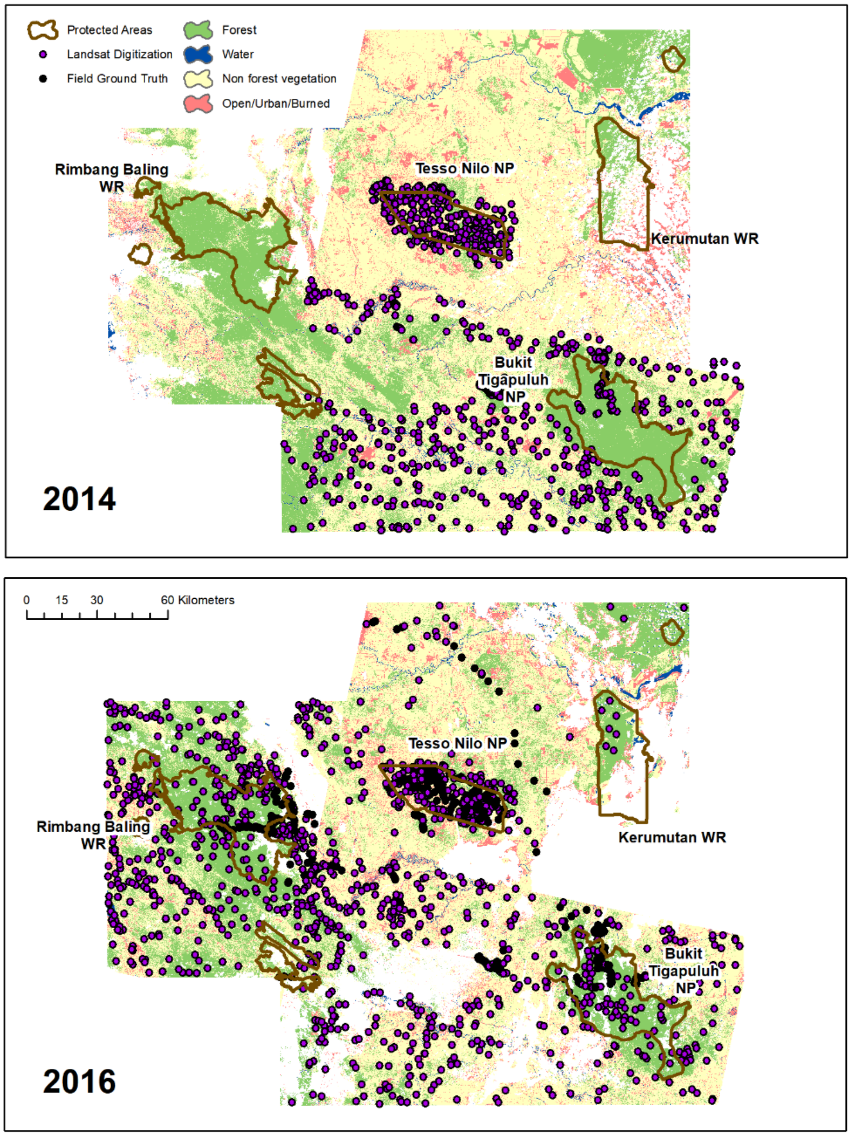

Figure S1. Locations of ground truth locations and digitized ground truth locations for the study area in Riau, Province, central Sumatra.

Table S1. Accuracy assessment for 2014 land cover classification for a region including Tesso Nilo National Park, Riau, Sumatra.

	Ground Truth					
Class	Burn/Open	Forest	Plantation	Shadow	Water	Total
Burn/Open	99	0	6	0	8	113
Forest	0	51	5	0	0	56
Plantation	2	3	91	0	0	96
Shadow	0	0	0	55	4	59
Water	1	0	0	1	52	54
Total	102	54	102	56	64	378

Table S2. Accuracy assessment for 2016 land cover classification for a region including Tesso Nilo National Park, Riau, Sumatra.

	Ground Truth				
Class	Burn/Open	Forest	Plantation	Water	Total
Burn/Open	101	0	44	2	147
Forest	5	118	43	1	167
Plantation	10	23	316	2	351
Water	2	0	2	38	42
Total	118	141	405	43	707

Table S3. Accuracy assessment for 2013 land cover classification for a region including Rimbang Baling Wildlife Reserve, Riau, Sumatra.

	Ground Truth						
Class	Burn/Open	Cloud	Forest	Plantation	Shadow	Water	Total
Burn/Open	141	10	0	11	0	14	176
Cloud	3	65	0	8	1	0	77
Forest	0	1	73	37	10	3	124
Plantation	0	3	2	169	0	3	177
Shadow	0	4	0	4	51	5	64
Water	1	0	0	1	2	46	50
Total	145	83	75	230	64	71	668

Table S4. Accuracy assessment for 2016 land cover classification for a region including Rimbang Baling Wildlife Reserve, Riau, Sumatra.

	Ground Truth						
Class	Forest	Burn/Open	Plantation	Shadow	Water	Cloud	Total
Forest	80	2	14	0	2	0	98
Burn/Open	0	84	0	0	28	0	112
Plantation	7	6	55	0	10	3	81
Shadow	1	0	0	109	3	2	115
Water	0	0	0	0	30	0	30
Cloud	0	2	0	0	2	85	89
Total	88	94	69	109	75	90	525

Table S5. Accuracy assessment for 2016 land cover classification for a region including Rimbang Baling Wildlife Reserve, Riau, Sumatra.

	Ground Truth						
Class	Cloud	Forest	Burn/Open	Plantation	Shadow	Water	Total
Cloud	44	0	0	0	0	0	44
Forest	0	45	1	9	0	0	55
Burn/Open	3	0	70	5	0	20	98
Plantation	12	1	7	57	0	6	83
Shadow	0	3	1	0	91	0	95
Water	0	0	2	0	1	24	27
Total	59	49	81	71	92	50	402

Table S6. Accuracy assessment for 2013 land cover classification for a region including Bukit Tigapuluh National Park, Riau, Sumatra.

	Ground Truth						
Class	Cloud	Shadow	Forest	Burn/Open	Plantation	Water	Total
Cloud	52	0	0	0	0	3	55
Shadow	0	9	0	0	0	0	9
Forest	0	0	83	0	29	0	112
Burn/Open	2	0	0	75	31	3	111
Plantation	3	0	2	0	227	0	232
Water	0	0	0	18	3	49	70
Total	57	9	85	93	290	55	589

Table S7. Accuracy assessment for 2016 land cover classification for a region including Bukit Tigapuluh National Park, Riau, Sumatra.

	Ground Truth						
Class	Cloud	Shadow	Burn/Open	Forest	Plantation	Water	Total
Cloud	48	0	10	10	6	2	76
Shadow	0	50	0	1	1	0	52
Burn/Open	3	0	111	1	28	4	147
Forest	0	0	3	84	9	0	96
Plantation	0	0	10	12	112	1	135
Water	1	0	4	0	0	49	54
Total	52	50	138	108	156	56	560

Table S8. Accuracy assessment for 2016 land cover classification for a region including Bukit Tigapuluh National Park, Riau, Sumatra.

	Ground Truth					
Class	Shadow	Burn/Open	Forest	Plantation	Water	Total
Shadow	63	1	9	2	0	75
Burn/Open	0	80	0	11	4	95
Forest	2	2	70	14	0	88
Plantation	0	22	11	103	1	137
Water	0	4	0	2	52	58
Total	65	109	90	132	57	453

Table S9. Accuracy assessment for 2016 land cover classification for a region including Bukit Tigapuluh National Park, Riau, Sumatra.

	Ground Truth					
Class	Burn/Open	Plantation	Water	Forest	Cloud	Total
Burn/Open	34	16	3	0	3	56
Plantation	1	71	0	0	1	73
Water	0	0	20	0	0	20
Forest	0	1	0	35	0	36
Cloud	1	2	0	0	13	16
Total	36	90	23	35	17	201

Chapter 3

Protected area effectiveness in a sea of palm oil: A Sumatran case study

Erin E. Poor, Emmanuel Frimpong, Muhammad Ali Imron, Marcella J. Kelly

Abstract

Despite the establishment of a national protected area system at the beginning of the 20th century to protect some of the world's most biodiverse forests, Indonesia has one of the highest deforestation rates in the world, due to the expansion of the global palm oil industry. The endemic ecosystems of Sumatra, Indonesia provide habitat for Critically Endangered Sumatran tigers (*Panthera tigris sumatrae*), Sumatran elephants (*Elephas maximus sumatrensis*), and two species of orangutans (*Pongo abelii* and *Pongo tapanuliensis*). In this study, we use a matching method with generalized boosted models to determine the effectiveness of three protected areas in preventing deforestation from 2002 -2016. We also examine leakage – an increase in deforestation directly outside of protected areas – to provide a clearer picture of the effects of the global palm oil industry. We found that Tesso Nilo National Park, with its lowland rain forest habitat and conditions suitable for oil palm, offered the least protection from deforestation (avoided deforestation rate = 4.18%, $p < 0.05$ 95% CI [1.97% - 6.45%]). Mountainous Bukit Tigapuluh National Park had the highest avoided deforestation rate (26.36%, $p < 0.05$ 95% CI [24.17 – 28.55]), but had relatively high leakage (10.21%, $p < 0.05$ 95% CI [7.51 – 12.98]). The quantitative evidence of deforestation and effectiveness of protected areas in this heavily modified landscape, supports the need for increased enforcement around protected areas locally and globally in other palm oil production regions. These actions are critical in the preservation of global tropical endemic flora and fauna.

Keywords

Matching, protected areas, generalized boosted regression models, Sumatra, palm oil, deforestation

Introduction

As the global human population continues to expand, agriculture has become a primary driver of deforestation (Henders et al. 2015). Global palm oil production has recently doubled, and as the world's cheapest vegetable oil, it is projected to continue to increase (FAPRI 2012). Indonesia and Malaysia produce 80% of the global palm oil supply. Oil palm is usually grown in a monoculture, which results in a lack of structural complexity compared to natural forests. Plantations contribute to significant changes in biodiversity and wildlife distributions, and reductions in species richness compared to native forests and other types of agriculture (Fitzherbert et al. 2008, Koh and Wilcove 2008, Barnes et al. 2017, Mendes-Oliveira et al. 2017, Spear et al. 2018). While endemic and globally threatened species have been seen in palm oil plantations, no evidence suggests that plantations can sustainably hold a breeding population of tigers (*Panthera tigris*), elephants (*Elephas maximus*), orangutans (*Pongo spp.*), or tapirs (*Acrocodia indica*). On Sumatra, Critically Endangered Sumatran tigers (*Panthera tigris sumatrae*) have been shown to prefer acacia plantations and secondary forests to palm oil plantations (Sunarto et al. 2012), and the presence of palm oil in surrounding protected areas can have substantial contribution to their persistence (Imron et al., 2011).

In Indonesia – an archipelago that spans two biodiversity hotspots (Myers et al. 2000) – only about 10% of oil palm plantations are certified by the Round Table on Sustainable Palm Oil (RSPO) (Carlson et al. 2018). However, in its attempt to provide more 'sustainable' palm oil, the

RSPO has often been criticized for allowing plantations on ‘degraded’ or ‘regenerating’ forests. In Indonesia, which contains about 44% of certified plantations globally, certification did reduce deforestation rates on certified plantations (Carlson et al. 2018), but increased deforestation before certification, offsetting some of its gains (Noojipady et al. 2017, Carlson et al. 2018). Furthermore, lack of traceability has been a recurring issue in RSPO certified plantation supply chains (Carlson et al. 2018), leaving doubt about certification’s true effectiveness. Although moving to such a certification scheme may be a step in the right direction in allowing multi-use landscapes while conserving unique and endemic species, enforcement of both the certification regulations and local land-use and protection regulations is key to protecting biodiversity.

Despite the establishment of a protected area (PA) system at the beginning of the 20th century to protect some of the world’s most diverse forests, Indonesia recently surpassed Brazil with the highest deforestation rate in the world, largely due to the expansion of the global palm oil industry since the mid/late 1990’s (Margono et al. 2014). The level of protection that PAs actually impart varies based on location, socio-economic factors, and political factors, to name a few (Joppa et al. 2018). In Indonesia, like in many other tropical developing countries where palm oil is grown, it is difficult to determine extent and level of protection due to incorrect or unavailable spatial boundaries of PAs or due to the remote nature of PAs – areas likely that would not face anthropogenic pressures even if they were not officially protected (‘de facto’ protection; Joppa et al. 2018).

In addition to protection inside a PA, it is important to prevent leakage. Leakage occurs when a PA is established, but underlying socioeconomic needs are not addressed and local communities intensify harvest and extraction activities outside the PA – thus displacing the negative impacts on biodiversity that motivated the creation of the PA (McDonald et al. 2007).

Oliviera et al. (2007) found that deforestation increased by 300-470% directly adjacent to a newly established protected area in the Amazon. If leakage is occurring in Sumatra, PAs are at risk of becoming isolated islands of forest in a sea of oil palm, leaving wildlife populations at higher risk of loss of genetic diversity, inbreeding depression and extinction due to declining dispersal rates across a potentially dangerous monoculture matrix.

Sumatran PA effectiveness has been studied before at an island-wide scale, where Gaveau et al. (2009; 2012) used a propensity score matching method to examine Sumatran PA effectiveness and found positive impacts of protection against deforestation. Shah and Baylis (2015) found that Tesso Nilo National Park in central Sumatra had higher deforestation inside the park than outside the park using a similar method. Compared to the broader landscape (Gaveau et al. 2009), and within a 10 km buffer around PAs to measure leakage (Gaveau et al. 2012), PAs had lower deforestation rates, island wide, from 1990 – 2000. Now, it is important to revisit these analysis due to several factors: 1) the large increase in oil palm plantations in this province since 2000 (50% of palm oil harvested in Indonesia in 2014 was planted in 2003 or later (FAO 2017)), 2) the availability of new, finer scale (30 m x 30 m), accuracy-assessed, land cover data, and 3) the general lack of research in central, lowland Sumatra in comparison to other areas on Sumatra.

On Sumatra, 27% of palm oil production comes from Riau Province (Secretariat Directorate of Estates, 2007), in central Sumatra, where lowland areas that once boasted unique eco-floristic zones, Sumatran tigers, elephants, orangutans, and rhinoceroses (*Dicerorhinus sumatrensis*), provide ideal oil palm growing conditions. Riau contains three geographically close protected areas, which vary in condition, habitat, and terrain. Tesso Nilo National Park was established on land suitable for oil palm, whereas Bukit Tigapuluh National Park and Rimbang

Baling Wildlife, are mountainous, difficult to access PAs, but Bukit Tigapuluh is surrounded by palm oil and Rimbang Baling is surrounded by pulp and paper plantations that may be affecting their protected forests. Although deforestation is currently rampant within Tesso Nilo, deforestation is also widespread across the non-protected landscape, thus the protected status of the national park may still confer some protection despite extreme human modification.

Like Rimbang Baling and Bukit Tigapuluh, PAs globally are often placed in disproportionately inaccessible areas, or areas where harvest and extraction activities are less likely to occur (Joppa et al. 2018). Thus, simply comparing deforestation rates inside and outside of a PA will provide a biased result due to the inherent differences in landscape heterogeneity and land use constraints. Studies that use such methods have resulted in artificially high estimates of effectiveness, and now statistical matching is the preferred method (Joppa and Pfaff 2011). Therefore, to determine whether Tesso Nilo, Bukit Tigapuluh, and Rimbang Baling are providing effective protection, we use a propensity score matching method.

Statistical matching has been used to determine the effect of a treatment (in medicine, policy, etc.) on a group of samples while controlling for covariate bias (Stuart 2010). Matching has been adapted to natural resources assessments, most notably when combined with a logistic regression post-matching, to examine PA effectiveness (Andam et al. 2008, Shah and Bayliss 2015). One of the important estimates from matching is 'avoided deforestation' – i.e., the lack of deforestation occurring in a PA due to protected status, compared to deforestation occurring in similar biophysical conditions outside of the PA. A high avoided deforestation rate indicates high protection effectiveness, and vice versa. Here, we use matching to determine whether PAs in central Riau Province have actually provided protection against deforestation from 2002 – 2016, in spite of the high human use and modification across central Riau.

Methods

Study Area

The climate of Riau is classified in the Koppen-Geiger system as Af, tropical. Average temperature is 27° C and average rainfall is 2696 mm per year. Tesso Nilo National Park (IUCN category II) was established in 2004 and expanded in 2009 and has lost >50% of its natural forest (Poor et al. in review). Bukit Tigapuluh National Park (IUCN category II) was established in 1995, is 1,276.98 km^2, and largely consists of tropical montane forest. While deforestation has encroached on the park's edges due to oil palm plantations, there is still a core of primary forest, which is connected to the Sumatra's western spine of forested and protected mountains (the Barisan mountain range) (Figure 1). Rimbang Baling Wildlife Reserve (IUCN category IV) was established in 1986 and is 1,360 km^2. Rimbang Baling is connected to Kerinci Seblat National Park along Sumatra's western Bukit Barisan mountain range and also faces encroachment, largely from pulp and paper plantations along its eastern and northern edges. In all of these PAs, locals routinely enter the forest to hunt, gather resin and fruit, and fish.

Matching

With the use of matching in the context of PA effectiveness, one draws samples inside (treatment, 1) and outside (control, 0) of a PA. Then, parametric methods such as logistic regression, mahalanobis distances (Abadie and Imbens 2006), or non-parametric methods such as a generalized boosted regression model are used to determine propensity scores (McCaffrey et al. 2004). A propensity score is the estimated probability of a sample receiving 'treatment', given the sample's landscape covariate values (slope, elevation, etc.). Generalized boosted regression models (gbm) are an improvement on a common non-parametric model, the genetic method (Diamond and Sekhon 2005), due to their incorporation of propensity scores. These scores should be ‘balanced’ across groups, that is, values of all of the chosen covariates should be as similar as possible between treatment and control groups. This process of attempting to achieve balance is termed ‘matching’, since the modeler is attempting to match the values of covariates at selected random locations inside a PA to those at random locations outside of a PA, thereby reducing any biases introduced by non-random locations of protected areas. If balance is not achieved, the selected model should be re-parameterized or adjusted until satisfactory balance is achieved. Further analysis such as logistic regression to determine avoided deforestation, can be completed using the matched sample set. Some samples may not match between groups and can be discarded.

We created random points in 2002 forested areas outside and inside of PAs, excluding the areas that were obstructed by clouds in 2002 or 2016 land cover imagery. We extracted the value of six covariates; distance to major roads, distance to cities, distance to open areas, distance to plantations, slope, and elevation, for 2002 and the presence or absence of forest in 2016 (to determine whether the 2002 forest samples remained forest in 2016) at each sample location (Andam et al. 2008).

To determine whether leakage was occurring outside of PAs, we used the same covariates and created random points within a 10 km buffer (Curran et al. 2004, Nepstad et al. 2006) outside of the PAs and, based on the values of the six covariates at the random points, matched these points, to points with similar covariate values in the wider landscape outside of this 10 km buffer zone. Covariate preparation was carried out in ArcGIS 10.5 (ESRI 2017).

We created propensity scores, the estimated probability of a sample receiving ‘treatment’, given the sample’s covariate values, using non-parametric generalized boosted regression models (Santika et al. 2017; Friedman 2001), implemented in the package twang (Ridgeway et al.

2017a) in R (R Development Team 2017). We matched 2000 sample points within PAs, and 20,000 locations for the broader landscape, outside of PAs. Propensity scores were identified for the average treatment effect on the treated (ATT; samples within PAs or buffer areas), and covariate weights were compared to determine what covariates influenced deforestation. Using the gbm, samples were matched with 100,000-500,000 regression trees and the mean effect size stopping method (Ridgeway et. al 2017b). Shrinkage was 0.02-0.03 and we set interaction depth to 2. After achieving balance, we used the presence or absence of forest in 2016 at the sampling locations from 2002 to determine the effectiveness of PAs. We then estimated a generalized linear model, with deforestation in 2016 (0 = no deforestation, 1 = deforestation), as the dependent variable and the gbm-generated propensity scores as the predictors to estimate the average treatment effect (protected versus un-protected, or, for leakage, within the 10 km buffer versus in the broader landscape) of the treated (ATT) on the presence or absence of forest in 2016. Results are provided as percent of forest remaining attributed to PA status – we interpret this as 'avoided deforestation' (Andam et al. 2008, Shah and Bayliss 2015).

Results

Maximum similarity between covariate propensity scores ('balance') was achieved using different parameters and settings for each PA (Table S1; Figure 2). Tesso Nilo showed the lowest amount of benefit from protection, with an avoided deforestation rate of only 4.19%, Rimbang Baling had 12.8% avoided deforestation, while Bukit Tigapuluh had 26.36% of forest remaining due to protection, the highest of our focal PAs (Table 1; Figure 3). Overall, 10.35% of forest maintained from 2002 – 2016 is attributable to protection status. In all PAs except Bukit Tigapuluh, distance to roads had the highest relative influence on deforestation (Figure 2). Effect of protection in Bukit Tigapuluh was most influenced by elevation.

For leakage, elevation and/or slope were the most important variables except for Bukit Tigapuluh, where distance to plantation had the highest relative influence on leakage (Figure 2). Overall, being within proximity of a PA brought approximately the same amount of protection as being inside a PA (Figure 3). There does appear to be leakage around Bukit Tigapuluh National Park, where only 10.21% of forest in the buffer existed in 2016 due to proximity to the PA (Table 1). The protection of Rimbang Baling seems to be conferring additional protection to areas adjacent to the park, with 16.77% of forest near the PA existing in 2016 due to proximity to the PA (Table 1; Figure 3).

Discussion

Globally, 85% of threatened vertebrates are not adequately protected as specified by the global targets created by the 2011 Convention on Biological Diversity and which specify minimum goals for the conservation of natural resources (Venter et al. 2014). Although about 13% of global terrestrial area is protected, the effectiveness of these areas in protecting wildlife species is unknown, and likely overestimated. In a study measuring global PA effectiveness in preventing deforestation, results changed significantly from pre-matching methods to post-matching methods (Joppa and Pfaff, 2011). Our study is the first to use gbm matching methods in central Sumatra to examine the effectiveness of PAs within a landscape heavily modified by palm oil and pulp and paper plantations. Although authorities rely heavily on the existence of the PAs themselves (and not enforcement) to confer protection to unique, endemic wildlife such as the Sumatran tiger, we found that PAs in this system are only slightly effective at providing protection, likely due to palm oil expansion over the past ~20 years.

Bukit Tigapuluh has the highest avoided deforestation at 26.36% (Table 2, Figure 3). This may be due to many factors including the presence of multiple conservation organizations conducting research within the park, many communities living in the park, a park office located within an hour of the park, seemingly more engaged park management that actively conducts research and monitoring, and the high elevation and rugged slopes found within the park. Globally, PAs fare better when empowered locals are allowed sustainable use options, or when PAs are co-managed, as opposed to a single top-down authority (Oldekop et al. 2015). Interestingly, Bukit Tigapuluh did not have the lowest amount of leakage. Palm oil plantations ring the park on the eastern side and these are likely the cause of the lower than expected avoided deforestation rate of 10.21% ($p < 0.05$, 95% CI [7.51% - 12.98%], Table 1; Figure 3) within 10 km of the park boundary. There are some areas on the northern side of the park where palm oil plantations have encroached into the park, and we believe this is likely to continue without immediate enforcement action. Though protection is currently relatively high given the other estimates of avoided deforestation in Bukit Tigapuluh, avoided deforestation is likely to decrease with increasing encroachment.

In spite of high elevations and rugged slopes potentially conferring ‘de facto’ protection, Rimbang Baling Wildlife Reserve is only slightly effective, with an avoided deforestation rate of 13.43%, ($p < 0.05$, 95% CI [11.14% - 15.65%], Table 1; Figure 3). This is likely due to the encroaching pulp and paper plantations in the eastern and northern part of Rimbang Baling. Unlike palm oil plantations, pulp and paper plantations, which use *Acacia mangium*, *acacia crassicarpa*, *Eucalyptus grandis* and *Eucalyptus globulus*, can be grown in a wider variety of soils and at higher slopes and elevations. Parts of this PA included former mining concessions and are still commonly used for local extraction of timber and non-timber forest products.

As expected, the avoided deforestation rate due to protected status was lowest in Tesso Nilo, (4.19%, $p < 0.05$ 95% CI [1.97% - 6.45%], Table 1; Figure 3) the PA with lowest average elevation and slope, the most suitable for growing oil palm, and the most contested park in our study area. The average effect of the treatment (ATT) Shah and Bayliss (2015) found for Tesso Nilo from 2000 – 2012 is within our 95% confidence intervals (2.69% vs. 4.10%), indicating corroboration with our results, i.e. no significant difference in estimated effectiveness between the two studies. Avoided deforestation rates inside and in the 10 km buffer area were the same (4.54%), so locals are using the entire Tesso Nilo area similarly.

Other studies cite potential policies and geographic variation as cause for variation in PA effectiveness (Shah and Baylis, 2015), and though we did not incorporate socio-economic or policy data, geographic variation can be seen as a cause of variation in effectiveness in this landscape as well. We controlled for provincial level policies in this analysis by selecting PAs from one province, but neglected to examine effects at a more local level – that of regency or settlement level. Matching on socio-economic and political covariates gleaned from interviews or local surveys could provide valuable information about local attitudes and their impacts on deforestation, and should be a direction for future research.

In Tesso Nilo, Rimbang Baling, and for all PAs combined, distance to roads had the highest influence on PA effectiveness, as determined through covariate propensity score weights resulting from the gbm (Figure 2). Only major roads were included in this study and results may change slightly if plantation roads are taken into account. Elevation and slope both had high influence throughout the landscape, likely due to the relationship between high slopes and elevation and the lack of suitability for plantations and ‘de facto’ protection (Joppa and Pfaff 2018) that these areas provide. We did not incorporate every available landscape covariate and results may differ slightly depending on the variables used in matching. However, we selected these variables based on results from a related study conducted to predict deforestation (Poor et al. in review).

Tesso Nilo, founded in 2004, has had a conflicted existence and locals did not support the formation of the PA. It is regularly used by locals for a variety of extractive purposes and there are still areas of contention between park management and palm oil plantation employees and locals. Community management has a positive impact on PA effectiveness (Santika et al. 2017). It is not surprising that while the other PAs also are regularly used by locals, that Tesso Nilo is the least effective PA on the landscape. It is unclear whether local attitudes or low elevation play a greater role in Tesso Nilo's lack of effectiveness because we were unable to incorporate socio-economic factors into our study. The government of Indonesia has proposed a 12 year plan to restore Tesso Nilo and relocate many of the locals who currently inhabit and make use of the park. However, another study estimates that very little forest will remain in Tesso Nilo in 12 years (Poor et al. in review). The proposed restoration is unlikely to be effective unless significant education, outreach, and capacity building regarding alternative livelihoods is consistently implemented as soon as possible.

Although Rimbang Baling and Bukit Tigapuluh have greater avoided deforestation estimates than Tesso Nilo, it is still important to increase enforcement of these PAs. Leakage is occurring around both, and as land becomes rare for new oil palm plantations in more ideal flat areas, encroachment into Rimbang Baling and Bukit Tigapuluh is likely to increase. The negative effects of oil palm monocultures and their associated infrastructure on biodiversity are well documented (Fitzherbert et al. 2008) and the continued expansion of oil palm in this landscape is detrimental to the native, endemic tropical forests there. Currently, Rimbang Baling and Bukit Tigapuluh are enjoying some 'de facto' protection (Joppa et al. 2008), but may face increased threats in the future. Globally there is a growing market for palm oil (Carter et al.

2009) and a continued financial incentive to grow oil palm in this landscape where it is the most lucrative crop, and many locals have few other viable livelihood options. Bribery, lack of funding for local agencies, and illegal deforestation are also common in this study area, making regulation enforcement even more difficult.

Biodiversity protection is a complex interdisciplinary issue globally and often is locally nuanced. Global awareness has increased regarding the negative impacts of oil palm plantations, but we still see significant impacts of the industry in this landscape and as the industry continues to grow, we are likely to see similar situations worldwide, specifically in PAs with conditions suitable for oil palm and in areas where enforcement is lacking. There may be little feasible opportunity to reduce the negative impacts of plantations and their associated infrastructure, such as roads that increase forest access and poaching (Fitzherbert et al. 2008). In our study landscape, we see enforcement and restoration of current PAs as critically important to the conservation of Sumatra’s unique, endemic and globally important species. The demand for palm oil is unlikely to disappear in the foreseeable future, so we must work to increase productivity of existing plantations, while ensuring the persistence of wildlife. The establishment of forested stepping stones and corridors could allow wildlife to move more freely across landscapes (Yaap et al. 2016), while the enforcement of the boundaries of existing PAs – especially those without ‘de facto’ protection – could ensure refuges for, and persistence of, wildlife in oil palm dominated landscapes. If swift action towards creating these wildlife friendly, mixed-use production landscapes is not taken, habitat will continue to decline and eventually, isolated wildlife populations will be unable to survive. Eventually, tropical wildlife worldwide may swallowed into a sea of palm oil.

References

Andam KS, Ferraro PJ, Pfaff A, Sanchez-Azofeifa GA, Robalino, JA. 2017. Measuring the effectiveness of protected area networks in reducing deforestation. Proceedings of the National Academy of Sciences **105**(42): 16089–16094.

Barnes AD, Allen K, Kreft H, Corre MD, Jochum M, Veldkamp E, Clough Y, Daniel R, Darras K, Denmead LH, et al. 2017. Direct and cascading impacts of tropical land-use change on multi-trophic biodiversity. Nature Ecology & Evolution **1**(10): 1511-1519.

Carlson KM, Heilmayr R, Gibbs HK, Noojipady P, Burns DN. 2018. Effect of oil palm sustainability certification on deforestation and fire in Indonesia. Proceedings of the National Academy of Sciences **115**(1): 121–126.

Carter C, Finley W, Fry J, Jackson D, Wills L. 2007. Palm oil markets and future supply. European Journal of Lipid Science and Technology **109**(4): 307–314.

Cattau ME, Marlier ME, DeFries R. 2016. Effectiveness of Roundtable on Sustainable Palm Oil (RSPO) for reducing fires on oil palm concessions in Indonesia from 2012 to 2015. Environmental Research Letters 11: 105007.

Curran LM, Trigg SN, McDonald AK, Astiani D, Hardiono YM, Siregar P, Caniago I, Kasischke E. 2004. Lowland forest loss in protected areas of Indonesian Borneo. Science **303**(5660): 1000-1003.

Dehejia RH, Wahba S. 2002. Propensity score-matching methods for nonexperimental causal studies. The Review of Economics and Statistics **84**(1): 151–161.

Diamond A, Sekhon JS. 2005. Genetic Matching for Estimating Causal Effects: A General Multivariate Matching Method for Achieving Balance in Observational Studies. Department of Political Science, UC Berkeley, Berkeley, CA.

Fitzherbert EB, Struebig MJ, Morel A, Danielsen F, Bruhl CA, Donald PF, Phalan B. 2008. How will oil palm expansion affect biodiversity? Trends in Ecology and Evolution **23**(10): 538-545.

Food and Agriculture Organization of the United Nations. 2017. FAOSTAT online statistical service. Rome, Italy. www.fao.org/faostat/en/#home.

Gaveau DLA, Curran LM, Paoli GD, Carlson KM, Wells P, Ratnasari D. 2012. Examining protected area effectiveness in Sumatra : importance of regulations governing unprotected lands. Conservation Letters **5**(2): 142–148.

Gaveau DLA, Epting J, Lyne O, Linkie M, Kumara I, Kanninen M, Leader-Williams N. 2009. Journal of Biogeography **36**(11): 2165-2175.

Henders S, Persson UM, Kastner T. 2015. Trading forests: Land-use change and carbon emissions embodied in production and exports of forest-risk commodities. Environmental Research Letters 10:125012

Imron MA, Herzog S, Berger U.2011. The influence of agroforestry and other land-use types on the persistence of a Sumatran tiger (*Panthera tigris sumatrae*) population: An Individual-Based Model Approach, Environmental Management, **48**: 276.

Joppa LN, Loarie SR, Pimm SL. 2018. On the protection of “protected areas”. Proceedings of the National Academy of Sciences **105**(18): 6673–6678.

Joppa L, Pfaff A. 2010. Reassessing the forest impacts of protection: The challenge of nonrandom location and a corrective method. Annals of the New York Academy of Sciences **1185**: 135–149.

Joppa LN, Pfaff A. 2011. Global protected area impacts. Proceedings of the Royal Society B

Biological Sciences doi:10.1098/rspb.2010.1713.

Koh LP, Wilcove DS. 2008. Is oil palm agriculture really destroying tropical biodiversity? Conservation Letters **1**(2): 60–64.

Margono BA, Potapov PV, Turubanova S, Stolle F, Hansen MC. 2014. Primary forest cover loss in Indonesia over 2000–2012. Nature Climate Change **4**: 730–736.

McCaffrey DF, Ridgeway G, Morral AR. 2004. Propensity score estimation with boosted regression for evaluating causal effects in observational studies. Psychological Methods **9**: 403–425.

McDonald RI, Yuan-Farrell C, Fievet C, Moeller M, Kareiva P, Foster D, et al. 2007. Estimating the effect of protected lands on the development and conservation of their surroundings. Conservation Biology **21**(6):1526–36.

Mendes-Oliveira AC, Peres CS, de Maues PCR, Oliveira GL, Mineiro IGB, de Maria SLS, Lima RCS. 2017. Oil palm monoculture induces drastic erosion of an Amazonian forest mammal fauna. PloS One **12**(11): e0187650.

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GA, Kent J. 2000. Biodiversity hotspots for conservation priorities. Nature **403**(6772): 853–858.

Nepstad D, Schwartzman S, Bamberger B, Santilli M, Ray D, Schlesinger P, Lefebvre P, Alencar A, Prinz E, Fiske G, et al. 2006. Inhibition of Amazon deforestation and fire by parks and indigenous lands. Conservation Biology **20**(1): 65–73.

Noojipady P, Morton DC, Schroeder W, Carlson KM, Huang C, Gibbs HK, Burns D, Walker NF, Prince SD. 2017. Managing fire risk during drought: the influence of certification and El Niño on fire-driven forest conversion for oil palm in Southeast Asia. Earth System Dynamics **8**: 749–771.

Poor EE, Shao Y, Kelly, MJ. In Review. Mapping and predicting forest loss in a Sumatran tiger landscape from 2002 – 2050. Journal of Environmental Management.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna Austria. www.R-project.org.

Ridgeway G, McCaffrey D, Morral A, Griffin BA Burgette L. 2017. Toolkit for weighting and analysis of nonequivalent groups. RAND Corporation, Santa Monica, CA.

Ridgeway G et al. 2017. gbm: Generalized Boosted Regression Models. R package version 2.1.3. https://CRAN.R-project.org/package=gbm

Rosenbaum PR, Rubin DB 1983. The central role of the propensity score in observational studies for causal effects. Biometrika **70**: 41–55.

Santika T, Meijaard E, Budiharta S, Law EA, Kusworo A, Hutabarat JA, Indrawan TP, Struebig M, Raharjo S, Huda I, et al. 2017. Community forest management in Indonesia : Avoided deforestation in the context of anthropogenic and climate complexities. Global Environmental Change **46**: 60–71.

Shah P, Baylis K. 2015. Evaluating heterogeneous conservation effects of forest protection in Indonesia. PloS One **10**(6): e0124872.

Spear DM, Foster WA, Advento AD, Naim M, Caliman JP, Luke SH, Snaddon JL, Ps S, Turner EC. 2018. Simplifying understory complexity in oil palm plantations is associated with a reduction in the density of a cleptoparasitic spider, Argyrodes miniaceus (Araneae: Theridiidae), in host (Araneae: Nephilinae) webs. Ecology and Evolution **8**(3): 1595-1603.

Stuart EA. 2010. Matching methods for causal inference: A review and a look forward. Statistical Science **25**: 1–21.

Sunarto S, Kelly MJ, Parakkasi K, Klenzendorf S, Septayuda E, Kurniawan H. 2012. Tigers need cover: multi-scale occupancy study of the big cat in Sumatran forest and plantation landscapes. PloS One, **7**(1): e30859.

Venter O, Fuller, RA, Segan, DB, Carwardine J, Brooks T, Butchart SH, Marco MD, Iwamure T, Joseph L, O'Grady D, et al. 2014. Targeting global protected area expansion for imperiled biodiversity. PLoS Biology **12**(6): e1001891.

Food and Agricultural Policy Research Institute. 2012. World Agricultural Outlook. Ames, Iowa. http://www.fapri.iastate.edu/outlook/2012/.

Yaap B, Magrach A, Clements GR, Mcclure CJW, Paoli GD, Laurance WF. 2016. Large mammal use of linear remnant forests in an industrial pulpwood plantation in Sumatra , Indonesia. Tropical Conservation Science October – December: 1-13.

Tables

Table 1. Model estimates (reported as percentages) and 95% confidence intervals (italicized; those not overlapping zero are bold) for model generalized boosted model intercept and covariates, including dependent variable, Treatment, for central Sumatran protected area effectiveness (EF) and leakage for three different protected areas (Tesso Nilo National Park, Bukit Tigapuluh National Park, Rimbang Baling Wildlife Reserve) and these three protected areas together. In the leakage case, ‘Treatment’ are those samples within a 10 km buffer around the protected area.

Covariate	Tesso Nilo NP – EF¶	Tesso Nilo Leakage	Bukit Tigapuluh NP-EF	Bukit Tigapuluh Leakage	Rimbang Baling WR-EF	Rimbang Baling Leakage	All PAs - EF	All PAs Leakage
Intercept	31.7848	26.1120	43.3329	-14.4400	41.6232	4.3834	-8.2938	-1.7250
	(26.38 – 37.39)	***(20.0344 – 32.3812)***	***(39.10 – 47.65)***	***(-17.0999 – -11.7151)***	***(35.40 - 48.09)***	***(1.0019 – 7.8851)***	***(-9.87 – -6.6904)***	*(-3.8218 – 0.4206)*
Treatment	4.1852	4.5400	26.3644	10.2081	13.4282	16.7658	10.3500	8.4805
	(01.97 – 06.45)	***(1.9135 – 7.2412)***	***(24.17 – 28.55)***	***(7.505 – 12.9782)***	***(11.1394 – 15.6537)***	***(13.8114 – 19.7634)***	***(8.7293 – 11.9861)***	***(6.7285 – 10.27)***
Cities	-	-0.0003	0.0002	-0.0003	0.0001	0.0001	-	-0.0004
	-	*(-0.0047 – 0.0001)*	*(0.0000 – 0.0004)*	*(-0.0005 – 0.000)*	***(0.0001 – 0.0004)***	*(-0.0002 – 0.0003)*	-	***(-0.0005 – -0.0003)***
Open	-0.0490	0.0066	-	0.0098	-	0.0107	-	0.0114
	(-0.0067 – -0.00308)	***(0.0029 – 0.0102)***	-	***(0.0074 – 0.0123)***	-	***(0.0090 – 0.0123)***	-	***(0.0108 – 0.0126)***
Plantations	-	-0.0006	0.0391	0.0716	0.0345	0.0919	0.0452	0.0666
	-	*(-0.0206 – 0.0193)*	***(0.0341 – 0.0441)***	***(0.0532 – 0.0900)***	***(0.0264 – 0.0425)***	***(0.0789 – 0.1049)***	***(0.0399 – 0.0505)***	***(0.0572 – 0.0761)***
Roads	0.0011	-0.0004	-	0.0013	0.0011	-	0.0025	-
	(0.0007 – 0.0014)	*(-0.0008 – 0.0001)*	-	***(0.0009 – 0.0016)***	***(0.0010 – 0.0013)***	-	***(0.0023 – 0.0027)***	-
Elevation	-0.1539	-0.1319	0.0589	0.1211	0.0337	-	0.0849	0.0585
	(-0.1845 – -0.1222)	***(-0.1716 – -0.9155)***	***(0.0511 – 0.0666)***	***(0.0993 – 0.1428)***	***(0.0287 – 0.0387)***	-	***(0.0791 – 0.0908)***	***(0.052 – 0.065)***
Slope	0.2077	0.2449	0.5857	0.9757	0.0469	1.5316	0.8451	1.2781
	(-0.116 – 0.532)	*(-0.1422 – 0.6335)*	***(0.4830 – 0.6875)***	***(0.7229 – 1.2294)***	***(0.3577 – 0.5828)***	***(1.4089 – 1.6612)***	***(0.7524 – 0.9379)***	***(1.1463 – 1.4014)***

Covariate names: Cities – Distance to cities; Open – Distance to open areas; Plantations – Distance to plantations; Roads – Distance to roads

¶ Effectiveness

Figures

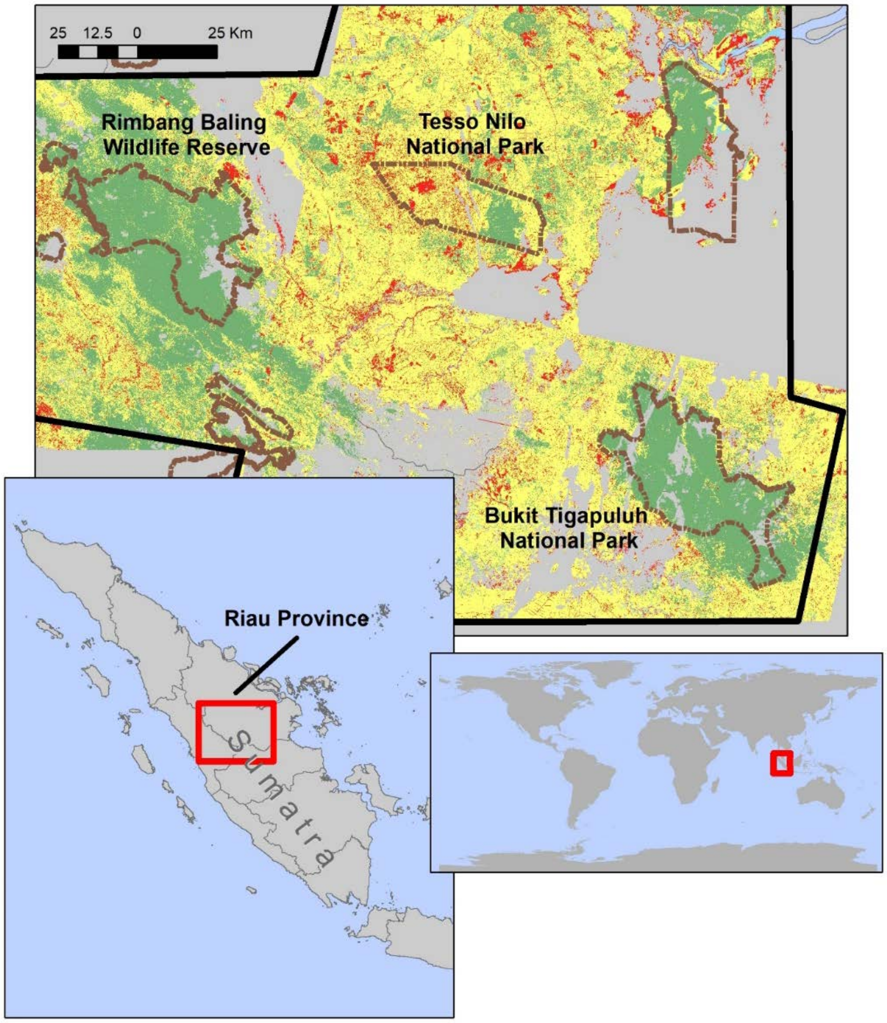

Figure 1. Study area. Location of focal protected areas and 2016 land cover; green: forest; yellow: plantation; red: open areas. Gray areas inside black study area boundary (top) were obstructed by cloud cover during land cover classification (Poor et al. in review).

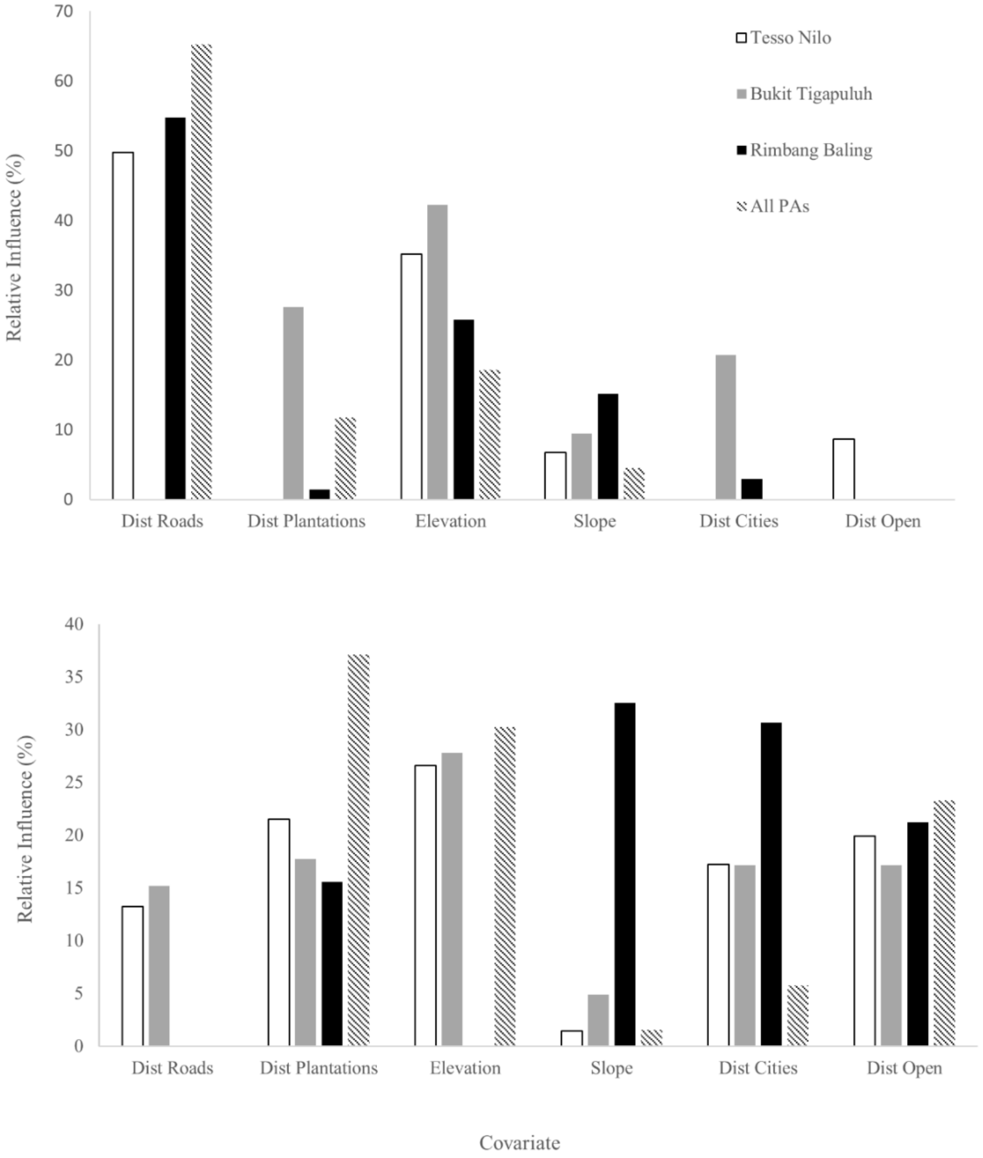

Figure 2. Covariate contributions. Relative covariate (distance to roads, distance to plantations, elevation, slope, distance to cities and distance to open areas) contributions (propensity scores) to central Sumatran protected area effectiveness (top) and leakage (bottom), resulting from a generalized boosted regression model. Values for individual protected area covariate contributions and the combined values are graphed (hashed bars). The direction of influence are provided in Table 1.

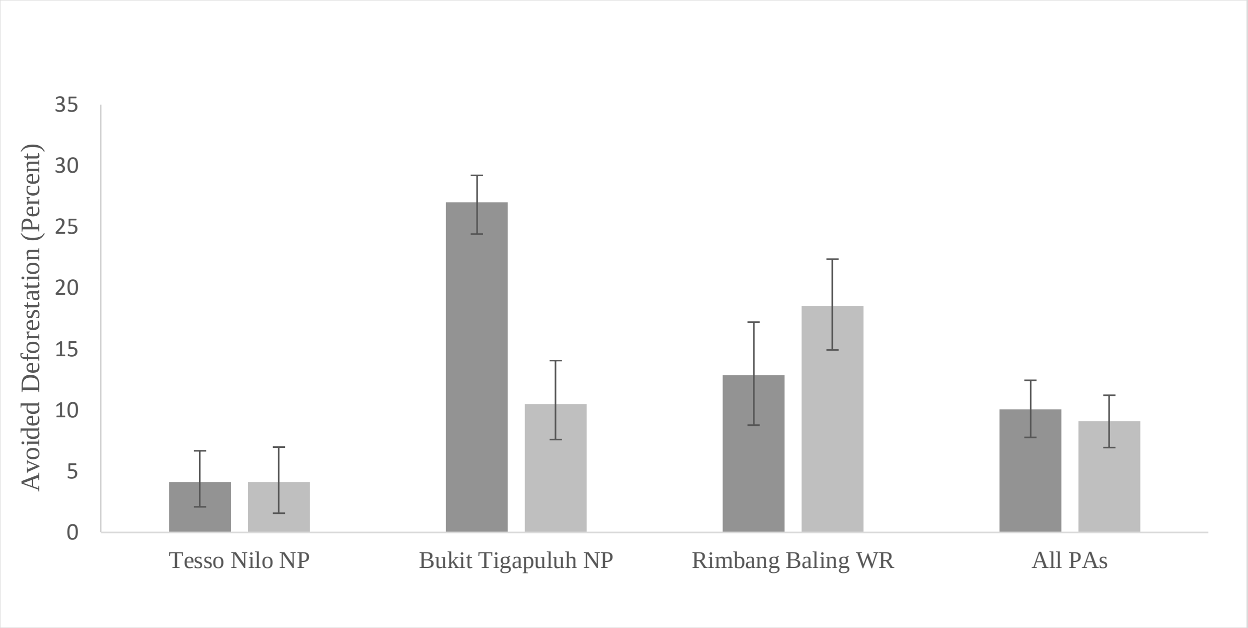

Figure 3. Treatment effects. Average treatment effect (ATT) and avoided deforestation estimates (dark gray) for each protected area, Tesso Nilo National Park, Rimbang Baling Wildlife Reserve and Bukit Tigapuluh National Park in central Sumatra, and estimates of leakage (light gray) as determined from a 10 km buffer area around each protected area, with 95% confidence intervals.

Supplementary Material

Table S1. Variable contributions (propensity scores) to central Sumatran protected area effectiveness and leakage, resulting from a generalized boosted regression model. Variables for which matching could not be achieved for group within protected areas and outside of protected areas are not included.

Variable	Tesso Nilo NP – EF^	Tesso Nilo Leakage	Bukit Tigapuluh NP -EF	Bukit Tigapuluh Leakage	Rimbang Baling WR - EF	Rimbang Baling Leakage	All PAs - EF	All PAs Leak.
Dist to cities*	-	17.25 (0.032)	20.70 (0.824)	17.19 (0.351)	2.93 (0.600)	30.66 (0.582)	-	5.77 (0.321)
Dist to open	8.61 (0.059)	19.91 (0.103)	-	17.16 (0.304)	-	24.22 (0.503)	-	23.29 (0.082)
Dist to plantations	-	21.51 (0.051)	27.59 (0.056)	17.73 (0.152)	1.44 (0.806)	15.57 (0.049)	11.75 (0.717)	39.12 (0.035)
Dist to roads	49.47 (0.044)	13.25 (0.991)	-	15.21 (0.268)	54.76 (0.223)	-	65.21 (0.05)	-
Elevation	35.17 (0.101)	26.63 (0.854)	42.22 (0.859)	27.82 (0.997)	25.76 (0.249)	-	18.56 (0.408)	30.24 (0.054)
Slope	6.74 (0.142)	1.45 (0.984)	9.48 (0.273)	4.9 (0.421)	15.12 (0.522)	32.54 (0.217)	4.48 (0.525)	1.57 (0.817)

* P values for generalized boosted models in parentheses

^ Effectiveness

Chapter 4

The road to deforestation: edge effects in an endemic ecosystem in Sumatra, Indonesia

Erin E. Poor, Virta I.M. Jati, Muhammad Ali Imron, Marcella J. Kelly

Abstract

Worldwide, roads are a main driver of deforestation and degradation as they increase forest access and cause changes in microclimate along the forest edge. In many tropical areas, unofficial roads go unreported and unrecorded, resulting in inaccurate estimates of total forested areas. This is the case in central Sumatra, which boasts populations of critically endangered Sumatran elephants and tigers and a suite of other endemic flora and fauna that make the area globally unique, but maps do not reflect reality. Here, we present new maps of digitized and ground-truthed roads, in one of Sumatra's unique lowland tropical protected areas, Tesso Nilo National Park, for 2002, 2014 and 2016. Using our newly created roads dataset, we determine how these roads impact forest extent and distribution, and using temperature and humidity data collected along roads and at interior forest locations, we determine whether roads have altered the remaining forests' microclimate. Our data show >2,400 km of roads within the national park - nearly a 10-fold increase from prior known roads – contributing to fragmentation and a minimum decrease in forested area of 15% when taking edge effects into account at multiple distances – 100 m, 500 m, 750 m and 1000 m. From 2002 – 2014, length of roads in the park increased 156%, and from 2014 – 2016 roads increased another 1.4%. Road density increased substantially from 1.06 km/km^2 to 2.63 km/km^2 from 2002 - 2016. We found significant differences in microclimate (i.e. higher temp and lower humidity) along roads compared to forest interior locations. Our results suggest that this endemic ecosystem is facing significant threat from roads and their associated impacts. Without swift management action such as road closures

and increased enforcements by park management, this ecosystem, and its endemic globally significant wildlife, could be lost.

Indonesian Abstract

Jalan raya secara global merupakan penyebab utama dari deforestrasi dan degradasi hutan dengan cara meningkatkan akses ke dalam hutan dan menyebabkan perubahan iklim mikro sepanjang batas hutan dan mengurangi penggunaan oleh satwa liar akibat cahaya lampu dan polusi dari kaki maupun kendaraan. Lebih lanjut, pada berbagai sistem di daerah tropis, jalan yang tidak resmi seringkali tidak dilaporkan dan tercatat, menyebabkan estimasi luasan tutupan hutan menjadi kurang akurat. Sumatra bagian tengah pada kasus ini, merupakan kawasan yang mampu menampung harimau Sumatera dan gajah Sumatera yang memiliki status kritis serta cocok untuk flora dan fauna endemik secara global, namun peta yang ada belum menggambarkan kenyataan yang ada. Pada makalah ini, kami menyajikan peta terbaru yang mempertimbangkan keberadaan jalan yang sudah didigitalisasi dan dicek di lapangan, termasuk jalan-jalan yang tidak resmi, di dalam Taman Nasional Tesso Nilo pada periode 2002, 2014 dan 2016. Menggunakan data set jalan terbaru yang kami buat, kami mengukur dampak keberdaan jalan terhadap luasan hutan dan juga distribusinya. Selain itu kami menggunakan data temperatur dan kelembaban sekitar jalan dan di dalam hutan, kami menentukan apakah jalan telah memengaruhi iklim mikro di dalam hutan yang tersisa. Dari data kami menunjukkan bahwa >2,400 km jalan dalam taman nasional telah dibangun – mendekati 10 kali dari jalan yang sudah ada sebelumnya – dan paling tidak mengurangi 15 % area hutan ketika kami mempertimbangkan dampak pinggir dengan menggunakan berbagai jarak– 100 m, 500 m, 750 m dan 1000 m. Dari tahun 2002 – 2014, panjang jalan di dalam taman nasional meningkat sebanyak 156%, dan dari 2014 – 2016 panjang jalan meningkat 1,4%. Kepadatan jalan meningkat sangat jelas dari 1,06

km/km^2 menjadi 2,63 km/km^2 dari tahun 2002 - 2016. Kami menemukan perbedaan yang signifikan dalam habitat mikro (antara lain temperatur yang lebih tinggi dan kelembaban yang lebih rendah) sepanjang jalan dibandingkan dengan lokasi-lokasi yang ada di dalam hutan. Hasil kami menunjukkan bahwa ekosistem endemic ini menghadapai ancaman yang signifikan akibat adanya jalan dan dampak yang diberikan. Tanpa mengganti cara mengelola seperti menutup jalan dan meningkatkan penegakan hukum, ekosistem dan satwa-satwa yang penting bagi dunia ini bisa menjadi punah.

Keywords

fragmentation, microclimate, roads, Sumatra, Tesso Nilo National Park

Introduction

Globally, forests have the highest rate of deforestation [1] and as forests are broken up into smaller patches of fragmented forest, biodiversity will be lost directly, as a result of forest clearing and indirectly, through increased poaching and increased forest access. In the Amazon, researchers found that extinction rates are negatively correlated with forest fragment area [2, 3] and that fragments 0.01 - 0.1 km^2 in size lost species across taxa at a higher rate than fragments 1 km^2 in size [4-7]. Furthermore, biodiversity is not only affected by the size of forest patches remaining, but the distance to, and habitat between, neighboring forest patches in the matrix can play roles in biodiversity persistence in a disturbed landscape [8, 2]. Thus, maintaining intact forested areas that are connected to other forested areas across a landscape is integral to maintaining global tropical biodiversity.

Within protected areas, there is ample evidence to suggest that roads have an overall negative affect on wildlife populations across taxa due to traffic, noise, light pollution, and/or increased human access [9]. Such effects are often measured by wildlife population abundance, species richness, or home range activity in relation to road density or distance from a road. For example, grizzly bears, cougars, and wolves, on average, have lower road density within their home ranges than outside of their home ranges [10-12]. Other species such as zebras, elephants, wildebeests, elands, jackals [13], salamanders, wood frogs [14], and woodland caribou [15] showed reduced animal sign or reduced abundance within a certain distance from roads, indicating that roads negatively affect habitat use by, or demography of, these taxa.

In addition to the impacts of increased human activity levels associated with roads [16-18], structural changes in forest edges along roads leading into or out of a forest or protected area [19], can negatively affect biodiversity through temperature variability, increased light, and reduced humidity, which can be found as far as 60 m from the edge of a rainforest patch [20, 5]. The increased tree mortality near forest edges also results in a decline in biomass and a decrease in carbon storage [21]. Lovejoy et al. (1986) found some trees were unable to handle these changes and died standing when an edge was created. These changes, along with an increased abundance of lianas [22], which decrease tree growth, can result in significant structural alterations at a patch's edge, ultimately resulting in a change in wildlife populations and diversity.

While protected areas may, in some cases, slow deforestation and confer protection within their borders, protected areas are not immune to the effects of deforestation that occur outside their borders. In fact, decreasing forest, increasing logging, and increasing fires outside of protected areas have been shown to negatively affect tropical forest protected area health (as measured by expert opinion on change in guild abundance) in 60 randomly selected protected areas [2]. Thus, as forest surrounding a protected area decreases, wildlife populations within the protected area may decline as well. If little natural forest remains outside of a protected area and the distance between neighboring patches is greater than a species' dispersal ability, these species may have low long-term persistence [23], ultimately resulting in decreased wildlife populations inside and outside of protected areas. While factors outside of protected areas were the most important factors negatively affecting protected area health, [2] also found that decreasing forest area within a protected area, increasing hunting, and increasing logging were also important drivers of protected area health.

In Indonesia, a tropical archipelago spanning the Sundaland and Wallacea biodiversity hotspots [24], many protected areas have been degraded due to the expansion of palm oil plantations in the past 20 years [25]. Indonesia contains 10% of the world's plant species and 17% of the world's bird species [26], and recently eclipsed Brazil as having the highest deforestation rate in the world [27]. Indonesia's protected area system was established at the beginning of the 20th century to preserve some of the world's highest diversity forests and unique species such as Critically Endangered Sumatran elephants (*Elephas maximum sumatrensis*) and Sumatran tigers (*Panthera tigris sumatrae*), which are found in Tesso Nilo National Park in Riau, Sumatra. Protection in many of the protected areas is not well enforced and the areas are easily accessed by the local human population, which hunts wildlife, tends oil palm plantations, and harvests timber and non-timber forest product resources. While [28] documented a decline in human activity in Tesso Nilo from 2005-2011, remote cameras still had a capture rate of seven photos of humans per 100 trap nights. As a former logging concession, Tesso Nilo can be accessed through small roads or pathways on motorbike or foot, most of which are not present in Tesso Nilo's official roads documentation. While the impacts of these smaller roads may be less than those from a highway, the impacts of access trails and roads in this landscape has yet to be studied and the increased access to the forest from such roads could be contributing significantly to biodiversity loss within this unique and endemic, eco-floristic zone [29]. In this study, we aim to 1) determine if, and how much, the roads within the park have increased from 2002-2016, 2) determine how much natural forest is left and its distribution when we take road effects into account, and 3) determine how roads affect microclimate within the park and relate that to potential impacts on flora/fauna.

Methods

Study Area

Tesso Nilo National Park was established in 2004 in Riau Province (pop. 6.3 million) (Figure 1) and contains some of the last remaining lowland tropical rainforest in Sumatra. The climate of Riau is classified in the Koppen-Geiger system as Af, tropical. Average temperature is 27° C while average rainfall is 2696 mm per year. The park, initially 386 km^2 was expanded to 830 km^2 in 2009 to better protect populations of Sumatran elephants and the endemic floristic community found within the park. Formerly a group of multiple adjacent timber concessions, Tesso Nilo was established with the intent to end the rapid deforestation that was beginning to occur, and to curb poaching of tigers and elephants as well as reduce human-wildlife conflict by providing a refuge for wildlife. Although community managers were involved in its foundation, indicating support for the protected area, strict protection and patrol measures are insufficient, as encroachment of the park has continued, and in recent years has likely gotten worse with the increased prevalence of palm oil in the region [30]. Tesso Nilo National Park is categorized Class III (of five) Tiger Conservation Landscapes (TCL) meaning it has habitat to support some tigers but also has moderate to high levels of threat and minimal conservation investment [31]. Tesso Nilo National Park is situated within a larger landscape-scale study area in central Riau

Province (Figure 1). Tropical forest fragmentation: synthesis of a diverse and dynamic discipline 1), which also includes Bukit Tigapuluh National Park and Rimbang Baling Wildlife Reserve. There could be a possibility of maintaining viable tiger populations in Tesso Nilo National Park if it remains connected to Bukit Tigapuluh National Park and Rimbang Baling Wildlife Reserve, as both have mountainous connections to the inaccessible Bukit Barisan mountains on Sumatra's western edge.

Road Length and Density 2002 – 2016

We digitized all roads possible within Tesso Nilo from 2002, 2014, and 2016 using composite infrared Landsat images and Digital Globe World View 2. To aid in clear identification, all Landsat images were pan-sharpened with Band 8 to 15 m accuracy. All geospatial data resulting from digitization projected to 48 S UTM 1984. Land cover data were created using Landsat data for 2002, 2013/2014 and 2016 and a maximum likelihood classification algorithm. All land cover maps had >85% accuracy and were created as a separate part of this study [30]. Additionally, from June – July 2016, we drove roads passable by motorbike within Tesso Nilo using GPS tracking to further identify roads and aid in creating the new roads data. We drove most roads within the forested area of Tesso Nilo, but were unable to drive all roads within the park, due to contested areas in illegal palm oil plantations within Tesso Nilo. We attempted to digitize roads regardless of size, as people often use trails, as well. We do not distinguish between large and small roads here.

To determine the density of roads, line density (kilometers of roads per square kilometer) was calculated for roads using a 1 km moving window. Analysis was restricted to within park boundaries so incomplete roads data outside of Tesso Nilo would not affect density estimates.

All spatial data were projected to UTM Zone 48N and analyses were completed using ArcGIS 10.3 [32].

Potential Impacts on Forest Distribution

After literature review, we determined that specific taxa would be affected by different buffer widths and thus we chose 100 m (small carnivores, [33]; birds, [18], [34]), 500 m (amphibians and mid-sized carnivores, [35]; [13]; [36]) 750 m (ungulates and large carnivores, [37]) and 1000 m (large carnivores, [38]) from roads to account for impacts of forest loss due to roads on various species. We created these buffers surrounding the roads data and subsequently removed these buffer areas first from the forested area of Tesso Nilo using GIS, to identify ‘core’ forest areas [39].

To identify natural forest patches, we used 2002, 2013/14 and 2016 land cover data, newly created for this area [30] and extracted land cover classes of ‘natural forest’. The area of each forest patch was then calculated and the distance from each patch to the closest neighboring forest patch was also calculated. We did this for forest patches across the greater Riau Province landscape, including Tesso Nilo, as well as for forest within Tesso Nilo, to determine how the forest distribution within Tesso Nilo compares to the landscape as a whole.

Potential Impacts on Microclimate

During our motorbike road surveys throughout Tesso Nilo at 100 randomly selected points, we measured the width, from edge to edge, of the road or path, temperature within the middle of the road, temperature 100 m from road edge (interior locations) and photographed canopy (for visual, ad-hoc canopy cover estimation) above each random sample location to begin to assess how roads are affecting the structure of Tesso Nilo’s forest. We conducted Wilcoxon paired rank sum tests to determine if there was a difference in temperature or humidity on roads vs. at interior locations. We used independent samples Wilcoxon rank sum tests in R software [40] to determine if there was a difference between temperature and humidity on plantation roads versus interior locations and between plantation roads and roads through other land cover types. We also compared humidity and temperature for sample locations with closed canopy road-ways (visually estimated directly above the sample location) to areas with open canopy road-ways. We then used this data combined with information gleaned from a literature search, to determine the potential impacts roads in Tesso Nilo may have on wildlife populations. Due to time and permitting constraints, we were unable to collect direct wildlife data.

Results

Road Length and Density 2002 – 2016

In 2002, we identified 954.06 km of previously unrecorded roads via digitization within Tesso Nilo, 321.32 km of which were within forested areas (Figure 2). In 2014, we identified an additional 1493.91 km of unrecorded roads using digitization, for a total length of 2447.97 km (a 156% increase), while the length of roads within the forested areas of Tesso Nilo decreased to 147.28 km, likely due to the overall decrease in natural forest left within the park. Roads increased slightly (1.4%) from 2014 – 2016, with a total of 2,483.8 km of roads found within the protected area via digitization and motorbike surveys in Tesso Nilo (Figure 2). In 2002, average road density within the entire park was 1.06 km road/km^2, while density increased to 2.58 km road/km^2 in 2014 and 2.63 km road/km^2 in 2016. Within just the natural forest of Tesso Nilo, we documented an overall doubling of average road density from 0.41 km road/km^2 in 2002 to 0.88 km road/km^2 in 2016.

Potential Impacts on Forest Distribution

Overall impacts – Without incorporating roads into land cover area calculations, from 2002-2016, natural forest in Tesso Nilo decreased 76.31% from 688.82 km^2 to 162.69 km^2 (Table 1). On the most conservative scale, taking the 100 m roads buffer into account, the amount of natural forest decreased by 79.90 % from 589.77 km^2 in 2002 to 118.49 km^2 in 2016 (Table 1). On the least conservative scale, taking 1000 m roads buffer into account, forest in Tesso Nilo declines 97% from 72.19 km^2 in 2002 to 2.12 km^2 in 2016.

Forest distribution without roads, 2002 – In 2002, in the larger study area (Figure 1), including Tesso Nilo, there were 409,602 forest patches (20,837.79 km^2) with a mean size of 0.05 km^2. Within Tesso Nilo, there were 1,305 natural forest patches, if roads are not taken into account (Table 1) with a much larger average patch size of 0.52 km^2 and an average of 200.6 m from the closest neighboring forest patch within the park, indicating a primarily intact natural forest area in the park (Table 2). Average distance from a forest patch within Tesso Nilo to the nearest patch outside Tesso Nilo was 2.39 km. Forest patches in Tesso Nilo were on average 166.54 m from the nearest open patch in the park, and 1.09 m from plantation.

Forest distribution with roads, 2002 – The impacts of using a 100 m road buffer distance resulted in decreasing the number of natural forest patches from 1,305 to 1,002, yielding an average patch size of 0.59 km^2within Tesso Nilo, and an average distance of 34.62 m between neighboring core forest patches (Figure 3). If road effects permeate 1 km into the forest, there were only natural forest 47 patches remaining, with an average 1.04 km between patches and an average size of 1.54 km^2 (Table 1; Figure 3).

Forest distribution without roads, 2016 – In 2016, across the greater study area, there were 403,414 natural forest patches with an average size of 0.02 km^2 with a total natural forested area of 10,198 km^2. Within Tesso Nilo, if roads are not taken into account, there are 4,313 forest patches with average patch size of 0.04 km^2. On average, patches are 37.79 m from the next closest patch within the park, and 2.2 km from the closest patch outside of the park, but an average 357 m from open areas, and 0.01 m from the nearest oil palm plantation – indicating isolation from natural forest (Table 2).

Forest distribution with roads, 2016 – On the most conservative end, if we assume road disturbances to wildlife extend 100 m from the road, the natural forest in 2016 was reduced from 4,313 to 3,321 core forest patches (Figures 3 & 4), with an average core patch size of 0.04 km^2 and an average distance of 49.79 m to the next closest forest patch. On the least conservative end, when using a 1 km road buffer, there was only 2.12 km^2 of core forest remaining within 109 patches, (nearly all remaining forest in Tesso Nilo is within 1 km of a road), 133.52 m to the nearest forest patch within the park. Average patch size dropped to 0.02 km^2 (Figure 4). Within Tesso Nilo and at the landscape scale, forest patches were clustered in the eastern section of Tesso Nilo (Figure 3).

Potential Impacts on Microclimate

Average width of roads was 1.27 m, but only 12% of randomly selected sample locations still had closed canopy cover. Across all sampled locations in Tesso Nilo, humidity at interior locations (100 m from road edge) was significantly higher than humidity on roads ($V = 2852$, $\bar{x}_{\text{Roads}} = 73.12$, $\bar{x}_{\text{Interior}} = 74.16$, $p < 0.001$) while temperature was significantly lower ($V = 704.5$, $\bar{x}_{\text{Roads}} = 29.78°$ C, $\bar{x}_{\text{Interior}} = 29.38°$ C, $p < 0.001$) at interior locations. Additionally, at interior plantation locations, humidity was significantly higher than at interior locations of forest ($W = 848.5$, $\bar{x}_{\text{Plantation}} = 77$, $\bar{x}_{\text{Nonplantation}} = 76$, $p < 0.005$). There was no significant difference in temperature or humidity on plantation roads compared with roads in other land cover types, nor was there a difference in temperature and humidity across open or closed canopy cover sampling locations, interior or road.

Discussion

It is widely recognized that Tesso Nilo has suffered from deforestation and degradation, but to our knowledge, this is the first study quantifying the impacts of roads. This research was motivated by receiving a roads dataset from the government of Indonesia that did not appear to reflect the on-the-ground situation, and we determined it necessary to create an updated and accurate roads dataset. Furthermore, to our knowledge there has not been any research focused on the impacts of human activity or roads in Tesso Nilo despite the declining populations of tigers and elephants and the anecdotal evidence of increased human activity in the park. Given the evidence that apex predators and large vertebrates are the most disturbance-sensitive species in protected areas in the global tropics [2], it is important to understand the impact roads have on multiple critically endangered species within Tesso Nilo, and to identify areas where impacts could be mitigated. We were unable to assess direct impacts to biodiversity during this study, but other studies have shown the negative impacts of road effects and human activity on tropical flora and fauna [9, 25].

From our digitization, it is clear that roads within Tesso Nilo have increased greatly, likely due to increased oil palm expansion within the park. In 2016, all of the natural forest within the protected area was within 2 km of a road, and thus within human reach. Prior research in temperate regions shows that road densities greater than 0.38 km/km^2 can have an impact on carnivore abundance and behavior [41, 42], but the impacts of road density vary by species. Research on the impacts of road density in tropical systems is lacking, but given that most of Tesso Nilo natural forest is within 2 km of a road and the road density, at 2.63 km road/km^2, is far beyond other densities that have demonstrated impact, roads are likely to have negative impacts on most wildlife species in Tesso Nilo. Because most of the roads within Tesso Nilo are accessible by motorbike only, we suspect the species that avoid roads due to habitat loss would be most affected [9], but, despite the potentially low activity level (frequent motorbike and foot traffic versus highway traffic), impacts are likely still occurring. In Northern Sumatra, [16] found that number of species, species diversity, and species evenness were lower in a high human traffic area versus a low traffic area and that wildlife altered activity patterns to avoid human activity, even though the roads were only either human or wildlife trails and not accessible by vehicle.

From 2002 – 2016, without taking roads effects into account, 76.31% of natural forest area within Tesso Nilo was lost. The remaining natural forest area is not large enough to support more than one tiger, notwithstanding use of non-forested areas. In addition, average patch size decreased and forest patches within the park are likely now losing species at an increased rate compared to intact forest due to small forest patch size [4-7]. From 2002 to 2016, patch size within Tesso Nilo decreased 92% from 0.52 km^2 to 0.04 km^2 and average patch size of forest outside of Tesso Nilo decreased by 60% from 0.05 km^2 to 0.03 km^2. These patch sizes reflect forest patches when roads effects are not taken into account. When conservative road effects (100 m buffer) are taken into account, the 2002 average patch size (i.e. core forest areas) increased 13% and the 2016 average patch size decreased slightly. The increase in size seen in 2002 is due to smaller patches overlapping areas within the 100 m road buffer, thus being removed from the calculations. Though the 2016 total amount of forest (162.69 km^2; without incorporating road effects) may be large enough to support wildlife, the average patch size in 2016 (0.04 km^2) indicates a highly patchy landscape. Furthermore, Tesso Nilo is not well connected with other forest patches outside of the protected area, (on average 2.2 km from nearest external forest), limiting not only habitat size, but also the dispersal and movement of wildlife across the landscape, and negatively affecting non-mobile, fragmentation-sensitive species such as reptiles, amphibians, and small mammals. When less conservative road effects are taken into account – at the 500 m, 750 m or 1000 m buffer widths, there is very little forest remaining, and is likely not suitable for wildlife in the long-term.

The decrease in forest within the park increases the importance of natural forest areas outside of the protected area to provide habitat and corridors for wildlife that have been pushed out of the park. Forest outside of the protected area decreased as well, but not as much as forest within the park, possibly due to the landscape's already degraded state in 2002 outside of the park. While small, the forest loss outside of Tesso Nilo is still alarming because if little natural forest remains outside of Riau's protected areas, and the distance between neighboring forest patches is greater than a species' dispersal ability, these species may have low long-term persistence [23]. Forest patches in Tesso Nilo are now >2 km from the closest forest patch outside of the park, which may be farther than some insect, small mammal and amphibian dispersal distances. This distance limits patch usefulness as stepping stones between protected areas. On the other hand, forested areas are within 0.01 km from, or nearly adjacent to, palm oil plantations. While some species may be able to survive in plantations, biodiversity in palm oil plantations is significantly lower than in natural forest [43], and we may be at risk of losing lowland Riau's unique assemblage of biodiversity.

We found a significant difference in temperature inside and outside of roads throughout Tesso Nilo and a significant difference between interior palm oil plantation locations and interior forest locations. Whether these differences are large enough to affect wildlife is as yet unclear. In other tropical ecosystems, small changes in microhabitat conditions have been found to effect reptile activity levels [44], bird activity and habitat selection [45, 46], and leaf litter decay rates [47], which is likely to impact amphibian and insect communities. The flora and fauna of Tesso Nilo are likely impacted by these differences in microhabitat, but there is no baseline on taxa present within this system, and research on non-mammalian communities in Riau is lacking. Furthermore, sample size is relatively small and we were restricted to measuring humidity and temperature. A larger sample size and additional data (soil moisture and temperature, canopy temperature, oxygen levels, wind speed, radiation, etc.) may provide better insights into the potential impacts of roads on wildlife. Additionally, sampling locations along palm oil plantation roads likely have similar microhabitat characteristics as locations 100 m from these roads, especially if the plantation is newly planted and does not have a closed canopy. We suggest future research focus on population distribution, dispersal, and persistence of the insect, small mammal, primate, bird, and arboreal species of Tesso Nilo.

To improve the accuracy of our newly created roads layers, we attempted to ground-truth the data by driving many of the roads within Tesso Nilo. However, some areas of Tesso Nilo have been illegally converted to oil palm and ownership is highly contested, thus the field team was unable to survey these areas. Nevertheless we were able to combine ground-truthing with GIS layers to substantially increase the accuracy of the road layers map. We digitized roads using Google Earth imagery, but do note that visibility in some images was restricted by smoke or cloud cover. Despite our limitations, our new digitized road maps provide fresh insight into the high level of disturbance happening in and around Tesso Nilo National Park. Since these roads are not officially documented by the government, and there is very little on-the-ground enforcement, Tesso Nilo may lose even more forest through the access these roads provide, and due to the lack of enforcement at access points.

It is clear that though often used as an example of protected lowland forest, due to roads, the forest in Tesso Nilo is now highly fragmented and likely impaired due to edge effects and impacts of degradation from outside of the park and it is fairly isolated from nearby forest. Further research should include how deforestation in the park compares to that outside of the park to determine if retaliatory deforestation is occurring. It is clear that Tesso Nilo is not providing the protection it was intended to, and is likely acting only as a ‘paper park’. The unique lowland Eastern Peneplain eco-floristic sector, one of 38 unique zones in Sumatra identified by floral and geologic features, which used to cover a large portion of Riau is now listed as Critically Endangered and only remains within Tesso Nilo National Park [29]. The most recent estimates (2007) indicate >70% of this zone has been lost, and today, given the further deforestation and degradation we have documented within Tesso Nilo, it is likely that very little of this zone remains. Few studies have focused on cataloging or quantifying changes in the insect, bird, small mammal, amphibian, and reptile communities that inhabit this lowland area of Sumatra. If deforestation and degradation within Tesso Nilo continue, it is likely this entire eco-floristic zone, any benefits it provides as a refuge for climate change or from poaching, and the endangered endemic wildlife species that inhabit it, will be lost [48].

Continued loss of Tesso Nilo and its endemic biodiversity, and its wildlife of global importance such as Sumatran elephant and tiger, could result in reduced wildlife dispersal across greater Riau, and an increased chance of isolation and extinction in the near future [48]. Without forest assessments that take road effects into account, Tesso Nilo and other tropical protected areas may appear to be ecologically healthier and larger than they actually are. The potentially far-reaching effects of roads can have significant negative impacts on forest extent and distribution that are not currently always taken into consideration – especially when roads are small, or in systems that are thought to be ‘intact’ [49]. Quantifying the effects of roads on forest distribution and microclimate and working to decrease human activity and mitigate road impacts in this unique, understudied system of Tesso Nilo National Park and the surrounding system should be a global conservation priority.

References

1. Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A et al. High-resolution global maps of 21st-century forest cover change. Science. 2013; 342(6160): 850-853.
2. Laurance WF, Useche DC, Rendeiro J, Kalka M, Bradshaw CJA, Sloan SP, et al. Averting biodiversity collapse in tropical forest protected areas. Nat. 2012, 489: 290-294.
3. MacArthur RH, Wilson EO. The theory of island biogeography. Princeton, New Jersey: Princeton University Press; 1967.
4. Laurance WF, Albernaz AKM, Schroth G, Fearnside PM, Bergen S, Venticinque EM, et al. Predictors of deforestation in the Brazilian Amazon. J Biogeogr. 2002; 29: 737-748.
5. Lovejoy TE, Bierregaard Jr RO, Rylands AB, Malcom JR, Quintela CE, Harper LH, et al. Edge and other effects of isolation on Amazon forest fragments. In M.E. Soule editor. Conserv Biol: The science of scarcity and diversity. Sunderland, Massachusetts: Sinauer Associates; 1986. pp 257-285.
6. Harper LH. The persistence of ant-following birds in small Amazonian forest fragments. Acta Amazon.1989; 19: 249-263,
7. Brown Jr KS, Hutchings RW. Disturbance, fragmentation, and the dynamics of diversity in Amazonian forest butterflies. In Laurance WF, Bierregaard Jr RO, editors. Tropical forest remnants: ecology, management, and conservation of fragmented communities. Chicago: University of Chicago Press; 1997. pp 91-110.
8. Gascon C, Lovejoy TE, Bierregaard Jr RO, Malcolm JR, Stouffer PC, Vasconcelos HL, et al. Matrix habitat and species richness in tropical forest remnants. Biol Conserv. 1999; 91(2-3): 223-229.
9. Fahrig L, Rytwinski T. Effects of roads on animal abundance: An empirical review and synthesis. Ecol Soc. 2009; http://www.ecologyandsociety.org/vol14/iss1/art21/Studies.
10. Mace RD, Waller JS, Manley TL, Lyon LJ, Zurring H. Relationships among grizzly bears, roads, and habitat in the Swan Mountains, Montana. J Appl Ecol. 1996; 33: 1395-1404.
11. Van Dyke FG, Brocke RH, Shaw HG. Use of road track counts as indices of mountain lion presence. J Wildl Manage. 1986; 50: 102-109.
12. Mech LD, Fritts SH, Radde GL, Paul WJ. Wolf distribution and road density in Minnesota. Wildl Soc Bull. 1988; 16: 85-87.
13. Newmark WD, Boshe JI, Sariko HI, Makumbule GK. Effects of a highway on large mammals in Mikumi National Park, Tanzania. Afr J Ecol. 1996; 34:15-31.
14. Porej D, Micacchion M, Hetherington TE. Core terrestrial habitat for conservation of local populations of salamanders and wood frogs in agricultural landscapes. Biol Conserv. 2004; 120: 399-409.
15. Dyer SJ, O'Neill JP, Wasel SM, Boutin S. Avoidance of industrial development by woodland caribou. J Wildl Manage. 2001; 65(3): 531-542.
16. Griffiths M, van Schaik CP. The impact of human traffic on the abundance and activity periods of Sumatran rainforest wildlife forest. Conserv Biol. 1993; 7(3): 623-626.
17. Carr LW, Fahrig L. Effect of road traffic on two amphibian species of differing vagility. Conserv Biol. 2001; 15: 1071-1078.
18. Reijnen R, Foppen R, Meeuwsen H. The effects of traffic on the density of breeding birds in dutch agricultural grasslands. Biol Conserv. 1996; 75: 255-260.

19. Goosem M. 2000. Fragmentation impacts caused by roads through rainforests. Curr Sci. 2000; 93(11): 1587-1596.
20. Kapos V. Effects of isolation on the water status of forest patches in the Brazilian Amazon. J Trop Ecol. 1989; 5:173-185.
21. Laurance WF, Birregaard Jr RO, Gascon C, Didham RK, Smith AP, Lynam AJ, et al. Tropical forest fragmentation: synthesis of a diverse and dynamic discipline. In Laurance WF, Bierregaard Jr RO, editors. Tropical forest remnants: ecology, management, and conservation of fragmented communities. Chicago: University of Chicago Press; 1997. pp. 502-514.
22. Laurance WF. Tropical logging and human invasions. Conserv Biol. 2001; 15: 4-5.
23. Laurance WF, Ferreira LV, Rankin-de Merona JM, Laurance SG. Rain forest fragmentation and dynamics of Amazonian tree communities. Ecol. 1998, 79(6): 2032-2040.
24. Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J. Biodiversity hotspots for conservation. Nat. 2000; 403: 853-858.
25. Kinnaird MF, Sanderson EW, Brien TGO, Wibisono HT, Woolmer G. Deforestation trends in a tropical landscape and implications for endangered large mammals. Conserv Biol. 2010; 17(1): 245–257.
26. Land Resources Department/Bina program. The land resources of Indonesia: A national overview from regional physical planning program for transmigration. Jakarta, Indonesia: Land Resource Department, Natural Resources Institute, Direktorat Jenderal Penyiapan Pemukiman, Departemen Transmigrasi; 1990.
27. Margono BA, Potapov PV, Turubanova S, Stolle F, Hansen MC. Primary forest cover loss in Indonesia over 2000–2012. Nat Clim Chang. 2014; 4: 730-735.
28. Sunarto S, Kelly MJ, Parakkasi K, Klenzendorf S, Septayuda E, Kurniawan H. 2012. Tigers need cover: multi-scale occupancy study of the big cat in Sumatran forest and plantation landscapes. PloS One. 7(1): e30859.
29. Laumonier Y, Uryu Y, Stüwe M, Budiman A, Setiabudi B, Hadian, O. Eco floristic sectors and deforestation threats in Sumatra: Identifying new conservation area network priorities for ecosystem-based land use planning. Biodivers Conserv. 2010; 19(4): 1153-1174.
30. Poor EE, Shao Y, Kelly MJ. Mapping and predicting forest loss in a Sumatra tiger landscape from 2002 – 2050. Environ Manage. Forthcoming.
31. Sanderson E, Forrest J, Loucks C, Ginsberg J, Dinerstein E, Seidensticker J, et al. Setting Priorities for the Conservation and Recovery of Wild Tigers: 2005–2015. Washington, DC: Wildlife Conservation Society, World Wildlife Fund, Smithsonian, and Save the Tiger Fund; 2006.
32. ESRI. ArcGIS Desktop: Release 10.5. Redlands, CA: Environmental Systems Research Institute. 2017.
33. Lovallo MJ, Anderson EM. Bobcat movements and home ranges relative to roads in Wisconsin. Wildl Soc Bull. 1996; 24: 71-76.
34. Rheindt FE. The impact of roads on birds: does song frequency play a role in determining susceptibility to noise pollution? J Ornithol. 2003; 144: 295-306.
35. Dickson BG, Beier P. Home-range and habitat selection by adult cougars in southern California. J Wildl Manage. 2002; 66: 1235-1245.
36. Eigenbrod F, Hecnar SJ, Fahrig L. Accessible habitat: an improved measure of the effects of habitat loss and roads on wildlife populations. Landsc Ecol. 2008; 23: 159-168.

37. Kunkel KE, Pletscher DH. Habitat factors affecting vulnerability of moose to predation by wolves in southeastern British Columbia. Can J Zool. 2000; 78: 150-157.
38. Reynolds-Hoglan MJ, Mitchell MS. Effects of roads on habitat quality for bears in the southern Appalachians: A long-term study. J Mammal. 2007; 88: 1050-1061.
39. Laurance WF, Yensen E. Predicting the impacts of edge effects in fragmented habitats. Biol Conserv. 1991; 55(1): 77-92.
40. R Core Development Team. R: A language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna Austria. https://www.R-project.org; 2016.
41. Jedrzejewski W, Niedzialkowska M, Nowak S, Jedrzejewska B. Habitat variables associated with wolf distribution and abundance in northern Poland. J Wildl Manage. 2002; 66: 1235-1245.
42. Fuller TK. Population dynamics of wolves in north-central Minnesota. Wildl Monogr. 1989; 105: 1-41.
43. Danielsen F, Beukema H, Burgess ND, Parish F, Bruhl CA, Donald PF, et al. Biofuel plantations on forested lands: double jeopardy for biodiversity. Conserv Biol. 2009; 23(2): 348-358.
44. Logan ML, Fernandez SG, Calsbeek R. 2015. Abiotic constraints on the activity of tropical lizards. Funct Ecol. 2015; 29: 694-700.
45. Janzen DH. Why mountain passes are higher in the tropics. Am Nat. 1967; 101:233-250.
46. Karr JR, Freemark KE. Habitat selection and environmental gradients: dynamics in the “stable” tropics. Ecol Monogr. 1983; 64: 1481-1494.
47. Salinas AN, Malhi Y, Meir P, Silman M, Cuesta RR, Huaman J, et al. The sensitivity of tropical leaf litter decomposition to temperature : results from a large-scale leaf translocation experiment along an elevation gradient in Peruvian forests. New Phytol. 2011; 189(4): 967-977.
48. Imron MA, Herzog S, Berger U. The influence of agroforestry and other land-use types on the persistence of a Sumatran tiger (*Panthera tigris sumatrae*) population: an individual-based model approach. Environ Manage. 2011; 48(2): 276–288.
49. Hughes AC. Have Indo-Malaysian forests hit the end of the road? Biol Conserv. 2018; 223: 129-137.

Tables

Table 1. Amount of forest, number of patches and distance between forest patches within Tesso Nilo National Park (TNNP) when taking roads into account at different buffer widths.

	No Roads			100 m Buffer			500 m Buffer			750 m Buffer			1000 m Buffer		
	2002	2014	2016	2002	2014	2016	2002	2014	2016	2002	2014	2016	2002	2014	2016
Amount of TNNP Forest (km2)	688.83	182.16	162.69	589.77	140.61	118.49	233.27	31.13	22.32	131.03	10.88	7.37	72.19	3.85	2.12
Number of Patches	1305	3611	4313	1002	2899	3321	224	662	726	90	278	278	47	84	109
Nearest Patch (m)	21.28	30.89	37.79	34.62	39.69	49.79	257.2	58.7	73.57	519.83	57.18	74.7	1040.59	167.02	133.52
Average Patch Size (km2)	0.52	0.05	0.04	0.59	0.05	0.04	1.04	0.05	0.03	1.45	0.04	0.03	1.54	0.05	0.02
Taxa affected				Small carnivores, birds			Amphibians, mid-sized carnivores			Ungulates, large carnivores			Large carnivores		

Table 2. Distance of forest patches within Tesso Nilo (without roads accounted for) to other land cover (LC) types inside and outside of Tesso Nilo.

	2002	2014	2016
TNNP Forest to Interior LC (km)			
Plantation	0.001	0.001	0.010
Open	0.167	0.419	0.357
Forest	0.200	0.114	0.111
TNNP Forest to Exterior LC (km)			
Plantation	2.40	2.86	2.14
Open	2.80	2.86	2.32
Forest	2.39	2.76	2.20

Figures

Figure 1. Location of entire the greater study area landscape (top), and Tesso Nilo National Park in relation to nearby protected areas, Rimbang Baling Wildlife Reserve and Bukit Tigapuluh National Park within central Sumatra, Indonesia.

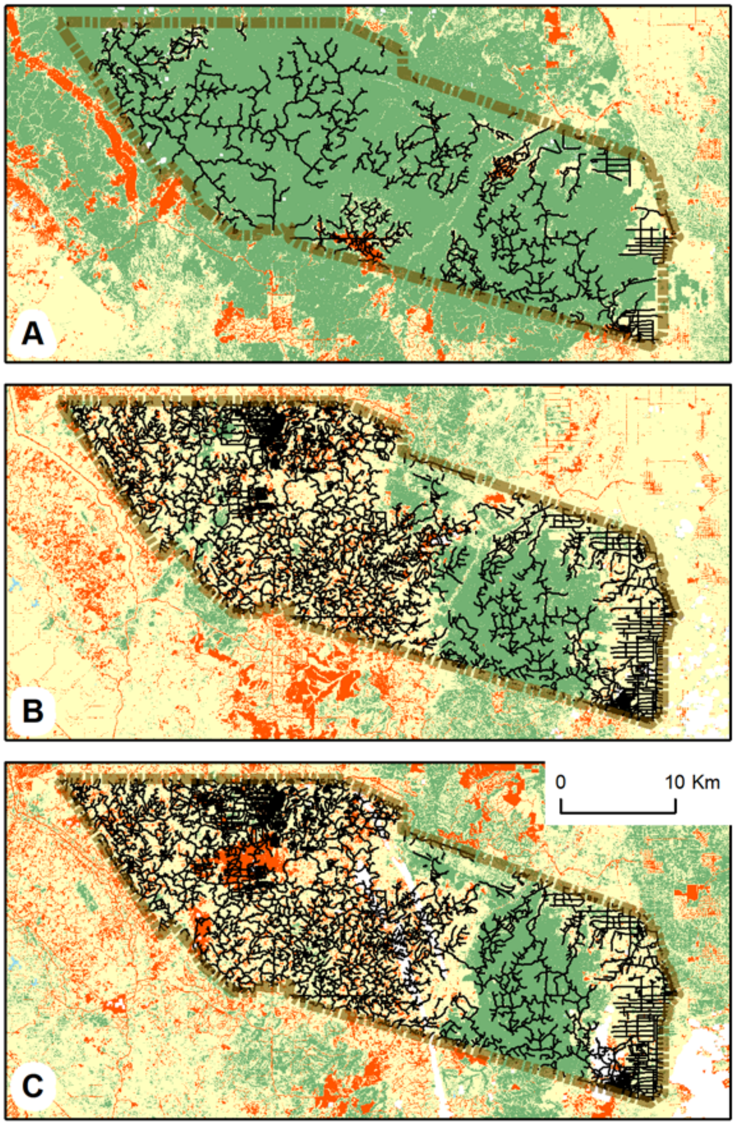

Figure 2. Roads (without impact buffers) and land cover (Yellow – plantation, Orange – bare land, Green – forest) within Tesso Nilo National Park, Riau, Sumatra in 2002 (A), 2014 (B) and 2016 (C).

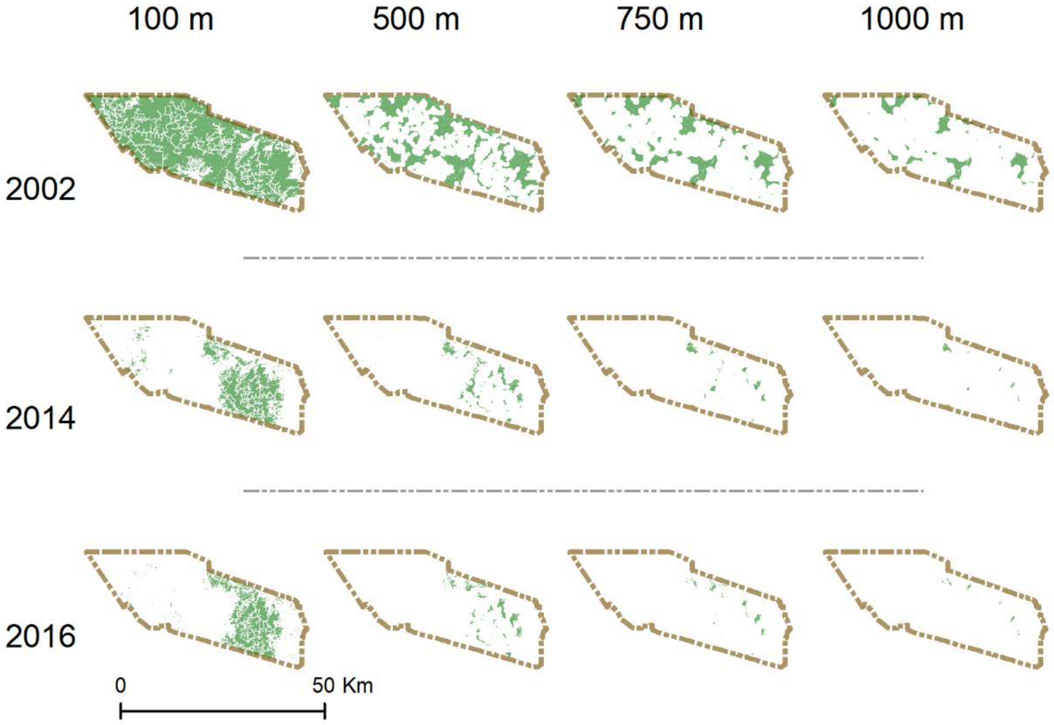

Figure 3. Natural forest distribution in Tesso Nilo National Park for 2002, 2014 and 2016 when the effects of roads are taken into account at three different distances: 100 m, 500 m, 750 m and 1000 m.

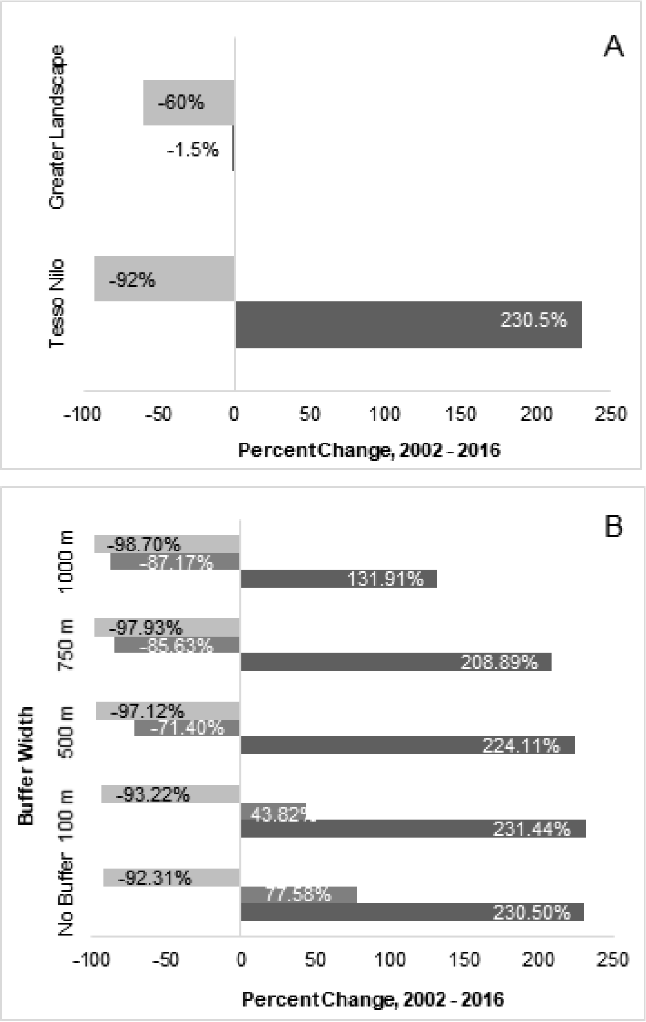

Figure 4. Percent change in average forest patch area (light gray) and the number of forest patches (dark gray) and average forest patch area (light gray) in Tesso Nilo and the greater landscape (A), and percent change in number of forest patches (dark gray), average forest patch area (light gray) and average distance to neighboring forest patches (medium gray) at different road impact widths (B) within Tesso Nilo.

Chapter 5

Pigs and space: A comparison of temporal and spatial partitioning of tiger prey in the presence of native and human predators in central Sumatra

Erin E. Poor, Marcella J. Kelly

Abstract

Globally, Southeast Asian mammals are some of the most threatened by both habitat destruction and poaching, and are also some of the least studied. While negative impacts of anthropogenic landscape changes are well documented, comparatively little research focuses on human impacts on predator-prey relationships in tropical systems. These relationships in Sumatra, Indonesia, are particularly understudied, even though the persistence of critically endangered species like the Sumatran tiger (*Panthera tigris sumatrae*) is tenuous. In this study, we compare spatial occupancy, or site use, and temporal activity changes, of three felid prey in the presence of felid and human predators. We use data from 33 camera traps deployed for five months to provide valuable new insights on prey behavior in a little-studied central Sumatra protected area, Rimbang Baling Wildlife Reserve. Wild pigs (*Sus scrofa* and *Sus barbatus*) had a high site use rate ($\Psi = 0.79$) and we found no evidence of avoidance behavior in response to humans (species interaction factor [SIF] = 1). Site use results for muntjac (*Muntiacus muntjak*) were inconclusive due to model failure. Mouse deer (*Tragulus napu* and *Tragulus kanchil*) detection declines drastically in areas with humans, showing potential avoidance behavior, and this relationship is mediated by habitat; mouse deer site use and SIF with humans decline with terrain ruggedness. However, mouse deer appeared to shift their daily activities to avoid felids ($\Delta_{hat1} = 0.52$) more than humans ($\Delta_{hat1} = 0.71$). We found evidence for the refuge effect for pigs, where pig site use appears positively impacted by humans, and the risk disturbance hypothesis for mouse deer, where prey shift behavior to avoid humans. Relatively little is known about tiger prey in Sumatra and across their range, and we recommend furthering this, and similar research, to more fully understand these systems. Doing so is key to increasing threatened carnivore populations in the future.

Keywords

human impacts, occupancy, predation, prey, site use, Sumatran tiger, temporal activity

Introduction

Worldwide, we are losing biodiversity at unprecedented rates (Pimm et al. 2014) largely due to habitat loss and unregulated hunting. Southeast Asian forests are experiencing the highest global deforestation rates and 13 – 85% of biodiversity in this region could be lost by 2100 (Sodhi et al. 2009). Not surprisingly, wildlife in Southeast Asia is at higher risk of extinction than in any other region (Dalerum et al. 2009) and the main threats to biodiversity include infrastructure development, leading to increases in roads and forest access that facilitate wildlife poaching for trade or consumption.

Human consumption of wildlife has occurred for thousands of years, but unregulated and illegal hunting has now reached unsustainable levels (Darimont et al. 2015). The illegal global wildlife trade has reached \$7 - \$23 billion annually, a figure that will likely continue to grow, given the inverse relationship between species abundance and market price (Nelleman et al. 2016). Southeast Asia as a region has the most species threatened by hunting, and Indonesia as a country, has an extremely high number of threatened endemic species (UNODC 2013). With the highest deforestation rate in the world (Margono et al. 2014), Indonesian species threatened by human consumption and trade are also threatened by habitat loss and degradation. Many of these threatened or endangered species such as sambar deer (*Rusa unicolor*) and langur (*Presbytis* spp.) are also primary prey of native predators, such as tigers (*Panthera tigris sumatrae*) and clouded leopards (*Neofelis diardi*). Carnivore abundance is widely thought to be tied to ungulate prey abundance (Karanth and Nichols 1998; Karanth et al. 2004), so even if carnivores are not targeted by hunting per se, decreases in prey availability can significantly impact carnivore populations.

Carnivores worldwide have been shown to avoid areas of human activity, especially roads (Mace et al. 1996, van Dyke et al. 1986, and Mech et al. 1988), but there is less consensus regarding the response of prey species, specifically ungulates, to the presence of humans (Stankowich 2008). There are multiple theories about how prey species alter behavior in response to human presence and each has been supported in different contexts. The risk disturbance theory (Frid and Dill 2002) posits that animals should respond to human-induced disturbances in the same manner that they respond to natural predators. Waldstein et al. (2016) found evidence supporting this theory in the southeastern US, where white-tailed deer (*Odocoileus virginianus*) avoided areas of humans on foot. Evidence has also been found to support the predator shelter hypothesis or the ‘refuge effect’ (Berger et al. 2001; Shannon et al. 2014). In the western US, pronghorn (*Antilocapra americana*) and elk (*Cervus canadensis*) spent more time foraging within 500 m of roads than they did at distances >500 m (Shannon et al. 2014), presumable due to the “shelter” provided them by humans who hunt predators. Ungulates have also been found to exhibit apparent attraction to areas of anthropogenic noise (Brown et al. 2012). It is clear that ungulate response to humans may depend on situational context, species, habitat, prior experiences with humans, and a number of other factors (Stankowich 2008).

Carnivores alter their behavior in the presence of humans or roads in a number of ways. In Thailand, after a popular national park was shut to tourism, leopards altered activity from a nocturnal to a diurnal pattern (Ngoprasert et al. 2017). Human presence can also lead to disruptions in hunting or feeding, thus leading to decreased calorie intake, which can lead to a decrease in fitness over a long period of time (Kerley et al. 2002). Roads have been shown repeatedly to have an effect on carnivore home range selection (Dickson and Beier 2002, Lovallo and Anderson 1996, Mace et al. 1996), potentially leading to restricted or sub-optimal home ranges, which could have negative consequences on survival and reproduction.

In Sumatra, tigers exist at naturally low densities (0.30 – 0.87/100 km^2) across about 88,000 km^2 of remaining forest (Sanderson et al. 2006; Sunarto et al. 2013). Protected areas in Sumatra that typically shelter remaining forest, have recently lost significant forest, especially in Riau, Province, where approximately 50% of natural forest has been lost to the oil palm plantations and other large-scale agricultural operations (Poor et al. in review). Tigers across their range prefer large bodied ungulate prey, such as sambar deer, guar (*Bos gaurus*), muntjac (*Muntius muntjakus*), and wild pig (*Sus scrofa* and *Sus barbatus*), but will also eat primates and smaller prey (Karanth and Sunquist 1995, Pakpien et al. 2017, O'Brien et al. 2003). In addition to tigers, Sunda clouded leopards (*Neofelis diardi*), Asiatic golden cats (*Catopuma temminckii*), marbled cats (*Pardofelis marmorata*), and leopard cats (*Prionailurus bengalensis*) also inhabit central Sumatra (Sunarto et al. 2015). While far less is known about these smaller bodied felids than tigers, Sunarto et al. (2015) found that all felids manage to co-exist in this landscape likely by partitioning space, time, or prey. The most common tiger prey species in this landscape that could be predated upon by other felids include (in order of body size) wild pig (~35 kg), muntjac (~20 kg), and mouse deer (~ 6 kg; *Tragulus kanchil* or *Tragulus napu*).

Humans in Sumatra often hunt on foot with guns, snares, and traps, sometimes remaining in the forest for several days. Anecdotal evidence suggests humans remain within a few kilometers of a village, but data to support this claim is still lacking. Humans can be frequently seen in protected areas in Riau, central Sumatra, hunting, collecting birds and non-timber forest products, and logging. While humans have been shown to have multiple negative impacts on wildlife worldwide and poaching is common in Sumatra, there is little research examining whether human activity is frequent enough to affect wildlife behavior in this system. We hypothesize that as natural predators such as wild felid species have declined in central Sumatra, humans have taken over the role of apex predator to maintain a landscape of fear for prey species. In this study, we use single species occupancy and two-species spatial co-occurrence modeling, and temporal overlap analyses to examine the predator-prey relationships of humans, felids, and three relatively common felid (and potential human) prey: wild pig, muntjac, mouse deer.

Methods

Study Area

Riau Province, in central Sumatra is extremely biodiverse, with mountains on its western edge and central lowland moist tropical forests leading to peat forests on its eastern edge. Riau's classified in the Koppen-Geiger system as Af, tropical, with an average temperature is 27° C and average rainfall is 2696 mm per year. Rimbang Baling Wildlife Reserve, established in 1986, is Riau Sumatra's largest protected area at 1,366 km^2. It lies on the border of Riau and West Sumatra provinces, remaining connected to the forested spine along the western side of Sumatra that makes up several national parks (Figure 1). Rimbang Baling is mountainous, with elevation ranging from 29 to 1200 m asl. Though there is moderate to severe encroachment around the reserve's edges, there is still lowland, moist, tropical forest and montane and scrub forests at higher elevations and prior research (Sunarto et al. 2012) indicates relatively high tiger occupancy probability in our study area. Encroachment areas largely are used for oil palm, rubber, acacia, and fruit plantations. Local villagers commonly use the bordering forest for logging, hunting, resin harvesting, and bird trapping. Rimbang Baling is extremely rugged, making travel to the interior difficult, thus we placed cameras around the edge, within one or two days' walk of a village. There is a local World Wildlife Fund camp on the eastern edge of Rimbang Baling, which was used as a basecamp from which to distribute cameras. This area consists of a matrix of ex-logging and mining roads, plantations, and forest edge (Figure 1).

Data Preparation

From March – July 2016, we conducted a camera trap survey in and around Rimbang Baling Wildlife Reserve. Thirty-three unpaired and un-baited camera traps were installed on trees or posts about 30 cm above the ground, and installation and removal took place throughout a three month period. Cameras were placed opportunistically along wildlife trails and old logging roads and were checked approximately every month. We used Bushnell Natureview HD Live View and Reconyx Hyperfire Pro cameras. The Bushnell Natureview cameras were set to take 15 or 20 s video with 10-15 s intervals between captures. Reconyx cameras were set to 3 photos per trigger and 30 s intervals between captures. Cameras were inside and outside of the protected area, within 4 km of the field camp and within 3.5 km of the protected area boundary (Figure 1).

Capture events were considered captures of all distinctly different individuals of the same species within 30 minute time periods. If there were multiple individuals captured in the same photo or video, we counted each individual as a separate capture (i.e. for calculating trap success as number of capture events per trap night *100). Due to difficulties in distinguishing between bearded and Eurasian pigs (*S. barbatus* and *S. scrofa*), and greater and lesser mouse deer (*T. napu* and *T. kanchil*) in some captures, we combined these species into groups, pigs (*spp.*) and mouse deer (*spp.*). We combined all cats into one group due to their low trap rates, and our aim to compare effects of native predators to human predators. Because our cameras were non-randomly distributed, approximately 500 m from each other at the closest, we consider our measure as probability of site use, rather than true occupancy.

For our site use analysis, we derived land cover covariates (Poor et al. in review) and Shuttle Radar Topography Mission (SRTM) elevation data (Farr et al. 2007). Candidate covariates included point and landscape scale measures of habitat (Farris et al. 2015) as identified within a 2 km radius surrounding each camera station (Table S1). We used average terrain ruggedness index (TRI; Riley et al. 1999) within 2 km, percent disturbed area within 2 km, percent forested area within 2 km, number of forest patches within 2 km, distance to forest larger than 500 km^2 (Sunarto et al. 2015), distance to disturbed areas, distance to plantations, distance to rivers, distance to roads, and distance to cities, and TRI at camera locations. We tested co-variates for correlation and removed those variables with Pearson correlation coefficients >0.7. Our final set of covariates included percent of area forested within 2 km, average TRI within 2 km of camera stations, distance to rivers, distance to roads, distance to forest >500 km^2, and distance to disturbed areas. We also included trap success of humans and all cats combined (tigers, clouded leopards, marbled cats, golden cats, and leopard cats) in our occupancy models. Field teams were not included in human detections. Covariate data was prepared in ArcGIS 10.5 (Esri 2018).

Predator Impacts on Prey Distribution

To determine whether humans affected focal prey species spatial distributions, we used single species and two species occupancy models, which allow us to incorporate imperfect detection into the modeling process (MacKenzie et al. 2006). For occupancy analysis, we created capture histories for each species by recording the presence or absence of each species, each day, at each station. We first created single species occupancy models for pigs, muntjac, mouse deer, cats, and humans, and analyzed data using program PRESENCE (Hines, 2006). We modeled each covariate's influence on occupancy while keeping detection constant, and then retained important occupancy covariates while modeling covariates on detection. However, we limited detection covariates to human and felid trap success because we were interested in identifying predation impacts, and with our relatively small data set, we wanted to avoid potentially spurious results from non-randomly distributed cameras and small sample sizes. We ranked models using Akaike's Information Criterion (AIC) adjusted for small sample size (AICc) (Akaike 1973), and we report top-ranking models with ΔAICc > 2.

After determining important covariates in single-species models, we retained those covariates in each top ranking model set within our predator-prey, co-occurrence models in order to assess impacts of both habitat and other species. In PRESENCE, we used the psiBa/rBa parameterization due to its increased stability and ability to incorporate covariates in comparison to the phi/delta parameterization (Richmond et al. 2010). The psiBa/rBa parameterization contains eight parameters: psiA, occupancy of the dominant species, A; psiBA, occupancy of the subdominant species (B) when the dominant species is present; psiBa, occupancy of the subdominant species when the dominant species is absent; pA, detection of the dominant species when the subdominant species is absent, and vice versa, pB; rA, detection probability of A if both species are present; rBA, detection of species B, if species A is detected, and finally, rBa, detection of the subdominant species when the dominant species is present but not detected. We were primarily interested in whether psiBA = psiBa, indicating no impact of the presence of the dominant species on the subdominant species (independent co-occurrence), or whether psiBA ≠

psiBa. If psiBA ≠ psiBa, site use of species B is influenced either positively or negatively by the presence of the dominant species. Furthermore, this relationship could be mediated by one or more covariates in the case of psiBA ≠ psiBa(Covariate), indicating a changing relationship between the dominant and subdominant species depending on covariate values.

We created co-occurrence models for six combinations of our two predator groups (felids and humans) and three prey groups (pigs, muntjac, and mouse deer). We assumed that predators would be dominant over prey species, assuming top-down regulation (Terborgh and Estes 2010). For each of the six site use models, we used an iterative process to identify candidate models, beginning with one covariate on one parameter and using covariates that improved model performance in subsequent models to identify our top models. Once top models were identified with additive effects of covariates, we tested interactions of the same covariates to determine if models could be further improved. We focused first on estimating occupancy parameters, before attempting to estimate detection parameters. As with single species models, to limit our number of candidate models, we restricted detection covariates to human and felid trap success. For each top model, we also calculated the species interaction factor (SIF, Φ), from model parameters. A SIF >1 (with CIs that do not overlap 1) indicates co-occurrence more often than expected by chance, while SIF < 1 indicates co-occurrence less often than expected by chance. A SIF = 1 indicates independent site use, which is the case when psiBA = psiBa. We calculated SIF in R (R Core Development Team, 2018) using the covariance-variance matrix created using program PRESENCE.

Predator Impacts on Prey Activity Patterns

To determine whether humans or cats impact prey species activity patterns throughout the day, we used package overlap (Meredith and Ridout 2018) in R v. 3.4 (R Core Development Team, 2018), following the methods of Ridout and Linkie (2009). We sorted pig, muntjac, and mouse deer captures by those that were at stations 1) with humans and without cats (human predator present), and 2) with cats and without humans (native predator present) and 3) those capture events at stations without humans or cats (predator free). We converted time of capture to circular data by using radians as time and calculated the overlap in prey activity densities in the presence and absence of both predator types. We then compared prey activity densities in the presence of both predator types. Because we had relatively small sample sizes, we used Δ_{Hat1} as the estimator of overlap (Ridout and Linkie 2009).

Results

In total, we had 2,186 trap nights. Trap success was low across all species (Table 1). Detections were highest for wild pigs and humans and lowest for tigers (Table 1). Trap success was highest for wild pigs (25.39/100 nights). For all felids combined, we had 46 total capture events and we had 76 human capture events. We did not test single species felid models due to low trap rates, and the combined cat occupancy model did not converge.

Predator Impacts on Prey Distribution

Site use in all single species, top models of prey occupancy was influenced by TRI, distance to roads, distance to forest > 500 km^2, percent of forest within 2 km of the camera, and distance to disturbed areas (Table 2). Pigs were at sites with less rugged terrain that were closer to disturbed areas, and in areas with less forest surrounding the camera site (Table 2). Muntjac habitat models did not converge and were excluded from further analysis. Mouse deer occupied sites closer to large forest patches, farther from roads, and with less rugged terrain. Detection in all top, single species site use models for prey species was influenced by human trap success. Pig detection was positively influenced by human trap success whereas mouse deer detection was negatively influenced by human trap success whereas.

Humans were found at sites closer to large forest patches, farther from roads, closer to disturbed areas, and where cat trap success was higher (Table 2). We did not test landscape and habitat variables for influence on human detection because we do not believe humans would alter their own behavior in response to any of these factors in this landscape.

Of the six predator-prey pairings in 2-species modeling, we were unable to estimate co-occurrence for muntjac-cats and muntjac-human due to non-convergence. Of the four relationships we were able to estimate, all exhibited independent relationships except mouse deer – human (psiBA ≠ psiBa; SIF > 1; Table 3; Figure 2). Mouse deer and humans occurred at the same locations more often than expected, but this and relationship was mediated by topographic ruggedness index (Figure 2). Mouse deer occupancy at sites with humans strongly declined as terrain ruggedness declined (Figure 2), whereas mouse deer occupancy at sites without humans remained relatively constant, declining slightly as TRI increased (Figure 2).

Predator Impacts on Prey Activity Patterns

In all cases, prey species activity patterns overlapped more with humans than with felids (Figure 3). Pigs reacted similarly to the presence of both felids and humans, increasing activity levels mid-day, and decreasing activity at dawn and dusk compared to when there were no predators present. For all prey species, pig activity overlapped the most with predators (Δ_{hat1} = 0.86; 95% CI = 0.74 – 0.86 for humans, Δ_{hat1} = 0.75 with cats; 95% CI = 0.68 – 0.92). When pig activity in the presence of humans is compared to activity in the presence of cats, activity levels and patterns are extremely similar (Δ_{hat1} = 0.86; 95% CI = 0.68 – 0.89). While confidence intervals are wide due to low sample size, point estimates suggest the presence of humans, pig activity shifts to a more diurnal pattern, compared to activity in the absence of predators.

Muntjac showed the smallest change in activity pattern in the presence humans, Δ_{hat1} = 0.83 (95% CI = 0.68 – 0.89) and only slight shifts in activity to more crepuscular patterns, as compared to muntjac activity in the presence of felids (Δ_{hat1} = 0.73; 95% CI = 0.64 – 0.94). In the presence of felids, activity is increased at dawn and dusk, and reduce during mid-day. Furthermore, when activity in the presence of humans and felids is compared, muntjac activity patterns are nearly the same as muntjac activity with and without felids, possibly indicating a lack of response to human presence in terms of temporal activity alone.

Mouse deer activity levels overlap the least with felids (Δ_{hat1} = 0.52; 95% CI = 0.59 – 0.99), though confidence intervals are wide due to a low sample size. Activity shifts only slightly in the presence of humans, and appears similar to mouse deer activity patterns in the absence of felid predators (Δ_{hat1} = 0.71; 95% CI = 0.66 – 0.90), showing a strong crepuscular pattern. In the presence of felids, there is a large shift in activity to diurnal activity (Figure 3). Mouse deer activity levels in the presence of humans and cats overlap the least of three focal prey, with a more apparent temporal shift and possible avoidance in response to felid presence versus human presence.

Discussion

Even though we were limited in this study by both the number of cameras and the number of captures, few studies have yet examined occupancy of tiger prey or humans in Sumatran tiger habitat, especially in light of the drastic conversion of forest to plantations in central Riau. Potential tiger-prey interactions and prey ecology research are lacking in this region in general, and in central Sumatra specifically (Ripple et al. 2016). Furthermore, many potential felid prey may be threatened by hunting, either directly or indirectly as by-catch (tapir, langur, sambar, wild pig, chevrotain), thus making it critical to understand predator-prey interactions, prey ecology, and human impacts on both. Unfortunately in this central Sumatran ecosystem, ecological communities likely have already been significantly impacted by forest loss and human activities resulting in a shifted community and thus preventing true assessment of original community relationships. Nevertheless, assessing shifted ecosystems now will allow us to better understand future changes and prevent further declines.

In other areas of the tiger's range, sambar and other large ungulates have been recorded as primary tiger prey (Karanth and Sunquist 1995, Linkie and Ridout 2011, Pakpien et al. 2017). However, we only captured one sambar over the entire duration of this study, prohibiting their inclusion in our analysis. Anecdotal reports from local researchers indicate a decline in sambar deer in this landscape, but sambar are sensitive to human presence and forest degradation and may be found in higher densities towards the interior of the park (O'Brien et al. 2003). We suspect that in an un-altered ecosystem, sambar would be at higher densities. With the decline of their primary prey, tigers may be forced to hunt non-preferred prey like wild pigs and muntjacs more frequently, leading to sub-optimal energetics, with fewer calories gained for increased effort. This could result in a declining tiger population, or one with rare reproduction (Sunquist et al. 1999).

Unsurprisingly, pigs had the highest occupancy of our focal species and had the largest temporal overlap with felid activity. Pigs are often recorded in large family groups or as single males. Our site use analysis did not reveal overt avoidance of humans by pigs, but there was a slight shift toward increased diurnal activity and less crepuscular activity in the presence of felids. Lack of human avoidance, and a positive impact of humans on pig detection, may be due to the predominately Muslim human population in our study area, resulting in lower hunting pressure than in other areas. Thus, in this landscape, humans may be sheltering pigs from felid predation. Research from other areas in Sumatra found different results with higher pig abundance at lower human densities, high spatial overlap with tiger habitat (O'Brien et al. 2003), and lower temporal overlap with tiger activity (Linkie and Ridout 2011). This suggests the use of temporal avoidance of tigers and spatial avoidance of humans, which was opposite to what we found. In our study area, pigs may not be need to avoid tigers due to low tiger density, or they may be using another strategy we were unable to detect. Additionally, many of our felid captures were of smaller cats that likely do not hunt pigs due to size limitations, thus biasing our results.

We were unable to analyze muntjac spatial site use or co-occurrence due to model failure, but we found that muntjac activity overlap results corroborated those of Linkie and Ridout (2011) who found $\Delta_{hat4} = 0.80$ (95% CI = 0.71 – 0.84) for overlap with tigers in mountainous western Sumatra. We found activity overlapped more with humans than with cats ($\Delta_{hat1} = 0.83$, 95% CI = 0.68 – 0.89 vs. $\Delta_{hat1} = 0.73$, 95% CI = 0.64 – 0.94, respectively), but CIs were overlapping. Muntjacs are hunted by humans in this landscape and may employ spatial avoidance with humans but temporal avoidance with felids. We recommend camera trapping in a wider area and for a longer duration to further elucidate the human-muntjac and felid-muntjac relationships.

Humans in our study area were regularly seen hunting mouse deer and caught on camera several times with mouse deer carcasses. We found that mouse deer and humans occur at the same sites more often than expected, but mouse deer likely alter their daily activity to regulate human encounters and their spatial relationship with humans changes depending on TRI. In the absence of predators, mouse deer have high activity peaks in the morning hours, and in the presence of both humans and cats, this morning activity is reduced, with a greater shift seen in the presence of felids alone. But, mouse deer were less likely to be detected with higher human trap success, indicating potential avoidance behavior, whereas felid trap success did not appear in any of the top models of prey site use. Interestingly, mouse deer exhibited a habitat mediated, co-occurrence relationship with humans. In areas with low TRI, presumably where hunting is easier, mouse deer site use was low and they seem to be avoiding humans, with SIF < 1 (Figure 2). But as TRI increased, so did mouse deer site use and overlap with humans (SIF > 1.0). Thus it appears mouse deer use a combination of temporal and spatial avoidance of both predators depending on where they are on the landscape.

We recognize that many other potential prey species exist in this landscape, including rodents, primates, birds, and lizards. We limited this study to those species that we had a sufficient number of captures, and to those that we thought might be targeted for hunting by humans. Expanding this study to include human impacts on other species is recommended. Even if humans are not targeting a species when hunting, non-target wildlife can be affected by light and noise pollution and caught as by-catch in traps and snares, which could negatively affect such wildlife populations.

Felid trap success was low in comparison to humans, and the apparent low impact of cats on prey detection and activity patterns may be due to low detections of cats in comparison with humans at all camera stations and throughout the day. This could bias our results in showing a larger apparent impact of humans on prey, versus the apparent impact of felids on prey. This could be due to the naturally low densities of felids, low density due to poaching, or our non-random camera placement near the protected area boundary. Of the five felid species included in this study, tigers are the most frequently studied in Sumatra. Clouded leopards and marbled cats are partially arboreal and not much is known about their hunting behavior or prey preferences in Sumatra. Golden cats and leopard cats are well known in other areas (Lynam et al. 2013, Vernes et al. 2015, McCarthy et al. 2015), but prey preferences and habitat use could differ in Riau due to high human modification in surrounding areas. Thus, our results on the impacts of native predators on prey could differ if we examined single species' impacts, or had a wider, randomly distributed camera array throughout the protected area, but we aimed to use our data as a starting point to provide new insights on the role humans may play in this at-risk landscape.

This is the first study of which we are aware, that uses an occupancy framework to study human use of a protected area. We recommend using such a framework in other areas threatened by high human presence or exploitation (particularly illegal) to enable quantification of human impacts on wildlife behaviors. By using humans as a parameter on detection, we not only found prey species altering space and/or time use in the presence of humans, but we can begin to assess such potential avoidance behaviors. In tying spatial data to temporal data, managers could better plan when and where to conduct activities so as to reduce impacts on wildlife. Furthermore, by pairing human site use data with wildlife site use data, managers can use limited resources more effectively to target poachers.

Although our study has small sample sizes and low trap success across all species (Table 1), we believe the results presented here provide valuable, new insights into tigers (or lack thereof), their prey, and the potential impacts of humans in this threatened, unique, and understudied Sumatran landscape. We recommend examining human, felid, and prey relationships more thoroughly throughout the entire protected area to better understand these dynamics. We believe our data provide evidence in support of the refuge effect for pigs and humans, and the risk disturbance hypothesis (Frid and Dill 2002) for mouse deer and humans.

However, more research is needed to fully investigate human-wildlife interactions in this landscape to determine whether humans have become an apex predator in this system. While focusing efforts on forest protection, poaching reduction is vitally important to the persistence of Sumatran tigers. It is also critically important to increase our knowledge of prey species in understudied ecosystems because tigers and other carnivore populations are driven by prey abundance (Karanth et al. 2004). Understanding how carnivore prey react to human presence as our human population continues to grow, in addition to habitat protection, could be the missing link to tiger long-term survival.

References

Akaike H. 1973. Information theory and an extension of the maximum likelihood principle. In Second international symposium on information theory: 267–281. Parzen, E., Tanabe, K. & Kitagawa, G. (Eds). New York: Akademinai Kiado.

Berger J, Stacey PB, Bellis L, Johnson MP. 2001. A mammalian predator/prey imbalance: grizzly bear and wolf extinction affect avian neotropical migrants. Ecological Applications, **11**: 967–980.

Brown CL, Hardy AR, Barber JR, Fristrup KM, Crooks KR, Angeloni LM. 2012. The effect of human activities and their associated noise on ungulate behavior. PLoS One, **7**: 38–40.

Dalerum F, Cameron EZ, Kunkel K, Somers MJ. 2009. Diversity and depletions in continental carnivore guilds. Implications for prioritizing global carnivore research. Biology Letters, **5**: 35-38.

Darimont CT, Fox CH, Bryan HM, Reimchen TE. 2015. The unique ecology of human predators. Science, **349**(6250): 858-860.

Dickson BG, Beier P. 2002. Home range and habitat selection by adult cougars in southern California. Journal of Wildlife Management, **66**: 1235–1245.

Farr TG, Rosen PA, Caro E, Crippen R, Duren R, Hensley S, Kobrick M, Paller M, Rodriguez E, Roth L, Seal D, Shaffer S, Shimada J, Umland J, Werner M, Oskin M, Burbank D, Alsdorf D. 2007. The shuttle radar topography mission. Reviews of Geophysics, **45**: RG2004.

Farris ZJ, Kelly MJ, Karpanty S, Ratelolahy F, Hall C, 2015. Patterns of spatial co-occurrence among native and exotic carnivores in north-eastern Madagascar. Animal Conservation, **19**:189–198.

Frid A, Dill L. 2002. Human-caused disturbance stimuli as a form of predation risk. Conservation Ecology, **6**(1): 11.

Hines JE (2006) PRESENCE – Software to estimate occupancy and related parameters. USGS-PWRC. http://www.mbr-pwrc.usgs.gov/ software/presence.html. Accessed

Karanth KU, Nichols JD. 1998. Estimation of tiger densities in India using photographic captures and recaptures. Ecology, **79**: 2852–2862.

Karanth KU, Nichols JD, Kumar NS, Link WA, Hines, JE. 2004. Tigers and their prey: Predicting carnivore densities from prey abundance. Proceedings of the National Academy of Sciences, **101**: 4854–4858.

Karanth KU, Sunquist ME. 1995. Prey selection by tiger, leopard and dhole in tropical forests. Journal of Animal Ecology, **64:** 439.

Kerley LL, Goodrich JM, Miquelle DG, Smirnov EN, Quigley HB, Hornocker MG. 2002. Effects of roads and human disturbance on Amur tigers. Conservation Biology, **16**: 97–108.

Linkie M, Ridout MS. 2011. Assessing tiger-prey interactions in Sumatran rainforests. Journal of Zoology, **284**: 224–229.

Lovallo MJ, Anderson EM. 1996. Bobcat movements and home ranges relative to roads in Wisconsin. Wildlife Society Bulletin, **24**: 71–76.

Lynam AJ, Jenks KE, Tantipisanuh N, Chutipong W, Ngoprasert D, Gale GA, Steinmetz

R, Sukmasuang R, Bhumpakphan N, Grassman Jr. LI et al. 2013. Terrestrial activity patterns of wild cats from camera trapping. The Raffles Bulletin of Zoology, **61**(1): 407-415.

Mace RD, Waller JS, Manley TL, Lyon LJ, Zuuring H. 1996. Relationship among grizzly bears, roads, and habitat in the Swan Mountains, Montana. Journal of Applied Ecology, **33**:1395-1404.

Margono BA, Potapov PV, Turubanova S, Stolle F, Hansen MC. 2014. Primary forest cover loss in Indonesia over 2000–2012. Nature Climate Change, **4**: 730–736.

McCarthy JL, Wibisono HT, McCarthy KP, Fuller TK, Andayani N. 2015. Assessing the distribution and habitat use of four felid species in Bukit Barisan Selatan National Park, Sumatra, Indonesia. Global Ecology and Conservation, **3:** 210–221.

Mech LD, Fritts SH, Radde GL, Paul WJ. 1988. Wolf distribution and road density in Minnesota. Wildlife Society Bulletin, **16**: 85–87.

Meredith M, Ridout M. 2016. Overlap: Estimates of Coefficient of Overlapping for Animal Activity Patterns. R Package Version 0.2.6. https://CRAN.R-project.org/ package=overlap.

Nellemann C, Henriksen R. Kreilhuber A, Stewart D, Kotsovou M, Raxter P, Mrema E, Barrat, S. (Eds). 2016. The Rise of Environmental Crime – A Growing Threat to Natural Resources Peace, Development and Security. A UNEP-INTERPOL Rapid Response Assessment. United Nations Environment Programme and RHIPTO Rapid Response–Norwegian Center for Global Analyses.

Ngoprasert D, Lynam AJ, Gale GA. 2017. Effects of temporary closure of a national park on leopard movement and behaviour in tropical Asia. Mammalian Biology, **82:** 65–73.

O’Brien TG, Kinnaird MF, Wibisono HT. 2003. Crouching tigers, hidden prey: Sumatran tiger and prey populations in a tropical forest landscape. Animal Conservation, **6:** 131–139.

Pakpien S, Simcharoen A, Duangchantrasiri S, Chimchome V, Pongpattannurak N, Smith JLD. 2017. Ecological Covariates at Kill Sites Influence Tiger (*Panthera tigris*) Hunting Success in Huai Kha Khaeng Wildlife Sanctuary, Thailand. Tropical Conservation Science, **10**:1-7.

Pimm SL, Jenkins CN, Abell R, Brooks TM, Gittleman JL, Joppa LN, Raven PH, Roberts CM, Sexton JO. 2014 The biodiversity of species and their rates of extinction, distribution, and protection. Science, **344**: 1246752.

Richmond OM, Hines JE, Beissinger SR. 2010. Two-species occupancy models: a new parameterization applied to co-occurrence of secretive rails. Ecol. Appl. **20**: 2036–2046.

Ridout MS, Linkie M. 2009. Estimating overlap of daily activity patterns from camera trap data. Journal of Agriculture, Biology and Environmental Statistics, **14**: 322–337.

Riley SJ, DeGloria SD, Elliot R. 1999. A terrain ruggedness index that quantifies topographic heterogeneity. Intermountain Journal of Science, **5**: 23-27.

Ripple WJ, Abernethy K, Betts MG, Chapron G, Dirzo R, Galetti M, Levi T, Lindsey PA, Macdonald DW, Machovina B, Newsome TM, Peres CA, Wallach AD, Wolf C, Young H. 2016. Bushmeat hunting and extinction risk to the world’s mammals. Royal Society Open Science, **3:** 1–16.

Sanderson EJ, Forrest J, Loucks C, Ginsberg J, Dinerstein E, Seidensticker J, Leimgruber P, Songer M, Heydlauff A, O'Brien T, Bryja G, Klenzendorf S, Wikramanayake E. 2006. Setting priorities for the conservation and recovery of wild tigers: 2005-2015. WCS, WWF, Smithsonian and NFWF-STF.

Shannon G, Cordes LS, Hardy AR, Angeloni LM, Crooks KR. 2014. Behavioral responses associated with a human- mediated predator shelter. PLoS One, **9**(4): e94630.

Sodhi NS, Lee TM, Koh LP, Brook BW. 2009. A meta-analysis of the impact of anthropogenic forest disturbance on Southeast Asia's biotas. Biotropica, **41:** 103-109.

Stankowich T. 2008. Ungulate flight responses to human disturbance : A review and meta-analysis. Biological Conservation, **141**(9): 2159 – 2173.

Sunarto, Kelly MJ, Klenzendorf S, Vaughan MR, Hutajulu MB, Parakkasi K. 2013. Threatened predator on the equator: multi-point abundance estimates of the tiger Panthera tigris in central Sumatra. Oryx, **47:** 211–220.

Sunarto S, Kelly MJ, Parakkasi K, Hutajulu MB. 2015. Cat coexistence in central Sumatra: ecological characteristics, spatial and temporal overlap, and implications for management. Journal of Zoology, **296**: 104 – 115.

Sunarto S, Kelly MJ, Parakkasi K, Klenzendorf S, Septayuda E, Kurniawan H. 2012. Tigers need cover: multi-scale occupancy study of the big cat in Sumatran forest and plantation landscapes. PLoS One, **7**: e30859.

Sunquist ME, Karanth KU, Sunquist F. 1999. Ecology, behaviour and resilience of the tigers and its conservation needs. In: Seidensticker, J., Christie, S., Jackson, P. (Eds.), Riding the Tiger: Tiger Conservation in Human-dominated Landscapes. Cambridge University Press, Cambridge, pp. 5–18.

Terborgh J, Estes JA. 2010. Trophic Cascades: Predators, Prey, and the Changing Dynamics of Nature. Island Press, Washington, D.C. UNODC, 2013. Chapter 7. The illegal wildlife trade in East Asia and Wildlife. In: Transnational Organized Crime in East Asia and the Pacific: A Threat Assessment, http://www.unodc.org/documents/southeastasiaandpacific//Publications/2013/TOCTA_EAP_web.pdf

van Dyke FG, Brocke RH, Shaw HG. 1986. Reactions of mountain lions to logging and human activity. Journal of Wildlife Management, **50:** 95–102.

Vernes K, Sangay T, Rajaratnam R, Singye R. 2015. Social interaction and co-occurrence of colour morphs of the Asiatic golden cat, Bhutan. Cat News, **62**: 18-20.

Waldstein A, Bland C, Forrester T, Baker-Whatton MC, Schuttler, SG, Mcshea WJ, Costello R, Kays R. 2016. The ecological impact of humans and dogs on wildlife in protected areas in eastern North America. Biological Conservation, **203**: 75–88.

Tables

Table 1. Detections and trap success of three focal prey species, wild felids, and humans, resulting from 33 camera trap stations active for 2,186 trap nights for five months in Riau, Sumatra, as well as other species of conservation interest. Detections are defined as number of individuals of each species captured within 30 minutes for all active trap nights and trap success is the number of capture events divided by trap nights, times 100.

Species	Total Detections	Trap Success	Humans Present/ No Felids	Felids Present/ No Humans	No Felids or Humans
Focal Species					
Wild pig (*Sus scrofa* or *Sus barbatus*)	555	25.39	261	28	237
Muntjac (*Montiacus montanus*)	160	7.32	45	29	58
Mouse deer (*Tragulus kancil* or *Tragulus napu*)	101	4.62	27	17	31
Sumatran tiger (*Panthera tigris sumatrae*)	4	0.18	-	-	-
Sunda clouded leopard (*Neofelis diardi*)	9	0.41	-	-	-
Asiatic golden cat (*Catopuma temminckii*)	12	0.55	-	-	-
Marbled cat (*Pardofelis marmorata*)	11	0.50	-	-	-
Leopard cat (*Prionailurus bengalensis*)	13	0.59	-	-	-
Human	364	16.65	-	-	-
Other Species of Interest					
Sambar (*Rusa unicolor*)	2	0.09	-	-	-
Sumatran dhole (*Cuon alpinus sumatrensis*)	2	0.09	-	-	-
Sunda pangolin (*Manis javanica*)	3	0.14	-	-	-
Malayan tapir (*Acrocodia indica*)	31	1.20	-	-	-
Sun bear (*Helarctos malayanus*)	68	3.11	-	-	-
Sumatran porcupine (*Hystrix sumatrae*)	141	6.45	-	-	-

Table 2. Single species occupancy (i.e. site use) model results for wild pigs (*Sus scrofa* and *Sus barbatus*), mouse deer (*Tragulus napu* and *Tragulus kanchil*), and humans in Rimbang Baling Wildlife Reserve, Riau, Sumatra. The combined cat occupancy models did not converge. Only top models (ΔAICc ≤ 2) are shown.

Pig						
Model	Psi (SE)	p (SE)	p(Cov1) (SE)	ΔAICc	AIC Wgt	No. Par
psi(TRI),p(TSHumans)	0.794 (0.092)	0.1295 (0.01057)	0.433 (0.077)	0	0.2396	4
psi(TRI+Dist_Disturbed),p(TSHumans)	0.791 (0.109)	0.1297 (0.0105)	0.432 (0.077)	0.38	0.1981	5
psi(TRI+ BigForest),p(TSHumans)	0.7928 (0.0920)	0.1296 (0.0106)	0.4326 (0.077)	1.34	0.1226	5
psi(TRI+%Forest2km),p(TSHumans)	0.793 (0.111)	0.1296 (0.0106)	0.433 (0.077)	1.38	0.1202	5
psi(.),p(TSHumans)	0.7912 (0.0715)	0.1296 (0.0105)	0.432 (0.077)	1.56	0.1098	3
Mouse Deer						
Model	Psi (SE)	p (SE)	p(Cov1) (SE)	ΔAICc	AIC Wgt	No. Par
psi(BigForest+TRI),p(TSHumans)	0.5699 (0.142)	0.0724 (0.0087)	-1.804 (1.047)	0	0.1087	5
psi(.),p(TSHumans)	0.5607 (0.0919)	0.0729 (0.0087)	-1.842 (1.047)	0.32	0.0926	3
psi(Dist_Roads),p(TSHumans)	0.559 (0.121)	0.0729 (0.0087)	-1.845 (1.048)	0.37	0.0903	4
psi(TRI),p(Humans)	0.567 (0.125)	0.0728 (0.0087)	-1.832 (1.047)	0.67	0.0929	4
psi(BigForest+TRI+Dist_Roads),p(TSHumans)	0.568 (0.164)	0.0724 (0.0087)	-1.81 (1.048)	1.21	0.0709	6
psi(BigForest),p(TSHumans)	0.557 (0.113)	0.0728 (0.0087)	-1.836 (1.048)	1.31	0.0675	4
psi(Dist_Roads+TRI),p(TSHumans)	0.565 (0.149)	0.0728 (0.0087)	-1.835 (1.047)	1.34	0.0665	5
psi(BigForest+Roads),p(TSHumans)	0.556 (0.142)	0.0728 (0.0087)	-1.838 (1.048)	1.55	0.0598	5
Human						
Model	Psi (SE)	p (SE)	p(Cov1) (SE)	ΔAICc	AIC Wgt	No. Par
psi(TSCats+Dist_Roads),p(.)	0.407 (0.133)	0.0759 (0.0085)	-	0	0.0789	4
psi(BigForest+Dist_Roads+Dist_Disturbed),p(.)	0.421 (0.151)	0.753 (0.0085)	-	0.07	0.0762	5
psi(Dist_Roads),p(.)	0.413 (0.117)	0.0756 (0.0085)	-	0.15	0.0732	3
psi(TSCats+Dist_Roads+Dist_Disturbed),p(.)	0.47 (.151)	0.0759 (0.0085)	-	0.22	0.0707	5
psi(Dist_Roads+Dist_Disturbed),p(.)	0.414 (0.138)	0.0756 (0.0085)	-	0.24	0.0700	4
psi(TSCats+Dist_Roads+BigForest),p(.)	0.408 (0.139)	0.0758 (0.0085)	-	0.41	0.0643	5

psi(.),p(.)	0.4158 (0.0892)	0.0756 (0.0085)	-	0.52	0.0609	2
psi(TSCats),p(.)	0.410 (0.11)	0.0758 (0.0085)	-	0.68	0.0562	3
psi(TSCats+BigForest),p(.)	0.412 (0.118)	0.0757 (0.0085)	-	0.85	0.0516	4
psi(TRI+Dist_Roads+Dist_Disturbed),p(.)	0.416 (0.160)	0.0756 (0.0085)	-	1.42	0.0388	5
psi(BigForest+Dist_Roads),p(.)	0.415 (0.135)	0.0755 (0.0085)	-	1.70	0.0337	4
psi(TRI+Dist_Roads),p(.)	0.413 (0.145)	0.0757 (0.0085)	-	1.75	0.0329	4
psi(Dist_Disturbed),p(.)	0.4167 (0.1174)	0.0756 (0.0085)	-	1.77	0.0326	3
psi(BigForest),p(.)	0.4180 (0.1085)	0.0755 (0.0085)	-	1.83	0.0316	3

Topographic ruggedness index (TRI), distance to forest > 500 km^2 (BigForest), distance to roads (Dist_Roads), distance to disturbed areas (Dist_Disturbed), trap success of felids (TSCats), trap success of humans (TSHumans) and percent of forest within 2 km of camera stations (%Forest2km).

Table 3. Two species occupancy (site use) model results for human-wild pig (*Sus scrofa* and *Sus barbatus*), felids (*Panthera tigris sumatrae*, *Neofelis diardi*, *Catopuma temminckii*, *Pardofelis marmorata* and *Prionailurus bengalensis*)-pig, human-mouse deer (*Tragulus napu* and *Tragulus kanchil*), and felids-mouse deer in Rimbang Baling Wildlife Reserve, Riau, Sumatra. Only top models (ΔAICc ≤ 2) are shown.

Humans-Pigs								
Model	PsiA (SE)	PsiBA (SE)	PsiBa (SE)	pA (SE)	pB (SE)	ΔAICc	AIC Wgt	SIF
psiA,psiBA(TRI)=psiBa(TRI),pA,pB(TSCats),rA,rBA(TSCats),rBa(TSCats)	0.4954 (0.0927)	0.8504 (0.0802)	0.8504 (0.0802)	0.0194 (0.0135)	0.09911 (0.01298)	0	0.1680	1
psiA(BigForest+TSCats),psiBA(TRI)=psiBa(TRI),pA,pB(TSCats),rA,rBA(TSCats), rBA(TSCats)	0.5099 (0.1257)	0.8503 (0.0804)	0.8503 (0.0804)	0.0158 (0.0111)	0.0992 (0.0131)	0.44	0.1348	1
psiA(BigForest),psiBA(TRI)=psiBa(TRI),pA,pB(TSCats),rA,rBA(TSCats), rBA(TSCats)	0.5010 (0.1120)	0.8496 (0.0811)	0.8496 (0.0811)	0.0183 (0.0129)	0.0991 (0.013)	0.67	0.1202	1
psiA(Road*TSCats),psiBA(TRI)=psiBa(TRI),pA,pB(TSCats),rA,rBA(TSCats), rBa(TSCats)	0.4938 (0.1147)	0.8512 (0.0795)	0.8512 (0.0795)	0.0192 (0.0134)	0.0992 (0.013)	0.70	0.1184	1
psi(TSCats),psiBA(TRI)=psiBa(TRI),pA,pB(TSCats),rA,rBA(TSCats),rBa(TSCats)	0.4982 (0.1162)	0.8510 (0.0979)	0.8510 (0.0797)	0.0181 (0.0128)	0.0992 (0.013)	1.12	0.0960	1
psiA(Dist_Road*Dist_Disturbed),psiBA(TRI)=psiBa(TRI),pA,pB(TSCats),rA, rBA(TSCats),rBa(TSCats)	0.4977 (0.1198)	0.8514 (0.0788)	0.8514 (0.0788)	0.0188 (0.0132)	0.0991 (0.013)	1.48	0.0802	1
Cats-Pigs								
Model	PsiA (SE)	PsiBA (SE)	PsiBa (SE)	pA (SE)	pB (SE)	ΔAICc	AIC Wgt	SIF
psiA(TRI),psiBA(TRI)=psiBa(TRI),pA,pB,rA,rBA(TSHumans)=rBa(TSHumans)	0.7367 (0.1017)	0.8410 (0.0820)	0.8410 (0.0820)	0.0307 (0.0096)	0.0284 (0.0083)	0	0.2106	1
psiA(TRI+TSHumans),psiBA(TRI)=psiBa(TRI),pA,pB,rA, rBA(TSHumans)=rBa(TSHumans)	0.7304 (0.1164)	0.8418 (0.0809)	0.8418 (0.0809)	0.0308 (0.0097)	0.0288 (0.0084)	0.01	0.2095	1
psiA(TRI),psiBA(TRI)=psiBa(TRI),pA,pB(TSHumans),rA, rBA(TSHumans)=rBa(TSHumans)	0.7110 (0.1056)	0.8424 (0.0801)	0.8424 (0.0801)	0.0308 (0.0097)	0.0613 (0.0099)	1.15	0.1185	1
Humans-Mouse Deer								
Model	PsiA (SE)	PsiBA (SE)	PsiBa (SE)	pA (SE)	pB (SE)	ΔAICc	AIC Wgt	SIF
psiA(TSCats*Dist_Road),psiBA(TRI),psiBa(TRI),pA,pB,rA,rBA(TSCats)=rBa(TSCats)	0.4145 (0.1145)	0.9325 (0.0775)	0.4323 (0.1701)	0.8636 (0.0732)	0.0638 (0.0109)	0	0.1055	1.4579
psiA(Dist_Roads),psiBA(TRI),psiBa(TRI),pA,pB,rA,rBA(TSCats)=rBa(TSCats)	0.4158 (0.1178)	0.9338 (0.0765)	0.4313 (0.1703)	0.8636 (0.0732)	0.0639 (0.0110)	0.78	0.0714	1.4585
psiA,psiBA(TRI),psiBa(TRI),pA,pB,rA,rBA(TSCats)=rBa(TSCats)	0.4175 (0.0898)	0.9337 (0.0771)	0.4323 (0.1707)	0.8636 (0.0732)	0.0637 (0.011)	0.87	0.0683	1.4552

psiA(Dist_Roads+TSCats),psiBA(TRI),psiBa(TRI),pA,pB,rA,rBA(TSCats)=rBa(TSCats)	0.4097 (0.1334)	0.9326 (0.0777)	0.4318 (0.1696)	0.8636 (0.0732)	0.0637 (0.0109)	1.01	0.0683	1.4641
psiA,psiBA,psiBa,pA,pB,rA,rBA,rBa	0.4203 (0.0905)	0.9279 (0.0696)	0.4330 (0.1236)	0.8636 (0.0732)	0.0618 (0.0119)	1.26	0.0562	1.4476
psiA(TSCats),psiBA(TRI)psiBa(TRI),pA,pB,rA,rBA(TSCats)=rBa(TSCats)	0.4127 (0.1109)	0.9327 (0.0780)	0.4315 (0.1702)	0.8636 (0.0732)	0.0637 (0.0110)	1.36	0.535	1.4610
psiA(TSCats),psiBA,psiBa,pA,pB,rA,rBA,rBa	0.4119 (0.1144)	0.9326 (0.0786)	0.4317 (0.1704)	0.8636 (0.0732)	0.0633 (0.0111)	1.66	0.0460	1.4617
psiA(Dist_Road*TSCats),psiBA(TRI),psiBa(TRI),pA,pB,rA,rBA,rBa	0.4119 (0.1144)	0.9326 (0.0786)	0.4317 (0.1704)	0.8636 (0.0732)	0.0633 (0.0111)	1.76	0.438	1.4616
psiA(Dist_Distrub+Dist_Road),psiBA,psiBa,pA,pB,rA,rBA,rBa	0.4113 (0.1386)	0.9263 (0.0710)	0.4401 (0.1220)	0.8636 (0.0732)	0.0626 (0.0111)	1.92	0.0404	1.4472
psiA,psiBA(TRI),psiBa,pA,pB,rA,rBA(TSCats)=rBa(TSCats)	0.4175 (0.898)	0.9337 (0.0771)	0.4323 (0.1707)	0.8636 (0.0732)	0.0637 (0.0110)	1.42	0.0617	1.4552

Cats-Mouse Deer

Model	PsiA (SE)	PsiBA (SE)	PsiBa (SE)	pA (SE)	pB (SE)	ΔAICc	AIC Wgt	SIF
psiA(Dist_River*Dist_Roads),psiBA(BigForest+TRI)=psiBa(DistBigForest+TRI), pA,pB,rA,rBA,rBa	0.7572 (0.1130)	0.5687 (0.1409)	0.5687 (0.1409)	0.0288 (0.0069)	0.0164 (0.0073)	0	0.0874	1
psiA(Dist_River*Dist_Roads),psiBA(BigForest+TRI)=psiBa(DistBigForest+TRI), pA, pB(TSHumans),rA,rBA,rBa	0.7456 (0.1131)	0.5864 (0.1411)	0.5864 (0.1411)	0.0295 (0.0070)	0.39 (0.006)	0.69	0.0619	1
psiA(Dist_River*Dist_Roads),psiBA=psiBa,pA,pB,rA,rBA,rBa	0.7661 (0.1145)	0.5474 (0.0967)	0.5474 (0.0967)	0.0283 (0.0068)	0.0176 (0.0079)	0.71	0.0613	1
psi(Dist_River),psiBA=psiBa,pA,pB,rA,rBA,rBa	0.7448 (0.1127)	0.5530 (0.0971)	0.5530 (0.0971)	0.0290 (0.0069)	0.0172 (0.0078)	1.15	0.0492	1
psiA(Dist_River*Dist_Road),psiBA=psiBa,pA,pB(TSHumans),rA,rBA,rBa	0.7543 (0.1152)	0.5653 (0.0993)	0.5653 (0.0993)	0.0291 (0.0070)	0.0401 (0.0073)	1.65	0.0383	1
psiA(Dist_River*Dist_Road),psiBA(Dist_Roads)=psiBa(Dist_Roads),pA,pB,rA,rBA,rBa	0.7641 (0.1135)	0.5497 (0.1284)	0.5497 (0.1284)	0.0284 (0.0068)	0.0173 (0.0078)	1.72	0.0370	1
psiA(Dist_River),psiBA(BigForest)=psiBa(BigForest),pA,pB,rA,rBA,rBa	0.7434 (0.1124)	0.5984 (0.1023)	0.5984 (0.1023)	0.0293 (0.0068)	0.0172 (0.0077)	1.81	0.0354	1
psiA(Dist_River),psiBA=psiBa,pA,pB(TSHumans),rA,rBA,rBa	0.7359 (0.1119)	0.5726 (0.0992)	0.5726 (0.0992)	0.0297 (0.0070)	0.0395 (0.0070)	1.88	0.0342	1

Topographic ruggedness index (TRI), distance to forest > 500 km2 (BigForest), distance to roads (Dist_Roads), distance to disturbed areas (Dist_Disturbed), trap success of felids (TSCats), trap success of humans (TSHumans) and percent of forest within 2 km of camera stations (%Forest2km).

Figures

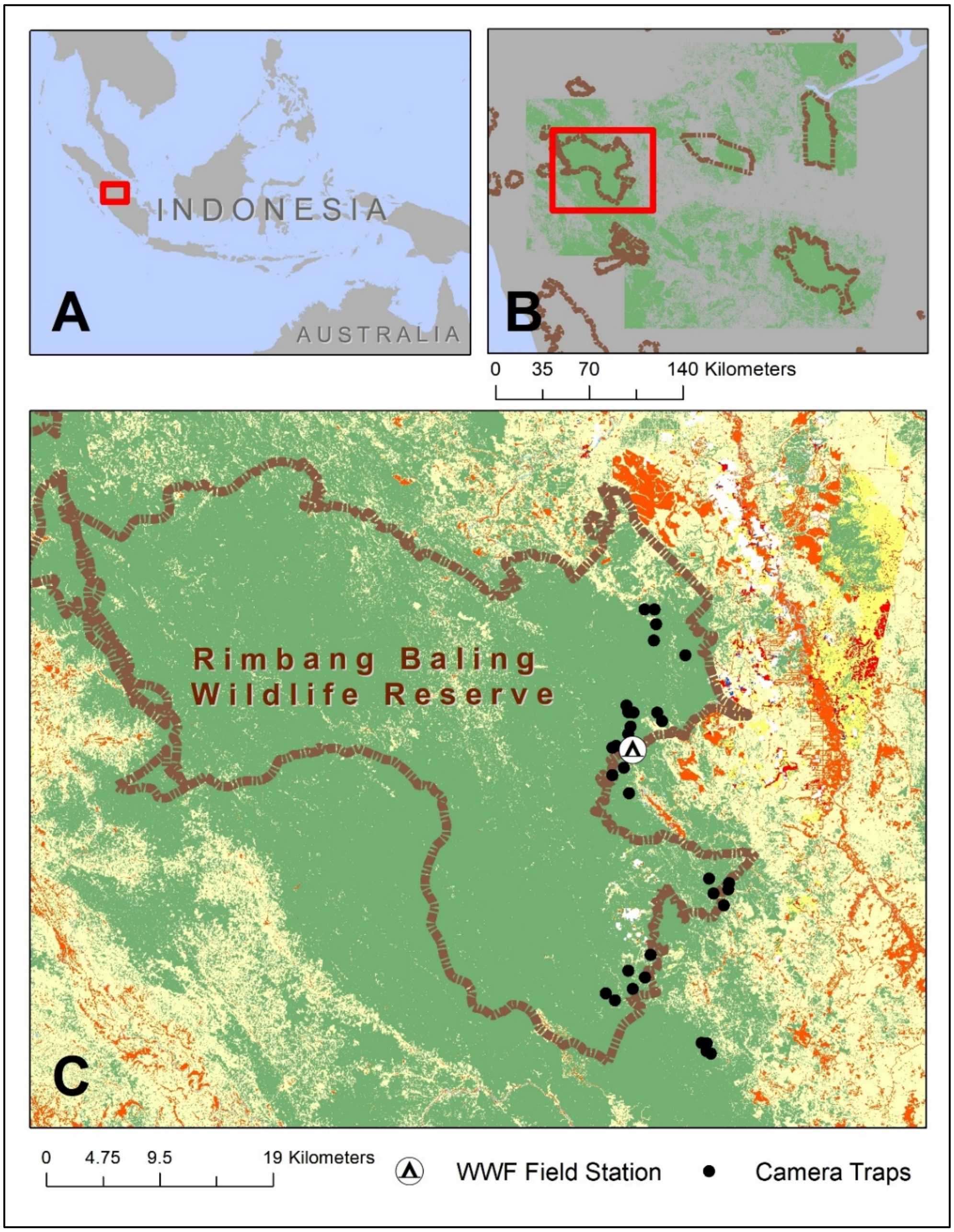

Figure 1. Location of Indonesia, (A) and Rimbang Baling Wildlife Reserve, Riau, Sumatra (B), where 33 camera traps were placed for a study lasting five months (C). Land cover is mainly forest (green) within the protected area, but Rimbang Baling is surrounded by plantation (yellow), and open areas (A; red). Rimbang Baling and its surrounds contain much of the forest remaining (B; green) in central Sumatra and is critical habitat for Sumatran tigers (*P. t. sumatrae*).

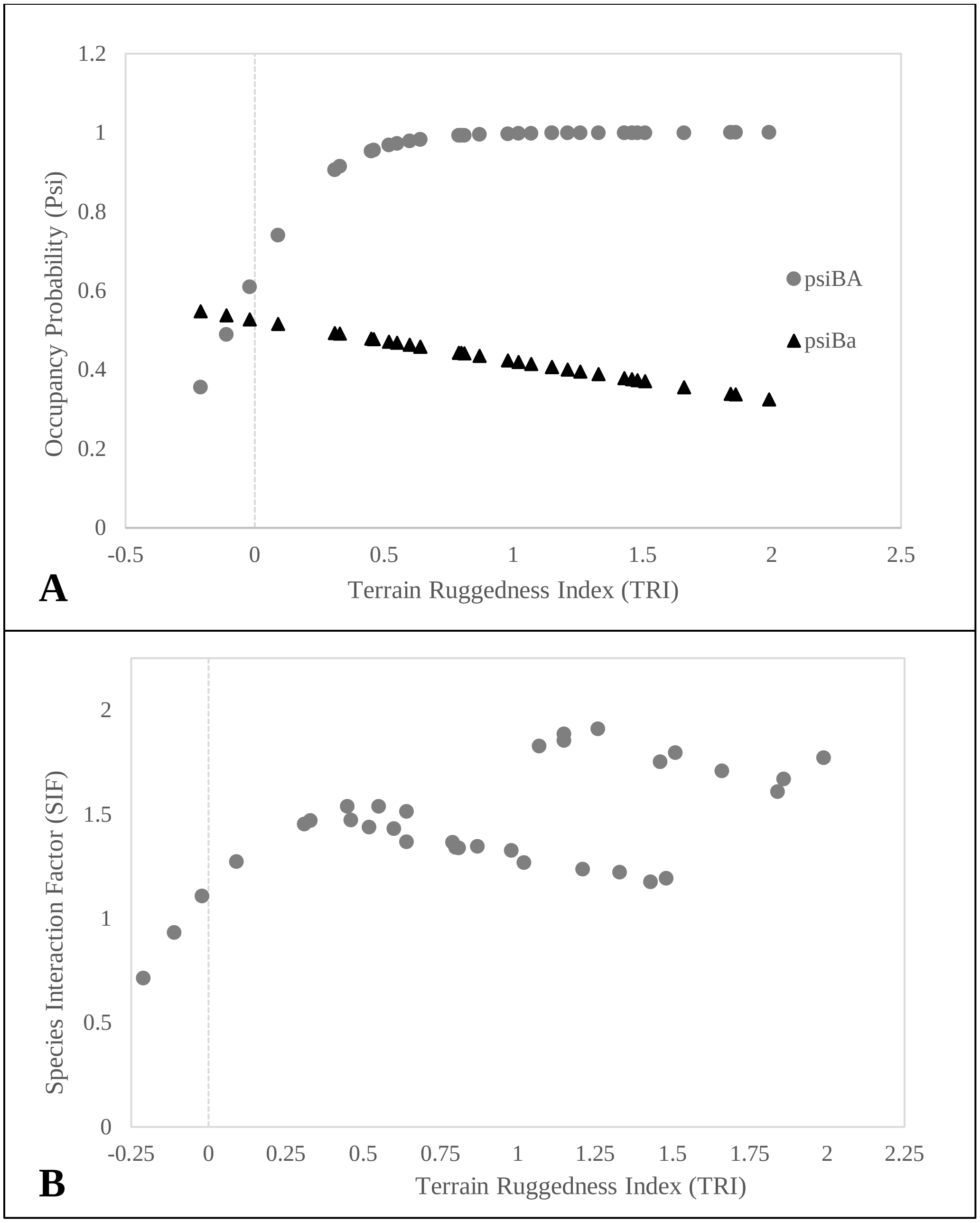

Figure 2. Mouse deer occupancy probability in the presence of (psiBA) and in the absence of (psiBa) humans (A) and the species interaction factor (SIF = 1 ≈ independence) for the relationship between mouse deer and human (B). Terrain ruggedness index (larger values = more rugged terrain) mediates this relationship.

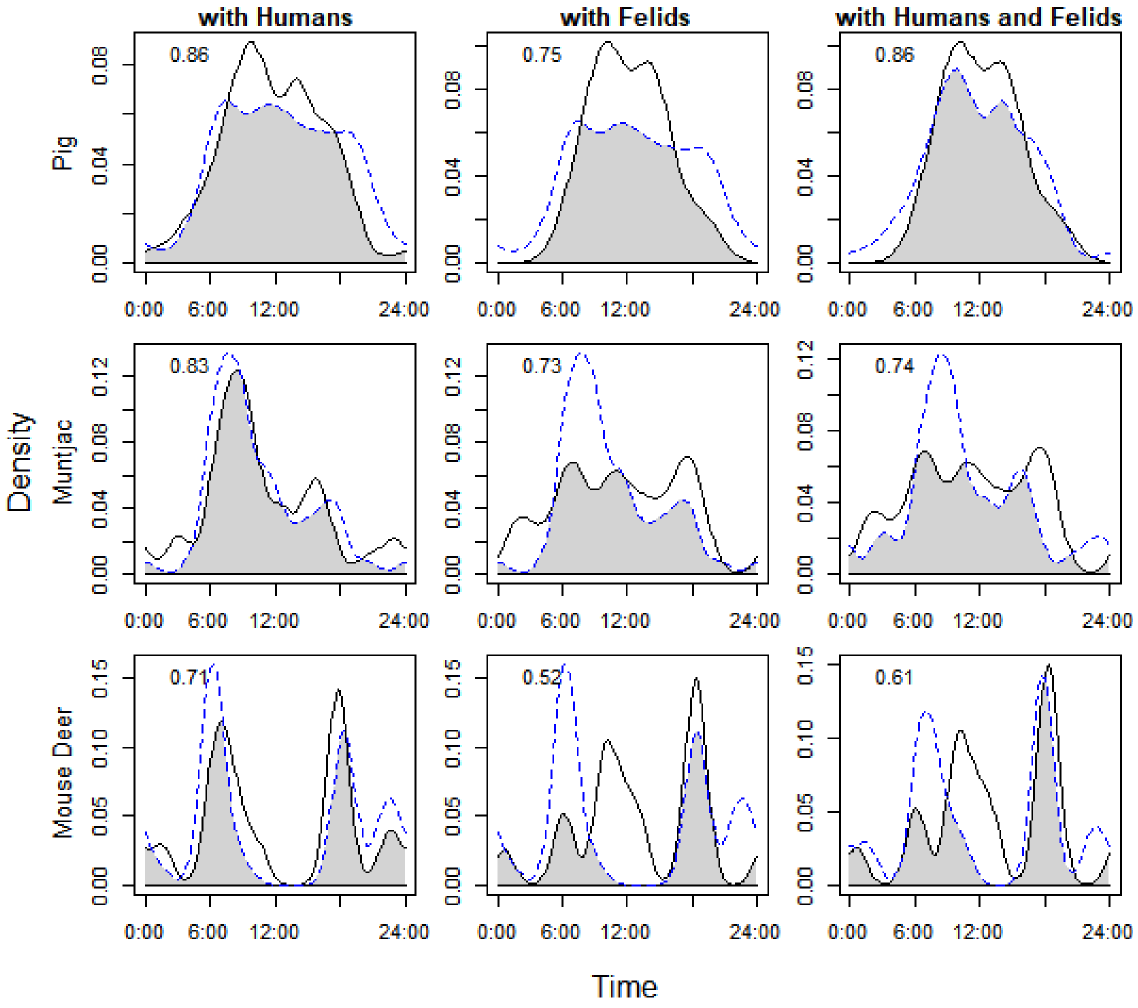

Figure 3. Activity patterns of pigs (*Sus* spp., top row), muntjacs (*Muntiacus muntjak;* middle row) and mouse deer *(Tragulus* spp.; bottom row) in the presence (solid lines) and absence (dashed lines) of humans (column 1) and cats (column 2) (*P.t. sumatra*, *Neofelis diardi*, *Catopuma temminckii*, *Pardofelis marmorata* and *Prionailurus bengalensis*), using kernel density estimator, $\Delta_{hat}1$, report in the top left corners of each graph. Activity patterns in the presence of cats (solid lines) vs. presence humans (dashed lines) are also compared in the right-most column to determine the level of similarity (i.e. overlap) in temporal response to predators.

Supplementary Material

Table S1. List of covariates tested for inclusion in occupancy models. Covariates were tested singularly and in combination. Those that did not appear in top (ΔAICc ≤ 2) single-species occupancy models were not included in co-occurrence models. Starred variables were highly correlated with other variables and not tested in occupancy modeling.

Variable	**Abbreviation**
Average topographic position index within 2 km radius of camera	TRI
Distance to cities*	Dist_City
Distance to disturbed land cover	Dist_Disturbed
Distance to patches of forest > 500 km2	BigForest
Distance to plantation*	Dist_Plant
Distance to river	Dist_River
Distance to roads	Dist_Roads
Felid trap success	TSCats
Human trap success	TSHumans
Percent disturbed areas within 2 km radius of camera*	%Disturbed
Percent of forest with 2 km radius of camera	%Forest2km
Number of forest patches within 2 km radius of camera*	NoPatches
Topographic position index at camera*	TRI_Camera

Chapter 6

Conclusions

It is apparent from this research, and much of the research cited herein, that Sumatran wildlife and natural forest is seriously, negatively impacted by humans. In my study area, in central Sumatra, which is known as a tiger stronghold and is important as a cross-island corridor area for wildlife, >35% of natural forest has been lost since 2002 (Chapter 2), and I predict that with a business-as-usual scenario, an additional 58.19% of forest existing in 2016 could be lost by 2050. Areas that are most susceptible to this loss are those that have low elevation, low slope, and soil conditions suitable for oil palm or acacia plantations, most notably Tesso Nilo National Park and Kerumutan Wildlife Reserve. Currently, all protected areas within our study area are suffering from deforestation and encroachment (Chapter 3).

Tesso Nilo National Park lies on land suitable for oil palm and is experiencing extreme deforestation, with nearly 2500 km of roads within Tesso Nilo (Chapter 4). If wildlife cannot use forest within 1 km of a road due to light and noise pollution, there is very little useable forest currently existing in Tesso Nilo, despite claims that Tesso Nilo is a stronghold for lowland rainforest wildlife. As it currently exists, Tesso Nilo can be considered a ‘paper park’.

In Rimbang Baling, a protected area that is supposedly better protected due to its rugged terrain, our results also show potential impacts on wildlife due to humans (Chapter 5). I recorded only 4 tiger detections and very few instances of preferred tiger prey in camera traps near the protected area boundary. Additionally, mouse deer detection rates fell with the presence of humans, and I recorded humans on camera with mouse deer carcasses and other wildlife on multiple occasions. However, I do note that this study focused on the edge of the park where detections were low across all species, and suggest future research move further towards the interior of the park to determine if detections increase farther inside.

Despite the destruction within Riau, there is still a possibility for the resurgence of wildlife and tigers. Tiger populations can likely recover if provided habitat and ample prey to provide energetics for breeding (Sunquist et al. 1999). Poaching in Sumatra is increasing (Risdianto et al. 2016, Ripple et al. 2016), and a small reduction in the tiger population through poaching can have significant impacts in small populations, so time is of the essence for saving the Sumatran tiger in Riau Province.

There are multiple actions that can help the Riau tiger population. First, I recommend securing Rimbang Baling Wildlife Reserve and Bukit Tigapuluh, including increasing efforts to reduce poaching and encroachment in these protected areas. My predictions show that by 2050, unless we reverse the current trends, there will be little forest remaining in these areas. These areas are still somewhat connected to tiger populations along the western edge of Sumatra and could provide habitat for migrants from the Bukit Barisan Mountains. Secondly, maintaining natural forest in Tesso Nilo is critical, and once Rimbang Baling and Bukit Tigapuluh are secure, authorities could focus on restoring Tesso Nilo, which would undoubtedly require substantially more resources. Restoring Tesso Nilo and Kerumutan Wildlife Reserve to their natural states could help localized and isolated populations of other wildlife like Asian elephants, but due to their current fragmented state, the healthy ecosystems such as in Rimbang Baling and Bukit Tigapuluh may be more immediately critical for the persistence of tigers. The government of Indonesia currently has a plan to relocate people living within Tesso Nilo, but it is yet unclear whether this plan will be effective.

In tandem with restoring Tesso Nilo, corridors such as those proposed in the Riau Conservation Vision could enable wildlife to migrate from Bukit Tigapuluh and Rimbang Baling. Unfortunately, I was unable to assess tiger use of potential corridor areas in this study, but others (Yaap et al. 2016) suggest that wildlife can and do use small forest fragments. Thus, managers should not be reluctant to set land aside for corridors, if large areas cannot be obtained – even small patches of forest can be used as stepping stones or refugia for smaller wildlife species. Due to our lack of wildlife movement data, managers should focus on obtaining and converting areas that are financially and socially appropriate in addition to those areas that are still forested outside of protected areas (Chapter 2, Figure 4).

Because we are unlikely to immediately stop deforestation, degradation, and loss of wildlife, research efforts should focus on less-studied species such as marbled cats and golden cats in this landscape. Riau is extremely biodiverse and many species of flora and fauna may still be unrecorded. Studying lesser known species – from primates to birds to the plethora of small carnivores – can not only aid in understanding tropical systems better, but also better plan restoration efforts so that limited conservation funding can be wisely spent. This area is one of the least studied in the world (Ripple et al. 2016), in part due to permitting and/or logistic difficulties, and losing the diversity extant in Riau today would be a global tragedy.

Finally, and most importantly, local citizens must be educated about the breadth of diversity of global importance that exists within several hours' drive of their homes. While many locals may not be able to aid in conservation due to the poverty that exists in Riau, and throughout Indonesia, some may be able to aid in or take part in conservation efforts and are unaware of the state of Indonesia's wildlife. Many locals are afraid of tigers or of being in the forest, and this perpetuates the idea that forest is 'bad'. I am unaware of any environmental education programs for grade-school children, but when I presented at local schools and shared my camera trap data, I was met by cheers and excitement. Of course not all Indonesians are nature-averse and there is a growing number of diverse, college-age students who form outdoor and nature clubs. Increasing the number young people who have knowledge of Riau's wildlife cannot be overstated in the importance in saving Indonesia's wildlife in the long run. Wildlife education programs for school-age children are needed to inspire future generations to want to protect biodiversity.

Addressing the social and economic issues tied to tiger conservation and the oil palm industry in Indonesia is unfortunately beyond the scope of this research, and such issues are complex vast, and global. Given the current state of tiger habitat as determined in this dissertation, it is unlikely that Indonesia will reach the 2022 goal of doubling their wild tiger population. There is an incredible amount of work needed to ensure tigers can even persist in Riau, but it is not impossible to consider growing the tiger population. With increased funding and participation from international NGOs and local Indonesian government agencies, hard work from passionate field teams, education programs reaching younger generations of men and women, restoration of habitat, and effective, well-placed land protection efforts including connections from the western mountains, tigers can be conserved, and possibly increased, in Riau. However, if efforts do not begin soon as possible to prevent further deforestation and degradation, it is likely we will lose tigers and much more biodiversity in central Sumatra.

References

Ripple WJ, Abernethy K, Betts MG, Chapron G, Dirzo R, Galetti M, Levi T, Lindsey PA, Macdonald DW, Machovina B, Newsome TM, Peres CA, Wallach AD, Wolf C, Young H. 2016. Bushmeat hunting and extinction risk to the world's mammals. Royal Society Open Science, **3**: 1–16.

Risdianto D, Martyr DJ, Nugraha RT, Harihar A, Wibisono HT, Haidir A, Macdonald DW, D'Cruze N, Linkie M. 2016. Examining the shifting patterns of poaching from a long-term law enforcement intervention in Sumatra. Biological Conservation, **204(**B): 306-312.

Sunquist ME, Karanth KU, Sunquist F. 1999. Ecology, behaviour and resilience of the tigers and its conservation needs. In: Seidensticker, J., Christie, S., Jackson, P. (Eds.), Riding the Tiger: Tiger Conservation in Human-dominated Landscapes. Cambridge University Press, Cambridge, pp. 5–18.

Yaap B, Magrach A, Clements GR, Mcclure CJW, Paoli GD, Laurance WF. 2016. Large mammal use of linear remnant forests in an industrial pulpwood plantation in Sumatra, Indonesia. Tropical Conservation Science, October-December: 1-13.