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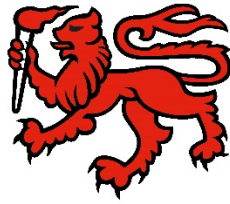
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Protected area planning for land-use change: Optimising outcomes for biodiversity

by

Carley Nichole Fuller

BSc (Hons)

University of Tasmania | College of Sciences and Engineering

Submitted in fulfilment of the requirements for the degree of Doctor of Philosophy

University of Tasmania

May 2022

Declaration of originality

I declare that this is my own work and has not been submitted in any form for another degree or diploma at any university or other institution of tertiary education. Information derived from the published or unpublished work of others has been duly acknowledged in the text and a list of references is given.

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Date: 22 December 2021

Publications associated with this thesis

Fuller, C., Ondei, S., Brook, B. W., & Buettel, J. C. (2019). First, do no harm: A systematic review of deforestation spillovers from protected areas. *Global Ecology and Conservation*, 18, e00591.

Fuller, C., Ondei, S., Brook, B. W., & Buettel, J. C. (2020). Protected-area planning in the Brazilian Amazon should prioritize additionality and permanence, not leakage mitigation. *Biological Conservation*, 248, 108673.

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The candidate was the primary author of all chapters and was responsible for the project conception, design, data sourcing, data analysis, interpretation, writing, and manuscript preparation. Members of the supervisory team contributed and guided at all stages.

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PAPER 2: Located in Chapter 3

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Statement of ethical conduct

The research associated with this thesis complies with the current laws of Australia, and all permits necessary for the project were obtained. Specifically, the University of Tasmania's Animal Ethics Committee approved mammal and ground bird monitoring by camera traps.

Ethics Approval No: A0017229

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Table of Contents

Declaration of originality	ii
Statement of authority of access.....	ii
Statement regarding published work	ii
Publications associated with this thesis.....	iii
Statement of co-authorship.....	iv
Statement of ethical conduct	vi
Acknowledgements.....	vii
List of Tables	x
List of Figures	xii
List of Appendices	xvii
Thesis Abstract.....	xviii
Chapter 1: General Introduction.....	20
I. Conservation science in context	21
II. Threats to effective protected areas	23
III. Thesis aims and case studies.....	26
IV. Thesis objectives	28
V. Thesis structure	28
Chapter 2: Assessing the prevalence and intensity of leakage from protected areas globally	32
Abstract.....	34
Introduction	35
Methods.....	37
Results.....	44
Discussion.....	51
Conclusion.....	54
Chapter 3: Assessing the magnitude of leakage from recently established protected areas in the Brazilian Amazon.....	55
Abstract.....	57
Introduction	58
Methods.....	60
Results.....	69
Discussion.....	73
Chapter 4: Thinking globally and acting locally to achieve protected area targets.....	77

Abstract.....	79
Introduction	80
Methods.....	82
Results.....	92
Discussion.....	100
Chapter 5: Integration of targeted, fine-scale biodiversity data to support broad-scale conservation decision making	105
Abstract.....	107
Introduction	108
Methods.....	110
Results.....	121
Discussion.....	133
Chapter 6: General Discussion	137
I. Aims.....	137
II. Summary of key findings.....	137
III. Quantitative methods within socioecological systems.....	139
IV. Research approaches and limitations.....	139
V. Opportunities for future research and for policy application	141
VI. Concluding remarks	142
References	144
Appendix A: Chapter 2 Supplementary materials.....	162
Appendix B: Chapter 3 Supplementary materials.....	167
Appendix C: Chapter 4 Supplementary materials.....	182
Appendix D: Chapter 5 Supplementary materials	255

List of Tables

Table 2.1 Search terms for retrieval of studies. The Boolean operator ‘OR’ linked terms within categories while ‘AND’ linked terms between categories; asterisks indicate wildcards and return any use of the root word; quotation marks prevent the search from splitting a phrase into its word components.....	37
Table 2.2 National scale meta-data factors, their sources, and rationales for their inclusion.....	42-43
Table 2.3 Predictors included in the best-selected generalised linear models of leakage and blockage magnitudes. Colour-shaded fields and the + and – symbols indicate the variables included in each model and direction of their effect.....	50
Table 3.1 Spillover calculated for 91 PAs.....	71
Table 4.1 Eight features calculated for each planning unit and used to rank their relative conservation priority. The simple multi-objective strategy and the jurisdictionally informed strategy were therefore distinguished only by the relative-importance weightings assigned to each feature. For example, the area of under-protected native grasslands within each planning unit was assigned 0.0833 and 0.0238, respectively (see also Appendix C, S13 and S14). Weights sum to 1.....	89
Table 4.2 Priority planning units selected by two strategies, constrained by three area budgets. At the time of this research, the protection proposed by the Tasmanian state government, upon which the BAU area budget is based, sought to add 25,209 ha (6.4% of the FPPF lands) to the PA network as part of the Tasmanian Wilderness World Heritage Area (DPIPWE, 2016). The ambitious area budget was set equivalent to the total area of planning units required to protect 30% (or, if not attainable, to maximise protection) of each of three ecoregions and nine bioregions (234,705 ha, or 59.7% of the FPPF zone). The conservative area budget was set equivalent to the total area of all planning units containing no suitable areas for <i>Eucalyptus nitens</i> , <i>Eucalyptus globulus</i> , or <i>Pinus radiata</i> plantations (10,277 ha, 2.6% of the FPPF zone). The percent values refer to the proportion of the total FPPF tenure’s number of planning units, area, reserve-adjacent boundaries, and area suitable for timber plantations is captured within the protection plan suggested by each strategy under each area budget. The greater connectivity and opportunity cost values are in bold.....	97

Table 5.1 Areas and proportions of ecoregions and bioregions of Tasmania that are protected and protectable. Within the state of Tasmania, there are three terrestrial ecoregions, as defined by the World Wildlife Fund and the Nature Conservancy in 2011, and nine IBRA units/bioregions, as defined by the Australian government in 2012. Red text flags where proportion protected is less than 30%; note that some planning units fall within multiple bioregions. These calculations are all based on spatial data from Olson et al. (2013), the Department of Sustainability, Environment, Water, Population and Communities (2013), and “LIST Future Potential Production Forest” from <http://www.thelist.tas.gov.au/> ©State of Tasmania.....112

Table 5.2 Pairwise comparison of the top-priority selected areas for each of the single-feature reserve design criteria rankings. Shown is the total area the compared rankings overlap (the maximum being 25,209 ha according to the business-as-usual area budget), the number of planning units selected by both scenarios, and the percent of total area that overlapped when the single-feature selections were compared. The pairwise comparisons are listed from highest (top) to lowest (bottom) percent overlap (see Appendix D, item S7 for the analogous table for the cadastral planning units).....123

Table 5.3 Percentage overlap of the top-ranked 25,209-ha of planning units for community-diversity metrics with those for optimising connectivity, representation, land-use threat mitigation, and cost efficiency, as well as for combinations of connectivity-plus-representation and threat mitigation-plus-cost efficiency. Each scenario is also compared with a scenario where the three community diversity metrics were combined.....128

Table 5.4. Pairwise comparison of scenarios’ top-priority selections; camera-informed scenarios are in bold text, and rows are sequenced from highest to lowest percent overlap.....132

List of Figures

- Figure 1.1** Conceptual map of how the thesis chapters relate and contribute to the overarching aim of spatially optimising land-use change by strategically establishing protected area networks where they can make the most difference for biodiversity. Where a protected area’s impact is defined as the difference made to the rates and locations of land-use changes as compared to the likely outcome without gazettelement, there are several interacting pitfalls to avoid. This thesis presents quantitative triage of these threats, jurisdictionally informed decision support tools and frameworks, and use of modern biodiversity monitoring technologies as key solutions.....29
- Figure 2.1** Screening protocol and numbers of papers included at each step in the systematic review process. Diagram modified from Moher et al. (2009).....39
- Figure 2.2** Map of a hypothetical landscape in which spillover from a PA to the unprotected adjacent surroundings (UAS) can be calculated. The sample of the broader landscape used to derive the baseline deforestation rate could be obtained a) as proposed by Ewers and Rodrigues (2008), or b) as provided by Spracklen et al. (2015).....40
- Figure 2.3** Map of the countries where deforestation rates within and around PAs, with which spillover could be calculated, were reported in the refereed literature. The shading indicates the number of PAs per country.....45
- Figure 2.4** Flowchart of the stepwise results of assessing PA impact on deforestation rates, indicating the number of PAs that were extracted from the evidence search, that passed data cleaning, that had positive versus negative impact within the target area, that had negative versus positive impact in nontarget area, and that were assigned each spillover outcome type (leakage, negligible spillover, or blockage). Blockage outcomes outnumbered leakage outcomes.....46
- Figure 2.5** Proportional distribution (and number per category) of spillover outcomes across PA-specific factors and continents. Results were broadly similar across: a) IUCN categories, b) land-use restriction types, whereby land use restriction is strict in PAs of IUCN categories I-IV and some degree of land use and resource extraction is allowed in PAs of IUCN categories V and VI, c) PA size quartiles, d) PA age quartiles, and e) continents. Each bar graph does not necessarily show the results for all 2,575 PAs (grand total assessed) because, for example, IUCN category was reported or

found for only 1,746 of the PAs; PA size was reported for only 2,545 PAs; the year of establishment was reported or found for only 1,816 PAs.....48

Figure 3.1 The inset map (a) shows the position of the Amazônia Legal (grey) within Brazil and the location of all the protected conservation units (blue) and indigenous lands (brown) established in the region. Indigenous lands were not assessed for impacts on deforestation, but were accounted for (see Appendix B, Table S1) where they overlapped with the spillover zones of the conservation unit PAs assessed. The main map (b) of the study region, Amazônia Legal, shows the 91 PAs that were established within it from 2005 to 2016 (blue), their adjacent zones tested for possible spillover (yellow), and areas of new deforestation reported for each year 2005 to 2017 (red). Government of the region is administered through nine Brazilian states labelled in the main map (b).....61

Figure 3.2 Maps of unprotected areas sampled to estimate two baseline deforestation rates for the Extractive Reserve Ariaca Pruano. A baseline is an estimate of the area of new deforestation that might have been expected in the PA and in its adjacent unprotected surroundings (yellow) had no protection ever been declared in the area. Deforestation (red) was reported during the period 2005 to 2016. The main map (a) shows the location of the Extractive Reserve Ariaca Pruano (purple) and the state of Pará (heavy outline) within Amazônia Legal. The left-hand inset map (b) was adapted from the non-target impact assessment method proposed by Ewers and Rodrigues (2008): five randomly generated circular samples (dark grey) of the nearby, unprotected landscape (light grey) were used to estimate expected deforestation for each year in the PA and in the spillover zone. The combined area of the five samples is equal to the area of the PA and thus also equal to the area of the spillover zone. The right-hand inset map (c) shows a novel spatial sample from which an alternative baseline is derived from the entire unprotected portion of the state within which the PA is located, which in this case is Pará.....64

Figure 3.3 Sequence of methods used to acquire relevant data, to assess the target (blue) and non-target (yellow) impacts of PA establishment, to classify PAs as leakage generating (orange) or blockage generating (green), and finally to analyse the results of the non-target impact assessments. The impact assessment, outcome classification, and statistical analysis steps were performed for two data sets, one generated using locally derived baselines (73 spillover results) and the other generated using the state-derived baselines (91 spillover results).....67

Figure 3.4 Maps of colour-coded target and non-target (i.e., spillover) impacts identified after the establishment of 91 PAs in the Amazônia Legal since 2005, showing the impact of protection within

the PAs as calculated (a) with locally derived baselines, and (b) with state-derived baselines. The impact of protection in the local spillover zones was also calculated (c) with locally derived baselines, and (d) with state-derived baselines. Spillover results are not shown for those PAs that experienced more deforestation within their boundaries than expected, as these PAs were unlikely to have influenced their surroundings if protection did not influence their target areas.....69

Figure 3.5 Plot of mean annual spillover value per PA age class, using both (a) locally derived baselines and (b) state-derived baselines, with the net outcome of either leakage (n = 9 and n = 11, respectively) in orange or blockage (n = 34 and 79, respectively) plotted separately. The error bars show the standard error of each mean. Note the difference in the y-axes' scales between plots (a) and (b).....72

Figure 4.1 Percentage of land protected for Tasmania's (a) three ecoregions and (b) nine bioregions. The temperate forest (TF) ecoregion and the Northern Midlands (TNM), the South East (TSE), Ben Lomond (BEL), the Northern Slopes (TNS), and King (KIN) bioregions all currently have less than 30% of their landscapes protected. Spatial data was obtained from Olson et al. (2001) and from the Department of the Environment (2012).....84

Figure 4.2 Tasmania's protected, timber production, and future potential production zones, including the area proposed for protection (orange outline), located adjacent the Tasmanian Wilderness World Heritage Area. The Tasmanian Forest Agreement Act of 2013 declared 392,969 ha to be "Future Reserve Lands;" in 2014, these areas were declared "Future Potential Production Forests" (yellow) upon the passage of the Forestry (Rebuilding the Forest Industry) Act, which aimed to secure a 'wood-bank' to provide resources for future production forestry. On 8 April 2020, these lands were opened for potential reallocation to either the reserve estate or to the Permanent Timber Production Zone (Future Potential Production Forest Fact Sheet, 2020). Grey areas are outside of the public forest domain. Spatial data was obtained from the "LIST Land Tenure" from <http://www.thelist.tas.gov.au/> ©State of Tasmania.....85

Figure 4.3 Four single-feature prioritisations. The (a) representation scenario selects all planning units required to protect 30% (or the maximum proportion where only <30% is possible) of each biogeographic unit; 92.44% (23,302 ha) of the area selected is congruent with the existing protection proposal (orange outline). The (b) connectivity scenario selects all planning units for which 50% or greater of the perimeter is adjacent existing protected areas; 88.30% (22,259 ha) of the area selected is congruent with the existing protection proposal (orange outline). The (c) zero-cost scenario selects all planning units that contain no area suitable for timber plantations. The (d) threat- mitigation scenario selects all planning units for which at least 30% of the 1 km buffer zone was modified land, i.e., not intact native vegetation. None of the areas selected by scenarios (c) and (d) were congruent with the government’s plan. 93

Figure 4.4 Two compromise prioritisations output by the ranking tool. Although the input features were identical, the differential importance weightings assigned by the simple multi-objective strategy (a) versus the jurisdictionally informed strategy (b) resulted in notable differences in the output rankings. 95

Figure 4.5 Conservation recommendations of the two multi-objective strategies, constrained by (a) conservative, (b) business-as-usual (BAU), and (c) ambitious area budgets. When constrained by an area budget (b), i.e., when limited to a maximum size equivalent to that of the BAU plan, neither compromise strategy recommended any of the current FPPF area that the Tasmanian state government has proposed for conversion to reserve tenure. The inset map under panel (c) shows that the areas the State has proposed for protection north of the TWWHA (which account for 24,172 ha or 96% of the total Government proposal) are recommended by the two multi-objective prioritisation strategies only at this most ambitious budget and alongside other priority areas, indicating that other (i.e., non-TWWHA adjacent) areas are required for the Tasmanian protected area system to meet multiple objectives.....100

Figure 5.1. A total of 201 camera traps were deployed within the Future Potential Production Forest tenure of the Tasmanian 25%-protected Temperate Forest ecoregion, with 119 in the 20%-protected Ben Lomond bioregion, 62 in the 19%-protected South East bioregion, and 20 cameras in the 31%-protected Furneaux bioregion. Spatial data was obtained from the “LIST Land Tenure” from <http://www.thelist.tas.gov.au/> ©State of Tasmania. Camera traps were placed in 58 subcatchments of FPPF tenure (equating to 69 cadastral units; see Appendix D, S1); I considered the more uniformly sized subcatchment-based planning units to be more ecologically relevant.....114

Figure 5.2 Cuddeback X-Change™ Color Model 1279 camera unit with Xenon strobe (i.e., white flash); the strobe can also be configured as an IR or black flash in areas of high human density to avoid detection and theft. Image source: <https://www.cuddeback.com/shop>.....115

Figure 5.3 The three extant members of the dasyuroid guild are the (a) Tasmanian devil (*Sarcophilus harrisii*), (b) the Eastern quoll (*Dasyurus viverrinus*), and (c) the spotted-tail quoll (*Dasyurus maculatus*), and the three problematic introduced species are (d) cats (*Felis catus*), (e) European fallow deer (*Dama dama*), and (f) black rats (*Rattus rattus*). 118

Figure 5.4 Maps of top-priority planning units for achieving either reserve network connectivity, bioregional representation, threat mitigation, or cost efficiency as sole objectives (see Appendix D, item S5 for maps of these four priority areas when rankings were generated for the cadastral planning units).....123

Figure 5.5 Maps of top-priority subcatchment-based planning units for maximising three measures of community biodiversity: species richness, Shannon-Wiener Index, and Simpson Index.....126

Figure 5.6 Map with top 25,209-ha of the community-diversity metric combination scenario (blue), the top 25,209-ha of the combination-criteria scenario (yellow), and the overlap between the two scenarios (green).....127

Figure 5.7 Map with top 25,209-ha identified for (a) high relative abundance of the members of the threatened dasyuroid predator guild and for connectivity, and for (b) high relative abundance of the three problematic introduced species and for the planning units most threatened by land uses.....130

List of Appendices

Appendix A: Chapter 2 Supplementary materials.....	162
Appendix B: Chapter 3 Supplementary materials.....	167
Appendix C: Chapter 4 Supplementary materials.....	182
Appendix D: Chapter 5 Supplementary materials.....	255

Thesis Abstract

Habitat destruction and degradation due to human land uses are core drivers of biodiversity loss, and area-based conservation efforts (e.g., protected areas) are a primary means of addressing these threats. The upcoming conference of the parties to the Convention on Biological Diversity is likely to endorse an increase in the protected area (PA) coverage target to 30% by 2030, with the goal of better mitigating ongoing biodiversity loss. However, the effectiveness of an expanded global PA network is threatened by problems like PA-impermanence, non-target 'leakage' effects, poor spatial prioritisation, and historically entrenched biases like unequal representation of habitat types. While each of these issues has been discussed in the literature, there are no clear guidelines for addressing leakage nor for coordinating local actions to build globally effective, expanded PA networks.

As framed in Chapter 1, the aim of my thesis was to provide real-world insights into the relative importance of the two emergent problems – leakage and globally uncoordinated prioritisation – in the context of global conservation, and to propose evidence-based solutions. The quantitative-triage methodology I apply seeks to ensure that area-based conservation efforts achieve maximal positive impact for irreplaceable and increasingly vulnerable biodiversity.

In Chapter 2, I assess the prevalence and magnitude of the spillover effects of PAs in a global systematic review of the literature. The first step in this review was to identify the most robust methods for measuring spillovers from PA establishment. Subsequent extraction and collation of all relevant data from the peer-reviewed literature provided generalised inferences on the existence, prevalence, magnitude, and direction of recorded spillover effects. I found that, in general, PAs have reduced deforestation rates effectively within their boundaries, without displacing deforestation pressures to nearby unprotected areas (i.e., without causing leakage, the problematic type of spillover). However, given the limitations of the available data, further confirmation of these results was required.

In Chapter 3, I use counterfactual methods identified in the systematic review to measure the target and non-target effects of recently established protected areas on deforestation rates within a global hotspot of habitat conversion: the Brazilian Amazon. A key methodological step for measuring the impacts of PA establishment is the estimation of a counterfactual a baseline rate of deforestation expected in a given landscape, from which deviations from the baseline in the PA and in its spillover zone were determined. The results from this regional study from the neotropics were aligned with those of the global systematic review, and I concluded that leakage effects from PA

establishment were not of sufficient magnitude to warrant direct mitigation efforts, compared to known problems of impermanence, location bias, and non-representativeness.

In Chapter 4, I address the problems that arise when local priorities drive PA expansion decisions at the expense of globally coordinated conservation efforts. I demonstrate a data-driven solution using a case study of Tasmanian forests using a transparent decision-support tool. I evaluated the potential allocation of land available for protection based on four objectives – representation of biogeographic units, connectivity, threat mitigation, and cost efficiency – each weighted equally. As a contrast to this ‘objective-neutral’ scenario, I also devised a jurisdictionally informed plan for tenure allocation, which prioritises features according to the origin of the target to which they contribute, where global conservation goals are valued over continental objectives over local concerns. Representational gap analyses and this jurisdictionally informed planning strategy identified the northeast of Tasmania as a target region for global and continental conservation goals.

In Chapter 5, I investigate the extent to which broad-scale design criteria for protected-areas (e.g., coverage of bioregions, network connectivity, threat mitigation, cost efficiency) reflect ecological priorities identified by fine-scale data: in this case, community-composition and focal species’ relative abundance data from a year of continual camera-trap monitoring for mammals and ground-dwelling birds collected from 201 sites. The focal area for this investigation was the northeast Tasmanian target region identified in Chapter 4. Machine-learning-assisted image classification to the species level enabled efficient calculation of distribution, diversity and relative-abundance metrics. Protectable sites were then ranked by these metrics and compared with other rankings generated using coarser proxies for biodiversity, validating the latter and grounding the global prioritisation of this region in the local ecology.

Collectively, the systematic review and the case studies within the complex socioecological contexts of Brazil and Tasmania yielded important contributions to the ‘crisis discipline’ of conservation science. This body of research revealed that known, entrenched problems are more important threats to effective area-based conservation than the widely perceived problem of PA leakage. Further, modern monitoring technology, computing power, machine- and statistical-learning methods and spatial decision support tools can be leveraged to achieve globally coordinated spatial prioritisation without losing sight of the ecological communities using the habitats being protected.

Chapter 1: General Introduction



Blue Tier of Tasmania, 2020

"Whether we and our politicians know it or not, Nature is party to all our deals and decisions, and she has more votes, a longer memory, and a sterner sense of justice than we do."

— Wendell Berry

I. Conservation science in context

The two defining events of the Anthropocene, biodiversity collapse and climate change, pose existential threats to the biosphere as we know it (IPBES 2019; IPCC 2021). Biodiversity and ecosystem loss are one of the five biggest threats facing humanity in the next ten years, according to the World Economic Forum (WEF, 2020). Current rates of extinction are about 1,000 times greater than background levels (Pimm et al., 2014) and are poised to increase, especially as the rate of climate change outpaces many species' adaptative capacity (Hoffmann & Sgrò, 2011; Visser, 2008) and habitat loss continues (Tilman, May, Lehman, & Nowak, 1994). Urgent, coordinated efforts must replace dispersed, incremental mitigation policies, despite the "attritional lethality" of these problems – the impacts are dispersed across time and space, and there is a gap between destructive practices and their deferred consequences. Furthermore, inaction can be actively facilitated by those that profit most from it (Nixon, 2011).

Despite powerful vested interests perpetuating business as usual, conservation interventions have become increasingly evidence based. Michael Soulé's foundational 1985 essay explained the emergence of the field of conservation biology as a "crisis discipline," promoting two key concepts: the diversity of organisms, ecological complexity, and evolution are good, and biotic diversity has intrinsic value irrespective of its instrumental or utilitarian value. Conservation biologists have enacted various interventions in hopes of perpetuating these two values: *ex situ* and *in situ* conservation efforts, reductions of human-wildlife conflict, species translocations, and the establishment and enforcement of anti-poaching laws, among others. The broader field of conservation science has since emerged (Kareiva & Marvier, 2012), and academics and practitioners alike are embracing more quantitative and systematic approaches akin to those that have already revolutionised the fields of modern medicine and public health.

Status report on the global protected-area network

The intensification of land-use change, and subsequent habitat destruction and degradation, are major drivers of the current biodiversity crisis (Pimm & Raven, 2000). Across all taxa, habitat loss is considered the greatest contributor to recent and potential future extinctions (Henle et al. 2004; Tilman et al. 1994). As such, the designation and enforcement of Protected Areas (PAs) are at the core of conservation efforts globally. Establishing PAs is a critical means of stopping habitat loss and thus combatting biodiversity loss. This thesis focuses on these *in situ* area-based conservation efforts, and it specifically assesses the advances in and pitfalls befalling systematic spatial planning of protected and conserved areas. For the purposes of this research, this land use, tenure, and

management classification, is defined in accordance with the International Union for Conservation of Nature:

“A protected area is a clearly defined geological space, recognised, dedicated, and managed, through legal or other effective means, to achieve long-term conservation of nature with associated ecosystem services and cultural values” (Dudley 2008).

Being aligned with the early understanding of “wilderness” as, by definition, ‘lacking people’ (Matthews, 2020), early national-park establishments were often accompanied by the forced displacement of local peoples (Agrawal & Redford, 2009). Since then, conservation science has slowly recognised the importance of social equity as a critical conservation objective, and the binary paradigm of the human-nature divide has been replaced with a more nuanced understanding of coupled socioecological systems (Mathevet et al., 2016; Young et al., 2006). Conservation science and practice ought to promote good governance, equity and effective conservation outcomes in a way that is inclusive of a diversity of contributions from non-state actors, especially those from First Nations peoples.

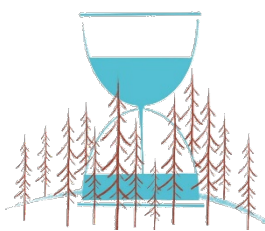
Expansion of the global PA network was a key goal of the Convention on Biological Diversity (CBD) of 2010, whereby the 196 Parties to the CBD adopted the Strategic Plan for Biodiversity 2011–2020, which contained 20 Aichi Biodiversity Targets. Target 11 called on the Parties to conserve 17% of terrestrial and inland water areas and 10% of coastal and marine areas by 2020 through “well connected systems of PAs and other effective area-based conservation measures.” Despite delays, the Parties to the CBD are expected to agree (imminently) to a post-2020 Global Biodiversity Framework that sets new, more ambitious targets for halting the loss of biodiversity and ecosystem integrity (Tsioumani, 2020).

The World Database of Protected Areas (WDPA) shows the size, location, and land-use restriction type of most PAs, but there are also some known gaps in this database, especially with PAs not being reported (Smallhorn-West & Govan, 2018). PAs may be State- (government-) designated, or they may be indigenous lands, community conservancies, or private PAs; they are subject to different strictness, and some allow for certain human activities within their boundaries. Currently, about 15% of land and 7% of marine areas are protected, making PAs one of the largest land uses in the world (Secretariat of the Convention on Biological Diversity, 2020). Yet, the extent to which PAs contribute to the persistence of all levels of biodiversity: genes, species, populations, and ecosystems, is strongly dependent upon their location, size, connectivity, management, permanence, and influences on land-use changes in the surrounding landscapes (Andam et al, 2008). Despite the recent, vast increase in the extent (area coverage) of official protected and conserved

areas, there remains much progress to be made: the increased extent of the global PA network stands in stark contrast to the available data on the increasing rates of loss and endangerment of species, suggesting that PA networks are failing to arrest the attrition of biodiversity.

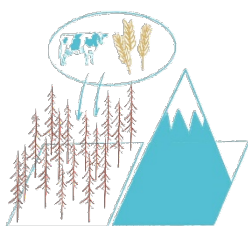
II. Threats to effective protected areas

PA networks can fail to effectively or sufficiently protect biodiversity due to impermanence (Mascia and Pailler 2011), location biases (Devillers et al., 2015; Joppa & Pfaff, 2009; R. Pressey et al., 1993; Venter et al., 2018) and non-representativeness in terms of habitat types protected (Joppa & Pfaff, 2009; Van Oosterzee, Blignaut, & Bradshaw, 2012; Venter et al., 2018), non-target leakage effects (Ewers & Rodrigues, 2008; Ford et al., 2020; Fuller, Ondeï et al., 2019), and globally uncoordinated spatial prioritisation of protection (Pouzols et al. 2014, Sala et al. 2021). Each of these is defined below.



Impermanence

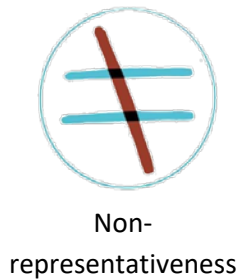
Although the conventional narrative regarding PAs is one of continuous growth in numbers and spatial extent, and indeed conservation policy and practice often assume that PAs are permanent fixtures on the landscape, PA downgrading, downsizing, and degazettement (PADD) events are widespread. The major threat posed by PA **impermanence** (Fund., 2021; Mascia & Pailler, 2011; Qin et al., 2019) has worsened following the COVID-19 pandemic (Kroner et al., 2021). Trends in efforts to scale back PAs are recorded in the online PADDTracker.org database, which shows both proposed and enacted downgrading, downsizing and degazettement events around the world. Research has assessed their extent (Cook et al., 2017), patterns (Pack et al., 2016), causes (Keles et al., 2020), and conservation implications (Mascia & Pailler 2011).



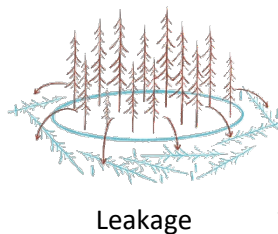
Location bias

Location bias is an entrenched historical trend by which PAs are preferentially located in areas where they are least likely to prevent land conversion (R. Pressey et al., 1993). Common attributes that have historically led to a PA being allocated reserve status include high elevation, steep slopes, long distance to roads and urban areas, as well as areas of low agricultural suitability (Joppa & Pfaff, 2009; Venter et al., 2018).). Indeed, such areas are ‘easy targets,’ being either unsuitable or too difficult to be used for any other purpose. Across many nations globally, PA networks

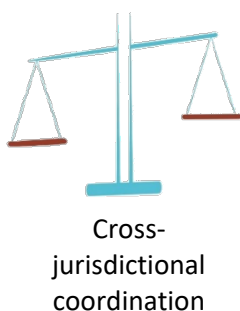
exhibit this bias, and the PAs with higher protection status are more biased than those with less strict protection status (Joppa & Pfaff, 2009). The global pattern of marine PAs is also characterised by location bias with recent expansion of large remote MPAs (Devillers et al., 2015).



Non-representative PA networks do not contain samples of all habitat types. Bioregionalisation frameworks have been developed to guide area-based conservation (e.g., global ecoregions, Australian bioregions). Yet, the vast increase in the extent of area-based conservation over the past decade has failed to achieve **ecological representation** within the global PA network (Kuempel et al., 2016). Kuempel and co-authors (2016) also went on to highlight that globally, the same level of ecological representation as has been achieved thus far by the existing network could have been achieved by completely random allocation of PAs.



Problematic spillover effects, called **leakage**, occurs when the establishment of a PA causes a 'spillover' of speculative behaviour in the nearby, unprotected areas. Establishment of the PA displaces habitat conversion to non-target, unprotected areas, reducing the net impact of the PA for conservation within the landscape. Leakage, coupled with non-representativeness, has the potential to generate perverse outcomes for biodiversity, whereby more, or more important, habitat is lost or degraded than would have been without the PA establishment (Bode et al., 2015; Ford et al., 2020; Renwick et al., 2015). A key step in measuring spillover effects is to estimate a counterfactual baseline – that is, what we expect would have happened both inside the PA and in the nearby unprotected surroundings if the PA had never been established. This can be estimated using representative samples of the unprotected, but otherwise similar, surrounding landscape.



Global scale conservation planning and international cooperation in area-based conservation efforts are vital for achieving globally meaningful conservation outcomes; indeed, a lack of international coordination would cause an efficiency loss much greater than the habitat loss expected from projected land-use change by 2040, whereas a globally coordinated PA

expansion effort could be nearly three times as efficient (Pouzols et al., 2014). Internationally **uncoordinated prioritisation** of where to protect can further engrain location biases, or can fail to meet habitat representation targets. Jurisdictions must reconcile their differences where compromise and trade-offs are inevitable.



Real-time
biodiversity
monitoring

The target of conservation efforts is the protection of biodiversity, but because it is impossible to assess all dimensions (e.g., at the genetic, species, and ecosystem levels; Socolar et al. 2016), broad-scale reserve design criteria (e.g., reserve network connectivity, representativeness of biogeographic units and habitat types, or vulnerability to human incursion) are often used to guide decisions. However, the downside of these easily accessible landscape-scale proxies is that they may not always capture on-the-ground changes: changes to direct community diversity measures (e.g., species richness) have been shown to be uncoupled from changes to surrogate measures (Hillebrand et al. 2018). In some contexts, a feature like PA-network connectivity may not, necessarily, improve gene flow and species persistence. The **lack of on-ground, time-series monitoring data** is limiting, but need not be: for mammals and some birds, improving technologies for camera-trapping hardware and image classification could inform planners about the extent to which reserve design criteria reflect differences in observed patterns of on-ground biodiversity.

Each of these challenges can reduce the efficiency of area-based conservation efforts for achieving impact for biodiversity on the ground. While all of these issues has been discussed in the conservation science literature, there are no clear guidelines for addressing two of them: leakage and uncoordinated spatial prioritisation. By 2017, leakage had been flagged as a potential concern by some papers (Bode et al., 2015); Renwick et al., 2015; Van Oosterzee et al., 2012), and a method for quantifying it had been proposed (Ewers & Rodrigues, 2008), but many fundamental questions regarding the prevalence, relative importance, and drivers remained unanswered. The need for more cross-jurisdictionally informed spatial prioritisation was recognised in the context of the terrestrial reserve expansion mandated by Aichi Target 11 within the 2010 Convention on Biological Diversity (Pouzols et al. 2014), as well as in the context of post-2020 marine PA expansion (Sala et al. 2021), but no decision support tool had yet been explicitly adapted for that purpose. Mitigation of

these two emergent threats first requires establishing an evidence base around the magnitude of threat they pose to PA effectiveness. Second, modern biodiversity-monitoring technology, computing power, statistical learning methods, and decision support tools can be leveraged to optimise appropriate mitigation strategies and interventions.

III. Thesis aims and case studies

The aim of this thesis is to provide real-world insights into the relative importance of the two emergent problems – leakage and globally uncoordinated prioritisation – in the context of global conservation, and to propose evidence-based solutions.

The four data chapters of this thesis investigated the land-use change dynamics around PAs at different spatial scales. Chapter 2 is a global systematic review, while Chapters 3 and 4 investigate two case studies at a regional scale, the first being focused on the Brazilian Amazon, and the second on the island-state of Tasmania (Australia). Chapter 5 further narrowed the focus by examining a specific conservation target area, the northeast of Tasmania. The systematic review of PAs and the three case studies, each located in hotspots of land clearance, have important implications for local PA planning and management, but their implications for conservation science apply broadly. This thesis focusses on the terrestrial realm, but some of the threats, and the proposed solutions, may also apply to marine PAs.

Brazil and Australia are two global hotspots of land clearance. These two megadiverse countries, of almost equal geographic size, have championed the establishment of some of the world's most enormous PA networks, although these networks have experienced downgrading, downsizing, and degazettement events (see Pack et al., 2016 and Cook et al., 2017). Ecotourism for landscapes and iconic species are economic boons in both focal areas assessed in this thesis, the Brazilian Amazon and Tasmania. The striking differences the historical context of land-use changes and habitat conversion in these two focal regions provide insight into the more general contrasts between strategies relevant to the Global South as compared to other parts of the world.

Historical context of conservation in the Brazilian Amazon

Amazônia Legal (the Brazilian Amazon) is a 5,016,136 km² administrative region of Brazil, home to a population of approximately 24 million people, that includes the entire states of Acre, Amapá, Amazonas, Pará, Rondônia, Roraima and Tocantins, as well as part of Mato Grosso and Maranhão. The Brazilian Amazon is not officially listed as a global biodiversity hotspot because threats are still kept in check by its (albeit dwindling) remoteness. Various actors are implicated in the deforestation dynamics of *Amazônia Legal*, including *grileiros* (illegal land grabbers who forge

fraudulent land-title documents), loggers and sawmillers, ranchers and their employees, disorganised landless migrants, organised “*sem terra*” landless rural workers, agribusiness-scale soy farmers, and miners (Fearnside, 2008). As of 2017 Amazônia Legal contained 340 conservation PAs and 459 Indigenous Lands. State-governed PAs are not the only conservation mechanism in place to regulate forest clearing in this region: a vast network of Indigenous Lands are important conservation actors, especially coupled with contributions from environmental legislation (e.g., the Forest Code of 2012), supply-chain regulations (e.g., the Soy Moratorium of 2006). Large-scale illegal land appropriation, e.g., by *grileiros*, and resource extraction by multi-national corporations are major drivers of land clearance in this region (Fearnside, 2008).

Historical context of conservation in Tasmania

Tasmania is an island state of Australia located 240 km south of the Australian mainland; it covers a land area of 68,150 km² between latitudes -40°S and -43°S and longitudes 144°W and 149°W. Indigenous Tasmanian Peoples used fire as a land management tool for several tens of thousands of years (Gammage, 2008). The arrival of British colonisers in the early 1800s brought vast, intensive land-use changes, invasive species, and disrupted Aboriginal land management. Until about 1970, the Tasmanian state tended to legally gazette PAs of low primary-resource value. Following the first application of systematic techniques for PA site selection (Kirkpatrick, 1983; R. Pressey, 2002), there was an increase in the representation of plant communities in the state’s PA network (Mendel & Kirkpatrick, 2002). In 1997 the Regional Forestry Agreement (RFA) process set a target of protecting 15% of the original (pre-colonisation) area of each widespread forest type, and it set higher targets for rare forest communities. As a result, Tasmania’s current PA network is more representative of its component vegetation communities than are the PA networks of most other regions in the world (Mendel & Kirkpatrick, 2002). Nonetheless, vegetation communities such as lowland tussock grasslands and grassy woodlands remain poorly represented. Furthermore, there is a large disparity in proportion protected across Tasmania’s three ecoregions (Olson et al., 2001) and nine nationally declared bioregions (Department of the Environment, 2012).

IV. Thesis objectives

The purpose of my PhD research was to contribute new knowledge to the field of systematic conservation planning, from the global to local scale. Generally, I assessed means of spatially optimising land-use change by strategically establishing PA networks where they can make the most difference for biodiversity. Specifically, I sought to provide real-world insights into the relative importance of the two emergent problems – leakage and globally uncoordinated prioritisation – in the context of global conservation, and to propose evidence-based solutions. I had three specific objectives:

1. To investigate the barriers to achieving conservation impact through the establishment of PAs.
2. To clarify the appropriate triage of those identified but poorly understood barriers.
3. To test quantitative means of providing data-driven conservation decision support for avoiding or mitigating these barriers.

V. Thesis structure

This thesis contains six chapters. Chapters 2-5 are written as stand-alone research papers. Chapters 2 and 3 are modified versions of publications in the peer-reviewed scientific journals *Global Ecology and Conservation* and *Biological Conservation*, respectively. Chapters 4 and 5 have been formatted for submission to the journal *Conservation Biology*, and the former was under review at the time of the submission of this thesis. This formatting has resulted in some repetition among chapters.

The data chapters were prepared with advice and feedback from my supervisory team, and their contributions are recognised with co-authorship where appropriate. I was primarily responsible for all aspects of the thesis, including formulating the research questions, literature reviews, planning and leading field data collection, data analyses, and writing. My primary supervisor provided the research funding for all activities. The chapters are designed to address knowledge gaps identified above and leveraging the unique case study attributes of Brazil and Tasmania.

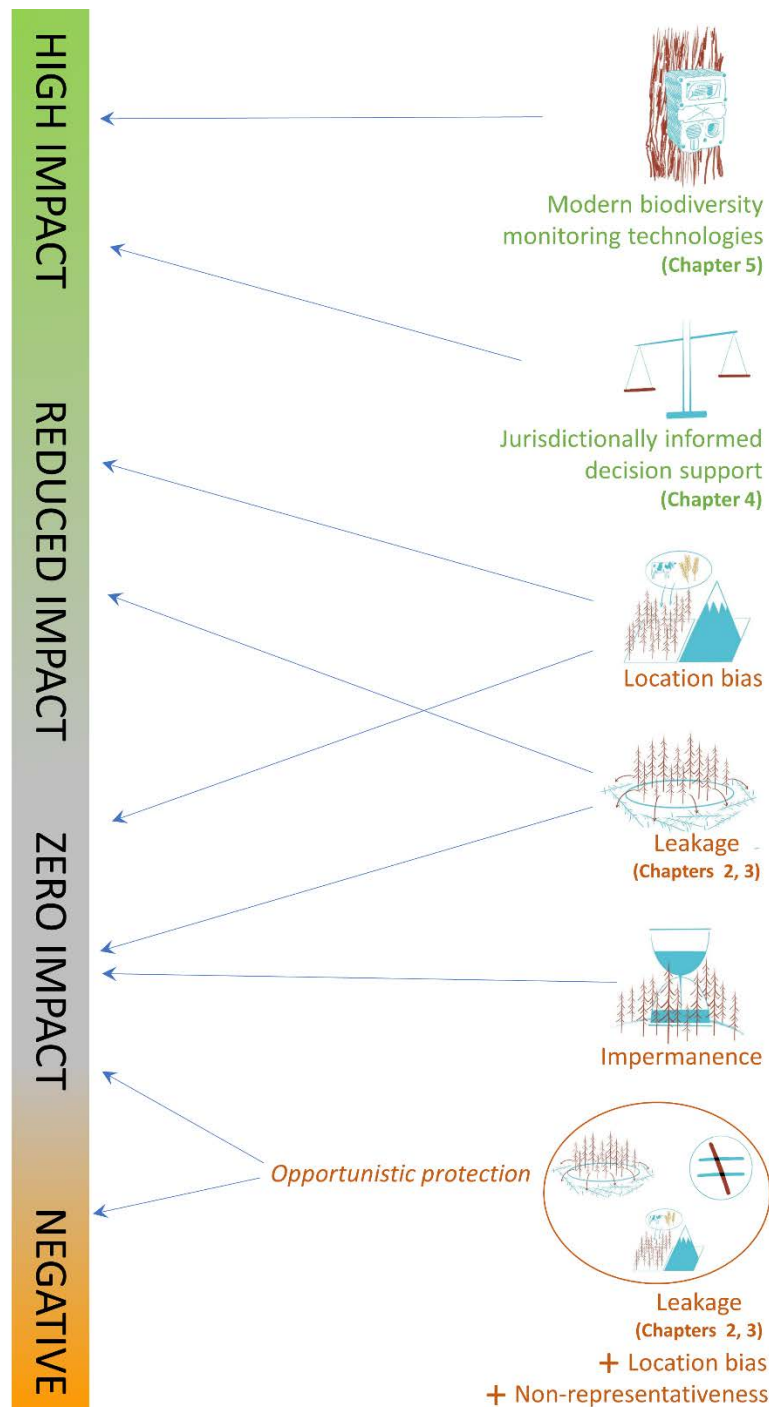


Figure 1.1 Conceptual map of how the thesis chapters relate and contribute to the overarching aim of spatially optimising land-use change by strategically establishing protected area (PA) networks where they can make the most difference for biodiversity. Where a PA's impact is defined as the difference made to the rates and locations of land-use changes as compared to the likely outcome in the landscape without that PA, there are several interacting pitfalls (orange text) to avoid. This thesis assesses these threats, particularly that of leakage, and presents two solutions (green text): more jurisdictionally informed decision support framework and the use of modern biodiversity monitoring technologies.

Chapter 1: General Introduction

Here in Chapter 1, I have introduced and defined the concepts, terminology, and remaining gaps in knowledge of the strategies that lead to effective PA networks. I have provided an overview of the research questions relevant to area-based conservation that have asked been since the advent of the formal discipline of conservation science, and I have summarised the extent to which they have been robustly answered. I have identified the frequency of calls for future research into the non-target effects of PAs and into the threat of parochialism in conservation decision making. I then provide a preview of the investigations I conducted with the goal of filling these acknowledged knowledge gaps and responding to these calls for future research.

Chapters 2-5: Data Chapters

In the four core chapters, I analyse data from the collated results and appendices of publications selected by a systematic literature review; satellite monitoring of deforestation; open-access spatial data repositories for vegetation communities; tenure cadastres; species observation databases; enterprise suitability maps; time-series land-use change data sets; bioregionalisation frameworks; and one year of camera trap monitoring.

Chapter 2 investigates whether PAs' official establishment have spurred speculative actions by natural resource extractors or developers, increasing the rate of land use change in the surrounding unprotected areas, such that the net impact of that PA on the landscape was reduced or negated. This chapter identifies the most robust, counterfactual methods by which the leakage problem can be detected, and then collates all evidence of this problem available in the scientific and grey literature. The outcome of this research is generalised inferences on the existence, prevalence, magnitude, and direction of recorded leakage effects globally.

Chapter 3 presents a retrospective assessment of the prevalence and intensity of leakage resulting from recently established PAs within a global hotspot of deforestation and clearing: the Brazilian Amazon. The goal of this study is to calculate the magnitude of observed deforestation leakages using satellite-monitored data. The quantified non-target damage inflicted by leakage is then compared to the magnitude of the region's PA downgrading, downsizing, and degazettement events of the same time period.

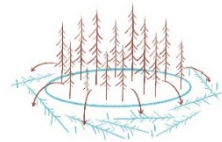
Upon concluding that leakage mitigation is of low urgency and consequence, I then assess the challenges of achieving maximal biodiversity impact in planning the expansion of PA networks to

meet globally agreed-upon objectives in Chapter 4. This chapter seeks to demonstrate how conservation decision support can be adapted to a more global lens, without losing sight of the particular needs of the local ecology, using an example from another global deforestation and clearing hotspot: Australia. With this example, I showcase the benefits of strategic conservation prioritisation compared to the opportunistic PA expansion adjacent the UNESCO-listed Tasmanian Wilderness World Heritage area, and I propose alternative solutions.

In Chapter 5, I identify a key limitation of systematic conservation planning: lack of direct measures of biodiversity values over time. In addressing this limitation, I advocate for long-term wildlife monitoring and collection of high-resolution spatial data for time-series measures of biodiversity. Specifically, I evaluate how spatial prioritisation of conservation target areas can be informed using data from one year of camera trapping in the Tasmanian temperate forest ecoregion. I compare these survey-informed prioritisations with that used broad-scale reserve design criteria.

Chapter 6: General Discussion

In the final chapter I synthesise the key findings of the data chapters, framing these outputs within the frame of quantitative triage of threats and the most appropriate solutions to be implemented in 21st-century systematic conservation planning. I discuss the importance of long-term ecological monitoring at fine spatial scales to inform conservation decision support and to improve target-setting within jurisdictions' key conservation policy mechanisms. Having addressed the relative importance of historical and emergent problems faced by PA planners and managers in three different contexts (globally, in the Brazilian Amazon, and in Tasmania), I conclude with suggestions for future directions in research, planning, and decision support. I also explain the opportunities for leveraging modern technologies to identify and prioritise likely climate refuges and refugia, to expand and intensify effective private-land conservation mechanisms, and to shift the planning, assessment, and reporting of conservation interventions to the units of impact.



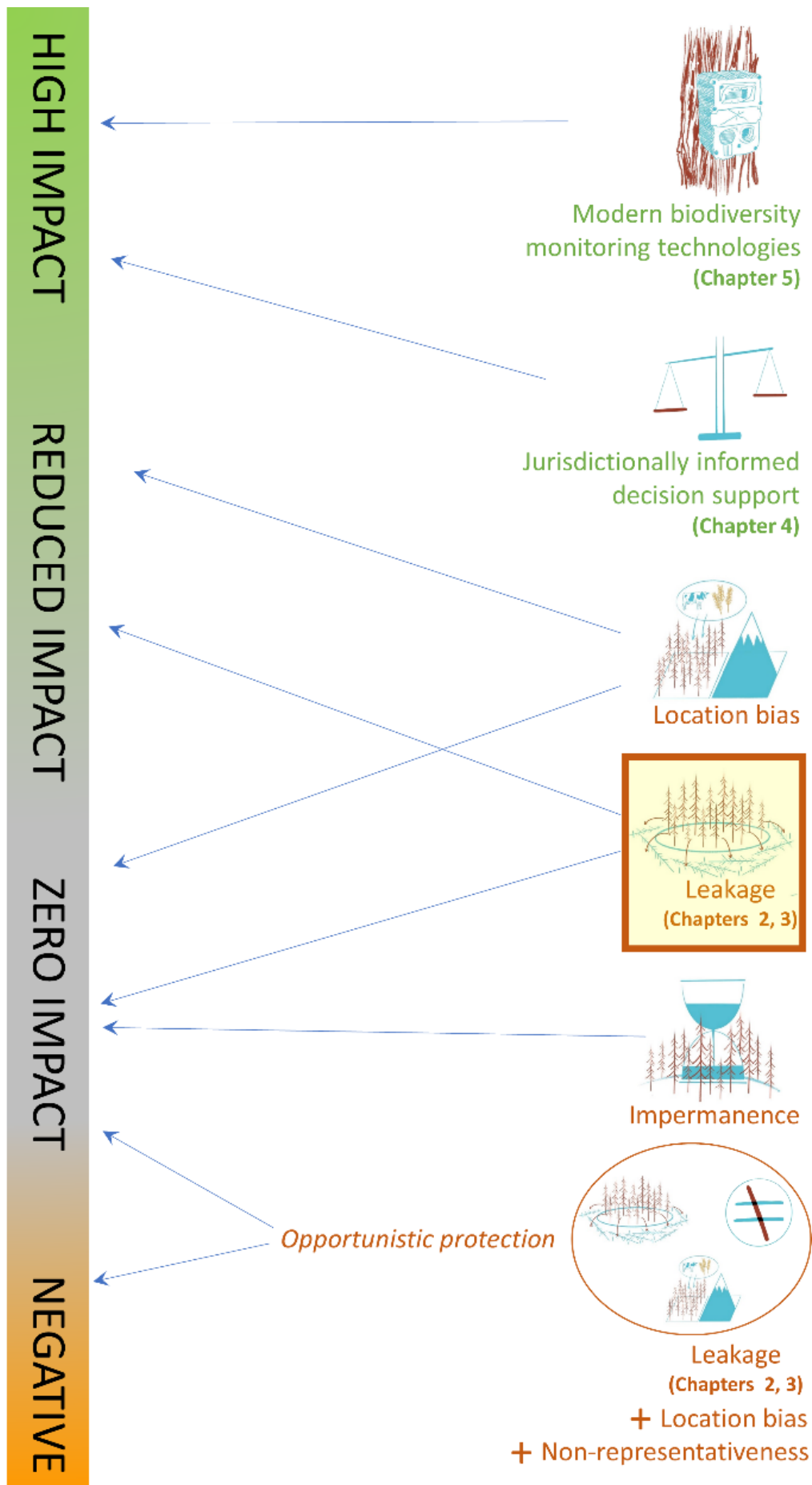
Chapter 2: Assessing the prevalence and intensity of leakage from protected areas globally¹



Te Papakura o Taranaki/Egmont National Park in Aotearoa/New Zealand. Image from Google Earth.

“Without reflection, we go blindly on our way, creating more unintended consequences, and failing to achieve anything useful.” — Margaret J. Wheatley

¹ Published as a peer-reviewed article: **Fuller, C.**, Ondej, S., Brook, B. W., & Buettel, J. C. (2019). First, do no harm: A systematic review of deforestation spillovers from protected areas. *Global Ecology and Conservation*, 18, e00591.



Abstract

Land-use restrictions in protected areas (PAs) might have unintended spillover effects on non-target, neighbouring areas. In the case of leakage, land-use change that would have occurred in the PA is displaced to an unprotected area where it would not have otherwise occurred. The resultant 'leakages' can offset benefits achieved inside PAs, confound impact assessments, and exacerbate the problems of opportunistic protection of lands. Conversely, in the case of 'blockage', the unprotected surroundings experience less land-use change than would have otherwise occurred due to a positive spillover effect from nearby protected areas. Little is known about the magnitude, ubiquity, and predictability of spillovers that have already occurred in the global PA network. Here I systematically review the literature and collate the existing evidence to quantify deforestation spillovers. I calculated deforestation rates within 3,398 PAs, most of which were found in tropical and subtropical moist forests, and in their unprotected adjacent surroundings and compared these rates to a baseline derived from the wider landscape. Of the 2,575 PAs that effectively restricted deforestation rates within their bounds relative to the baseline, 11.8% showed leakage and 54.8% exhibited blockage. Deforestation rates in the remaining 33.4% were indistinguishable from their respective baselines. Linear modelling of the correlates of leakage and blockage showed that PA-specific characteristics like size and IUCN category were uninformative, whereas national-scale socioeconomic factors like population density and GDP were useful predictors. Although spillovers from land-use restrictions are ultimately driven by socioeconomic factors, their ecological consequences are such that PA assessments should routinely and explicitly account for displaced impacts to the unprotected surroundings.

Introduction

The expanding global protected area (PA) network acts as a primary bulwark against encroaching anthropogenic threats to biodiversity, and its importance is codified into international law: Aichi Target 11 of the 2010 Convention on Biological Diversity (CBD; <http://www.cbd.int/sp/targets>) called for the protection of 17% of the planet's terrestrial surface by 2020, and this goal is likely to be achieved (Tittensor et al., 2014). The well-developed field of systematic conservation planning (Margules and Pressey, 2000) provides a framework for implementing and managing this significant investment by governments to restrict or ban human land use for the benefit of biodiversity. Evaluating PAs' effectiveness is difficult, because their establishment can be motivated by multiple agendas (Naughton-Treves et al., 2005), but their ability to prevent forest conversion can easily be assessed empirically; as such, this is a commonly used metric for ensuring that these conservation efforts provide tangible benefits. To date, published assessments have demonstrated repeatedly that PAs experience significantly lower rates of forest clearing in comparison to their unprotected surroundings (Nagendra, 2008; Naughton-Treves et al., 2005). Yet, such assessments of PA effectiveness rarely acknowledge the possibility of spillover effects, whereby protection in one area impacts the nontarget neighbouring areas, with potentially negative implications for biodiversity (Bode et al., 2015; Renwick et al., 2015).

The possible impact of terrestrial PAs on their unprotected adjacent surroundings¹ (UAS) can be divided into three categories: leakage, blockage, or no effect. Leakage, the problematic type of spillover, occurs when land use restricted in one area is displaced to an unprotected area where it would not otherwise have occurred, diminishing or even entirely offsetting habitat conservation achieved within a PA (Ewers and Rodrigues, 2008). Leakages are well understood in the context of carbon offsetting (Murray et al., 2004). Conversely, blockage incurs an unintended beneficial outcome; for example, in the context of marine protected reserves, where the primary nontarget effect observed is increased fisheries productivity near reserve bounds (Harmelin-Vivien et al., 2008), spillover effects are generally considered to be a positive side effect. In the terrestrial context, blockage occurs when land use is restricted in the target area whilst also occurring at lower rates than would be expected in the corresponding UAS (García, 2015). Blockage can manifest, in the context of a forested landscape, as a “halo” of limited clearing directly surrounding a PA in a more heavily cleared broader landscape; such halos may be due to protection in the region discouraging investment in transportation infrastructure and industry (García, 2015). Although de facto protection associated with a blockage effect is ostensibly beneficial for habitat conservation, it is unlikely to be permanent. Where no spillover occurs, rates of land use in the UAS are simply concordant with those that would be expected without nearby protection (e.g., Andam et al., 2008).

Land-use spillovers from terrestrial PAs are rarely measured and reported directly (Miteva et al., 2012; Pfaff and Robalino, 2017) although spillovers are both quantifiable (Ewers and Rodrigues, 2008) and have important implications for biodiversity (Bode et al., 2015; Renwick et al., 2015). If PA impact assessments do not account for leakage where it is occurring, the difference between the deforestation rate in the PA and the rate in the surroundings can be inflated as a consequence, and thus the assessment will overstate the PA's land-use restriction success and will fail to acknowledge the negative impact that results in the UAS (Ewers and Rodrigues, 2008). Furthermore, leakage could involve displacement of human land use from areas of low biodiversity value to areas containing more important habitats of more threatened species (Renwick et al., 2015). This perverse effect is plausible, given that PAs have historically been established in places unlikely to be developed for human land use rather than in top priority areas for biodiversity conservation (Joppa and Pfaff, 2009; Venter et al., 2018).

An understanding of the direction, magnitude, ubiquity, and predictability of past spillovers is critical for informing the future placement and maintenance of the PAs, if they are to reach the CBD's goals for landscape protection and biodiversity conservation. Here I systematically review the evidence for deforestation spillovers, especially leakages, from land-use restrictions associated with PAs to determine the extent to which PA establishment has done “more good than harm” (Pullin and Knight, 2009). I sought to answer the following questions:

- (a) Of PAs that successfully restricted deforestation within their bounds, how many demonstrated a positive (blockage) versus negative impact (leakage) on their unprotected adjacent surroundings?
- (b) Is the magnitude of spillover, whether leakage or blockage, related to characteristics of the PA (e.g., size or IUCN category) or to national-scale socioeconomic factors related to deforestation pressure (e.g., population density and the extracted forest products' economic value)?

The “unholy trinity” of challenges facing efforts to reduce deforestation is to simultaneously achieve additionality, permanence, and avoidance of leakage (Van Oosterzee et al., 2012). Failure to account for leakage where it is likely to occur would violate conservation's version of medicine's principle of “first, do no harm”.

Methods

Search strategy and evidence screening

An a priori systematic review protocol was devised following guidelines specific to conservation and environmental management (Pullin and Stewart, 2006). Four databases were searched between January and March 2018: Web of Science, Scopus, Google Scholar, and the ProQuest thesis and dissertation database (Table 1.1). No topical or geographic restrictions were imposed.

Table 2.1 Search terms for retrieval of studies. The Boolean operator ‘OR’ linked terms within categories while ‘AND’ linked terms between categories; asterisks indicate wildcards and return any use of the root word; quotation marks prevent the search from splitting a phrase into its word components.

Intervention	Outcome	Metric
“protected area*”	leakage	“land use*”
OR...	OR...	OR...
"PA*"	displacement	"deforestation*"
"national park*"	spillover	clearing
"conservation*"	"displaced effort"	"human impact*"
"habitat*"	"protected area isolation"	"anthropogenic pressure*"
"biodiversity*"	"buffer zone*"	"anthropogenic threat*"
"Indigenous land*"	"buffer area*"	

Unique records identified by the database searches were screened manually by title for relevance to conservation science or policy. Those records relevant to conservation were then screened using the abstract/summary for any mention of spillover (or alternate terminology; Table 1) caused by a conservation or environmental policy or intervention. Finally, full-text screening was used on the filtered list to identify records that provided a sufficient quantitative measure of PA impacts to enable data extraction (or data request). To ensure the comprehensiveness of the literature search, I also screened the bibliographies of the resultant records. An independent reviewer affirmed inter-rater screening reliability.

Data extraction

According to Ewers and Rodrigues (2008), spillover that occurs as a consequence of PA establishment can be quantified given a measure of land use (e.g., deforestation rate) over the same

period in three separate areas of the landscape: a) within the PA, b) in the unprotected adjacent surroundings (UAS) of the PA, and c) in representative parcels from the wider landscape, from which a 'baseline' deforestation rate can be derived (Fig. 2.2a). If the area, or deforestation rate was not reported in the main text or supplementary information, these data were requested from the corresponding author. A few studies reported alternative proxy measures of land use, like human footprint (Tapia-Armijos et al., 2017), but these (Beresford et al., 2013; Borgström et al., 2012; Joppa and Pfaff, 2011; Paiva et al., 2015; Seiferling et al., 2012) were excluded during the full-text screening (Fig. 2.1). Although both forest degradation and forest clearing indicate human land use, only deforestation values were extracted because it was the most consistently reported metric. Data were extracted regardless of the number of years between PA establishment and the deforestation monitoring period due to data availability, but ideally, deforestation monitoring would begin prior to, or immediately following, PA establishment. I converted extracted deforestation data to average annual percent of area deforested in order to standardise across PAs of varying size and duration of monitoring.

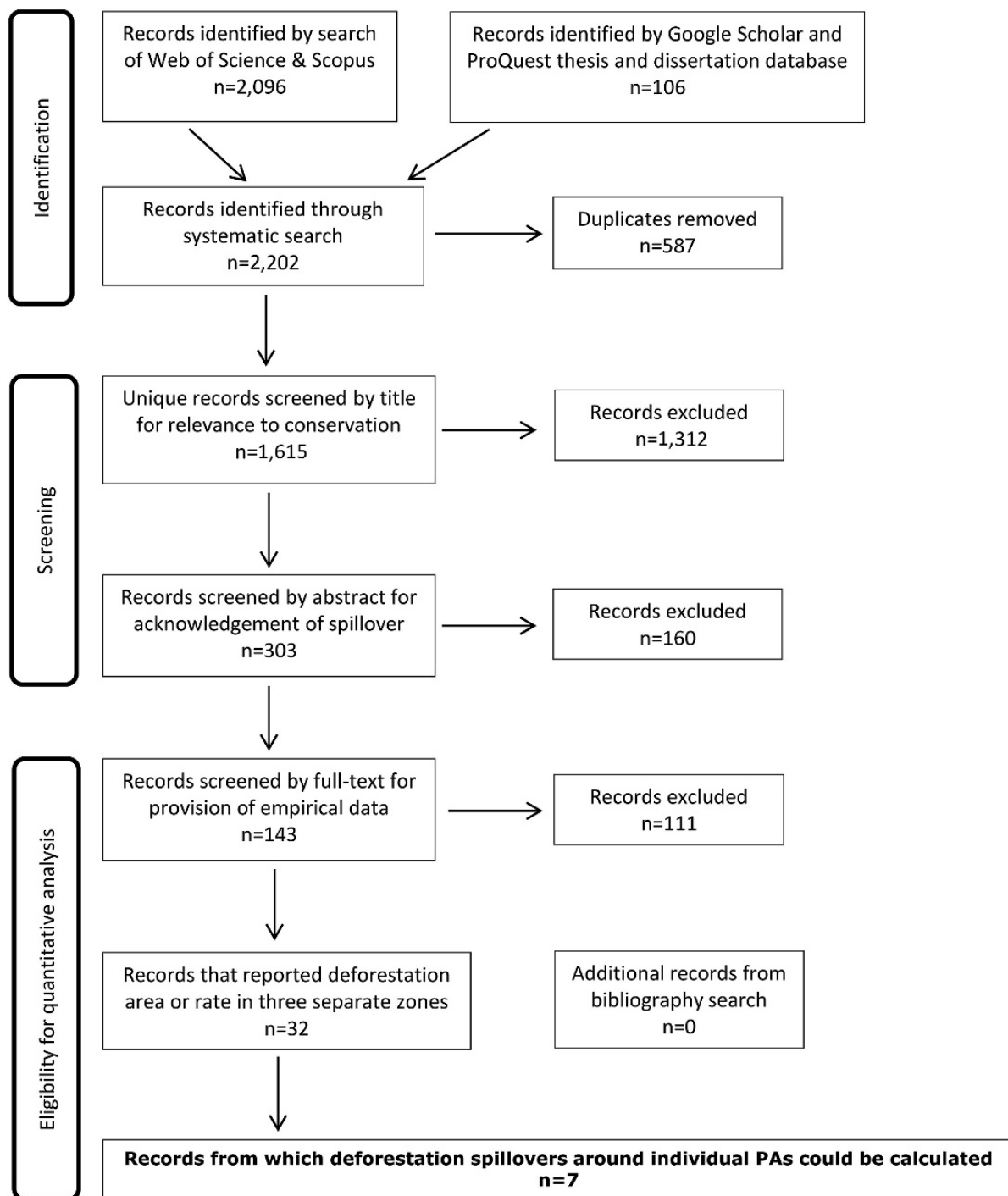


Figure 2.1 Screening protocol and numbers of papers included at each step in the systematic review process. Diagram modified from Moher et al. (2009).

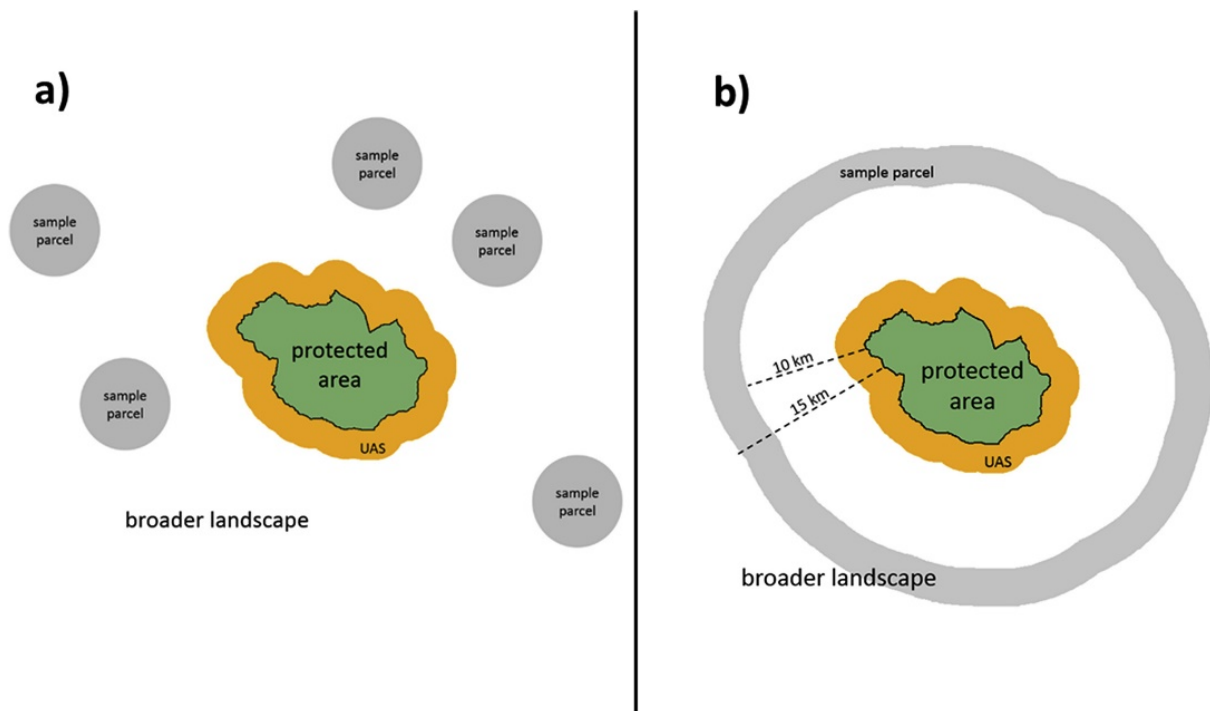


Figure 2.2 Map of a hypothetical landscape in which spillover from a PA to the unprotected adjacent surroundings (UAS) can be calculated. The sample of the broader landscape used to derive the baseline deforestation rate could be obtained a) as proposed by Ewers and Rodrigues (2008), or b) as provided by Spracklen et al. (2015).

PAs' target and non-target impact assessment

Some studies, including that of Spacklen et al., did not quantify spillovers directly, because leakage detection was not the chief objective of the study; however, they did report relevant data from which I calculated a PA's unintended, non-target deforestation outcomes. I first derived a counterfactual baseline rate of deforestation, i.e., an estimate of what would have happened without legal protection or nearby legal protection, based on the assumption that the average deforestation rate reported in the sample of the broader landscape (Fig. 2.2) was representative of the likelihood of deforestation in the PA and UAS in the same time period. Given the paucity of studies, I extracted or calculated the spatial baseline in this way regardless of how the landscape sample was chosen, so long as the sample was separate from the UAS zone. However, these samples would ideally be selected with matching methods in order to improve the comparability of this “control” area and the protected “treatment” area (Andam et al., 2008; Mas, 2005).

I then confirmed beneficial PA impact within the target area by identifying those PAs within which the deforestation rate was lower than for the wider-landscape samples. Those for which this

was not true may be considered “paper parks,” PAs for which gazettal is not concomitant with staffing, management, or enforcement (Di Minin and Toivonen, 2015). Since in these PAs protected designation did not inhibit deforestation, I infer that displacement of deforestation pressure to the surroundings is unlikely.

Following removal of the possible paper parks (Fig. 2.4), PAs’ non-target impact was calculated as the difference of the average deforestation rate observed in the UAS and the baseline rate for the same period. These nontarget impact values, i.e., the spillover magnitudes, were categorised as blockage when a smaller deforestation rate was observed in the UAS than was expected; as leakage when a greater deforestation rate was observed than was expected; or as negligible spillover when the difference in the observed and expected average annual percent deforestation rates was smaller than 0.01.

Statistical analysis

I used generalised linear modelling to evaluate fifteen potential predictors of spillover magnitude. Those specific to individual PAs (area, age at the time of deforestation monitoring, and IUCN category) were obtained directly from the papers’ main text, from their online supplementary information, or from the World Database on Protected Areas (WDPA). National-scale socioeconomic and political factors ($n = 11$) relevant to deforestation pressure (Table 2) were collected for each country containing one or more of the PAs for which a spillover value was calculated. Where possible, the average over the years 2000–2012 was calculated to reflect the period over which deforestation was monitored for most of the PAs. The continent in which a PA was located was also evaluated as a predictor of spillover magnitude. Three ostensibly important study-specific factors—deforestation monitoring duration in years, UAS definition, and wider landscape sampling method—could not be investigated due to the small number of studies.

Table 2.2 National scale meta-data factors, their sources, and rationales for their inclusion.

Potential national-scale predictor	Source	Rationale
- Population density (people per sq. km of land area) in 2017 - Population growth (annual %) in 2017 - Rural population growth (annual %) in 2017 - GDP growth (annual %) in 2017 - GDP (USD) in 2017 - Ratio of percent of land area under agriculture to percent land area under forest	The World Bank	Human development pressures include economic, agricultural and population growth (Waldron et al. 2017). The most recent data for these development indicators were collected.
Total value in USD of all forest products removed in 2005 Value in USD of the forest products removed per hectare in 2005	The Food and Agriculture Organisation of the United Nations	These factors are specific indicators of timber and other forest resource extraction pressure.
Carbon dioxide emissions in the unit of 1000 metric tons for the year 2009	Carbon Dioxide Information Analysis Centre	Changes in land use, mostly deforestation, have been a significant component of anthropogenic carbon emissions from 1960 to 2015 (Le Quéré et al. 2016).
Percentage of the total energy supply from solid biofuels in 2010	International Energy Agency	Biofuel plantations can cause indirect land-use changes resulting in deforestation (Lapola et al. 2010) perhaps especially if a country derives a large proportion of its energy supply from biofuels.
2017 Corruption Perceptions Index	Transparency International	Corruption in national governance may affect the outcomes of conservation projects (Smith et al. 2003).

Annual conservation spending	<i>Nature</i> publication by Waldron et al., 2017	This factor was included as an indicator of conservation effort, a counterpart to the indicators of human development pressure.
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Leakage and blockage magnitudes (including those classified as negligible in Fig. 2.4) were analysed separately as logit-transformed response variables. Two approaches were used to remove observations with missing data, as all fifteen potential predictor variables were not available for all 2,575 PAs. The first was a subset of observations with complete cases, whereby every predictor was available for the observation (no missing values). A second, larger data set was created by removing two factors, PA age and IUCN category (for which many case studies lacked data), and then deleting individual observations that were missing other predictors. Using these two data frames, I fitted generalised linear models to predict: (i) leakage, and (ii) blockage outcomes. All subsets of predictor variables were tested using the leaps library (Lumley, 2017) in R (R Core Team, 2018), with k-fold (k = 10) cross-validation (James et al., 2013) used to select the combination that build the model with the best ‘hold-out’ predictive performance, thereby identifying the most important variables related to spillover magnitude.

Results

Evidence from the systematic search

The literature search yielded 1,615 unique papers, 143 of which addressed spillovers in the context of conservation interventions (Fig. 2.1); these interventions ranged from payments for ecosystem services in Mexico (Alix-García et al., 2012) and zero-deforestation cattle agreements in Brazil (Alix-García and Gibbs, 2017) to the establishment of PA networks in various countries—the latter being the focus of this review. I identified studies that directly addressed the possibility of land-use leakage from PAs, but which deemed quantitative assessment beyond their scope (Barber et al., 2014; Nepstad et al., 2006; Pfeifer et al., 2012). After excluding records that did not report, nor provide sufficient data to calculate deforestation spillovers around PAs, only seven studies remained (Fig. 2.1). These (Ament and Cumming, 2016; Apan et al., 2017; Dewi et al., 2013; Phua et al., 2008; Sanchez-Azofeifa et al., 2003, Sánchez-Azofeifa et al., 2009; Spracklen et al., 2015) contained relevant data for 3,404 PAs (Appendix A, S2), of which 3,374 were reported by Spracklen et al. (2015) global-scale study of moist subtropical and tropical forest PAs. The literature search yielded no case studies from PAs in Europe and relatively few (n = 66; 1.9%) from Australia or Oceania (Fig. 2.3).

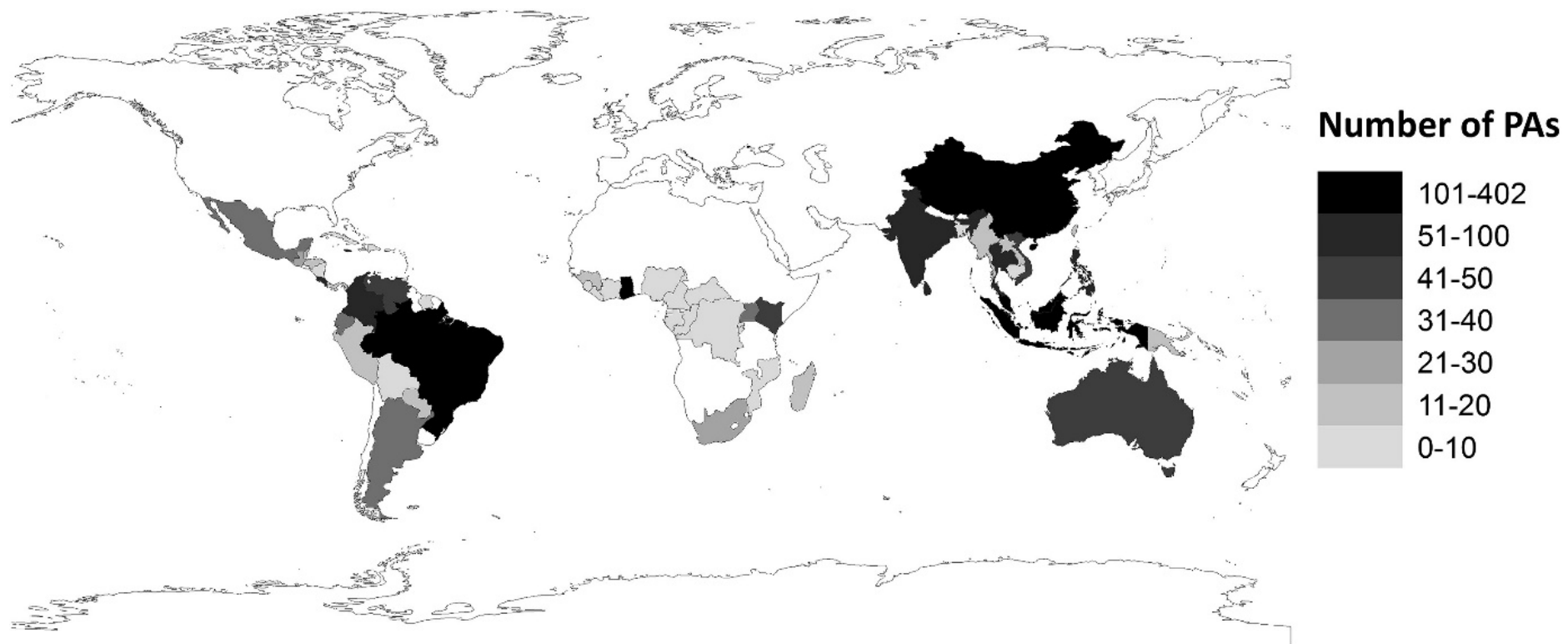


Figure 2.3 Map of the countries where deforestation rates within and around PAs, with which spillover could be calculated, were reported in the refereed literature. The shading indicates the number of PAs per country.

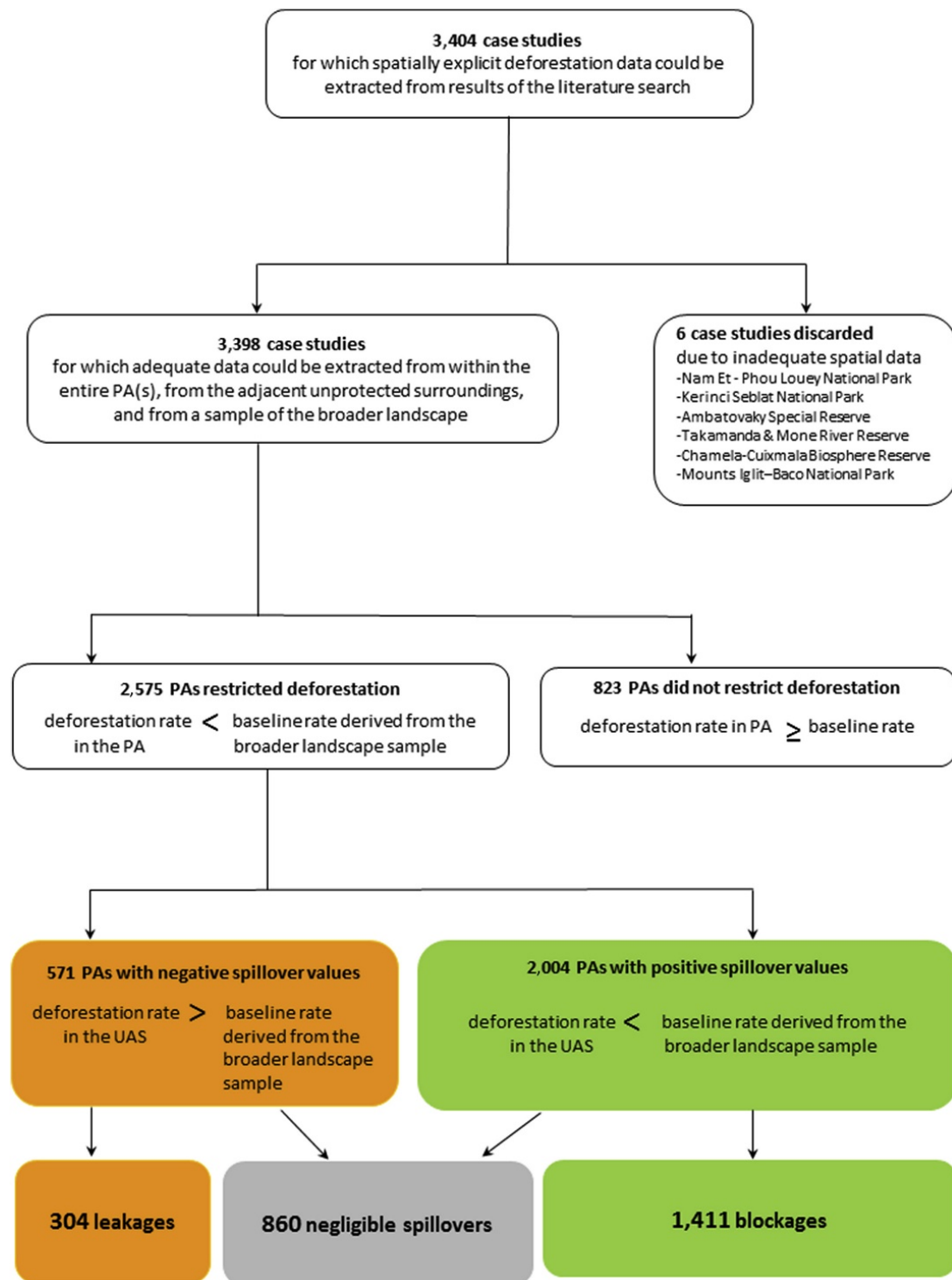


Figure 2.4 Flowchart of the stepwise results of assessing PA impact on deforestation rates, indicating the number of PAs that were extracted from the evidence search, that passed data cleaning, that had positive versus negative impact within the target area, that had negative versus positive impact in nontarget area, and that were assigned each spillover outcome type (leakage, negligible spillover, or blockage). Blockage outcomes outnumbered leakage outcomes.

Although all seven relevant studies reported deforestation rates within the PA(s), within the UAS, and in a sample of the wider landscape, the definition of the latter two areas was not standardised across studies. The UAS was defined as either 1, 2, 5, or 10-km of radial distance out from PA bounds, depending on the authors' idiosyncratic choice, with the most frequent being 1 km, as chosen by Spracklen et al. (2015). The area chosen to represent the broader landscape likewise varied across studies, with the most common choice being a band spanning 10–15 km beyond the given PA's boundary (Fig. 2.2b).

Target and nontarget impacts on deforestation rates

After removal of six PAs due to inadequate spatial data and 823 PAs that did not reduce deforestation rates within their boundaries relative to the rates in the wider landscape samples (Appendix A, S3), spillover effects to the UAS were quantified for 2,575 PAs. Blockage outcomes far outnumbered leakage outcomes. The deforestation rates in 2,004 (77.8%) of these 2,575 PAs were associated with lower-than-expected deforestation rates in the UAS, whereas for 571 (22.2%) they were higher than the associated baselines. After removing 860 (33.4%) PAs where the protective effect of the UAS was close to zero, I identified 304 PAs (11.8%) with evidence of leakage and 1,411 (54.8%) PAs showing blockage (Fig. 2.4). Leakages were least frequent for PAs irrespective of IUCN category, land-use restriction types, sizes, ages or continents (Fig. 2.5).

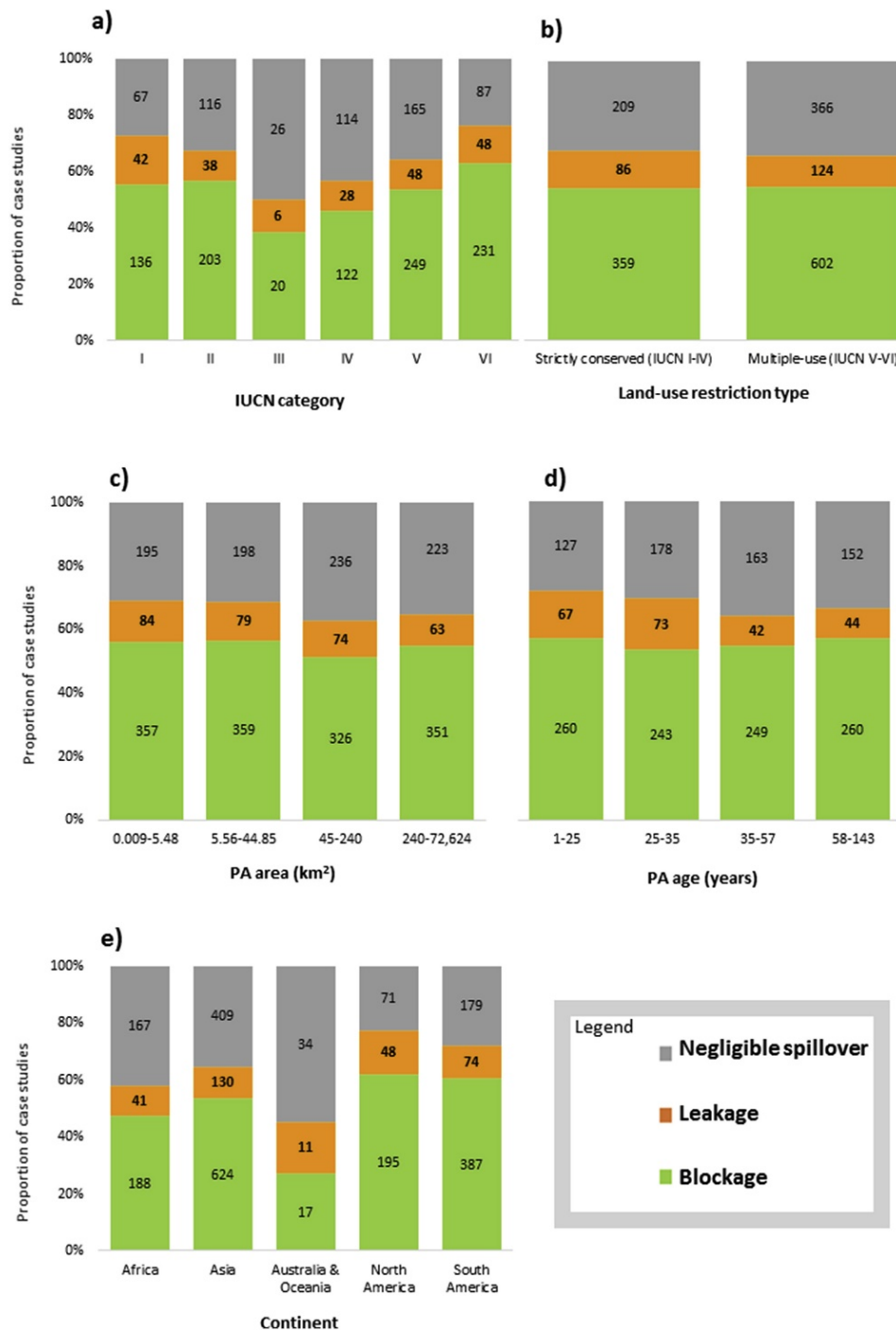


Figure 2.5 Proportional distribution (and number per category) of spillover outcomes across PA-specific factors and continents. Results were broadly similar across: a) IUCN categories, b) land-use restriction types, whereby land use restriction is strict in PAs of IUCN categories I-IV and some degree of land use and resource extraction is allowed in PAs of IUCN categories V and VI, c) PA size quartiles, d) PA age quartiles, and e) continents. Each bar graph does not necessarily show the results for all 2,575 PAs (grand total assessed) because, for example, IUCN category was reported or found for only 1,746 of the PAs; PA size was reported for only 2,545 PAs; the year of establishment was reported or found for only 1,816 PAs.

Drivers of spillover

I identified 165 leakage observations for which all the chosen predictor variables were known (complete cases). The best-selected model had a structural goodness-of-fit of 14.4 %DE (percent deviance explained) and included six of the potential predictors (Table 3; 'Leakage - Complete cases of 15 variables'). After removal of two data-poor predictors (PA age at time of deforestation monitoring and land-use restriction type), the best model (from 395 observations), with 13.4 %DE, included just two coefficients (Table 3; 'Leakage - Complete cases of 13 variables'). Leakage magnitudes were positively correlated with population growth rate, the ratio of the percent of the country under agricultural cover to the percent under forest cover, and the total and per-hectare values of a country's forest products removed annually. Leakages were negatively correlated with national population density, rural population growth rate, and GDP annual growth rate.

Table 2.3 Predictors included in the best-selected generalised linear models of leakage and blockage magnitudes. Colour-shaded fields and the + and – symbols indicate the variables included in each model and direction of their effect.

		Leakage		Blockage	
		Complete cases	Complete cases	Complete cases	Complete cases
			2		2
Number of observations		163	390	549	1,442
Percent of deviance explained		28.7	11.7	16.8	15.4
PA area				–	
PA age upon deforestation monitoring			N/A		N/A
Land-use restriction type			N/A		N/A
Population density		–	–	–	–
Population growth rate					
Rural population growth rate					
GDP		–		+	+
GDP growth rate				+	+
Ratio: % land area under agriculture to forest				–	
Value of forest products removed		+		–	
Value of forest products removed per hectare		+	+	+	+
Carbon dioxide emissions				–	–
Percentage of energy supply from biofuels					–
Corruption perceptions index				–	
Annual conservation spending				+	
Africa					
Asia				+	
Continent	Australia & Oceania			–	–
	North America				
	South America			+	

For the 549 observations of blockage where all predictors were available, the best-selected model had 17.3 %DE and included six predictors (Table 3; ‘Blockage – Complete cases with 15 variables’). The removal of the same two data-poor variables as for leakage enabled modelling of 1,443 blockage observations, with a 15.9 %DE for the best model and ten predictors (Table 3; ‘Blockage- Complete cases with 13 variables’). Blockage was positively correlated with population growth rate, GDP, the ratio of the percent of the country under agricultural cover to percent under forest cover, the per-hectare value of forest products removed in the country annually, and whether a PA was located in Asia or in Australia and Oceania. They were negatively correlated with national

population density, rural population growth rate, GDP growth rate, annual carbon dioxide emissions, and a country's annual conservation spending.

The only predictor common to all four best-selected models was the associated country's population density. All showed >13 %DE of their respective data sets (see Appendix A, S7), and all included only national-scale factors. None of the four best models included coefficients for PA area, PA age, land-use restriction type (derived from IUCN category), the percentage of a country's energy supply from solid biofuels, or PA location within Africa, North America, or South America.

Discussion

The conservation-science literature has recognised the possibility and implications of spillover effects, but few studies have attempted to quantify them. My collated PA assessments, despite being the result of an exhaustive search, constituted <2% of the global PA network (UNEP-WCMC, 2016). Results from these 3,398 case studies affirmed the generally accepted view that most PAs reduce deforestation effectively within their borders relative to the wider landscapes in which they are embedded. Although I also identified some PAs that experienced higher deforestation within compared to without (the surrounding baseline), a more detailed case-by-case assessment is needed to draw firm conclusions about these possible “paper parks,” as prohibition of forest clearing might not have been the goal of protection (Appendix A, S4). Importantly, blockage occurred almost five times more frequently than leakage outcomes.

Because leakage comprises one pillar of the “unholy trinity” of challenges facing efforts to reduce deforestation (Van Oosterzee et al., 2012), I sought to characterise leakage-prone PAs by determining how features like IUCN category, size, and age since gazettal drove spillover direction towards either blockage or leakage. I was unable, however, to identify any consistent characteristics of highly leakage-susceptible PAs: the three outcomes—blockage, leakage, or negligible spillover—were distributed without apparent pattern across IUCN categories, land-use restriction types, size quartiles, and age quartiles (Fig. 2.5). I also expected that one or more of these intrinsic PA characteristics might influence spillover magnitude. Instead, my modelling pointed to national-scale socioeconomic and political factors as being most relevant to spillover magnitude, for both leakage and blockage (Table 2.3). There is a logic to this outcome: displacement of human development pressure by land-use restriction is, after all, a socioeconomic phenomenon (Renwick et al., 2015), albeit one with ecological consequences. Attempts to predict spillover direction and magnitude for proposed or recently established PAs might therefore necessitate interdisciplinary partnerships

between social scientists, economists, and natural scientists (Miteva et al., 2012) and assessment of potential predictors not considered in this study.

Our modelling results challenge the assumption of a trade-off between predictors associated with high leakage and low blockage values, or vice-versa (Table 3). None of the potential predictor variables demonstrated a relationship to leakage and to blockage indicative of a trade-off: PAs located in countries with higher population densities and higher rural population growth rates were associated with both lower leakage and lower blockage magnitudes (i.e., smaller-magnitude spillovers in both directions). PAs located in countries with higher population growth rates, higher ratios of land under agriculture compared to forest cover, and higher value-per-hectare forest products were all associated with both high leakage and blockage magnitudes (i.e., larger-magnitude spillovers in both directions). Case-specific analyses incorporating a socio-ecological framework (Ostrom, 2009) might be necessary to explain these complex dynamics.

Whilst significant uncertainty remains as to the magnitude and ubiquity of the leakage problem on global, national, and ecoregional scales, my results are promising and suggest that, at least in PAs characterised by subtropical and tropical moist forest cover, PA assessments should recognise the possibility not only of a problematic leakage outcome in the UAS, but also of a beneficial blockage outcome. Future studies could seek evidence to determine whether blockages are indeed generated by nearby protection changing perceptions of the development potential of the UAS, perhaps influenced by ecotourism (García, 2015). There is clearly a need for more case studies of the non-target impacts of land-use restrictions associated with PA creation, and future spillover analyses should seek to calculate the most meaningful baseline possible. My results were mostly extracted from deforestation data reported by one study (Spracklen et al., 2015) for which the wider landscape samples were arbitrarily chosen (Fig. 2.2b) and for which only PAs characterised by subtropical or tropical moist forest cover were assessed. Because the samples of the wider landscape (Fig. 2.2) did not necessarily experience the same likelihood of being deforested as the associated PA or the UAS experienced, and because the PA and UAS may have been less suitable for clearing due to slope, elevation, or some other factor (Joppa and Pfaff, 2009), it is possible that my impact calculations overstate the beneficial effect of protection and/or of nearby protection. Baseline estimates can be improved using statistical tools like propensity score matching (Andam et al., 2008; Chomitz, 2002; Joppa and Pfaff, 2010) and panel regression techniques (Jones and Lewis, 2015). Although my systematic literature search did identify a few studies that implemented propensity score matching methods, these studies did not meet the data-extraction criteria.

A baseline deforestation rate could also be derived temporally (Oliveira et al., 2007) rather than from samples of the wider landscape during the same period, although this involves potentially problematic assumptions due to the often non-linear dynamics of land-use change pressures over time. In the case of a temporally derived baseline, the deforestation rates in the PA and in the UAS prior to PA gazettal constitute the counterfactual expected deforestation rates. Comparison of temporally and spatially derived baselines was not possible in this review because the identified records did not report deforestation data for the period before PAs were created. A difference-in-difference baseline-estimation method (see Fig. 3 in Pattanayak, 2009), combining both the spatial and temporal means of informing the baseline estimate, offers a potentially useful way in which future spillover analyses might calibrate baselines. Further, I recommend that spillover monitoring begin when a PA is established and continue over longer durations than those typically reported in this review, as there is likely to be a time lag between protected status designation and the displacement of land-use pressures. My sample did not provide enough variation in duration of deforestation monitoring to model its importance; all the case studies extracted in this review had 12 or fewer years of deforestation monitoring. Longer-term studies, beginning the year of PA establishment, would be ideal for clarifying the spatio-temporal displacement of land-use pressures.

In this review I quantified PA impacts using the metric of area deforested over time, but I acknowledge that human land use is more complex than the binary presence or absence of forest clearing. Furthermore, non-forest environments are also home to much of the world's biodiversity, and these habitats are also being degraded and destroyed due to the expansion and intensification of human land uses (Bond and Parr, 2010). Spillover analyses, and indeed PA assessments that reflect their net impact on the entire landscape, from a variety of perspectives and values, should reach beyond forested areas. Ideally, they should seek to capture the graded intensities of anthropogenic landscape modification using spatially explicit land-use and land-use-change (LULC) data, which is increasingly available from government initiatives such as Europe's Coordination of Information on the Environment (CORINE) and the United States Geological Survey. In addition to land-use restrictions by public PAs, protection associated with conservation covenants on private lands, with indigenous land governance, and with community managed forests, could also generate measurable spillover effects that may warrant quantification, collation, and meta-analysis.

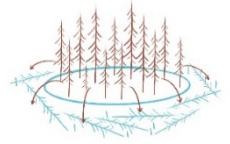
Following identification of spillover effects around areas of land-use restriction, it is essential to then determine their biological and ecological consequences. As noted by Miteva et al. (2012), the degree to which measures of PA impacts on land-cover dynamics are biologically and ecologically useful is determined by the degree to which land cover is a good proxy for species richness and ecosystem function. Spatially explicit biodiversity data, like that provided by the IUCN for red-listed

species' ranges (<http://www.iucnredlist.org>), could be employed to this end. The increasingly common practice of lodging such data in open-access repositories online (Ondei et al., 2018) will facilitate meta-analyses of the impacts of land-use restrictions on threatened species and habitats. Future studies may assess the relative strength of the mechanisms through which human land uses outside of PAs influence ecological function within them in terms of the effective size of the ecosystem being protected in part by the PA, flows of ecological process zones, crucial habitats, and exposure to humans at reserve edges (Hansen and DeFries, 2007).

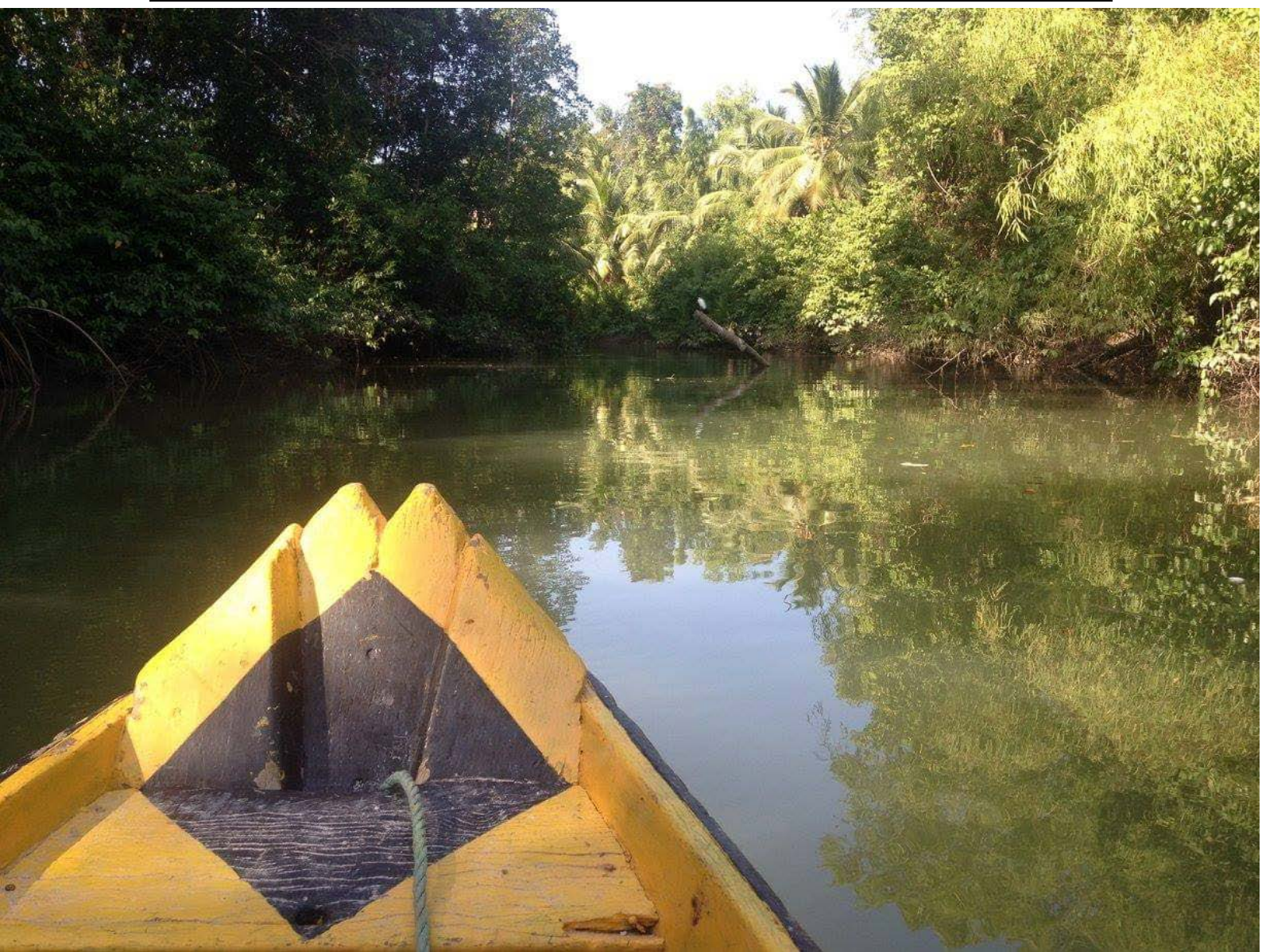
Finally, this review focussed on local spillovers, sometimes called 'neighbourhood spillovers', which although poorly understood are being addressed with increasing frequency in the conservation-science literature. Although difficult, future studies should also look beyond the problem of local leakage to track cross-biome and transnational leakages (Gan and McCarl, 2007; Meyfroidt et al., 2013). Cross-biome leakage might have occurred, for example, when sugarcane expansion in the state of Sao Paulo, Brazil affected land use in distant and disjunct Amazonas, due to the consequent displacement of cattle ranching (De Sá et al., 2013). Indeed, Dou et al. (2018) found evidence that reduced deforestation in the Amazon Forest has come at the cost of increased deforestation in the Cerrado savanna. In this context, implementation of a tele-coupling framework (Bruckner et al., 2015) might be necessary to track non-local leakages and thus, ultimately, to balance trade-offs between development and conservation goals on a global scale.

Conclusion

Although the published global-ecology literature mostly discusses terrestrial spillovers in terms of the potential for leakages, I identified blockage as the most common outcome. It is promising that deforestation leakages were the least frequent consequence for the 2,575 PAs whose non-target impacts were assessed in this review. Nonetheless, PA spillover analyses ought to be done routinely given how critical they are for landscape-scale PA impact assessments and for monitoring the consequences of opportunistic, rather than systematic, protection of lands (Renwick et al., 2015). The goal of expansion of the global PA network is codified into international law by means of Target 11 of the 2010 Convention on Biological Diversity, but simply adding more hectares under protected designation does not necessarily translate to positive impacts for habitat and biodiversity, if they act to displace threats to areas where biodiversity is more highly vulnerable and irreplaceable. This should be a key focus of research as new regions are gazetted for conservation.



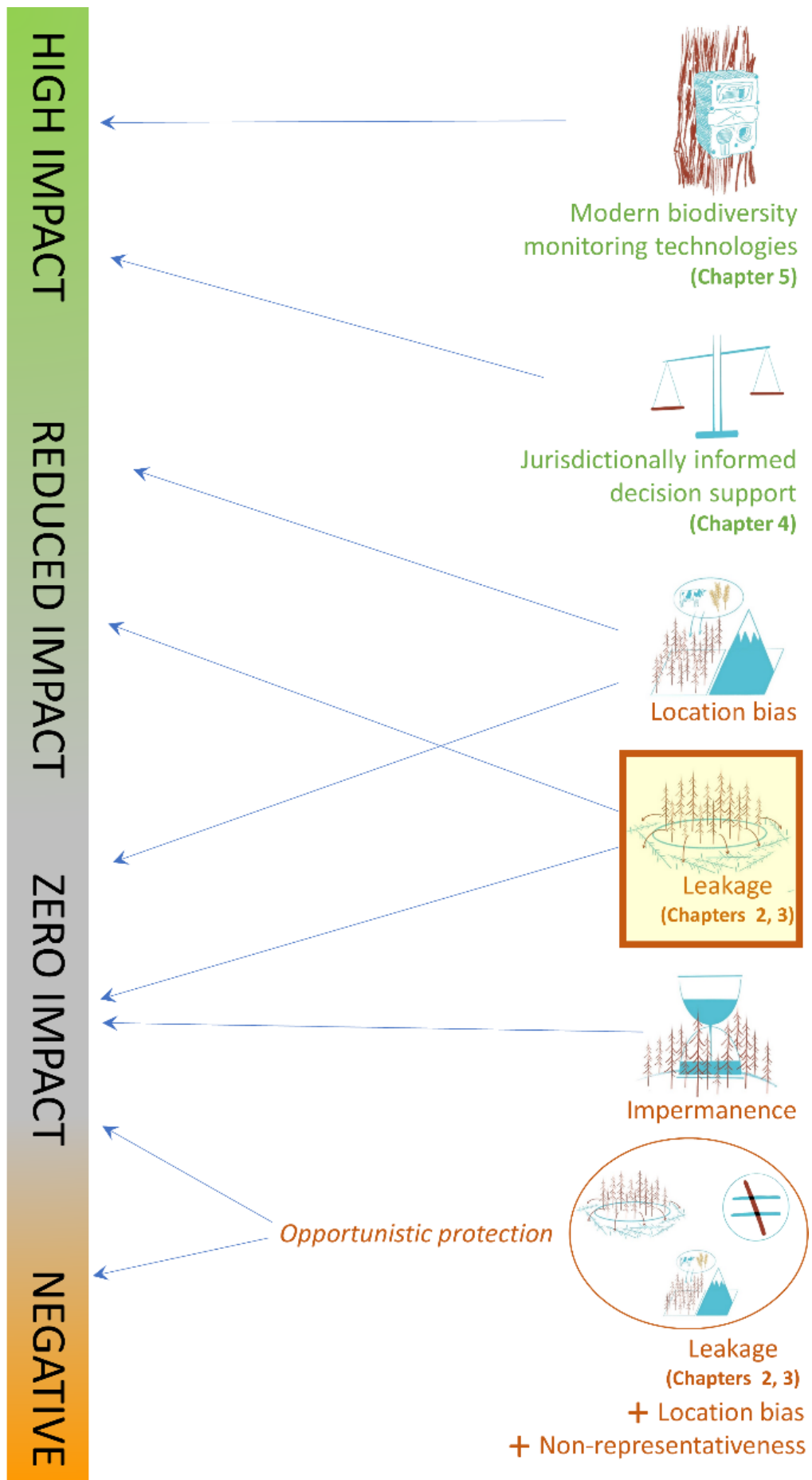
Chapter 3: Assessing the magnitude of leakage from recently established protected areas in the Brazilian Amazon.²



Marajó Archipelago, gateway to the Amazon River, 2016

Rubber tappers, ribeirinhos (riverside dwellers), and other Indigenous groups have occupied the forest for many years. They never threatened her. The real threat are the agribusiness projects, large-scale loggers and hydroelectric dams with their floodings. — Chico Mendes

² Published as a peer-reviewed article: **Fuller, C., Ondej, S., Brook, B. W., & Buettel, J. C. (2020).** Protected-area planning in the Brazilian Amazon should prioritize additionality and permanence, not leakage mitigation. *Biological Conservation*, 248, 108673.



Abstract

The establishment of protected areas (PAs) where human land uses are restricted, especially in Amazonian forests, is an important bulwark against global biodiversity collapse and climate change. Here I assess first if recently established PAs have reduced deforestation within their boundaries, and second if restriction of land clearing displaced deforestation pressure to adjacent forests, a spillover effect called leakage, or if it further inhibits deforestation around PAs, a spillover effect called blockage. Leakage can undermine the net impact achieved by PA establishment. I calculated the non-target and target impacts of 91 state-governed (non-indigenous) PAs established between 2005 and 2016 on satellite monitored deforestation rates. I then assessed how well the features of PAs, like size and restriction level, explain the spillovers. On average, 2% or less of the assessed spillover zone was deforested more extensively (leakage) or less extensively (blockage) than the area expected to have been deforested without nearby PA establishment. Blockages outnumbered leakages regardless of the method adopted to calculate the deforestation baseline rate. My results suggest the establishment of PAs has inhibited deforestation both within their boundaries and in their adjacent surroundings. The 17 PAs that did not inhibit deforestation within their boundaries and the 19 PAs where leakage occurred were geographically concentrated in southern and western protected regions of Amazônia Legal. High-magnitude leakage of deforestation pressure from PAs to their adjacent surroundings seems therefore to be rare in the Brazilian Amazon. For the most effective conservation of Amazonian forests, priority should be given to achieving additionality with any new PA and ensuring the permanence and enforcement of those already established.

Introduction

The global protected area (PA) estate is an important contributor to both biodiversity conservation and climate change mitigation goals, especially in the humid tropics (Nelson and Chomitz, 2011). PAs are the principal defence of animal habitat as global mass extinction unfolds (Pimm and Raven, 2000), and safeguarding forests from unrestricted human encroachment and development is an effective, low cost, natural climate change mitigation strategy (Griscom et al., 2017). While the conservation impact of PA establishment within the target area is presumably positive (Nagendra, 2008; Naughton-Treves et al., 2005), the associated impact on its surroundings (i.e., the non-target impact, or spillover) may be negative, positive, or negligible; both the target and non-target impacts are components of the net impact of establishing a PA within a landscape (Bode et al., 2015; Ewers and Rodrigues, 2008; Renwick et al., 2015).

In the case of “leakage,” the negative type of spillover, clearing or other land-use change that is restricted in the target area is displaced to an area where it would not otherwise have occurred (Aukland et al., 2003; Ewers and Rodrigues, 2008). Leakage can undermine or entirely offset PAs' net impact for conservation (Bode et al., 2015; Renwick et al., 2015) and carbon sequestration (Ostwald and Henders, 2014), whilst falsely exaggerating assessments of PA effectiveness and accelerating the rate at which PAs become isolated and the landscape becomes fragmented (Ewers and Rodrigues, 2008), shortening the window of opportunity to achieve a representative and well-connected PA network (Fearnside, 2009). Further, if clearing is displaced from locations opportunistically protected, i.e., under protection because of their remoteness and/or low economic value, rather than their environmental value (Pressey et al., 1993), to other areas containing more highly irreplaceable and vulnerable habitat or greater carbon storage capacity, PA establishment could do more harm than good (Bode et al., 2015; Renwick et al., 2015). Despite these concerns, leakages from PAs have rarely been systematically quantified (Fuller et al., 2019). Leakage monitoring and mitigation should be a component of a global push to target quality over quantity in PA target development, as leakage is amongst the perverse outcomes of prioritising area-based targets (Barnes et al., 2018).

The beneficial type of spillover, termed “blockage”, can also occur, whereby conservation impact is achieved beyond the legal boundary of a PA. In terrestrial forested ecosystems, blockage may occur if deforestation is discouraged by the presence of a nearby PA, perhaps due to a flourishing ecotourism industry or reduced incentive for regional infrastructure investment (García, 2015). Blockage effects may be visible from remote sensed imagery as “haloes” of intact vegetation

around a PA's border, demonstrating that adjacent forest cover has been protected from clearing by de facto rather than de jure means (García, 2015).

Rigorous, counterfactual measurement of spillover, crucial for determining the net impact of PA establishment, can inform systematic conservation planning (Ferraro and Pattanayak, 2006; Margules and Pressey, 2000) by ensuring spatial prioritisation of a PA network that is the best possible bulwark against the planetary scale problems of biodiversity collapse and climate change. Yet, in spite of the recent availability of widespread spatially explicit, multi-temporal deforestation and land use/land-use change data, spillover effects are infrequently accounted for in PA assessments (Fuller et al., 2019; Miteva et al., 2012; Pfaff and Robalino, 2017). As such, little is known about why some land-use change restrictions, like those assessed in Peru (Oliveira et al., 2007), have generated leakage whilst others, like those assessed in Costa Rica and Ecuador, have generated either negligible spillovers or blockages (Andam et al., 2008; Robalino et al., 2017; Tapia-Armijos et al., 2017). An understanding of the prevalence, direction, magnitude, and drivers of past spillovers is critical for predicting and avoiding future leakage events. Such understanding is vital for avoiding scenarios where investments in PA establishment result in only nominal gains for habitat conservation and carbon storage, and for avoiding a worst-case scenario whereby land clearing is displaced from PAs to more vulnerable and irreplaceable habitats or to areas more important for climate change mitigation. To that end, I calculate and analyse the target and non-target impacts of recent PA establishments in an especially biodiversity- and carbon-rich forested region: Brazil's Amazônia Legal.

The PA network of Amazônia Legal is an ideal case study as it is recognised as a globally important area for biodiversity conservation (Mittermeier et al., 2003) and climate change mitigation (Nepstad et al., 2008; Soares-Filho et al., 2010). Brazil's 3201 terrestrial PAs make up one of the largest networks in the world; approximately 30% of the country's area is protected, much of which is located within Amazônia Legal (UNEP-WCMC and IUCN, 2019). Impact assessment that accounts for spillover is possible here because deforestation has been monitored and reported in the Brazilian Amazon since 2005, and PAs in this region have significantly inhibited deforestation within their target areas (Fearnside, 2008b; Ferreira et al., 2005; Nepstad et al., 2006; Vitel et al., 2009). Unlike previous research on deforestation inhibition by PAs (Nagendra, 2008), in this study I aimed to account for deviation of post-PA establishment clearing rates from the baseline, i.e., the deforestation rate that would have been expected without PA establishment, when calculating impact values (Pressey et al., 2017). I counterfactually quantified both components of PAs' net impact on the landscape: (1) deforestation within PAs, and (2) deforestation in the unprotected,

adjacent surroundings. I identified PAs with evidence of leakage or blockage and then tested key PA features for association with the two kinds of spillover.

Methods

Study area

Amazônia Legal is a 5,016,136 km² administrative region of Brazil that includes the entire states of Acre, Amapá, Amazonas, Pará, Rondônia, Roraima and Tocantins, as well as part of Mato Grosso and Maranhão (Fig. 3.1); it is home to a population of approximately 24 million people. As of 2017 Amazônia Legal contained 340 conservation PAs and 459 Indigenous Lands (Fig. 3.1). I assessed the 91 state-governed (non-indigenous) PAs, including 15 privately owned PAs, established during the period 2005 to 2016, which range in size from less than 1 km² to 36,102 km², with a mean size of 4666 km² across the sample (see Appendix B, Table S1). High-resolution deforestation data is not reported prior to 2005, and thus PAs established before 2005 were not assessed. Indigenous lands were not assessed because they may not be managed specifically for deforestation inhibition, although they have been shown to be effective for that purpose (Nolte et al., 2013; Nepstad et al., 2006) as well as being important for vertebrate biodiversity conservation in Brazil (Schuster et al., 2019).

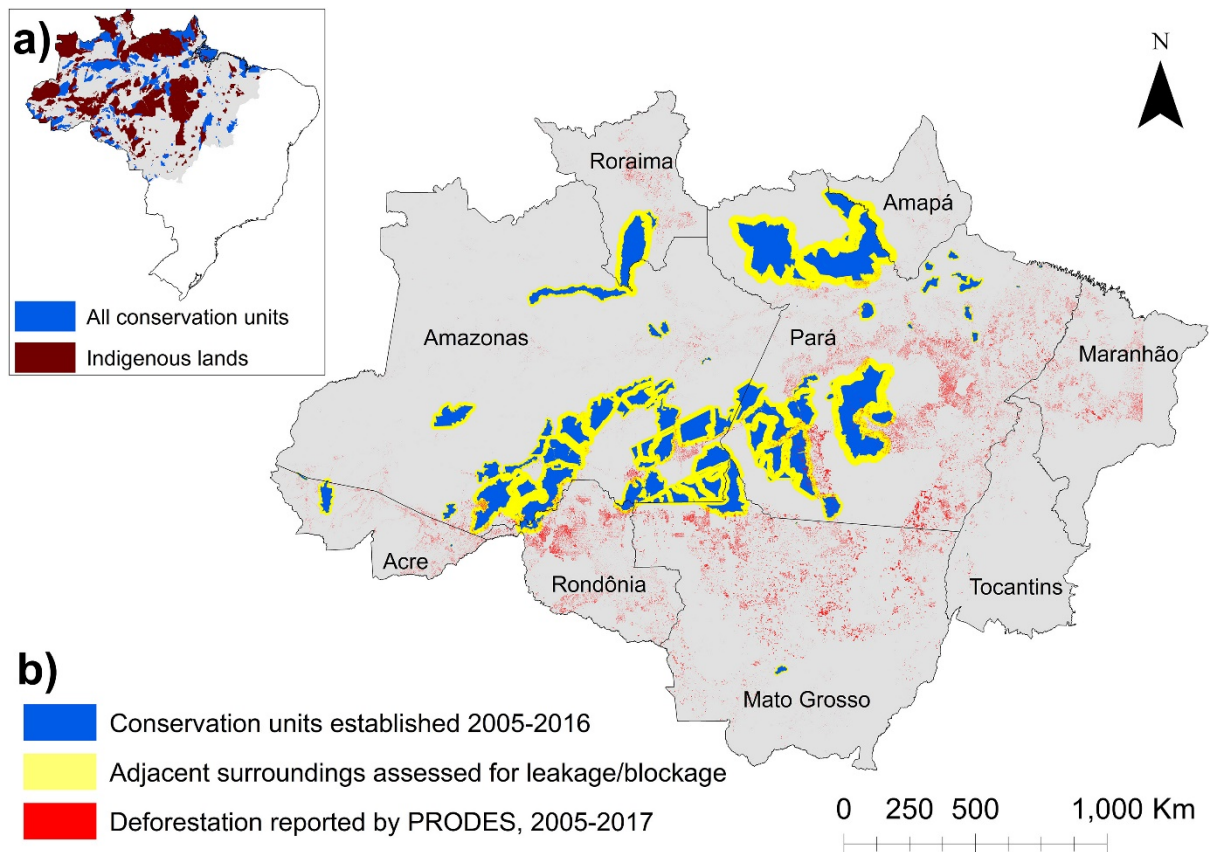


Figure 3.1 The inset map (a) shows the position of the Amazônia Legal (grey) within Brazil and the location of all the protected conservation units (blue) and indigenous lands (brown) established in the region. Indigenous lands were not assessed for impacts on deforestation, but were accounted for (see Appendix B, Table S1) where they overlapped with the spillover zones of the conservation unit PAs assessed. The main map (b) of the study region, Amazônia Legal, shows the 91 PAs that were established within it from 2005 to 2016 (blue), their adjacent zones tested for possible spillover (yellow), and areas of new deforestation reported for each year 2005 to 2017 (red). Government of the region is administered through nine Brazilian states labelled in the main map (b).

Various actors are implicated in the deforestation dynamics of Amazônia Legal, including grileiros or illegal land grabbers, loggers and sawmillers, ranchers and their employees, unorganised landless migrants, organised “sem terra” landless rural workers, agribusiness-scale soy farmers, and miners (Fearnside, 2008a). Land tenure is not always uniformly regulated under the rule of law; and changing commodity markets affect human migration and thus clearing (Fearnside, 2008a). Leakage may occur where small farmers are displaced from newly protected forests or where crop production is displaced and affects supply and demand dynamics (Fearnside, 2009). The type of

leakage whereby *grilagem* (illegal land appropriation and clearing) becomes concentrated in the unprotected portions of forest where land tenure is especially unregulated is considered to be the most significant leakage mechanism within the Amazonian PA network (Fearnside, 2009). Despite the variety of actors involved, PA location within a given state and PA administration by a particular governance tier warrant testing as drivers of deforestation impacts (García et al., 2019; Pfaff et al., 2014). As Amazonian protected areas' forest impacts have been shown to vary significantly with development pressure (Pfaff et al., 2015a; Pfaff et al., 2015b), I assess six possible factors that may contribute to the intensity of deforestation pressure faced by a given area, and Table S2 in Appendix B provides the justifications for their inclusion in modelling the impact of PA establishment on deforestation dynamics.

Data acquisition

I obtained data on the location and extent of new clear-cut deforestation for each year from 2005 to 2017 from the Program for Monitoring Deforestation of the Amazon (*PROgrama de Cálculo do DESflorestamento na Amazônia*, or *PRODES*, in Portuguese) data set, based on 30-m resolution LandSat imagery. The government agency responsible for carrying out PRODES, the Instituto Nacional de Pesquisas Espaciais (INPE), made these data publicly available (<http://www.dpi.inpe.br/prodesdigital/prodes.php>) at the behest of Brazil's Environment Ministry since 2002; this website was discontinued in April 2019. The monitoring program has used a consistent methodology for collecting and reporting these data only since 2005, and consequently, only data from the period since 2005 were used in this study. Because PRODES only detects and reports clear cuts of 0.0625 km² or larger, some smaller scale clearing may be missing from this analysis; notably, PRODES does not distinguish between illegal deforestation such as that carried out by *grileiros*, which all PAs seek to reduce, and the local inhabitants' subsistence tree felling, which may be legally sanctioned in some of the sustainable use units that I assess ($n = 47$). Further, PRODES does not monitor the portion of the Amazon biome that extends beyond Brazil's national borders, and it is possible that some agents of deforestation could have shifted clearing activities to those areas to avoid detection (Richards et al., 2017).

I obtained data on locations and characteristics of all PAs within Amazônia Legal from Brazil's Environment Ministry (<http://mapas.mma.gov.br/i3geo/datadownload.htm>). The geographic location of the PAs, as well as associated metadata including year of establishment, whether the Brazilian PA classification is strictly protected or sustainable-use approved, governing tier (federal, state, or municipal), and total area protected, were cross-checked with similar spatial data provided

by Protected Planet's World Database on Protected Areas (<https://www.protectedplanet.net/>). The 91 PAs (Fig. 3.1) I assessed were chosen because they were established within the spatial and temporal range of PRODES deforestation reporting; that is, they were all established 2005–2016 and are located in the states of Amazonas (n = 38), Pará (n = 33), Mato Grosso (n = 7), Tocantins (n = 5), Acre (n = 3), Rondônia (n = 3), and Roraima (n = 2). No PAs established in Maranhão or Amapá met both my temporal (2005–2016) and spatial (satellite monitoring range) criteria. PA downgrading, downsizing, and degazettement (see Appendix B, Table S1) data for this region was obtained from <https://www.paddtracker.org/>.

Assessment of PAs' target and non-target impacts on deforestation rates

I defined a PA's target impact as the difference between the observed deforestation and the expected deforestation within protection's boundaries; non-target impact is the difference between observed and expected deforestation in the spillover zone, located in the immediately adjacent, unprotected surroundings of the PA. Using QGIS and ArcMap 10.6 software, I defined the spillover zones by creating a buffer around each PA equal in area to the PA (Fig. 3.1); any non-terrestrial portions of the buffers, in the few cases of coastal PAs, were removed. I then calculated the area newly deforested each year within each PA and within each spillover zone; these values constitute observed annual deforestation.

Estimates, for each year, of the amount of deforestation that would have occurred had the PA never been established, both within the PA and in the spillover zone, were ascertained from deforestation that occurred in samples of the unprotected wider landscape during the same year (Fig. 3.2); generation of counterfactual baselines that define the wider landscape are necessary to quantify impacts (Ferraro, 2009; Pressey et al., 2017). I could only use the baseline estimation method suggested by Ewers and Rodrigues (2008) for 73 of the 91 PAs (Fig. 3.2b and Section 3.1 of this chapter), as the remaining 18 PAs were in close proximity, meaning there was not enough 'unprotected land' in their surrounds (Fig. 3.1) from which to sample. As such, I also used a second baseline generation method (Fig. 3.2c, and see Table S3 in Appendix B) to allow for the assessment of all 91 PAs. I attempted to account for variation in likelihood of clearing by using one highly localised baseline to minimise environmental differences, and one state-derived baseline to minimise differences in anthropogenic factors.

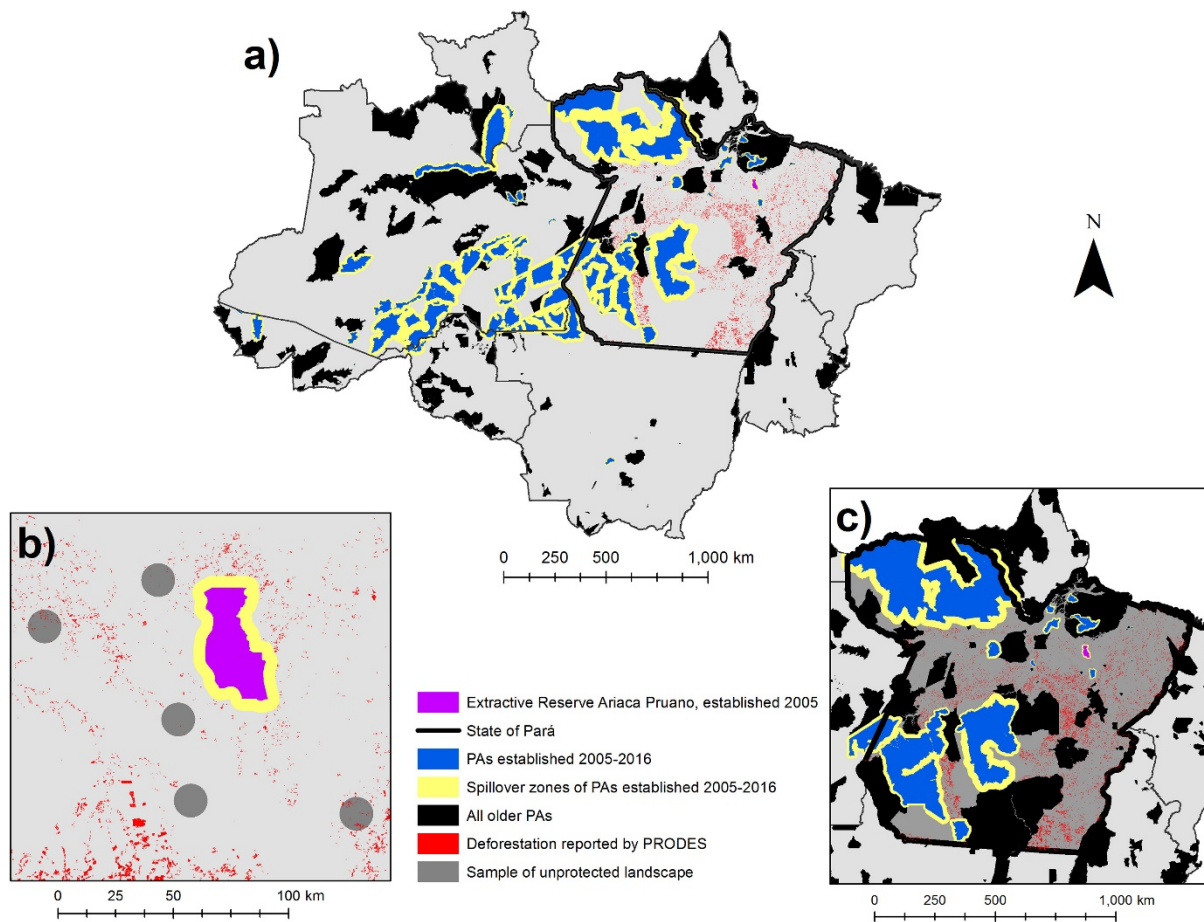


Figure 3.2 Maps of unprotected areas sampled to estimate two baseline deforestation rates for the Extractive Reserve Ariaca Pruano. A baseline is an estimate of the area of new deforestation that might have been expected in the PA and in its adjacent unprotected surroundings (yellow) had no protection ever been declared in the area. Deforestation (red) was reported during the period 2005 to 2016. The main map (a) shows the location of the Extractive Reserve Ariaca Pruano (purple) and the state of Pará (heavy outline) within Amazônia Legal. The left-hand inset map (b) was adapted from the non-target impact assessment method proposed by Ewers and Rodrigues (2008): five randomly generated circular samples (dark grey) of the nearby, unprotected landscape (light grey) were used to estimate expected deforestation for each year in the PA and in the spillover zone. The combined area of the five samples is equal to the area of the PA and thus also equal to the area of the spillover zone. The right-hand inset map (c) shows a novel spatial sample from which an alternative baseline is derived from the entire unprotected portion of the state within which the PA is located, which in this case is Pará.

Estimating locally derived baselines

Following the spillover quantification method suggested by Ewers and Rodrigues (2008), for each of the 73 PAs I generated five random circular sample parcels, located within the unprotected wider landscape but beyond the spillover zone (Fig. 3.2b). The total area of the five sample parcels was equal to that of the associated spillover zone to allow for direct comparison of areas newly deforested each year in the unprotected landscape, in the spillover zone, and in the PA itself. I constrained the random sample parcels to (i) the area within a maximum distance of 10 times their radii from the associated PA's boundary, and (ii) outside the spillover zones of any of the other 90 PAs. I then calculated the area of deforestation reported for each year in each set of landscape samples.

Estimating state-derived baselines

A second set of expected annual deforestation for each PA and associated spillover zone was derived from the percent area deforested each year within the unprotected portion of the state in which the given PA is located (see Appendix B, Table S3). Forty of the 91 PAs I assessed were administered by state governments, and other key deforestation policies in place in the Amazônia Legal have been applied and enforced at a state-by-state level. For example, the 2012 Forest Code legislation states that the percent of landholders' property that can be legally deforested depends on the state within which that property is located (Freitas et al., 2018). Given the state-by-state nature of legal forest conservation requirements, I estimated these novel baselines for each state and assigned them to PAs and their spillover zones according to PA location. For example, the Extractive Reserve Ariaca Pruano was established in Pará in 2005; the percent of the PA deforested, and the percent of the spillover zone deforested during the years 2006 through 2017, were compared with the percent of the entire unprotected portion of Pará that was deforested during the same years (Fig. 3.2c). In the few cases of a single PA being located in more than one state ($n = 5$), the baseline values for each year were derived from the state containing the largest portion of the PA's area.

Impact calculation

Spillover (i.e., non-target impact) was calculated for each PA for each year (see Appendix B, Table S4), starting from the year after the given PA's year of establishment through to 2017. However, any of these spillover calculations were not analysed if the associated PA did not impact deforestation within its boundaries. I obtained a net spillover value for each PA by using the following formula:

$$I_{total} = \sum_{i=1}^n (O_i - E_i)$$

where I_{total} is the summed impact of all n years of the given PA's existence through to 2017, O_i is the observed area (km²) of new deforestation in the i^{th} year since the given PA's gazettal, and E_i is the counterfactual estimate of the area (km²) of new deforestation expected in the i^{th} year had no PA ever been gazetted.

A negative spillover value, indicating that there was less deforestation observed than expected, was interpreted as a possible blockage effect; conversely, a positive spillover value indicated possible leakage (Fig. 3.3). Blockage observed in some years could have offset leakage observed in other years, and vice versa. Note that some PAs' (those established in 2005) spillover results were informed by 12 years of deforestation monitoring, whereas others (established in 2016) were only informed by a single year of monitoring.

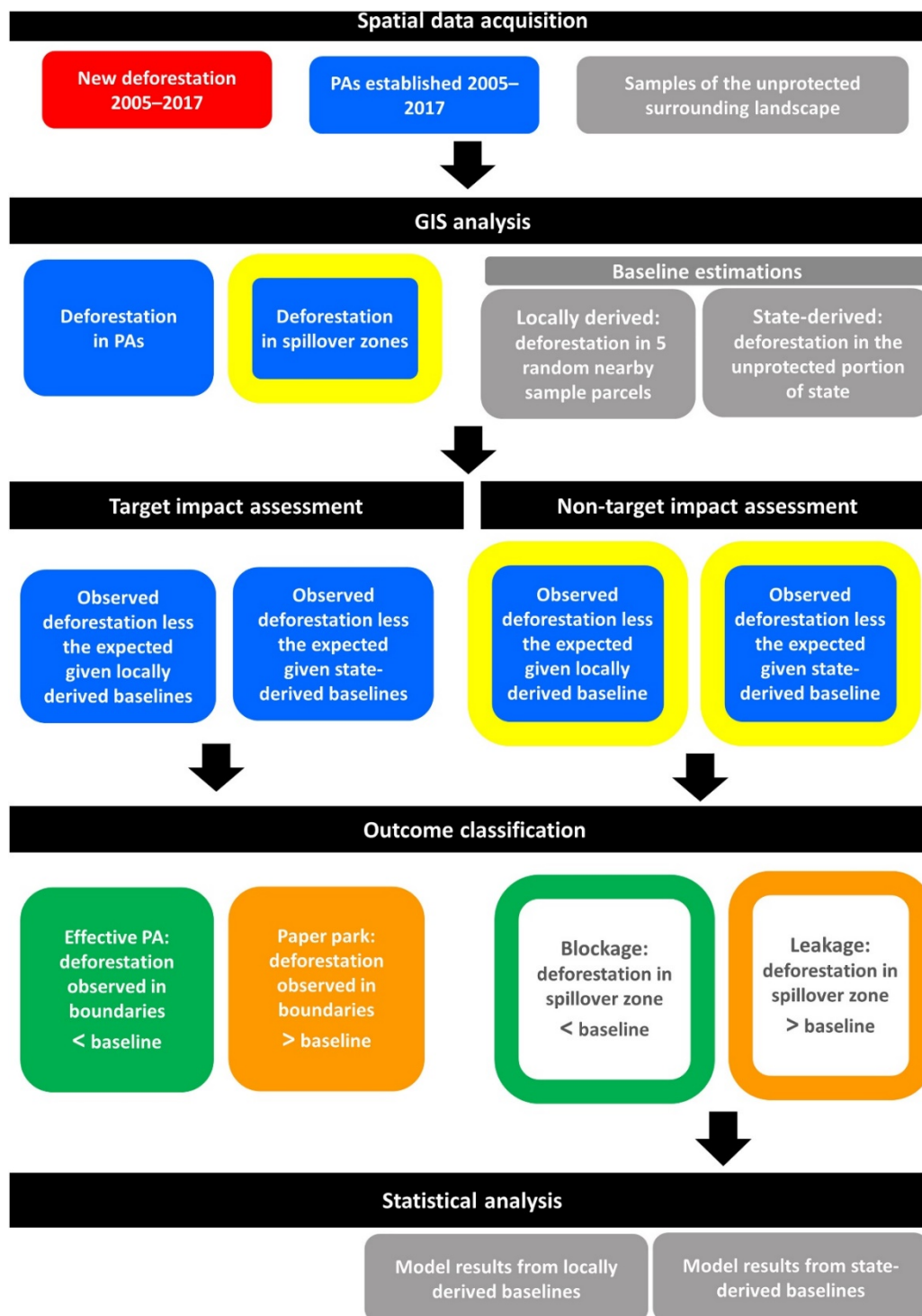


Figure 3.3 Sequence of methods used to acquire relevant data, to assess the target (blue) and non-target (yellow) impacts of PA establishment, to classify PAs as leakage generating (orange) or blockage generating (green), and finally to analyse the results of the non-target impact assessments. The impact assessment, outcome classification, and statistical analysis steps were performed for two data sets, one generated using locally derived baselines (73 spillover results) and the other generated using the state-derived baselines (91 spillover results).

An analogous calculation was performed to assess whether each PA achieved a beneficial target impact, i.e., the area inside protection's boundaries experienced less deforestation than expected, or conversely if the PA may be a “paper park” (Di Minin and Toivonen, 2015). Paper park status is suggested if protection did not reduce deforestation compared to the baseline expected rate.

Evaluation of potential predictors

I used generalised linear modelling to gain insight into possible drivers of the direction and magnitude of the spillover results obtained. Six predictors were chosen, five of which were features of the PAs: age, area, state, governing tier (federal/state/municipal), category within the Brazilian PA classification system (see Appendix B, Table S1 for PA characteristics and Table S2 for rationales for their inclusion). The sixth predictor was a characteristic of the spillover zones rather than of the PAs themselves: the percent of the spillover zone that overlaps with other PA(s) or with Indigenous Lands (ILs). It might be expected that high percent overlap would bias spillover calculations towards what would otherwise appear to be blockage-directional effects. I calculated overlap both with other conservations units and with ILs because, in the Brazilian Amazon, ILs have achieved significant forest conservation impact within their boundaries (Nepstad et al., 2006; Nolte et al., 2013) and as such should be considered as equivalent to conservation units in terms of possibly influencing deforestation dynamics in the spillover zones.

All possible combinations of potential predictor variables were tested using the `leaps` library (Lumley 2017) in R (R Core Team 2018) with k-fold ($k = 10$) cross-validation (James et al., 2013) to select the model, and thereby identify the most important variables related to spillover magnitude across PAs. After removing observations for which the PAs' did not have a deforestation-inhibiting impact within their boundaries, and thus were considered unlikely to have displaced deforestation pressures to their unprotected surroundings, I fitted a model to the remaining net spillover values calculated with the locally derived baseline and repeated this process for the spillover values calculated with state-derived baselines.

Results

Target-area impacts of PA establishment

For the subset of 73 PAs for which a locally derived baseline could be quantified, impact assessment within PA boundaries showed tangible forest conservation benefits. Satellite monitoring by PRODES did not report deforestation in nor around 17 PAs, and as such their impact could not be quantitatively assessed. Less deforestation was reported inside 43 of the remaining 56 PAs assessed with the locally derived baselines (Fig. 3.4a), suggesting successful protection. For these 43 cases with evidence of successful protection, the net differences between expected and observed deforestation within PA boundaries by 2017 ranged from less than 0.1 km² to 1,250 km²; the mean was 127 km² less deforestation than would be expected without protection. As percentages of the PA's extent, avoided deforestation ranged from close to 0% to more than 9% with a mean of 2.19%.

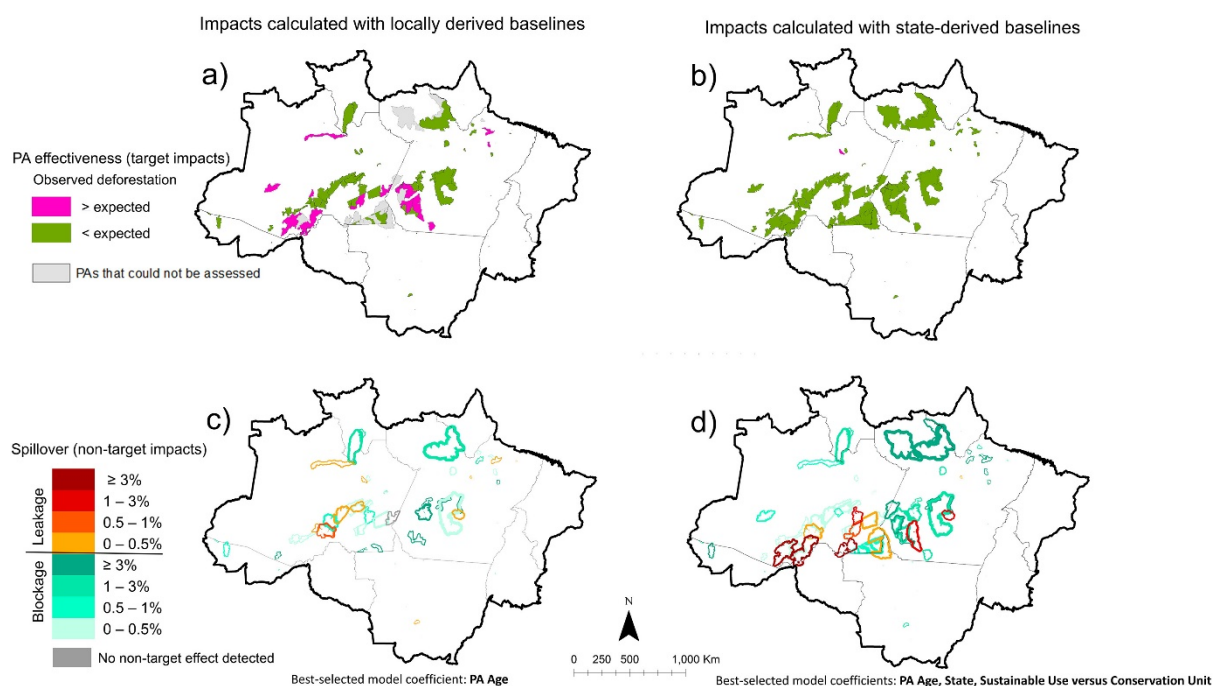


Figure 3.4 Maps of colour-coded target and non-target (i.e., spillover) impacts identified after the establishment of 91 PAs in the Amazônia Legal since 2005, showing the impact of protection within the PAs as calculated (a) with locally derived baselines, and (b) with state-derived baselines. The impact of protection in the local spillover zones was also calculated (c) with locally derived baselines, and (d) with state-derived baselines. Spillover results are not shown for those PAs that experienced more deforestation within their boundaries than expected, as these PAs were unlikely to have influenced their surroundings if protection did not influence their target areas.

Assessment with the locally derived baselines indicated that only 13 PAs failed to inhibit deforestation (Fig. 3.4a). Within the boundaries of these possible paper parks (Di Minin and Toivonen, 2015), the area deforested over the baseline expectation ranged from less than 1 to 434 km², with a mean value of 55 km². As percentages of the paper parks' areas, the excess deforestation ranged from less than 0.01% to 3.33% with a mean of 0.60%. Within two of these PAs, both located in the state of Pará, excess deforestation was of relatively high magnitude: National Forest Jamanxim, established in 2006, experienced clearing of 3.34% of its area above the expected rate; Biological Reserve of the Springs of the Cachimbo Mountains, established in 2005, saw clearing of 2.53% of its area above the expected rate.

When all 91 PAs target impacts were assessed using the state-derived baseline, only the Sustainable Development Reserve of Rio Negro in Amazonas, established in 2008, was flagged as a possible paper park (Fig. 3.4b): it experienced 1.8 km² more deforestation inside its boundaries than expected. The remaining 90 PAs were all calculated to have experienced less deforestation than expected given the state-derived baseline calculated.

Most PAs' target impacts were greater than or equal their non-target impacts (see Appendix B, Table S4).

Non-target area impacts of PA establishment

Overall, blockages outnumbered leakages regardless of the method adopted to calculate the deforestation baseline rate (Table 1). There were only six discrepancies in net spillover outcome (excluding spillovers calculated for paper parks) between the two methods: in five of the six cases, the locally derived baseline calculation yielded a net leakage result for a PA for which the state-derived baseline calculation yielded blockage; only Acari National Park yielded the opposite type of outcome discrepancy (see Appendix B, Table S4). The first five cases may be explained by the bias of the state-derived baseline method towards indicating blockage rather than leakage. That is, because at least some deforestation occurred in every Amazonian state each year 2006 to 2017 (see Appendix B, Table S3) state-derived baselines expect some deforestation to occur in all PAs' target and spillover zones.

Table 3.1 Spillover calculated for 91 PAs.

	Locally derived baseline		State-derived baseline method	
	Number of PAs	Mean annual deforestation (% of PA) above or below baseline rate	Number of PAs	Mean annual deforestation (% of PA) above or below baseline rate
Leakage	9	0.240%	11	1.747%
Blockage	34	-2.019%	79	-1.690%
Not possible to calculate	18	NA	0	NA
Paper park - spillover irrelevant	30	NA	1	NA

Spillover outcome types grouped by PA characteristics

See Appendix B (Fig. S1) for the proportional distribution and number of blockages/leakages by predictor. For those PAs where a net leakage effect was identified by 2017, I observed an increasing trend of (non-cumulative) annual leakage magnitudes, regardless of the baseline method used (Fig. 3.5). Conversely, for those PAs where a net blockage effect was detected by 2017, no clear temporal trend was observed (Fig. 3.5). The lack of reported deforestation in and around the PAs that were typically more recently established, small, or municipally administered (see Appendix B, Fig. S1) prohibited calculation of either type of spillover using locally derived baselines for those areas. Regardless of the baseline used in calculation, PA age, size, state, governmental tier, or land-use restriction intensity as denoted by the Brazilian PA classification system, blockage outcomes far outnumbered leakage outcomes (see Appendix B, Table S4 and Fig. S1). Leakages were slightly more prevalent in the subset of PAs in the largest size quartile group (see Appendix B, Fig. S1b).

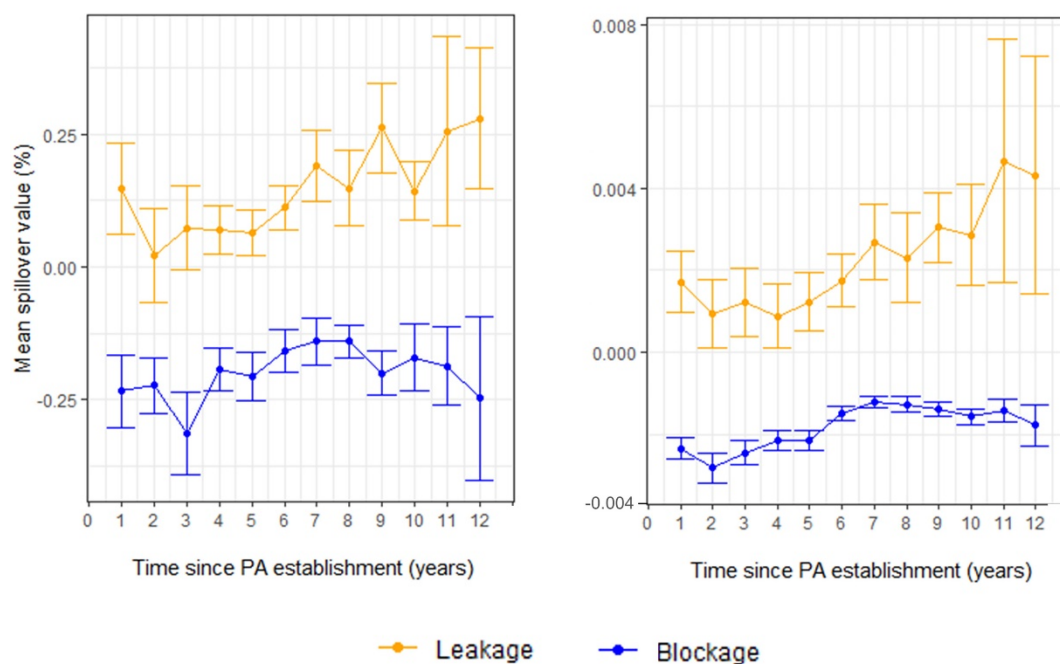
a) Locally derived**b)** State-derived

Figure 3.5 Plot of mean annual spillover value per PA age class, using both (a) locally derived baselines and (b) state-derived baselines, with the net outcome of either leakage ($n = 9$ and $n = 11$, respectively) in orange or blockage ($n = 34$ and 79 , respectively) plotted separately. The error bars show the standard error of each mean. Note the difference in the y-axes' scales between plots (a) and (b).

Results of model selection with k -fold cross-validation

The best-selected model of the 43 net spillover values associated with PAs effective within their boundaries, calculated with locally derived baselines, had a structural goodness-of-fit of 14.1%DE (percent deviance explained) and included one of the potential predictors (see Appendix B, Fig. S2a) as a coefficient: PA age (coefficient = -0.0022 , p -value = 0.0217). Older PAs were associated with more negative spillover values, i.e., blockages.

The best-selected model of 90 net spillover values calculated with state-derived baselines had a structural goodness-of-fit of 33.6%DE, and it included three significant coefficients (see Appendix B, Fig. S2b): PA age (coefficient = -0.0014 , p -value = 0.0026), PA type in the Brazilian classification system (coefficient = -0.0080 , p -value = 0.0297), and state (coefficient = 0.0177 , p -value = 0.0428) where only Amazonas was significant. Older PAs and the sustainable use PAs, i.e.,

those where some resource extraction is allowed under Brazilian law, were significantly associated with negative model coefficients, i.e., blockage outcomes, in the best-selected model. PA location within the state of Amazonas was significantly associated with a positive model coefficient, i.e., leakage outcomes.

In summary (see also Fig. 3.4), model selection with k-fold ($k = 10$) cross-validation indicated that PA age was negatively correlated with spillover value and thus implicated in blockage-directional outcomes, regardless of which of the two baseline estimation methods was used to generate the spillover values modelled. Neither the PA's governing tier (federal, state, or municipal) nor the percent of the spillover zone overlapped by other PAs or ILs was an important predictor of the spillover value calculated (see Appendix B, Table 3.2 and Fig. S2).

Discussion

A growing body of literature calls for explicit spillover accounting in conservation policy impact assessments (Ewers and Rodrigues, 2008; Miteva et al., 2012; Pfaff and Robalino, 2017). Even if only legally protected forests are likely to remain in the Amazônia Legal in the long term, e.g., 100 years from now, leakage from PAs established now still poses a problem: leakage reduces the temporal 'window of opportunity' to establish a biologically representative PA network with maximised connectivity within the remaining unprotected landscape (Fearnside, 2009). That crucial window of opportunity is reduced in a 100% leakage scenario, whereby displaced deforestation pressure causes clearing to occur in an unprotected area equal in size to the area under protected status; a 0% leakage scenario does not reduce that window of opportunity (Fearnside, 2009). Given that the mean value of the net leakages I detected was less than 2% of the associated PAs' areas, regardless of the baseline used in calculation, leakage in this region and during this period is unlikely to have shortened the window of opportunity for designing and building an optimised PA network in Amazônia Legal before a future equilibrium state of forested and cleared land is reached. Indeed, detection of leakage was less frequent than detection of blockage, regardless of the baseline used to calculate impact. Model selection analysis showed that older PAs were more likely to be associated with net blockage outcomes by 2017, but it is possible that my sample of 91 recently established Amazonian PAs may not be representative of spillover dynamics of yet older PAs.

The results from my sample of PAs nonetheless further affirm the beneficial target impact of recent PA establishments in Amazônia Legal for forest conservation. Congruent with past studies (Nagendra, 2008), my assessment of the areas within PAs' boundaries detected overwhelmingly

more evidence of effective deforestation inhibition than evidence of paper parks, regardless of which baseline was used to calculate impact. Given the general effectiveness of these PAs to inhibit deforestation within their boundaries, it is important that they not be downgraded, downsized, or degazetted: PA downgrading, downsizing, or degazettement (Kroner et al., 2019; Mascia and Pailler, 2011; Pack et al., 2016) was proposed (but not enacted) at some point after the establishment of 13 of the 91 PAs assessed. Two PAs, Floresta Estadual do Paru and Parque Nacional Mapinguari, saw downsizing proposals enacted in 2012.

The few possible paper parks, i.e., those PAs associated with being ineffective within their boundaries, and the PAs associated with the highest magnitude leakages are geographically aggregated in the south and west of the study area (Fig. 3.4). This trend is congruent with the model selection analysis' identification of PAs location within the state of Amazonas being more associated with leakage outcomes than PAs located in other states. It is logical that sustainable-use PAs, where some resource extraction is allowed, showed little evidence of leakages and were associated with blockages in the best model, as looser land-use restriction is less likely to displace clearing pressure (i.e., create leakage) than is strict bans on human land uses. Assuming that PA permanence is upheld, strict conservation units in the southern and western regions of Amazônia Legal should continue to be monitored for leakage over time, especially considering the worsening magnitude of annual leakage for the PAs that showed net leakage by 2017 (Fig. 3.5).

Although I identified few paper parks and few leakages, two major threats related to leakage of deforestation pressure from PA establishment remain. First, the few cases of high magnitude leakage I detected could have displaced habitat conversion pressure to non-target areas that are more vulnerable or irreplaceable than the PA target area itself. Thus, the key principles of systematic conservation planning, irreplaceability and vulnerability (Margules and Pressey, 2000), should still be rigorously applied to prioritise areas for conservation in the Amazon, and conservation planners should make a concerted effort to consider the species richness in the area targeted for protection as well as the species richness of the associated spillover zones (Renwick et al., 2015) to avoid a perverse outcome of opportunistic, area-based protection priorities (Barnes et al., 2018). Second, the increasing magnitude of (non-cumulative) leakage over time (Fig. 3.5) indicates that net leakage may continue to increase in the future; as such, I recommend that as new data becomes available, spillover should be routinely monitored. Third, although I found little evidence of displaced deforestation pressure in the adjacent surroundings of PAs, deforestation that would have occurred in the PA could have been displaced further afield. Indeed, two supply-chain agreements implemented to reduce deforestation in the Amazon, the Soy Moratorium and zero-deforestation beef agreement, have been found to have successfully reduced deforestation in the target region at

the cost of increased clearing of native vegetation in the Cerrado region to the south (Dou et al., 2018); it is possible that PAs established in the Amazon could also displace land-use change pressure to the Cerrado. Implementation of a telecoupling framework would be required to identify such trans-biome spillover; the same holds for trans-national scale spillover (Bruckner et al., 2015; Dou et al., 2018). Total leakage could consist of clearing displaced both locally and further afield; I only measured local leakage. It is nonetheless important to measure the local component of leakage, especially considering that deforestation displaced from the PA is likely to occur nearby because adjacent areas are likely characterised by the same accessibility and other characteristics as the protected land in question (Bode et al., 2015).

There is a need for clear, widely applicable methodological guidelines for counterfactual evaluation of these local non-target impacts of PA establishment (Ferraro and Pattanayak, 2006; Fuller et al., 2019; Miteva et al., 2012; Pfaff and Robalino, 2017). The impossibility of estimating locally derived baselines for 18 of the 91 PAs due to the crowded nature of this region's reserve network demonstrates the practical limitations of method proposed by Ewers and Rodrigues (2008). Further, the estimation of a baseline deforestation rate from random landscape samples suggested by this method does not account for other untested environmental and anthropogenic factors that could be driving intensity of deforestation pressure. Statistical tools like propensity score matching (Andam et al., 2008; Chomitz, 2002; Joppa and Pfaff, 2010) and panel regression techniques (Jones and Lewis, 2015) could improve these counterfactual baseline estimates. My study attempted to account for this possible variation in likelihood of clearing by using two baselines: one highly localised to minimise environmental differences, and one state-based to minimise differences in anthropogenic factors. My strategy may not have accounted for all possible confounding effects (Ferraro and Pattanayak, 2006); the true counterfactual annual deforestation that would have occurred if no PA had ever been established cannot, by definition, be known but only estimated.

Policies targeting deforestation, such as legislation like the Forest Code of 2012 (Soares-Filho et al., 2014) or supply-chain agreements like the Soy Moratorium initiated in 2006 (Dou et al., 2018), were indirectly accounted for through generation of a baseline expectation of deforestation for each PA separately for each year. By calculating target and non-target PA impact with these annual counterfactual estimates, my analyses captured year-to-year variation in deforestation pressures. It is likely that a combination of such legislative and supply-chain regulations with permanent and enforced PAs and Indigenous Lands is necessary to optimise trade-offs between conservation and development in the Amazônia Legal region, and it is important that future studies distinguish the varying roles of these regulations and associated human migrations (e.g., illegal land appropriation by *grileiros*). I assessed the impact of 91 PAs' only upon deforestation rates, but it is important to

acknowledge that the land-use restrictions associated with PAs may also impact other anthropogenic pressures on biodiversity and carbon, such as forest degradation, hunting, introduction of invasive species, hydroelectric energy generation and transportation, mining, or road construction. It is important to also account for the social and ethical dimensions of PA establishment for local residents and indigenous peoples (Miller et al., 2011).

The extent to which the results of my impact assessments and spillover accounting for deforestation inhibition by PAs can be generalised beyond the study region is unclear: most of the PAs I assessed are relatively remote from densely populated cities or transportation infrastructure (for now). Future research on this region, and research on less remote regions, should pursue fine-scale analyses of accessibility, using proxies such as distance to nearest road or town. For example, it cannot be assumed that the same minimal threat that leakage of clearing pressure has posed recently in Amazônia Legal translates to habitats that are already highly fragmented by development, such as the Atlantic Forest, or that are characterised by other types of land cover, such as the Cerrado. These two other Brazilian biomes are both global biodiversity hotspots where habitats are highly threatened by land-use changes (Myers et al., 2000); assessing the target and non-target impacts of the PAs and other conservation interventions in these and other biodiversity hotspots, where data is available, should be a global priority.

Conclusion

Regardless of location, it is crucial to ensure that PAs are truly additional, permanent (Kroner et al., 2019; Mascia and Pailler, 2011; Pack et al., 2016), enforced, and that their achievements are not offset by leakage. PAs are the cornerstone of conservation of ecosystem, species, and genetic diversity globally; impactful forested PAs are also important for climate change mitigation goals. My counterfactual impact assessments suggests that most PAs established since 2005 in Amazônia Legal have successfully inhibited deforestation within their boundaries, and, in the majority of cases, this inhibition of clearing has not displaced deforestation pressures to PAs' adjacent, unprotected surroundings. Leakage mitigation as part of the systematic conservation planning for this region is, therefore, not imperative. For the most effective conservation of Amazonian forests, priority should instead be given to achieving additionality with any new PA and ensuring the permanence and enforcement of those already established.



Chapter 4: Thinking globally and acting locally to achieve protected area targets.³

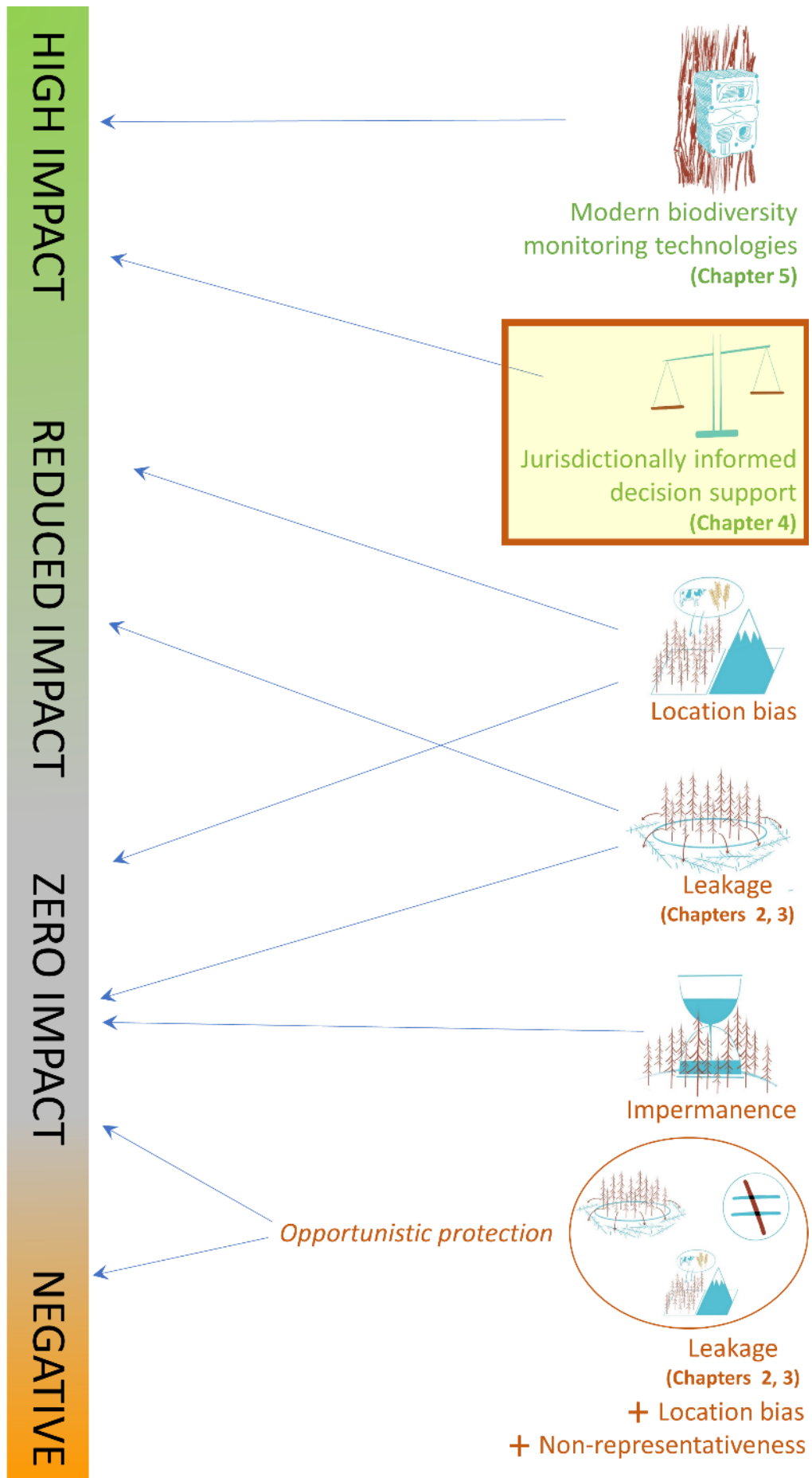


Banksia branch, Ben Lomond, Tasmania, 2020

“No matter how much one may love the world as a whole, one can live fully in it only by living responsibly in some small part of it. Where we live and who we live there with define the terms of our relationship to the world and to humanity. We thus come again to the paradox that one can become whole only by the responsible acceptance of one's partiality.”

— Wendell Berry, *The Art of the Commonplace: The Agrarian Essays*.

³ Pending submission to *Biological Conservation* as a contributed paper authored by Fuller, C., Ondeï, S., Adams, V.M., Brook, B. W., & Buettel, J. C.



Abstract

Expanding the network of protected areas (PA) is one of the core targets of the Convention on Biological Diversity's post-2020 Global Biodiversity Framework. Yet, there is no clear protocol for decision makers to translate international aspirations for conservation into local actions. To this end, I devised a way to account for multiple priorities across different jurisdictions to improve transparency and accountability. As a case study, I applied this strategy to planning the expansion of the Tasmanian PA network. I ranked planning units across nearly 400,000 ha of unallocated public lands, with the priority assigned to spatial features reflecting the relevant targets' jurisdictional origin, weighting international priorities more heavily than localised ones. For comparison, I also ranked the planning units based on a simple multi-objective strategy, by equally prioritising representation of habitat types, PA connectivity, cost efficiency, and threat mitigation. The future Tasmanian PA network suggested by the jurisdictionally informed prioritisation was largely incongruent with the simple multi-objective approach, and neither overlapped at all with the business-as-usual Government protection proposal in place at the time of this research. Across three different area budgets, the jurisdictionally informed plan achieved greater connectivity but came at a higher opportunity cost than the simple regional approach, and both achieved less connectivity and a higher opportunity cost than the business-as-usual plan. Given the importance of reconciling local and international priorities for post-2020 PA expansion, I recommend including jurisdictionally informed scenarios in conservation planning to ensure greater transparency and accountability where compromises and trade-offs are inevitable.

Introduction

The most important global policy mechanism for conservation, the Convention on Biological Diversity (CBD), will soon finalise the post-2020 Global Biodiversity Framework for halting the loss of biodiversity and ecosystem integrity (Tsoumani, 2020). A key target of this decade's conservation plan is to *“protect and conserve through a well-connected and effective system of protected areas and other effective area-based conservation measures at least 30% of the planet with the focus on areas particularly important for biodiversity”* by 2030 (Secretariat of the CBD, 2020), henceforth the '30x30' target. This aspiration for expanding the global protected area (PA) network is to be collectively implemented by the national governments of the CBD's 196 signatory countries, which will track locally driven PA establishment to report national progress against the international target.

Historically, PA networks have been geographically biased towards remote, high-altitude, and/or agriculturally unsuitable areas (Pressey, Humphries, Margules, Vane-Wright, & Williams, 1993; Joppa & Pfaff, 2009). If post-2020 PA expansion is to make a difference for irreplaceable, vulnerable biodiversity, this bias must not be perpetuated (Venter et al., 2018). Instead, data-driven prioritisation of *“areas particularly important for biodiversity”* should be guided by systematic conservation planning (Margules & Pressey, 2000; Moilanen, Wilson, & Possingham, 2009).

One way to identify priority areas is to consider the extent to which a habitat type is (under)represented in the existing PA network (Kukkala & Moilanen, 2013; Sarkar & Illoldi-Rangel, 2010); planning for more representative PA networks is facilitated by international and national bioregionalization frameworks (e.g., the global ecoregions of Olson et al., 2001 and the bioregions of the Interim Biogeographic Regionalisation of Australia, 2012). The selection of new areas to legally protect is often constrained by the degree to which a planner seeks to maximise connectivity between PAs (Beger et al., 2015; CBD 2020) and to minimise costs (Naidoo et al., 2006). The mitigation of threats to biodiversity (e.g., forest loss due to timber harvesting, climate change, poaching), also critical to the conservation impact of a PA (Sacre, Weeks, Bode, & Pressey, 2020), is another common goal of systematic conservation planning.

The conservation-science literature provides some guidance for navigating trade-offs where PA objectives of representativeness, connectivity, cost efficiency, and threat mitigation compete (Beger et al., 2015), but PA expansion is typically further complicated by the involvement of multiple jurisdictions (e.g., municipal, state, regional, national, or international designating and managing authorities), each with their own local priorities. For example, declaring scenic and recreational areas as reserves may generate tourism for the local economy, historical protection for local hunting grounds or water supply may constrain what remains available for protection. Lobby groups

promoting parochial causes may also exert a significant influence in decision-making processes. These priorities of local jurisdictions can complicate adherence to national strategies for PA-network representativeness (e.g., Australia's 'comprehensive, adequate and representative' reserve system) or international agreements (e.g., Ramsar, Bonn, and World Heritage Conventions and the CBD).

Problems of transparency and accountability in the decision-making process can also arise, especially if local governments are pressured to select new areas for protection that might cover only small extents or be biogeographically irrelevant, as a means of justifying decisions that reflect stakeholder values. To address the need for cross-jurisdictional compromise in efforts to meet the '30x30' target, I here applied a simple yet potentially powerful decision-support framework that offers transparency about what is prioritised, and therefore also what is compromised, during the selection of new PAs. This conservation-decision-support tool (Ondei, Brook, & Buettel, 2019) requires explicit assignment of importance weightings to each spatial feature, ensuring transparency and accountability. The tool's flexibility means that this approach can be applied to PA-expansion planning anywhere that sufficient data are available.

I illustrate the importance of considering jurisdictional priorities with a broad-scale decision support framework by applying it to contested public lands of the Australian island state of Tasmania. The Tasmanian State Government's recent proposal to convert a portion of lands allocated for potential future forestry to reserve tenure (see Appendix C, S3) provides a prime opportunity for comparison against alternative, data-driven solutions. I identify these alternative conservation plans by (1) individually prioritising representation, connectivity, cost efficiency, and threat mitigation features; (2) assigning equal importance weighting to each objective to produce a simple multi-objective strategy; and (3) alternatively assigning hierarchical importance weightings that reflect jurisdictional priorities to produce a jurisdictionally informed compromise. The third strategy accounts explicitly for the differential importance of certain features to the international, national, regional, and municipal authorities involved, thereby enabling planners to reconcile local constraints (e.g., costs) and objectives (e.g., threat mitigation) with international and national extent-and-representation goals. I then assessed the extent to which the jurisdictionally informed PA expansion plan differs from the other reasonable protection plans, and from the Tasmanian government's business-as-usual (western-biased) proposal.

Methods

Case study

Tasmania, a large temperate continental island (68,401 km²) situated off the southern coast of mainland Australia, has a mild maritime climate, and its physical complexity contains a large diversity of terrestrial, freshwater, and coastal habitats. It is recognised as a centre of floristic endemism (Crisp, Laffan, Linder, & Monro, 2001; Kirkpatrick & Brown, 1984a, 1984b) and is home to iconic animal species that are extinct on the rest of the continent. With a small and geographically concentrated human population, much of the landscape of Tasmania is protected (Fig. 4.1), and the public PA network is complemented by a growing private reserve estate of properties under permanent conservation covenants. Emblematic of the island's relatively successful history of conservation (Mendel & Kirkpatrick, 2002) is Kirkpatrick's application of systematic techniques for PA site selection to targets (1983), the first ever published in the scientific literature and an advance that paved the way for the emergence of systematic conservation planning as a discipline (R. Pressey, 2002). A large western tract of the island is internationally recognised as the Tasmanian Wilderness World Heritage Area (DASETT, 1989; see Fig. 4.2).

Tasmania's current PA network is more representative of its component vegetation communities than are the PA networks of most other regions in the world (Mendel & Kirkpatrick, 2002). Nonetheless, vegetation communities such as lowland tussock grasslands and grassy woodlands remain poorly represented. Furthermore, there is a large disparity in proportion protected (Fig. 4.1) across Tasmania's three ecoregions (Olson et al., 2001) and nine nationally declared bioregions (Department of the Environment, 2012; see also Appendix C, S2). Ideally, the '30×30' extent goal should be implemented to improve the representation of biogeographic units in the Tasmanian PA network.

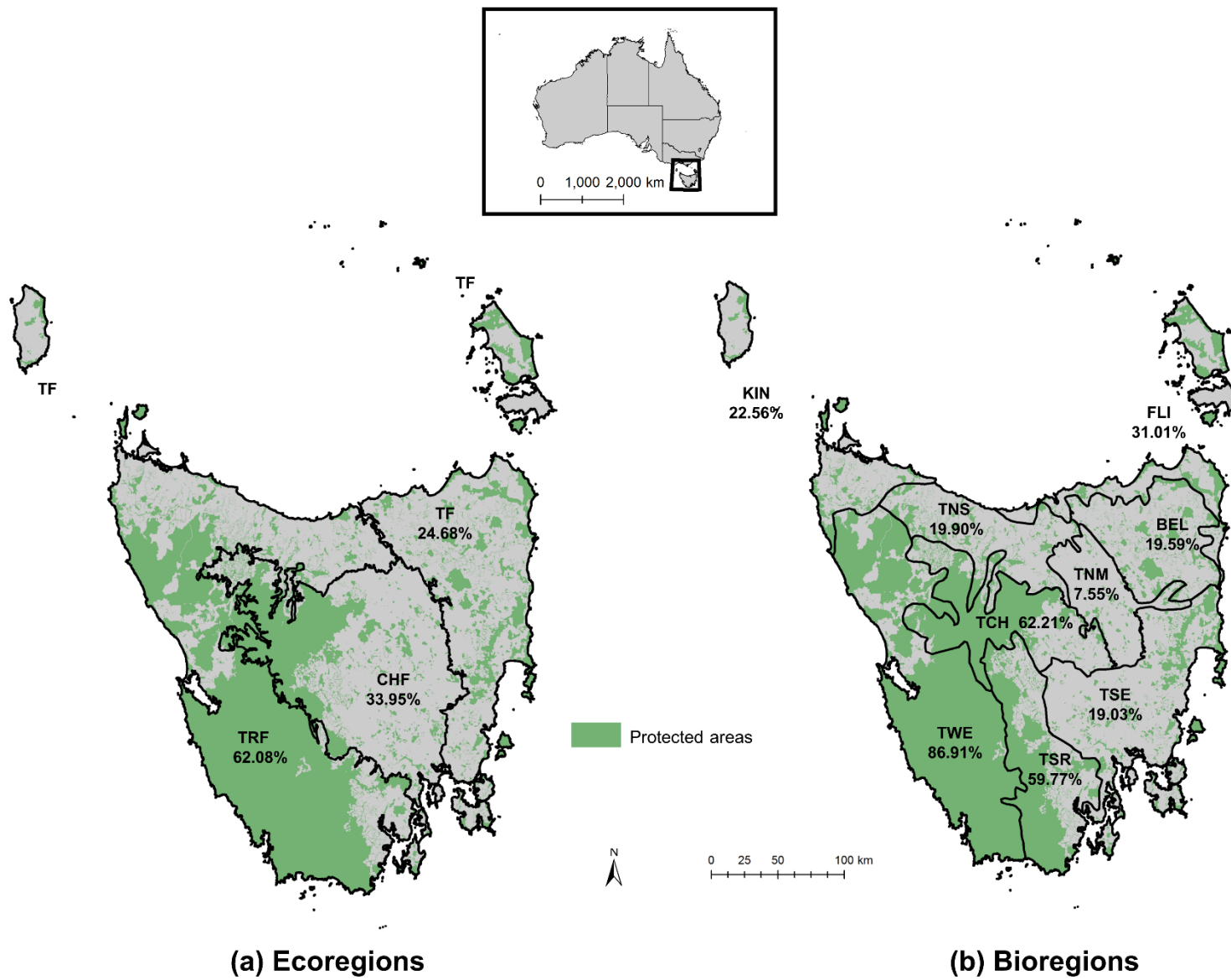


Figure 4.1 Percentage of land protected for Tasmania’s (a) three ecoregions and (b) nine bioregions. The temperate forest (TF) ecoregion and the Northern Midlands (TNM), the South East (TSE), Ben Lomond (BEL), the Northern Slopes (TNS), and King (KIN) bioregions all currently have less than 30% of their landscapes protected. Spatial data was obtained from Olson et al. (2001) and from the Department of the Environment (2012).

The conversion of the 392,969 ha of “Future Reserve Lands” to “Future Potential Production Forests” (Fig. 4.2) in 2013 aimed to secure a “wood-bank” to provide additional resources for future production forestry (*Rebuilding the Forest Industry Act of 2013*) to supplement timber resources available in the Permanent Timber Production Zone (approximately 800,00 ha; Fig. 4.2). It also enabled immediate harvest of specialty species, such as celery-top pine (*Phyllocladus aspleniifolius*) and sassafras (*Atherosperma moschatum*). On 8 April 2020, these FPPF lands were opened for reallocation to either the reserve estate (clearing prohibited) or to the Permanent Timber Production Zone (clearing allowed; see the Future Potential Production Forest Fact Sheet, 2020). Return of Tasmania’s Future Potential Production Forest (FPPF) lands to reserve status could contribute to representative post-2020 PA expansion. For example, protecting the FPPF lands within the underrepresented temperate forest ecoregion and Ben Lomond bioregion (Fig. 4.1) would meet or exceed the 30% protection target for those biogeographic units (Appendix C, S2).

Given this potential opportunity, and the demonstrated likelihood of their protection (Fig. 4.2; see also Appendix C, S3), I chose to apply a decision-support framework specifically to the FPPF lands. This subcategory of the state’s “unallocated crown lands” is predominantly forested; those of the northwest are mostly rainforest and related scrub, whilst those of the eastern half of Tasmania largely consist of dry eucalypt forests. Some areas of native grassland, moorland, sedgeland, rushland, and peatland also fall within the FPPF tenure (see Appendix C, S13 and S14). The FPPF lands have a variety of land-use histories: some are undisturbed whereas others have undergone recent land-use change or experienced edge effects from adjacent private lands. Clearing for timber harvest is prohibited within the Future Potential Production Forest (FPPF) lands, although special permits may be issued for low-intensity, selective harvest of speciality tree species. The 243 polygons delineated in the spatial dataset for the FPPF tenure (<http://www.thelist.tas.gov.au> ©State of Tasmania)—the boundaries of which are in-part artefacts of the previous forestry coupes—were chosen as the planning units of my prioritisation analyses because they are the most-likely parcels of land by which the Tasmanian government will allocate to one of two possible future tenures: protected or timber production (Fig. 4.2).

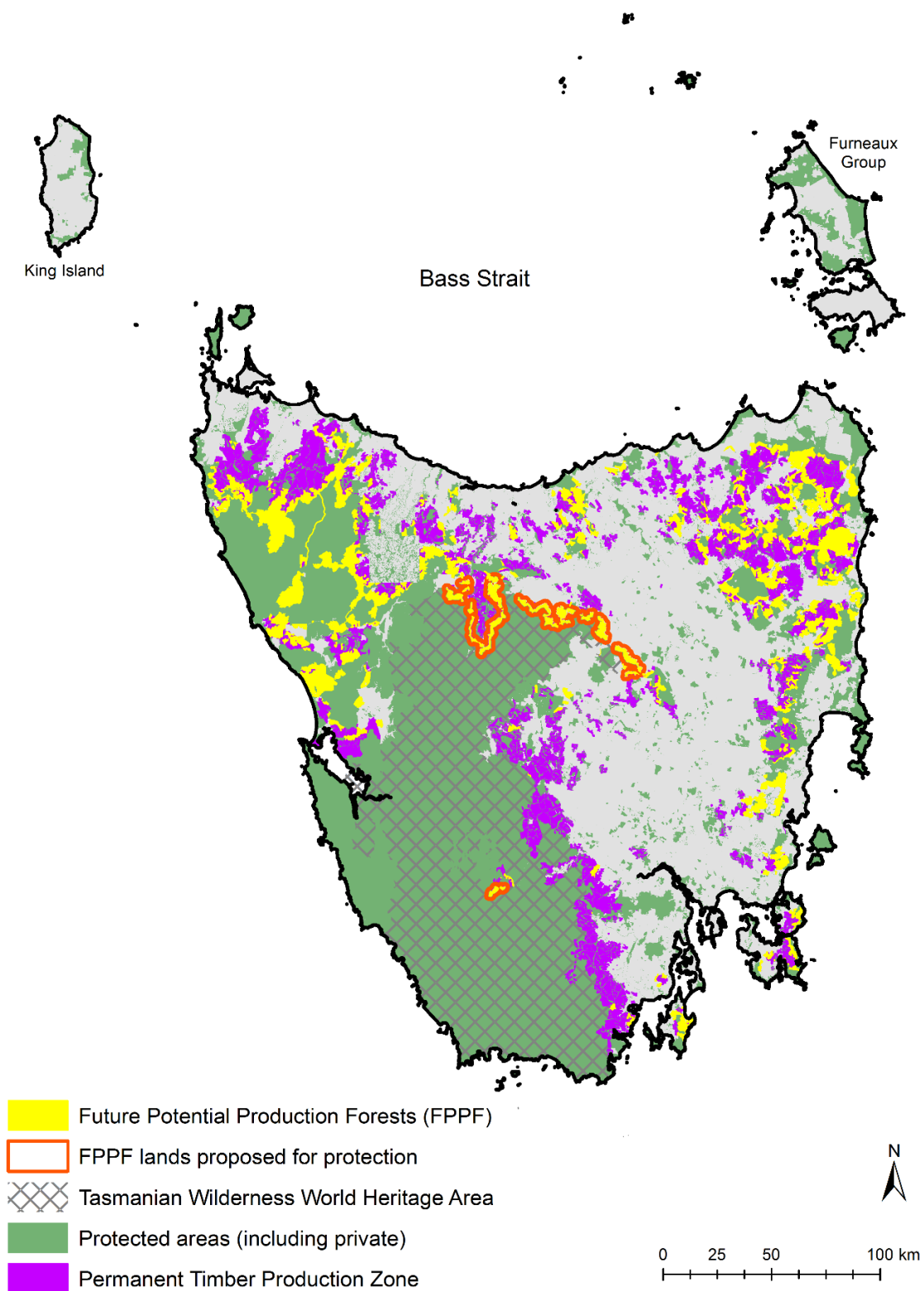


Figure 4.2 Tasmania's protected, timber production, and future potential production zones, including the area proposed for protection (orange outline), located adjacent the Tasmanian Wilderness World Heritage Area. The Tasmanian Forest Agreement Act of 2013 declared 392,969 ha to be "Future

Reserve Lands;” in 2014, these areas were declared “Future Potential Production Forests” (yellow) upon the passage of the Forestry (Rebuilding the Forest Industry) Act, which aimed to secure a ‘wood-bank’ to provide resources for future production forestry. On 8 April 2020, these lands were opened for potential reallocation to either the reserve estate or to the Permanent Timber Production Zone (Future Potential Production Forest Fact Sheet, 2020). Grey areas are outside of the public forest domain. Spatial data was obtained from the “LIST Land Tenure” from <http://www.thelist.tas.gov.au/> ©State of Tasmania. See also Appendix C, S1 and S3.

Spatial features within planning units

In my PA-options analysis, I considered representation, connectivity, cost efficiency, and threat mitigation objectives. Features that contribute to each of these objectives (Table 1) were calculated for each planning unit using ESRI ArcMap 10.6 software and open-access sources. Spatial information for global ecoregions and bioregions was obtained from Olson et al. (2001) and from the Australian government (Department of the Environment, 2012), respectively. Spatial data in vector format was obtained from <http://www.thelist.tas.gov.au> (©State of Tasmania) for land tenures, land cover, and vegetation groups. Tasmanian enterprise suitability maps produced by Kidd, Webb, Malone, Minasny, and McBratney (2015) provided data for the suitability for three major plantation forestry species: *Eucalyptus nitens*, *Eucalyptus globulus*, and *Pinus radiata* (see Appendix C, S6, S7, and S8). Although the native-forestry-harvest potential of the planning units would constitute another reasonable opportunity-cost metric, the relevant data are not open access. See Appendix C, A20 for more on data limitations and methodological caveats.

Four simple rankings for single features

Having calculated the extent of each of these features contained within each FPPF, high-priority planning units were then identified on the basis of maximising their contribution to representation, connectivity, cost efficiency, and threat mitigation features. I defined single-objective scenarios for each of these four alternative goals:

1. The **representation scenario** sought to achieve 30% protection of each ecoregion and bioregion, or where 30% is unattainable with only FPPFs, to maximise representation; planning units with greater areas in under-protected biogeographic units were ranked higher (see Appendix C, S2, S4, and S5).
2. The **connectivity scenario** sought to create the most contiguous resultant PA network by selecting planning units with minimum 50% of their perimeter adjacent to existing PAs (see Appendix C, S11, S12).
3. The **cost-efficiency scenario** sought to minimise opportunity costs to the timber-plantation industry by avoiding planning units containing suitable areas for timber plantations for three species – *Eucalyptus nitens*, *Eucalyptus globulus*, and *Pinus radiata* – as reported by Kidd et al. (2015; see Appendix C, S6, S7, and S8).
4. The **threat-mitigation scenario** prioritised the planning units most surrounded by modified land, calculated as the proportion of the 1-km buffer zone around the planning unit classified as ‘modified land’ by the TasVeg4.0 spatial layer (see Appendix C, S8 and S9 for the land cover classes included in this category); planning units whose buffer zones were more than 50% modified were considered highly threatened (see Appendix C, S9 and S10).

Two multi-objective rankings

Tenure allocation to PA networks requires more practical decision-support guidance than the simple identification of priority areas for individual objectives; ideally, conservation planners can recommend strategic compromises that account for all relevant goals using the best data available. A generalised tool for integrating many spatial features into conservation-decision-support must be flexible, as the relevant goals and available data often vary across spatial scales and planning units. To that end, I applied a simple tool developed by Ondeï, Brook, and Buettel (2019) for use in $\mathbb{R}$ (R

Core Team, 2018), which combines multiple features with adjustable importance weightings to generate rankings of planning units, upon which reasonable PA establishment plans can be based.

In brief, to rank planning units, this method normalises the metric-by-planning unit's raw values using z-scores, calculated as the difference between the value for one planning unit and the mean, divided by the standard deviation. It calculates the overall z-score for each planning unit by multiplying the z-score of each metric by the weight it has been assigned by the user, summed across all parameters. I used this tool to produce two rankings of the 243 FPPF planning units, accounting for the same suite of eight key features (summarised Table 1) in both scenarios, defined as:

(1) Simple/equal weighting – features were grouped based on their relevance to the goal category (representation, connectivity, cost efficiency, or threat mitigation); equal (i.e., 0.25) weight was assigned to each goal category; importance weightings were distributed evenly amongst the component features of each category (Table 1).

(2) Jurisdictionally informed – features were grouped according to the international, national, or local origin of the target to which it contributes or detracts, with each group of features assigned importance weightings hierarchically. I chose to assign an internationally relevant feature twice the weight of a nationally relevant feature, which in turn was assigned twice the weight of the suite of locally relevant features (Table 1). The rationale for this hierarchy of weights was that coordinated global PA network expansion efforts have been shown to be more efficient than uncoordinated, national-level conservation planning (Pouzols et al., 2014; Sala et al., 2021). In the example of choosing which of Tasmania's FPPF lands to convert to reserves, ostensibly relinquishing the rest to timber production, I assigned top priority to the international goal of equitably representing all global ecoregions; I assigned as my second priority the continental goal of equitably representing all Australian bioregions. Features contributing to more local goals in the Tasmanian context were each assigned the same, significantly lower priority weighting.

The two multi-objective scenarios are thus distinguished only by the differential importance weightings assigned to the eight features. Note that the jurisdictional weightings applied here are intended to provide an example where international goals are considered twice as important as national considerations, etc., but are ultimately somewhat arbitrary: weightings should be assigned on a case-by-case basis, ideally based on principals agreed by the various stakeholders.

Table 4.1 Eight features calculated for each planning unit and used to rank their relative conservation priority. The simple multi-objective strategy and the jurisdictionally informed strategy were therefore distinguished only by the relative-importance weightings assigned to each feature. For example, the area of under-protected native grasslands within each planning unit was assigned 0.0833 and 0.0238, respectively (see Appendix C, S13 and S14). Weights sum to 1.

Objective	Spatial feature	Simple multi-objective prioritisation	Jurisdictionally informed prioritisation
Representation		$\frac{1}{4}$	
	Area (ha) within ecoregions <30% protected (i.e., the temperate forests)		$\frac{4}{7}$ <i>international</i>
	Area (ha) within bioregions <30% protected (i.e., TNM, TSE, BEL, TNS, KIN)		$\frac{2}{7}$ <i>national</i>
	Area (ha) cover by vegetation groups <30% protected (i.e., native grasslands)		$\frac{1}{42}$
Connectivity		$\frac{1}{4}$	
	Length (km) of the planning unit's boundary shared with PA(s)		$\frac{1}{42}$
Cost efficiency		$\frac{1}{4}$	
	Area (ha) unsuitable for timber plantations		$\frac{1}{42}$
Threat mitigation		$\frac{1}{4}$	
	Percent area classified as 'modified land' within 1km from the planning unit		$\frac{1}{42}$
	Area (ha) classified as 'modified land' within planning unit		$\frac{1}{42}$
	No. plant 'weed' species or phytophthora recorded in 1km buffer		$\frac{1}{42}$

$\frac{1}{7}$ *local*

Identifying high priority planning units

I applied three reasonable area budgets (Appendix C, S19)—business-as-usual (BAU), ambitious, or conservative—to translate the two rankings into protection plans that can be compared with one another and, in the case of the BAU area budget, with the protection allocation recently proposed by the state government (Fig. 4.2; see also Appendix C, S1 and S3). At the time of this research, the proposed protection (upon which the BAU area budget is based) sought to add 25,209 ha (6.4% of the FPPF lands) to the PA network as part of the Tasmanian Wilderness World Heritage Area (TWWHA; DPIPWE, 2016).

An ambitious area budget was set equivalent to the total area of planning units required to protect 30% (or, if not attainable, to maximise protection) of each of three ecoregions and nine bioregions (234,705 ha or 59.7% of the FPPF lands) (Fig. 4.1; see also Appendix C, S2, S4, and S5). A third, conservative area budget of 10,277 ha (2.6% of the FPPF zone) was chosen to reflect the total area of all planning units containing no suitable areas for *Eucalyptus nitens*, *Eucalyptus globulus*, or *Pinus radiata* plantations (see Appendix C, S6 and S8). The application of multiple (but all reasonable) area budgets provides insight into the sensitivity of the six scenarios' PA expansion recommendations to the total area of planning units available for protection. Priority selections were limited to whole planning units, assuming tenure decisions are to be made for each of the 243 lots and not for components of each; as such, the total area selected per scenario differ because I ensured that selections did not exceed the area budget. Selection recommendations were therefore as close in total area to the given area budget as allowed for by the size of the planning units prioritised.

Assessing congruencies.

I calculated congruence as the extent of spatial overlap of the sets of planning units selected for protection in each scenario. Because the two multi-objective rankings were generated using the same spatial input data, the extent to which the two priority sets do not overlap can be attributed solely to the difference in feature-importance weightings, assigned either according to their conservation goal category (representation, connectivity, cost efficiency, or threat mitigation) or jurisdictional origin (international, national, or local) of the target to which they contribute or detract.

Comparing efficiencies.

To assess the efficiency of the PA plans recommended by the six strategies, I first calculated a proxy metric for the resultant PA network's manageability: connectivity. Connectivity is measured by calculating the total length (km) of selected planning units' perimeters that are adjacent to existing PAs. A more contiguous PA network provides greater habitat connectivity (Saura et al., 2018), requires less boundary enforcement by managers, and is less likely to be subject to the edge effects experienced by those boundaries adjacent to unprotected areas, as unprotected adjacent areas are more likely to be modified for various land uses or exposed to higher levels of disturbance.

Gains in efficiency are also possible if economic information is considered when designing PA networks (Carwardine et al., 2008; Faith, Margules, & Walker, 2000; Stewart, Noyce, & Possingham, 2003). For each scenario's selected planning unit, I also estimated the opportunity cost incurred by the timber-plantation industry as the area suitable for any of the three major Tasmanian plantation species (*Eucalyptus nitens*, *Eucalyptus globulus*, or *Pinus radiata*), as reported in the Enterprise Suitability Map data (Kidd et al., 2015). Land valuations are irrelevant to opportunity cost in this case because the FPPF lands are not available for private purchase. Differences in the connectivity and opportunity costs of alternative protection plans are important considerations for decision makers, as is the sensitivity of these differences to area budgets.

Results

Suggested protection versus likely protection

Spatial selections of priority areas were different for each of the four single-objective scenarios (Fig. 4.3; see also Appendices S5, S8, S10, S12), although there are overlaps. Notably, more than 92% of the area selected for the representation scenario, i.e., the planning units with area in under-protected biogeographic units (Fig 3a), coincide with the existing protection proposal by the Tasmanian state government. The high congruence reflects that 12,568 ha (49.86%) of the 25,209-ha of FPPF land proposed for protection falls within the Northern Slopes (TNS) bioregion, which is currently less than 20% protected (Fig. 4.1), and 717.74 ha (2.83%) fall within the critically under-protected Northern Midlands bioregion, bringing proportion protected there from 7.6 to 7.7%. Similarly, more than 83% of the area selected for protection under the connectivity scenario (Fig. 4.3b) is congruent with the existing proposal. This congruence is unsurprising, as the 25,209-ha proposed for protection were selected by the state government due to their adjacency to the TWWHA. However, the government's proposal would not protect any areas within the Ben Lomond bioregion (19.6% protected) nor the Tasmanian temperate forest ecoregion (24.7% protected), two biogeographic units for which "quick wins" are otherwise possible through conversion of the FPPF lands to PAs, which alone would be sufficient to meet and even exceed the 30% target (Appendix C, S2).

None of the area selected for protection under the zero-cost scenario (Fig. 4.3c) is congruent with the existing protection proposal, as the planning units likely to be protected by the state government all contain area suitable for timber plantations. There was also no overlap between the area selected for protection under the threat-mitigation scenario (Fig. 4.3d) and the existing proposal, as the latter is adjacent to the already strictly protected TWWHA. Indeed, none of the areas selected by either of the two multi-objective strategies overlapped with the government's proposal when constrained to the BAU (25,209-ha) area budget. Both of these multi-objective plans, when constrained to the BAU area budget, achieved less connectivity and came at a higher opportunity cost compared to the government's plan to protect only TWWHA-adjacent FPPF planning units (Table 2; see also Appendix C, S18).

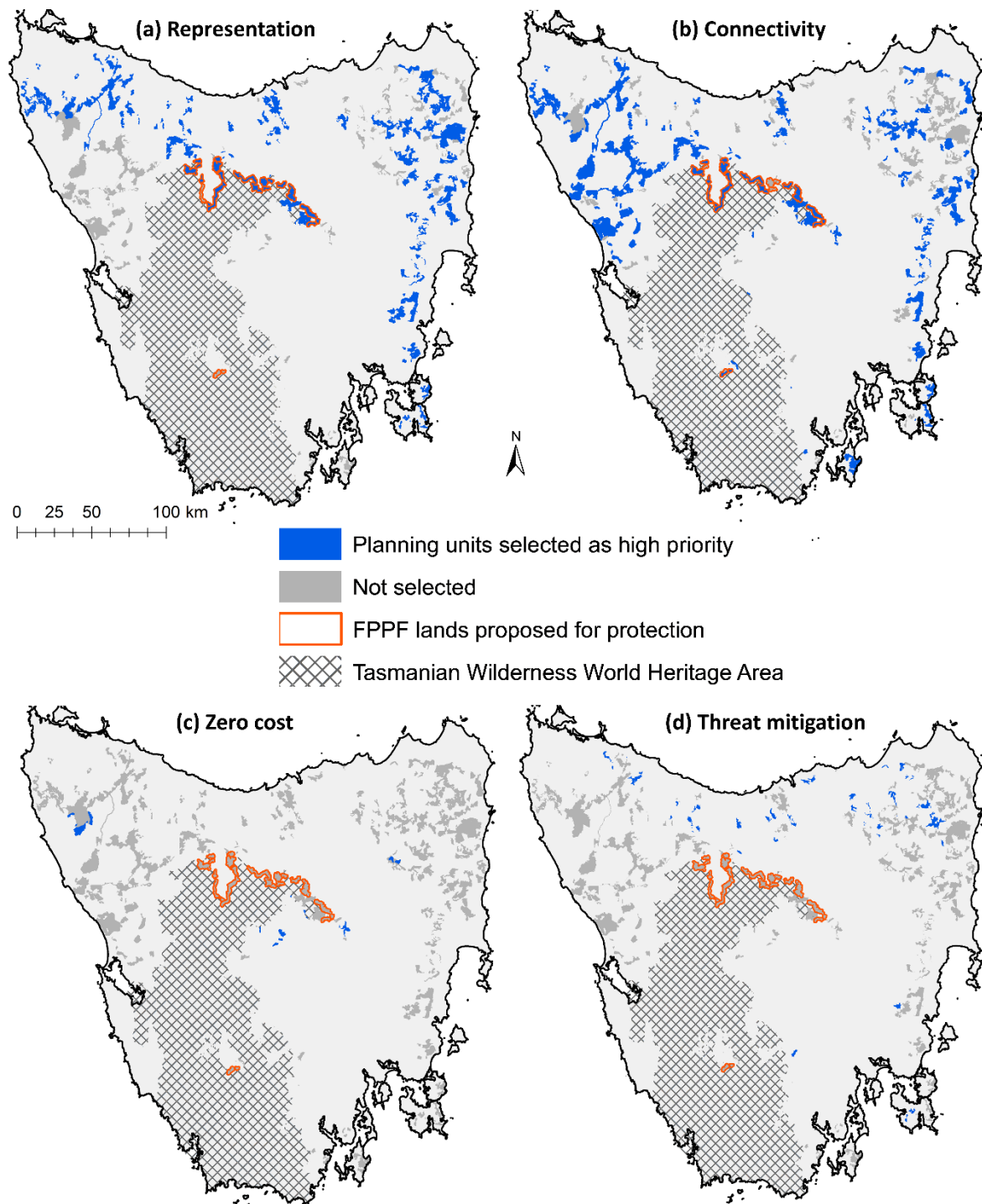
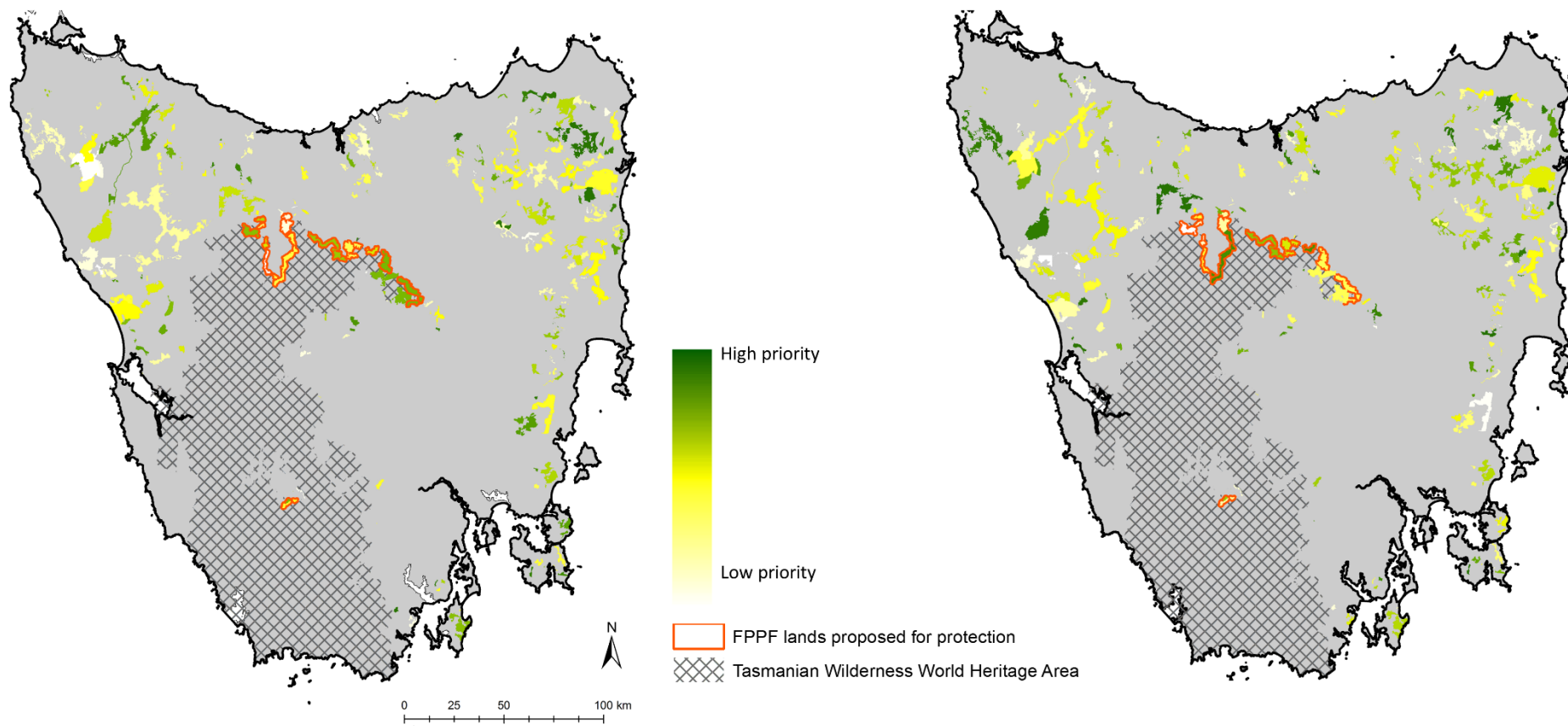


Figure 4.3 Four single-feature prioritisations. The (a) representation scenario selects all planning units required to protect 30% (or the maximum proportion where only <30% is possible) of each biogeographic unit; 92.44% (23,302 ha) of the area selected is congruent with the existing protection proposal (orange outline). The (b) connectivity scenario selects all planning units for which 50% or greater of the perimeter is adjacent existing protected areas; 88.30% (22,259 ha) of the area selected is congruent with the existing protection proposal (orange outline). The (c) zero-cost scenario selects all planning units that contain no area suitable for timber plantations. The (d) threat- mitigation

scenario selects all planning units for which at least 30% of the 1 km buffer zone was modified land, i.e., not intact native vegetation. None of the areas selected by scenarios (c) and (d) were congruent with the government's plan.

Comparing the two compromise rankings

Although the same eight features were input to the decision-support tool for both the simple multi-objective prioritisation and the jurisdictionally informed alternative, the two resultant compromise rankings of the 243 planning units were dissimilar (Fig. 4.4; see also Appendix C, S16 and S17). The highest-priority areas identified using the simple multi-objective strategy, for which each of the four objectives were weighted equally, were concentrated in the northeast of the island (Fig. 4.4a). By contrast, the highest-priority areas flagged by the jurisdictionally informed ranking were distributed more evenly across all regions (Fig. 4.4b) despite the heavy weight this strategy assigned to planning units within the area of the north-eastern temperate forest ecoregion (Fig. 4.1, Table 4.1).



(a) Ranking by the simple multi-objective strategy

(b) Ranking by the jurisdictionally informed strategy

Figure 4.4 Two compromise prioritisations output by the ranking tool. Although the input features were identical, the differential importance weightings assigned by the simple multi-objective strategy (a) versus the jurisdictionally informed strategy (b) resulted in notable differences in the output rankings.

Priority areas within area budgets

Regardless of the area budget applied, the jurisdictionally informed conservation plans achieved greater connectivity with existing PAs and contained more suitable area for timber plantations than did their non-jurisdictionally informed counterparts; this was true for both the raw values as well as proportionally (Table 2). The greater connectivity embodied in the jurisdictionally informed plans is preferable and unexpected, given the lighter importance weighting it assigned to connectivity. The jurisdictionally informed plans' slightly (Table 2) higher opportunity cost to the timber plantation industry is undesirable, but predictable, given the lighter importance weighting it assigned to cost efficiency. The jurisdictionally informed plan's superior connectivity and greater suitable timber plantation area across the three area budgets (Table 2) suggests that, in this planning context, the prioritisation of global and continental goals generally leads to better reserve network manageability, but this comes at a higher opportunity cost, requiring greater compromise by the Tasmanian plantation forestry industry.

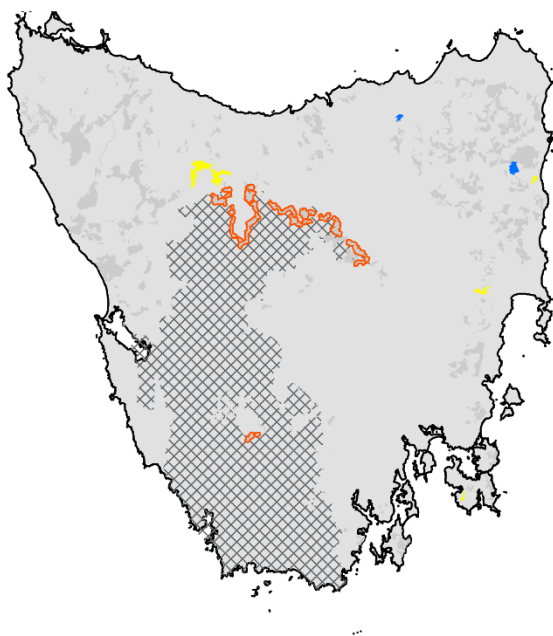
Table 4.2 Priority planning units selected by two strategies, constrained by three area budgets. At the time of this research, the protection proposed by the Tasmanian state government, upon which the BAU area budget is based, sought to add 25,209 ha (6.4% of the FPPF lands) to the PA network as part of the Tasmanian Wilderness World Heritage Area (DPIPWE, 2016). The ambitious area budget was set equivalent to the total area of planning units required to protect 30% (or, if not attainable, to maximise protection) of each of three ecoregions and nine bioregions (234,705 ha, or 59.7% of the FPPF zone). The conservative area budget was set equivalent to the total area of all planning units containing no suitable areas for *Eucalyptus nitens*, *Eucalyptus globulus*, or *Pinus radiata* plantations (10,277 ha, 2.6% of the FPPF zone). The percent values refer to the proportion of the total FPPF tenure's number of planning units, area, reserve-adjacent boundaries, and area suitable for timber plantations is captured within the protection plan suggested by each strategy under each area budget. The greater connectivity and opportunity cost values are in bold.

		Selected by the simple multi- objective strategy		Selected by the jurisdictionally informed strategy	
Conservative area budget	No. planning units	3	1.2%	4	1.7%
	Area (ha)	4,008	1.0%	12,233	3.1%
	Length of PA-adjacent boundary (km)	36	0.9%	135	3.2%
	Area (ha) suitable for timber plantations	235	1.2%	359	1.8%
BAU area budget	No. planning units	13	5.4%	10	4.1%
	Area (ha)	24,959	6.4%	24,991	6.4%
	Length of PA-adjacent boundary (km)	202	4.8%	244	5.8%
	Area (ha) suitable for timber plantations	1,340	6.6%	1,355	6.7%
Ambitious area budget	No. planning units	142	58.4%	139	57%
	Area (ha)	224,131	57.0%	230,059	58.5%
	Length of PA-adjacent boundary (km)	2,301	54.9%	2,613	62.3%
	Area (ha) suitable for timber plantations	11,110	54.5%	12,785	62.7%

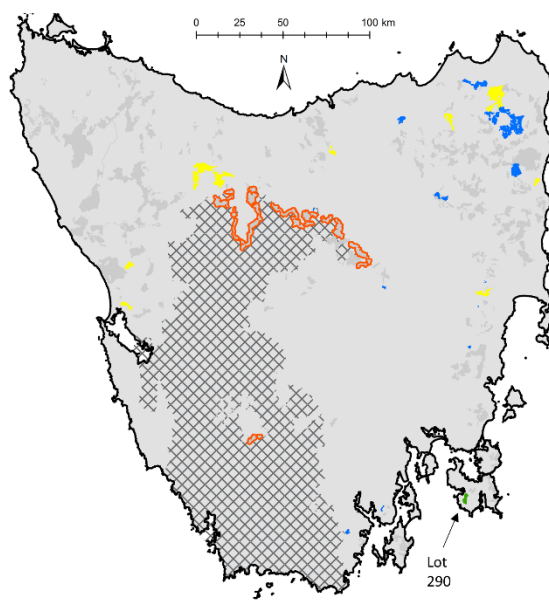
Congruencies

When constrained to either the conservative (Fig. 4.4a) or BAU area budget (Fig. 4.4b), the two multi-objective strategies recommend protection of entirely disparate planning units, except for Lot 290 on the Tasmanian Peninsula under the BAU area budget (Fig. 4.4b); Lot 290 is bordered on the west by the dry and wet eucalypt forest and woodlands of the Tasman National Park, and it is bordered on the east by modified pastures for grazing and timber production forestry. Within the ambitious area budget, 80 of the 243 planning units are identified as priorities by both multi-objective strategies; these 80 units are distributed across the state and include units on the Tasman Peninsula and Bruny Island (Fig. 4.4c).

(a) Conservative area budget



(b) Business-as-usual area budget



(c) Ambitious area budget

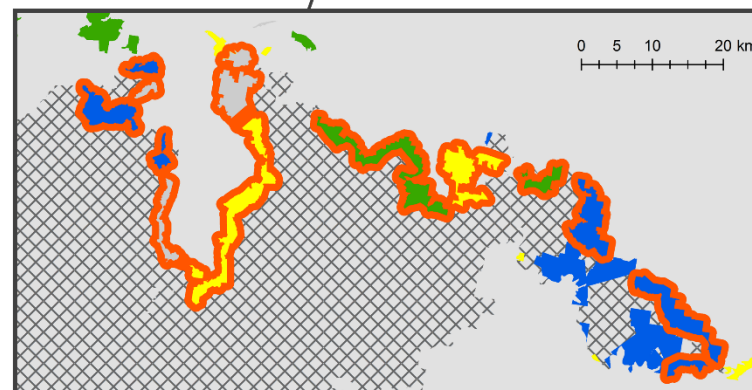
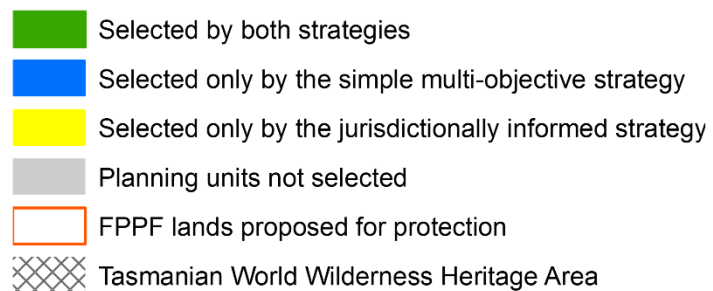
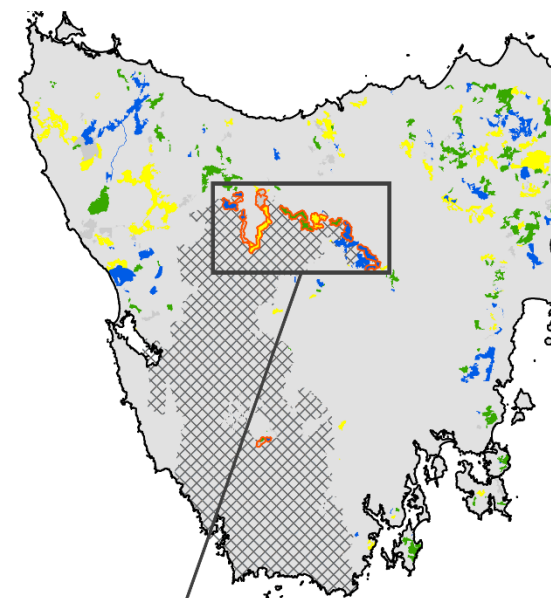


Figure 4.5 Conservation recommendations of the two multi-objective strategies, constrained by (a) conservative, (b) business-as-usual (BAU), and (c) ambitious area budgets. When constrained by an area budget (b), i.e., when limited to a maximum size equivalent to that of the BAU plan, neither compromise strategy recommended any of the current FPPF area that the Tasmanian state government has proposed for conversion to reserve tenure. The inset map under panel (c) shows that the areas the State has proposed for protection north of the TWWHA (which account for 24,172 ha or 96% of the total Government proposal) are recommended by the two multi-objective prioritisation strategies only at this most ambitious budget and alongside other priority areas, indicating that other (i.e., non-TWWHA adjacent) areas are required for the Tasmanian protected area system to meet multiple objectives.

Discussion

The conservation-science literature has recognised and warned of the threat that parochialism in land-tenure decision making poses to the establishment of an effective and representative network of global protected areas (Pouzols et al., 2014; Sala et al., 2021). However, few studies have tested or implemented quantitative solutions. To fill this gap, I developed a jurisdictionally informed framework for protected area-expansion planning that demonstrates how it is possible to prioritise across multiple, competing objectives while also accounting for the overarching importance of global and continental coordination in area-based conservation efforts. By doing this, I was able to illustrate how local (e.g., State-focused) decisions can be made in a way that ensures they are justifiable within the context of global conservation crises and solutions, while acknowledging that it will likely be challenging for conservationists, ecologists, and policymakers to reach agreements on criteria and priority weightings.

In this case study, I adapted an existing, data-driven tool to explicitly account for the primacy of global and continental objectives recommended unique target areas within the tenure-planning context of Tasmania. I found that the resultant jurisdictionally informed recommendation for converting planning units (Future Potential Production Forests – FPPFs) to reserves did not coincide at all with a jurisdictionally blind recommendation, which attempted to suggest a ‘fair’ compromise between four traditional PA-network goals. Further, the jurisdictionally informed recommendation also did not overlap with the government’s plan to extend the Tasmanian Wilderness World Heritage area (TWWHA). The government’s plan instead had the strongest alignment with the single-objective recommendations targeting increased connectivity and bioregional representation of the Tasmanian PA network, which are both reasonable given the operational advantages of protecting

these FPPF planning units for Tasmanian Parks and Wildlife Service, and considering the mandates for strict TWWHA protection by UNESCO (DASETT, 1989). This work also exposed the potential opportunities missed by the government's plan: forgoing two 'quick win' biogeographic units in terms of the 30×30 target, perpetuating historic location bias to the more remote west of the island, and failing to optimally mitigate threats.

The jurisdictionally informed approach proposed addresses local constraints (e.g., costs) and objectives (e.g., threat mitigation) as well as international and national extent-and-representation goals, thus reconciling potentially competing goals. By adapting prioritisation approaches so as to ensure that land-tenure decisions are locally relevant but not overly parochial, local arbiters of land tenure can be made accountable to international agreements to which their nation is a signatory – chiefly, to the Convention on Biological Diversity (Secretariat of the CBD, 2020). Such accountability is not only possible, but can be achieved through this transparent, apolitical spatial prioritisation based on ecological and economic data. Transparency in land-tenure decision making can reduce the pressure on governments to select new areas for protection to appease influential stakeholders, e.g., environmentalists, politicians, hunters, the wood-product industry, or the tourism industry. Transparency has recognised as an explicit objective of marine PA planning (Yates & Schoemann, 2015), and I advocate for similar treatment in terrestrial area-based conservation.

Transparency is inherent in jurisdictionally informed approaches, whereby each conservation feature contributing to decision making is explicitly assigned an importance value relative to other features. In my example, assignment of importance weightings to the features available and relevant to Tasmania were not intended to be generally prescriptive. Indeed, the hierarchical breakdown of jurisdictional tiers will be situation specific, and many other conservation features for which data is available may be included and importance-weighted on a case-by-case basis as agreed upon by all stakeholders. The z-score based mechanism by which prioritisation tool I adapted for this purpose allows for compromise across multiple objectives is straightforward compared to the complex simulated annealing algorithm of Marxan software (Ball & Possingham, 2000). However, I do not claim that this leads to a superior outcome to a Marxan-type optimization. Rather, this simplicity suited my goal of demonstrating how a jurisdictionally informed strategy can recommend a PA-expansion plan that is transparently traced back to the spatial features contained within selected planning units, ensuring that the priorities underlying the plan are obvious to all stakeholders. Because no single decision-support tool or framework, used in isolation, could hope to cover the full spectrum of potential conservation planning challenges (Schwartz et al., 2018) I recommend that other decision-support software could be adapted similarly, to consider interactions in jurisdictional needs from local to global.

In any systematic conservation planning exercise, the areas defined as planning units (i.e., the parcels which are considered feasibly protectable) influence the analyses and the resultant recommendation for PA-network expansion. In the Tasmanian case study, three major caveats accompany my choice to limit what were considered protectable planning units. First, the FPPF lands—chosen as planning units due to the prompt legal mechanism for their conversion to PAs—are predominantly forested lands (Appendix C, S15), and thus my scenarios could not address shortfalls in the grassland-dominated regions, such as the critically under-represented Northern Midlands bioregion. Indeed, my preliminary gap analyses (Appendix C, S2 and S13) affirmed that the Northern Midlands bioregion and the native grasslands vegetation group, both of which have much less than 30% of their area protected, are major hurdles to achieving a representative PA network in Tasmania, echoing past research in the region (McQuillan, Watson, Fitzgerald, Leaman, & Obendorf, 2009; Mendel & Kirkpatrick, 2002). Moreover, my gap analyses were optimistic in that I assumed all categories of PAs to be effective contributors to the PA network; if only public, IUCN category I, II, and III (i.e., dedicated and not multi-use) PAs are considered, then the task of achieving ‘30×30’ becomes more daunting. Second, the FPPFs are public lands, and therefore my prioritisations do not acknowledge that representation of under-protected regions in the Tasmanian PA network will require private-land conservation; for example, much of the more agriculturally suitable grasslands are predominantly located on freehold lands. Third, by limiting my analyses to the largely unmodified FPPF lands, I do not acknowledge that some of the under-represented habitats are so degraded that restoration is needed, nor do I assess the relative conservation priority of restoration works versus protection of intact areas. The pathway to the ‘30×30’ target through restoration of degraded areas should be seriously contemplated because, contrary to conservation orthodoxy, restoration can be more cost-effective than habitat protection in some circumstances (Possingham, Bode, & Klein, 2015). Furthermore, the risk posed by climate change varies across soil and vegetation types and topographical zones; climate change susceptibility is a critical measure of threat. See Appendix C, S20 for a comprehensive list of data limitations and associated methodological caveats.

The uncertain future of the Tasmanian FPPF lands nonetheless makes for an excellent example to demonstrate the utility of jurisdictionally informed conservation planning, due to the relatively abundant and high-quality spatial data that are publicly available for this region. However, I acknowledge that several key features were notably absent from the cases study prioritisations. Because of the simplicity and flexibility of the jurisdictionally informed framework, additional data can be quickly and easily incorporated into decision-support tools. I suggest three reasonable

features to consider including in future land tenure decision support exercises (but see also Appendix C, S20):

(i) The relative carbon storage benefits of protecting some planning units and re-allocating others (e.g., to the timber production zone) have important implications for climate change, an emerging shaper of ecological and evolutionary processes that underpin and sustain the biodiversity we wish to protect (Klein et al., 2009; McQuillan et al., 2009). In Tasmania, existing estimates of carbon masses contained within Tasmanian forest types and age classes (Moroni, Kelley, & McLarin, 2010) is not yet refined enough to be meaningfully assigned to the FPPF planning units. Data for old-growth forests, sometimes used as a proxy for carbon storage, is not publicly available in a recent version in Tasmania, and old-growth status may not necessarily equate to maximum carbon sequestration (Moroni, Kelley, & McLarin, 2010). Furthermore, the relative carbon storage capacity of some non-forested areas (e.g., peatland and wetland) store relatively large amounts of carbon but are especially vulnerable to land-use changes (Dawson & Smith, 2007). Climate refugia and centres of paleo-endemism should also be included.

(ii) Social equity is an important consideration in land tenure decisions, but I was unable to identify a clear terrestrial analogue for the equity metrics used for marine conservation and fisheries trade-off planning (Halpern et al., 2013) for which there is publicly available spatial data relevant to Tasmania. One possible social equity features to consider in this context are the yet unmet proposals by the Tasmanian Aboriginal Land Council for the return of areas containing cultural heritage sites to the traditional owners (see the Tasmanian Aboriginal Lands Act 1995), although this data is also not publicly accessible.

(iii) The relative economic opportunity costs of protecting some planning units over others are important practical considerations in systematic conservation planning (Naidoo et al., 2006), but I only considered the opportunity cost to one industry (plantation forestry). Alternative reasonable opportunity-cost metrics include land valuations and opportunity cost to the native forestry industry, although the spatial data for native forestry production potential of Tasmanian forests are not open access. If such data were available, the planning units' binary futures (intensive plantation or strict PA) that I defined as the possible futures for the FPPF lands could be broadened. With such data, optimum 'triad' zoning, whereby the

landscape is divided into three management-defining tenures (reserve, low-intensity native forestry, and intensive plantation; see Betts et al., 2021) would be possible within jurisdictionally informed land tenure/management assignment.

The failures of the past decade to achieve a large-scale and representative expansion of the global PA network have made clear the paramount importance of international and national-to-regional coordination in spatial conservation planning and, conversely, the trade-offs invoked by overtly parochial land-tenure decisions. I present jurisdictionally informed conservation planning as a modern solution, leveraging spatial data for all relevant ecological, economic, and social features and still tailoring conservation decision support to local contexts with customizable assignment of importance to each feature of interest. Taking a jurisdictionally informed approach, which can be applied to land tenure decisions in any context, will be critical if we are to learn from the lessons of the past decade of area-based conservation and turn the tide of mass biodiversity loss.

Chapter 5 has been removed for
copyright or proprietary reasons.

It is the following published article:
Lamba, A., Cassey, P., Segaran, R. R., Koh,
L. P., 2019. Deep learning for
environmental conservation. *Current
biology*, 29(19), R977-R982.

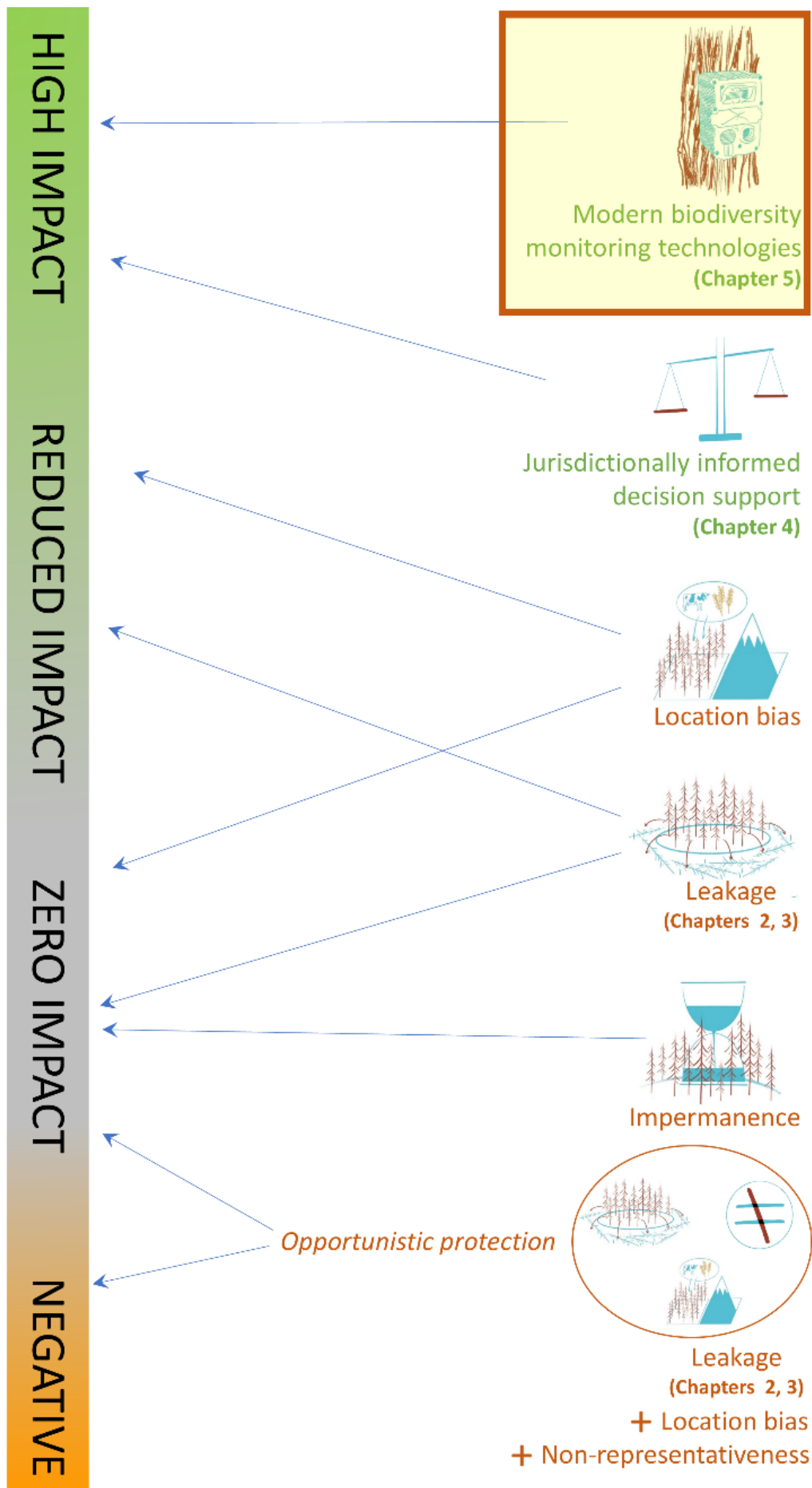


Chapter 5: Integration of targeted, fine-scale biodiversity data to support broad-scale conservation decision making

Great Forester River, Tasmania. Photo courtesy of Amy Coghlan, 2020

“Deep learning is expected to permeate into almost every sphere of life, bringing with it benefits that can potentially save thousands of hours of human effort. Conservation scientists and practitioners need to be ready to ride this wave” (Lamba et al., 2019).⁴

⁴ Lamba, A., Cassey, P., Segaran, R. R., & Koh, L. P. (2019). Deep learning for environmental conservation. *Current Biology*, 29(19), R977-R982.



Chapter 6: General Discussion



Eastern quoll
(*Dasyurus viverrinus*)



Tasmanian devil
(*Sarcophilus harrisii*)



Short-beaked echidna
(*Tachyglossus aculeatus*)

Northeast Tasmania, 2020

I. Aims

Broadly, this thesis has explored pathways to achieving high conservation impacts through the expansion of the global protected area (PA) network. The research outcomes articulated herein provide real-world insights into the relative importance of leakage and globally uncoordinated prioritisation in the context of global conservation

Related to my overall thesis aim, I addressed three objectives:

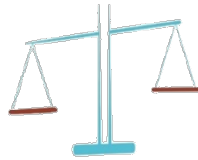
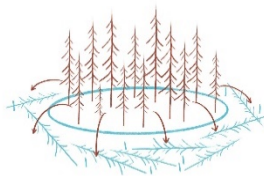
1. To investigate the barriers to achieving conservation impact through the establishment of protected areas.
2. To clarify the appropriate triage of those identified but poorly understood barriers.
3. To test quantitative means of providing data-driven conservation decision support for avoiding or mitigating these barriers.

II. Summary of key findings

My research aims were fulfilled, first through assessment of the prevalence and magnitude of non-target 'leakage' effects in a global systematic review of the literature and within the Brazilian Amazon, using counterfactual methods to measure the target and non-target impacts of recently established PAs on deforestation rates. This quantification of gross impact within and gross impact

outside the new PAs enabled calculation of net impact. Second, the utility of a jurisdictionally informed approach to PA-planning was demonstrated by adapting a conservation decision-support tool to generate a globally effective and coordinated plan for the protection of contested Tasmanian forests. Then, to contextualise the land tenure decisions for this region in the local ecology, I assessed the extent to which common PA-network design criteria (e.g., coverage of bioregions/ecoregions, connectivity, cost efficiency) reflect ecological priorities identified by community diversity and focal species' relative abundances, calculated using data from one year of camera-trap monitoring undertaken at 201 sites. This final study validated the utility of reserve network connectivity and biogeographic representation as reserve design criteria for alignment with mammals' and ground-dwelling birds' species richness and community diversity in northeast Tasmania; it showed vulnerability to human impacts to be a less useful reserve design criteria.

Together, my findings shed light how to shift from opportunistic to data-driven, systematically planned area-based conservation. I affirm that the pervasive possibility of PA impermanence warrants the strongest mitigation efforts: PA downgrading, downsizing, and degazettement (PADDD) events can reduce or completely nullify the habitat conserved by the stricter, larger, or pre-existing reserve, respectively. Of secondary concern are the problems of location bias and associated non-representativeness of PAs in terms of coverage of habitats. Where the likelihood of habitat conversion for human land uses approaches zero, the impact of a protected area on habitat conservation within the landscape also approaches zero (Joppa & Pfaff, 2009). Where location bias is coupled with high-magnitude leakage of human land-use change efforts to areas more critical or irreplaceable for biodiversity, the net impact of a PA for the biodiversity of the landscape can become negative (Bode et al, 2015; Renwick et al., 2015). Except in these cases, local or 'neighbourhood' leakage is unlikely to pose a significant threat to PA effectiveness, at least in the global tropics to which the global review and the Amazonian case study were constrained. More concerning are transnational or trans-biome leakage events (Dou et al., 2018). Where these distantly externalised spillover effects exacerbate existing biases in the global PA network, its representativeness is likely to be poor and heavily biased towards nations capable of importing primary resources (Bruckner et al., 2015; Friis et al., 2016). However, whilst global representation targets (e.g., protection of 30% of each of the global ecoregions) are helpful guidelines, land-tenure decisions must also incorporate local, fine-resolution data such that decisions are aligned with the needs of ecological communities and habitats that irreplaceable endemic taxa depend upon.



III. Quantitative methods within socioecological systems

Environmental management and conservation policy have lagged behind medicine and public health in the methodologies of evidence-based interventions assessed through the units of impact (Pullin & Stewart, 2006) because the complex social and economic factors often confound clear patterns of the causes and effects of land-use changes. Nonetheless, this thesis is quantitative in nature. In each of my chapters I used methods that recognise the complexity of the socioecological systems within which the policy mechanism of PA establishment and the ecological consequences subsequent land-use dynamics are embedded. The introductory components of Chapters 2 through 5 each describe the anecdotal understanding of the conservation problem each addressed; the methods sections explain how these anecdotal accounts were tested from the available evidence-base; the results sections determine if they hold water or are misguided. Specifically, Chapter 2 assesses the validity of the idea that PAs are potentially displacing land-use changes such that their net impact for landscapes is nullified. Chapter 3 corrects the widely claimed fallacy that the difficulty of managing and enforcing such enormous PAs as those recently established in the Brazilian Amazon means they are probably not reducing deforestation rates within their boundaries. Chapter 4 questions the assumption that a proposed extension (more accurately, the addition of a less-strict reserve buffer) to a Wilderness World Heritage Area is inherently a globally effective way to allocate tenure-contested public forests. Chapter 5 critiques and refines the generally accepted wisdom of the goal of equitably representing the established units of bioregionalisation frameworks within a well-connected and cost-efficient PA network.

IV. Research approaches and limitations

In Chapters 2 and 3, I attempted to isolate the impact of effective PAs causing non-target spillover by calculating a baseline of deforestation for each PA separately for each year, capturing year-to-year variation in the other deforestation pressures. However, it is also important to distinguish the contribution of non-PA forest conservation mechanisms (e.g., environmental legislation and supply chain regulations agreed upon by multinational corporations and governments) to any reduced deforestation rates observed. For example, two supply-chain agreements implemented to reduce

deforestation in the Amazon, the Soy Moratorium and zero-deforestation beef agreement, have been found to have successfully reduced deforestation in the target region at the cost of increased clearing of native vegetation in the Cerrado region to the south (Dou et al., 2018). The telecoupling framework and economic supply-chain analyses required to quantify such distantly externalised spillover effects are beyond the scope of this thesis. It is nonetheless important to measure the local manifestations of leakage, as I did in Chapters 2 and 3, especially considering that deforestation displaced from the PA is likely to occur nearby because adjacent areas are likely characterized by the same accessibility and other features as the protected land in question.

That said, the estimation of a baseline deforestation rate from random sample parcels within the landscape, as demonstrated in Chapter 3, does not account for other untested environmental and anthropogenic factors that could be driving intensity of deforestation (or other human development) pressures. Statistical tools like propensity score matching (Andam et al., 2008; Chomitz, 2002; Joppa and Pfaff, 2010) and panel regression techniques (Jones and Lewis, 2015) could improve these counterfactual baseline estimates. My study attempted to account for this possible variation in likelihood of clearing by using two baselines: one highly localized to minimize environmental differences, and one state-based to minimize differences in anthropogenic factors. My strategy may not have accounted for all possible confounding effects (Ferraro and Pattanayak, 2006); the true counterfactual annual deforestation that would have occurred if no PA had ever been established cannot, by definition, be known but only estimated.

The downgrading of existing PAs in Tasmania, the conservation values of which were assessed in Chapter 4 and Chapter 5, can be considered an example of state-sanctioned leakage of timber resource extraction. The limited supply of areas within which native forestry is permitted on the well-protected island state of Tasmania caused the State Government to declare these buffer zones adjacent the existing reserves to be potential timber extraction zones. While this thesis did not attempt to address the existence of legislated, state-sanctioned blockage (positive spillover) around protected areas, there is an obvious example of such legislated blockage: since their inception in 1971 through UNESCO, 727 official biosphere reserves have been declared in 131 countries. Biosphere reserves typically have a strictly protected core area, a buffer zone within which limited human activities are allowed, and a transition area where people live in ecologically sustainable ways.

The major inference of Chapter 4 was limited by the lack of high-resolution spatial data to provide values or relative rankings of features that are direct analogues to the international, national, and global goals for conservation, as conservation targets are not often clearly quantitative. It is critically important that policymakers set quantitative targets such that success can

be measured. Data deficiencies, confounded by biases in the locations and taxa that receive research effort, are prominent limitations in any conservation planning and management scenario, and Chapter 4 was no exception. Conservation scientists may find themselves in a perverse cycle whereby funding is needed to collect monitoring data, but it is challenging to convince funding sources of the need for the research without data to back up the pitch; ensuring that data (and software tools) are publicly available at no costs is a critical strategy for exiting the cycle (Hoffman, 2021).

Furthermore, as anthropogenic pressures and the ecological responses to them are not static, consistent monitoring over long time periods is needed (Larson et al, 2013). Chapter 5 sought to fill a small component of this gap in knowledge from monitoring efforts, but the taxa captured by camera traps were primarily the large-bodied, charismatic ones (mammals) that already receive a disproportionate of conservation attention and monitoring (Berti et al., 2020). Long-term monitoring of plants and invertebrates would lead to a better understanding of local patterns of endemism and biogeographically distinct forest patches in Tasmania (McQuillan et al., 2009).

V. Opportunities for future research and for policy application

The Brazilian Amazon and Tasmania are unique in that a relatively large proportion of their area is under some degree of reserve tenure. The Brazilian Amazon is not officially listed as a global biodiversity hotspot because threats are still kept in check by its (albeit dwindling remoteness). Non-target impacts of recent PAs should be quantified also for the actual two biodiversity hotspots found within these countries – the Cerrado savanna to the south and the highly fragmented, highly urbanised Atlantic Forest of the east coast of Brazil, and the southwest coast and east coast of the Australian mainland. Tasmania is an unusually successful conservation story even within the Australian context, and means of mitigating high rates of land clearance on the Australian mainland's biologically rich east coast should be a priority. It is important to assess the contribution of leakage and of uncoordinated spatial prioritisation in these regions. Furthermore, the land-use dynamics surrounding the vast Indigenous Protected Areas (IPAs) found in both Brazil and Australia should be monitored through quantitative impact assessments without losing sight of the socio-cultural and bio-cultural benefits of the legal establishment and enforcement of these non-State managed PAs.

Policymakers should therefore primarily work to ensure the permanence, representativeness, and the prioritisation of highly vulnerable (e.g., agriculturally suitable) areas within conserved area networks that include State-governed and Indigenous-governed zones.

Conservation scientists, as mentioned in the previous section, should work towards quantifying trans-biome and trans-national leakage events. Such research will involve complex tele-coupling frameworks and transdisciplinary unravelling of convoluted, rarely transparent supply chains and market forces (Bruckner et al., 2015; Dou et al., 2018; Friis et al., 2016). It is increasingly important to embrace the complexities of such trans-disciplinary research, despite its unique challenges, as we move deeper into the Anthropocene.

My adaptation and application of the simple z-score based ranking tool (Ondei et al., 2019) for planning the future of contested public lands' tenure, and thus their future management, was a useful demonstration of a more globally coordinated approach to futures thinking for PA network spatial prioritisation. However, I did not attempt retrospective assessment of how existing PA-tenure allocations align with optimal conservation target areas, as Pouzols et al. did (2014). Here I see an opportunity to use the prevalence of "paper parks" (Di Minin & Toivonen, 2015) and of PA de-gazettement events to our advantage to become more biospherically minded conservation planners. Ineffective PAs, whether they be paper parks or PAs located in places very unlikely to experience land-use changes, could be effectively "swapped out" to justify the creation of new PAs that would more effectively contribute to biodiversity conservation. Such unconventional approaches may be necessary to achieve conservation goals where funding and political will are limited.

Policymakers should take advantage of conservation decision support tools adapted to produce more jurisdictionally informed results, if they are to meaningfully expand the global PA network in response to the 30x30 target of the post-2020 Biodiversity Framework of the Convention on Biological Diversity (CBD). Conservation scientists should expand the frontiers of real-time ecological monitoring with the synergistic application of modern camera trapping hardware, machine- and statistical-learning methods, and data-driven conservation decision support tools.

VI. Concluding remarks

Area-based conservation offers solutions to the defining problems of the Anthropocene: protected and conserved areas are the preferred mechanism for tackling the twin crises of biodiversity loss and climate change (Hagerman and Satterfield 2014). The national reports on signatories' implementation of conservation's most important global policy mechanism, the Convention on Biological Diversity, submitted for the decadal review of the Strategic Plan for Biodiversity 2011-2020, collectively indicate slow progress. However, progress occurred at levels generally insufficient to achieve the Aichi Biodiversity Targets, including Target 11 (Secretariat of the Convention on Biological Diversity, 2020). Although there has been significant expansion over the 2000-2020

period, the fifth report of the Global Biodiversity Outlook deems progress towards Target 11 to have been “modest” regarding the extent to which it safeguards the most important areas for biodiversity, is ecologically representative, is connected, and is made up of are equitably and effectively managed PAs. This decade’s area-based conservation efforts must be more than modest if we are to turn the tide of mass biodiversity loss: we must actively learn from the lessons of the past decade. The adherence of CBD signatory parties to the goals outlined by the post-2020 Biodiversity Framework, particularly to targets set for PA network expansion, could change the trajectory of the current mass extinction crisis. The known and emerging threats to effective area-based conservation must be formally acknowledged, quantified, triaged, and ultimately avoided or mitigated through a synergy of conservation science, planning, and practice.

There is a largely undisputed need to shift conservation planning, target setting, management, and progress reporting to impact-based measures, where impact is defined as the sum of avoided biodiversity loss and promoted recovery relative to outcomes without protection (Pressey et al. , 2017; Pressey et al., 2021). I conclude the thesis by urging conservation scientists to urgently shift the planning, assessment, and reporting of conservation interventions to the units of impact. Furthermore, conservation science must embrace and leverage modern technologies to identify and prioritise likely climate refuges and refugia, and conservation practitioners must act to expand and intensify effective private-land conservation mechanisms where public policies do not apply.

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Appendix A: Chapter 2 Supplementary materials

Table S1. Additional details of the seven empirical studies that were included in the systematic review. All papers acknowledged spillover and provided extractable deforestation data.

Authors	Year of publication	Journal	Title	Number of PAs
Spracklen, B.D., Kalamandeen, M., Galbraith, D., Gloor, E. & Spracklen, D.V.	2015	<i>PLoS ONE</i>	A global analysis of deforestation in moist tropical forest protected areas	3,374
Ament, J.M. & Cumming, G.S.	2016	<i>Conservation Biology</i>	Scale dependency in effectiveness, isolation, and social-ecological spillover of protected areas	8
Apan, A., Suarez, L.A., Maraseni, T. & Castillo, J.A.	2017	<i>Applied Geography</i>	The rate, extent and spatial predictors of forest loss (2000–2012) in the terrestrial protected areas of the Philippines	7
Sanchez-Azofeifa, G.A., Daily, G.C., Pfaff, A.S.P. & Busch, C.	2003	<i>Biological Conservation</i>	Integrity and isolation of Costa Rica's national parks and biological PAs: examining the dynamics of land-cover change	6
Dewi, S., van Noordwijk, M., Ekadinata, A. & Pfund, J.L.	2013	<i>Land Use Policy</i>	Protected areas within multifunctional landscapes: Squeezing out intermediate land use intensities in the tropics?	4
Phua, M.H., Tsuyuki, S., Furuya, N. & Lee, J.S.	2008	<i>Journal of Environmental Management</i>	Detecting deforestation with a spectral change detection approach using multitemporal Landsat data: A case study of Kinabalu Park, Sabah, Malaysia	1
Sánchez-Azofeifa, G.A., Quesada, M., Cuevas-Reyes, P., Castillo, A. & Sánchez-Montoya, G.	2009	<i>Forest Ecology and Management</i>	Land cover and conservation in the area of influence of the Chamela-Cuixmala Biosphere PA, Mexico	1

Table S2. Data set extracted from the results of the literature search with additional meta-data.

Visit <https://www.sciencedirect.com/science/article/pii/S2351989419301143#appsec1> to download .xlsx file.

Table S3. List of 823 protected areas (PAs) excluded from spillover analysis due to lack of evidence of deforestation restriction (.xlsx)

Visit <https://www.sciencedirect.com/science/article/pii/S2351989419301143#appsec1> to download .xlsx file.

Table S4. Summary of the IUCN categories of 823 PAs excluded from spillover analysis for lack of evidence of deforestation restriction. The majority (67.1%) of these “paper parks” fall within the category of multiple-use (IUCN V or VI) reserves, and as such, extraction of timber and other forest products and/or forest clearing for some land uses may be legally permitted (see McDonald and Boucher 2011; Naughton-Treves et al. 2005; Shafer 2015). This is a possible explanation for the relatively high deforestation rates reported within these PAs.

IUCN Category	# PAs that did not reduce deforestation relative to the broader landscape	% (of 489) ineffective PAs for which IUCN category is known
1	49	9.8%
2	40	8.0%
3	7	1.4%
4	68	13.7%
5	255	51.2%
6	79	15.9%
Total Known	489	
Unknown	325	
Total	1312	
Strictly conserved, i.e., IUCN I-IV	164	32.9%
Multiple-use, i.e., IUCN V-VI	334	67.1%

Table S5. Coefficients of the predictors included in the best model for leakage and blockage magnitudes, respectively. Model dimension was selected by k-fold cross-validation.

		Leakage		Blockage	
		Complete cases of 15 variables	Complete cases of 13 variables	Complete cases of 15 variables	Complete cases of 13 variables
Number of observations		165	395	549	1,443
%Deviance explained		14.4	13.4	17.3	15.9
PA area					
PA age upon deforestation monitoring			NA		NA
Land-use restriction type			NA		NA
Population density		-0.000418147	-0.004849274	-0.004728695	0.003611273
Population growth rate		1.65194806			0.814584507
Rural population growth rate		-0.480372808		-0.139139626	-
GDP					6.40458E-13
GDP growth rate		-0.010071738		-0.153199933	-
Ratio: % land area under agriculture to forest		0.068612444			0.162384078
Value of forest products removed in 1 year		2.43088E-10			
Value of forest products removed per ha			0.017691559	0.006621321	
Carbon dioxide emissions					-1.92722E-07
Percentage of energy supply from biofuels					
Annual conservation spending				-0.002382286	-
Continent					0.028176069
	Africa				
	Asia			0.944604537	1.065018265
	Australia & Oceania				7.773005962
	North America				
	South America				

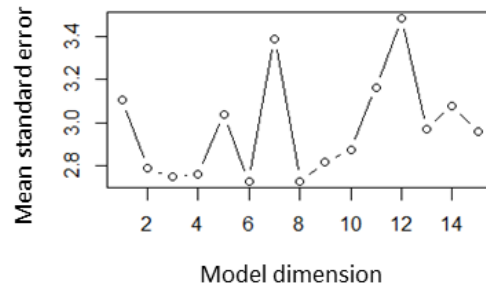
Table S6. Summary of coefficient estimates, standard error, t values, percent of deviance explained for the saturated model with all potential predictors as coefficients, and percent of deviance explained by the model whose dimension was chosen by k-fold cross-validation. Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘.’

Model Input	Predictor	Estimate	Std. Error	t value	Significance code	%DE of saturated model	%DE of best model
Complete cases of leakage observations given 15 potential predictors	Population density	-0.000418147	0.001266788	-0.33		39.7	14.4
	Population growth rate	1.65194806	0.32748357	-0.33	***		
	Rural population growth rate	-0.480372808	0.137191238	-3.501	***		
	GDP growth rate	-0.010071738	0.101363314	-0.099			
	Ratio: % land area under agriculture to forest	0.068612444	0.066069232	1.038			
	Value of forest products removed in 1 year	2.431E-10	1.044E-10	2.328	*		
Complete cases of leakage observations given 13 potential predictors	Population density	-0.0048493	0.0008451	-5.738	***	20.0	13.4
	Value of forest products removed per ha	0.0176916	0.0025024	7.07	***		
Complete cases of blockage observations given 15 potential predictors	Population density	-0.0048807	0.0007461	-6.542	***	21.6	17.3
	Rural population growth rate	-0.0935124	0.0987659	-0.947			
	GDP growth rate	-0.1418354	0.0557482	-2.544	*		
	Value of forest products removed per ha	0.0076507	0.0026777	2.857	**		
	Annual conservation spending	0.0004554	0.0082243	0.055			
	PA located within Asia	1.2237164	0.4013176	3.049	**		
Complete cases of blockage observations given 13 potential predictors	Population density	-0.003745087	0.000720227	-5.2	***	17.4	15.9
	Population growth rate	0.887553465	0.175426892	5.059	***		
	Rural population growth rate	-0.466955745	0.060156101	-7.762	***		
	GDP	7.538E-13	2.074E-13	3.634	***		
	GDP growth rate	-0.044596715	0.054032052	-0.825			
	Ratio: % land area under agriculture to forest	0.19282719	0.054427015	3.543	***		
	Carbon dioxide emissions	-2.76046E-07	1.43598E-07	-1.922			
	Annual conservation spending	-0.028774647	0.011520875	-2.498	*		
	PA located within Asia	1.258087318	0.240395871	5.233	***		
	PA located within Australia & Oceania	8.08986301	3.400663239	2.379	*		

Figure S7. Relationship between mean standard error and model dimension as predicted by k-fold cross-validation

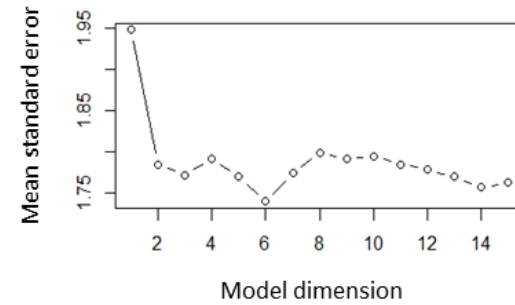
Dependent variable: leakage magnitudes

Best model dimension $n=6$ given 15 potential predictor variables



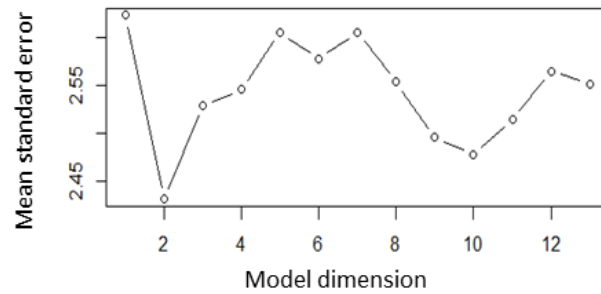
Dependent variable: blockage magnitudes

Best model dimension $n=6$ given 15 potential predictor variables



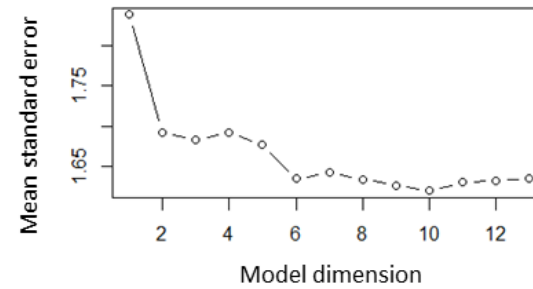
Dependent variable: leakage magnitudes

Best model dimension $n=2$ given 13 potential predictor variables



Dependent variable: blockage magnitudes

Best model dimension $n=10$ given 13 potential predictor variables



Appendix B: Chapter 3 Supplementary materials

Table S1. Potential spillover predictor variables and PA downgrading, downsizing, and degazettement (PADDD) events associated with each of 91 protected areas (PAs): age in 2017, state, area (km^2), percent of PA's spillover zone that overlaps existing PAs or Indigenous Lands, administrative governance tier, and Brazilian PA classification category. Areas vary widely across the sample of 91 PAs: 25 PAs are smaller than 100 km^2 , 12 are between 100 and 1000 km^2 , 43 are between 1000 and 10 000 km^2 , and 11 are larger than 10 000 km^2 . PADDD was proposed for 13 PAs and downsizing was enacted in two PAs during the study period 2005-2017.

Protected area (PA) Name	PA Age in 2017	State	PA Area (km^2)	Percent of spillover zone overlapping other PA(s) or Indigenous Lands	Government Tier	Classification within the Brazilian PA system	PA downgrading, downsizing, or degazettement Event	Proposed or enacted	Year of PADDD
Área de Proteção Ambiental Lago do Amapá	12	Acre	53.24	0.00%	state	Sustainable use	NA	NA	NA
Estação Ecológica Terra do Meio	12	Para	33732.40	21.81%	federal	Strictly protected	Downgrade	Proposed	2006
Floresta Estadual Apuí	12	Amazonas	1833.33	0.00%	state	Sustainable use	Degazette	Proposed	2005
Floresta Estadual Aripuanã	12	Amazonas	3304.83	14.50%	state	Sustainable use	NA	NA	NA
Floresta Estadual Manicoré	12	Amazonas	838.92	25.95%	state	Sustainable use	NA	NA	NA
Floresta Estadual Sucunduri	12	Amazonas	4831.09	25.28%	state	Sustainable use	NA	NA	NA
Floresta Nacional de Anauá	12	Roraima	2612.83	8.79%	federal	Sustainable use	NA	NA	NA
Floresta Nacional de Balata-Tufari	12	Amazonas	10961.10	21.84%	federal	Sustainable use	NA	NA	NA
Parque Estadual Guariba	12	Amazonas	715.76	32.46%	state	Strictly protected	NA	NA	NA
Parque Estadual Sucunduri	12	Amazonas	7986.88	16.30%	state	Strictly protected	NA	NA	NA
Parque Nacional da Serra do Pardo	12	Para	4454.64	0.06%	federal	Strictly protected	Downsize	Proposed	2006
Reserva Biológica Nascentes da Serra do Cachimbo	12	Para	3422.16	30.47%	federal	Strictly protected	Downgrade	Proposed	2016
Reserva de Desenvolvimento Sustentável de Itatupã-Baquiaí	12	Para	645.11	1.59%	federal	Sustainable use	NA	NA	NA
Reserva De Desenvolvimento Sustentável Bararati	12	Amazonas	1111.25	0.00%	state	Sustainable use	NA	NA	NA

Reserva de Desenvolvimento Sustentável do Rio Amapá	12	Amazonas	2163.00	27.82%	state	Sustainable use	NA	NA	NA
Reserva de Desenvolvimento Sustentável Canumã	12	Amazonas	228.43	39.18%	state	Sustainable use	NA	NA	NA
Reserva de Desenvolvimento Sustentável de Uacari	12	Amazonas	6379.34	16.27%	state	Sustainable use	NA	NA	NA
Reserva de Desenvolvimento Sustentável Aripuanã	12	Amazonas	2191.45	7.36%	state	Sustainable use	NA	NA	NA
Reserva Extrativista Ariaca Pruno	12	Para	840.22	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista do Guariba	12	Amazonas	1489.83	11.09%	state	Sustainable use	NA	NA	NA
Reserva Extrativista Ipaú-Anilzinho	12	Para	559.88	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Mapuá	12	Para	939.67	79.11%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Riozinho da Liberdade	12	Acre	3408.83	22.71%	federal	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Klagesi	12	Para	0.23	0.00%	federal	Sustainable use	NA	NA	NA
Área de Proteção Ambiental Baixo Rio Branco	11	Roraima	15780.00	65.85%	state	Sustainable use	NA	NA	NA
Área de Proteção Ambiental Nascentes do Rio Paraguai	11	Mato Grosso	709.18	0.00%	state	Sustainable use	NA	NA	NA
Área de Proteção Ambiental do Tapajós	11	Para	20425.70	22.67%	federal	Sustainable use	Degazette	Proposed	2006
Floresta Estadual do Paru	11	Para	36102.90	35.09%	state	Sustainable use	Downsize	Enacted	2012
Floresta Estadual do Trombetas	11	Para	31472.20	14.09%	state	Sustainable use	NA	NA	NA
Floresta Nacional do Amaná	11	Para	6839.50	32.10%	federal	Sustainable use	Degazette	Proposed	2006
Floresta Nacional do Crepori	11	Para	7414.71	28.56%	federal	Sustainable use	Degazette	Proposed	2006
Floresta Nacional de Jamanxim	11	Para	13021.80	0.00%	federal	Sustainable use	Degazette	Proposed	2006
Floresta Nacional do Trairão	11	Para	2576.27	28.88%	federal	Sustainable use	Degazette	Proposed	2006
Monumento Natural Morro de Santo Antonio	11	Mato Grosso	2.58	0.00%	state	Strictly protected	NA	NA	NA
Parque Nacional de Jamanxim	11	Para	8633.05	40.80%	federal	Strictly protected	Degazette	Proposed	2006
Parque Nacional do Juruena	11	Amazonas	19630.40	60.80%	federal	Strictly protected	NA	NA	NA
Parque Nacional do Rio Novo	11	Para	5386.02	15.95%	federal	Strictly protected	Degazette	Proposed	2006
Parque Nacional dos Campos Amazônicos	11	Amazonas	9687.72	15.90%	federal	Strictly protected	Degazette	Proposed	2008
Reserva Biológica Maicuru	11	Para	11731.50	58.53%	state	Strictly protected	NA	NA	NA

Reserva de Desenvolvimento Sustentável do Rio Madeira	11	Amazonas	2814.43	0.00%	state	Sustainable use	NA	NA	NA
Reserva de Desenvolvimento Sustentável do Juma	11	Amazonas	5842.91	0.00%	state	Sustainable use	NA	NA	NA
Reserva Extrativista Arapixi	11	Amazonas	1375.54	46.10%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Rio Unini	11	Amazonas	8610.93	52.00%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Gurupá-Melgaço	11	Para	1455.73	5.88%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Rio Iriri	11	Para	3989.91	56.23%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Terra Grande-Pracuúba	11	Para	1953.44	99.30%	federal	Sustainable use	NA	NA	NA
Reserva Particular do Patrimônio Natural do Cristalino III	10	Mato Grosso	16.18	15.12%	state	Sustainable use	NA	NA	NA
Floresta Nacional do Iquiri	9	Amazonas	15056.70	20.86%	federal	Sustainable use	NA	NA	NA
Parque Nacional Mapinguari	9	Amazonas	18070.60	22.14%	federal	Strictly protected	Downgrade	Enacted	2010
Parque Nacional Nascentes do Lago Jari	9	Amazonas	8219.38	40.90%	federal	Strictly protected	NA	NA	NA
Reserva de Desenvolvimento Sustentável do Rio Negro	9	Amazonas	1036.90	85.73%	state	Sustainable use	NA	NA	NA
Reserva Extrativista do Médio Purus	9	Amazonas	6174.88	49.26%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Ituxi	9	Amazonas	7911.67	20.84%	federal	Sustainable use	NA	NA	NA
Reserva Extrativista Rio Xingu	9	Para	3030.60	91.57%	federal	Sustainable use	NA	NA	NA
Área De Relevante Interesse Ecológico Japiim Pentecoste	8	Acre	270.33	13.94%	state	Sustainable use	NA	NA	NA
Floresta Estadual Canutama	8	Amazonas	1530.66	0.00%	state	Sustainable use	NA	NA	NA
Floresta Estadual Tapauá	8	Amazonas	8933.66	39.03%	state	Sustainable use	NA	NA	NA
Parque Estadual do Matupiri	8	Amazonas	5136.61	27.91%	state	Strictly protected	NA	NA	NA
Reserva de Desenvolvimento Sustentável do Matupiri	8	Amazonas	1781.91	37.65%	state	Sustainable use	NA	NA	NA
Reserva De Desenvolvimento Sustentável Igapó-Açu	8	Amazonas	3980.67	12.56%	state	Sustainable use	NA	NA	NA
Reserva Extrativista Canutama	8	Amazonas	2013.49	48.83%	state	Sustainable use	NA	NA	NA
Reserva Extrativista Renascer	8	Para	2096.67	33.42%	federal	Sustainable use	Downsize	Proposed	2013
Reserva Particular Do Patrimônio Natural Osório Reimão	8	Para	0.09	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Particular do Patrimônio Natural Fazenda Calixto	8	Tocantins	3.46	0.00%	federal	Sustainable use	NA	NA	NA

Reserva Particular do Patrimônio Natural Ilhas Alexandre Rodrigues Ferreira	8	Amazonas	3.06	99.69%	federal	Sustainable use	NA	NA	NA
Parque Estadual do Charapucu	7	Para	654.69	75.38%	state	Strictly protected	NA	NA	NA
Refúgio de Vida Silvestre Metrópole da Amazônia	7	Para	64.01	8.09%	state	Strictly protected	NA	NA	NA
Reserva Particular Do Patrimônio Natural Peugeot-Onf-Brasil	7	Mato Grosso	17.65	0.00%	state	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Catedral Do Jalapão	7	Tocantins	3.28	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Sonhada	7	Tocantins	8.89	0.00%	federal	Sustainable use	NA	NA	NA
Área De Proteção Ambiental Da Serra Do Saubal	6	Para	1.67	0.00%	municipal	Sustainable use	NA	NA	NA
Parque Natural Municipal Veredas de Carajás	6	Para	8.33	0.00%	municipal	Strictly protected	NA	NA	NA
Reserva Particular Do Patrimônio Natural Fazenda Loanda	6	Mato Grosso	5.15	0.00%	state	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Cristalino I	6	Mato Grosso	24.52	70.36%	state	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Bico do Javali	6	Tocantins	27.69	34.14%	federal	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Nova Aurora	6	Rondonia	0.19	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Irmãos Satelis	5	Rondonia	0.41	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Gibeão	5	Rondonia	0.32	0.00%	federal	Sustainable use	NA	NA	NA
Reserva Particular Do Patrimônio Natural Aurora Natural	5	Tocantins	0.15	0.00%	federal	Sustainable use	NA	NA	NA
Parque Natural Municipal Parque Florestal De Sinop	3	Mato Grosso	1.06	25.12%	municipal	Strictly protected	NA	NA	NA
Reserva de Desenvolvimento Sustentável Puranga Conquista	3	Amazonas	865.60	77.68%	state	Sustainable use	NA	NA	NA
Área de Proteção Ambiental dos Campos de Manicoré	1	Amazonas	1536.25	27.62%	federal	Sustainable use	NA	NA	NA
Floresta Nacional de Urupadi	1	Amazonas	5396.34	10.42%	federal	Sustainable use	NA	NA	NA
Floresta Nacional do Aripuanã	1	Amazonas	7561.95	9.75%	federal	Sustainable use	NA	NA	NA
Parque Nacional do Acari	1	Amazonas	9002.10	1.98%	federal	Strictly protected	NA	NA	NA
Refúgio De Vida Silvestre Lago Mole	1	Para	6.53	0.00%	municipal	Strictly protected	NA	NA	NA
Refúgio de Vida Silvestre Tabuleiro do Embaubal	1	Para	40.38	0.00%	state	Strictly protected	NA	NA	NA

Refúgio De Vida Silvestre Padre Sérgio Tonetto	1	Para	3.41	0.00%	state	Strictly protected	NA	NA	NA
Reserva Biológica do Manicoré	1	Amazonas	3617.79	0.73%	federal	Strictly protected	NA	NA	NA
Reserva de Desenvolvimento Sustentável Campo das Mangabas	1	Para	70.93	39.06%	state	Sustainable use	NA	NA	NA
Reserva de Desenvolvimento Sustentável Vitória de Souzel	1	Para	229.83	0.00%	state	Sustainable use	NA	NA	NA

Table S2. Predictor variables tested for relationship with spillover outcome and the hypotheses tested by including them in the model selection. PA age and area are likely to affect deforestation dynamics in most geographic contexts. The percent overlap with other PAs and/or Indigenous Lands, Amazonian state, governmental administrative tier, and land-use restriction type are specific to the context of the Brazilian PA system.

Potential predictor	Predictor type	Hypotheses	Included as a coefficient in best model of spillovers calculated with locally derived baselines?	Included as a coefficient in best model of spillovers calculated with state-derived baselines?
PA age in 2017 (years)	Integer	Whether the direction of the effect of nearby protection is leakage or blockage, older PAs may have a stronger effect on their unprotected adjacent surroundings.	Yes – older PAs were associated with more negative spillover values, i.e. blockages.	Yes – older PAs were associated with more negative spillover values, i.e., blockages.

PA area (km^2)	Continuous numeric		Larger PAs may generate more blockages and higher magnitude blockages than smaller PAs because they are more likely to discourage investment in regional infrastructure for development.	No	No
Percent overlap of spillover zone with other PAs and/or with Indigenous Lands	Continuous numeric		The proportion of the spillover zone already under designation as a PA or as an Indigenous Land may generate false blockage outcomes where there was less deforestation than expected in the spillover zone (Nolte et al. 2013; Nepstad et al. 2006).	No	No
State in which PA is located	Categorical (7-level)	Amazonas (n=38) Pará (n=33) Mato Grosso (n=7) Tocantins (n=5) Acre (n=3) Rondônia (n=3) Roraima (n=2)	Some states may be associated with a disproportionately high number of leakage outcomes if development pressures are high and land-use restrictions within PAs are successfully enforced; I expect that this may be the case for Mato Grosso, Rondônia, and Pará because more than 90% of deforestation in the Amazônia Legal from 2001 to 2003 occurred in these three states (Ferreira et al. 2005; Pfaff et al. 2014; Pfaff 2015b).	No	Yes – PA location within the state of Amazonas was significantly associated with a positive model coefficient, i.e., leakage outcomes.
Public administrative tier of PA government	Categorical (3-level)	Federal (n=47) State (n=40) Municipal (n=4)	Different administrative tiers might experience varying degrees of success in enforcing land-use restrictions; those that enforce restrictions more successfully are more likely to encourage leakages (García 2015; García et al. 2019).	No	No

Land-use restriction type in Brazilian PA classification system	Categorical (2-level)	Conservation unit (n=23) Sustainable use area (n=68)	Conservation units will more often be associated with leakage outcomes than will sustainable-use areas, because protection is more strictly and successfully enforced (Nolte et al. 2013; Pfaff 2015b) and therefore more intensively displaces deforestation pressures.	No	Yes – sustainable use PAs were associated with negative model coefficients, i.e., blockage outcomes.
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Table S3. Baseline deforestation rates for each year 2005–2017 derived from the unprotected, PRODES-monitored portion of each state.

State	Baseline deforestation (%)												
	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Mato Grosso	1.14	0.35	0.24	0.45	0.11	0.11	0.14	0.10	0.15	0.15	0.20	0.19	0.19
Amazonas	0.06	0.06	0.02	0.06	0.03	0.07	0.05	0.04	0.04	0.05	0.06	0.08	0.08
Acre	0.27	0.34	0.25	0.38	0.21	0.30	0.32	0.29	0.19	0.30	0.07	0.27	0.78
Rondônia	2.34	0.79	0.55	0.71	0.28	0.27	0.47	0.41	0.57	0.46	0.55	0.69	0.73
Pará	0.78	0.82	0.52	0.91	0.54	0.55	0.42	0.26	0.30	0.25	0.33	0.37	0.39
Tocantins	0.16	0.01	0.01	0.04	0.02	0.02	0.01	0.02	0.02	0.02	0.02	0.02	0.01
Roraima	0.00	0.27	0.19	0.90	0.15	0.30	0.17	0.14	0.20	0.25	0.21	0.30	0.15

Table S4. List of the non-target area impact (i.e., spillover) types and magnitudes of the establishment of 91 PAs in the Amazônia Legal since 2005, calculated using a baseline derived locally from the deforestation observed in nearby random landscape parcels and calculated using a baseline derived from the deforestation observed in the unprotected portion of the state in which the PA is located. PA names with an asterisk indicate that the net spillover outcome calculated was different depending on which baseline method was used, although only six discrepancies are notable given that the others were cases of paper parks for which the spillover outcome is irrelevant. Also listed for each PA is the ratio of the calculated net target impact to the calculated net spillover. Only 14 (if considering calculations using the local baseline) or 9 (if considering calculations that used the state-derived baselines) of 91 PAs had a ratio of target to non-target impacts of less than 1, i.e., only these PAs were calculated to have larger non-target impacts than target impacts. It is reasonable that most PAs' target impacts were nearly equivalent to or larger in magnitude than their non-target impacts. NB: This table reports spillover values for all PAs, including those that did not impact their target area; spillover values for these possible paper parks were not included in analyses.

PA Name	Results with locally derived baselines			Results with state-derived baseline		
	Spillover outcome type	Spillover value	Ratio of target impact to spillover	Spillover outcome type	Spillover value	Ratio of target impact to spillover
Área de Proteção Ambiental Lago do Amapá	blockage	-2.59%	1.30	blockage	-2.06%	0.97
Estação Ecológica Terra do Meio	blockage	-0.49%	6.82	blockage	-2.58%	2.34
Floresta Estadual Apuí	NA	NA	NA	blockage	-0.63%	1.09
Floresta Estadual Aripuanã	NA	NA	NA	blockage	-0.63%	1.08
Floresta Estadual Manicoré	blockage	-6.46%	0.99	blockage	-0.63%	1.00
Floresta Estadual Sucunduri	NA	NA	NA	blockage	-0.63%	1.08
Floresta Nacional de Anauá	blockage	-1.09%	2.11	blockage	-2.02%	1.60
Floresta Nacional de Balata-Tufari	leakage	0.67%	-0.11	leakage	0.16%	-3.86
Parque Estadual Guariba	NA	NA	NA	blockage	-0.63%	1.09
Parque Estadual Sucunduri	NA	NA	NA	blockage	-0.62%	1.09
Parque Nacional da Serra do Pardo	leakage	0.22%	-36.81	leakage	2.96%	-1.93
Reserva Biologica Nascentes da Serra do Cachimbo*	leakage	3.74%	0.68	blockage	-1.83%	1.54
Reserva de Desenvolvimento Sustentável de Itatupã-Baquiá	blockage	-0.14%	1.00	blockage	-5.65%	1.14
Reserva De Desenvolvimento Sustentável Bararati	NA	NA	NA	blockage	-0.63%	1.09

Reserva de Desenvolvimento Sustentável do Rio Amapá	blockage	-0.09%	1.42	blockage	-0.59%	1.16
Reserva de Desenvolvimento Sustentável Canumã	blockage	-1.42%	1.07	blockage	-0.07%	3.46
Reserva de Desenvolvimento Sustentável de Uacari*	no spillover	0.00%	2.17	blockage	-0.61%	1.08
Reserva de Desenvolvimento Sustentável Aripuanã	blockage	-7.24%	1.00	blockage	-0.63%	1.09
Reserva Extrativista Ariaca Pruano	blockage	-0.12%	-6.96	blockage	-4.33%	0.97
Reserva Extrativista do Guariba	NA	NA	NA	blockage	-0.63%	1.09
Reserva Extrativista Ipaú-Anilzinho	blockage	-3.08%	0.83	blockage	-3.89%	1.06
Reserva Extrativista Mapuá	blockage	-0.01%	-11.41	blockage	-5.54%	1.12
Reserva Extrativista Riozinho da Liberdade	blockage	-0.55%	1.37	blockage	-3.22%	1.14
Reserva Particular Do Patrimônio Natural Klagesi*	no spillover	0.00%	NA	blockage	-5.65%	1.14
Área de Proteção Ambiental Baixo Rio Branco	blockage	-2.77%	1.01	blockage	-2.92%	1.10
Área de Proteção Ambiental Nascentes do Rio Paraguai*	leakage	0.04%	-1.12	blockage	-1.95%	1.22
Área de Proteção Ambiental do Tapajós*	leakage	1.72%	0.43	blockage	-2.62%	1.68
Floresta Estadual do Paru	blockage	-2.55%	1.36	blockage	-3.82%	1.45
Floresta Estadual do Trombetas	NA	NA	NA	blockage	-4.45%	1.27
Floresta Nacional do Amaná	NA	NA	NA	blockage	-3.79%	1.39
Floresta Nacional do Crepori	NA	NA	NA	blockage	-4.10%	1.35
Floresta Nacional de Jamanxim	leakage	5.03%	0.66	leakage	1.94%	0.00
Floresta Nacional do Trairão	blockage	-4.05%	1.79	blockage	-1.25%	4.17
Monumento Natural Morro de Santo Antonio*	no spillover	0.00%	NA	blockage	-2.03%	1.17
Parque Nacional de Jamanxim	blockage	-8.16%	1.22	blockage	-2.56%	2.00
Parque Nacional do Juruena	NA	NA	NA	leakage	0.17%	-3.23
Parque nacional do rio novo	blockage	-3.21%	1.15	blockage	-4.01%	1.31
Parque Nacional dos Campos Amazônicos	NA	NA	NA	leakage	3.34%	-0.10
Reserva Biológica Maicuru	NA	NA	NA	blockage	-4.83%	1.17
Reserva de Desenvolvimento Sustentável do Rio Madeira	blockage	-0.21%	1.46	blockage	-0.33%	1.42
Reserva de Desenvolvimento Sustentável do Juma*	blockage	-0.65%	4.53	leakage	1.77%	-0.31
Reserva Extrativista Arapixi	blockage	-4.73%	1.05	blockage	-0.04%	7.22
Reserva Extrativista Rio Unini*	leakage	0.02%	1.01	blockage	-0.55%	1.10
Reserva Extrativista Gurupá-Melgaço	blockage	-2.74%	1.04	blockage	-4.59%	1.20

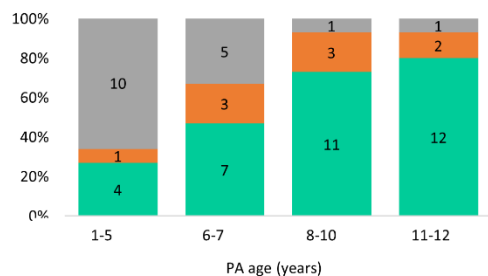
Reserva Extrativista Rio Iriri	blockage	-3.04%	0.94	blockage	-4.78%	1.11
Reserva Extrativista Terra Grande-Pracuúba*	leakage	0.03%	3.58	blockage	-4.66%	1.16
Reserva Particular do Patrimônio Natural do Cristalino III	blockage	-0.12%	4.49	blockage	-1.17%	1.73
Floresta Nacional do Iquiri	leakage	4.66%	0.01	leakage	4.19%	-0.10
Parque Nacional Mapinguari	leakage	3.86%	0.02	leakage	3.44%	-0.10
Parque Nacional Nascentes do Lago Jari*	leakage	0.14%	-0.46	blockage	-0.29%	1.91
Reserva de Desenvolvimento Sustentável do Rio Negro	leakage	0.18%	-1.14	leakage	0.83%	-2.93
Reserva Extrativista do Médio Purus	blockage	-0.45%	1.38	blockage	-0.22%	1.98
Reserva Extrativista Ituxi	NA	NA	NA	blockage	-0.42%	1.11
Reserva Extrativista Rio Xingu	blockage	-3.35%	1.01	blockage	-3.31%	1.28
Área De Relevante Interesse Ecológico Japiim Pentecoste	blockage	-1.38%	2.01	blockage	-1.13%	2.40
Floresta Estadual Canutama*	leakage	0.16%	0.19	blockage	-0.30%	1.50
Floresta Estadual Tapauá	blockage	-0.57%	1.58	blockage	-0.11%	4.46
Parque Estadual do Matupiri*	leakage	0.027%	-0.33	blockage	-0.42%	1.16
Reserva de Desenvolvimento Sustentável do Matupiri	blockage	-0.42%	1.00	blockage	-0.46%	1.07
Reserva De Desenvolvimento Sustentável Igapó-Açu	blockage	-0.50%	1.00	blockage	-0.46%	1.07
Reserva Extrativista Canutama*	leakage	0.24%	-0.03	blockage	-0.20%	2.39
Reserva Extrativista Renascer	blockage	-0.10%	5.78	blockage	-1.34%	1.65
Reserva Particular Do Patrimônio Natural Osório Reimão*	no spillover	0.00%	NA	blockage	-2.86%	1.19
Reserva Particular do Patrimônio Natural Fazenda Calixto*	no spillover	0.00%	NA	blockage	-0.13%	1.18
Reserva Particular do Patrimônio Natural Ilhas Alexandre* Rodrigues Ferreira	no spillover	0.00%	NA	blockage	-0.46%	1.07
Parque Estadual do Charapucu	NA	NA	NA	blockage	-2.32%	1.24
Refúgio de Vida Silvestre Metrópole da Amazônia*	leakage	0.16%	-0.99	blockage	-1.80%	1.48
Reserva Particular Do Patrimônio Natural Peugeot-Onf-Brasil	blockage	-4.46%	1.00	blockage	-1.12%	1.10
Reserva Particular Do Patrimônio Natural Catedral Do Jalapão*	no spillover	0.00%	NA	blockage	-0.11%	1.20
Reserva Particular Do Patrimônio Natural Sonhada	NA	NA	NA	blockage	-0.11%	1.20
Área De Proteção Ambiental Da Serra Do Saubal*	no spillover	0.00%	NA	blockage	-1.89%	1.22
Parque Natural Municipal Veredas de Carajás*	no spillover	0.00%	NA	blockage	-1.89%	1.22
Reserva Particular Do Patrimônio Natural Fazenda Loanda*	no spillover	0.00%	NA	blockage	-0.98%	1.15

Reserva Particular Do Patrimônio Natural Cristalino I*	no spillover	0.00%	NA	blockage	-0.98%	1.15
Reserva Particular Do Patrimônio Natural Bico do Javali	blockage	-1.71%	1.00	blockage	-0.09%	1.15
Reserva Particular Do Patrimônio Natural Nova Aurora*	no spillover	0.00%	NA	blockage	-3.41%	1.14
Reserva Particular Do Patrimônio Natural Irmãos Satelis*	no spillover	0.00%	NA	blockage	-3.00%	1.14
Reserva Particular Do Patrimônio Natural Gibeão*	no spillover	0.00%	NA	blockage	-3.00%	1.14
Reserva Particular Do Patrimônio Natural Aurora Natural*	no spillover	0.00%	NA	blockage	-0.08%	1.19
Parque Natural Municipal Parque Florestal De Sinop*	no spillover	0.00%	NA	blockage	-0.58%	1.26
Reserva de Desenvolvimento Sustentável Puranga Conquista	blockage	-0.17%	1.05	blockage	-0.19%	1.29
Área de Proteção Ambiental dos Campos de Manicoré	blockage	-0.02%	1.53	blockage	-0.05%	2.40
Floresta Nacional de Urupadi*	no spillover	0.00%	NA	blockage	-0.08%	1.70
Floresta Nacional do Aripuanã	leakage	0.84%	0.03	leakage	0.76%	-0.16
Parque Nacional do Acari*	blockage	-0.10%	4.41	leakage	0.28%	-0.55
Refúgio De Vida Silvestre Lago Mole*	no spillover	0.00%	NA	blockage	-0.39%	1.94
Refúgio de Vida Silvestre Tabuleiro do Embaubal	NA	NA	NA	blockage	-0.39%	1.94
Refúgio De Vida Silvestre Padre Sérgio Tonetto	NA	NA	NA	blockage	-0.39%	1.94
Reserva Biológica do Manicoré	blockage	-0.00%	1.00	blockage	-0.08%	1.93
Reserva de Desenvolvimento Sustentável Campo das Mangabas*	no spillover	0.00%	NA	blockage	-0.39%	1.94
Reserva de Desenvolvimento Sustentável Vitória de Souza	leakage	0.49%	-0.25	leakage	0.23%	-3.37

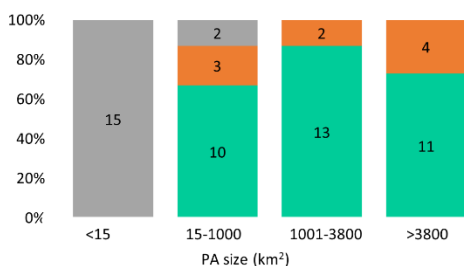
■ Blockage ■ Leakage ■ No net effect

Results of assessment with locally derived baselines

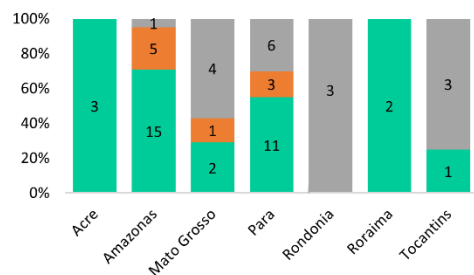
a) PA age quartiles



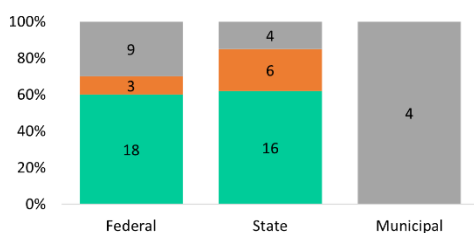
b) PA size quartiles



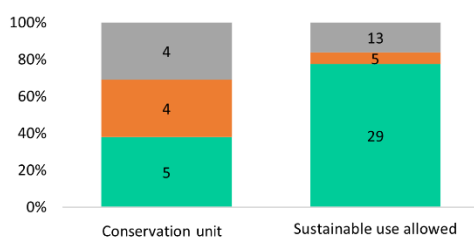
c) by state within which PA is located



d) by administrative tier of government



e) by type in the Brazilian PA classification system



Results of assessment with state-derived baselines

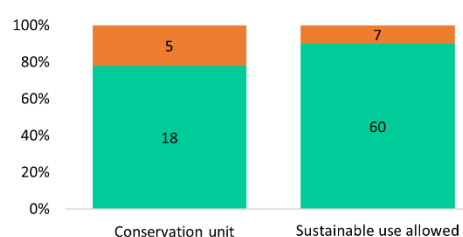
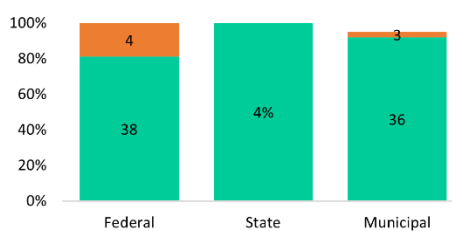
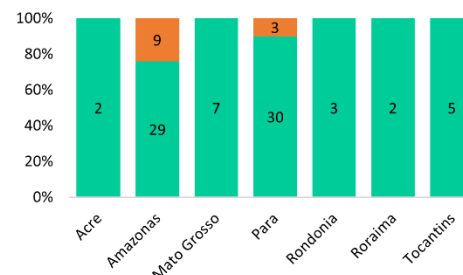
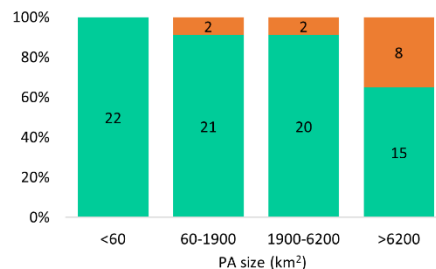
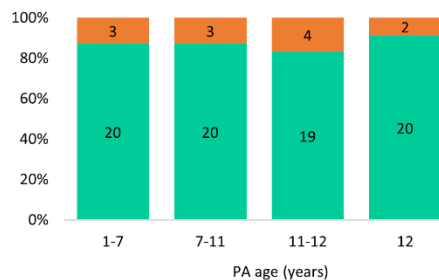
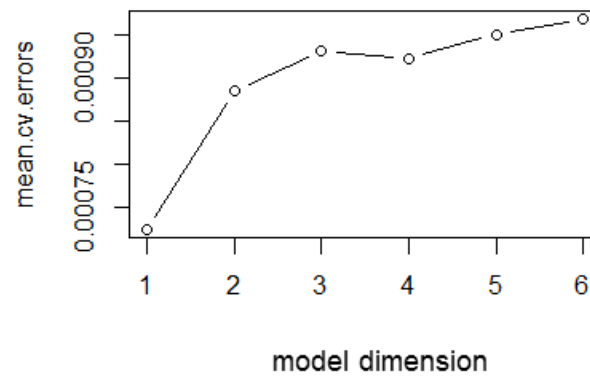


Figure S1. Proportional distribution and number of blockages/leakages per category of net spillover outcomes generated from locally derived baselines (left-hand column) and state-derived baselines (right-hand column) across a) PA age quartiles, b) PA size quartiles, c) states, d) administrative tiers of government, e) PA type in the Brazilian PA classification system, where ‘sustainable use allowed’ means that some human land use and resource extraction is permitted. The left-hand column of graphs does not include outcomes of the 18 PAs to which assessment method 1 could not be applied, nor does the information presented here include spillover outcomes for those PAs where more deforestation was observed inside the target area than expected given the baseline.

a)

Response variable: spillover calculated with locally-derived baselines
Best model dimension of 1 (given 43 observations)



b)

Response variable: spillover calculated with state-derived baselines
Best model dimension of 6 (given 90 observations)

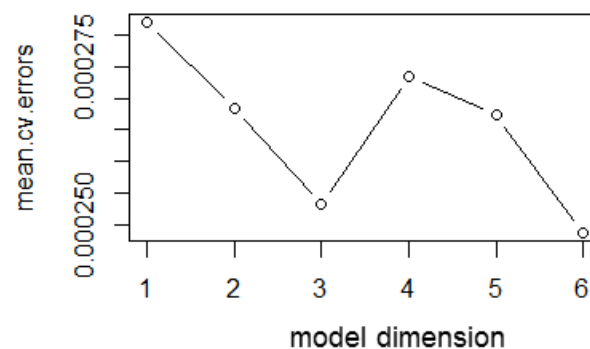


Figure S2. Relationship between mean standard error and model dimension as predicted by k-fold cross-validation. The best model in set, i.e., the one with the lowest mean standard error, included **(a)** only one coefficient (PA age) for the spillover outcomes calculated with locally derived baselines, and **(b)** three coefficients (PA age, PA type in the Brazilian classification system, and state within which the PA is located) for the spillover outcomes generated with state-derived baselines.

Appendix C: Chapter 4 Supplementary materials

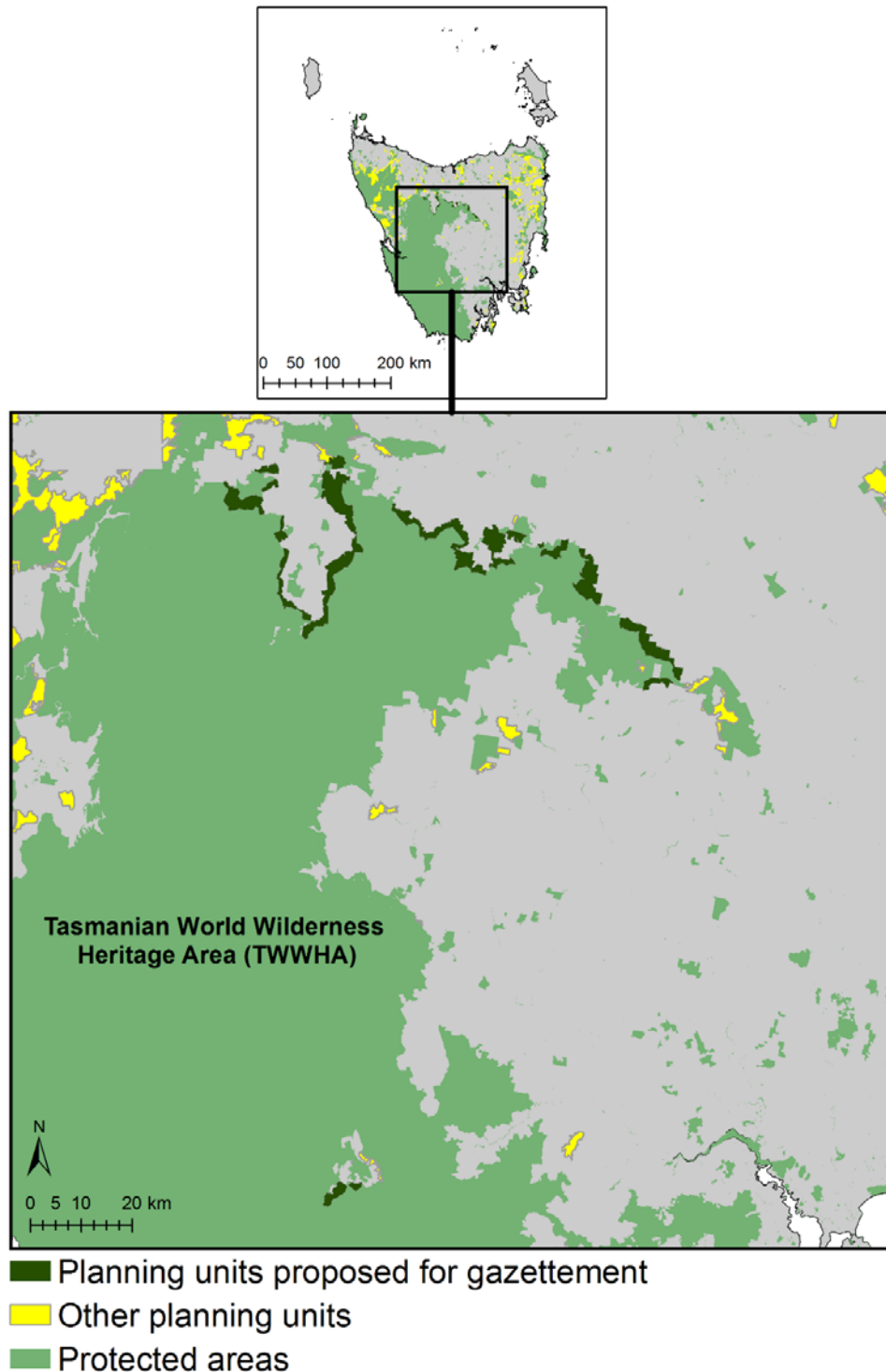
Appendix S1. Planning units to be protected under business as usual. Spatial data from “LIST Future Potential Production Forest” from <http://www.thelist.tas.gov.au/> ©State of Tasmania and the gazettment proposal website <https://dpipwe.tas.gov.au/about-the-department/reservation-of-future-potential-production-forest-land>).

Planning unit	Planning unit area (ha)	Area to be protected (ha)	Length (km) of PA-contiguous boundary	Area suitable for timber plantations (ha)	Cumulative planning unit area (ha)	Cumulative length (km) PA-adjacent boundaries	Cumulative area (ha) suitable for timber plantations
24	249.91	249.91	2.881	9.774	249.91	2.881	9.77
62	551.27	551.27	15.639	2.063	801.17	18.520	11.84
79	15,804.70	5,924.95	120.795	418.514	6,726.12	139.315	430.35
118	235.63	235.63	6.198	0.602	6,961.76	145.513	430.95
183	3,602.10	3,602.10	57.253	132.334	10,563.86	202.765	563.29
210	869.80	869.80	26.758	12.503	11,433.66	229.524	575.79
213	2,933.06	2,933.06	25.552	77.294	14,366.72	255.076	653.08
214	2,420.51	2,420.51	25.041	134.550	16,787.23	280.117	787.63
215	418.38	418.38	11.211	43.301	17,205.61	291.328	830.94
217	279.64	279.64	3.213	18.312	17,485.25	294.541	849.25
218	776.46	776.46	19.362	7.232	18,261.71	313.903	856.48
222	4,406.31	4,406.31	70.964	8.300	22,668.02	384.867	864.78
232	2,077.43	2,077.43	30.724	0.164	24,745.45	415.591	864.94
234	463.50	463.50	9.085	12.155	25,208.95	424.676	877.10

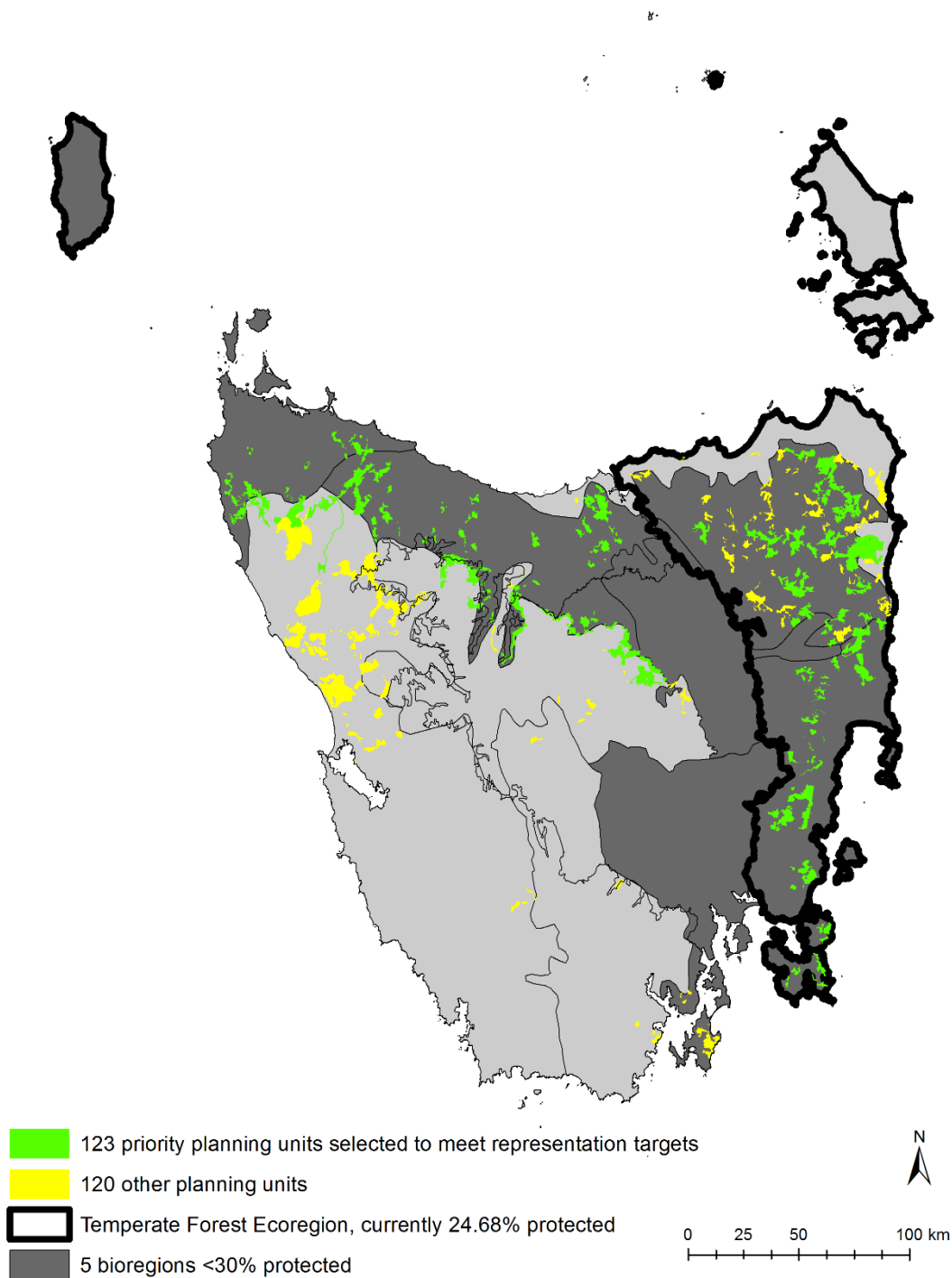
Appendix S2. Areas and proportions of ecoregions and bioregions of Tasmania that are protected and protectable. Within the state of Tasmania, there are three terrestrial ecoregions, as defined by the World Wildlife Fund and the Nature Conservancy in 2011, and nine IBRA units/bioregions, as defined by the Australian government in 2012. This study excluded the small islands in the Bass Strait that are part of the Wilson’s Promontory bioregion, the bulk of which is located on mainland Australia. Red text flags where percent protected is less than 30%. Only the temperate forest ecoregion (which spans the east coast, King Island, and the Furneaux group islands) is less than 30% protected. The temperate rain forest ecoregion is very well represented within the protected area network at more than 67% protected; the Central Highland Forest ecoregion has also met the 30% target. Five of the nine bioregions are less than 30% protected; Ben Lomond could reach the 30% target by only re-gazetting some of the FPPF lands within that bioregion. The other four under-protected bioregions cannot reach the 30% goal by re-gazettement of the FPPF lands alone. Note that some of the 243 FPPF units fall within multiple bioregions. These calculations were all based on spatial data from Olson et al. (2013), the Department of Sustainability, Environment, Water, Population and Communities (2013), and “LIST Future Potential Production Forest” from <http://www.thelist.tas.gov.au/> ©State of Tasmania.

		Area (ha)	Protected areas (ha)	% PAs	FPPF area (ha)	% FPPFs	FPPF+PAs (ha)	FPPF+PAs (%)
Global Ecoregions	TAS temperate forest ecoregion	1,815,040	447,891	24.68%	167,924	9.25%	615,815	33.93%
	TAS Central Highland Forest ecoregion	1,870,940	635,098	33.95%	50,644	2.71%	685,742	36.65%
	TAS temperate rain forest ecoregion	3,120,490	1,937,210	62.08%	173,296	5.55%	2,110,506	67.63%
Australian Bioregions	Northern Midlands	415,121	31,322	7.55%	1,691	0.41%	33,012	7.95%
	South East	1,131,000	215,239	19.03%	49,878	4.41%	265,117	23.44%
	Ben Lomond	657,040	128,688	19.59%	110,021	16.74%	238,709	36.33%
	Northern Slopes	622,700	123,924	19.90%	55,797	8.96%	179,721	28.86%
	King	425,631	96,038	22.56%	12,735	2.99%	108,773	25.56%
	Furneaux	496,234	153,859	31.01%	13,450	2.71%	167,309	33.72%
	Southern Ranges	756,643	452,237	59.77%	3,587	0.47%	455,824	60.24%
	Central Highlands	767,330	477,329	62.21%	39,220	5.11%	516,549	67.32%

West	1,564,330	1,359,630	86.91%	105,528	6.75%	1,465,158	93.66%
Tasmania	6,836,030	3,038,266	44.44%	391,944	5.73%	3,430,210	50.18%



Appendix S3. Existing gazettelement proposal for the FPPF lands. In its Strategic Management Statement attached to the 2016 Tasmanian Wilderness World Heritage Area (TWWHA) Management Plan, the Tasmanian government proposed that 25,209 hectares (6.4%) of the total 392,969 ha of Future Potential Production Forest be gazetted as reserves; each of these selected land parcels is adjacent to the TWWHA. At the time of this research, the proposal had undergone public consultation, and the Tasmanian government was in the process of determining appropriate reservation classes for each parcel. Source of information: <https://dpiwwe.tas.gov.au/about-the-department/reservation-of-future-potential-production-forest-land>



Appendix S4. Planning units (234,705 ha) selected as priorities by the representation strategy, whereby the fewest planning units are protected to achieve 30% ecoregional and bioregional protection in Tasmania for all biogeographic units for which this is possible by gazetting the planning units. Where there were insufficient planning units to achieve this goal in a biogeographic unit, the proportion of that unit protected was maximised. Spatial data from Olson et al. (2013), the Department of Sustainability, Environment, Water, Population and Communities (2013), and “LIST Future Potential Production Forest” from <http://www.thelist.tas.gov.au/> ©State of Tasmania.

Appendix S5. Planning units (234,705 ha) selected as priorities for gazettelement by the representation strategy, whereby the fewest planning units are protected to achieve 30% protection in Tasmania for all biogeographic units for which this is possible by gazetting the planning units. Where there are insufficient planning units to achieve this goal in a biogeographic unit, the proportion of that unit protected was maximised. The selected planning units are listed from smallest to largest in area. Spatial data from Olson et al. (2013), the Department of Sustainability, Environment, Water, Population and Communities (2013), and “LIST Future Potential Production Forest” from <http://www.thelist.tas.gov.au/> ©State of Tasmania.

Planning unit selected for representation	Planning unit area (ha)	Length (km) of PA- contiguous boundary	Area suitable for timber plantations (ha)	Cumulative planning unit area (ha)	Cumulative length (km) PA-adjacent boundaries	Cumulative area (ha) suitable for timber plantations
129	3.12	1.377	1.41	3.12	1.377	1.41
190	8.84	2.149	0.25	11.95	3.526	1.66
9	10.13	2.043	0.60	22.08	5.570	2.26
50	12.34	1.227	0.85	34.41	6.797	3.11
135	14.95	1.979	5.49	49.36	8.776	8.60
48	15.67	3.105	1.52	65.03	11.881	10.12
54	16.50	1.236	4.13	81.53	13.116	14.25
90	17.25	0.784	2.71	98.78	13.900	16.97
86	18.33	1.749	4.67	117.11	15.650	21.63

39	31.46	1.241	7.83	148.57	16.891	29.46
96	32.89	0.881	7.19	181.46	17.772	36.65
72	37.20	3.441	4.65	218.66	21.213	41.30
55	38.91	3.090	18.34	257.57	24.304	59.64
92	46.87	2.180	10.18	304.44	26.483	69.83
128	49.01	2.657	16.83	353.46	29.141	86.66
81	50.88	1.458	28.22	404.34	30.598	114.87
119	63.94	0.256	18.28	468.28	30.854	133.16
151	65.81	2.208	8.10	534.09	33.062	141.26
91	65.98	3.376	0.25	600.06	36.438	141.51
65	67.80	5.690	42.91	667.86	42.128	184.42
68	72.07	0.937	0.18	739.93	43.065	184.60
212	79.02	4.126	43.08	818.95	47.192	227.68
145	82.25	5.472	57.11	901.20	52.663	284.79
156	91.42	2.352	21.27	992.62	55.015	306.05
70	92.05	11.282	3.96	1,084.68	66.297	310.02
204	92.99	1.435	41.93	1,177.67	67.732	351.95
206	95.74	1.808	3.12	1,273.41	69.540	355.07

51	103.98	3.942	23.31	1,377.39	73.483	378.38
66	104.73	3.943	25.57	1,482.12	77.426	403.95
131	135.07	4.616	59.37	1,617.19	82.042	463.33
93	143.48	2.319	6.11	1,760.67	84.360	469.44
108	167.98	4.464	76.90	1,928.65	88.824	546.34
85	177.52	3.071	5.92	2,106.17	91.896	552.26
181	183.75	7.311	36.24	2,289.92	99.207	588.50
69	187.80	3.068	29.32	2,477.72	102.275	617.82
127	190.22	11.407	31.75	2,667.95	113.681	649.57
80	204.50	14.631	15.68	2,872.45	128.313	665.25
205	214.51	2.006	1.65	3,086.96	130.319	666.91
176	226.01	7.240	29.87	3,312.96	137.559	696.78
77	226.57	7.328	8.43	3,539.53	144.887	705.20
123	228.58	3.951	38.08	3,768.11	148.838	743.28
78	228.93	3.128	6.51	3,997.04	151.966	749.79
217	279.64	3.213	18.31	4,276.68	155.179	768.10
114	298.61	9.372	38.83	4,575.29	164.550	806.93
10	306.27	9.615	52.43	4,881.56	174.165	859.36

162	306.85	0.773	30.91	5,188.41	174.938	890.27	
89	332.99	1.797	63.18	5,521.40	176.735	953.44	
104	346.54	7.574	118.81	5,867.94	184.310	1,072.25	
126	349.18	8.883	56.25	6,217.12	193.193	1,128.51	
103	355.79	8.777	53.23	6,572.91	201.970	1,181.74	
124	369.30	6.568	6.91	6,942.21	208.538	1,188.65	
82	380.38	7.583	52.20	7,322.59	216.121	1,240.85	
8	412.04	5.873	55.42	7,734.62	221.993	1,296.27	
87	415.51	9.729	0.51	8,150.14	231.722	1,296.78	
215	418.38	11.211	43.30	8,568.51	242.933	1,340.08	
242	439.52	2.761	104.19	9,008.04	245.694	1,444.27	
234	463.50	9.085	12.15	9,471.54	254.780	1,456.42	
209	485.20	4.795	47.61	9,956.74	259.574	1,504.04	
224	497.06	3.982	79.05	10,453.80	263.556	1,583.08	Exceeds cheapest selection
35	498.88	10.780	6.50	10,952.68	274.336	1,589.58	
64	510.08	8.126	11.84	11,462.75	282.462	1,601.42	
137	520.85	13.226	17.14	11,983.61	295.688	1,618.56	
139	535.88	10.268	47.44	12,519.49	305.956	1,666.00	

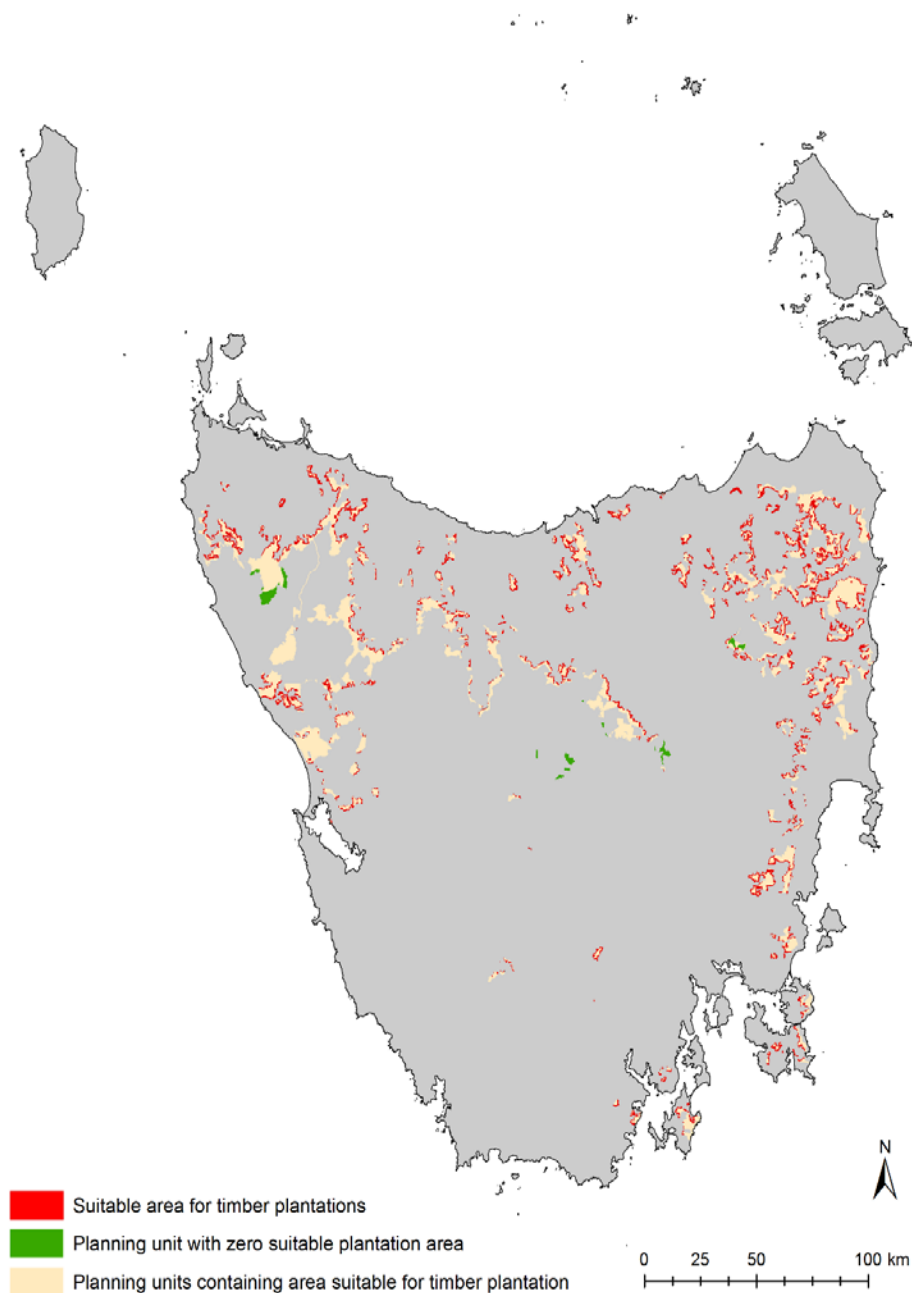
150	547.06	9.154	38.02	13,066.55	315.110	1,704.01
146	579.05	8.225	3.08	13,645.60	323.335	1,707.09
157	613.68	13.763	84.69	14,259.27	337.098	1,791.77
47	624.55	8.034	22.95	14,883.83	345.132	1,814.72
196	696.92	10.961	95.05	15,580.75	356.093	1,909.77
160	707.89	14.142	13.98	16,288.64	370.235	1,923.75
202	733.29	6.881	32.66	17,021.93	377.116	1,956.41
164	765.01	3.716	153.82	17,786.95	380.832	2,110.23
218	776.46	19.362	7.23	18,563.40	400.194	2,117.46
180	782.93	19.664	76.97	19,346.33	419.858	2,194.44
117	840.55	11.152	115.69	20,186.88	431.010	2,310.12
230	970.38	12.157	173.55	21,157.25	443.168	2,483.68
75	990.74	13.305	54.47	22,148.00	456.473	2,538.14
223	1,024.08	7.684	246.07	23,172.08	464.156	2,784.22
109	1,079.69	8.480	77.90	24,251.77	472.637	2,862.11
67	1,092.41	2.732	111.48	25,344.18	475.368	2,973.60
49	1,104.14	12.419	56.24	26,448.32	487.787	3,029.83
221	1,167.18	14.076	206.84	27,615.50	501.863	3,236.67

Exceeds BAU

143	1,175.75	11.431	152.22	28,791.25	513.295	3,388.89
134	1,196.61	37.302	100.31	29,987.86	550.596	3,489.20
219	1,258.43	14.185	75.86	31,246.29	564.781	3,565.06
38	1,303.35	14.782	79.61	32,549.64	579.563	3,644.68
74	1,358.62	28.628	34.53	33,908.26	608.191	3,679.21
168	1,414.49	11.265	177.21	35,322.75	619.456	3,856.41
229	1,568.88	20.496	90.81	36,891.63	639.951	3,947.22
142	1,640.78	11.777	75.11	38,532.41	651.728	4,022.33
232	2,077.43	30.724	0.16	40,609.84	682.452	4,022.49
3	2,148.74	31.242	105.63	42,758.58	713.694	4,128.12
144	2,167.07	24.045	110.44	44,925.65	737.739	4,238.56
155	2,414.67	22.231	353.44	47,340.32	759.970	4,592.01
214	2,420.51	25.041	134.55	49,760.83	785.011	4,726.56
133	2,712.15	33.459	61.44	52,472.98	818.470	4,787.99
158	2,712.40	15.190	281.87	55,185.38	833.661	5,069.86
225	2,808.06	24.636	321.65	57,993.44	858.297	5,391.50
213	2,933.06	25.552	77.29	60,926.50	883.849	5,468.80
61	2,987.90	23.101	80.36	63,914.40	906.950	5,549.16

132	3,093.58	26.124	263.78	67,007.98	933.074	5,812.94
241	3,155.38	42.439	425.06	70,163.36	975.513	6,238.00
122	3,388.68	38.439	181.52	73,552.04	1,013.952	6,419.52
169	3,567.51	30.115	197.13	77,119.55	1,044.067	6,616.65
183	3,602.10	57.253	132.33	80,721.65	1,101.319	6,748.99
113	3,769.88	30.338	314.70	84,491.53	1,131.657	7,063.68
46	4,359.85	41.442	303.51	88,851.38	1,173.099	7,367.20
178	4,404.05	27.193	419.52	93,255.43	1,200.292	7,786.71
222	4,406.31	70.964	8.30	97,661.74	1,271.256	7,795.01
116	4,554.09	23.537	133.59	102,215.83	1,294.793	7,928.61
130	4,843.24	52.540	69.78	107,059.07	1,347.333	7,998.39
203	5,688.32	46.138	272.98	112,747.39	1,393.471	8,271.37
102	5,926.88	9.939	195.10	118,674.27	1,403.410	8,466.47
193	6,258.26	67.024	211.22	124,932.53	1,470.433	8,677.69
227	6,650.27	68.939	239.44	131,582.80	1,539.372	8,917.13
41	7,338.91	36.133	603.72	138,921.71	1,575.506	9,520.86
53	7,378.28	50.780	264.63	146,299.99	1,626.285	9,785.49
110	7,407.46	67.037	290.20	153,707.45	1,693.322	10,075.69

76	9,797.42	91.912	200.19		163,504.87	1,785.234	10,275.88
42	11,789.00	105.854	876.18		175,293.87	1,891.088	11,152.05
148	12,592.40	255.050	1,048.56		187,886.27	2,146.138	12,200.61
211	14,389.80	43.570	341.85		202,276.07	2,189.708	12,542.47
79	15,804.70	120.795	418.51		218,080.77	2,310.503	12,960.98
63	16,624.20	233.618	670.51		234,704.96	2,544.121	13,631.49



Appendix S6. Suitable areas for timber plantations of *Eucalyptus nitens*, *Eucalyptus globulus*, and/or *Pinus radiata* (red) within the planning units (beige). Only 14 (10,276.55 ha, or 2.6%) of the 243 planning units contain no suitable area for any of the three plantation species (green); these polygons in green are considered together to represent the gazettement selection with the minimum opportunity cost. Spatial data from “LIST Future Potential Production Forest” and from “LIST Enterprise Suitability Maps Using Climate Futures Projections” from <http://www.thelist.tas.gov.au/>, specifically *Eucalyptus nitens* suitability, 2018; *Eucalyptus globulus* suitability, 2018; and *Pinus radiata* suitability, 2018. ©State of Tasmania.

Appendix S7. Suitable areas for timber plantations of *Eucalyptus nitens*, *Eucalyptus globulus*, and/or *Pinus radiata* within the planning units. Spatial data from “LIST Future Potential Production Forest” and from “LIST Enterprise Suitability Maps Using Climate Futures Projections” from <http://www.thelist.tas.gov.au/>, specifically *Eucalyptus nitens* suitability, 2018; *Eucalyptus globulus* suitability, 2018; and *Pinus radiata* suitability, 2018. ©State of Tasmania.

Timber plantation species	Suitable area (ha)	% Total FPPF Land
<i>Eucalyptus nitens</i>	9,060.95	2.31%
<i>Eucalyptus globulus</i>	5,447.30	1.39%
<i>Eucalyptus nitens</i> and <i>Eucalyptus globulus</i>	4,593.66	1.17%
<i>Pinus radiata</i>	872.62	0.22%
<i>Eucalyptus nitens</i> and <i>pinus radiata</i>	282.59	0.07%
<i>E. nitens</i> and <i>P. radiata</i> and <i>E. globulus</i>	89.97	0.02%
<i>Pinus radiata</i> and <i>Eucalyptus globulus</i>	53.36	0.01%
Total	20,400.44	5.20%

Appendix S8. Area suitable for timber plantations (combination of *Eucalyptus nitens*, *Eucalyptus globulus*, and *Pinus radiata*) within each planning unit, listed from smallest to largest suitable area. Spatial data from “LIST Future Potential Production Forest” and from “LIST Enterprise Suitability Maps Using Climate Futures Projections” from <http://www.thelist.tas.gov.au/>, specifically *Eucalyptus nitens* suitability, 2018; *Eucalyptus globulus* suitability, 2018; and *Pinus radiata* suitability, 2018. ©State of Tasmania.

Planning unit	Planning unit area (ha)	Area suitable (ha)	% Planning unit suitable	Cumulative planning unit area (ha)	Cumulative suitable area (ha)	Percent total planning unit area suitable for plantation
2	348.80	0.00	0.00%	348.80	0.00	0.00%
4	1,110.16	0.00	0.00%	1,458.97	0.00	0.00%
5	271.26	0.00	0.00%	1,730.22	0.00	0.00%
11	1,487.00	0.00	0.00%	3,217.22	0.00	0.00%
12	167.68	0.00	0.00%	3,384.90	0.00	0.00%
13	22.37	0.00	0.00%	3,407.27	0.00	0.00%
18	49.69	0.00	0.00%	3,456.96	0.00	0.00%
71	1,206.05	0.00	0.00%	4,663.01	0.00	0.00%
98	1,372.94	0.00	0.00%	6,035.95	0.00	0.00%
100	3,384.38	0.00	0.00%	9,420.33	0.00	0.00%
101	502.13	0.00	0.00%	9,922.46	0.00	0.00%

141	4.83	0.00	0.00%	9,927.29	0.00	0.00%
153	270.27	0.00	0.00%	10,197.55	0.00	0.00%
179	79.00	0.00	0.00%	10,276.55	0.00	0.00%
1	4.30	0.01	0.31%	10,280.85	0.01	0.00%
232	2,077.43	0.16	0.01%	12,358.28	0.18	0.00%
68	72.07	0.18	0.25%	12,430.34	0.36	0.00%
190	8.84	0.25	2.80%	12,439.18	0.61	0.00%
91	65.98	0.25	0.38%	12,505.16	0.86	0.01%
44	135.70	0.28	0.21%	12,640.85	1.14	0.01%
173	577.99	0.28	0.05%	13,218.84	1.43	0.01%
94	124.06	0.42	0.34%	13,342.90	1.85	0.01%
87	415.51	0.51	0.12%	13,758.41	2.36	0.02%
118	235.63	0.60	0.26%	13,994.04	2.96	0.02%
9	10.13	0.60	5.95%	14,004.17	3.56	0.03%
50	12.34	0.85	6.85%	14,016.50	4.41	0.03%
129	3.12	1.41	45.24%	14,019.62	5.82	0.04%
48	15.67	1.52	9.69%	14,035.29	7.34	0.05%
166	10,113.40	1.59	0.02%	24,148.69	8.93	0.04%

Zero opportunity cost

205	214.51	1.65	0.77%	24,363.20	10.59	0.04%
30	1,375.58	1.80	0.13%	25,738.78	12.39	0.05%
62	551.27	2.06	0.37%	26,290.04	14.45	0.05%
15	212.92	2.55	1.20%	26,502.96	17.00	0.06%
90	17.25	2.71	15.73%	26,520.21	19.71	0.07%
27	136.63	2.87	2.10%	26,656.84	22.59	0.08%
146	579.05	3.08	0.53%	27,235.89	25.66	0.09%
206	95.74	3.12	3.26%	27,331.63	28.78	0.11%
184	343.62	3.30	0.96%	27,675.25	32.09	0.12%
26	12.81	3.42	26.67%	27,688.06	35.50	0.13%
73	164.23	3.69	2.24%	27,852.29	39.19	0.14%
70	92.05	3.96	4.30%	27,944.35	43.15	0.15%
54	16.50	4.13	25.04%	27,960.85	47.28	0.17%
57	36.70	4.45	12.13%	27,997.55	51.73	0.18%
72	37.20	4.65	12.51%	28,034.75	56.39	0.20%
86	18.33	4.67	25.46%	28,053.07	61.05	0.22%
97	141.31	4.91	3.48%	28,194.38	65.96	0.23%
19	171.53	4.91	2.86%	28,365.91	70.88	0.25%

Exceeds BAU

135	14.95	5.49	36.76%	28,380.86	76.37	0.27%
85	177.52	5.92	3.33%	28,558.38	82.29	0.29%
93	143.48	6.11	4.26%	28,701.86	88.40	0.31%
35	498.88	6.50	1.30%	29,200.74	94.90	0.32%
78	228.93	6.51	2.84%	29,429.67	101.40	0.34%
36	10,230.10	6.61	0.06%	39,659.76	108.01	0.27%
111	229.76	6.73	2.93%	39,889.52	114.73	0.29%
124	369.30	6.91	1.87%	40,258.82	121.65	0.30%
96	32.89	7.19	21.87%	40,291.71	128.84	0.32%
218	776.46	7.23	0.93%	41,068.17	136.07	0.33%
197	61.62	7.78	12.63%	41,129.79	143.85	0.35%
39	31.46	7.83	24.87%	41,161.25	151.68	0.37%
151	65.81	8.10	12.31%	41,227.06	159.78	0.39%
60	1,064.21	8.28	0.78%	42,291.27	168.06	0.40%
222	4,406.31	8.30	0.19%	46,697.58	176.36	0.38%
77	226.57	8.43	3.72%	46,924.15	184.79	0.39%
24	249.91	9.77	3.91%	47,174.06	194.56	0.41%
192	96.37	9.94	10.31%	47,270.43	204.50	0.43%

92	46.87	10.18	21.73%	47,317.30	214.68	0.45%
88	3,496.87	10.64	0.30%	50,814.17	225.32	0.44%
170	29.08	11.76	40.43%	50,843.25	237.08	0.47%
240	539.87	11.78	2.18%	51,383.12	248.86	0.48%
64	510.08	11.84	2.32%	51,893.20	260.69	0.50%
234	463.50	12.15	2.62%	52,356.70	272.85	0.52%
210	869.80	12.50	1.44%	53,226.50	285.35	0.54%
28	747.89	13.33	1.78%	53,974.38	298.68	0.55%
187	807.64	13.92	1.72%	54,782.02	312.60	0.57%
160	707.89	13.98	1.98%	55,489.92	326.58	0.59%
125	139.03	14.00	10.07%	55,628.95	340.58	0.61%
56	1,715.09	14.04	0.82%	57,344.04	354.62	0.62%
14	953.15	14.10	1.48%	58,297.19	368.72	0.63%
84	190.46	14.48	7.60%	58,487.65	383.20	0.66%
185	276.17	14.86	5.38%	58,763.82	398.06	0.68%
220	575.29	14.93	2.60%	59,339.11	412.99	0.70%
80	204.50	15.68	7.67%	59,543.61	428.68	0.72%
191	104.27	16.23	15.57%	59,647.87	444.91	0.75%

186	548.64	16.66	3.04%	60,196.51	461.57	0.77%
128	49.01	16.83	34.34%	60,245.52	478.40	0.79%
237	65.53	16.83	25.69%	60,311.06	495.24	0.82%
137	520.85	17.14	3.29%	60,831.91	512.38	0.84%
236	92.57	17.64	19.05%	60,924.48	530.01	0.87%
95	634.19	17.90	2.82%	61,558.67	547.92	0.89%
238	1,075.14	18.08	1.68%	62,633.81	566.00	0.90%
119	63.94	18.28	28.59%	62,697.75	584.28	0.93%
217	279.64	18.31	6.55%	62,977.39	602.59	0.96%
55	38.91	18.34	47.13%	63,016.31	620.93	0.99%
167	693.30	18.77	2.71%	63,709.61	639.71	1.00%
121	205.92	18.94	9.20%	63,915.53	658.65	1.03%
156	91.42	21.27	23.26%	64,006.95	679.91	1.06%
159	89.78	22.18	24.70%	64,096.73	702.09	1.10%
99	10,942.30	22.91	0.21%	75,039.03	725.00	0.97%
47	624.55	22.95	3.67%	75,663.58	747.94	0.99%
51	103.98	23.31	22.42%	75,767.56	771.26	1.02%
201	48.33	24.50	50.70%	75,815.89	795.76	1.05%

66	104.73	25.57	24.41%	75,920.62	821.33	1.08%
40	3,618.57	26.45	0.73%	79,539.19	847.78	1.07%
81	50.88	28.22	55.46%	79,590.07	875.99	1.10%
69	187.80	29.32	15.61%	79,777.87	905.31	1.13%
176	226.01	29.87	13.22%	80,003.88	935.18	1.17%
152	300.24	29.98	9.98%	80,304.12	965.16	1.20%
162	306.85	30.91	10.07%	80,610.97	996.07	1.24%
136	1,940.75	31.19	1.61%	82,551.72	1,027.26	1.24%
127	190.22	31.75	16.69%	82,741.94	1,059.01	1.28%
105	187.94	31.87	16.96%	82,929.88	1,090.88	1.32%
202	733.29	32.66	4.45%	83,663.17	1,123.54	1.34%
140	867.34	34.40	3.97%	84,530.51	1,157.93	1.37%
182	430.61	34.53	8.02%	84,961.12	1,192.46	1.40%
74	1,358.62	34.53	2.54%	86,319.74	1,226.99	1.42%
174	176.50	35.50	20.11%	86,496.24	1,262.50	1.46%
181	183.75	36.24	19.72%	86,679.99	1,298.74	1.50%
45	165.26	37.96	22.97%	86,845.25	1,336.70	1.54%
7	189.83	37.99	20.01%	87,035.08	1,374.70	1.58%

150	547.06	38.02	6.95%	87,582.14	1,412.71	1.61%
123	228.58	38.08	16.66%	87,810.72	1,450.79	1.65%
114	298.61	38.83	13.01%	88,109.33	1,489.63	1.69%
58	270.75	39.41	14.56%	88,380.07	1,529.04	1.73%
147	258.32	40.16	15.55%	88,638.39	1,569.20	1.77%
208	135.31	40.29	29.78%	88,773.70	1,609.50	1.81%
6	80.13	40.44	50.47%	88,853.83	1,649.94	1.86%
204	92.99	41.93	45.09%	88,946.83	1,691.87	1.90%
65	67.80	42.91	63.29%	89,014.63	1,734.78	1.95%
212	79.02	43.08	54.51%	89,093.65	1,777.86	2.00%
215	418.38	43.30	10.35%	89,512.03	1,821.16	2.03%
34	878.91	44.31	5.04%	90,390.93	1,865.47	2.06%
112	431.31	44.55	10.33%	90,822.24	1,910.03	2.10%
139	535.88	47.44	8.85%	91,358.12	1,957.47	2.14%
209	485.20	47.61	9.81%	91,843.32	2,005.08	2.18%
29	223.60	49.54	22.16%	92,066.93	2,054.62	2.23%
233	423.91	51.25	12.09%	92,490.84	2,105.87	2.28%
171	1,674.88	51.31	3.06%	94,165.72	2,157.18	2.29%

82	380.38	52.20	13.72%	94,546.10	2,209.38	2.34%
10	306.27	52.43	17.12%	94,852.37	2,261.81	2.38%
103	355.79	53.23	14.96%	95,208.16	2,315.04	2.43%
107	458.77	53.90	11.75%	95,666.93	2,368.94	2.48%
75	990.74	54.47	5.50%	96,657.68	2,423.41	2.51%
8	412.04	55.42	13.45%	97,069.71	2,478.82	2.55%
49	1,104.14	56.24	5.09%	98,173.85	2,535.06	2.58%
126	349.18	56.25	16.11%	98,523.03	2,591.31	2.63%
145	82.25	57.11	69.43%	98,605.29	2,648.42	2.69%
22	409.93	57.66	14.07%	99,015.22	2,706.08	2.73%
20	1,051.11	58.58	5.57%	100,066.33	2,764.66	2.76%
131	135.07	59.37	43.96%	100,201.40	2,824.03	2.82%
207	238.17	59.45	24.96%	100,439.57	2,883.49	2.87%
133	2,712.15	61.44	2.27%	103,151.72	2,944.92	2.85%
172	331.54	61.70	18.61%	103,483.25	3,006.63	2.91%
120	768.96	62.56	8.14%	104,252.21	3,069.19	2.94%
89	332.99	63.18	18.97%	104,585.20	3,132.37	3.00%
189	758.48	64.18	8.46%	105,343.68	3,196.54	3.03%

154	290.29	67.12	23.12%	105,633.96	3,263.66	3.09%
31	483.19	67.85	14.04%	106,117.15	3,331.51	3.14%
115	1,693.42	68.84	4.06%	107,810.57	3,400.35	3.15%
52	665.28	69.17	10.40%	108,475.85	3,469.52	3.20%
130	4,843.24	69.78	1.44%	113,319.09	3,539.30	3.12%
17	228.66	72.35	31.64%	113,547.75	3,611.65	3.18%
142	1,640.78	75.11	4.58%	115,188.53	3,686.76	3.20%
219	1,258.43	75.86	6.03%	116,446.96	3,762.62	3.23%
108	167.98	76.90	45.78%	116,614.94	3,839.52	3.29%
180	782.93	76.97	9.83%	117,397.87	3,916.50	3.34%
213	2,933.06	77.29	2.64%	120,330.93	3,993.79	3.32%
106	1,002.33	77.33	7.71%	121,333.26	4,071.12	3.36%
109	1,079.69	77.90	7.21%	122,412.95	4,149.01	3.39%
37	293.99	78.67	26.76%	122,706.93	4,227.68	3.45%
224	497.06	79.05	15.90%	123,203.99	4,306.73	3.50%
38	1,303.35	79.61	6.11%	124,507.34	4,386.34	3.52%
61	2,987.90	80.36	2.69%	127,495.24	4,466.70	3.50%
161	1,198.78	81.26	6.78%	128,694.02	4,547.97	3.53%

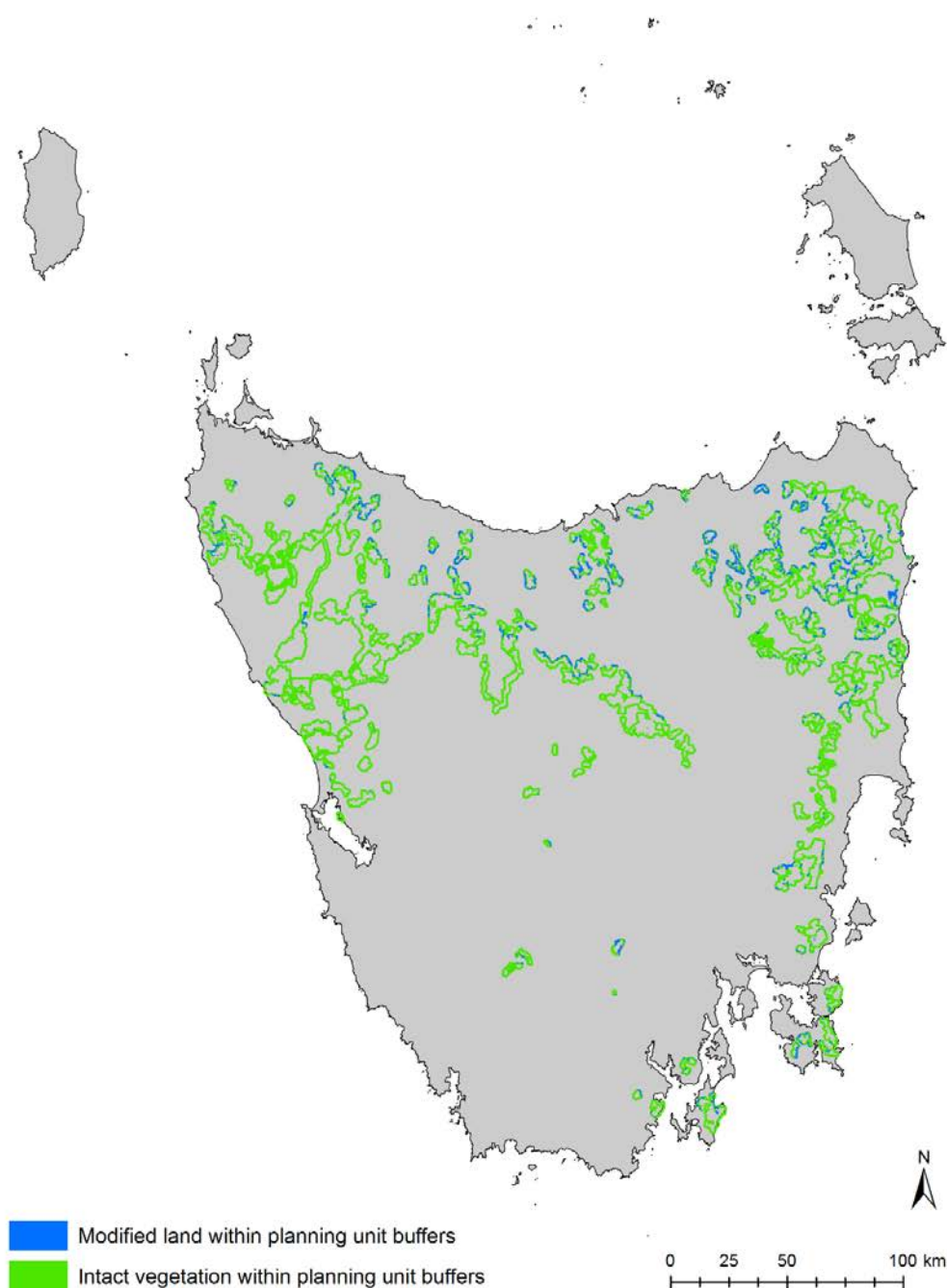
195	2,515.05	81.70	3.25%	131,209.07	4,629.67	3.53%
163	193.31	84.33	43.63%	131,402.38	4,714.00	3.59%
157	613.68	84.69	13.80%	132,016.06	4,798.68	3.63%
33	746.33	85.37	11.44%	132,762.39	4,884.05	3.68%
229	1,568.88	90.81	5.79%	134,331.27	4,974.86	3.70%
199	461.90	91.80	19.87%	134,793.17	5,066.65	3.76%
16	3,942.60	93.14	2.36%	138,735.77	5,159.80	3.72%
196	696.92	95.05	13.64%	139,432.69	5,254.84	3.77%
83	224.75	98.68	43.91%	139,657.45	5,353.53	3.83%
134	1,196.61	100.31	8.38%	140,854.06	5,453.83	3.87%
235	1,402.67	102.77	7.33%	142,256.73	5,556.60	3.91%
242	439.52	104.19	23.71%	142,696.25	5,660.79	3.97%
3	2,148.74	105.63	4.92%	144,844.99	5,766.43	3.98%
144	2,167.07	110.44	5.10%	147,012.06	5,876.86	4.00%
21	1,978.33	110.59	5.59%	148,990.39	5,987.46	4.02%
67	1,092.41	111.48	10.21%	150,082.80	6,098.94	4.06%
200	931.17	112.92	12.13%	151,013.97	6,211.86	4.11%
23	901.65	113.59	12.60%	151,915.62	6,325.45	4.16%

117	840.55	115.69	13.76%		152,756.17	6,441.14	4.22%
216	2,762.52	117.95	4.27%		155,518.69	6,559.09	4.22%
104	346.54	118.81	34.28%		155,865.23	6,677.90	4.28%
194	510.83	121.86	23.86%		156,376.05	6,799.76	4.35%
231	2,269.95	122.81	5.41%		158,646.00	6,922.57	4.36%
25	1,295.19	130.94	10.11%		159,941.19	7,053.51	4.41%
183	3,602.10	132.33	3.67%		163,543.29	7,185.84	4.39%
116	4,554.09	133.59	2.93%		168,097.38	7,319.44	4.35%
214	2,420.51	134.55	5.56%		170,517.89	7,453.99	4.37%
243	838.36	144.07	17.18%		171,356.25	7,598.06	4.43%
138	1,126.33	145.93	12.96%		172,482.58	7,743.99	4.49%
143	1,175.75	152.22	12.95%		173,658.33	7,896.21	4.55%
59	1,004.01	153.57	15.30%		174,662.34	8,049.78	4.61%
164	765.01	153.82	20.11%		175,427.35	8,203.61	4.68%
226	1,350.11	160.17	11.86%		176,777.46	8,363.78	4.73%
165	1,095.37	162.08	14.80%		177,872.83	8,525.85	4.79%
230	970.38	173.55	17.89%		178,843.21	8,699.41	4.86%
168	1,414.49	177.21	12.53%		180,257.70	8,876.62	4.92%

228	2,019.40	178.77	8.85%	182,277.10	9,055.38	4.97%
122	3,388.68	181.52	5.36%	185,665.78	9,236.90	4.98%
239	4,909.19	181.86	3.70%	190,574.97	9,418.76	4.94%
102	5,926.88	195.10	3.29%	196,501.85	9,613.86	4.89%
169	3,567.51	197.13	5.53%	200,069.36	9,811.00	4.90%
76	9,797.42	200.19	2.04%	209,866.78	10,011.19	4.77%
221	1,167.18	206.84	17.72%	211,033.96	10,218.03	4.84%
193	6,258.26	211.22	3.38%	217,292.22	10,429.25	4.80%
177	2,374.05	216.17	9.11%	219,666.27	10,645.41	4.85%
32	2,107.69	219.04	10.39%	221,773.96	10,864.46	4.90%
188	1,674.55	230.35	13.76%	223,448.51	11,094.81	4.97%
227	6,650.27	239.44	3.60%	230,098.78	11,334.25	4.93%
223	1,024.08	246.07	24.03%	231,122.86	11,580.32	5.01%
198	1,682.98	258.38	15.35%	232,805.84	11,838.70	5.09%
132	3,093.58	263.78	8.53%	235,899.42	12,102.48	5.13%
53	7,378.28	264.63	3.59%	243,277.70	12,367.11	5.08%
203	5,688.32	272.98	4.80%	248,966.02	12,640.09	5.08%
158	2,712.40	281.87	10.39%	251,678.42	12,921.95	5.13%

Exceeds representational selection

110	7,407.46	290.20	3.92%		259,085.88	13,212.16	5.10%
175	360.86	295.90	82.00%		259,446.74	13,508.05	5.21%
46	4,359.85	303.51	6.96%		263,806.59	13,811.57	5.24%
113	3,769.88	314.70	8.35%		267,576.47	14,126.26	5.28%
149	6,651.87	317.69	4.78%		274,228.34	14,443.95	5.27%
225	2,808.06	321.65	11.45%		277,036.40	14,765.60	5.33%
211	14,389.80	341.85	2.38%		291,426.20	15,107.45	5.18%
155	2,414.67	353.44	14.64%		293,840.87	15,460.89	5.26%
79	15,804.70	418.51	2.65%		309,645.57	15,879.40	5.13%
178	4,404.05	419.52	9.53%		314,049.62	16,298.92	5.19%
241	3,155.38	425.06	13.47%		317,205.00	16,723.98	5.27%
43	26,394.10	454.67	1.72%		343,599.10	17,178.65	5.00%
41	7,338.91	603.72	8.23%		350,938.01	17,782.38	5.07%
63	16,624.20	670.51	4.03%		367,562.21	18,452.89	5.02%
42	11,789.00	876.18	7.43%		379,351.21	19,329.07	5.10%
148	12,592.40	1,048.56	8.33%		391,943.61	20,377.63	5.20%



Appendix S9. Modified land and intact vegetation within 1-km radius buffers around planning units. Spatial data from “LIST Future Potential Production Forest” and from “TASVEG 4.0 from www.theLIST.tas.gov.au ©State of Tasmania. TasVeg4.0 defines ‘modified land’ as agricultural land, improved pasture with native tree canopy (greater than or equal to 5% cover), extra-urban miscellaneous, the introduced Marram (*Ammophila arenaria*) grassland, permanent easements, plantations for silviculture – hardwood, plantations for silviculture – softwood, bracken (*Pteridium esculentum*) fernland, regenerating cleared land, *Spartina* marshland, unverified plantations for silviculture, urban areas, and weed infestation. The presence of modified land areas in the 1-km buffer was considered a metric of threat faced by each planning unit.

Appendix S10. Area and proportion of modified land within planning units and within the 1-km radius buffer zones immediately adjacent each planning unit, listed from largest to smallest proportion of buffer under modified land classes. Spatial data from “LIST Future Potential Production Forest” and from “TASVEG 4.0 from www.theLIST.tas.gov.au @State of Tasmania. TasVeg4.0 defines ‘modified land’ as agricultural land, improved pasture with native tree canopy (greater than or equal to 5% cover), extra-urban miscellaneous, the introduced Marram (*Ammophila arenaria*) grassland, permanent easements, plantations for silviculture – hardwood, plantations for silviculture – softwood, bracken (*Pteridium esculentum*) fernland, regenerating cleared land, Spartina marshland, unverified silviculture, urban areas, and weed infestation. The presence of modified land areas in the 1-km buffer was considered a metric of threat faced by each planning unit.

Planning unit	Planning unit area (ha)	Modified area		Area of 1km buffer (ha)	Area modified within buffer (ha)	% Buffer modified	Cumulative planning unit area (ha)
		within planning unit (ha)	% Planning unit modified				
175	360.86	26.82	7.43%	1,779.76	1,437.76	80.78%	360.86
59	1,004.01	9.8	0.98%	1,998.29	1,360.23	68.07%	1,364.87
194	510.83	18.89	3.70%	2,314.94	1,517.84	65.57%	1,875.70
157	613.68	13.05	2.13%	1,798.54	1,084.89	60.32%	2,489.38
83	224.75	7.06	3.14%	1,589.74	953.07	59.95%	2,714.13
201	48.33	5.73	11.86%	885.17	505.64	57.12%	2,762.46
92	46.87	1.84	3.92%	687.42	345.6	50.28%	2,809.33
155	2,414.67	7.87	0.33%	4,730.90	2,367.23	50.04%	5,224.00
154	290.29	1.91	0.66%	1,677.84	823.63	49.09%	5,514.29
143	1,175.75	22.34	1.90%	2,515.79	1,167.35	46.40%	6,690.04

168	1,414.49	20.26	1.43%	2,486.91	1,134.87	45.63%	8,104.53
224	497.06	4.81	0.97%	1,547.09	678.18	43.84%	8,601.59
189	758.48	2.76	0.36%	2,183.47	939.39	43.02%	9,360.07 Exceeds cheapest selection
228	2,019.40	18.56	0.92%	3,956.51	1,692.44	42.78%	11,379.47
230	970.38	5.77	0.59%	2,671.81	1,134.31	42.45%	12,349.85
126	349.18	4.76	1.36%	1,804.53	759.13	42.07%	12,699.03
150	547.06	0.44	0.08%	1,510.42	628.6	41.62%	13,246.09
199	461.9	19.67	4.26%	1,757.90	727.23	41.37%	13,707.99
147	258.32	0	0.00%	1,244.69	514.38	41.33%	13,966.31
120	768.96	0.1	0.01%	1,859.02	766.66	41.24%	14,735.27
23	901.65	1.2	0.13%	2,116.53	838.85	39.63%	15,636.92
236	92.57	4.15	4.48%	1,159.77	456.43	39.36%	15,729.49
138	1,126.33	7.3	0.65%	2,210.42	848.51	38.39%	16,855.82
113	3,769.88	22.49	0.60%	5,267.47	2,021.34	38.37%	20,625.70
164	765.01	18.54	2.42%	1,965.66	727.63	37.02%	21,390.71
89	332.99	1.11	0.33%	1,293.90	475.9	36.78%	21,723.70
223	1,024.08	5.12	0.50%	3,021.52	1,107.42	36.65%	22,747.78
78	228.93	0.21	0.09%	1,142.80	416.87	36.48%	22,976.71

67	1,092.41	10.45	0.96%	1,979.48	699.15	35.32%	24,069.12
170	29.08	1.05	3.62%	542.24	189.4	34.93%	24,098.20
8	412.04	3.32	0.81%	1,674.87	579.36	34.59%	24,510.24
124	369.3	6.52	1.77%	1,320.08	454.06	34.40%	24,879.54
196	696.92	11.3	1.62%	2,307.11	790.64	34.27%	25,576.46 Exceeds BAU
75	990.74	1.26	0.13%	2,550.03	855.13	33.53%	26,567.20
123	228.58	0.01	0.01%	1,053.77	347.75	33.00%	26,795.78
182	430.61	0.01	0.00%	1,511.53	493.8	32.67%	27,226.39
151	65.81	0	0.00%	676.62	219.17	32.39%	27,292.20
121	205.92	3.01	1.46%	1,315.37	425.45	32.34%	27,498.12
188	1,674.55	30.41	1.82%	3,667.46	1,171.61	31.95%	29,172.67
185	276.17	0.82	0.30%	1,110.35	346.14	31.17%	29,448.84
190	8.84	0.24	2.77%	608.65	188.79	31.02%	29,457.68
198	1,682.98	55.21	3.28%	3,797.21	1,171.77	30.86%	31,140.66
200	931.17	0.58	0.06%	2,739.69	800.71	29.23%	32,071.83
211	14,389.80	10.25	0.07%	8,841.78	2,579.00	29.17%	46,461.63
112	431.31	2.75	0.64%	1,799.17	520.44	28.93%	46,892.94
45	165.26	2	1.21%	1,130.26	325.5	28.80%	47,058.20

158	2,712.40	10.23	0.38%	5,307.24	1,525.88	28.75%	49,770.60
93	143.48	0.11	0.08%	818.41	234.8	28.69%	49,914.08
237	65.53	0	0.00%	690.86	190.78	27.62%	49,979.61
174	176.5	0.14	0.08%	869.89	237.27	27.28%	50,156.11
177	2,374.05	18.86	0.79%	4,339.81	1,169.42	26.95%	52,530.16
9	10.13	0.05	0.51%	625.56	168.38	26.92%	52,540.29
226	1,350.11	23.05	1.71%	2,876.77	767.55	26.68%	53,890.40
243	838.36	15.57	1.86%	2,604.92	693.24	26.61%	54,728.76
125	139.03	0.31	0.22%	910.86	238.17	26.15%	54,867.79
214	2,420.51	7.27	0.30%	4,613.08	1,175.90	25.49%	57,288.30
180	782.93	23.78	3.04%	2,438.03	614.64	25.21%	58,071.23
152	300.24	1.5	0.50%	1,670.28	398.77	23.87%	58,371.47
57	36.7	0.66	1.79%	859.61	204.65	23.81%	58,408.17
77	226.57	1.57	0.69%	1,191.80	281.2	23.59%	58,634.74
6	80.13	13.43	16.76%	774.79	181.92	23.48%	58,714.87
241	3,155.38	62.52	1.98%	5,663.45	1,325.80	23.41%	61,870.25
242	439.52	1.89	0.43%	1,219.80	284.49	23.32%	62,309.77
109	1,079.69	27.25	2.52%	2,479.85	575.2	23.19%	63,389.46

42	11,789.00	139.85	1.19%	14,302.90	3,285.21	22.97%	75,178.46
225	2,808.06	10.36	0.37%	4,891.83	1,121.66	22.93%	77,986.52
172	331.54	14.64	4.41%	1,152.06	261.31	22.68%	78,318.06
156	91.42	0.45	0.49%	706.03	159.67	22.61%	78,409.48
165	1,095.37	11.05	1.01%	2,879.43	627	21.77%	79,504.85
84	190.46	1.62	0.85%	1,476.24	315.42	21.37%	79,695.31
39	31.46	0	0.00%	558.13	118.72	21.27%	79,726.77
204	92.99	7.65	8.22%	791.05	165.33	20.90%	79,819.76
139	535.88	0.59	0.11%	1,745.24	359.82	20.62%	80,355.64
234	463.5	0.5	0.11%	1,551.49	313.61	20.21%	80,819.14
119	63.94	0.01	0.01%	670.46	133.93	19.98%	80,883.08
183	3,602.10	25.88	0.72%	7,846.26	1,555.88	19.83%	84,485.18
161	1,198.78	2.86	0.24%	1,892.46	372.05	19.66%	85,683.96
122	3,388.68	10.15	0.30%	5,931.28	1,149.99	19.39%	89,072.64
41	7,338.91	745.96	10.16%	7,443.16	1,434.78	19.28%	96,411.55
144	2,167.07	4.96	0.23%	3,897.70	748.12	19.19%	98,578.62
178	4,404.05	7.17	0.16%	5,446.49	1,036.64	19.03%	102,982.67
209	485.2	0.06	0.01%	1,399.18	264.73	18.92%	103,467.87

96	32.89	0	0.00%	553.17	103.72	18.75%	103,500.76
22	409.93	0	0.00%	1,224.80	226.04	18.46%	103,910.69
104	346.54	4.76	1.37%	1,618.48	296.76	18.34%	104,257.23
208	135.31	8.54	6.31%	1,284.89	232.6	18.10%	104,392.54
117	840.55	59.17	7.04%	1,912.35	345.42	18.06%	105,233.09
17	228.66	0.11	0.05%	1,021.18	180.89	17.71%	105,461.75
91	65.98	0	0.00%	686.51	117.39	17.10%	105,527.73
213	2,933.06	18.44	0.63%	3,904.37	657.79	16.85%	108,460.79
227	6,650.27	53.71	0.81%	9,459.88	1,560.17	16.49%	115,111.06
90	17.25	0	0.00%	488.41	78.3	16.03%	115,128.31
220	575.29	1.38	0.24%	1,807.26	289.53	16.02%	115,703.60
231	2,269.95	0.52	0.02%	3,947.81	611.03	15.48%	117,973.55
197	61.62	4.32	7.01%	716.86	110.59	15.43%	118,035.17
193	6,258.26	8.65	0.14%	9,563.08	1,470.82	15.38%	124,293.43
76	9,797.42	13.3	0.14%	13,057.10	1,954.01	14.97%	134,090.85
1	4.3	0.07	1.52%	413.06	60.88	14.74%	134,095.15
141	4.83	0.19	4.03%	429.05	63.13	14.71%	134,099.98
221	1,167.18	2.46	0.21%	3,203.75	468.06	14.61%	135,267.16

181	183.75	0	0.00%	1,307.77	189.65	14.50%	135,450.91
176	226.01	0	0.00%	1,253.88	179.53	14.32%	135,676.92
192	96.37	2.92	3.03%	879.5	124.59	14.17%	135,773.29
16	3,942.60	8.71	0.22%	4,505.89	618.97	13.74%	139,715.89
142	1,640.78	0.2	0.01%	2,839.53	382.54	13.47%	141,356.67
239	4,909.19	40.43	0.82%	6,677.53	872.67	13.07%	146,265.86
162	306.85	0.18	0.06%	1,188.97	151.64	12.75%	146,572.71
203	5,688.32	5.7	0.10%	6,823.34	854.2	12.52%	152,261.03
10	306.27	0.04	0.01%	1,892.33	236.85	12.52%	152,567.30
52	665.28	0.39	0.06%	1,813.70	224.3	12.37%	153,232.58
218	776.46	0	0.00%	2,408.86	295.59	12.27%	154,009.04
184	343.62	0.42	0.12%	1,318.48	152.86	11.59%	154,352.66
29	223.6	0	0.00%	1,408.13	163.01	11.58%	154,576.26
219	1,258.43	0.45	0.04%	3,119.11	344.52	11.05%	155,834.69
7	189.83	0	0.00%	1,232.97	133.65	10.84%	156,024.52
169	3,567.51	17.03	0.48%	5,105.39	540.66	10.59%	159,592.03
207	238.17	0.09	0.04%	1,224.55	129.5	10.58%	159,830.20
58	270.75	0	0.00%	1,090.29	113.77	10.44%	160,100.95

118	235.63	5	2.12%	992.73	102.97	10.37%	160,336.58
26	12.81	0	0.00%	478.97	48.62	10.15%	160,349.39
206	95.74	0	0.00%	788.39	79.61	10.10%	160,445.13
68	72.07	0	0.00%	672.76	67.76	10.07%	160,517.20
240	539.87	2.42	0.45%	1,617.23	161.73	10.00%	161,057.07
159	89.78	0	0.00%	930	92.47	9.94%	161,146.85
233	423.91	2.17	0.51%	2,236.31	209.22	9.36%	161,570.76
35	498.88	0.7	0.14%	1,601.07	147.47	9.21%	162,069.64
148	12,592.40	390.28	3.10%	21,612.30	1,976.72	9.15%	174,662.04
238	1,075.14	0.78	0.07%	2,469.52	212.75	8.62%	175,737.18
163	193.31	3.7	1.91%	1,207.18	98.55	8.16%	175,930.49
53	7,378.28	14	0.19%	7,806.36	632.27	8.10%	183,308.77
27	136.63	0	0.00%	938.57	75.9	8.09%	183,445.40
115	1,693.42	0.01	0.00%	2,278.07	183.64	8.06%	185,138.82
116	4,554.09	18.64	0.41%	5,164.95	387.79	7.51%	189,692.91
79	15,804.70	203.72	1.29%	16,875.10	1,256.81	7.45%	205,497.61
73	164.23	3.76	2.29%	1,222.09	89.69	7.34%	205,661.84
140	867.34	1.46	0.17%	1,641.71	118.62	7.23%	206,529.18

88	3,496.87	39.73	1.14%	3,356.96	237.73	7.08%	210,026.05	
3	2,148.74	12.49	0.58%	4,352.97	301.09	6.92%	212,174.79	
235	1,402.67	17.49	1.25%	3,717.51	255.85	6.88%	213,577.46	
128	49.01	0.27	0.56%	872.73	59.97	6.87%	213,626.47	
46	4,359.85	4.61	0.11%	5,231.76	344.78	6.59%	217,986.32	
31	483.19	3.41	0.71%	1,459.82	95.63	6.55%	218,469.51	
48	15.67	0	0.00%	786.69	51.13	6.50%	218,485.18	
195	2,515.05	9.52	0.38%	7,180.87	425.77	5.93%	221,000.23	
38	1,303.35	0.52	0.04%	2,227.94	128.11	5.75%	222,303.58	
86	18.33	0	0.00%	504.84	28.5	5.64%	222,321.91	
133	2,712.15	0.01	0.00%	4,383.31	242.73	5.54%	225,034.06	
87	415.51	1.08	0.26%	1,368.31	75.05	5.49%	225,449.57	
171	1,674.88	9.36	0.56%	3,292.61	180.24	5.47%	227,124.45	
229	1,568.88	7.66	0.49%	3,554.04	187.66	5.28%	228,693.33	
114	298.61	0.01	0.00%	1,519.88	77.88	5.12%	228,991.94	
63	16,624.20	268.92	1.62%	25,935.20	1,316.74	5.08%	245,616.14	Exceeds representational selection
136	1,940.75	92.22	4.75%	4,285.92	209.23	4.88%	247,556.89	
82	380.38	0.01	0.00%	1,488.07	72.55	4.88%	247,937.27	

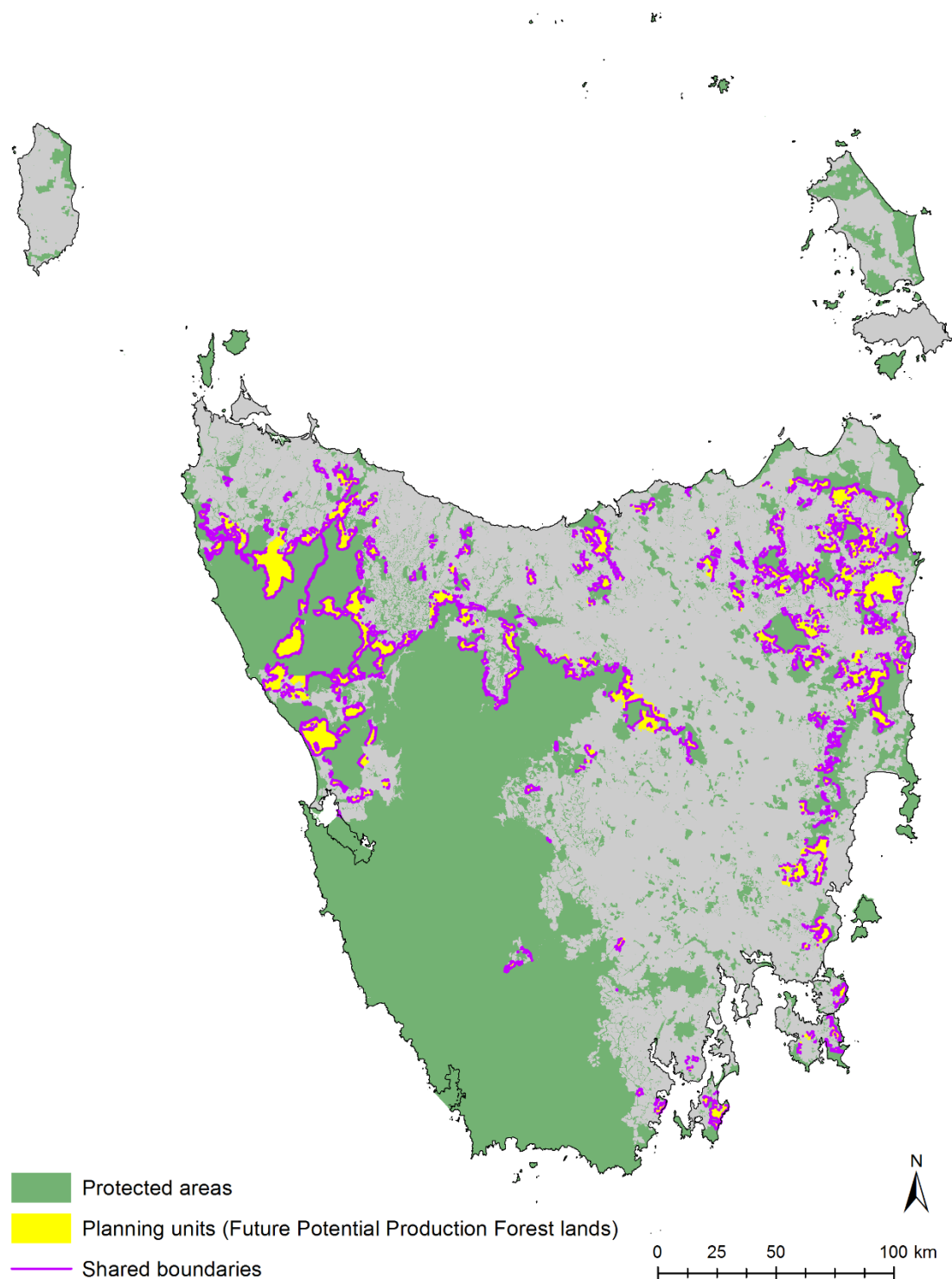
36	10,230.10	181.13	1.77%	5,999.11	290.9	4.85%	258,167.37
149	6,651.87	3.23	0.05%	5,866.46	283.84	4.84%	264,819.24
44	135.7	0.26	0.19%	1,166.03	49.91	4.28%	264,954.94
25	1,295.19	0.02	0.00%	2,823.70	116.43	4.12%	266,250.13
107	458.77	12.99	2.83%	2,297.06	86.31	3.76%	266,708.90
69	187.8	0	0.00%	1,012.08	37.39	3.69%	266,896.70
110	7,407.46	5.32	0.07%	9,039.33	327.93	3.63%	274,304.16
187	807.64	0.9	0.11%	2,116.77	75.69	3.58%	275,111.80
85	177.52	0	0.00%	863.45	30.62	3.55%	275,289.32
132	3,093.58	10.29	0.33%	6,122.96	191.16	3.12%	278,382.90
62	551.27	6.14	1.11%	1,726.16	50.33	2.92%	278,934.17
145	82.25	2.02	2.46%	1,040.28	30.21	2.90%	279,016.42
43	26,394.10	259.37	0.98%	30,594.80	886.69	2.90%	305,410.52
71	1,206.05	10.93	0.91%	2,749.09	74.5	2.71%	306,616.57
186	548.64	0.02	0.00%	1,853.68	47.83	2.58%	307,165.21
167	693.3	7.93	1.14%	1,621.88	39.89	2.46%	307,858.51
24	249.91	3.8	1.52%	1,055.68	24.34	2.31%	308,108.42
173	577.99	17.24	2.98%	1,838.26	34.47	1.88%	308,686.41

146	579.05	0.08	0.01%	1,694.15	31.59	1.86%	309,265.46
40	3,618.57	58.58	1.62%	4,565.44	82.76	1.81%	312,884.03
131	135.07	0	0.00%	1,121.25	19.51	1.74%	313,019.10
215	418.38	1.09	0.26%	1,685.14	27.79	1.65%	313,437.48
74	1,358.62	0	0.00%	3,417.01	56.13	1.64%	314,796.10
217	279.64	0	0.00%	1,251.18	20.41	1.63%	315,075.74
5	271.26	0	0.00%	1,032.00	14.94	1.45%	315,347.00
21	1,978.33	2.05	0.10%	3,524.85	50.93	1.44%	317,325.33
47	624.55	0	0.00%	1,848.08	25.73	1.39%	317,949.88
166	10,113.40	1.21	0.01%	5,294.50	73.32	1.38%	328,063.28
232	2,077.43	11.09	0.53%	3,198.02	38.08	1.19%	330,140.71
20	1,051.11	0	0.00%	2,378.89	25.91	1.09%	331,191.82
202	733.29	0	0.00%	1,536.23	15.32	1.00%	331,925.11
97	141.31	0.86	0.61%	969.68	8.44	0.87%	332,066.42
4	1,110.16	0	0.00%	1,999.83	15.7	0.79%	333,176.58
14	953.15	0.45	0.05%	2,019.26	14.86	0.74%	334,129.73
28	747.89	6.18	0.83%	1,514.94	11.06	0.73%	334,877.62
30	1,375.58	0.5	0.04%	1,917.66	13.7	0.71%	336,253.20

222	4,406.31	10.86	0.25%	8,859.15	55.97	0.63%	340,659.51
32	2,107.69	0.01	0.00%	3,233.73	20.16	0.62%	342,767.20
60	1,064.21	3.19	0.30%	2,056.61	12.51	0.61%	343,831.41
205	214.51	0	0.00%	1,047.80	6.11	0.58%	344,045.92
216	2,762.52	18.73	0.68%	5,516.69	29.57	0.54%	346,808.44
111	229.76	0	0.00%	1,205.59	6.38	0.53%	347,038.20
33	746.33	2.34	0.31%	2,251.94	11.72	0.52%	347,784.53
103	355.79	1.5	0.42%	1,758.15	7.41	0.42%	348,140.32
212	79.02	0	0.00%	1,093.00	4.09	0.37%	348,219.34
210	869.8	0.23	0.03%	3,505.68	12.65	0.36%	349,089.14
100	3,384.38	3.89	0.11%	4,239.59	12.64	0.30%	352,473.52
130	4,843.24	3.22	0.07%	5,282.40	14.59	0.28%	357,316.76
56	1,715.09	14.87	0.87%	2,313.29	6.06	0.26%	359,031.85
13	22.37	0	0.00%	809.15	1.56	0.19%	359,054.22
2	348.8	0	0.00%	1,359.41	2.39	0.18%	359,403.02
80	204.5	0	0.00%	1,975.28	2.69	0.14%	359,607.52
18	49.69	0	0.00%	635.86	0.8	0.13%	359,657.21
102	5,926.88	0.21	0.00%	5,298.90	5.76	0.11%	365,584.09

134	1,196.61	1.01	0.08%	4,085.30	3.98	0.10%	366,780.70
15	212.92	0	0.00%	1,003.59	0.95	0.10%	366,993.62
153	270.27	0	0.00%	1,176.65	1.11	0.09%	367,263.89
99	10,942.30	24.67	0.23%	8,087.13	3.89	0.05%	378,206.19
95	634.19	0.38	0.06%	1,430.80	0.67	0.05%	378,840.38
37	293.99	1.76	0.60%	1,160.61	0.5	0.04%	379,134.37
108	167.98	1.63	0.97%	1,057.81	0.18	0.02%	379,302.35
12	167.68	0.69	0.41%	1,051.72	0.24	0.02%	379,470.03
11	1,487.00	0	0.00%	0	0	0.00%	380,957.03
19	171.53	0	0.00%	0	0	0.00%	381,128.56
34	878.91	0	0.00%	0	0	0.00%	382,007.47
49	1,104.14	0	0.00%	2,422.72	0.07	0.00%	383,111.61
50	12.34	0	0.00%	0	0	0.00%	383,123.95
51	103.98	0	0.00%	0	0	0.00%	383,227.93
54	16.5	0	0.00%	0	0	0.00%	383,244.43
55	38.91	0	0.00%	0	0	0.00%	383,283.34
61	2,987.90	0	0.00%	0	0	0.00%	386,271.24
64	510.08	0	0.00%	0	0	0.00%	386,781.32

65	67.8	0	0.00%	0	0	0.00%	386,849.12
66	104.73	0	0.00%	0	0	0.00%	386,953.85
70	92.05	0	0.00%	0	0	0.00%	387,045.90
72	37.2	0	0.00%	0	0	0.00%	387,083.10
81	50.88	0	0.00%	0	0	0.00%	387,133.98
94	124.06	0	0.00%	0	0	0.00%	387,258.04
98	1,372.94	0	0.00%	0	0	0.00%	388,630.98
101	502.13	0	0.00%	0	0	0.00%	389,133.11
105	187.94	0	0.00%	0	0	0.00%	389,321.05
106	1,002.33	0	0.00%	0	0	0.00%	390,323.38
127	190.22	0	0.00%	0	0	0.00%	390,513.60
129	3.12	0	0.00%	0	0	0.00%	390,516.72
135	14.95	0	0.00%	0	0	0.00%	390,531.67
137	520.85	0	0.00%	0	0	0.00%	391,052.52
160	707.89	0	0.00%	0	0	0.00%	391,760.41
179	79	0	0.00%	0	0	0.00%	391,839.41
191	104.27	0	0.00%	0	0	0.00%	391,943.68



Appendix S11. Extents of shared boundaries between protected areas and planning units. Spatial data from “LIST Future Potential Production Forest” and from “LIST Land Tenure” from <http://www.thelist.tas.gov.au/> ©State of Tasmania.

Appendix S12. Lengths (km) of planning units' boundary adjacency with the existing protected area network, listed from largest to smallest proportion of perimeter. Spatial data from "LIST Future Potential Production Forest" and from "LIST Land Tenure" from <http://www.thelist.tas.gov.au/> ©State of Tasmania.

Planning unit	Planning unit area (ha)	Planning unit perimeter (km)	Length (km) of PA-adjacent boundary	Percent perimeter PA-adjacent	Cumulative planning unit area (ha)	Cumulative length (km) PA-adjacent boundaries
160	707.89	14.142	14.142	100.00%	707.89	14.14
129	3.12	1.386	1.377	99.34%	711.01	15.52
36	10,230.10	73.245	71.847	98.09%	10,941.11	87.37
153	270.27	9.004	8.657	96.14%	11,211.38	96.02
232	2,077.43	32.673	30.724	94.03%	13,288.81	126.75
118	235.63	6.832	6.198	90.71%	13,524.44	132.94
130	4,843.24	58.416	52.540	89.94%	18,367.68	185.48
60	1,064.21	25.726	23.025	89.50%	19,431.89	208.51
70	92.05	12.984	11.282	86.89%	19,523.94	219.79
86	18.33	2.014	1.749	86.85%	19,542.27	221.54
91	65.98	3.894	3.376	86.70%	19,608.25	224.92

135	14.95	2.290	1.979	86.44%	19,623.19	226.90
87	415.51	11.314	9.729	85.99%	20,038.70	236.63
218	776.46	22.868	19.362	84.67%	20,815.16	255.99
44	135.70	9.498	8.018	84.42%	20,950.86	264.01
20	1,978.33	22.782	18.813	82.58%	22,929.19	282.82
62	551.27	19.017	15.639	82.24%	23,480.46	298.46
64	510.08	9.969	8.126	81.51%	23,990.53	306.58
48	15.67	3.821	3.105	81.25%	24,006.20	309.69
35	498.88	13.271	10.780	81.22%	24,505.08	320.47
51	103.98	4.950	3.942	79.64%	24,609.05	324.41
80	204.50	18.394	14.631	79.55%	24,813.55	339.04
77	226.57	9.294	7.328	78.85%	25,040.12	346.37
22	409.93	9.992	7.868	78.74%	25,450.06	354.24
56	1,715.09	21.368	16.644	77.89%	27,165.15	370.88
11	1,487.00	40.607	31.345	77.19%	28,652.15	402.23
97	141.31	6.686	5.085	76.05%	28,793.45	407.31
27	136.63	6.625	5.030	75.93%	28,930.08	412.34
210	869.80	35.541	26.758	75.29%	29,799.88	439.10

57	36.70	6.158	4.618	74.99%	29,836.58	443.72
190	8.84	2.887	2.149	74.43%	29,845.42	445.87
157	613.68	18.524	13.763	74.30%	30,459.10	459.63
222	4,406.31	95.709	70.964	74.15%	34,865.41	530.59
43	26,394.10	413.870	306.522	74.06%	61,259.51	837.12
55	38.91	4.199	3.090	73.60%	61,298.42	840.21
74	1,358.62	39.277	28.628	72.89%	62,657.04	868.83
1	4.30	1.027	0.748	72.79%	62,661.34	869.58
238	1,075.14	29.418	21.341	72.54%	63,736.48	890.92
63	16,624.20	322.205	233.618	72.51%	80,360.68	1,124.54
61	2,987.90	32.021	23.101	72.14%	83,348.58	1,147.64
133	2,712.15	46.708	33.459	71.63%	86,060.73	1,181.10
215	418.38	15.664	11.211	71.57%	86,479.10	1,192.31
72	37.20	4.832	3.441	71.23%	86,516.30	1,195.75
38	1,303.35	20.828	14.782	70.97%	87,819.65	1,210.54
100	3,384.38	42.492	29.604	69.67%	91,204.03	1,240.14
154	290.29	16.311	11.353	69.61%	91,494.31	1,251.49
148	12,592.40	366.507	255.050	69.59%	104,086.71	1,506.54

26	12.81	1.650	1.147	69.54%	104,099.52	1,507.69
173	577.99	16.304	11.279	69.18%	104,677.51	1,518.97
166	10,113.40	57.200	39.401	68.88%	114,790.91	1,558.37
112	431.31	19.749	13.561	68.67%	115,222.22	1,571.93
176	226.01	10.587	7.240	68.39%	115,448.23	1,579.17
111	229.76	10.266	7.010	68.28%	115,677.99	1,586.18
73	164.23	9.452	6.442	68.16%	115,842.22	1,592.62
145	82.25	8.037	5.472	68.08%	115,924.47	1,598.09
152	300.24	14.709	9.853	66.99%	116,224.71	1,607.95
134	1,196.61	56.114	37.302	66.47%	117,421.32	1,645.25
183	3,602.10	86.811	57.253	65.95%	121,023.42	1,702.50
149	6,651.87	67.671	44.547	65.83%	127,675.29	1,747.05
180	782.93	29.959	19.664	65.64%	128,458.22	1,766.71
181	183.75	11.230	7.311	65.10%	128,641.97	1,774.02
150	547.06	14.220	9.154	64.38%	129,189.03	1,783.18
101	502.13	18.943	12.157	64.17%	129,691.16	1,795.33
29	223.60	11.525	7.344	63.72%	129,914.76	1,802.68
216	2,762.52	61.703	39.243	63.60%	132,677.28	1,841.92

54	16.50	1.945	1.236	63.55%	132,693.79	1,843.16
137	520.85	20.858	13.226	63.41%	133,214.64	1,856.38
34	878.91	17.622	11.168	63.37%	134,093.54	1,867.55
200	931.17	28.199	17.861	63.34%	135,024.72	1,885.41
98	1,372.94	23.795	15.017	63.11%	136,397.66	1,900.43
110	7,407.46	106.390	67.037	63.01%	143,805.12	1,967.46
114	298.61	15.035	9.372	62.33%	144,103.72	1,976.84
10	306.27	15.427	9.615	62.33%	144,410.00	1,986.45
203	5,688.32	74.161	46.138	62.21%	150,098.32	2,032.59
239	4,909.19	86.362	53.716	62.20%	155,007.51	2,086.31
234	463.50	14.636	9.085	62.07%	155,471.01	2,095.39
3	2,148.74	50.522	31.242	61.84%	157,619.75	2,126.63
227	6,650.27	112.541	68.939	61.26%	164,270.02	2,195.57
139	535.88	16.846	10.268	60.95%	164,805.90	2,205.84
151	65.81	3.635	2.208	60.74%	164,871.71	2,208.05
236	92.57	7.870	4.766	60.56%	164,964.28	2,212.81
127	190.22	18.852	11.407	60.51%	165,154.50	2,224.22
9	10.13	3.405	2.043	60.02%	165,164.63	2,226.26

14	212.92	20.524	12.312	59.99%	165,377.54	2,238.58
94	124.06	8.888	5.331	59.98%	165,501.60	2,243.91
12	167.68	7.430	4.447	59.85%	165,669.27	2,248.35
156	91.42	3.934	2.352	59.78%	165,760.69	2,250.71
194	510.83	25.000	14.921	59.68%	166,271.52	2,265.63
53	7,378.28	85.523	50.780	59.38%	173,649.80	2,316.41
124	369.30	11.130	6.568	59.01%	174,019.09	2,322.97
88	3,496.87	43.796	25.789	58.88%	177,515.96	2,348.76
136	1,940.75	53.903	31.443	58.33%	179,456.71	2,380.21
141	4.83	1.277	0.745	58.31%	179,461.54	2,380.95
16	228.66	46.179	26.817	58.07%	179,690.20	2,407.77
213	2,933.06	44.308	25.552	57.67%	182,623.26	2,433.32
228	2,019.40	46.530	26.791	57.58%	184,642.66	2,460.11
49	1,104.14	21.576	12.419	57.56%	185,746.80	2,472.53
46	4,359.85	72.360	41.442	57.27%	190,106.65	2,513.97
117	840.55	19.556	11.152	57.03%	190,947.20	2,525.12
122	3,388.68	67.747	38.439	56.74%	194,335.88	2,563.56
171	1,674.88	35.888	20.224	56.35%	196,010.76	2,583.79

182	430.61	13.702	7.695	56.16%	196,441.37	2,591.48
65	67.80	10.239	5.690	55.57%	196,509.17	2,597.17
85	177.52	5.565	3.071	55.19%	196,686.69	2,600.24
79	15,804.70	219.798	120.795	54.96%	212,491.39	2,721.04
18	171.53	3.218	1.764	54.83%	212,662.92	2,722.80
235	1,402.67	38.214	20.925	54.76%	214,065.59	2,743.73
144	2,167.07	44.154	24.045	54.46%	216,232.66	2,767.77
229	1,568.88	37.746	20.496	54.30%	217,801.54	2,788.27
184	343.62	13.355	7.251	54.29%	218,145.16	2,795.52
193	6,258.26	123.996	67.024	54.05%	224,403.42	2,862.54
83	224.75	15.152	8.148	53.77%	224,628.18	2,870.69
15	3,942.60	7.182	3.856	53.69%	228,570.78	2,874.55
159	89.78	6.312	3.353	53.11%	228,660.56	2,877.90
179	79.00	4.809	2.524	52.49%	228,739.56	2,880.42
52	665.28	15.699	8.240	52.49%	229,404.84	2,888.66
30	1,375.58	18.308	9.607	52.48%	230,780.42	2,898.27
233	423.91	23.333	12.222	52.38%	231,204.32	2,910.49
76	9,797.42	177.074	91.912	51.91%	241,001.74	3,002.40

115	1,693.42	22.660	11.752	51.86%	242,695.16	3,014.16
147	258.32	11.352	5.866	51.67%	242,953.48	3,020.02
82	380.38	14.753	7.583	51.40%	243,333.86	3,027.61
126	349.18	17.295	8.883	51.36%	243,683.04	3,036.49
131	135.07	9.013	4.616	51.22%	243,818.12	3,041.10
39	31.46	2.440	1.241	50.85%	243,849.58	3,042.35
40	3,618.57	58.065	29.387	50.61%	247,468.15	3,071.73
108	167.98	8.830	4.464	50.55%	247,636.13	3,076.20
146	579.05	16.340	8.225	50.34%	248,215.18	3,084.42
125	139.03	7.147	3.580	50.09%	248,354.21	3,088.00
84	190.46	17.585	8.787	49.97%	248,544.67	3,096.79
103	355.79	17.566	8.777	49.97%	248,900.46	3,105.57
196	696.92	21.945	10.961	49.95%	249,597.38	3,116.53
13	22.37	5.009	2.498	49.88%	249,619.76	3,119.02
241	3,155.38	85.444	42.439	49.67%	252,775.14	3,161.46
92	46.87	4.421	2.180	49.30%	252,822.01	3,163.64
172	331.54	9.718	4.782	49.21%	253,153.54	3,168.43
202	733.29	13.985	6.881	49.20%	253,886.83	3,175.31

220	575.29	18.101	8.891	49.12%	254,462.12	3,184.20
237	65.53	4.084	1.991	48.74%	254,527.65	3,186.19
47	624.55	16.498	8.034	48.70%	255,152.20	3,194.22
169	3,567.51	61.999	30.115	48.57%	258,719.71	3,224.34
163	193.31	13.637	6.611	48.48%	258,913.02	3,230.95
75	990.74	27.629	13.305	48.16%	259,903.76	3,244.25
143	1,175.75	23.973	11.431	47.68%	261,079.51	3,255.68
199	461.90	20.125	9.476	47.09%	261,541.41	3,265.16
45	165.26	9.450	4.419	46.76%	261,706.67	3,269.58
66	104.73	8.509	3.943	46.34%	261,811.41	3,273.52
187	807.64	24.219	11.212	46.29%	262,619.05	3,284.73
214	2,420.51	54.670	25.041	45.80%	265,039.56	3,309.78
186	548.64	20.094	9.094	45.26%	265,588.20	3,318.87
192	96.37	7.417	3.319	44.75%	265,684.57	3,322.19
25	1,295.19	30.106	13.379	44.44%	266,979.76	3,335.57
123	228.58	8.898	3.951	44.40%	267,208.34	3,339.52
90	17.25	1.788	0.784	43.86%	267,225.59	3,340.30
174	176.50	5.700	2.473	43.39%	267,402.09	3,342.78

188	1,674.55	45.868	19.684	42.92%	269,076.64	3,362.46
42	11,789.00	247.986	105.854	42.69%	280,865.64	3,468.31
212	79.02	9.756	4.126	42.29%	280,944.67	3,472.44
59	1,004.01	21.735	9.082	41.78%	281,948.68	3,481.52
69	187.80	7.346	3.068	41.77%	282,136.48	3,484.59
189	758.48	25.497	10.586	41.52%	282,894.96	3,495.18
113	3,769.88	74.088	30.338	40.95%	286,664.84	3,525.51
168	1,414.49	27.623	11.265	40.78%	288,079.33	3,536.78
230	970.38	30.177	12.157	40.29%	289,049.70	3,548.94
225	2,808.06	61.208	24.636	40.25%	291,857.76	3,573.57
120	768.96	18.228	7.308	40.09%	292,626.72	3,580.88
198	1,682.98	47.946	19.088	39.81%	294,309.70	3,599.97
37	293.99	9.375	3.710	39.57%	294,603.69	3,603.68
231	2,269.95	41.929	16.473	39.29%	296,873.64	3,620.15
31	483.19	12.924	5.068	39.21%	297,356.83	3,625.22
104	346.54	19.329	7.574	39.19%	297,703.37	3,632.79
93	143.48	5.925	2.319	39.13%	297,846.85	3,635.11
211	14,389.80	112.262	43.570	38.81%	312,236.65	3,678.68

128	49.01	6.858	2.657	38.74%	312,285.66	3,681.34
8	412.04	15.194	5.873	38.65%	312,697.70	3,687.21
221	1,167.18	36.701	14.076	38.35%	313,864.88	3,701.29
191	104.27	8.483	3.251	38.33%	313,969.14	3,704.54
19	1,051.11	10.005	3.725	37.23%	315,020.25	3,708.27
23	901.65	24.232	8.973	37.03%	315,921.90	3,717.24
206	95.74	4.894	1.808	36.95%	316,017.64	3,719.05
219	1,258.43	38.515	14.185	36.83%	317,276.07	3,733.23
132	3,093.58	71.536	26.124	36.52%	320,369.65	3,759.36
195	2,515.05	96.367	35.030	36.35%	322,884.70	3,794.39
50	12.34	3.398	1.227	36.11%	322,897.03	3,795.61
24	249.91	7.993	2.881	36.04%	323,146.94	3,798.49
41	7,338.91	100.681	36.133	35.89%	330,485.85	3,834.63
209	485.20	13.426	4.795	35.71%	330,971.05	3,839.42
243	838.36	31.487	11.171	35.48%	331,809.41	3,850.59
58	270.75	8.640	3.055	35.36%	332,080.16	3,853.65
167	693.30	15.967	5.588	35.00%	332,773.46	3,859.24
116	4,554.09	67.326	23.537	34.96%	337,327.55	3,882.77

178	4,404.05	78.208	27.193	34.77%	341,731.60	3,909.97
5	271.26	7.179	2.487	34.65%	342,002.86	3,912.45
96	32.89	2.548	0.881	34.58%	342,035.75	3,913.33
240	539.87	16.283	5.594	34.35%	342,575.62	3,918.93
138	1,126.33	34.931	11.939	34.18%	343,701.95	3,930.87
155	2,414.67	67.052	22.231	33.15%	346,116.62	3,953.10
208	135.31	10.352	3.413	32.96%	346,251.93	3,956.51
78	228.93	9.524	3.128	32.84%	346,480.86	3,959.64
99	10,942.30	91.000	29.714	32.65%	357,423.16	3,989.35
140	867.34	14.880	4.849	32.59%	358,290.50	3,994.20
142	1,640.78	36.227	11.777	32.51%	359,931.28	4,005.98
33	746.33	28.340	9.082	32.05%	360,677.61	4,015.06
207	238.17	12.835	4.073	31.73%	360,915.77	4,019.13
81	50.88	4.609	1.458	31.63%	360,966.65	4,020.59
217	279.64	10.205	3.213	31.48%	361,246.30	4,023.80
95	634.19	13.332	4.186	31.40%	361,880.49	4,027.99
107	458.77	29.866	9.097	30.46%	362,339.26	4,037.09
226	1,350.11	31.474	9.559	30.37%	363,689.37	4,046.64

177	2,374.05	51.830	15.710	30.31%	366,063.42	4,062.35
201	48.33	6.738	2.036	30.21%	366,111.74	4,064.39
109	1,079.69	28.732	8.480	29.52%	367,191.43	4,072.87
204	92.99	5.030	1.435	28.52%	367,284.43	4,074.31
21	1,978.33	59.508	16.904	28.41%	369,262.76	4,091.21
28	747.89	13.045	3.688	28.27%	370,010.64	4,094.90
161	1,198.78	19.556	5.527	28.26%	371,209.42	4,100.42
2	348.80	15.086	4.150	27.51%	371,558.23	4,104.57
224	497.06	14.615	3.982	27.25%	372,055.29	4,108.56
242	439.52	10.303	2.761	26.80%	372,494.82	4,111.32
205	214.51	7.641	2.006	26.25%	372,709.33	4,113.32
158	2,712.40	59.006	15.190	25.74%	375,421.73	4,128.51
165	1,095.37	42.548	10.889	25.59%	376,517.10	4,139.40
6	80.13	5.451	1.368	25.10%	376,597.22	4,140.77
68	72.07	3.767	0.937	24.87%	376,669.29	4,141.71
7	189.83	10.103	2.462	24.37%	376,859.12	4,144.17
121	205.92	10.897	2.562	23.51%	377,065.04	4,146.73
223	1,024.08	34.727	7.684	22.13%	378,089.12	4,154.41

185	276.17	8.819	1.941	22.02%	378,365.29	4,156.36
4	1,110.16	18.093	3.939	21.77%	379,475.45	4,160.30
197	61.62	5.363	1.020	19.01%	379,537.07	4,161.31
106	1,002.33	20.830	3.678	17.66%	380,539.40	4,164.99
164	765.01	21.799	3.716	17.05%	381,304.41	4,168.71
102	5,926.88	60.741	9.939	16.36%	387,231.29	4,178.65
170	29.08	2.439	0.393	16.13%	387,260.37	4,179.04
67	1,092.41	18.195	2.732	15.01%	388,352.78	4,181.77
89	332.99	12.831	1.797	14.01%	388,685.76	4,183.57
17	49.69	8.274	1.022	12.35%	388,735.46	4,184.59
32	2,107.69	48.998	5.502	11.23%	390,843.15	4,190.09
162	306.85	9.515	0.773	8.12%	391,150.00	4,190.87
175	360.86	17.759	1.377	7.75%	391,510.86	4,192.24
71	1,206.05	23.847	1.703	7.14%	392,716.91	4,193.95
119	63.94	3.783	0.256	6.77%	392,780.85	4,194.20
105	187.94	6.728	0.262	3.89%	392,968.79	4,194.46

Appendix S13. Vegetation groups from TasVeg4.0 in the Tasmanian protected area network and within the Future Potential Production Forest Land. Of the ten major vegetation groups listed by TasVeg4.0, only native grasslands are less than 30% protected.

Vegetation Group from TasVeg4.0	Area in TAS (ha)	Area in PAs (ha)	Percent Protected	Area in FPPFs	Percent in FPPFs
Rainforest and related scrub	757,121	605,331	79.95%		
Moorland, sedgeland, rushland	596,639	533,376	89.40%		
Wet eucalypt forest and woodland	1,123,146	605,463	53.91%		
Scrub, heathland, coastal complexes	468,469	331,628	70.79%		
Non eucalypt forest and woodland	191,136	99,061	51.83%		
Dry eucalypt forest and woodland	1,557,855	548,415	35.20%		
Native grassland	128,291	19,147	14.92%	519.663	0.41%
Saltmarsh and wetland	34,559	18,407	53.26%		
Highland and treeless vegetation	135,549	123,359	91.01%		
Other natural environments	249,095	114,366	45.91%		

Appendix S14. The area of native grasslands found within planning units represents the only vegetation group listed by TasVeg4.0 that is less than 30% protected by the existing protected area network in Tasmania. Native grasslands are less than 15% protected.

Planning Unit	Native grassland area (ha)
1	0
2	0
3	0
4	0
5	0
6	0
7	0
8	0
9	0
10	0
11	0
12	0
13	0
14	0
15	0
16	0
17	0
18	0
19	0
20	0
21	0
22	0
23	0.041625202
24	0

25	0
26	0
27	0
28	0
29	0
30	0
31	0
32	0
33	0
34	0
35	0
36	0
37	0
38	0
39	0
40	0
41	12.0067997
42	8.384070396
43	5.907020092
44	0
45	0
46	0
47	0
48	0
49	0
50	0
51	0
52	0

53	0.189279005
54	0
55	0
56	0
57	0
58	0
59	0
60	0
61	0
62	0
63	0
64	0
65	0
66	0
67	0
68	0
69	0
70	0
71	0.024143299
72	0
73	0
74	3.385200024
75	0
76	2.319499969
77	0
78	0
79	226.8099976
80	0

81	0
82	0
83	0
84	1.910040021
85	1.842929959
86	0
87	0
88	0
89	0
90	0
91	0
92	0
93	0
94	0
95	0
96	0
97	0
98	0
99	0
100	0
101	0
102	0
103	0
104	0
105	0
106	0
107	0
108	0

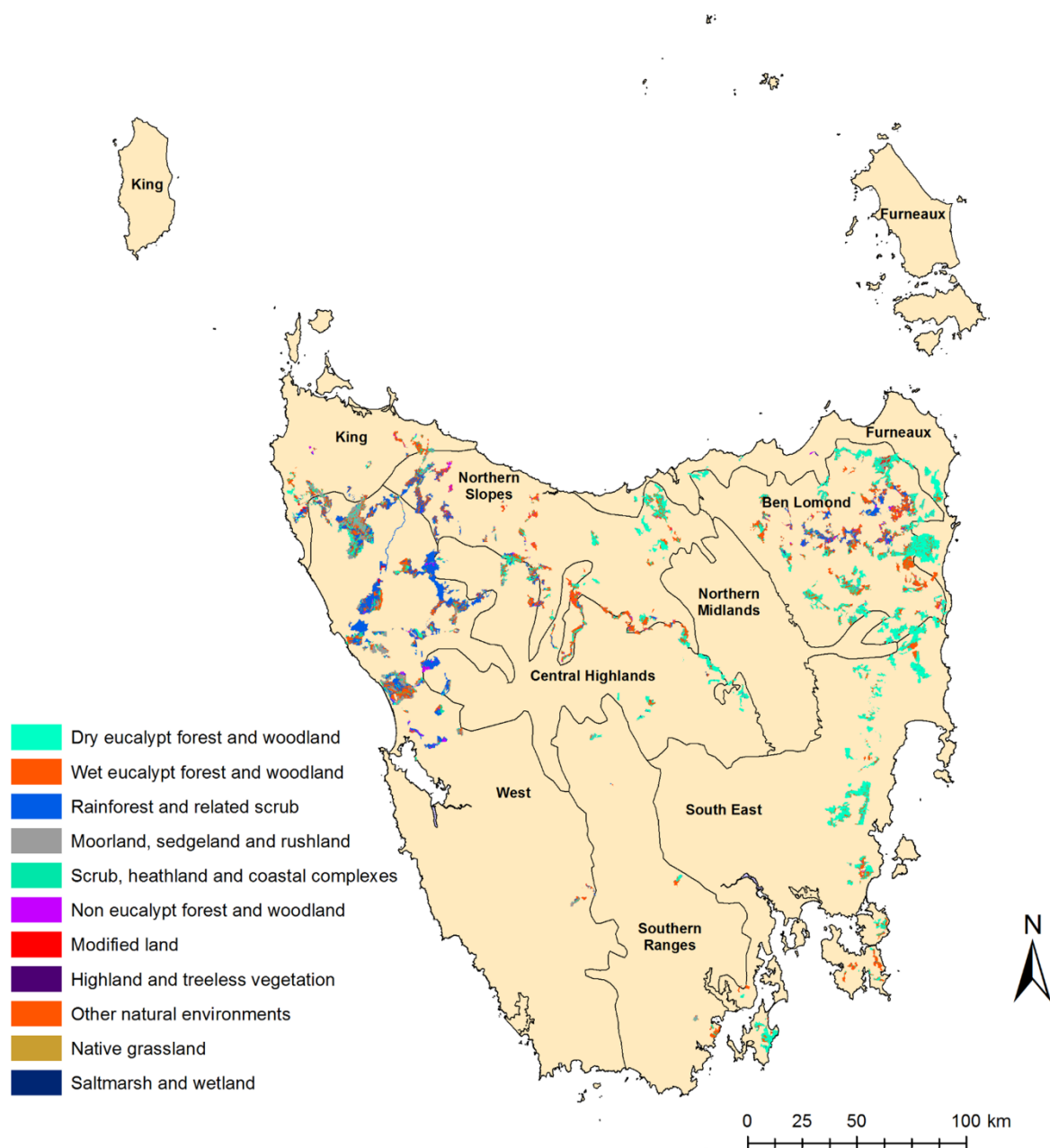
109	0
110	0.0225772
111	0
112	0
113	5.825419903
114	0
115	0
116	0.414568007
117	0
118	0
119	0
120	0
121	0
122	0
123	0
124	0
125	0
126	0
127	0
128	0
129	0
130	0
131	0
132	0
133	5.436289787
134	0
135	0
136	0.000476012

137	0
138	0
139	0
140	0.409972996
141	0
142	0
143	0
144	0
145	0
146	0
147	0
148	0
149	0
150	0
151	0.00186428
152	0
153	0.105233997
154	0
155	0.00853229
156	0
157	0
158	2.548860073
159	0
160	0
161	0
162	0
163	0
164	0

165	0
166	0
167	0
168	0
169	6.542250156
170	0
171	0
172	0
173	0
174	0
175	0.00883776
176	0
177	0
178	2.732039928
179	0
180	0
181	0
182	0
183	0.289970994
184	0
185	0
186	0
187	2.71984005
188	0
189	0
190	0
191	0
192	0

193	80.59490204
194	2.087280035
195	5.543529987
196	0.0151407
197	8.538069725
198	0
199	10.06700039
200	0.285829008
201	0
202	0
203	0
204	0
205	0
206	0
207	0
208	0
209	0
210	0
211	0
212	0
213	0
214	0.0424853
215	0
216	0
217	0
218	0
219	0
220	0

221	0.51463902
222	0.493656993
223	0
224	0
225	0
226	6.244249821
227	39.68420029
228	71.2276001
229	0
230	0
231	0
232	0
233	0
234	0
235	1.378939986
236	0
237	0
238	0
239	0
240	0
241	0.785265982
242	0
243	2.272900105



Appendix S15. Vegetation groups of the Future Potential Production Forests of Tasmania. Of these, only native grasslands are less than 30% protected, and thus considered underrepresented, in the existing protected area network of Tasmania. Spatial data from “TASVEG 4.0 from www.theLIST.tas.gov.au ©State of Tasmania.

Appendix S16. Planning units ranked by the simple multi-objective strategy.

Contact PhD candidate or see the pending publication to download the .xlsx file.

Appendix S17. Planning units ranked by the jurisdictionally informed strategy. See .xlsx file.

Contact PhD candidate or see the pending publication to download the .xlsx file.

Appendix S18. Three area budgets derived from extreme cost aversion, the business-as-usual approach, and a commitment to protecting 30% of biogeographic units, respectively, can be used to translate planning unit rankings into reasonable gazettelement plans.

Budget	Area (ha) protectable	No. planning units to protect	Connectivity: total length (km) of PA-adjacent boundaries	Area (ha) suitable for timber plantations
Conservative	12,255	14	220.791	0
Business as usual	25,209	11	424.676	877
Ambitious	234,705	123	2,544.121	13,631

Appendix S19. The areas suggested for protection by each of the six data-driven scenarios ranged from 0% to 92% congruence with the 25,209 ha that have been proposed for gazettment as PAs by the Tasmanian state government.

Scenario	Area (ha) overlap	Percent overlap
Biogeographic representation	23,302	92.44%
Zero opportunity cost	0	0%
Threat mitigation	0	0%
Connectivity	22,259	88.30%
Multiple objectives (simple) – BAU budget	0	0%
Multiple objectives (jurisdictionally informed) – BAU budget	0	0%

Appendix S20. List of acknowledged caveats in the data and methodology.

- Observation records of weed species and phytophthora were obtained from the Tasmanian Natural Values Atlas database; these records were not collected systematically, and thus planning units containing roads or walking tracks were more likely to have observance records.
- Threatened species' geographic distributions are not available at a fine enough resolution to be relevant to the scale of the FPPF planning units and therefore were not included as conservation features in the prioritisations.
- Calculations of percent of bioregions protected exclude the Bass Strait islands that are components of Wilson's Promontory bioregion, the majority of which is located on mainland Australia. All other Bass Strait Islands, which belong to the King and Furneaux bioregions, are included in all calculations.
- Eight "iconic" walking track upgrades were undertaken as part of a federally funded Community Forest Walks Program, all with sections located within "Future Potential Production Forests": the Arm River Track, Warners Track, Higgs Track, Junee Cave Track, Lake Skinner Track, Twisted Sister Track (in the Florentine), Blue Tier Giant Walk and the Halls Falls Track. The Giant Walk and Halls Falls Track are wholly (and only) in FPPFs. Sections of the Three Capes Track, an international tourist attraction, also pass through FPPF land, including the Mt Fortescue section and the current start of the track at Denman's Cove. Although I did not include the presence nor extent of "iconic" walking tracks as conservation priority features in my analyses, the Commonwealth funding of these very local outdoor recreation sites, as well as the catering to international tourists, exemplifies that a given feature of FPPF planning units may contribute to or detract from the objectives of multiple jurisdictions.
- Area budgets are only one possible constraint on the potential conversion of FPPF land to reserves. Budgets may also be set for costs (e.g., of forgoing forestry opportunities).
- Old growth status, suitable swift parrot habitat, and centres of paleo-endemism were not included in my prioritisations, but their inclusion should be considered in a more in-depth analysis that may influence tenure allocation decisions in Tasmania.