Watershed scale conservation planning across Amazonia

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Abstract

Amazonia is a megadiverse tropical forest biome of continental dimensions. Although still largely intact, the biome is highly threatened and future infrastructure projects associated with the low governance in Amazonian countries suggest a bleak scenario for the region in years to come. Biodiversity conservation in Amazonia has focused on creating a robust protected area (PA) network to face human-related threats. At present, some 34 % of the hydrologic domain of the entire Amazon Basin is under some category of protection. However, total acreage may not be the best way to assess levels of protection against anthropogenic pressures. This thesis considers the vulnerability to anthropogenic threats and the distribution of PAs throughout Amazonia at the scale of major watersheds. Watersheds are presented not just as a scale of analysis but also as a viable option for conservation planning units across Amazonia. Analysis of 23 major watersheds indicated that high-vulnerability are widespread all over Amazonia. However, the most threatened areas are located in the southeastern and western portions of the biome and there is clear evidence of a mismatch between PAs and high vulnerability areas. This imbalance in the allocation of conservation investments within the biome leads to either under-protection or over-protection, creating redundancy in setting aside similar habitats, thereby misusing scarce available resources. In addition, the systematic PA avoidance of high vulnerability areas is a strategy that is both deceptive and risky, because the future prospects of biodiversity conservation performance of existing hinterland reserves are less than sanguine once they eventually confront severe threats from advancing development frontiers. Creating PAs far from high pressure areas is a reasonable strategy to meet global conservation goals, but often merely serve to justify political objectives with questionable impacts on biodiversity protection.

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Declaration of research independence

The research presented in this thesis is all my own work. I conceived the research ideas, sampling design, led the fieldwork and other data collection, and am responsible for the data analysis and write-up. However, my supervisor (Carlos Peres) provided advices during all the steps of my research, including the write-up process. This contribution is recognised through co-authorship in each of the chapters. Dr. Laura Hess provided the flood map used in the thesis and for this reason is considered one of the co-authors of Chapter 5. Despite I used the phrase “we” in the data chapters as a way to acknowledge the contributions made by my supervisor, I am the responsible for the contents and result interpretations presented by this thesis.

Acknowledgements

This thesis marks the end of five incredible years of my life. I had the opportunity to work with my major passion, the Amazonian biome, and to live in an amazing country. England received me with gentleness and understanding, turning my distrust into love after the years living here. I learnt the value of a blue sky after many cloudy days and the beauty of each one of the seasons (not sure yet about winter). I grew fond of the pubs and beers (sure), little birds, English artists and the people from this country. This PhD was as much a personal as it was a professional experience and I am very grateful for that.

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Fieldwork was a very important part of my PhD. Although not all was eventually included in the final thesis, the knowledge acquired during the long and tough field campaigns, complete with their logistical problems, was of crucial importance for the professional I am nowadays. I thanks HRT and CEUC staff for all the help provided during fieldwork campaigns and all the friends I made in this period. Sebastião, Marcela, Seu Dimas, Bahia, Marcelo, Soldado, Neguinho, Donguinho and Dongão are people I could not fail to mention for all the help given and for their trust in me to conduct missions that on many occasions could have put their life at risk. Felipe and Ricardo were also of great importance in supporting my fieldwork activities.

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Chapter 1

Introduction

Photo: *Panramic canopy view of Amazonia rainforest. Photograph: Davi Teles*

Amazonia is the world's largest remaining area of continuous tropical forest, extending in its *sensu latissimo* definition over 8.12 million km^2 across nine countries of South America (Eva et al. 2005). The biome is widely acknowledged for its exuberant biodiversity (Hopkins 2007; Hoorn et al. 2010; Nores 1999), the wide range of ecosystem services provided (Portela & Rademacher 2001), and its crucial role in climate regulation (Soares-Filho et al. 2010). Amazonia's importance to global scale environmental and biodiversity assets is only rivalled by the extent to which it is threatened.

1.1 Amazonia under threat

The downward trend observed in Amazonian deforestation (Nepstad et al. 2014) and the success of supply-chain governance actions (Gibbs et al. 2015; Assunção & Rocha 2014) can give the false impression that the battle against deforestation has been won in Amazonia (Tollefson 2015b). However, deforestation rates are very sensitive to political decisions and recent indications on this front are not promising. Changes in environmental laws (de Marques & Peres 2015), choice of representatives for important ministries (Tollefson 2015a), and future infrastructure projects already sanctioned beyond a blueprint (Nepstad et al. 2014; Lees et al. 2016) are expected to have an important impact, increasing deforestation across the biome. Deforestation rates are also influenced by currency exchange rates (Richards et al. 2012) and market demands for commodity goods (Lourival et al. 2008). All Amazonian countries are primarily commodity exporters and therefore the biome's deforestation rate can be considered intrinsically related to national economies. For example, the Brazilian currency is in free-fall and recent deforestation rates are several times higher than rates observed in 2013-2014 (Fearnside 2015). Lesser impacts generated by smallhold farmers (Imbernon & Branthomme 2001), settlement projects (Barni et al. 2015), and timber extraction with the leakage effect from legal forest concessions (Oliveira et al. 2007) also contribute to the deforestation in the biome.

Nonetheless, deforestation is not the only threat to Amazonian biodiversity. Although with a lesser intensity than in the 1980s (Goulding et al. 2003), gold mining is still a real threat in Amazonia (Alvarez-Berríos & Mitchell Aide 2015). The high recent prices of gold in global markets make it advantageous to extract the ore from areas previously considered unprofitable, such as those from low-grade deposits beneath Amazonia, leading to habitat conversion and river and soil contamination (Swenson et al. 2011). Other mining activities are also present in Amazonia with similar negative ecosystem impacts (Mendes-Oliveira et al. 2013). Oil and gas exploitation fields are spread across all of lowland Amazonia and millions of hectares of new areas have recently received concessions for exploitation (EPE 2016). Oil spill accidents are not rare in exploitation areas and such events could have catastrophic consequences to local biodiversity (Yapa & Tao Shen 1994). In fact, oil exploitation generates diverse detrimental social and environmental impacts throughout its entire production chain (O'Rourke & Connolly 2003) and the large amount of oil concessions in Amazonia is now a major threat to long-term conservation of the biome.

Seasonally-flooded várzea and igapó forest habitats for many endemic species (Junk & Soares 2001; Wittmann et al. 2004), and an important source of fish, timber and fibre products for local populations (Parolin 2000), are threatened by illegal selective logging, removal of native vegetation for small scale agriculture and cattle ranching (Junk et al. 2010; de Queiroz 2010), and construction of hydroelectric dams (Moser et al. 2014). Indeed, 246 hydroelectric dams are currently planned or under construction in Amazonia (Lees et al. 2016) despite their perverse effects on long-term biodiversity maintenance (Benchimol & Peres 2015), carbon emission (Fearnside 1995), connectivity between habitats (Finer & Jenkins 2012) and fish migrations (Barthem et al. 1991).

All the aforesaid facts, not to mention the hard to quantify problems of overfishing (Alho et al. 2015) and overhunting (Peres et al. 2006), are symptomatic of who has the upper hand in ongoing battles between economic development and environmental conservation in Amazonia. The biome is clearly in need of conservation interventions. So far, such conservation efforts in Amazonia have been focussed on the creation of a robust Protected Area (PA) network (Soares-Filho et al. 2006), which is considered critical to safeguard

Amazonian biodiversity under any anticipated future scenario (Soares-Filho et al. 2006; Laurance et al. 2001).

1.2 Protected Areas in Amazonia

The history of PA creation in Amazonia dates back to the 1970s, with the creation of the Amazonia National Park (Rylands & Brandon 2005) Since then, excluding any overlap between PAs, circa 34% of the hydrologic domain of Amazonia is now included under some category of protection across the nine Amazonian countries (Table 1.1). However, total acreage might not be the best way of assessing the degree of biodiversity protection.

Table 1.1. Number and size (ha) of Protected Areas in the hydrological domain of Amazonia. The figures presented here represent values for "Designated" PAs in the WPDA[1] database after the removal of overlapping areas between PAs. The category of the most restrictive use was prioritised, while maintaining the overall original area. For legal reasons, Indigenous Lands were always prioritised in relation to other PA categories.

	Category	Definition[2]		N°	Size
Strictly protected	Ia	Strict Nature Reserve		35	15,903,462.7
	Ib	Wilderness Area		1	1,500,405.3
	II	National Park		101	43,330,498.7
	III	Natural Monument or Feature		14	438,633.8
	I–III		Sub-total	151	61,173,000.6
Sustainable use	IV	Habitat/Species Management Area		72	3,614,140.9
	V	Protected Landscape/Seascape		105	32,291,989.3
	VI	Managed Resource Protected Area		179	62,282,069.4
	IV - VI		Sub-total	356	98,188,199.7
	Indigenous Land		Sub-total	347	116,572,497.5
			Total	854	275,933,697.8

[1] IUCN - World Database on Protected Areas [2] Lausche & Burhenne-Guilmin 2011

Brazil controls almost two thirds of the entire Amazonian forest. Within this country, the selection of priority areas for conservation has always been advanced using expert opinions (Schulman et al. 2007). Over the years, there have been changes in the criteria used to define priority areas but the process has been largely based on scientific understanding (Rylands & Brandon 2005). However, the priority areas identified in several workshops by scientists (Schulman et al. 2007) were not ultimately used as the major foundation in the implementation of PAs. The current PA network seems to be the result of several *ad hoc* decisions, possibly as result of the myriad of pre-existing conflicting interests whenever a reserve is created in Amazonia (Fearnside 2003).

Regardless of the reason, the existing PA network does not adequately cover Amazonian biodiversity (Schulman et al. 2007), habitat types (Gaston et al. 2008), or endemism centres (Da Silva et al. 2005). PAs have been relatively successful in containing deforestation in the biome (Soares-Filho et al. 2010; Nolte et al. 2013). However, this effectiveness is associated with the physical distance between PAs and highly threatened areas (Joppa & Pfaff 2010; Pfaff et al. 2015), which are generally avoided in the selection of PA sites (Joppa & Pfaff 2009). In addition, a large proportion of PAs in Amazonia have no management plan in place (Campos-Silva et al. 2015) and present a low density of either in situ or ex situ reserve staff to enforce their protection (Peres & Terborgh 1995), jeopardizing the PA's ability to safeguard the biodiversity assets they contain. In short, the present Amazonian PA network is inefficiently distributed and ineffectively managed. However, there is still time, albeit not much time, to adequately protect Amazonian biodiversity assets. To that end, it will be necessary to establish a systematic conservation planning programme, where all conservation actions have clearly established goals and are well coordinated (Margules & Pressey 2000).

The major goal of this thesis is to propose the watershed as the basic unit for systematic conservation planning in Amazonia. The idea is by no means an original concept, with Peres & Terborgh (1995) proposing the alignment of PA boundaries with watershed dividers across the biome over 20 years ago.

However, here I provide a more detailed examination of this approach, expanding the idea and highlighting the advantages presented by it, consequently making the concept more accessible and applicable to the reality of Amazonia. I expect that by the end of this thesis, the reader will be convinced of the suitability of this approach to address the gap between knowledge and application, a major limitation in modern conservation planning (Knight et al. 2008). Hopefully, the reader will realise the considerable conceptual potential of watershed scale planning and implementing of future conservation actions.

1.3 Watersheds as conservation planning units

The principal difference of using watersheds as planning units between freshwater and terrestrial conservation programmes resides in their species distribution patterns. Watersheds can be considered as geographic units in processes such as gene flow, gene drift and natural selection for aquatic species (Wishart & Davies 2003). In contrast, rivers can act as geographic barriers for terrestrial species, possibly leading to speciation between opposite river margins. Complementarity is a key principle in systematic reserve planning and design (Margules & Pressey 2000) wherein this difference does not justify why watersheds are largely used as a conservation planning and management unit for freshwater and water resource conservation (Thieme et al. 2007), and yet rarely in terrestrial conservation. Moreover, natural ecosystems are not closed and static entities (Pickett et al. 1992), and it therefore makes little sense to consider aquatic and terrestrial environments separately during conservation planning. In fact, it is preeminent for the generation of effective actions to consider both of these coupled realms simultaneously during land/waterscape conservation planning (Thieme et al. 2007).

In addition to their passive defensibility (Peres & Terborgh 1995), other watershed characteristics also contribute to making them an exceptional choice for conservation planning unit. Current remote sensing and GIS technologies and widely available terrain data permit an accurate delimitation of watersheds, which can be used as boundaries for PAs. The modular cross-scale aspect of watersheds permit larger units to be subdivided into smaller units that can be more tractable for different conservation actions (Thieme et al. 2007), while maintaining the relationship with the larger scale. This relationship demonstrates the connectivity between watersheds, a useful trait in generating less fragmented solutions for PA networks (Linke et al. 2008). Maintaining connectivity between planning units and between habitats is a major objective in biodiversity conservation (Thieme et al. 2007), and safeguarding a whole river basin protects aquatic, semi-aquatic and terrestrial environments, as well as any lateral migration of organisms between them.

The abovementioned facts, plus their suitability for use in “coarse filter” conservation strategies (Hunter 1991), makes watersheds an ideal planning unit for a vast region such as Amazonia, which still lacks reliable information on biodiversity distribution (Hopkins 2007; Kress et al. 1998; Peres 2005). Furthermore, Amazonia remains to a large extent roadless, and rivers are both the primary means of transportation for hunting, fishing and other subsistence activities, and the major route for threat contagion and expansion, making this approach more relevant still.

1.4 Study region

In this PhD thesis, I conducted analyses at three different scales (Fig. 1.1). Chapter 2 was the only chapter not directly related to the watershed concept, focussing instead on the interfluvial region between the rivers Purus and Jutaí, an area larger than 40 million hectares. Chapters 3 and 4 examined the entire hydrological domain of Amazonia (almost 715 million hectares), comprising the neighbouring basins of the Amazon (= Solimões), Araguari and Tocantins rivers, extending from lowland forests at sea level to elevations of ~6,400 m in the tropical Andes and embracing seven of the nine Amazonian countries, namely Brazil, Bolivia, Peru, Ecuador, Colombia, Venezuela and Guyana. Finally, Chapter 5 focused on the Rio Tefé watershed, a roadless area spanning about 2.4 million hectares with regional economic importance for the Brazilian state of Amazonas.

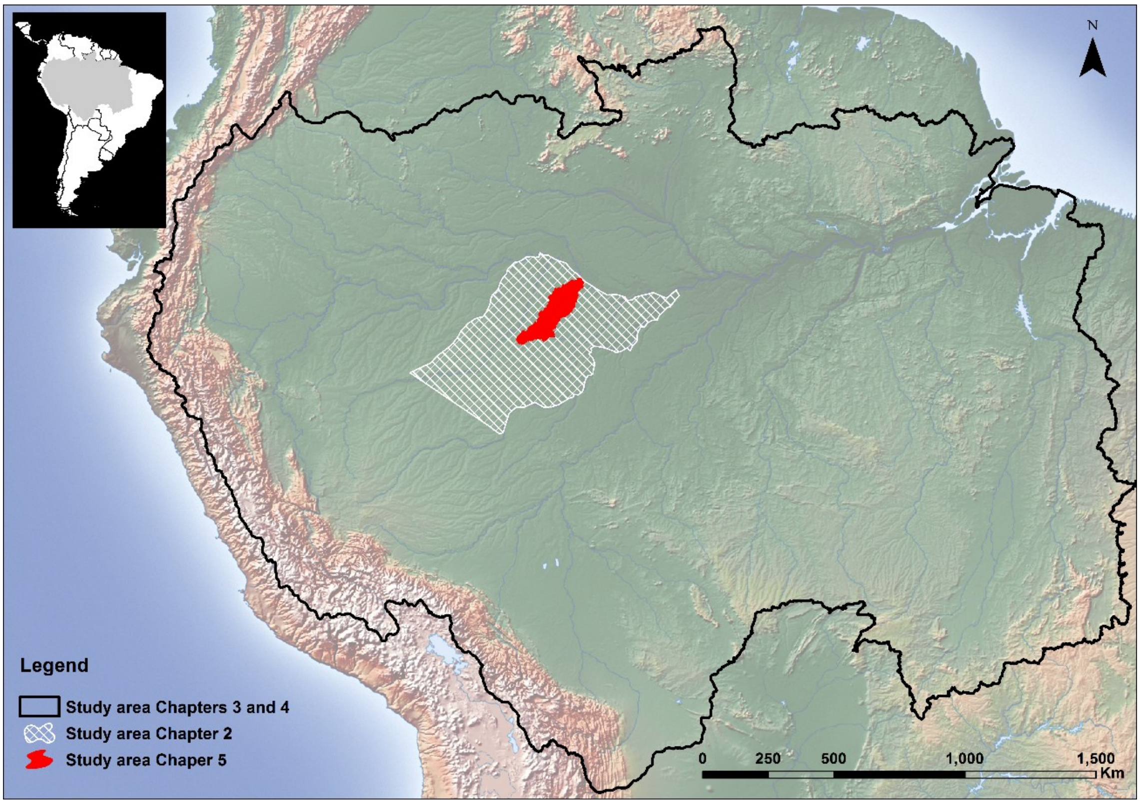

Fig 1.1. Location of the study area, including the nested scales of analysis for different thesis chapters. The black line represents the extent of the entire hydrological domain of Amazonia, comprising the study area for Chapters 3 and 4. The white crosshatches area encompasses more than 40 million of hectares between the Jutaí, Juruá and Purus rivers, and comprised the study area for Chapter 2. The red polygon represents the study area including the entire Tefé river watershed, which was addressed in Chapter 5. The inset map (top left) shows the extent of the Amazonian hydrological domain within South America.

1.5 Thesis structure

The chapters in this thesis are deliberately arranged to gradually show how relevant and advisable is the use of watersheds as conservation planning units in Amazonia. In addition to the main goal of the thesis, this document is also an assessment of how anthropogenic threats are distributed across the biome and the current degree of protection for all major Amazonian watersheds. A brief description of each chapter in provided below.

Chapter 1 introduces the thesis and contextualises the reader about the urgency of promoting coordinated actions of biodiversity conservation in Amazonia. Data from the RADAMBRASIL project (DNPM 1973-83) are widely used to estimate biomass in Amazonia (Brown & Lugo 1992; Fearnside 1997; Sales et al. 2007; Houghton et al. 2008; Peres et al. 2016) and Chapter 2 examines potential sampling biases associated with tree plots inventoried by this project. RADAMBRASIL is arguably the most comprehensive project ever conducted in Amazonia but demonstrating the limitations of its data helps to contextualise how data-poor the biome remains. In Chapter 3, I segmented the Amazonian hydrological domain into 23 major watersheds and assessed the major threats to the biome under this segmentation. This chapter also presents the justification for why Amazonian biodiversity conservation should be planned at the watershed scale. Chapter 4 analyses the existing mismatches between degree of protection and degree of vulnerability across the biome. Using the segmentation generated in Chapter 3, Chapter 4 explores the real drivers of recent PA siting throughout Amazonia and discusses the consequences of such *modus operandi*. Chapter 5 is a case study in which I seek to display how a planning programme based on watersheds and their subdivisions can generate practical solutions for biodiversity conservation in a data-poor tropical forest region. This chapter also continues the discussion started in Chapter 4 about PA siting and political convenience. Chapter 6 presents the concluding remarks, with some final conservation recommendations and suggestions of future steps for how the use of watersheds as conservation planning units can help build successful conservation polices in Amazonia.

The four data chapters (Chapter 2 to 5) are written as stand-alone manuscripts and will be published separately as peer-reviewed papers following further review incorporating comments from my examiners. Consequently, some small sections in the “Methods” and “Results” may present text redundancy in relation to other chapters. The bibliography references used in the text are also provided separately at the end of each chapter. By organizing the chapters as stand-alone manuscripts, I intend to make the reading easier and more fluid, allowing readers to access each chapter individually. All chapters will be submitted to peer-reviewed journals in appropriate time.

1.6 References

Alho, C.J.R., Reis, R.E. & Aquino, P.P.U., 2015. Amazonian freshwater habitats experiencing environmental and socioeconomic threats affecting subsistence fisheries. *Ambio*, 44(5), pp.412–425. Available at: http://www.ncbi.nlm.nih.gov/pubmed/25572836.

Alvarez-Berríos, N.L. & Mitchell Aide, T., 2015. Global demand for gold is another threat for tropical forests. *Environmental Research Letters*, 10(1), p.014006. Available at: http://stacks.iop.org/1748-9326/10/i=1/a=014006?key=crossref.6ccc347f82912ddd78f8fb396c95da22.

Assunção, J. & Rocha, R., 2014. *Getting Greener by Going Black : The Priority Municipalities in Brazil*, Rio de Janeiro, Brazil.

Barni, P.E. et al., 2015. Deforestation and Forest Fires in Roraima and Their Relationship with Phytoclimatic Regions in the Northern Brazilian Amazon. *Environmental Management*, 55(5), pp.1124–1138. Available at: http://link.springer.com/10.1007/s00267-015-0447-7.

Barthem, R.B., de Brito Ribeiro, M.C.L. & Petrere, M., 1991. Life strategies of some long-distance migratory catfish in relation to hydroelectric dams in the Amazon Basin. *Biological Conservation*, 55(3), pp.339–345. Available at: http://linkinghub.elsevier.com/retrieve/pii/000632079190037A.

Benchimol, M. & Peres, C.A., 2015. Widespread forest vertebrate extinctions induced by a mega hydroelectric dam in Lowland Amazonia M. K. Pandit, ed. *PLOS ONE*, 10(7), p.e0129818. Available at: http://dx.plos.org/10.1371/journal.pone.0129818.

Brown, S. & Lugo, A., 1992. Aboveground biomass estimates for tropical moist forests of the Brazilian Amazon. *Interciencia*, 1(17), pp.8–18.

Campos-Silva, J.V., da Fonseca Junior, S.F. & da Silva Peres, C.A., 2015. Policy reversals do not bode well for conservation in Brazilian Amazonia. *Natureza & Conservação*, 13(2), pp.193–195. Available at: http://linkinghub.elsevier.com/retrieve/pii/S1679007315000444.

DNPM, 1973. *Projeto RADAMBRASIL: Levantamento de Recursos Naturais*, Rio de Janeiro, Brazil.

Eva, H. et al., 2005. *a Proposal for Defining the Geographical*, Office for Official Publications of the European Communities.

Fearnside, P.M., 2003. Conservation Policy in Brazilian Amazonia: Understanding the Dilemmas. *World Development*, 31(5), pp.757–779. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0305750X03000111.

Fearnside, P.M., 2015. Environment: Deforestation soars in the Amazon. *Nature*, 521(7553), pp.423–423.

Fearnside, P.M., 1997. Greenhouse gases from deforestation in Brazilian Amazonia: net committed emissions. *Climatic Change*, 35(3), pp.321–360. Available at: http://link.springer.com/10.1023/A:1005336724350.

Fearnside, P.M., 1995. Hydroelectric Dams in the Brazilian Amazon as Sources of "Greenhouse" Gases. *Environmental Conservation*, 22(01), p.7. Available at: http://www.journals.cambridge.org/abstract_S0376892900034020.

Finer, M. & Jenkins, C.N., 2012. Proliferation of Hydroelectric Dams in the Andean Amazon and Implications for Andes-Amazon Connectivity M. Krkosek, ed. *PLoS ONE*, 7(4), p.e35126. Available at: http://dx.plos.org/10.1371/journal.pone.0035126.

Gaston, K.J. et al., 2008. The Ecological Performance of Protected Areas. *Annual Review of Ecology, Evolution, and Systematics*, 39(1), pp.93–113.

Gibbs, H.K. et al., 2015. Brazil's Soy Moratorium. *Science*, 347(6220), pp.377–378. Available at: http://www.sciencemag.org/content/347/6220/377.summary.

Goulding, M., Barthem, R. & Ferreira, E., 2003. *The Smithsonian atlas of the Amazon*,

Hoorn, C. et al., 2010. Amazonia Through Time: Andean Uplift, Climate Change, Landscape Evolution, and Biodiversity. *Science*, 330(6006), pp.927–931. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1194585.

Hopkins, M.J.G., 2007. Modelling the known and unknown plant biodiversity of the Amazon Basin. *Journal of Biogeography*, 34(8), pp.1400–1411. Available at: http://doi.wiley.com/10.1111/j.1365-2699.2007.01737.x [Accessed March 19, 2016].

Hunter Jr, M.., 1991. Coping with ignorance: the coarse-filter strategy for maintaining biodiversity. In *Balancing on the brink of extinction*. Washington: Island press, pp. 266–281.

Imbernon, J. & Branthomme, A., 2001. Characterization of landscape patterns of deforestation in tropical rain forests. *International Journal of Remote Sensing*, 22(9), pp.1753–1765. Available at: http://www.catchword.com/cgi-bin/cgi?body=linker&ini=xref&reqdoi=10.1080/014311601300176033.

Joppa, L. & Pfaff, A., 2010. Reassessing the forest impacts of protection. *Annals of the New York Academy of Sciences*, 1185(1), pp.135–149. Available at: http://doi.wiley.com/10.1111/j.1749-6632.2009.05162.x.

Joppa, L.N. & Pfaff, A., 2009. High and Far: Biases in the Location of Protected Areas J. Moen, ed. *PLoS ONE*, 4(12), p.e8273. Available at: http://dx.plos.org/10.1371/journal.pone.0008273.

Junk, W.J. et al., 2010. The Role of Floodplain Forests in an Integrated Sustainable Management Concept of the Natural Resources of the Central Amazonian Várzea. In *Amazonian Floodplain Forests*. Springer Netherlands, pp. 485–509. Available at: http://link.springer.com/10.1007/978-90-481-8725-6_23 [Accessed March 19, 2016].

Junk, W.J. & Soares, M.G.M., 2001. Freshwater Fish Habitats in Amazonia: State of Knowledge, Management, and Protection. *Aquatic Ecosystem Health & Management*, 4(4), pp.437–451. Available at: http://www.tandfonline.com/doi/abs/10.1080/146349801317276107 [Accessed March 19, 2016].

Knight, A.T. et al., 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology*, 22(3), pp.610–617.

Kress, W. et al., 1998. Amazonian biodiversity: assessing conservation priorities with taxonomic data. *Biodiversity & Conservation*, 7(12), pp.1577–1587.

Laurance, W. et al., 2001. The future of the Brazilian Amazon. *Science (Washington)*, 291(5503), pp.438–439.

Lausche, B.J. & Burhenne-Guilmin, F., 2011. *Guidelines for protected areas legislation*, IUCN.

Lees, A.C. et al., 2016. Hydropower and the future of Amazonian biodiversity. *Biodiversity and Conservation*, 25(3), pp.451–466. Available at: http://link.springer.com/10.1007/s10531-016-1072-3.

Linke, S., Norris, R.H. & Pressey, R.L., 2008. Irreplaceability of river networks: towards catchment-based conservation planning. *Journal of Applied Ecology*, 45(5), pp.1486–1495. Available at: http://doi.wiley.com/10.1111/j.1365-2664.2008.01520.x.

Lourival, R. et al., 2008. Getting fourteen for the price of one! Understanding the factors that influence land value and how they affect biodiversity conservation in central Brazil. *Ecological Economics*, 67(1), pp.20–31. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0921800908001547.

Margules, C.R. & Pressey, R.L., 2000. Systematic conservation planning. *Nature*, 405(6783), pp.243–253. Available at: http://www.nature.com/doifinder/10.1038/35012251.

de Marques, A.A.B. & Peres, C.A., 2015. Pervasive legal threats to protected areas in Brazil. *Oryx*, 49(01), pp.25–29. Available at: http://www.journals.cambridge.org/abstract_S0030605314000726 [Accessed March 19, 2016].

Mendes-Oliveira, A.C. et al., 2013. Efeitos antrópicos sobre comunidades de pequenos mamíferos não-voadores na amazônia brasileira. In A. C. Mendes-Oliveira & C. Miranda, eds. *Os Pequenos Mamíferos Não Voadores da Amazônia Brasileira*. pp. 257–274.

Moser, P. et al., 2014. Tree Species Distribution along Environmental Gradients in an Area Affected by a Hydroelectric Dam in Southern Amazonia. *Biotropica*, 46(3), pp.367–376. Available at: http://doi.wiley.com/10.1111/btp.12111.

Nepstad, D. et al., 2014. Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, 344(6188), pp.1118–1123. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1248525.

Nolte, C. et al., 2013. Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon. *Proceedings of the National Academy of Sciences*, 110(13), pp.4956–4961. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.1214786110\nhttp://www.ncbi.nlm.nih.gov/pubmed/23479648.

Nores, M., 1999. An alternative hypothesis for the origin of Amazonian bird diversity. *J Biogeography*, 26(3), pp.475–485. Available at: http://www.blackwell-synergy.com/links/doi/10.1046/j.1365-2699.1999.t01-1-00311.x/abs.

O’Rourke, D. & Connolly, S., 2003. Just oil? The distribution of environmental and social impacts of oil production and consumption. *Annual Review of Environment and Resources*, 28(1), pp.587–617.

Oliveira, P.J.C. et al., 2007. Land-Use Allocation Protects the Peruvian Amazon. *Science*, 317(5842), pp.1233–1236. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1146324.

Parolin, P., 2000. Growth, productivity and use of trees in white-water floodplains. In W. J. Junk et al., eds. *The Central Amazon Floodplain: Actual Use and Options for a Sustainable Management*. Leiden: Backhuys Publisher, pp. 375–391.

Peres, C. A, Barlow, J. & Laurance, W.F., 2006. Detecting anthropogenic disturbance in tropical forests. *Trends in ecology & evolution (Personal edition)*, 21(5), pp.227–9. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16697907.

Peres, C. A., 2005. Why We Need Megareserves in Amazonia. *Conservation Biology*, 19(3), pp.728–733. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00691.x.

Peres, C. A & Terborgh, J.W., 1995. Amazonian Nature Reserves: An Analysis of the Defensibility Status of Existing Conservation Units and Design Criteria for the Future. *Conservation Biology*, 9(1), pp.34–46. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1995.09010034.x.

Peres, C.A. et al., 2016. Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *Proceedings of the National Academy of Sciences*, 113(4), pp.892–897. Available at: http://www.pnas.org/lookup/doi/10.1073/pnas.1516525113 [Accessed March 21, 2016].

Pfaff, A. et al., 2015. Protected Areas' Impacts on Brazilian Amazon Deforestation: Examining Conservation – Development Interactions to Inform Planning K. Bawa, ed. *PLOS ONE*, 10(7), p.e0129460. Available at: http://dx.plos.org/10.1371/journal.pone.0129460.

Pickett, S.T., Parker, V.T. & Fiedler, P.L., 1992. The new paradigm in ecology: implications for conservation biology above the species level. In *Conservation biology*. Springer US, pp. 65–78.

Portela, R. & Rademacher, I., 2001. A dynamic model of patterns of deforestation and their effect on the ability of the Brazilian Amazonia to provide ecosystem services. *Ecological Modelling*, 143(1-2), pp.115–146. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0304380001003593.

de Queiroz, H.L., 2010. Protected Areas in the Amazonian Várzea and their Role in its Conservation: The Case of Mamirauá Sustainable Development

Reserve (MSDR). In *Amazonian Floodplain Forests*. Springer Netherlands, pp. 465–483. Available at: http://link.springer.com/10.1007/978-90-481-8725-6_22 [Accessed March 19, 2016].

Richards, P.D. et al., 2012. Exchange rates, soybean supply response, and deforestation in South America. *Global Environmental Change*, 22(2), pp.454–462. Available at: http://dx.doi.org/10.1016/j.gloenvcha.2012.01.004.

Rylands, A.B. & Brandon, K., 2005. Brazilian Protected Areas. *Conservation Biology*, 19(3), pp.612–618. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00711.x.

Sales, M.H. et al., 2007. Improving spatial distribution estimation of forest biomass with geostatistics: A case study for Rondônia, Brazil. *Ecological Modelling*, 205(1), pp.221–230.

Schulman, L. et al., 2007. Amazonian biodiversity and protected areas: do they meet? *Biodiversity and Conservation*, 16(11), pp.3011–3051. Available at: http://link.springer.com/10.1007/s10531-007-9158-6.

Da Silva, J.M.C., Rylands, A.B. & da FONSECA, G. a. B., 2005. The Fate of the Amazonian Areas of Endemism. *Conservation Biology*, 19(3), pp.689–694. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00705.x.

Soares-Filho, B. et al., 2010. Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences*, 107(24), pp.10821–10826. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.0913048107.

Soares-Filho, B.S. et al., 2006. Modelling conservation in the Amazon basin. *Nature*, 440(7083), pp.520–523. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16554817.

Swenson, J.J. et al., 2011. Gold Mining in the Peruvian Amazon: Global Prices, Deforestation, and Mercury Imports G. J.-P. Schumann, ed. *PLoS ONE*, 6(4), p.e18875. Available at: http://dx.plos.org/10.1371/journal.pone.0018875.

Thieme, M. et al., 2007. Freshwater conservation planning in data-poor areas: An example from a remote Amazonian basin (Madre de Dios River, Peru and Bolivia). *Biological Conservation*, 135(4), pp.484–501. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0006320706004733.

Tollefson, J., 2015a. Political appointments spur concerns for Amazon. *Nature*, 517(7534), pp.251–252. Available at: http://www.nature.com/doifinder/10.1038/517251a.

Tollefson, J., 2015b. Stopping deforestation: Battle for the Amazon. *Tollefson, J. (2015). Stopping deforestation: Battle for the Amazon. Nature, 520(7545), 20–23. http://doi.org/10.1038/520020aNature*, 520(7545), pp.20–23. Available at: http://eutils.ncbi.nlm.nih.gov/entrez/eutils/elink.fcgi?dbfrom=pubmed&id=25832388&retmode=ref&cmd=prlinks\npapers2://publication/doi/10.1038/520020a.

Wishart, M.J. & Davies, B.R., 2003. Beyond catchment considerations in the conservation of lotic biodiversity. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 13(5), pp.429–437. Available at: http://doi.wiley.com/10.1002/aqc.600.

Wittmann, F., Junk, W.J. & Piedade, M.T.., 2004. The várzea forests in Amazonia: flooding and the highly dynamic geomorphology interact with natural forest succession. *Forest Ecology and Management*, 196(2), pp.199–212.

Yapa, P.D. & Tao Shen, H., 1994. Modelling river oil spills: a review. *Journal of Hydraulic Research*, 32(5), pp.765–782. Available at: http://www.tandfonline.com/doi/abs/10.1080/00221689409498713.

Chapter 2

Landscape-scale plot selection affects above ground biomass estimates in Amazonian forests

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2.1 Introduction

Past emissions of carbon dioxide (CO_2) have changed Earth’s climate (IPCC 2015) and the prognostic of additional greenhouse-gas (GHG) emissions does not bode well for years to come (Friedlingstein et al. 2014). Developing countries are expected to account for most increases in future GHG emissions (IEA 2009). Ensuring that developing nations can reduce their emissions remains of the most intractable challenges of our time (Wara 2007).

The Kyoto Protocol established the first global carbon global market; by selling carbon credits on international markets, it aimed to reduce global GHG emissions through monetary incentives aimed at developing countries (Gibbs et al. 2007). REDD+ (Reducing Emissions from Deforestation and Forest Degradation) initiatives were subsequently proposed as a tool to generate large, inexpensive and rapid reductions in GHG emissions in developing countries by involving national governments, farmers, companies, and forest landowners (Angelsen 2009). One of the most critical requirements in developing policies for REDD+ mechanisms includes measuring the total carbon stock of a given area (Gibbs et al. 2007; Angelsen 2008; Réjou-Méchain et al. 2014). However, debate has ensued over how best to measure forest carbon, the basis on which carbon can be valued and traded (Gibbs et al. 2007; Goetz & Dubayah 2011; Baraloto et al. 2013).

The potential to implement effective strategies to mitigate carbon emissions is limited by our ability to accurately estimate forest carbon stocks (Malhi et al. 2006; Réjou-Méchain et al. 2014), in particular its temporal dynamics (Chave et al. 2003; Chave et al. 2008; Muller-Landau et al. 2014). Tropical forests are critical for climate change mitigation efforts because they contain the highest carbon density of any forest type (Pan et al. 2011), storing nearly half of the carbon found in all terrestrial flora worldwide (Houghton 2005). High deforestation rates in tropical forests (Achard et al. 2002; Geist & Lambin 2002) exacerbate the urgent need to effectively assess these carbon stocks (Saatchi et al. 2011). About half of world's remaining tropical forests are in South America (FAO 2001), mostly within Amazonia (Hansen et al. 2013), highlighting the importance of the region for global scale forest carbon policies. However, regional assessments of carbon stocks conducted to date in Amazonia lack precision, either because of insufficient sampling, due to poor spatial distribution of available forest inventories, or the inherent uncertainty associated with these inventories (Houghton 2005).

Floristic plots, permanent or otherwise, where aboveground biomass (AGB) is estimated through allometric equations, is the most common method used to quantify carbon stocks (Clark et al. 2001; FAO 2007). The ideal size and shape of these plots is still matter of discussion (Chave et al. 2003; Wagner et al. 2010; Baraloto et al. 2013; Holdaway et al. 2014) with plots of 1 ha being most widely used (Baraloto et al. 2013), following recommendations from the United Nations Food and Agriculture Organization (FAO 1981).

The largest data source of Amazonian forest inventories is the RADAMBRASIL Project (DNPM 1973-1983), which was ambitiously designed to map the forest and mineral resources of the entire Brazilian Amazon (Brown & Lugo 1992; Fearnside 1997; Sales et al. 2007). The vast majority of forest inventories, which consisted of 2,795 widely distributed plots spread across the entire Brazilian Amazon, were carried out using 1-ha tree plots. Subsequently, the project was renamed as RADAMBRASIL and expanded to the whole of Brazil. However, despite the broad spatial distribution of tree plots, these data may not be ideal to estimate forest AGB because small vegetation plots may lead to biases that systematically inflate AGB estimates. This may come about through an inherent tendency to select portions of the forest containing particularly large trees (Chave et al. 2003), which has been referred to as the "majestic forest" effect

(Phillips et al. 2004; Malhi et al. 2006; Maniatis et al. 2011), or a systematic tendency to avoid tree-fall gaps (Chave et al. 2004).

Here, I evaluate the inferential adequacy of data derived from one-hectare forest plots to generate forest AGB estimates for two major interfluvial regions of Central-Western Brazilian Amazonia. To do so, I compare data obtained by the RADAMBRASIL program with those from more recent forest inventories commissioned by a private oil and gas company, which were conducted within the same geographic region using variable-sized tree plots of up to 9.7 ha. I also discuss issues of scale in tropical forest inventories, particularly the sampling caveats in estimating tropical forest AGB using small-sized tree plots.

2.2 Methods

2.2.1 Forest inventories

This study used 321 forest assessments carried out south of the Amazon river within the interfluvial region between the Purus, Juruá and Jutaí rivers (Purus – Juruá, Juruá- Jutaí), encompassing a combined area of ~40 million hectares (Fig 2.1). Our data comes from two datasets: the RADAMBRASIL project (DNPM 1973-1983), which sampled this part of the Amazon between 1973 and 1983, and more recent forest inventories (2006-2011) conducted by a professional team of botanists, parabotanists and local field assistants commissioned by an oil company (HRT Oil & Gas). The latter floristic surveys were part of a wider oil and gas drilling and prospecting operation, as required by law to comply with baseline environmental licensing standards sanctioned by the State of Amazonas, Brazil.

We used information from 272 one-hectare plots inventoried by RADAMBRASIL (hereafter, RB data). This dataset is based on forest inventories carried out within plots of 500m x 20m, which were laid out and sampled by botanists and para-botanists hired by the Brazilian government. All trees ≥100 cm cbh (circumference at breast height) or ≥31.8 cm dbh (diameter at breast height) other than arborescent palms were measured and reliably identified to the level of genus. The HRT Oil and Gas forest inventories (hereafter, HRT data) were carried out in 49 forest plots ranging from 1.6 to 9.7 ha, 96% of which were over 3 ha (median = 7.56 ha, mean = 7.15 ha, sd= 1.81) by BRASA (BRA Soluções Ambientais), a specialized environmental consultancy enterprise. All trees ≥15 cm DBH were measured and identified to the highest possible level of taxonomic resolution and at least to the level of genus. Data were standardized to allow for any comparative analyses; therefore, only trees from the HRT dataset equal or above the minimum cut-off size used in the RB dataset (31.8 cm DBH) were used in the analyses. The spatial distribution of all floristic plots is shown in Figure 2.1.

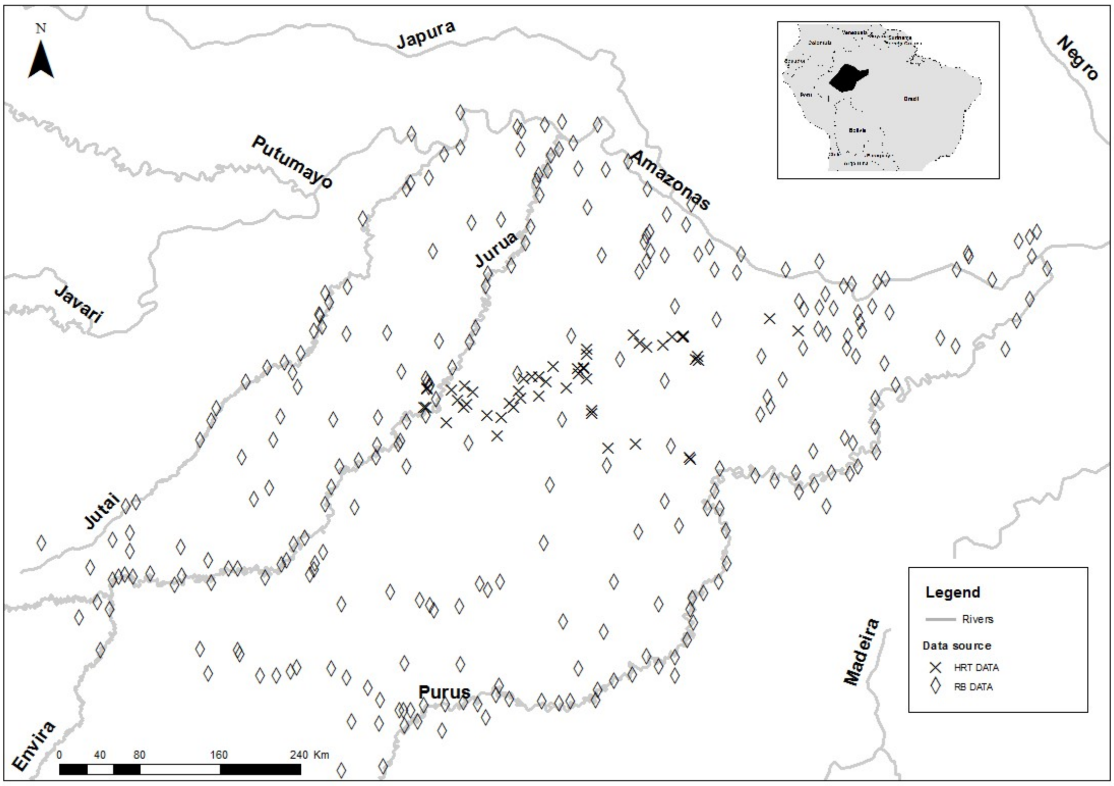

Fig. 2.1 – Study region comprising areas between the Jutaí, Juruá and Purus rivers. The area comprises more than 40 million hectares and shows forest inventory sites. Xs indicate plots sampled by the HRT Oil & Gas company, whereas diamonds indicate those sampled by the RADAMBRASIL Project. Grey lines represent major Amazonian rivers. Inset map (top right) shows the position and extent of the study region within South America.

2.2.2 Above Ground Biomass measurement

The AGB of each tree was calculated following the allometric equation in Chave et al. 2005:

$$AGB = p \times \exp(0.0634 - 1.562 + (2.148 \times \log(DBH)) + (0.207 \times \log(DBH))^2 + (-0.0281 \times \log(DBH))^3)$$

Where: DBH = Diameter at breast high in cm
p = wood specific gravity (WSG) in g/cm^3.

Data describing tree heights were unavailable, and we assumed stems were circular at the point of measurement. We used tree WSG values at the most specific taxonomic resolution possible, which in 99.9% of cases was either genus or species. For trees classified as "unidentified", we attributed mean WSG values in their plot. We prioritised species-specific WSG values found in the literature and also took into account the geographic proximity of studies that quantified WSG in relation to our study area. WSG values were collected from peer-reviewed literature (Wittmann et al. 2006; Nogueira et al. 2008), unpublished dissertations, as well as the Global Wood Database and from a landscape scale Amazonian study in which wood density measurements were extracted from samples of >300 trees species (Jari Florestal 2013).

2.2.3 Subsampling tree plots

The HRT data were obtained from plots of different sizes compared to those of RB. Before running analyses therefore we standardised the data by bootstrapping, with no replacement, the trees sampled in each HRT plot (31.8 cm minimum DBH) 999 times selecting 60 trees (which corresponds to the mean number of trees in the RB plots) from each of the HRT plots.

2.2.4 Assessing differences between datasets

We evaluated differences between the RB and HRT datasets by considering descriptors of forest structure (stand density, basal area, AGB), tree size distribution using DBH, and the distance between inventories and the nearest major perennial stream or river.

RB and HRT were compared in terms of their total AGB ha^{-1}, stem density (per ha), and basal area density (m^2/ha) through one-way Anovas. In these analyses, we used the raw data (without bootstrapped subsamples) from each forest plot and the value of each variable was defined as the sum of the values inside each plots divided by the plot area.

We compared the size structure of trees between the two datasets using the cumulative DBH distribution of all trees occurring within their respective plots. We also subsequently calculated the area under the curve (AUC) for each cumulative distribution function and cross-referencing their values with the plot basal area to provide an insight on the effect of smaller trees on the cumulative DBH distribution. To ensure that AUC estimates were representative of the shape of the curve rather than the range of DBH values, tree DBH was standardised using the maximum DBH value encountered in each plot.

In order to understand which size classes exert the strongest effect on the tree DBH distribution, we grouped trees by ranked DBH class (31.8-39.9, 40-49.9, 50-59.9, 60-69.9, 70-79.9, 80-89.9, 90-99.9, >100 cm). Because RB and HRT plots differed greatly in their size variance and sample sizes, comparison between datasets were carried out using Welch's *t*-tests across all possible pairwise combinations of DBH classes. We also compared the AGB of each DBH class across the datasets.

Tree plots in both datasets were concentrated in areas near stream or river channels (Fig. 2.1). Because spatially structured distributions of sample units can generate regional biased estimates (Fisher et al. 2008), we examined whether RB and HRT are similarly distributed in relation to fluvial landmarks. Euclidian distances to rivers were calculated for only the largest streams and rivers mapped within the entire study region, and were measured from the geometric centroid of each plot. Results were compared between datasets using a t-test

2.2.5 Effect of plots size on AGB estimates

To assess the effect of plot size on AGB values we modelled the aggregate AGB estimate for each plots in both databases (RB or HRT) alongside climatic, geological and physical variables with known effects on AGB. We used climatic variables such as Isothermality and the E index created by Chave et al. (2014). Isothermality (Hijmans et al. 2004) (downloaded from: http://www.worldclim.org) is a product of mean diurnal temperature range and the annual temperature range was multiplied by 100. The E index reflects environmental stress and is calculated as following:

$$E=(0,178*TS-0.938*CWD-6.61*PS)/1000$$

Where: TS = Temperature seasonality (Worldclim dataset)
CWD = Long-term climatic water deficit (Chave et al. 2014)
PS = Precipitation seasonality (Worldclim dataset)

The other variables present in the model were: Elevation, Vegetation type, Soil type, Soil fertility and Drainage density. A Vegetation-type map was created by the Brazilian National Institute for Space Research (INPE) based on information from the RADAMBRASIL project posteriorly updated by the Brazilian government. We calculated Elevation based on images with a spatial resolution of 3 arc-seconds (~1 km) from the Shuttle Radar Topography Mission (SRTM). Soil type and a measure of soil fertility (see Schneider & Peres 2015). Drainage density was created by the AMBDATA project (http://www.dpi.inpe.br/Ambdata) and was based on a spatial interpolation (Kernel) of drainage networks obtained from the HydroSHEDS project (Lehner et al. 2008). Drainage density, Soil type and Isothermality were downloaded from the AMBDATA project website (available at: http://www.dpi.inpe.br/Ambdata/download.php). Temperature range (Index E) was extracted from a 2.5 arc-minute (~5km) resolution map while all other data were extracted from maps of 30 arc-seconds (~1 km).

Accounting for the variability in the size and shape of tree plots in the RB and HRT datasets, we calculated the centroid of known plot locations and created 500-m buffers around each plot. Average values for Isothermality, E index, Elevation, Soil Fertility index and Drainage network within those buffers were used in the models, whereas we used the most dominant class of Vegetation type and Soil type within those buffers to define these variables.

Moran’s Test I confirmed the expected autocorrelation for both datasets, but it was possible to define the type of spatial dependence present in each dataset and select the most adequate model through the Lagrange Multiplier Test Diagnostics for Spatial Dependence (Anselin 1988). We used a Mixed Regressive-Spatial Autoregressive Model (MRSAR) that ensures that the spatial dependence (rho) is incorporated into the model by removing its effects on the environmental variables (Anselin & Rey 1991).

We determined the importance of each explanatory variables describing AGB by determining their frequency of occurrence in the models on the basis of a Δ AIC ≤ 3 likelihood threshold (equally likely models). All analyses and plots were carried out using R 3.1.3 software (R Core Team 2015). Maps and GIS processing were done using the ARCGIS 10.1 software (ESRI 2011).

2.3 Results

In total, our database consists of 38,736 trees, including 16,221 trees from RB and 22,515 trees from the HRT dataset. Approximately 0.01% (441) of those trees could not be identified, whereas all other trees were identified to at least the genus level. Collectively, these large canopy stems represent 264 tree genera. The minimum convex polygon circumscribed by all surveyed plots measure 40,338,172 hectares and most of this area overlaps a subregion of central-western Brazilian Amazonia that is considered severely data-deficient for plant inventories (Hopkins 2007).

2.3.1 Differences between 1-ha and larger plots

The total AGB calculated across all plots was 110,313 Mg, with a mean of 198 Mg per hectare. These plots are distributed over a region of over 40 million hectares which reflect the differences found between the highest (483 Mg) and lowest (66 Mg) AGB values across all 321 plots examined here. Plots larger than 1 hectare (in the HRT data) exhibited lower AGB and basal area values per hectare ($p<0.001$), even though they had a higher density of stems ($p= 0.002$) (Fig. 2.2). The fact that 1-ha plots present higher AGB values despite containing a sparser set of tree density supports the expected trend of higher per stem AGB in the RD dataset, especially considering that the average wood density (WSG) was also higher in plots larger than 1 ha (0.69 g/cm^3 against 0.64 g/cm^3). This indicates that RB plots on average had larger trees than HRT plots. Large-girthed trees are scarce in tropical forest and therefore larger plots should have, by chance, a higher probability of capturing large trees than 1-ha plots, suggesting a consistent bias in RB data towards large-girthed trees.

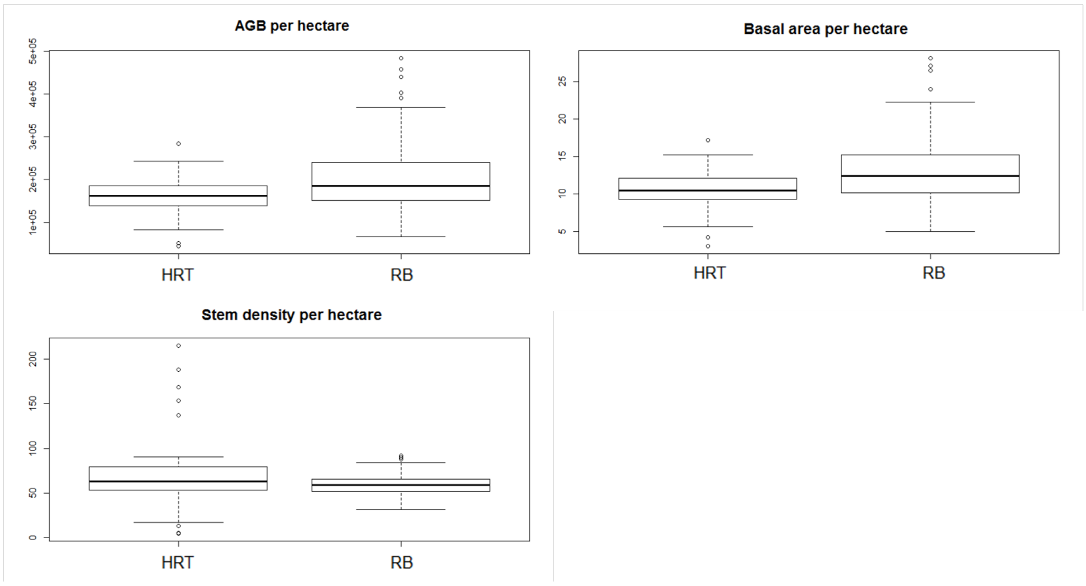

Fig. 2.2 – Differences between key forest attributes estimated by either 1-ha tree plots (RB dataset) or plots of 1.6 - 9.7 ha (HRT dataset). All comparisons yielded significant differences (ANOVAs, $P<0.01$) in which AGB and basal area density estimates were higher in 1-ha plots despite their lower stem densities.

Difference in the size distribution of trees sampled in plots within each dataset are also confirmed in their cumulative DBH distribution (Fig. 2.3). The line of best fit for HRT plots had a steeper slope than that of RB plots, showing a lower prevalence of large trees in the former. The relatively lower importance of large-girthed trees in larger plots compared to 1-ha plots are consistent with the fact that large trees (≥60 cm DBH) always account for the last size distribution quartile in the HRT dataset while it is possible to find trees ≥80 cm DBH in the RB dataset (Fig. 2.3).

Conversely, relatively small trees (<40 cm DBH) in the HRT dataset are far more prevalent than in the RB dataset (Fig. 2.4). The AUC of DBH cumulative distribution is a powerful representation of the tree size distribution within any given plot. AUC values near 1 indicate that smaller trees are more important in explaining the overall DBH variance within each plot. The pattern encountered shows that, in comparison with RB, the HRT dataset had much higher AUC values, which also rised faster as a function of plot scale basal area density, indicating that basal area increments in larger tree plots were more to be due to a larger proportion of smaller trees.

We can also assess differences in tree size distributions between the datasets (HRT and RB) by segmenting them into size classes (Fig. 2.5). The low size variance in the HRT plots is due to the bootstrapped subsampling, but there were significant differences in tree DBH between 1-ha and larger plots for almost all size classes (Welch's t-test, Table 2.1), with larger plots containing more trees in only the smallest DBH class. This again suggests that RB plots were associated with areas containing large-girthed trees and perhaps avoided large clusters of smaller trees.

The biased selection of trees also changes the contribution of each DBH size class to the AGB values encountered in the plots (Fig. 2.6). With the exception of the largest class, which accounts for a wider DBH range than the other classes, the impact of DBH size on total AGB decreases towards larger trees in HRT plots. Conversely, RB plots do not present a clear pattern in this regard (Table 2.2). However, the contribution of very large trees to AGB estimates was much larger in 1-ha plots than in larger plots. In the HRT data, less than 10% of the total AGB was contributed by trees >90 cm DBH, and >50% of AGB was contained in trees smaller than 50 cm DBH, compared to ~20% and 30%, respectively, in the RB dataset (Table 2.2). Trees defined as very large (over >70cm DBH) (Clark & Clark 1996) accounted for >40% of the biomass in the RB plots but for only ~22% of the biomass in HRT plots.

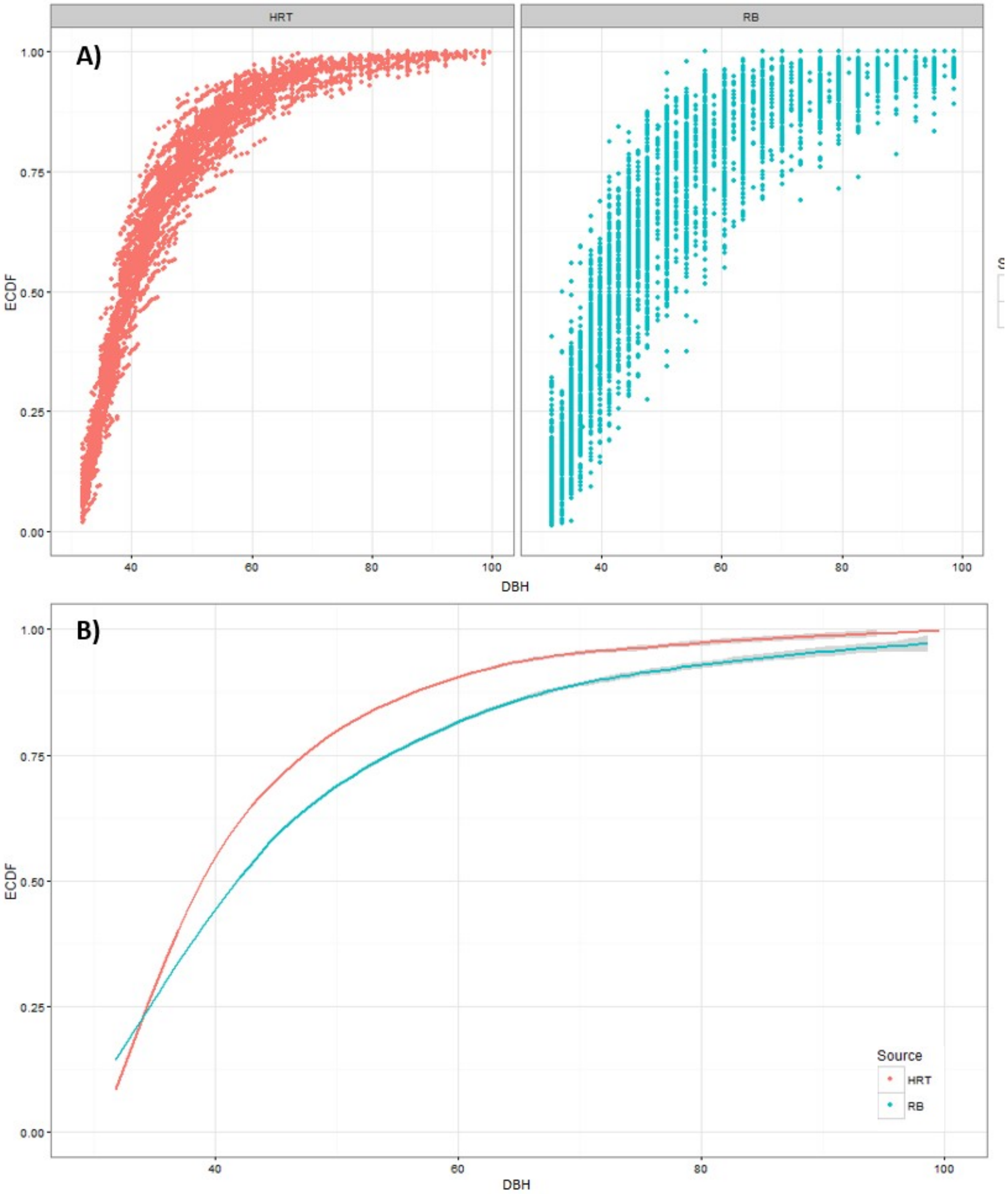

Fig. 2.3 – Cumulative size (DBH) distribution of trees in HRT and RB plots . A) Data collected in 1-ha plots are depicted in red (RB) and data collected in plots between 1.6 - 9.7 hectares (HRT) are in blue. Each dot represents the ECDF generated for a tree within a plot;B) Lines of best fit and 95% confidence intervals for the tree size distributions found in each dataset. The HRT curve is steeper than the RB curve, indicating the higher prevalence of very large canopy trees in the RB plots.

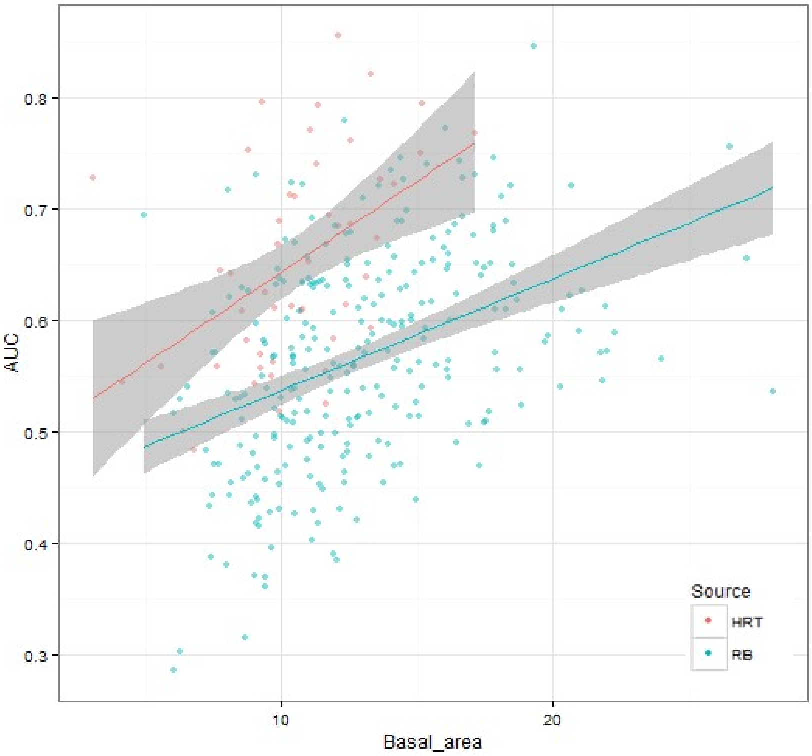

Fig. 2.4 – Relationship between the AUC of tree DBH cumulative distribution functions and basal area for 1-ha and 1.6 - 9.7 ha plots. AUC values near 1 indicate higher prevalence of small-girthed trees to any given plot. Larger plots (red dots) had higher AUC values and a steeper slope in relation to basal area density compared to 1-ha plots (blue dots). Lines of best fit are colour-coded and shaded areas indicate 95% confidence interval regions. There was a significant difference in slopes between AUC and forest basal area across the two datasets.

Table 2.1 – Welch's t-tests comparing number of trees by DBH classes encountered in 1-ha and 1.6 - 9.7 ha plots.

Class	**Mean of RB**	**Mean of HRT**	**delta mean**	**T**	**p**
31.8 - 39.9	23.16176	31.91527	-8.75351	-11.94	< 2.2e-16
40-49.9	15.79044	15.47686	0.31358	0.7013	0.4841
50-59.9	8.775735	6.770082	2.005653	5.6029	1.31E-07
60-69.9	5.113971	3.054163	2.059808	8.5772	2.43E-15
70-79.9	3.176471	1.32698	1.849491	10.9053	< 2.2e-16
80-89.9	1.286765	0.739551	0.547214	5.641	5.28E-08
90-99.9	0.8786765	0.3463265	0.53235	5.8799	1.41E-08
>100	1.4522059	0.3707755	1.0814304	8.8413	< 2.2e-16

Distance to major rivers from the plot centroids were not significantly different between the datasets (*p= 0.2634*). However, HRT plots were consistently placed farther from rivers (Table 2.3).

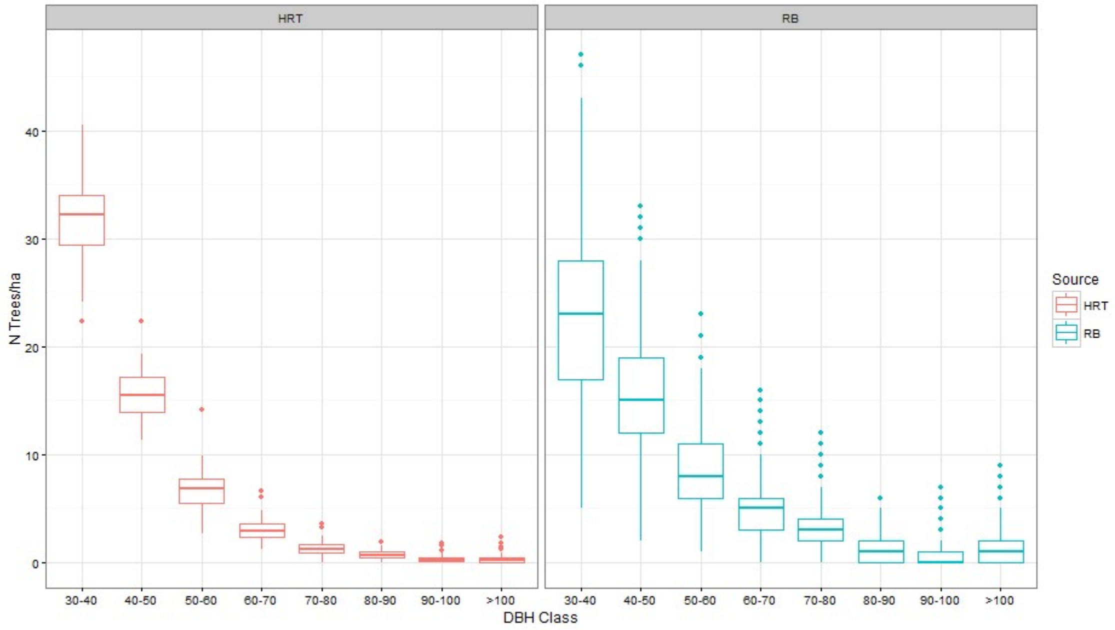

Fig. 2.5 – Tree distribution by DBH size classes for 1-ha plots (RB) and larger plots (HRT). Class intervals on the X axis are expressed in DBH (cm). HRT plots were larger areas than RB plots so values presented for HRT plots are the mean values across 999 bootstraps (without replacement), where 60 trees (mean number of trees in RB plots) were subsampled.

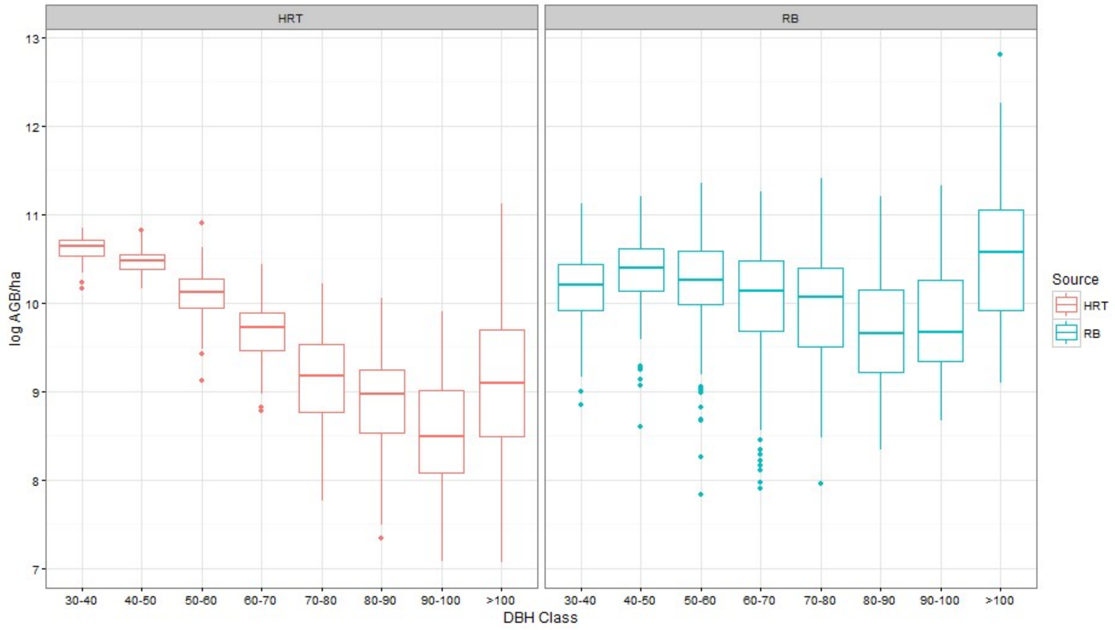

Fig. 2.6 – Total AGB in Mg (expressed as log10 per hectare) for 1-ha plots (RB dataset) and 1.6 - 9.7 ha plots (HRT dataset). The results are presented by DBH classes measured in cm. HRT plots (red boxplots) show a clear decline in the contribution of larger trees to total AGB, with the exception of the highest DBH class that includes a wider size range than the other classes. One-hectare plots (blue boxplots) exhibit a more evenly distributed AGB contribution through the classes. Note the marked difference across the two datasets in the importance of the smallest trees to the overall AGB.

Table 2.2 – Distribution of total AGB in DBH classes (in cm) across forest inventories sampled in one-hectare plots (RB) and plots between 1.6 - 9.7 hectares (HRT). Both the stand-alone contribution of each DBH class to the dataset total AGB and cumulative value of AGB contained in the DBH size classes (summed from the largest class to the smallest one) are presented here.

Class (cm)	**Biomass (% total)**		**Cumulative Biomass (% total)**	
	RB	**HRT**	**RB**	**HRT**
31.8-39.9	14.03	26.83	100	100
40-49.9	17.03	23.32	85.97	73.17
50-59.9	15.39	16.75	68.94	49.85
60-69.9	13.30	10.95	53.55	33.1
70-79.9	11.84	6.76	40.25	22.15
80-89.9	6.54	5.40	28.41	15.39
90-99.9	5.55	3.08	21.87	9.99
>100	16.32	6.91	16.32	6.91

Table 2.3 – Distance (km) of 321 forest inventories sampled in Central Amazonia to the nearest major river. Distances were calculated from plot centroids to the nearest river/stream bank.

Parameter	**Source**	
	RB	**HRT**
Median	2.54	3.35
Average	4.02	5.13
sd	4.58	4.67
90 perc	10.00	12.21
10 perc	1.02	1.56

2.3.2 AGB responses to plot size and environmental variables

The data source (β = .133, *p* = 0.008) in addition to Vegetation Type (β =0.604, *p* = 0.040 ; β =0.784, *p* = .008; β = 1.092, *p* = 0.003), Elevation (β = .001, *p* = 0.030) and Isothermality (β = 0.020, *p* = 0.011) constitute the four major predictors of AGB across all 321 plots (Table 2.4). Rho had a positive effect and was also highly significant in our model, and reflects the inherent spatial dependence of our data sample, and measures the average influence on observations by their neighbouring observations. There were a total of seven plausible models (Δ AIC ≤ 3) and the only variable that does not appear in any of them is Soil Type.

Isothermality is a metric that reflects temperature stability throughout the year. It is a product of the day-to-night temperature oscillation and the temperature oscillation across months where higher values indicate more constant month-to-month temperatures. The positive relationship presented by Isothermality indicates that AGB is higher in areas where temperatures are more constant throughout the year. The model also shows a positive relationship between AGB and some vegetation types, elevation and the RB (1-ha plots) dataset.

Table 2.4 – Summary of best ranked Mixed Regressive-Spatial Autoregressive Models (MRSARs) of AGB in 321 forest plots sampled in Central-Western Amazonia.

Parameter	**Estimate**	**Standard error**	**95% Confidence interval**	**Relative importance**
rho**	.142	.056	.032, .252	
(Intercept)***	7.965	1.022	5.962,9.968	
Elevation*	.001	.001	.000,.003	.89
E	1.066	.652	-.212,2.343	.50
Isothermality*	.020	.008	.005,.036	1.00
Source**	.133	.050	.035,.232	1.00
Veg_Type1*	.604	.295	.027,1.181	1.00
Veg_Type2	.507	.301	-.082,1.097	
Veg_Type3**	.784	.295	.206,1.362	
Veg_Type4	.442	.336	-.216,1.100	
Veg_type5	.560	.293	-.015,1.134	
Veg_Type6**	1.092	.363	.380,1.804	
Veg_Type7	.573	.304	-.023,1.168	
Fertility	.008	.018	-.028,.043	.19
Drainage density	.001	.003	-.005,.006	.18

Significance codes: *** .001; ** .01; * .05

2.4 Discussion

This study provides important insights of the effects of floristic plot size and placement on estimates of forest AGB based on ground surveys. We analysed 321 forest plots sampled in Central Amazonia (Fig. 2.1) and found significant differences in tree size structure between one-hectare and larger plots sampled in the same general region by two different field programs. One-hectare plots contained proportionally fewer small trees (31.8 - 40 cm DBH), lower tree density per hectare, and a larger contribution of very large emergent trees contributing to estimates of forest biomass.

Decreasing the uncertainties surrounding the global carbon budget is considered crucial for the success of carbon emission mitigation strategies, so accurate measures of forest carbon stocks are critical (Réjou-Méchain et al. 2014). However, despite several recommended methodological approaches (Houghton et al. 2001; Baraloto et al. 2013), official guidelines for in situ carbon assessments based on ground surveys have yet to be defined (Wertz-Kanounnikoff et al. 2008). To date, 1-ha plots are the most common sampling unit in tropical forest carbon assessments based on tree plots (Baraloto et al. 2013) so I elaborate on a number of conscious or unconscious effects of plot size, selection and placement that can generate bias and imprecision in estimates of forest AGB.

2.4.1 'Majestic forest' and canopy gap avoidance

We present a robust methodology to analyse the accuracy of 1-ha plots in estimating AGB for a given region. A large number of replicate plots from the RB and HRT inventories were widely distributed over the same interfluvial region of central Amazonia. HRT data were collected to fulfil strict environmental licensing requirements in Brazil surrounding hydrocarbon exploration. These guidelines are rigorous in relation to the exact geographic positioning of the plots, which in any case is pre-determined by the oil company to match specific potential impact zones of onshore drilling sites regardless of terrain structure, slope, and forest profile. HRT plot locations included areas with steep slopes, natural clearings, small streams and otherwise difficult terrain that could have been avoided had field plots been selected within a general area at the scale of tens of hectares.

Since RB plots are smaller and more widely dispersed, they are expected to have a wider range of basal area values than HRT, contrary to what was observed (Fig. 2.2). AGB is positively correlated with stem size, stem density and wood density

(Malhi et al. 2006; Hunter 2015) but RB plots had consistently higher AGB values, even though they had lower tree densities and a higher prevalence of light-wooded trees (Fig. 2.2). Slik et al. (2010) found similar patterns for Bornean forests, where AGB was correlated with basal area, rather than with stem density and wood density. However, the density of trees >70 cm DBH in Borneo is three-fold greater than in the Neotropics (Paoli et al. 2008), which suggests that the unusual relationship between stem density, WSG and AGB in RB plots could be attributed to areas containing a higher than random incidence of large-girthed trees. Yet the ratio between small and large trees in plots within the same region should be similar in the RB and HRT datasets.

The HRT and RB inventories were carried out in similar areas and there is no reason other than sampling design or procedures that could yield different tree size distributions. However, the higher prevalence of large canopy and emergent trees in the RB plots, evidenced by the cumulative DBH distribution (Fig. 2.4), suggests that the placement of RB plots may have been biased in the field.

The relatively low density of small trees in the RB plots (Fig. 2.5) suggests a "majestic forest" effect (Phillips et al. 2004), in which plots are consciously or inadvertently selected to capture areas of larger-girthed or higher statured trees. The RB plots had a more evenly spread size distribution whereas most trees in the HRT dataset were in the 31.8 – 40 cm DBH size class, which fits the inverse-J curve of tree size distributions in other tropical forest (Clark & Clark 1996; Chave et al. 2003). The so-called 'majestic forest effect' has important implications to field estimates of AGB in that surveyers may bypass sites dominated by smaller trees and positively select sites containing large trees. AGB is strongly correlated with bole diameter (Clark et al. 2001), whereby exceptionally large trees will have a disproportionate impact on AGB despite being relatively rare (Clark & Clark 1996; Chave et al. 2003). Thus, if for whatever reason plot location is non-random and systematically chosen to capture even a single very large tree, this effect will scale up over a sampling program and total forest AGB can be overestimated at large spatial scales.

The absence of clearings within the 231 one-hectare plots examined here, inferred by the stems density found in the plots (RB parcel with smaller density has 32 stems per hectare, value bigger than 15% of HRT parcels - 20 percentile ~ 36 stems/ha) is another anomaly with potentially marked effects on AGB estimation. This absence of clearings in the RB plots was also highlighted by Houghton et al. (2001). Canopy gaps alter the forest topology and have a strong role in improving AGB estimates (Clark & Clark 2000; Chave et al. 2008). Physical access to areas with treefall gaps can be difficult and requires considerable more time from field teams under tight schedules. Even small plots in areas containing a large treefall gap can substantially augment the workload allocated to that plot, increasing costs and risks. Avoiding these areas is therefore understandable from a logistical perspective alone. Estimating above and below ground carbon stocks in tropical forests is challenging for several reasons, including difficulties in accessing remote field sites (Phillips et al. 2003), which in the case of the RADAMBRASIL program was primarily via motorized boats and small aircraft operated by the Brazilian Air Force. However, tropical forest carbon stocks are characterised by a “slow in, rapid out” dynamic (Korner 2003) so that if biomass has been lost from an area due to natural process (e.g. blowdowns from convective storms, large treefall) gap recovery could take many years, further increasing the probability of AGB overestimates through gap avoidance sampling biases.

Differences in the tree size distribution between the HRT and the RB datasets also affect the contribution of each size class to the total AGB estimated for each plot (Fig. 2.6). The distribution of AGB across all tree size classes in the 1-ha plots can lead to a misunderstanding of the importance of including small trees in AGB assessments. Smaller trees (31.8-39.9 cm DBH) contributed most to the total AGB in the larger tree plots, which is consistent with most forest inventories where all trees ≥10cm DBH are measured (Chave et al. 2003). Four different size classes dominated contributions to AGB in the RB dataset. Given the urge to promote rapid, efficient and cheap methods to assess forest carbon stocks (Houghton et al. 2001), it would be plausible to recommend the use of small plots but exclude small trees. However, doing so would likely underestimate plot scale AGB, sacrificing precision in the interest of practicality. I also note that small trees as defined here (31.8 – 39.9 cm DBH) are considered as medium trees by others (Malhi et al. 2006) and most plot-based AGB assessments include all trees larger than 10 cm DBH.

However, differences encountered here relate to plot size and it is not possible to rule out other sampling problems in the RB protocol, such as a failure to measure all small trees within a plot due to severe time constraints. Those plots were sampled during long field work campaigns, so it is possible that the standardised sampling protocol was not always adhered to strictly to meet regional coverage targets.

Both datasets also present a bias towards areas that could be accessed by fluvial transport (Table 2.3) and there was no significant difference between datasets in distrance to the nearest river ($p = 0.263$). Yet a systematic selection of low-lying areas near rivers affects forest type (Malhi et al. 2006) and compensatory tree height (Detto et al. 2013), and should therefore be avoided. Access to many remote areas of Amazonia is very resource demanding in terms of time and money and setting up plots too far from rivers may not be feasible. One mitigating option would be to set up larger plots, place plots in the nearest upland plateaus, or sample a topographic gradient by establishing long rectangular plots (1000 x 10 m).

2.4.2 Effect of plot size on AGB estimate

AGB estimates were highly affected by the size of the plot. Dataset identification, which is a proxy of plot size, was present in all seven top ranking models and presented one of the highest contributions to these models, in addition to vegetation type and isothermality. Elevation also had a significant but smaller effect in the models (Table 2.4).

The fact that data source strongly affected total AGB estimates in our models, confirms the notion that plot size influences AGB estimates. Although all variables with significant effects in our models are considered to be important predictors of AGB in tropical forests, their effects are beyond the scope of this chapter. As the RB dataset is much better replicated, most conclusions here could result from overestimated AGB values. This could also explain the negligible importance of variables that are often considered to be decisive in AGB estimates in Amazonian forests, such as soil type (Castilho et al. 2006) and soil fertility (Malhi et al. 2006).

Forest AGB estimates are calculated under the assumption that plots are completely random in a given landscape and field measurements are largely error-free. In addition to sampling biases, methodological problems in data collection such as missed trees, trees measured twice (Chave et al. 2004), and over-representation of spatially aggregated species (Phillips et al. 2003) have a stronger effect in small plots. Large plots will likely average out these types of error, diluting their effects on AGB estimates.

The future of REDD+ and related climate policies are inextricably linked to our ability to appropriately estimate tropical forest carbon stocks (Gibbs et al. 2007). This has generated a growing demand to establish a standardised protocol for forest carbon inventories (Angelsen 2009). The definition of plot size to be used in this protocol is a trade-off between the quality of estimates generated and the resource/time spent to carry out the assessment on the ground (Angelsen 2008). However, our results show that plot size will hugely affect field measurements of AGB and frameworks based on small plots (≤ 1 ha) may not be a reliable option.

The recent use of remote sensing techniques in carbon assessments has become increasingly popular (Asner et al. 2010; Asner et al. 2013; Réjou-Méchain et al. 2014) given the high cost and logistical issues surrounding methods based entirely on fieldwork, suggesting that remote sensing will eventually dominate landscape scale carbon assessments (Hese et al. 2005; Goetz & Dubayah 2011). However, current technologies depend crucially on field data to calibrate imagery-generated models (Asner et al. 2010; Gonzalez et al. 2010; Réjou-Méchain et al. 2014) thereby requiring less but more precise field measurements than ground-based only methods. As such, 1-ha plots should be used carefully in independently validating remote sensing techniques.

2.5 Conclusion

Our study in a poorly known region of Brazilian Amazonia was conducted with a large dataset that is widely used to predict tropical forest AGB. We found evidence that 1-ha plots are susceptible to conscious and/or unconscious sampling bias, reducing the estimate precision of forest structure. Circumstantial evidence suggest that 1-ha plots deployed by the RADAMBRASIL program systematically avoided clearings and were placed in areas with fewer small stems, an effect described in the literature as the “majestic forest” syndrome. Both effects lead to significant AGB overestimates.

Sampling errors are often considered inherently random and are not assumed to have a large impact on forest biomass estimates (Muller-Landau et al. 2014). However, the sampling errors described here are systematic and specific, and can propagate across large spatial scales, distorting AGB estimates. Studies aiming to define a standard protocol to measure AGB should consider the problem of true randomisation in plot establishment in the field and the sampling units should be sufficiently large to avoid poor decisions in selecting unrepresentative sites. Large plots can subsume variation created by clearings and by site selection biases (Chave et al. 2008), and should be preferable in tropical forest environments. If the use of small plots is inevitable, plot positioning guidelines should consider pre-determined random placement and orientation of long rectangular plots regardless of terrain structure and topography.

2.6 *References*

Achard, F. et al., 2002. Determination of Deforestation Rates of the World’s Humid Tropical Forests. *Science*, 297(5583), pp.999–1002. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1070656 [Accessed March 25, 2016].

Angelsen, A., 2008. *Moving ahead with REDD: issues, options and implications*, Cifor.

Angelsen, A., 2009. *Realising REDD+: National strategy and policy options*, Cifor.

Anselin, L., 1988. Lagrange Multiplier Test Diagnostics for Spatial Dependence and Spatial Heterogeneity. *Geographical Analysis*, 20(1), pp.1–17. Available at: http://doi.wiley.com/10.1111/j.1538-4632.1988.tb00159.x [Accessed March 25, 2016].

Anselin, L. & Rey, S., 1991. Properties of Tests for Spatial Dependence in Linear Regression Models. *Geographical Analysis*, 23(2), pp.112–131. Available at: http://doi.wiley.com/10.1111/j.1538-4632.1991.tb00228.x [Accessed March 25, 2016].

Asner, G.P. et al., 2013. High-fidelity national carbon mapping for resource management and REDD+. *Carbon Balance and Management*, 8(1), p.7. Available at: http://www.cbmjournal.com/content/8/1/7 [Accessed March 25, 2016].

Asner, G.P. et al., 2010. High-resolution forest carbon stocks and emissions in the Amazon. *Proceedings of the National Academy of Sciences*, 107(38), pp.16738–16742. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.1004875107 [Accessed March 26, 2016].

Baraloto, C. et al., 2013. Rapid Simultaneous Estimation of Aboveground Biomass and Tree Diversity Across Neotropical Forests: A Comparison of Field Inventory Methods. *Biotropica*, 45(3), pp.288–298. Available at: http://doi.wiley.com/10.1111/btp.12006 [Accessed March 25, 2016].

Brown, S. & Lugo, A., 1992. Aboveground biomass estimates for tropical moist forests of the Brazilian Amazon. *Interciencia*, 1(17), pp.8–18.

de Castilho, C. V. et al., 2006. Variation in aboveground tree live biomass in a central Amazonian Forest: Effects of soil and topography. *Forest Ecology and Management*, 234, pp.85–96.

Chave, J. et al., 2008. Assessing Evidence for a Pervasive Alteration in Tropical Tree Communities G. M. Mace, ed. *PLoS Biology*, 6(3), p.e45. Available at: http://dx.plos.org/10.1371/journal.pbio.0060045 [Accessed March 25, 2016].

Chave, J. et al., 2004. Error propagation and scaling for tropical forest biomass estimates. , (February), pp.409–420.

Chave, J. et al., 2014. Improved allometric models to estimate the aboveground biomass of tropical trees. *Global Change Biology*, 20(10), pp.3177–3190. Available at: http://doi.wiley.com/10.1111/gcb.12629 [Accessed March 25, 2016].

Chave, J. et al., 2003. Spatial and temporal variation of biomass in a tropical forest: results from a large census plot in Panama. *Journal of Ecology*, 91(2), pp.240–252. Available at: http://doi.wiley.com/10.1046/j.1365-2745.2003.00757.x [Accessed March 25, 2016].

Chave, J. et al., 2005. Tree allometry and improved estimation of carbon stocks and balance in tropical forests. *Oecologia*, 145(1), pp.87–99. Available at: http://link.springer.com/10.1007/s00442-005-0100-x [Accessed March 25, 2016].

Clark, D.. & Clark, D.., 2000. Landscape-scale variation in forest structure and biomass in a tropical rain forest. *Forest Ecology and Management*, 137(1), pp.185–198.

Clark, D.A. et al., 2001. Measuring net primary production in forests: concepts and field methods. *Ecological Applications*, 11(2), pp.356–370. Available at: http://doi.wiley.com/10.1890/1051-0761(2001)011[0356:MNPPIF]2.0.CO;2 [Accessed March 25, 2016].

Clark, D.B. & Clark, D.A., 1996. Abundance, growth and mortality of very large trees in neotropical lowland rain forest. *Forest Ecology and Management*, 80(1), pp.235–244.

Detto, M. et al., 2013. Hydrological Networks and Associated Topographic Variation as Templates for the Spatial Organization of Tropical Forest Vegetation G. Bohrer, ed. *PLoS ONE*, 8(10), p.e76296. Available at: http://dx.plos.org/10.1371/journal.pone.0076296 [Accessed March 24, 2016].

DNPM, 1973. *Projeto RADAMBRASIL: Levantamento de Recursos Naturais*, Rio de Janeiro, Brazil.

ESRI, 2011. ArcGIS Desktop: Release 10.

FAO, 2001. Global Forest Resources Assessment. *FAO Forestry Paper*, 140.

FAO, 1981. *Manual of forest inventory with special reference to mixed tropical forests*, Rome, Italy.

FAO, 2007. *State of the World's Forests*, Rome, Italy.

Fearnside, P.M., 1997. Greenhouse gases from deforestation in Brazilian Amazonia: net committed emissions. *Climatic Change*, 35(3), pp.321–360. Available at: http://link.springer.com/10.1023/A:1005336724350.

Fisher, J.I. et al., 2008. Clustered disturbances lead to bias in large-scale estimates based on forest sample plots. *Ecology Letters*, 11(6), pp.554–563. Available at: http://doi.wiley.com/10.1111/j.1461-0248.2008.01169.x.

Florian Wittmann et al., 2006. Wood specific gravity of trees in Amazonian white-water forests in relation to flooding. *IAWA Journal*, 27(3), pp.255–268.

Friedlingstein, P. et al., 2014. Persistent growth of CO2 emissions and implications for reaching climate targets. *Nature Geoscience*, 7(10), pp.709–715. Available at: http://www.nature.com/doifinder/10.1038/ngeo2248 [Accessed March 25, 2016].

Geist, H.J. & Lambin, E.F., 2002. Proximate Causes and Underlying Driving Forces of Tropical Deforestation. *BioScience*, 52(2), p.143. Available at:

http://bioscience.oxfordjournals.org/cgi/doi/10.1641/0006-3568(2002)052[0143:PCAUDF]2.0.CO;2.

Gibbs, H.K. et al., 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters*, 2(4), p.045023. Available at: http://stacks.iop.org/1748-9326/2/i=4/a=045023?key=crossref.4118e8af5a9a3ac02c1bb32f8a92c50f [Accessed March 25, 2016].

Goetz, S. & Dubayah, R., 2011. Advances in remote sensing technology and implications for measuring and monitoring forest carbon stocks and change. *Carbon Management*, 2(3), pp.231–244. Available at: http://www.tandfonline.com/doi/abs/10.4155/cmt.11.18 [Accessed March 25, 2016].

Gonzalez, P. et al., 2010. Forest carbon densities and uncertainties from Lidar, QuickBird, and field measurements in California. *Remote Sensing of Environment*, 114(7), pp.1561–1575.

Hansen, M.C. et al., 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, 342(6160), pp.850–853. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1244693 [Accessed March 20, 2016].

Hese, S. et al., 2005. Global biomass mapping for an improved understanding of the CO2 balance—the Earth observation mission Carbon-3D. *Remote Sensing of Environment*, 94(1), pp.94–104.

Hijmans, R.J. et al., 2004. The WorldClim interpolated global terrestrial climate surfaces.

Holdaway, R.J. et al., 2014. Propagating Uncertainty in Plot-based Estimates of Forest Carbon Stock and Carbon Stock Change. *Ecosystems*, 17(4), pp.627–640. Available at: http://link.springer.com/10.1007/s10021-014-9749-5 [Accessed March 25, 2016].

Hopkins, M.J.G., 2007. Modelling the known and unknown plant biodiversity of the Amazon Basin. *Journal of Biogeography*, 34(8), pp.1400–1411. Available at: http://doi.wiley.com/10.1111/j.1365-2699.2007.01737.x.

Houghton, R.A., 2005. Aboveground Forest Biomass and the Global Carbon Balance. *Global Change Biology*, 11(6), pp.945–958. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2005.00955.x [Accessed March 24, 2016].

Houghton, R.A. et al., 2001. The spatial distribution of forest biomass in the Brazilian Amazon: a comparison of estimates. *Global Change Biology*, 7(7), pp.731–746. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2001.00426.x [Accessed March 19, 2016].

Hunter, J.T., 2015. Changes in Allometric Attributes and Biomass of Forests and Woodlands across an Altitudinal and Rainfall Gradient : What Are the Implications of Increasing Seasonality due to Anthropogenic Climate Change ? , 2015.

IEA, 2009. World Energy Outlook 2009. Available at: http://www.worldenergyoutlook.org/2009.asp.

IPCC, 2015. *Special Report on Carbon Dioxide Capture and Storage*, Cambridge, UK: Cambridge University Press.

Jari Florestal, 2013. *Wood density of canopy trees data set*, Monte Dourado, Brasil.

Korner, C., 2003. ATMOSPHERIC SCIENCE: Slow in, Rapid out--Carbon Flux Studies and Kyoto Targets. *Science*, 300(5623), pp.1242–1243. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1084460 [Accessed March 25, 2016].

Lehner, B., Verdin, K. & Jarvis, A., 2008. New Global Hydrography Derived From Spaceborne Elevation Data. *Eos, Transactions American Geophysical Union*, 89(10), p.93. Available at: http://doi.wiley.com/10.1029/2008EO100001 [Accessed March 25, 2016].

Malhi, Y. et al., 2006. The regional variation of aboveground live biomass in oldgrowth Amazonian forests. *Global Change Biology*, 12(7), pp.1107–1138. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2006.01120.x.

Maniatis, D. et al., 2011. Evaluating the Potential of Commercial Forest Inventory Data to Report on Forest Carbon Stock and Forest Carbon Stock Changes for REDD+ under the UNFCCC. *International Journal of Forestry Research*, 2011, pp.1–13. Available at: http://www.hindawi.com/journals/ijfr/2011/134526/ [Accessed March 25, 2016].

Muller-Landau, H. et al., 2014. Detecting and projecting changes in forest biomass from plot data. In D. Coomes, D. Burslem, & W. Simonson, eds. *Forests and Global Change*. pp. 381–415.

Nogueira, E.M. et al., 2008. Estimates of forest biomass in the Brazilian Amazon: New allometric equations and adjustments to biomass from wood-volume inventories. *Forest Ecology and Management*, 256(11), pp.1853–1867.

Pan, Y. et al., 2011. A Large and Persistent Carbon Sink in the World's Forests. *Science*, 333(6045), pp.988–993. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1201609 [Accessed March 25, 2016].

Paoli, G.D., Curran, L.M. & Slik, J.W.F., 2008. Soil nutrients affect spatial patterns of aboveground biomass and emergent tree density in southwestern Borneo. *Oecologia*, 155(2), pp.287–299. Available at: http://link.springer.com/10.1007/s00442-007-0906-9 [Accessed March 25, 2016].

Phillips, O.L. et al., 2003. Efficient plot-based floristic assessment of tropical forests. *Journal of Tropical Ecology*, 19(6), p.S0266467403006035. Available at: http://www.journals.cambridge.org/abstract_S0266467403006035 [Accessed March 25, 2016].

Phillips, O.L. et al., 2004. Pattern and process in Amazon tree turnover, 1976-2001. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 359(1443), pp.381–407. Available at: http://rstb.royalsocietypublishing.org/cgi/doi/10.1098/rstb.2003.1438 [Accessed March 25, 2016].

R Core Team, 2015. R: A language and environment for statistical computing. Available at: http://www.r-project.org/.

Réjou-Méchain, M. et al., 2014. Local spatial structure of forest biomass and its consequences for remote sensing of carbon stocks. *Biogeosciences Discussions*, 11(4), pp.5711–5742. Available at: http://www.biogeosciences-discuss.net/11/5711/2014/ [Accessed March 25, 2016].

Saatchi, S.S. et al., 2011. Benchmark map of forest carbon stocks in tropical regions across three continents. *Proceedings of the National Academy of Sciences*, 108(24), pp.9899–9904. Available at:

Sales, M.H. et al., 2007. Improving spatial distribution estimation of forest biomass with geostatistics: A case study for Rondônia, Brazil. *Ecological Modelling*, 205(1), pp.221–230.

Schneider, M. & Peres, C.A., 2015. Environmental Costs of Government-Sponsored Agrarian Settlements in Brazilian Amazonia R. Zang, ed. *PLOS ONE*, 10(8). Available at: http://dx.plos.org/10.1371/journal.pone.0134016.

Slik, J.W.F. et al., 2010. Environmental correlates of tree biomass, basal area, wood specific gravity and stem density gradients in Borneo's tropical forests. pp.50–60.

Wagner, F. et al., 2010. Effects of Plot Size and Census Interval on Descriptors of Forest Structure and Dynamics. , 42(6), pp.664–671.

Wara, M., 2007. Is the global carbon market working? *Nature*, 445(7128), pp.595–596. Available at: Wertz-Kanounnikoff, S. et al., 2008. How can we monitor, report and verify carbon emissions from forests. In *Moving Ahead with REDD: Issues, Options and Implications*. Bogor, Indonesia: Center for International Forestry Research (CIFOR).

Chapter 3

Watershed scale assessment of anthropogenic threats and conservation opportunities across the Amazon basin

To be submitted as:

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3.1 Introduction

Establishing effective policies for conservation planning and action present huge challenges, yet it is critical for regional scale biodiversity conservation. Although several authors agree that the most efficient way to approach biodiversity conservation is through systematic conservation planning (Margules & Pressey 2000; Pressey & Bottrill 2008), there have been lively debates over how the systematization should be applied in practice. Defining the level of interest in any analysis — be it species (Lindenmayer et al. 2002; Mace et al. 2007), site (Cowling et al. 2003; Eken et al. 2004) or landscape (Grumbine 1994; Beissinger & Westphal 1998) — has been a matter of discussion for several years (but see Brooks (2010) for a comprehensive discussion on the strengths and shortcomings of each strategy). Moreover, defining the optimal analytical pathway in prioritizing conservation areas has also been contentious, with prioritization algorithms of either minimum-set or maximum-coverage considered under different situations (Cabeza & Moilanen 2001; Arponen et al. 2005; Wilson et al. 2009). Protected areas (hereafter, PAs) are widely considered as the cornerstone of conservation planning strategies (Margules & Pressey 2000) and there have been a number of discussions about PA design (Brandon & Wells 1992; Sanderson et al. 2002; Crouzeilles et al. 2011), the choice of biodiversity surrogates used to define PA location and distribution (Noss et al. 1996; Caro & O’Doherty 1999; Caro et al. 2004; Roberge & Angelstam 2004), and the inclusion of ecosystem services in prioritizing PAs (Millennium Ecosystem Assessment 2003).

Based on the conservation planning principles of irreplaceability, vulnerability, rarity, representation and adequacy, different prioritization approaches have been proposed to define priority areas for conservation (Eken et al. 2004; Sanderson et al. 2002; Ricketts et al. 2005). However, the strength of each of those approaches depends on the scale of analysis (Wilson et al. 2009), limiting the generalization of the concept to a broader context. Moreover, different sectors of society have different perceptions on what exactly should be prioritized in biodiversity conservation policies, so that successful policies should be designed based on multiple priority setting methods (Mace et al. 2007). Considering these needs, large-scale biophysical features of any geographic region could form the basic unit of conservation planning, allowing multiple prioritization approaches to converge onto a common landscape template.

The alignment of PAs borders with watershed dividers has long been proposed by some authors (Peres & Terborgh 1995; Shafer 1999), but the potential for using watersheds in biodiversity conservation policies goes far beyond this application. Large watersheds can have a decisive role as biodiversity conservation planning units not just offering a solution for the debated PAs design problem but also maximizing the use of information already available in the prioritization of areas to conservation. With the availability of remote sensing data and development of GIS technologies, watershed and sub-watershed boundaries can be readily identified, facilitating the compartmentalisation of extensive regions into manageable subunits. Watersheds can also function as a meaningful and ecologically relevant target for biodiversity assessments (Naiman et al. 2010), particularly in terms of nutrient cycling (Vannote et al. 1980; Junk et al. 1989; Montgomery 1999). These may include freshwater biotas, which have been largely ignored in conservation planning efforts (Brooks 2010). Performing a conservation planning evaluation at the watershed level also allows consideration of different scales including sub-watersheds, thereby ensuring a more adequate management strategy according to the conservation target. A modular cross-scale planning unit that retains a measurable relationship with the larger scales is also a key advantage in systematic conservation planning that considers the variation in spatial requirements of vulnerable and threatened species (Boyd et al. 2008).

Amazonia accounts for over half of the remaining tropical forests worldwide and is the largest continental hydrogeological unit, spanning over $7x10^6$ km^2 (Goulding et al. 2003). Biodiversity conservation planning in this biome from the perspective of major watersheds is both logical and appealing. To fulfil their conservation goals, PAs must be effectively protected from external threats, with staff size and parkguard density often considered as strong predictors of the effectiveness of strictly-protected tropical reserves (Bruner et al. 2001). Considering that Amazonian PAs are severely underfunded and understaffed, aligning those PAs with watershed boundaries would greatly enhance their defensibility through strategic deployment of scarce reserve personnel (Peres & Terborgh 1995). Moreover, most of lowland Amazonia remains roadless (Carvalho et al. 2001; Bass et al. 2010), rendering rivers the most important route for human colonization and population growth, and an important vector of threat expansion. Rivers are also the most relevant geographic feature for Amazonian rural households, which are primarily riverine (Parry et al. 2010), as rivers are used as landmarks and the primary means of transportation for hunting, fishing and other subsistence activities.

Conservation planning in tropical forest regions is also severely hindered by massive knowledge gaps concerning biodiversity, with all biogeographic studies showing profound biases in sampling effort for a wide range of taxonomic groups (Kress et al. 1998; Junk & Piedade 2004; Peres 2005; Oliveira 2006; Hopkins 2007). In contrast, major watersheds are unbiased in their delimitation and present a wide diversity of habitat types along topographic gradients (e.g. high plateaus, slopes, floodplains, rivers and lakes) and can be used as “coarse filters” (Hunter Jr. 1991) to allocate protection to areas lacking sufficient reliable information on biodiversity. In addition, rivers can function as important biogeographic barriers to dispersal for many species, so that designing PAs coinciding with watershed boundaries presumably maximizes complementarity in species diversity. Yet, there are no quantitative assessments of biodiversity conservation planning at the watershed scale across the wildlands of Amazonia.

This paper assesses the vulnerability status of all major watersheds across the Amazon and neighbouring river basins, and discusses the importance of applying the watershed concept as a biodiversity conservation planning unit. Our analyses were based on a detailed vulnerability surface created considering multiple indicators of human related threats. Beyond mapping and discussing current threats, our analysis projected the likely expansion of vulnerable areas by 2050. In doing so, we identify areas that are more prone to rapid increases in their vulnerability levels with direct consequences for conservation planning in the biome. Finally, we consider the conservation opportunities and challenges of conservation planning in Amazonia at the watershed scale.

3.2 Methods

3.2.1 Study region

Our study region comprises the entire hydrologic domain of the Amazon River (*sensu lato*) and two adjacent watersheds (Araguari and Tocantins) that are part of the ecoregional domain of Amazonia (Goulding et al. 2003). This area was segmented into 23 sub-domains representing major watersheds (Fig. 3.1), collectively encompassing >6,580,000 km^2, including 83% of the formally recognized phytogeographic boundaries of Amazonia (Eva et al. 2005). All watersheds flow directly into the Amazon River, except for the Araguari, Tocantins and Branco. We decided to subdivide the Branco and the Negro watersheds despite their direct hydrologic Guianan Shields connectivity, due to their very distinctive geochemical and human development characteristics. All areas located in the immediate neighbourhood of the Amazon River but which do not belong to any of our 23 hydrologic subdomains were disregarded in the analysis, but in aggregate the entire disregarded area accounted for only 8.5% of the study region (Fig. 3.1).

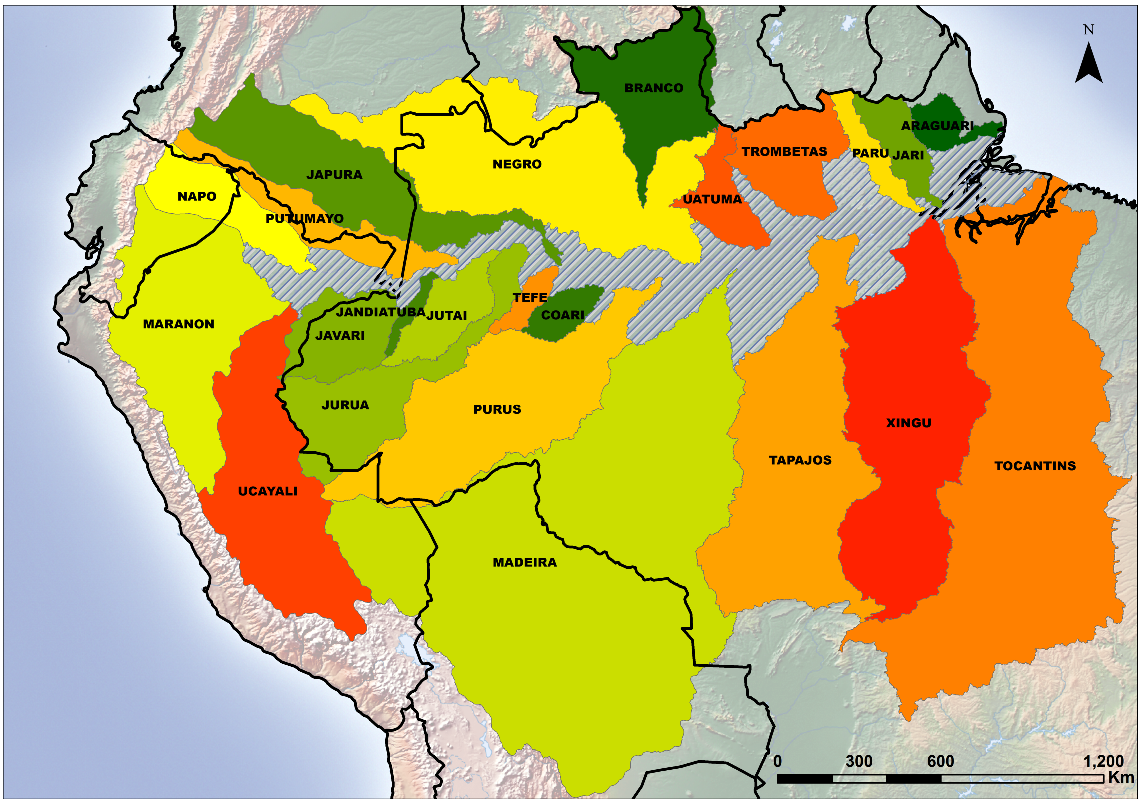


Fig. 3.1 – The 23 major Amazonian hydrologic domains examined in this analysis. Each domain was segmented on the basis of their hydrological boundaries and represents the entire watershed of rivers and their tributaries. Shaded areas along the main channel of the Amazon River do not belong to any of the major watersheds and were excluded from the analyses. Black lines indicate the boundaries of the seven Amazonian countries included in the study region.

Watershed delimitation was defined on the basis of hydrological shapefiles (Ottobacias) from ANA (Agência Nacional de Águas; http://www2.ana.gov.br/), the Brazilian federal agency responsible for implementing the management of water resources. These shapefiles delimit Amazonian watersheds based on the classification method created by Otto Pfafstetter, a Brazilian hydrogeologist who adapted the hierarchical hydrology concept to South American river basins. We subsequently redrew and corrected this classification for some portions of the Greater Amazonian Watershed to more accurately align all watershed boundaries as visualised by a medium-resolution (3 arc-seconds or approximately 90m) STRM (Shuttle Radar Topography Mission) imagery.

3.2.2 Human population distribution

To examine how Amazonian watersheds are settled from the least to the most remote areas, we constructed cumulative distribution functions (CDF) of the entire human population within each of the 23 major Amazonian watersheds, which were segmented into 50-km length sections considering their main axis from the mouth to the headwaters. CDFs therefore represent the cumulative population size as a function of distance from the lowest segment of the main tributary near the Amazon River, which typically coincides with a market town (Parry & Peres 2015). Rapidly escalating CDF asymptotes indicate greater population concentration near the river mouth, whereas sigmoidal curves indicate large populations in headwater regions. We also calculated the AUC (area under the curve) for each CDF distribution, with higher AUC values indicating higher levels of population aggregation near the lower reaches of each watershed. Using Pearson's correlation coefficients, we then compared those human occupation patterns in relation to the road density (km/km^2) across each watershed.

HPD values were extracted from a digital map created based on two sources: the Brazilian Institute of Geography and Statistics (IBGE 2000) for the Brazilian territory, and the Global Rural-Urban Mapping Project (CIESIN et al. 2011) for all other Amazonian countries. We used human population data standardized to the year 2000, and both the IBGE and CIESIN databases were combined into a single raster with a 30 arc-second spatial resolution (~1km).

3.2.3 Vulnerability status

Using the Protected Area Tools extension in ArcGIS (Schill & Raber 2012), we created an Environmental Risk Surface (hereafter, ERS; see McPherson et al. 2008) as a measure of vulnerability to anthropogenic threats throughout the entire study area. To model this surface, we used deforestation data from 2000 to 2013 (Hansen et al. 2013), and the following surrogates of human settlements: human population density (HPD, see below), areas of persistent nocturnal lights (image and data processing by NOAA's National Geophysical Data Center - http://ngdc.noaa.gov/eog/dmsp/downloadV4composites.html), and paved and unpaved road networks (http://www.openstreetmap.org). Deforestation was modelled as a fixed positional value, whereas roads and night lights were a function of distances from sources (decreasing values with 5-km radii to roads and 20-km radii to night lights). HPD was assigned into six logarithmic bins as ordinal (rank) categories (0, 0.01-1, 1.01-10, 10.01-25, 25.01-50, >50 persons/km^2). The final ERS value was then defined as the sum of the values for all overlapping threat layers.

3.2.4 Vulnerability projection

We generate a vulnerability prediction for 2050 considering the deforestation projection under the BAU (business as usual) scenario of Soares-Filho et al. (2006). Their deforestation map was created considering expansion of paved roads, the existing PA network and its effectiveness, deforestation rates and areas currently deforested. We also considered national and transnational roads planned by Brazilian National Department of Infrastructure and Transportation (DNIT) when generating the ERS surface for 2050.

HPD density and distance from settlements were kept constant between the two vulnerability maps created. Both variables depend on existing and new infrastructure projects and without a more detailed analysis that can count of strategic planning data from national governments it would be inaccurate to include those variables in the model. We believe the inclusion of planned roads and projected deforestation in our vulnerability model is sufficient to allow a clear identification of which areas are most prone to succumb to elevated vulnerability values by 2050.

3.2.5 Watershed political divisions

To evaluate the influence of administrative regions on the degree of vulnerability of each watershed, we segmented the 23 watersheds into local and regional political subdivisions according to the Global Administrative Areas map (GADM, http://www.gadm.org/home). As different countries do not have the same political units, we used the most similar subdivision to represent local and regional governments (Table 3.1).

Table 3.1 – National administrative regions of the seven Amazonian countries included in this study.

COUNTRY	REGIONAL DIVISION	LOCAL DIVISION
Bolivia	Department	Municipality
Brazil	State	Municipality
Colombia	Department/Commissary/Intendancy	Municipality
Ecuador	Province	Canton
Guyana	Region	Neighbourhood Councils[1]
Peru	Region/Province	District
Venezuela	State	Municipality

[1] Some of the administrative districts are listed as "non-classified" according to the GADM database.

Watershed area, administrative subdivisions and countries were used to model the mean ERS value of each watershed. Because ERS values of local and regional administrative districts were highly correlated (r = 0.840), we retained only regional divisions in the final model. This also avoided high levels of correlation between local divisions and distance from settlements, one of the variables incorporated into the final ERS map. Degree of vulnerability was modelled using the log-transformed mean watershed scale ERS value to fit a Gaussian distribution, as following:

$$ERS(log) \sim Area(km2) + Country(n) + States(n)$$

Unless explicitly stated, all GIS processing and statistical analyses were performed using ArcGIS 10.1 (ESRI 2011) and R 3.1.3 (R Core Team 2015), respectively.

3.3 Results

We delimited a total of 23 major river basins across seven of the nine Amazonian countries, namely Brazil, Bolivia, Peru, Ecuador, Colombia, Venezuela and Guyana, extending from lowland forest at sea level to high elevations of ~6400 m in the tropical Andes (Fig. 3.1). The area of these 23 major watersheds ranged across two orders of magnitude from the relatively small Jandiatuba river basin (~15,000 km^2) to the vast Madeira river basin (>1,460,000 km^2).

3.3.1 Effect of roads on human population distribution

Patterns of human occupancy changed according to the density of road networks within each watershed (r = -0.528). Human populations in entirely roadless watersheds were highly concentrated in the lowest part of the watersheds. Conversely, watersheds that already exhibit a large road network, and are often segmented in their upper reaches, had more evenly distributed populations along the main watershed axis being generally more densely settled in their headwaters (Fig. 3.2). The heavily populated Rio Negro basin has major part of its inhabitants largely concentrated around Manaus, the largest urban center in the Amazon (~2 million people; IBGE 2010), resulting in a steeply rising CDF, which is consistent with roadless watersheds elsewhere. The Xingu, Tapajós and Tocantins watersheds are criss-crossed by roads, resulting in more gentle CDF curves, while the other watersheds with high road density concentrate most of their human populations in their upper portions.

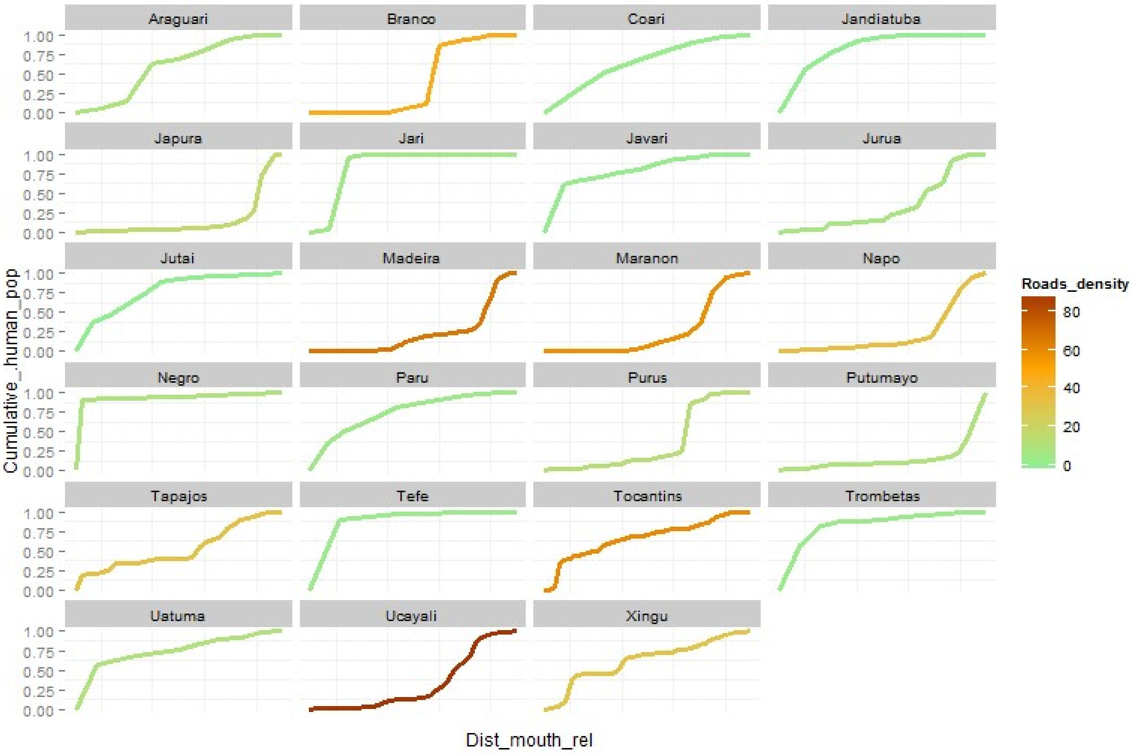


Fig. 3.2 – Longitudinal pattern of human occupation across all major Amazonian watersheds. CDF curves were created along a longitudinal gradient, from the mouth of the main river to its headwaters, with fluvial distances shown on the x-axis representing the entire length of the main watershed axis. The lines indicate cumulative functions for the human population size, which were colour-coded according to the road density in each watershed (with road density increasing from green to dark orange).

3.3.2 Vulnerability of major Amazonian watersheds

We estimated the probability surface of potential human impact within each major watershed, as shown in a basin-wide vulnerability map (Fig. 3.3). The map presents the geographic variation in levels of threat, as defined by the Environmental Risk Surface (ERS) metric. This measure of vulnerability across the entire Amazon Basin ranged from 0 to 402 (mean =47.60; SD = 47.56). The Ucayali, Tocantins, Maranon and Madeira watersheds had on average the highest vulnerability values (70.0, 64.8, 63.1 and 56.9, respectively), whereas the Jari, Trombetas, Javari and Jutai were the least vulnerable (15.0, 21.4, 22.3 and 22.5, respectively). Beyond the large geographic asymmetry in the overall distribution of vulnerability, the most vulnerable areas were concentrated in the peripheral portions of the biome, particularly along more accessible broad transitions to seasonally drier and higher agricultural value regions of eastern, southern and southwestern Amazonia.

The south-eastern watersheds (Tocantins, Xingu, Tapajós and Madeira) are predicted to succumb to the highest levels of vulnerability by 2050 with a noticeable advance of development frontiers to the headwaters of the Purus and Juruá watersheds (Fig. 3.4). However, the most dramatic changes in vulnerability patterns will occur in the northern watersheds. Several areas of presently low vulnerability will become highly threatened by 2050, highlighting the imperative of a proactive approach in the conservation of those subregions. The straight lines in the map (Fig. 3.4) represent planned paved roads, so we can expect much higher vulnerability values than those shown here for areas alongside those roads.

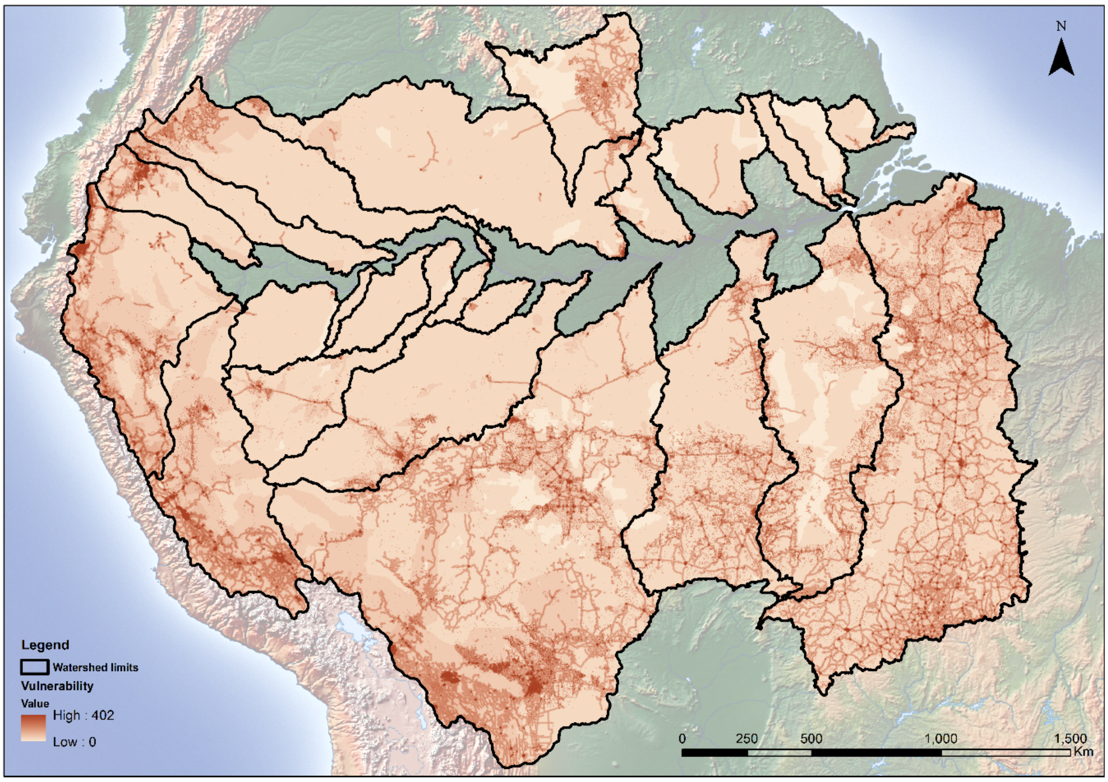


Fig. 3.3 – Geographic distribution of vulnerability to anthropogenic habitat disturbance throughout the study region. Vulnerability scores are colour-coded from higher (darker) to lower (lighter) values associated with each ~1-km^2 pixel. Differences between vulnerability levels in watersheds located either north or south of the Amazon River are noteworthy as road density to date remains far lower to the north of the Amazon. Black lines indicate the geographic boundaries of all 23 major river basins considered here.

Our projection of vulnerable areas for Amazonia presents a tendency for human related activities to become for diffuse. However, this is a very conservative estimate that does not take into account currently planned large national scale infrastructure projects, which will likely have devastating effects on forest biodiversity, eroding wilderness areas in a long-term scenario.

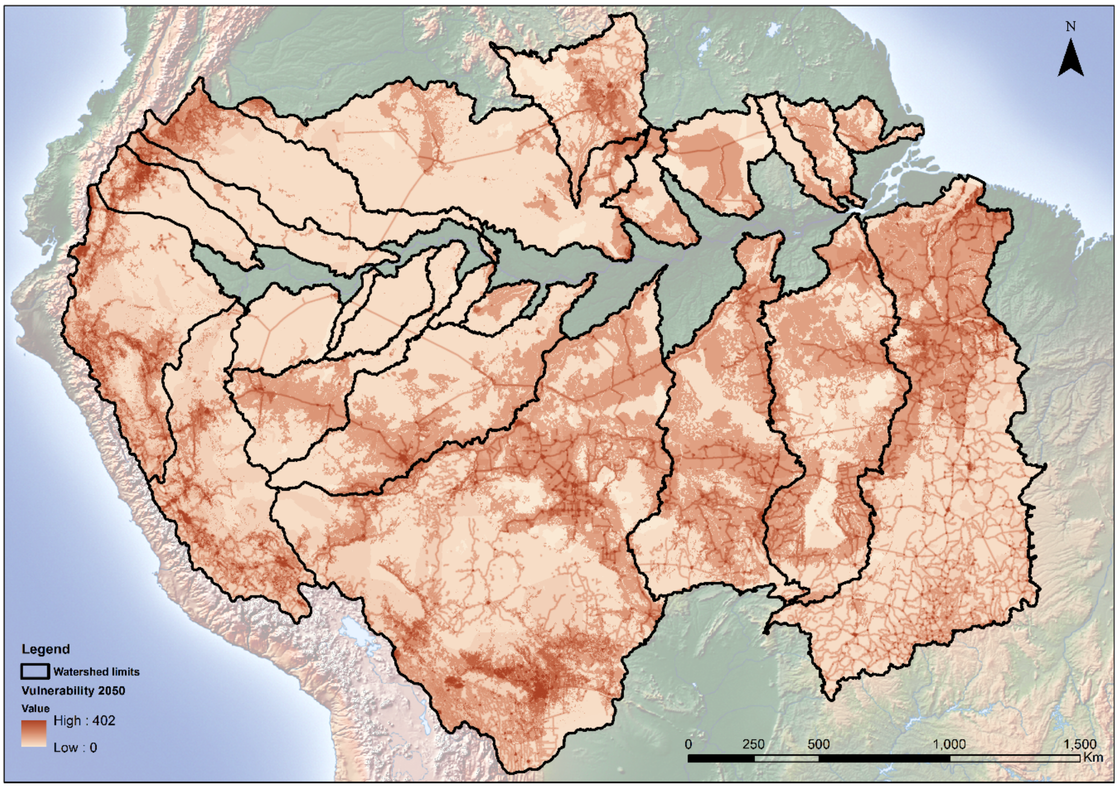


Fig. 3.4 – Vulnerability prediction based on an anthropogenic habitat disturbance scenario forecasted to 2050. The map represents vulnerability values (colour-coded as in Fig. 3.3) generated using the same data plus the deforestation predictions of Soares-Brito et al. (2006) and the paved roads planned to be constructed by the Brazilian government within Brazilian Amazonia, some of which connecting other Amazonian countries.

3.3.3 Political segmentation and vulnerability

Given the vast territorial extent of some Amazonian watersheds, it comes as a surprise that they are largely contained within only one or a few countries. Only three watersheds (Madeira, Napo and Putumayo) have more than a quarter of their area within two different countries, with ~70% of all watersheds with >90% of their areas inside a single country (Table 3.2). The political fragmentation of any major watershed can be decisive in the implementation of transboundary conservation strategies, with conservation policy accords more likely to be achieved in less fragmented watersheds.

States were a significant predictor ($\beta = 0.02$, $p < 0.001$) of overall degree of vulnerability within the watersheds evidencing that in addition to imposing greater challenges to enforcing protection measures, more politically fragmented areas are also more likely to be threatened. The number of countries sharing a watershed or watershed area had no effect on watershed vulnerability. We note that the absence of a country effect can reflect the small variance in our dataset.

Table 3.2 – Distribution of major Amazonian watersheds across national scale political subdivisions of seven Amazonian countries. Percentage values and numbers indicate the proportion of each watershed area within different countries; and the number of political regions (states and municipal counties) within each watershed, respectively.

BASIN	AREA (km^2)	Countries (n)	COUNTRY	%	REGIONAL DIVISIONS		LOCAL DIVISIONS	
					By country	Total	By country	Total
ARAGUARI	37,422	1	Brazil	100	1	1	13	13
BRANCO	194,043	3	Brazil	93.5	2	7	16	37
			Guyana	6.2	3		16	
			Venezuela	0.2	2		5	
COARI	35,770	1	Brazil	100	1	1	3	3
JANDIATUBA	14,977	1	Brazil	100	1	1	4	4
JAPURA	261,152	2	Brazil	19.3	1	10	11	65
			Colombia	80.7	9		54	
JARI	58,187	1	Brazil	100	2	2	6	6
JAVARI	109,473	2	Brazil	76.9	2	3	8	21
			Peru	23.1	1		13	
JURUA	192,932	2	Brazil	93.7	2	4	26	38
			Peru	6.3	2		12	
JUTAI	77,925	1	Brazil	100	1	1	9	9
MADEIRA	1,460,976	3	Bolivia	53	8	16	211	356
			Brazil	38.9	4		85	

BASIN	AREA (km^2)	Countries (n)	COUNTRY	%	REGIONAL DIVISIONS		LOCAL DIVISIONS	
					By country	Total	By country	Total
			Peru	8.1	4		60	
MARANON	368,784	2	Ecuador	17.8	11	23	57	563
			Peru	82.2	12		506	
NAPO	110,040	3	Colombia	1.7	2	10	5	42
			Ecuador	55.9	7		24	
			Peru	42.4	1		13	
NEGRO	522,766	3	Brazil	76.5	2	7	20	40
			Colombia	14.1	4		16	
			Venezuela	9.4	1		4	
PARU	40,452	1	Brazil	100	1	1	4	4
PURUS	389,322	3	Bolivia	0.5	1	6	4	44
			Brazil	93.4	3		33	
			Peru	6.1	2		7	
PUTUMAYO	112,594	4	Brazil	12.3	1	6	6	41
			Colombia	51.1	3		26	
			Ecuador	2.9	1		2	
			Peru	33.8	1		7	
TAPAJOS	511,086	1	Brazil	100	4	4	66	66
TEFE	24,648	1	Brazil	100	1	1	7	7
TOCANTINS	981,052	1	Brazil	100	10	10	445	445
TROMBETAS	126,278	2	Brazil	99.8	3	5	9	11
			Guyana	0.2	2		2	
UATUMA	68,656	2	Brazil	99.9	3	4	13	14
			Guyana	0.1	1		1	
UCAYALI	357,749	1	Peru	100	14	14	576	576
XINGU	524,146	1	Brazil	100	2	2	64	64

3.4 Discussion

In this chapter, we assessed the present and future major anthropogenic threats to the integrity of Amazonian watersheds, highlighting the conservation opportunities for biodiversity protection policies at this scale of analysis.

Vulnerability is considered a key factor in systematic conservation planning (Wilson et al. 2005). By generating a vulnerability map we quantify and pinpoint anthropogenic threats throughout Amazonia, permitting effective comparative assessment of the vulnerability status across major watersheds, which can serve as an effective tool for conservation planning. The variation in vulnerability was accessed at the watershed scale because we believe this is the most adequate scale for conservation planning across Amazonia, and the advantages of this approach are discussed below.

3.4.1 Watershed colonization

The current human occupation of Amazonia declines rapidly with distance from urban centres (Parry et al. 2010) and this is the strongest indicator of its colonization vector. At the watershed scale the human occupation of Amazonia, barring a few exceptions, followed two patterns depending on road density (Fig. 3.2). Roadless watersheds were colonised following an upriver pattern, in which the first towns were founded near the mouth of rivers and human settlements spreading into more remote sites farther upstream. However, the majority of Amazonian roads are located at peripheral portions of the biome. In watersheds with a more developed road network the process of colonization was therefore inverted with areas near large market centers located outside the biome, being colonised first.

The influx of human settlements is expected to bring about dramatic land use changes (Meyer & Turner 1992; Bass et al. 2010; Geist & Lambin 2002), having direct effects on the biological (Lopes & Ferrari 2000) and physical environment (Roulet et al. 1999) in Amazonia. Understanding the dynamics of watershed scale population expansion is therefore important because it allows conservation planners to forecast how threats will spread throughout the region. To assess how human colonization affects landscapes is challenging and involves several other factors (e.g. climate, politics, economics, vegetation) than the vector of colonization alone (Arce-Nazario 2007). However, we have shown that at a large scale the vector of colonization is an effective predictor of the human threat expansion. For an efficient application of a proactive conservation approach the spread of human threats would have to be contained by strategic placement of PAs. Therefore, the longitudinal pattern of human occupation shown here provides important information that can be used for large scale conservation planning through watersheds.

3.4.2 Vulnerability status of Amazonian watersheds

Our vulnerability map (Fig. 3.3) is highly influenced by roads since distance to roads is one of the variables utilised to generate the map, in conjunction with two other variables, deforestation and distance to settlements, which are also correlated with distance to roads. The effect of roads on deforestation (Laurance et al. 2002; Nepstad et al. 2001; Carvalho et al. 2001) and biodiversity (Lambert et al. 2014; Ahmed et al. 2014) is well documented, reaffirming the relevance of roads in a threat analysis such as presented here. However, beyond the marked expected effect of roads on our metric of vulnerability, the map also captures high vulnerability areas in watersheds with poorly developed road networks. This fact is evidenced by the Ucayali and Marañon watersheds, which exhibit some of the highest mean values of vulnerability (only lower than Tocantins) although most of their road networks are restricted to headwater regions. Both watersheds cover large parts of the tropical Andes in Peru and Ecuador, with lowland Amazonia being the natural next development frontier, with threats to the biome projected to intensify in the next years (Fig. 3.4).

Existing roads in the south-eastern watersheds (Tocantins, Xingu, Tapajós and Madeira) provide a template for agricultural expansion, as is the case in any other parts of Amazonia. The long string of cities connected by many kilometres of roads in those watersheds, connect them with economically more developed extra-Amazonian regions. This places these watershed in a bleak future scenario if effective conservation actions are not taken by national governments. This cluster of watersheds host the Amazonian 'Arc of Deforestation', the world's largest tropical agricultural frontier (Nepstad et al. 1999; Laurance et al. 2001; Morton et al. 2006). Assuming the rapid expansion of the soybean frontier (Fearnside 2001), it is the most likely region to be the focus of intensive deforestation in the future (Laurance et al. 2002) (Fig. 3.4). The high vulnerability pattern encountered in the Madeira headwaters (Fig. 3.3) represents consolidated soy plantations and cattle ranches in Bolivia (Kruijt & Leaders 2014). Given the similarities in agricultural activities and the roads connecting these areas with the Deforestation Arc, there is a real possibility of both areas merging, creating a larger deforestation zone reaching beyond Brazil. The future escalation in vulnerability forecasted for the Purus and Juruá watersheds (Fig. 3.4) is also associated with the road network in the south-eastern watersheds and links between with developed consumer markets elsewhere.

Another evident spot of high vulnerability in our map is located in the Rio Branco watershed, which is mostly within the Brazilian state of Roraima. Deforestation in this watershed is mainly associated with the creation and strengthening of settlement projects and associated infrastructure, in particular the construction and paving of access roads (Barni et al. 2015). Despite the clear socio-environmental impacts that these activities entail, they count on the support of the state government (Diniz & Santos 2005) and have the potential to become a larger problem in years to come (Peres & Schneider 2012). In Amazonia, vulnerability spreads as an inward-centred movement. As a result, restricting the emergence of new high vulnerability areas in the Roraima state is a precaution not just to protect the Branco watershed but all watersheds located north of the Amazonas River (Fig. 3.4). All northern watersheds drain the Guiana Shield, with the Negro and Uatumã having black-water drainages, while the other watersheds consist of clear-water rivers (Goulding et al. 2003). The increase in vulnerability levels in those watersheds is alarming due to the low intrinsic resilience of oligotrophic regions (Henderson 1990) with perhaps irreversible damage caused in those areas.

We have presented a worrisome scenario of the vulnerability status of most watersheds considered here. While we included important human-related threats in our analysis, due to the lack of reliable data for all Amazonia, we did not include all activities with important negative impacts on biodiversity such as mining (Alvarez-Berríos & Mitchell Aide 2015; Mendes-Oliveira et al. 2013), hydroelectric dams (Benchimol & Peres 2015; Palmeirim et al. 2014), oil and gas exploitation (Yapa & Tao Shen 1994; O'Rourke & Connolly 2003), illegal logging (Laurance et al. 2001) and overfishing/overhunting (Alho et al. 2015; Peres et al. 2006). Including these activities would result in most areas being even more vulnerable in our map. Even disregarding these activities, few of the easily accessible places in our map have low vulnerability. This indicates that low levels of vulnerability in Amazonia are restricted to remote areas. Considering that several infrastructure projects plan of creating or expanding existing road networks, pre-emptive conservation actions have to be urgently considered before it is too late for the world's largest tropical forest.

3.4.3 Effects of administrative regions on watershed vulnerability

Regional administrative subdivision was a significant predictor of vulnerability ($\beta = 0.02$, $p < 0.000$), showing that watersheds would benefit from an integrated management approach across these regions. Each administrative region is vested with a particular political agenda and the urge for economic development can lead to resource overexploitation, which could be avoided if management and planning took place at the landscape scale. Our vulnerability analysis has indicated several perturbation hotspots in areas linking different municipal counties (or districts), suggesting that economic development of local to regional administrative regions can be “contagious” and increase an area’s vulnerability in the process. This leads us to believe that without a holistic approach to managing and planning conservation activities right across the biome, Amazonian ecosystems will be gradually degraded by a diffuse set of local threats generated throughout the biome.

3.4.4 Challenges and opportunities in watershed-based conservation planning in Amazonia

The idea of watersheds being managed in an integrated way is well developed in terms of water resources. The concept of integrated management of water resources (IMWR) emerged in the 1980s and became a new international paradigm around the 1990s (Hooper 2005; Biswas 2008), since then serving as guidance for water management plans in different countries (Braga & Lotufo 2008). IMWR seeks to manage watersheds by reconciling human activities and ecosystem processes and establishing socially acceptable trade-offs (Falkenmark 2004), which represents the major challenge in the most threatened Amazonian subregions. Despite the criticism concerning the feasibility of this approach in managing hydrological resources (Biswas 2008), we believe that a heuristic application of the concept can provide a comprehensive scale for Amazonian conservation planning. This would also allow different sectors of society to participate in the creation of biodiversity conservation plans (discussed in more detail below), thereby closing the current gap between available knowledge and applications on the ground (Knight et al. 2008).

The way watersheds are organized facilitate their application for the scheduling of conservation actions, whereby a list of priorities concerning which features should be protect first is developed to reduce biodiversity loss (Rodrigues & Brooks 2007; Pressey et al. 2007; Sarkar et al. 2006). Watersheds are fractal dimensions of rivers and require specific managed strategies at each of their different spatial scales. A major river contains several small watersheds that can still be segmented into ever smaller sub-watersheds. It is at the largest scale that policies should be defined; conservation actions should be taken (e.g. PA set asides, reforestation, land tenure organization); and long-term action plans should be developed. The smallest sub-watersheds can be used as building blocks of conservation units and as management areas.

The compartmentalized structure is not the only asset a watershed scale analysis brings to biodiversity conservation. In fact aligning PA boundaries with watershed water dividers would automatically favour some of the most important features when creating large PAs capturing heterogeneous landscape mosaics: habitat and species diversity, ecological sustainability, defensibility and associated costs. Amazonian forests are heterogeneous even at the smallest scales (Tuomisto et al. 1995) and to protect an entire watershed requires protecting several different habitats and ecological relationships within and between these habitats (Naiman et al. 2010). In Amazonia, several species rely heavily on lateral migration from upland (terra firme) forests to adjacent floodplains (Haugaasen & Peres 2007; Fernandes 1997; Castello 2008) and on longitudinal migrations from the mouth to headwaters of rivers (Peres 2005; Barthem & Goulding 1997). These ecological processes can only be fully protected if the entire watershed is protected. In addition, riparian areas are affected by all threats located upstream, so PAs that exclude headwater regions outside their boundaries jeopardise their own long-term viability.

How effectively a PA can be defended against external threats is a direct reflection of how effective these areas will be in protecting biodiversity (Bruner et al. 2001). By offering a few access points (often only one) watersheds can be protected by a small number of staff (Peres & Terborgh 1995), increasing the probability of successful protection in an environment where PAs are severely underfunded and understaffed. Watersheds can provide passive protection as well, since water dividers are often situated in steep terrains which are often inaccessible, thereby reducing deforestation rates immediately outside their domain, a trait shared by most PAs worldwide (Spracklen et al. 2015), avoiding the eventual isolation of those PAs from their forest matrix.

Floodplain areas in Brazil are legally defined as under the jurisdiction of the state or federal government, depending on the characteristics of the river. Using drainage dividers rather than navigable rivers as limits of PAs would increase the total floodplain area inside PAs, reducing the costs of land acquisition, which represents the largest cost of reserve implementation (Espirito Santo et al. 1991).

Any discussion on large-scale conservation strategies in Amazonia should consider the fact that its biological diversity remains poorly known (Peres 2005). Considering the high levels of threats the region is already facing it does not appear to be sensible to wait for more complete biodiversity inventories before creating new PAs. Therefore, reserve design in the biome should be done through coarse filter approaches (Hunter Jr. 1991). The use of watersheds as conservation planning units would therefore adequately fit this purpose.

The spatial alignment of protected areas and meso-scale watershed boundaries across Amazonia was proposed over two decades ago (Peres & Terborgh 1995). This concept was justified by design criteria to improve the defensibility of PAs and protection of seasonally flooded ecosystems and related ecological process. However, nothing has appreciably changed since this proposal in relation to conservation policies in Amazonia. In fact, there are few instances in which these design concepts have tangibly contributed to the adjustment of PAs network around the globe (Saunders et al. 2002). In addition, to our knowledge, there are no examples of the watershed concept being used to support biodiversity conservation planning unit at a national or biome level in any part of the world.

One of the reasons for the absence of conservation planning policies created at a watershed scale is the intractable governance of their management systems. Watersheds can extend over vast areas and embrace regions under different cultural, political, administrative and institutional contexts. This complicates the governance of an integrated management approach (Hooper 2005), further made difficult through issues of political cooperation wherever watersheds straddle national or other important jurisdiction boundaries (Wolf et al. 1999). However, although Amazonian watersheds are large, the results shown in Table 02 suggest that, with the exception of few cases, watershed scale biodiversity conservation in Amazonia can be conducted at the national level.

Building the political structures to conduct necessary actions is considered a major challenge to implement integrated watershed management initiatives (McDonnell 2008), but this should not be a problem in Amazonia. PAs have a crucial role in biodiversity conservation policies (Margules & Pressey 2000) and an internal reshuffle of the environmental national agencies with watersheds being considered the focal point of PA design and land use directives would represent a strong base for other conservation-related actions. Watersheds transcending neighbouring administrative districts could be protected under federal conservation units (CUs) with their management council counting on representatives from different localities, which already takes place in Amazonian forest reserves contained by two or more municipalities. In the case of transnational watersheds, different reserves could be created with an additional design. This is not without precedent as shown by the Parque Nacional Alto Purus (Peru) and the neighbouring Parque Estadual Chandless (Brazil) in the upper Rio Purus. This wider planning scale must be conducted with the participation of the diverse stakeholders representing different sectors of society.

The management of hydrological resources is already organised based on watershed boundaries in some countries (Brazilian law, EU Water Framework Directive) through the formation of management committees that deliberate the actions and the schedule of the plans developed for the watersheds. This format could be used as a blueprint for biodiversity conservation plans wherever political cooperation between different government agencies is possible. When most of a watershed is contained in a single country, national action plans could be followed by the other country containing a smaller portion of the watershed. Alternatively, in the worst case scenario (when the other country disagrees with the proposed policy) the largest part of the watershed would be protected under a particular strategy. The Madeira, Napo and Putumayo are the only three watersheds that contain less than 75% of their area inside a single country and their management would require a strong cooperation agreement between different countries.

Rather than a mere academic exercise, this paper aims to propose watersheds as a viable scale of analysis for biodiversity conservation planning within Amazonia. In some regions watersheds are the only meaningful geographic compartment of Amazonia and to some extent environmental agencies already take into account the internal geographic subdivisions based on waterways. This idea now needs to be expanded to consider major watersheds as central to planning exercises, trickling down to smaller watersheds, which become especially relevant for more local conservation actions. The overall concept is both simple and feasible, and would benefit not just the creation of biodiversity conservation policies but also the assessment of their outcomes. With minor adjustments and political will the concept we proposed here could be implemented, and represent a significant advance for conservation in Amazonia.

3.5 References

Ahmed, S.E. et al., 2014. Road networks predict human influence on Amazonian bird communities. *Proceedings of the Royal Society B: Biological Sciences*, 281(1795), pp.20141742–20141742. Available at: http://rspb.royalsocietypublishing.org/content/281/1795/20141742.abstract?etoc.

Alho, C.J.R., Reis, R.E. & Aquino, P.P.U., 2015. Amazonian freshwater habitats experiencing environmental and socioeconomic threats affecting subsistence fisheries. *Ambio*, 44(5), pp.412–425. Available at: http://www.ncbi.nlm.nih.gov/pubmed/25572836.

Alvarez-Berríos, N.L. & Mitchell Aide, T., 2015. Global demand for gold is another threat for tropical forests. *Environmental Research Letters*, 10(1), p.014006. Available at: http://stacks.iop.org/1748-9326/10/i=1/a=014006?key=crossref.6ccc347f82912ddd78f8fb396c95da22.

Arce-Nazario, J. a., 2007. Human landscapes have complex trajectories: reconstructing Peruvian Amazon landscape history from 1948 to 2005. *Landscape Ecology*, 22(S1), pp.89–101. Available at: http://link.springer.com/10.1007/s10980-007-9123-5.

Arponen, A. et al., 2005. The Value of Biodiversity in Reserve Selection: Representation, Species Weighting, and Benefit Functions. *Conservation Biology*, 19(6), pp.2009–2014. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00218.x.

Barni, P.E. et al., 2015. Deforestation and Forest Fires in Roraima and Their Relationship with Phytoclimatic Regions in the Northern Brazilian Amazon. *Environmental Management*, 55(5), pp.1124–1138. Available at: http://link.springer.com/10.1007/s00267-015-0447-7.

Barthem, R. & Goulding, M., 1997. *The catfish connection: ecology, migration, and conservation of Amazon predators*, Columbia University Press.

Bass, M.S. et al., 2010. Global Conservation Significance of Ecuador's Yasuní National Park A. Hector, ed. *PLoS ONE*, 5(1), p.e8767. Available at: http://dx.plos.org/10.1371/journal.pone.0008767.

Beissinger, S. & Westphal, M., 1998. On the use of demographic models of population viability in endangered species management. *The Journal of Wildlife Management*, 62(3), pp.821–841. Available at: http://www.jstor.org/stable/3802534 [Accessed November 2, 2010].

Benchimol, M. & Peres, C.A., 2015. Widespread Forest Vertebrate Extinctions Induced by a Mega Hydroelectric Dam in Lowland Amazonia M. K. Pandit, ed. *PLOS ONE*, 10(7), p.e0129818. Available at: http://dx.plos.org/10.1371/journal.pone.0129818.

Biswas, A.K., 2008. Integrated Water Resources Management: Is It Working? *International Journal of Water Resources Development*, 24(1), pp.5–22. Available at: http://www.tandfonline.com/doi/abs/10.1080/07900620801902918\nhttp://www.informaworld.com/openurl?genre=article&doi=10.1080/07900620701871718&magic=crossref\nhttp://www.tandfonline.com/doi/abs/10.1080/02508060802272812\nhttp://www.informaworld.com/openurl.

Boyd, C. et al., 2008. Spatial scale and the conservation of threatened species. *Conservation Letters*, 1(1), pp.37–43. Available at: http://doi.wiley.com/10.1111/j.1755-263X.2008.00002.x.

Braga, B.P.F. & Lotufo, J.G., 2008. Integrated River Basin Plan in Practice: The São Francisco River Basin. *International Journal of Water Resources Development*, 24(1), pp.37–60. Available at: http://www.tandfonline.com/doi/abs/10.1080/07900620701722978 [Accessed March 25, 2016].

Brandon, K.E. & Wells, M., 1992. Planning for people and parks: Design dilemmas. *World Development*, 20(4), pp.557–570.

Brooks, T., 2010. Conservation planning and priorities. In *Conservation Biology for All*. Oxford: Oxford University Press, pp. 199–219.

Bruner, A.G. et al., 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, 291(5501), pp.125–128. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.291.5501.125.

Cabeza, M. & Moilanen, A., 2001. Design of reserve networks and the persistence of biodiversity. *Trends in Ecology & Evolution*, 16(5), pp.242–248. Available at:

http://linkinghub.elsevier.com/retrieve/pii/S0169534701021255.

Caro, T. et al., 2004. Preliminary assessment of the flagship species concept at a small scale. *Animal Conservation*, 7(1), pp.63–70. Available at: http://doi.wiley.com/10.1017/S136794300300115X [Accessed March 25, 2016].

Caro, T.M. & O'Doherty, G., 1999. On the Use of Surrogate Species in Conservation Biology. *Conservation Biology*, 13(4), pp.805–814. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1999.98338.x [Accessed March 25, 2016].

Carvalho, G. et al., 2001. Sensitive development could protect Amazonia instead of destroying it. *Nature*, 409(6817), pp.131–131. Available at: http://www.nature.com/doifinder/10.1038/35051794.

Castello, L., 2008. Lateral migration of Arapaima gigas in floodplains of the Amazon. *Ecology of Freshwater Fish*, 17(1), pp.38–46. Available at: http://doi.wiley.com/10.1111/j.1600-0633.2007.00255.x [Accessed March 25, 2016].

Center for International Earth Science Information Network - CIESIN et al., 2011. Global Rural-Urban Mapping Project, Version 1 (GRUMPv1): Coastlines.

Cowling, R.. et al., 2003. A conservation plan for a global biodiversity hotspot—the Cape Floristic Region, South Africa. *Biological Conservation*, 112(1-2), pp.191–216. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0006320702004251.

Crouzeilles, R., Lorini, M.L. & De Viveiros Grelle, C.E., 2011. Applying Graph Theory to Design Networks of Protected Areas: Using Inter-Patch Distance for Regional Conservation Planning. *Brazilian Journal of Nature Conservation Research Letters Natureza & Conservação ABECO Handling Editor*, 9(2), pp.1–6.

Diniz, A.M.A. & Santos, R. dos, 2005. O vertiginoso crescimento populacional de Roraima e seus impactos socioambientais. *Cad Geografia*, 15(25), pp.23–44.

Dourojeanni, M., Barandiarán, A. & Dourojeanni, D., 2009. *Amazonía peruana en 2021: explotación de recursos naturales e infraestructuras:¿ qué está pasando?¿ Qué es lo que significan para el futuro?*, undación Peruana para la Conservación de la Naturaleza.

Eken, G. et al., 2004. Key Biodiversity Areas as Site Conservation Targets. *BioScience*, 54(12), p.1110. Available at: http://bioscience.oxfordjournals.org/cgi/doi/10.1641/0006-3568(2004)054[1110:KBAASC]2.0.CO;2.

Espirito Santo, C. do, Faleiros, A.A. & Fundaçao Pró-Natureza-FUNATURA, 1991. *Cost of implantation of conservation units in legal Amazonia*, Brasília, DF.

Eva, H. et al., 2005. *a Proposal for Defining the Geographical*, Office for Official Publications of the European Communities.

Falkenmark, M., 2004. Towards integrated catchment management: opening the paradigm locks between hydrology, ecology and policy-making. *International Journal of Water Resources Development*, 20(3), pp.275–281. Available at: http://www.tandfonline.com/doi/abs/10.1080/0790062042000248637.

Fearnside, P.M., 2015. Hidrelétricas na Amazônia brasileira: Questões ambientais e sociais. In D. Floriani & A. E. Hevia, eds. *América Latina Sociedade e Meio Ambiente: Teorias, Retóricas e Conflitos em Desenvolvimento*. Curitiba, Brasil: Editora da Universidade Federal do Paraná.

Fearnside, P.M., 2001. Soybean cultivation as a threat to the environment in Brazil. *Environmental Conservation*, 28(01). Available at: http://www.journals.cambridge.org/abstract_S0376892901000030 [Accessed March 25, 2016].

Fernandes, C.C., 1997. Lateral migration of fishes in Amazon floodplains. *Ecology of Freshwater Fish*, 6(1), pp.36–44. Available at: http://doi.wiley.com/10.1111/j.1600-0633.1997.tb00140.x [Accessed March 25, 2016].

Geist, H.J. & Lambin, E.F., 2002. Proximate Causes and Underlying Driving Forces of Tropical Deforestation. *BioScience*, 52(2), p.143. Available at: http://bioscience.oxfordjournals.org/cgi/doi/10.1641/0006-3568(2002)052[0143:PCAUDF]2.0.CO;2 [Accessed March 25, 2016].

Goulding, M., Barthem, R. & Ferreira, E., 2003. *The Smithsonian atlas of the Amazon*,

Grumbine, R.E., 1994. What Is Ecosystem Management? *Conservation Biology*, 8(1), pp.27–38. Available at: http://doi.wiley.com/10.1046/j.1523-

1739.1994.08010027.x [Accessed March 25, 2016].

Hansen, M.C. et al., 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, 342(6160), pp.850–853. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1244693 [Accessed March 20, 2016].

Haugaasen, T. & Peres, C.A., 2007. Vertebrate responses to fruit production in Amazonian flooded and unflooded forests. *Biodiversity and Conservation*, 16(14), pp.4165–4190. Available at: http://link.springer.com/10.1007/s10531-007-9217-z [Accessed March 25, 2016].

Henderson, P.A., 1990. Fish of the Amazonian Igapó: stability and conservation in a high diversity-low biomass system. *Journal of Fish Biology*, 37, pp.61–66. Available at: http://doi.wiley.com/10.1111/j.1095-8649.1990.tb05021.x [Accessed March 25, 2016].

Hooper, B.P., 2005. *Integrated river basin governance: learning from international experiences*, IWA publishing.

Hopkins, M.J.G., 2007. Modelling the known and unknown plant biodiversity of the Amazon Basin. *Journal of Biogeography*, 34(8), pp.1400–1411. Available at: http://doi.wiley.com/10.1111/j.1365-2699.2007.01737.x [Accessed March 19, 2016].

Hunter Jr., M.L., 1991. Coping with ignorance: the coarse-filter strategy for maintaining biological diversity. In K. Kohm, ed. *Balancing on the brink of extinction*. Washington, D.C.: Island press, pp. 266–281.

IBGE, 2000. Censo Demográfico 2000 - Resultados do universo. Available at: http://www.ibge.gov.br [Accessed March 20, 2015].

IBGE, 2010. Censo Demográfico 2010 - Resultados do universo. Available at: www.ibge.gov.br [Accessed June 14, 2015].

Junk, W.J., Bayley, P.B. & Sparks, R.E., 1989. The flood pulse concept in river-floodplain systems. T. Mailund, ed. *Canadian special publication of fisheries and aquatic sciences*, 106(1), pp.110–127. Available at: http://dx.plos.org/10.1371/journal.pone.0028909.

Junk, W.J. & Piedade, M.T.F., 2004. Status of knowledge, ongoing research, and research needs in Amazonian wetlands. *Wetlands Ecology and Management*, 12(6), pp.597–609. Available at: http://link.springer.com/10.1007/s11273-005-1767-9.

Knight, A.T. et al., 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology*, 22(3), pp.610–617.

Kress, W. et al., 1998. Amazonian biodiversity: assessing conservation priorities with taxonomic data. *Biodiversity & Conservation*, 7(12), pp.1577–1587.

Kruijt, B. & Leaders, A. work-package, 2014. *Impactos del clima y el uso del uso en los bosques tropicales de la Amazonía*,

Lambert, T.D. et al., 2014. Roads as barriers to seed dispersal by small mammals in a neotropical forest. *Tropical Ecology*, 55(2), pp.263–269. Available at: www.tropecol.com [Accessed March 25, 2016].

Laurance, W., 2001. Synergistic effects in fragmented landscapes. *Conservation Biology*, 15(6), p.1488. Available at: http://si-pddr.si.edu/dspace/bitstream/10088/1508/1/scb_intro.pdf [Accessed November 1, 2010].

Laurance, W. et al., 2001. The future of the Brazilian Amazon. *Science (Washington)*, 291(5503), pp.438–439.

Laurance, W.F. et al., 2002. Predictors of deforestation in the Brazilian Amazon. *Journal of Biogeography*, 29(5-6), pp.737–748. Available at: http://doi.wiley.com/10.1046/j.1365-2699.2002.00721.x.

Lindenmayer, D.B. et al., 2002. The Focal-Species Approach and Landscape Restoration: a Critique. *Conservation Biology*, 16(2), pp.338–345. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2002.00450.x [Accessed March 25, 2016].

Lopes, M.A. & Ferrari, S.F., 2000. Effects of Human Colonization on the Abundance and Diversity of Mammals in Eastern Brazilian Amazonia. *Conservation Biology*, 14(6), pp.1658–1665. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2000.98402.x.

Mace, G.M., Possingham, H.P. & Leader-Williams, N., 2007. Prioritising choices in conservation. In *Key Topics in Conservation Biology*. pp. 17–34.

Margules, C.R. & Pressey, R.L., 2000. Systematic conservation planning. *Nature*, 405(6783), pp.243–253. Available at: http://www.nature.com/doifinder/10.1038/35012251.

McDonnell, R.A., 2008. Challenges for Integrated Water Resources Management: How Do We Provide the Knowledge to Support Truly Integrated Thinking? *International Journal of Water Resources Development*, 24(1), pp.131–143. Available at: http://www.tandfonline.com/doi/abs/10.1080/07900620701723240.

McPherson, M. et al., 2008. GIS-based modeling of environmental risk surfaces (ERS) for conservation planning in Jamaica. *Journal of conservation planning*, 4, pp.60–89.

Mendes-Oliveira, A.C. et al., 2013. Efeitos antrópicos sobre comunidades de pequenos mamíferos não-voadores na amazônia brasileira. In A. C. Mendes-Oliveira & C. Miranda, eds. *Os Pequenos Mamíferos Não Voadores da Amazônia Brasileira*. pp. 257–274.

Meyer, W.B. & Turner, B.L., 1992. Human population growth and global land-use/cover change. *Annual review of ecology and systematics*, 23, pp.39–61.

Millennium Ecosystem Assessment, 2003. *Ecosystems and human well- being: A framework for assessment*, Washington, D.C.: Island press.

Montgomery, D.R., 1999. PROCESS DOMAINS AND THE RWER CONTINTUUM. *Journal of the American Water Resources Association*, 35(2), pp.397–410. Available at: http://doi.wiley.com/10.1111/j.1752-1688.1999.tb03598.x [Accessed March 25, 2016].

Morton, D.C. et al., 2006. Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences*, 103(39), pp.14637–14641. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.0606377103 [Accessed March 25, 2016].

Naiman, R.J., Decamps, H. & McClain, M.E., 2010. *Riparia: ecology, conservation, and management of streamside communities*, Academic Press.

Nepstad, D. et al., 2001. Road paving, fire regime feedbacks, and the future of Amazon forests. *Forest Ecology and Management*, 154(3), pp.395–407.

Nepstad, D.C. et al., 1999. Large-scale impoverishment of Amazonian forests by logging and fire. *Nature*, 398(6727), pp.505–508. Available at:

http://www.nature.com/doifinder/10.1038/19066.

Noss, R.F. et al., 1996. Conservation Biology and Carnivore Conservation in the Rocky Mountains. *Conservation Biology*, 10(4), pp.949–963. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1996.10040949.x [Accessed March 25, 2016].

O'Rourke, D. & Connolly, S., 2003. Just oil? The distribution of environmental and social impacts of oil production and consumption. *Annual Review of Environment and Resources*, 28(1), pp.587–617.

Oliveira, T.G. de, 2006. Research in terrestrial Carnivora from Brazil: current knowledge and priorities for the new millennium. In R. G. Morato et al., eds. *Manejo e conservação de carnívoros neotropicais*. São Paulo, Brazil: Edições IBAMA, pp. 39–45.

Palmeirim, A.F., Peres, C.A. & Rosas, F.C.W., 2014. Giant otter population responses to habitat expansion and degradation induced by a mega hydroelectric dam. *Biological Conservation*, 174, pp.30–38.

Parry, L. et al., 2010. Rural-urban migration brings conservation threats and opportunities to Amazonian watersheds. *Conservation Letters*, 3(4), pp.251–259. Available at: http://doi.wiley.com/10.1111/j.1755-263X.2010.00106.x.

Parry, L. & Peres, C.A., 2015. Evaluating the use of local ecological knowledge to monitor hunted tropical-forest wildlife over large spatial scales. *Ecology and Society*, 20(3). Available at: http://www.ecologyandsociety.org/vol20/iss3/art15/.

Peres, C. a, Barlow, J. & Laurance, W.F., 2006. Detecting anthropogenic disturbance in tropical forests. *Trends in ecology & evolution (Personal edition)*, 21(5), pp.227–9. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16697907.

Peres, C. a., 2001. Synergistic Effects of Subsistence Hunting and Habitat Fragmentation on Amazonian Forest Vertebrates. *Conservation Biology*, 15(6), pp.1490–1505. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2001.01089.x.

Peres, C. a., 2005. Why We Need Megareserves in Amazonia. *Conservation Biology*, 19(3), pp.728–733. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00691.x.

Peres, C. a. & Terborgh, J.W., 1995. Amazonian Nature Reserves: An Analysis of the Defensibility Status of Existing Conservation Units and Design Criteria for the Future. *Conservation Biology*, 9(1), pp.34–46. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1995.09010034.x.

Peres, C.A. & Schneider, M., 2012. Subsidized agricultural resettlements as drivers of tropical deforestation. *Biological Conservation*, 151(1), pp.65–68.

Pressey, R.L. et al., 2007. Conservation planning in a changing world. *Trends in Ecology & Evolution*, 22(11), pp.583–592.

PRESSEY, R.L. & BOTTRILL, M.C., 2008. Opportunism, Threats, and the Evolution of Systematic Conservation Planning. *Conservation Biology*, 22(5), pp.1340–1345. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2008.01032.x [Accessed March 25, 2016].

Ricketts, T.H. et al., 2005. Pinpointing and preventing imminent extinctions. *Proceedings of the National Academy of Sciences*, 102(51), pp.18497–18501. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.0509060102 [Accessed March 25, 2016].

Roberge, J.-M. & Angelstam, P., 2004. Usefulness of the Umbrella Species Concept as a Conservation Tool. *Conservation Biology*, 18(1), pp.76–85. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2004.00450.x.

Rodrigues, A.S.L. & Brooks, T.M., 2007. Shortcuts for Biodiversity Conservation Planning: The Effectiveness of Surrogates. *Annual Review of Ecology, Evolution, and Systematics*, 38(1), pp.713–737. Available at: http://www.annualreviews.org/doi/abs/10.1146/annurev.ecolsys.38.091206.095737.

Roulet, M. et al., 1999. Effects of Recent Human Colonization on the Presence of Mercury in Amazonian Ecosystems. *Water, Air, and Soil Pollution*, 112(3/4), pp.297–313. Available at: http://link.springer.com/10.1023/A:1005073432015.

Sanderson, E.W. et al., 2002. A conceptuals model for concervation planning based on landscape species requirements. *Landscape and Urban Planning*, 58, pp.41–56. Available at: Sanderson_E-W_et-al_2002 a conceptual model for concervation.pdf.

Sarkar, S. et al., 2006. Biodiversity Conservation Planning Tools: Present Status and Challenges for the Future. *Annual Review of Environment and Resources*, 31(1), pp.123–159. Available at: http://www.annualreviews.org/doi/abs/10.1146/annurev.energy.31.042606.085844 [Accessed March 25, 2016].

Saunders, D.L., Meeuwig, J.J. & Vincent, A.C.J., 2002. Freshwater Protected Areas: Strategies for Conservation. *Conservation Biology*, 16(1), pp.30–41. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2002.99562.x [Accessed March 25, 2016].

Schill, S. & Raber, G., 2012. Protected Area Tools (PAT) for ArcGIS software. Available at: http://www.gispatools.org.

Shafer, C.L., 1999. National park and reserve planning to protect biological diversity: some basic elements. *Landscape and Urban Planning*, 44(2-3), pp.123–153. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0169204698001157.

Soares-Filho, B.S. et al., 2006. Modelling conservation in the Amazon basin. *Nature*, 440(7083), pp.520–523. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16554817.

Spracklen, B.D. et al., 2015. A Global Analysis of Deforestation in Moist Tropical Forest Protected Areas R. Zang, ed. *PLOS ONE*, 10(12), p.e0143886. Available at: http://dx.plos.org/10.1371/journal.pone.0143886.

Tuomisto, H. et al., 1995. Dissecting amazonian biodiversity. *Science*, 269(5220), pp.63–66. Available at: http://www.sciencemag.org/cgi/content/abstract/sci;269/5220/63 [Accessed November 2, 2010].

Vannote, R.L. et al., 1980. The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences*, 37(1), pp.130–137. Available at: http://www.nrcresearchpress.com/doi/abs/10.1139/f80-017 [Accessed March 25, 2016].

Wilson, K. et al., 2005. Measuring and Incorporating Vulnerability into Conservation Planning. *Environmental Management*, 35(5), pp.527–543. Available at: http://link.springer.com/10.1007/s00267-004-0095-9.

Wilson, K. a., Carwardine, J. & Possingham, H.P., 2009. Setting Conservation Priorities. *Annals of the New York Academy of Sciences*, 1162(1), pp.237–264.

Available at: http://doi.wiley.com/10.1111/j.1749-6632.2009.04149.x.

Wolf, A.T. et al., 1999. International River Basins of the World. *International Journal of Water Resources Development*, 15(4), pp.387–427. Available at: http://www.tandfonline.com/doi/abs/10.1080/07900629948682.

Yapa, P.D. & Tao Shen, H., 1994. Modelling river oil spills: a review. *Journal of Hydraulic Research*, 32(5), pp.765–782. Available at: http://www.tandfonline.com/doi/abs/10.1080/00221689409498713.

Chapter 4

Basin-wide conservation planning in Amazonia: degree of protection fails to match anthropogenic threats

To be submitted as:

Teles, D., Peres, C. A., Basin-wide conservation planning in Amazonia: degree of protection fails to match anthropogenic threats.

4.1 Introduction

Protected areas (PAs) comprise the cornerstone of global scale biodiversity conservation (Terborgh & Schaik 2002). Whether they inhibit deforestation (Nepstad et al. 2006; Andam et al. 2008), avoid species extinctions (ter Steege et al. 2015), maintain ecological services (Gaston et al. 2008), protect traditional livelihoods and cultures (Naughton-Treves et al. 2005; Constantino et al. 2012) or serve as pivotal components of biodiversity conservation policies (Margules & Pressey 2000), PAs are critical in shaping the future of Earth's biodiversity. PAs currently cover over 32,000,000 km^2, encompassing 14.0% of the world's terrestrial ecosystems (Deguignet et al. 2014), rapidly approaching the target set by the Convention on Biological Diversity (CBD) in their strategic plan for 2011-2020 (Aichi Biodiversity Target 11 - at least 17 per cent of terrestrial and inland water areas should be under protection by 2020). However, the geographic position and representativeness of PAs has been widely questioned, particularly in the tropics (Joppa & Pfaff 2009; Joppa et al. 2013; Knight et al. 2008; Prendergast et al. 1999; Cowling et al. 2004).

Considering the rapid declines in biodiversity worldwide (Butchart et al. 2010), actions to supress the spread of mounting biodiversity threats, such as PA allocation, are urgently required. However, regional scale PA distribution should be as efficient as possible given the limited conservation resources available (James et al. 1999). Although habitat conversion rates are generally lower within PA boundaries than in their surrounding areas (Nepstad et al. 2006; Bruner et al. 2001; Nelson & Chomitz 2011), it is difficult to assess their effectiveness in protecting the biodiversity of any given region due to their inherently biased distribution (Joppa & Pfaff 2010). PAs are often created in areas of low agricultural potential and far from population centres (Joppa & Pfaff 2009), where they typically benefit from passive protection by virtue of remoteness and low feasibility of economic development options (Soares-Filho et al. 2006; Peres

2001a). Conversely, conservation interventions could be better allocated to areas that are more directly threatened at present, where those investments would be least redundant and most urgent (Master 1991; Wilson et al. 2009; Pfaff et al. 2015). This creates a site selection dilemma in designing nature reserve networks whereby the optimal allocation of PAs should be governed by both costs (e.g. land acquisition) and benefits (e.g. avoided biodiversity loss).

Over 38% of the Greater Amazon Basin has already been set aside under Strictly Protected (SP) areas, Sustainable Use (SU) reserves and Indians Lands (IL) (Deguignet et al. 2014). Creating a robust PA network has been the focus of biodiversity conservation strategies across Amazonia (Soares-Filho et al. 2006), whereby geographic priority areas for conservation have been pinpointed based on centres of species endemism (Mayr & O'Hara 1986), coarse-filter approaches such as broad classifications of vegetation types (Olson et al. 2001; Ferreira et al. 2001), and regional conservation planning workshops involving hundreds of experts (Rylands & Brandon 2005). As a result, a large proportion of Amazonian forests and non-forest biomes have been proposed as conservation priority areas, yet there are no basin-wide assessments of the major determinants of how existing reserves have been selected and designed. Although conservation gap analyses performed across Amazonia (Fearnside & Ferraz 1995; Rodrigues et al. 2004) have highlighted some of the weaknesses in the existing PA distribution, a comprehensive assessment of how the current reserve network relates to the threats they face remains unavailable.

Natural ecosystems cannot be viewed as closed and static entities (Pickett et al. 1992), so greater emphasis in biodiversity conservation policies needs to be placed on preserving the full integrity of biological diversity and ecosystem processes (Angermeier & Karr 1994). These must be assessed in a regional context and protected by an integrated network of PAs ensuring the capture of key ecosystem targets (Barrett & Barrett 1997). In Amazonia the most adequate mesoscale to perform regional assessments and conservation planning is represented by major watersheds (CHAPTER 3). As such, given the high levels of anthropogenic habitat disturbance already confronting several parts of Amazonia (CHAPTER 3) and realistic long-term scenarios of growing threats forecasted for some watersheds (Soares-Filho et al. 2006), it is critical to implement efficient basin-wide conservation planning to understand how levels of protection and vulnerability relate to one another both within and between major watersheds.

The objective of this chapter is to evaluate how the existing 'safety net' of Amazonian protected areas has adjusted to subregional levels of vulnerability to anthropogenic habitat disturbance occurring within major watersheds. To achieve this, we focused on two major tasks. We first quantified the levels of protection and vulnerability status of each major river basin, classifying them according to both threat and protection criteria. Second, we examined the degree to which basin-scale vulnerability was spatially correlated with levels of protection for different major categories of reserve management. We also examined the recent history of protected area creation (2000 - 2015) right across Amazonia to assess the degree to which variables describing the degree of human disturbance are related to the geographic cover of PAs throughout the Amazonian biome. Finally, based on these findings, we consider the current protection status and future conservation prospects of all major Amazonian watersheds to inform the policy of reserve design and implementation in the world's largest tropical forest domain.

4.2 Methods

4.2.1 Study region

Our study region comprises the entire hydrologic domain of the Amazon River (sensu lato) and two adjacent watersheds (Araguari and Tocantins) that are part of the ecoregional domain of Amazonia (M Goulding et al. 2003). This area was segmented into 23 domains representing major watersheds (Fig. 4.1), collectively encompassing >6,580,000 km2, including 83% of the formally recognized phytogeographic boundaries of Amazonia (Eva et al. 2005). All of these major drainages flow directly into the Amazon (= Solimões) River, except for the Araguari, Tocantins, and Branco river basins. We decided to subdivide the Branco from the Negro watershed despite their direct hydrologic connectivity in the Guianan Shields, due to their very divergent geochemical and human development characteristics. All areas located along the immediate neighbourhood of the Amazon River but which do not belong to any of our 23 hydrologic subdomains were disregarded in this analysis. In aggregate this combined area accounted for only 8.5% of the study region (Fig. 4.1).

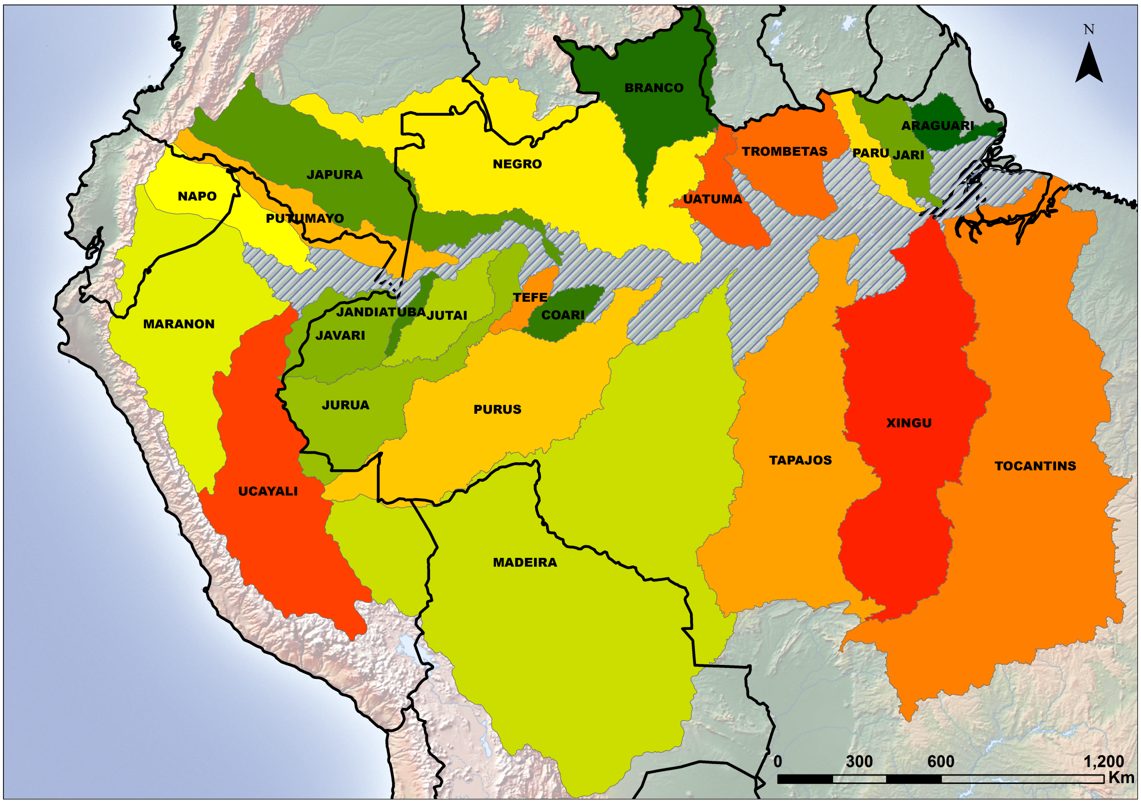


Fig 4. 1 - The 23 major Amazonian hydrologic domains examined in this study. Each domain was segmented on the basis of their hydrological boundaries and represents the aggregate drainage areas of all rivers and their tributaries. Shaded areas along the main channel of the Amazon River do not belong to any of the major watersheds and were excluded from the analyses. Black lines indicate the boundaries of the seven Amazonian countries included in the entire study region.

The analyses were conducted at the scales of both entire river basins and a total of 14,320 hexagonal cells of 500-km^2 each. Watershed delimitation was defined on the basis of hydrological shapefile (Ottobacias) sourced from ANA (Agência Nacional de Águas; http://www2.ana.gov.br/), the federal agency responsible for managing Brazilian water resources. These shapefiles delimit Amazonian watersheds based on the classification method created by Otto Pfafstetter, a Brazilian hydrogeologist who adapted the hierarchical hydrological concept to South American river basins. We subsequently redrew and corrected this classification in some portions of the Greater Amazon Basin to more accurately align all upland watershed dividers as visualised by a high-resolution (3 arc-seconds ≈ 90m) version of a global SRTM (Shuttle Radar Topography Mission) layer. Hexagonal cells were allocated to all watersheds using the Marxan software (Ball et al. 2009), but cells with less than 90% of their area inside the boundaries of the study region were excluded from the analyses.

4.2.2 Vulnerability status

Using the ARCGIS extension Protected Area Tools (Schill & Raber 2012), we created an Environmental Risk Surface (hereafter, ERS; see McPherson et al. 2008) as a measure of vulnerability to anthropogenic threats throughout the entire study area. To model this surface, we used information on deforestation between 2000 and 2013 (Hansen et al. 2013), and the following surrogates of human settlement distribution: human population density (HPD, see below), areas of persistent nocturnal lights (imagery and data processing by NOAA's National Geophysical Data Center - http://ngdc.noaa.gov/eog/dmsp/downloadV4composites.html), and paved and unpaved road networks (http://www.openstreetmap.org). Deforestation was modelled as a fixed positional value, whereas roads and night lights were a function of distances from sources (assuming decreasing values with radii of up to 5 km from roads and up to 20 km from night lights). HPD was assigned into six bins as ordinal (rank) categories (0, 0.01-1, 1.01-10, 10.01-25, 25.01-50, >50 persons/km^2). HPD values were extracted from a composite map derived from two different sources. We used data from the Brazilian Institute of Geography and Statistics (IBGE 2000) for the Brazilian territory, and the Global Rural-Urban Mapping Project (CIESIN et al. 2011) for all Amazonian countries other than Brazil. We used human population data standardized to the year 2000, and both the IBGE and GRUMP databases were combined into a single raster with spatial resolution of 30 arc-second (~1km). The final ERS value was then defined as the sum of the values of all overlapping threat layers.

4.2.3 Protected areas

We used the World Database on Protected Areas (WDPA 2015) as the main data source for PA location, size, shape, and management category. Overlaps between neighbouring PAs were examined and removed on a case-by-case basis, with the category of the most restrictive use being prioritised, while maintaining the overall original area of each PA. For legislative reasons, overlapping Indigenous Lands were always prioritised in relation to other PA categories. We restricted our data filter to PAs listed as "Designated" within the WPDA database, and PAs designated under IUCN categories I, II and III were subsequently grouped as Strictly Protected reserves, whereas those under categories IV, V and VI were grouped as Sustainable Use reserves.

We assigned values describing the total spatial extent (and proportion of the watershed area) set aside as protected areas and a mean vulnerability score to each hexagonal cell, which were subsequently grouped by major watershed. The relationship between vulnerability scores and extent of protection was then examined using Pearson's correlation coefficients.

To understand how allocation of conservation investments in terms of protected areas under different categories are distributed throughout Amazonia, we also plotted levels of vulnerability and protection against each other for all major watersheds.

4.2.4 Protected areas vs human populations

We assessed how PAs relate to the frontier of human population expansion across Amazonia by comparing the cumulative distribution function (CDF) curves of both degree of protection and human population size along the main axis of each of the 23 major Amazonian watersheds. The degree of large-scale convergence between both of these cumulative longitudinal patterns provide an indication of the degree to which PAs within watersheds have been allocated to either contain the spread of anthropogenic threats or systematically avoid densely settled areas. To do so, each watershed main axis (from its mouth to the headwaters) was segmented into sections of 50 km in length. CDFs therefore represent the cumulative values of each variable (human population and PAs) as a function of distance from the mouth of the main tributary near the Amazon River. HPD values were extracted from the same composite map used to generate vulnerability values, and PA values were calculated considering the proportion of each 50-km sector encompassed by reserves of any protection category.

4.2.5 Assessing PA responses to human disturbance and biodiversity uniqueness

We evaluated how human disturbance variables influenced the geographic positioning of recently created PAs (2000 – 2015) by fitting a generalized linear mixed model using Markov chain Monte Carlo (MCMC) methods for inference. Model predictors included the same variables used to create the vulnerability surface and a measure of biodiversity uniqueness of each 500-km^2 hexagonal cell, and PA presence was considered a binary response under a zero-inflated distribution. Hexagonal cells were thus nested within major watersheds, here considered as random factor in each model. The process of positional and landscape design of Amazonian PAs has changed considerably over the decades (Rylands & Brandon 2005). As the focus of this analysis is to understand the recent trends and dynamics of PAs creation, we restricted this analysis to PAs created after 2000. The dataset used consisted of 13,211 hexagonal cells distributed across the 23 major Amazonian watersheds. We also included habitat uniqueness in the analysis because high-uniqueness areas comprise a desirable asset in the design of PA networks (Cabeza & Moilanen 2001).

The binary level of protection was calculated as a response variable for each 1-km^2 pixels within each 500-km^2 hexagonal cell. Amazonia is widely variable in terms of HPD, including urban centers with over 2 million people and vast tracts of completely uninhabited areas, so HPD was $\log_{10}(x + 1)$ transformed for the analyses. HPD for each cell was then calculated as the average of all HPD values occurring within that sample unit. Distance to the nearest settlement or road (km) represents the mean distance of values within each hexagonal cell. Deforestation is represented as the percentage of forest cover that had been removed within each cell. Both the HPD and deforestation data refer to the year 2000 and the 2000-2013 period, respectively.

A habitat uniqueness value was assigned to each hexagonal cell on the basis of a normalized Relative Biodiversity Index (nRBI) calculated by the Protected Area Tools (Schill & Raber 2012). This index is calculated using an area-weighted function for which the distribution of biodiversity surrogates within each sample unit is related to the global distribution of the surrogate across the entire study region, thereby representing a measure of relative uniqueness or rareness of conservation targets.

As biodiversity surrogates, this index considered (1) the spatial distribution of all threatened bird and mammal species across Amazonia, the geographic range polygons of which were obtained from two sources (IUCN 2015; BirdLife International and NatureServe 2014), and (2) the ecoregion boundaries, representing maximum regional scale levels of species turnover, as defined by the World Wildlife Fund (Olson et al. 2001). nRBI was therefore calculated as following:

$$nRBI = \frac{RBI}{RAI}$$

Where RBI = abundance (planning unit) / abundance (study area); and RAI = area (planning unit) / area (study area).

The values obtained for each target (i.e. bird and mammal species, and ecoregions) were summed within each hexagonal cell and divided by the number of targets within the cells, deriving an aggregate nRBI value, which was then used as a measure of biodiversity uniqueness (see biodiversity uniqueness map in Appendix 4.1).

To evaluate how the geographic distribution of PAs managed under different designation categories can be explained by threat variables within each of the major Amazonian watersheds, MCMCglmms were performed considering just the human disturbance variables. We used the best linear unbiased prediction (BLUP) coefficient as an indicator of the degree to which within-watershed PA allocation over the last 15 years has been proactive or reactive in responding to anthropogenic threats.

Unless explicitly stated, all GIS processing and statistical analyses were performed using ARCGIS 10.1 (ESRI 2011) and R version 3.1.3 (R Core Team 2015), respectively.

4.3 Results

We delimited a total of 23 major river basins across the entire study area of 6,578,013 km^2. These ranged in size from the Jandiatuba basin (~15,000 km^2) to the Madeira basin (>1,460,000 km^2). These major watersheds are part of seven of the nine Amazonian countries, namely Brazil, Bolivia, Peru, Ecuador, Colombia, Venezuela and Guyana, extending from lowland forests at sea level to elevations of ~6,400 m in the tropical Andes.

4.3.1 Protection vs vulnerability

Protected areas cover more than 2,500,000 km^2 of the entire study area, most of which represented by indigenous territories (~42%), followed by sustainable-use (~36%) and strictly-protected reserves (~22%) (Fig 4.2). Although all categories of protected areas accounted for ~38% of the aggregate area across the 23 major river basins, the basin-scale degree of protection was highly heterogeneous, ranging from 90.8% in the Trombetas basin to only 0.5% in the Coari basin. As we shall see, this variation increases even further if we consider the numeric and area distribution of protected areas in terms of their officially denominated categories, particularly if we consider the extent to which major river basins are threatened by different human activities.

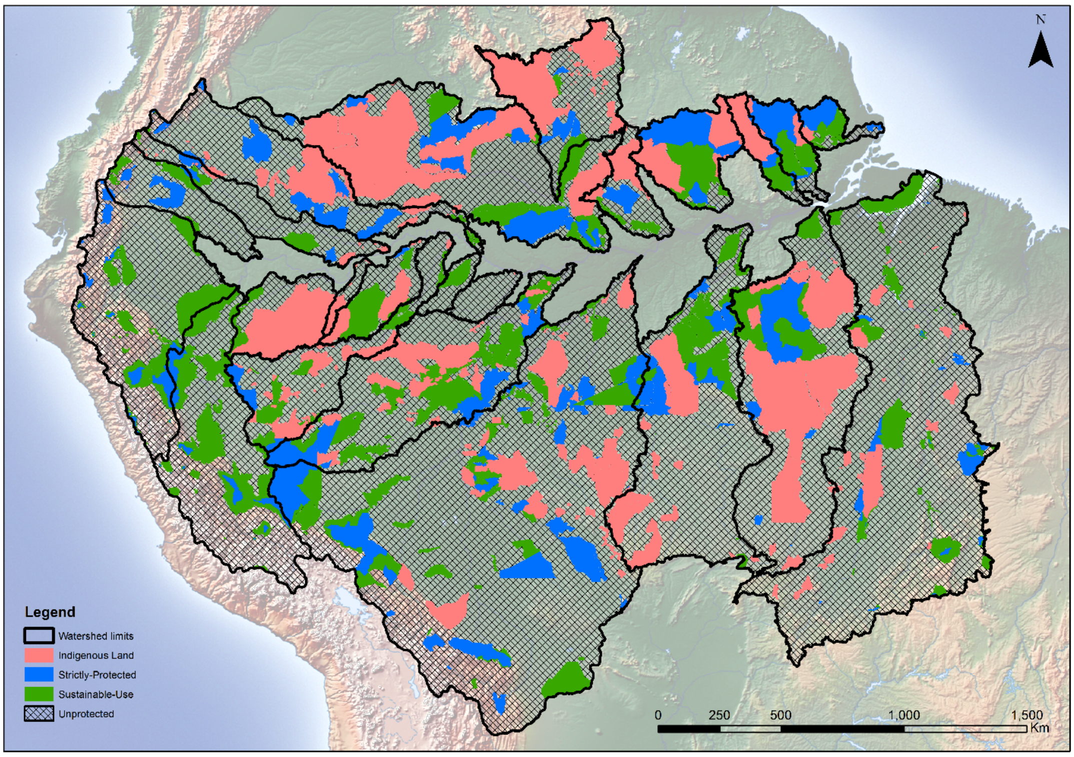


Fig 4. 2 - The three main classes of protected areas considered in this analysis, and their spatial distribution across all major Amazonian river basins. Strictly-Protected and Sustainable-Use Reserves are shown as blue and green polygons, respectively; Indigenous territories are shown as pink polygons. Black lines indicate the geographic boundaries of all 23 river basins.

We estimated the probability surface of potential human impact within each major watershed, as shown in a basin-wide vulnerability map (Fig. 4.3). This map presents the geographic variation in levels of threat, as defined by the Environmental Risk Surface (ERS) metric. Vulnerability scores across the entire study region ranged from 0 to 402 (mean =47.6; SD = 47.6). The Ucayali, Tocantins, Marañon and Madeira basins on average had the highest vulnerability values across all watersheds (69.96, 64.81, 63.06 and 56.87, respectively), whereas the Jari, Trombetas, Javari and Jutai were the least vulnerable (15, 21.41, 22.25 and 22.52, respectively). Beyond the large geographic asymmetry in the overall distribution of vulnerability, the most vulnerable areas were concentrated in the peripheral portions of the biome, particularly along broad transitions to seasonally-dry vegetation in the eastern, southern and southwestern Amazon. Another clear difference becomes evident between watersheds located either north or south of the main channel of the Amazon, which largely reflects the distribution of modern road access and infrastructure development (Chapter 3). The vulnerability surface is highly influenced by distance to roads, which was one of the variables used to generate the map coupled with deforestation and distance to settlements, which are also positively correlated with proximity to roads.

In general, the large-scale spatial distribution of PAs within watersheds does not track that of the human population (Fig. 4.4). Cumulative values of both overall human population size and areas under protection differ greatly within almost all watersheds, and are often strongly negatively correlated. River basins for which CDF curves exhibit a steeper slope represent areas where PA cover and human populations increase fastest over short longitudinal gradients. Therefore, the observed absence of congruence between these CDFs indicate that PAs are consistently allocated to less populated areas, regardless of their degree of vulnerability.

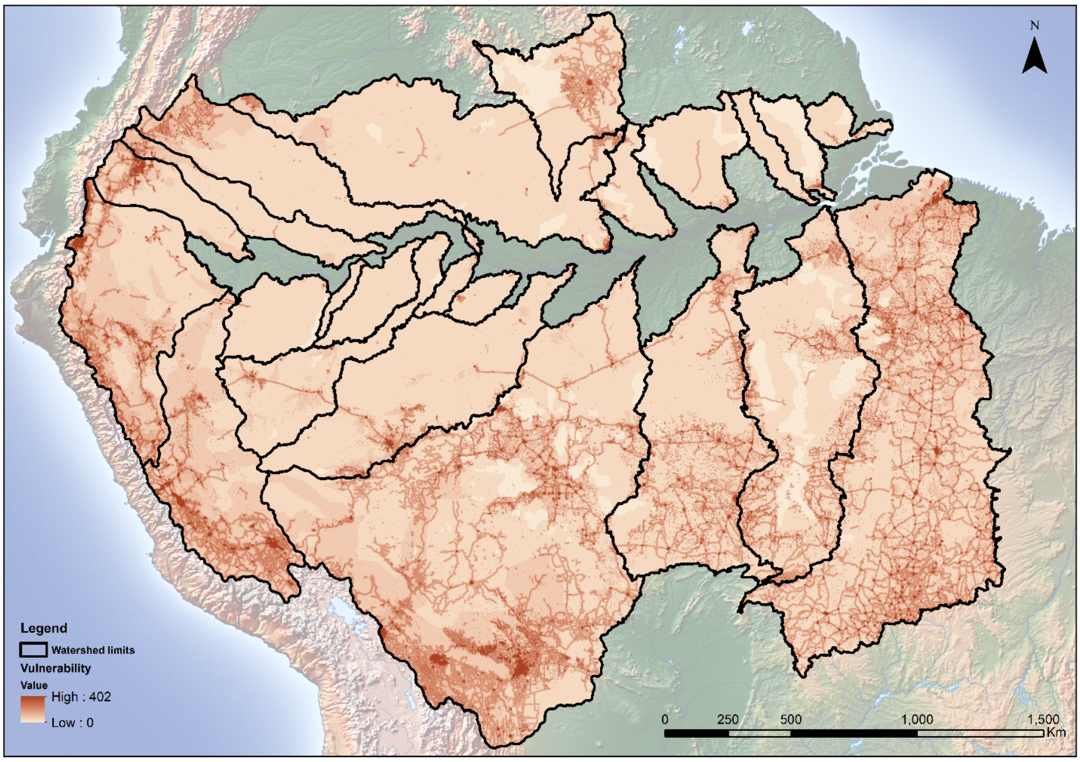


Fig 4. 3 - Geographic distribution of vulnerability to anthropogenic habitat disturbance throughout the study region. Vulnerability values are colour-coded from higher (darker) to lower (lighter) values associated with each ~1-km^2 pixel. Differences between vulnerability levels in watersheds located north and south of the Amazon River is noteworthy as road density to date remains far lower to the north of the Amazon. Black lines indicate the boundaries of river basins.

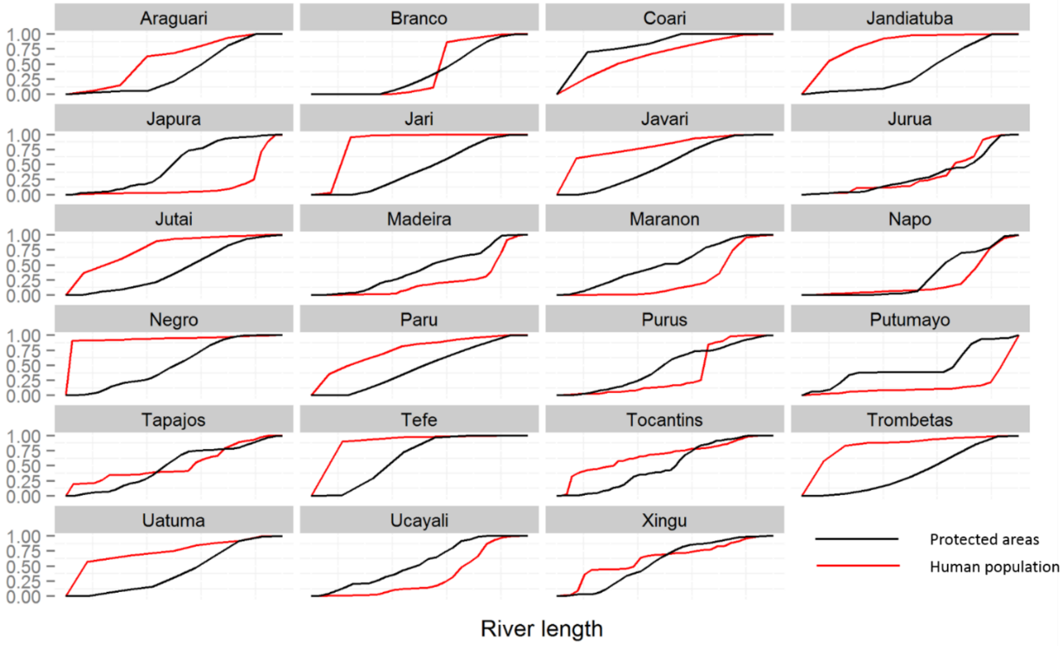


Fig 4. 4 - Comparisons of longitudinal patterns of protection and human occupation for the 23 major river basins considered in this study. CDF curves were created along a longitudinal gradient, from the mouth of the main river to its headwaters, with fluvial distances shown on the x-axis representing the entire relative length of the main watershed axis. Black and red lines indicate cumulative functions for protection area cover and total human population size, respectively.

There was a clear mismatch between the degree of vulnerability and total protection targets within major river basins, with negative correlations between these variables in 22 of the 23 river basins (Fig. 4.5). The only exception for which this relationship was weakly positively correlated was the Coari basin, where protected areas are virtually absent despite threats from growing oil and gas developments. Considering the 500-km^2 cells throughout the study region, there was an overall negative correlation between the percentage of area under protection and mean vulnerability ($r = -0.403$, $P < 0.001$, $N = 14,320$), again suggesting that PAs tend to be set aside in areas of low vulnerability. Furthermore, negative correlations between vulnerability and protection within watersheds are often moderate to strong, regardless of the degree of vulnerability.

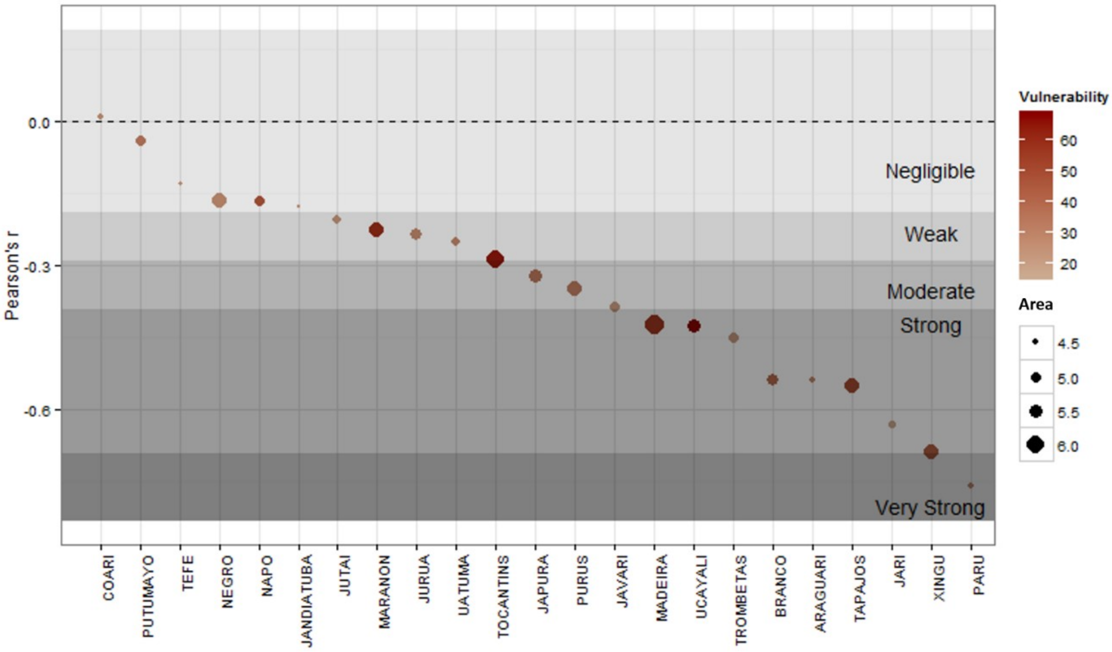


Fig 4. 5 - Pearson correlation values between levels of vulnerability and proportional extent of protection for 500-km^2 hexagonal cells within each of the 23 major Amazonian watersheds, ranked from the most positive to the most negative. Correlation strength is indicated by increasingly darker colour bands; solid circles are colour-coded according to the mean vulnerability value of the watershed, where darker indicates higher vulnerability. Symbol sizes are proportional to the total size (log) of the basin.

4.3.2 Watershed protection status

In addition to the protection-vulnerability mismatch at the scale of 500-km^2 cells, extent of protection fails to track vulnerability at the entire basin scale, whereby several high-vulnerability watersheds remain currently underprotected, whereas several low-vulnerability watersheds are overprotected (Fig. 4.6). For example, nine of the 11 basins with >50% of their area already protected are exposed to very low levels of vulnerability. Conversely, most high-vulnerability watersheds have set aside less than 37% of their areas into reserves of any category. This bias is largely consistent with patterns of physical access, human occupation, and agricultural productivity throughout the region, in that many sparsely-settled, low-nutrient white- and black-water river basins are overprotected, while several southern clear-water and upper Amazonian white-water basins succumbing to a myriad of environmental threats remain underprotected.

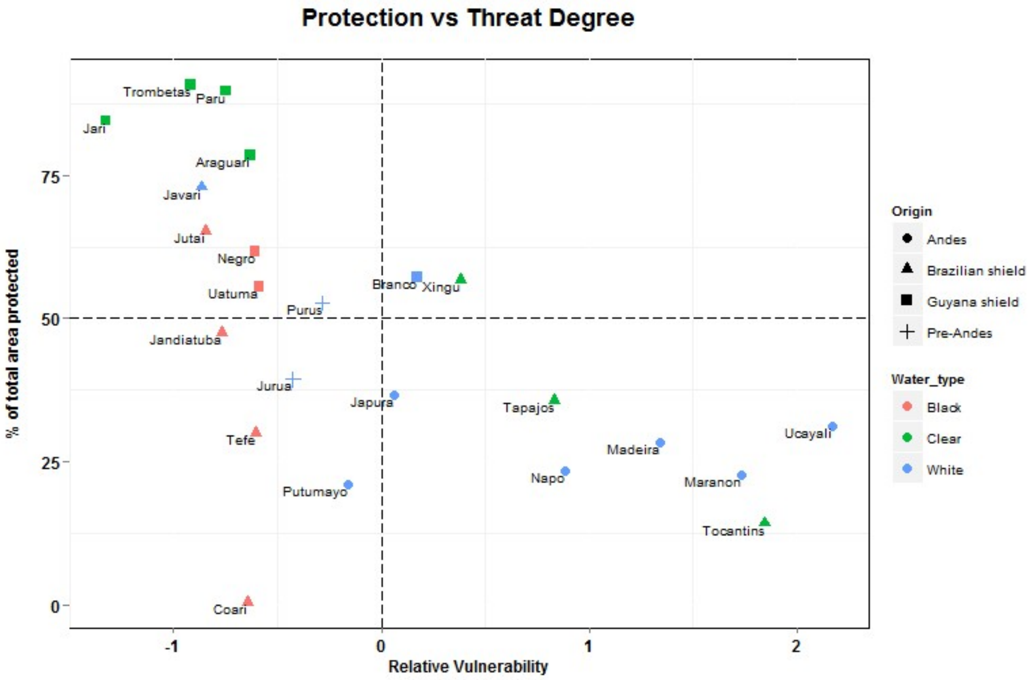


Fig 4. 6 - Relationship between relative vulnerability and extent of protection across the 23 major Amazonian watersheds. Degree of vulnerability was standardized (mean vulnerability values are zero-centered) to provide an easier visualisation across all watersheds. Symbol type and colour are coded according to the geological origin and prevailing water geochemistry of the overall river basin (*sensu* Goulding et al 2003).

This protection asymmetry becomes even more pronounced when we consider protected area cover by broad designation category. Indigenous territories account for 40.5% of the number and 42.1% of the total acreage of all Amazonian PAs considered here, so that strictly-protected and sustainable-use reserves represent a modest fraction of several high-vulnerability watersheds (Fig. 4.7). Yet the legislation ensuring the long term future of Indian Lands in most Amazonian countries is open to constitutional challenges, and depends on upholding of additional laws to guarantee their effectiveness. This, in practice, renders the long-term future of ILs less certain than officially sanctioned parks and reserves. Four of the 23 river basins did not contain any SP reserve, and SUs exceeded SPs in both total numbers and total area in 14 of the 23 watersheds, again confirming that most of the Amazonian protected acreage has been set aside as 'soft reserves' that may not preclude future biodiversity loss.

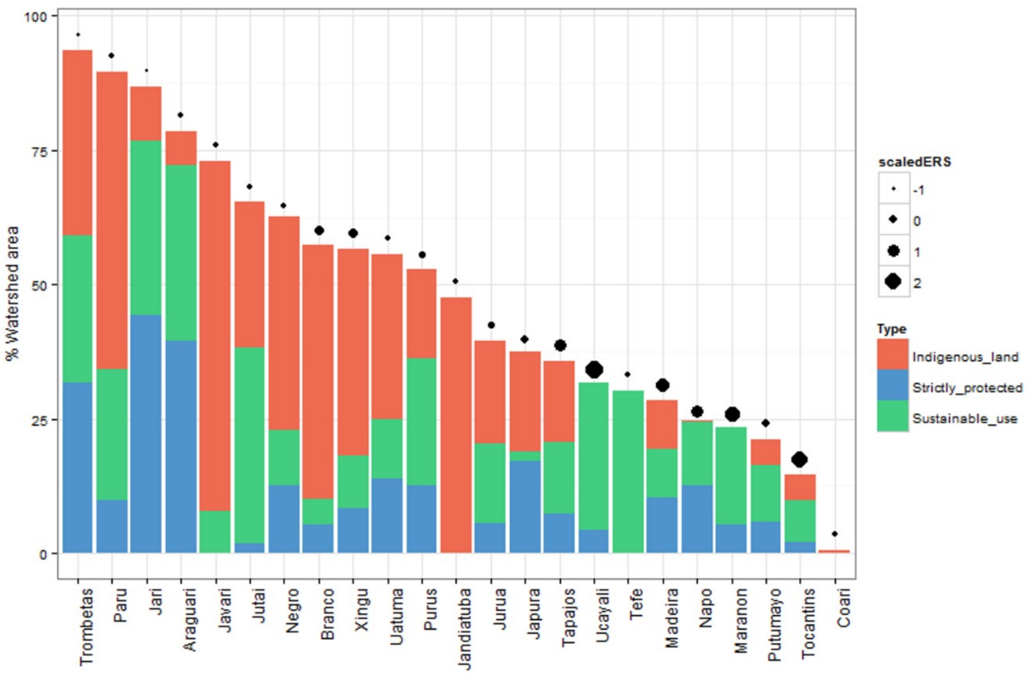


Fig 4. 7 - Breakdown of PAs according to their broad designation categories within the 23 major Amazonian watersheds. Indian Lands (ILs), Sustainable-Use reserves (SU) and Strictly-Protected (SP) reserves are shown in red, green, and blue, respectively. Vulnerability values are scaled for each watershed according to variable-sized circles above stacked bars (see Fig. 4.6). Major watersheds are rank-ordered from the most to least protected

Even if ILs are not taken into account, Amazonia already captures a large proportion of its domains within PAs, with 18 of the 23 watersheds exceeding the 17% minimum protection level proposed by the 11th Aichi Target of the Convention on Biological Diversity (CBD 2011) (Fig. 4.8). Sustainable-Use reserves, where many human interventions are legally permitted, are now the dominant class of PAs after ILs. IUCN Category V is the least restrictive of all Amazonian conservation units, but was the most abundant class of reserves in the three most vulnerable watersheds (Ucayali, Tocantins e Marañón). All but three major watersheds (Jari, Araguari and Trombetas) thus failed to meet the Aichi Target 11, if we redefine conservation targets in terms of strictly-protected reserves only, as advocated by some conservationists (Locke & Dearden 2005).

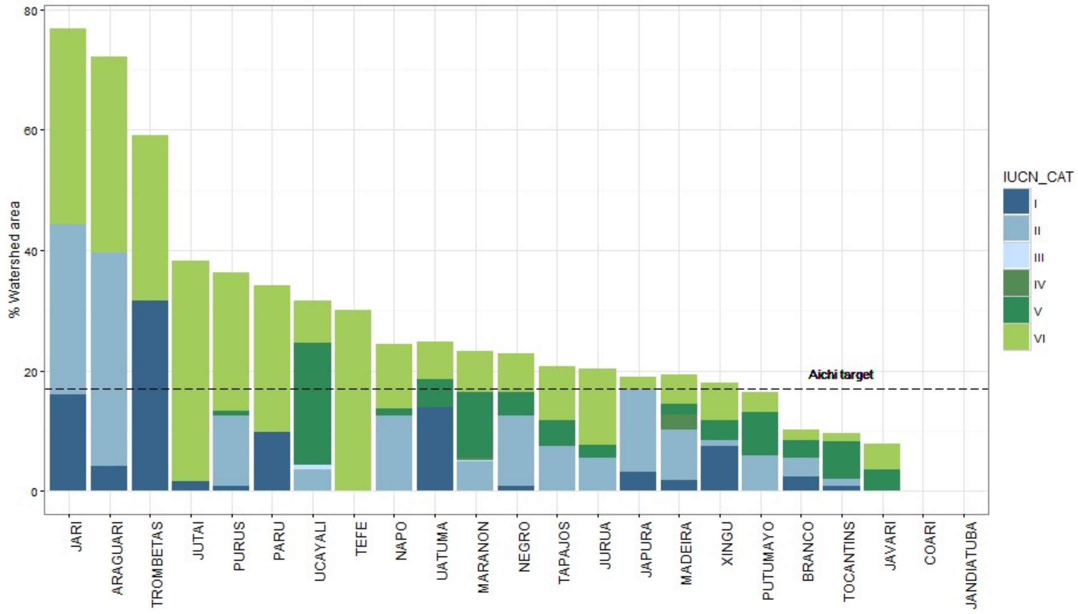


Fig 4. 8 - Percentage of total area protected (excluding indigenous territories) under different IUCN categories within the 23 major Amazonian watersheds. IUCN categories I–III (Strictly-Protected) and IV–VI (Sustainable-Use) are represented in blue and green, respectively. The dashed line indicates the minimum level of protection proposed by Aichi's Biodiversity Target 11 of the Convention on Biological Diversity (CBD) in their 2011-2020 strategic plan. Major watersheds are rank-ordered left to right from the most to the least protected in terms of their aggregate protected area (excluding ILs).

4.3.3 PA distribution, anthropogenic threats and habitat uniqueness

Both HPD and deforestation rates were consistently negatively related to PA cover within watersheds (pMCMC< 0.001 in all models; Fig. 4.9), regardless of protection category. Road distance has also been an important predictor of PAs positioning lately in SP reserves and IL, with no effect observed in SU reserves. Settlement distance presents effects in different direction, according category of protection, when predicting PAs positioning. Human settlements are allowed (even required) inside SU, explaining the negative relation with settlement distance. ILs are generally big and sparsely populated rendering several hex cells far from populated centres while we were not able to find any significative relation in the SPs. Areas of high biodiversity uniqueness has been constantly associated with SP and SU but not with IL. Differently of SU and SP reserves, the creation of ILs is driven majorly by historical occupation of indigenous people and for that high biodiversity uniqueness is not necessarily an asset required.

The direction and magnitude of PA distribution responses to human related threats for different watersheds across all three major reserve categories can be seen in Figure 4.10. In most watersheds, PAs under different management categories have avoided anthropogenic threats. Rio Branco and Jutai are the only watersheds were SPs have largely coincided with threatened areas, with all the SPs present in these watersheds being created by federal governments. ILs comprised is the PA category most likely to be positioned in highly vulnerable areas, with four watersheds exhibiting a positive relationship with human threat variables.

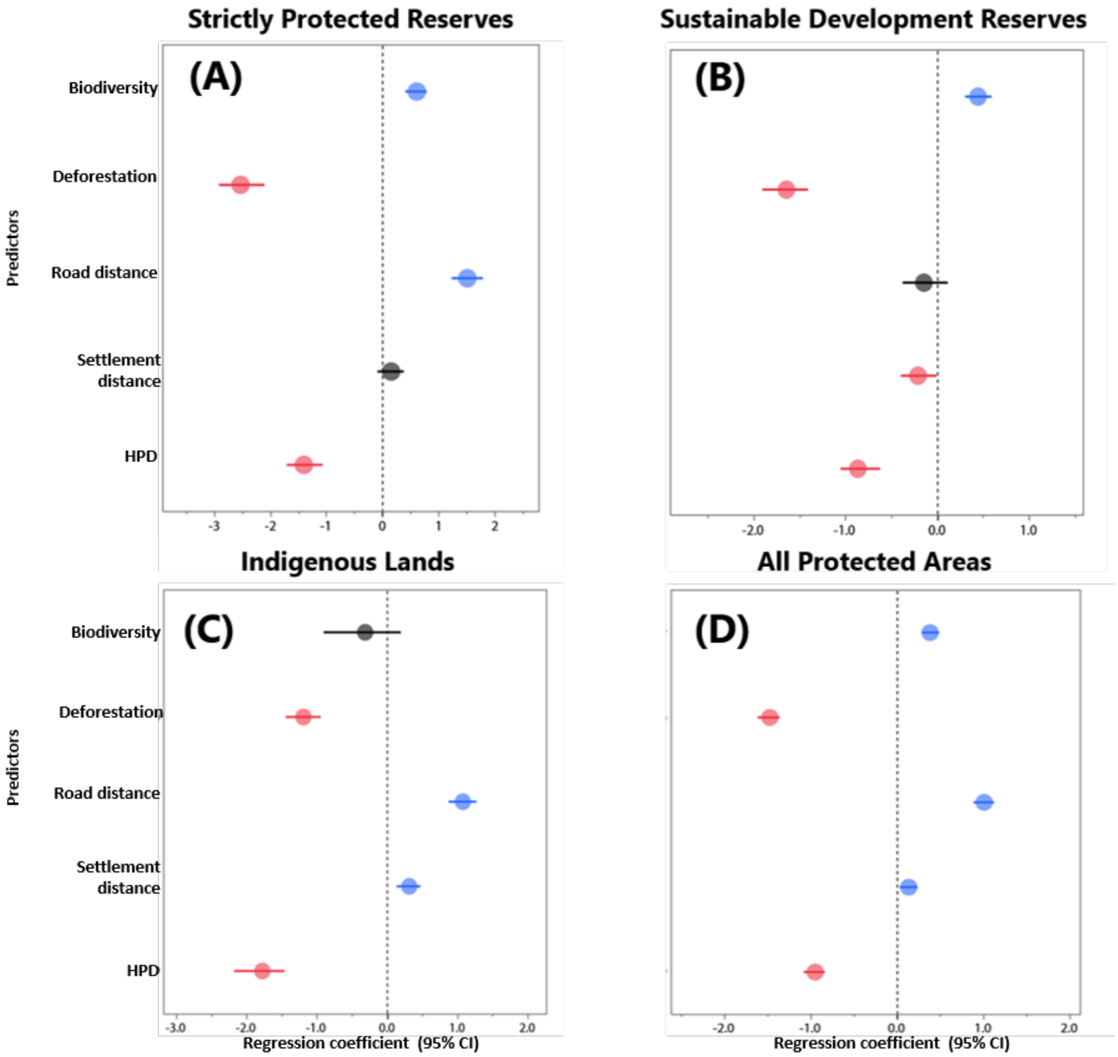


Fig 4. 9 - Estimates and 95% confidence intervals of the impact of human related threats indicators and biodiversity uniqueness in the positioning of PAs exhibited by category of protection being A) Strictly-protected reserves, B) Sustainable Development Reserves, C) Indigenous Land and D) results disregarding the category type. The results come from a Monte Carlo Markov Chain general linear mixed model of the presented variables as predictors of hexagonal cells' area covered by PAs, considering watersheds as random factor. Grey dots represents non-significative correlation between predictor and the response variable while red and blue represents negative and positive correlation respectively. The X axis shows 95% confidence intervals considering the values of the variables standardised.

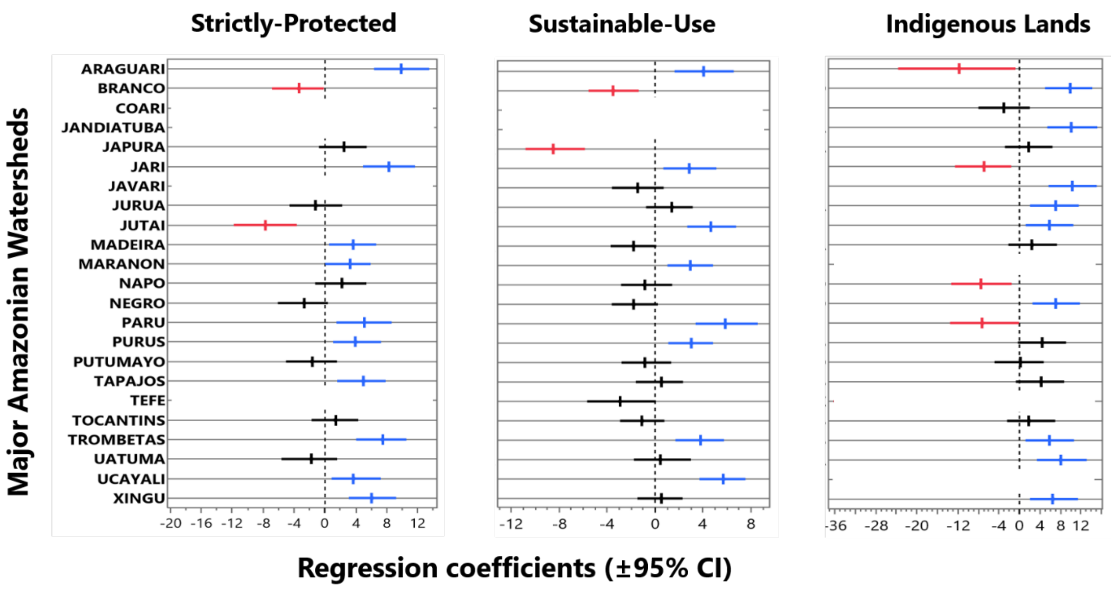


Fig 4. 10 - Best linear unbiased predictions (BLUPs) of PAs positioning in function of human related threats modelled by MCMCglmm for the major Amazonian watersheds. BLUP is the estimate of the random effect in the model and positive values indicate the random effect has same direction of the fixed effects for the model, conversely negative values indicate the fixed effects have opposite direction for this watershed than the encountered in the model. Our model presented a negative relation between PAs positioning (for all categories of protection) and the human population related threats. In this case the red values shown on the graph represent significative negative values of BLUP, indicating that the PAs existent in the watershed are positioned more frequently in areas of high vulnerability. Blue values indicate PAs being positioned far from human related threats, while black values indicate watersheds where human threat is not a good predictor of PA distribution. Results are presented by PA management category. Axis X represents 95% confidence interval of BLUP. The graph does not present values for watersheds lacking areas under the PA category

4.4 Discussion

We provided the first quantitative assessment of the degree to which all major river basins across lowland and upper Amazonia are either threatened or protected, or both, at present. Protected areas worldwide are typically positioned in areas of low human population and development pressure (Joppa & Pfaff 2009), and our study shows that Amazonia is no exception.

Modern human colonization along Amazonian watersheds exhibits a clear longitudinal pattern governed by the degree of river and road access from population centres (Parry et al. 2010; Chapter 3). The conspicuous lack of congruence between the distribution of human populations and protected areas within major watersheds (Fig. 4.4) indicates that the existing PA network was not designed to match the threats resulting from frontier expansion. Therefore, we show that despite the considerable political and financial investments allocated to a large number (and total area) of forest reserves, there is a profound mismatch in the geographic distribution of PAs, rendering them ineffective at containing the spread of human threats. This is typical of tropical regions lacking systematic conservation planning, where low-vulnerability areas are preferentially selected for new reserves largely because high-vulnerability areas are avoided, not least because of their high opportunity costs derived from present or future development prospects (Joppa & Pfaff 2009). We elaborate further on the risks of this strategy below.

At the 500-km^2 cell scale, there was negative correlation between vulnerability and degree of protection within all watersheds containing significant PA cover (Fig. 4.5), regardless of levels of vulnerability, suggesting that PA allocation respond to local to subregional political interests. Because PAs are often seen as an obstacle to economic development (Ferreira et al. 2014) siting them far away from centres of vulnerability appears to be the most expedient option to meet political targets, with our data suggesting that areas of economic potential are systematically avoided even if no imminent development plans are in place. In general, economically desirable areas (especially for mechanised agriculture) are sidestepped for PA allocation (Joppa & Pfaff 2009). However, the fact that even reasonably inaccessible areas appear to experience the same trend is a testament of the political influence of local economic sectors in the policy design of wildland conservation.

Brazil, which accounts for 66% of our study region, is a good example of incongruent Amazonian conservation policies. Despite the robust environmental and protected area legislation in the country (e.g Law 9985/00, Decree 4339/02, Decree 4340/02), recent political developments have weakened the PA system (Bernard et al. 2014; de Marques & Peres 2015), paving the way to higher deforestation rates (Soares-Filho et al. 2014). This contradicts global scale environmental policies ratified by Brazil and other Amazonian countries (Loyola 2014), casting shadows on their true political will in guarding the interests of protected areas.

4.4.1 Over-protection, under-protection and redundancy

The mismatch between degree of protection and vulnerability can be seen both within and across watersheds (Fig. 4.6). With over two-thirds of all Amazonian watersheds either over- or under-protected, conservation investments could be described as poorly distributed across the biome. Resources allocation to low-vulnerability areas under no imminent threat to local biodiversity can be considered inefficient and justifiable only if resources are super-abundant (Wilson et al. 2009), which is clearly not the case of Amazonia. Under the ubiquitous scenario of limited resources in tropical countries, this can be seen as a detrimental zero-sum game, whereby subregions in critical need of conservation investments rival one another, so that overprotection at one basin effectively suppresses PA expansion in other basins. Using conservation funds to allocate PAs to low-vulnerability areas thus subtracts resources that could otherwise be used in more critical areas.

PAs biased towards watersheds with low vulnerability, mostly located in northern Amazonia, also creates an additional redundancy problem (Walker 1992). The Amazon River is a strong geographic barrier for several taxa creating a marked species turnover along the >6,500 km east-west gradient, which becomes more species-rich farther west (Gascon et al. 2000; Ter Steege et al. 2003; Bass et al. 2010). Currently, the Amazonian PA network is biased to low-productivity regions with similar geological history (Fig. 4.6), whose watershed areas are often already >75% protected. In contrast, all Andean watersheds failed to reach a 40% protection threshold, which is significant considering that these are the most diverse watersheds (e.g. Bass et al. 2010). This redundancy towards more uniform biotas or lower diversity subregions is noteworthy given that complementarity is a key desirable criterion to be maximized by PA networks (Kati et al. 2004).

High biodiversity distinctiveness overlapping high levels of human stressors has been described as a "cruel twist of fate" (Pimm & Lawton 1998), but this can be largely pre-empted with rational proactive conservation planning. Otherwise, a systematic protection bias against vulnerable areas may lead to massive impoverishment of the Amazonia biota in the medium-term future.

4.4.2 Fool's gold

The notion of Amazonia as a well-protected biome is a false impression created by the raw figures on proportional PA cover. We have already noted that resources have been badly allocated in the absence of a systematic strategy to guide reserve deployment. However, the distribution of the PAs is not the only problem. When PAs are disaggregated into different protection categories (Fig. 4.7 and 4.8) the inherent frailties of an otherwise robust PA network becomes even more apparent.

A significant fraction of PAs in highly vulnerable watersheds (especially the Xingu and Branco) are indigenous territories, which misleads assessments of the effective area protected within different watersheds. Although indigenous territories and sustainable-use reserves inhibit deforestation and have a positive effect on Amazonian biodiversity conservation (Nepstad et al. 2006; Nelson & Chomitz 2011), they should be considered separately in any biodiversity conservation analysis. ILs have their own set of management criteria and will not necessarily serve the long-term interests of biodiversity conservation (Schwartzman & Zimmerman 2005; Zimmerman et al. 2001). ILs are designated to ensure the demographic and socio-cultural persistence of indigenous peoples (Davis & Wali 1994), with biodiversity protection as an incidental outcome.

Apart from Guyana, which contains no ILs within the boundaries of our study region, all other countries constitutionally ensure indigenous land rights and are signatories to the ILO 169 (ILO Convention 169 on Indigenous and Tribal Peoples in Independent Countries, 1989). However, indigenous land tenure and sovereignty principles are widely variable across countries. In general, the constitutional rules regulating aborigine territories are vague, confusing and ambiguous with most national constitutions requiring an additional set of laws to regulate the demarcation, creation and management of ILs (Ortega 2004). Whenever specific regulations are lacking, ILs are typically governed by the national civil code. They can therefore converge with nominally unprotected areas in terms of jurisdictional defences against external threats. These legislative uncertainties poses serious questions over the fate of ILs because of the realistic possibility that new legal instruments will eventually weaken indigenous territorial rights. This has already been santioned in Brazil by Law Decree 1775/96, which provides for civil administrative grievances, permitting non-indigenous peoples to challenge the identification and delimitation of ILs. Likewise, the 1993 constitution in Peru legally renders ILs vulnerable to private land purchases (Ortega 2004).

Some sectors of society also see indigenous territories as a threat to national security, an obstacle to advancing economic development, and a social injustice given the low population density of many ILs (e.g. indigenous people in Brazil occupy ~12% of the territory but represent only 2% of the population). While indigenous people in some countries, such as Bolivia and Colombia, have won battles against powerful sectors of society, there are many examples of how the law can be silent in protecting ILs against disputes with farmers and large infrastructure projects (Zhouri 2010).

Regardless of legal and ethical issues, it is both unsafe and unfair to transfer the burden of protecting biodiversity from the State to indigenous peoples within areas they retain to ensure their demographic and cultural survival. Although ILs can lead to potentially positive biodiversity conservation outcomes, conservation efforts should be co-developed with native Amazonians, explicitly considering their rights to self-determination and autonomy in deciding the ultimate fate of their own resources.

In terms of total area, Amazonian watersheds are still well protected on paper according to current international targets, even if we ignore the contribution of ILs (Fig. 4.8). However, the 17% Aichi target is merely a political practicality, rather than a minimum sufficiency threshold to ensure biodiversity conservation (Joppa et al. 2013). In fact, maintaining full complements of species, the hydrological viability of closed-canopy forest cover, and the complete integrity of ecological processes would require relatively intact forests in at least 50-60% of each drainage watershed (Peres 2005; Silva Dias et al. 2002). Yet all credible scenarios predict that Amazonian forest cover by 2050 will be restricted to either extremely remote hinterlands or well protected areas (Laurance et al. 2001; Soares-Filho et al. 2006). It is therefore imperative that post-frontier PAs are able to secure enough area across the optimal basin-wide distribution to ensure the persistence of biodiversity and key ecological process (Junk & Soares 2001).

Reasonable estimates of how much is enough to avoid wholesale extinctions are context-dependent with targets often set at around 50% or more of each biome (Soulé & Sanjayan 1998). Tropical forests are the most species-rich and complex terrestrial biomes, where average populations densities are typically low, so that vast areas are likely required to sustain in perpetuity even some of the most basic biophysical processes, such as water cycling. For example, Silva Dias et al.

(2002) indicate that maintaining the intricacies of the Amazonian rainfall regime requires that >70% of different subregions being retained as forest cover.

At present, only three major Amazonian watersheds have at least half of their areas set aside as SP or SU reserves, but no watershed reaches this threshold when only strictly-protected areas are considered (Fig. 4.8). Yet only strictly-protected reserves are considered capable of fully protecting entire plant and animal communities in tropical biotas (Terborgh 1999; Locke & Dearden 2005; Bruner et al. 2001). Therefore, the fact that about two thirds of all watersheds failed to reach 10% of their area under this class of reserves shows just how underprotected Amazonia still is.

Even very low human population densities can result in declines of highly sensitive game species (Levi et al. 2011), affecting the structure of whole vertebrate communities (Peres 2000). This emphasizes the critical importance of strictly-protected reserves to biodiversity conservation. Assessing levels of overhunting requires extensive fieldwork and cannot rely on remote sensing approaches (Peres et al. 2006). However, even subsistence hunting can be a problem in many parts of Amazonia (Valsecchi et al. 2014; Peres 2001b). Therefore, the future prognosis of sustainable-use reserves in retaining a full complement of wildlife species is questionable, given rapid population growth and affordable fire-weapon technology (Peres 2011). A possible option to ensure the population viability of harvest-sensitive species would be to enhance connectivity between SPs, SUs and ILs, thereby ensuring that demographic deficits (sinks) in overhunted areas are subsidised by vast neighbourhoods of unhunted source areas.

4.4.3 Human related threats as architects of PA design in Amazonia

All variables associated with our vulnerability map were good predictors of recent trends in PA positioning in Amazonia (Fig 4.9). The PA creation process has to overcome several stages of legal public consultation, so the negative effects of HPD on extent of protection is not surprising and confirms the results found at the watershed scale. Proximity to roads was also a key negative predictor of PA positioning for SPs and ILs. Distance of roads is related with predisposition of land use change (Laurance et al. 2002; Nepstad et al. 2001) and the fact of SP and IL being positioned far from the roads can indicate a strategy by governments to avoid areas with potential of human expansion when creating those areas. SU reserves are far less restrictive about land transformation than SP and IL representing a smaller threat to economic development or colonization plans what possibly explaining the absence of relevance of distance of roads predicting SU location. Indeed, such road repulsion effect was even stronger in the last decade than during the peak road-building period of 1970 – 2005, which militates against the best effectiveness of PAs in containing deforestation (Pfaff et al. 2015). Human settlements had varying effects on each of the protection categories, but those appear to be more related to the legal characteristics of the PAs than to their distribution. The positive effect of habitat uniqueness in recently created SP and SU reserves suggests that to some extent biodiversity conservation science has been incorporated into reserve network design. However, it is important to note that the biodiversity surrogates used here are simplistic and have little significance in local conservation actions. The likelihood of regional scale extinctions cannot be assessed by such surrogates, neither can they ensure the integrity of ecological processes within high uniqueness regions. In fact, due to our collectively disconcerting ignorance about large scale patterns of biotic distribution in Amazonia (Peres 2005), its PA network for now should be guided by a modular, coarse-filter conservation approach (Chapter 3), which could be regionally updated and fine-tuned by future biodiversity assessments.

Over the last 15 years, Amazonian PAs have rarely been positioned to contain the spread of human related threats (Fig 4.10). National, rather than state or local, government appears to be less resistant in placing PAs near densely-settled areas. For instance, ILs comprised the management category most positively associated with high-pressure indicators, and the only two watersheds in which SPs were spatially correlated with such areas (Branco and Jutaí) have no such protected areas created by state and municipal governments. Nonetheless, the pattern of reserve allocation observed throughout Amazonia represent a proactive, rather than reactive, approach of PA network expansion, whereby low-vulnerability areas are consistently prioritized (Brooks 2010). This strategy is both deceptive and risky, as we cannot predict the future biodiversity conservation performance of currently hinterland reserves once they are eventually confronted with severe threats. There are many examples of Amazonian PAs that are being downsized or even completely degazzeted once they face powerful competing economic interests; over 2,000,000 hectares of Amazonian protected areas are currently being assessed for downgrading, downsize or degazzetement by the Brazilian government (Bernard et al. 2014).

Protecting wildlands facing little or no development pressure at the expense of highly vulnerable sites may be politically appealing, since the latter are more expensive and typically involve greater public resistance to perceived economic straitjacketing imposed by PAs. Land prices are affected by both existing and speculative land use (Plantinga et al. 2002). Agricultural soil fertility, topographic slope, length of wet seasons, distance to major urban centers, road infrastructure, and land cover are all factors that can influence land prices in Amazonia. For example, transaction prices for a hectare of forest or pasture in Brazilian Amazonia can range over three orders of magnitude (DT and CAP, unpubl. data). Despite confounding assessments of PA performance (*sensu* Joppa and Pfaff, 2010), consistently allocating resources to reserve creation in remote areas ensures that national conservation targets committed to international conventions and funders can be fulfilled in terms of total designated areas under protection. This also satisfies conservation-minded public opinion in terms of political commitments to biodiversity conservation without displeasing local political leaders and key economic sectors, including mining, agriculture, forestry and fisheries. Given this reality, Amazonia has lately been protected as a 'Maximum Coverage Problem' (Church & Velle 2005), whereby the only important criterion is the total amount of land sequestered into PAs, with governments seeking to set aside as much land as possible under limited budgets.

Large tropical forest reserves are critical in both maintaining large-scale ecosystem phenomena (Peres 2005) and deterring the advance of anthropogenic threats(Pfaff et al. 2015), so a conservation planning compromise must be reached. Considering that land acquisition accounts for the highest cost of reserve implementation (Espirito Santo et al. 1991) resources could be allocated considering the trade-offs between numeric and reserve size targets across different levels of vulnerability. Amazonian conservation planning should therefore be systematically designed at the watershed scale (Chapter 3) with high-vulnerability, high-biodiversity areas and land prices used to optimize reserve selection and design. This should replace the practice of geographic allocation of reserves based on ephemeral political criteria, which may create the false impression of successful targets but often fail to allocate PAs where they are most needed.

Finally, the efficacy of *de jure* PAs in truly protecting biodiversity, once they are sited anywhere, is well beyond the scope of this assessment. We have only shown that there is a large protection gap to be filled even if we can assume that all reserves are well implemented, fully functional, and perform well under potentially adverse conditions that are likely to arise in the future. Indeed, Amazonian protected areas are gradually facing more hostile political and development threats (Bernard et al. 2014; Ferreira et al. 2014; Campos-Silva et al. 2015), so that sustained political will and conservation investments will be required to ensure that future PAs remain truly protected.

4.5 References

Andam, K.S. et al., 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences*, 105(42), pp.16089–16094. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.0800437105 [Accessed March 20, 2016].

Angermeier, P.L. & Karr, J.R., 1994. Biological Integrity Versus Biological Diversity as Policy Directives: Protecting Biotic Resources. In *Ecosystem Management*. New York, NY: Springer New York, pp. 264–275. Available at: http://link.springer.com/10.1007/978-1-4612-4018-1_24 [Accessed March 20, 2016].

Ball, I.R., Possingham, H.P. & Watts, M., 2009. Marxan and relatives: Software for spatial conservation prioritisation. In A. Moilanen, K. A. Wilson, & H. P. Possingham, eds. *Spatial conservation prioritisation: Quantitative methods and computational tools*. Oxford: Oxford University Press, pp. 185–195.

Barrett, N.E. & Barrett, J.P., 1997. Reserve design and the new conservation theory. In *The ecological basis of conservation*. Springer US, pp. 236–251.

Bass, M.S. et al., 2010. Global Conservation Significance of Ecuador's Yasuní National Park A. Hector, ed. *PLoS ONE*, 5(1), p.e8767. Available at: http://dx.plos.org/10.1371/journal.pone.0008767.

Bernard, E., Penns, L.A.O. & Araújo, E., 2014. Downgrading, Downsizing, Degazettement, and Reclassification of Protected Areas in Brazil. *Conservation Biology*, 28(4), pp.939–950. Available at: http://doi.wiley.com/10.1111/cobi.12298.

BirdLife International and NatureServe, 2014. Bird species distribution maps of the world.

Brooks, T., 2010. Conservation planning and priorities. In *Conservation Biology for All*. Oxford: Oxford University Press, pp. 199–219.

Bruner, A.G. et al., 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, 291(5501), pp.125–128. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.291.5501.125.

Butchart, S.H.M. et al., 2010. Global Biodiversity: Indicators of Recent Declines. *Science*, 328(5982), pp.1164–1168. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1187512 [Accessed March 20, 2016].

Cabeza, M. & Moilanen, A., 2001. Design of reserve networks and the persistence of biodiversity. *Trends in Ecology & Evolution*, 16(5), pp.242–248. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0169534701021255.

Campos-Silva, J.V., da Fonseca Junior, S.F. & da Silva Peres, C.A., 2015. Policy reversals do not bode well for conservation in Brazilian Amazonia. *Natureza & Conservação*, 13(2), pp.193–195. Available at: http://linkinghub.elsevier.com/retrieve/pii/S1679007315000444.

Center for International Earth Science Information Network - CIESIN et al., 2011. Global Rural-Urban Mapping Project, Version 1 (GRUMPv1): Coastlines.

Church, R. & Velle, C.R., 2005. The maximal covering location problem. *Papers in Regional Science*, 32(1), pp.101–118. Available at: http://doi.wiley.com/10.1111/j.1435-5597.1974.tb00902.x [Accessed March 20, 2016].

Claude Gascon et al., 2000. Riverine barriers and the geographic distribution of Amazonian species. *Proceedings of the National Academy of Sciences*, 97(25), pp.13672–13677.

Constantino, P. et al., 2012. Empowering local people through community-based resource monitoring: a comparison of Brazil and Namibia. *Ecology and Society*, 1(17), p.22.

Convention on Biological Diversity, 2011. *Conference of the Parties Decision X/2: Strategic plan for biodiversity 2011–2020*,

Cowling, R.. et al., 2004. Nature conservation requires more than a passion for species. *Conservation Biology*, 18(6), pp.1674–1676.

Davis, S.H. & Wali, A., 1994. Indigenous land tenure and tropical forest management in Latin America. *Ambio*, pp.485–490.

Deguignet, M. et al., 2014. 2014 United Nations List of Protected Areas. *UNEP-WCMC: Cambridge, UK*.

Espirito Santo, C. do, Faleiros, A.A. & Fundaçao Pró-Natureza-FUNATURA, 1991. *Cost of implantation of conservation units in legal Amazonia*, Brasília, DF.

ESRI, 2011. ArcGIS Desktop: Release 10.

Eva, H. et al., 2005. *a Proposal for Defining the Geographical*, Office for Official Publications of the European Communities.

Fearnside, P.M. & Ferraz, J., 1995. A conservation gap analysis of Brazil's

Amazonian vegetation. *Conservation Biology*, pp.1134–1147.

Ferreira, J. et al., 2014. Brazil's environmental leadership at risk. *Science*, 346(6210), pp.706–707. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1260194.

Ferreira, L. et al., 2001. Identificação de áreas prioritárias para a conservação da biodiversidade por meio da representatividade das unidades de conservação e tipos de vegetação nas ecorregiões da Amazônia brasileira. In *Biodiversidade na Amazônia brasileira: avaliação e ações prioritárias para a conservação, uso sustentável e repartição de benefícios*. São Paulo: Instituto Socioambiental, pp. 268–86.

Gaston, K.J. et al., 2008. The Ecological Performance of Protected Areas. *Annual Review of Ecology, Evolution, and Systematics*, 39(1), pp.93–113.

Hansen, M.C. et al., 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, 342(6160), pp.850–853. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1244693 [Accessed March 20, 2016].

IBGE, 2000. Censo Demográfico 2000 - Resultados do universo. Available at: http://www.ibge.gov.br [Accessed March 20, 2015].

IUCN, 2015. The IUCN Red List of Threatened Species - Version 2015-4. Available at: http://www.iucnredlist.org [Accessed July 7, 2015].

James, A.N., Gaston, K.J. & Balmford, A., 1999. Balancing the Earth's accounts. *Nature*, 401(6751), pp.323–324. Available at: http://www.nature.com/doifinder/10.1038/43774 [Accessed March 20, 2016].

Joppa, L. & Pfaff, A., 2010. Reassessing the forest impacts of protection. *Annals of the New York Academy of Sciences*, 1185(1), pp.135–149. Available at: http://doi.wiley.com/10.1111/j.1749-6632.2009.05162.x.

Joppa, L.N. et al., 2013. Achieving the Convention on Biological Diversity's Goals for Plant Conservation. *Science*, 341(6150), pp.1100–1103. Available at: http://www.sciencemag.org/content/341/6150/1100.abstract\nhttp://www.sciencemag.org/content/341/6150/1100.full.pdf.

Joppa, L.N. & Pfaff, A., 2009. High and Far: Biases in the Location of Protected Areas J. Moen, ed. *PLoS ONE*, 4(12), p.e8273. Available at: http://dx.plos.org/10.1371/journal.pone.0008273.

Junk, W.J. & Soares, M.G.M., 2001. Freshwater Fish Habitats in Amazonia: State of Knowledge, Management, and Protection. *Aquatic Ecosystem Health & Management*, 4(4), pp.437–451. Available at: http://www.tandfonline.com/doi/abs/10.1080/146349801317276107 [Accessed March 19, 2016].

Kati, V. et al., 2004. Hotspots, complementarity or representativeness? designing optimal small-scale reserves for biodiversity conservation. *Biological Conservation*, 120(4), pp.471–480. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0006320704001454.

Knight, A.T. et al., 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology*, 22(3), pp.610–617.

Laurance, W. et al., 2001. The future of the Brazilian Amazon. *Science (Washington)*, 291(5503), pp.438–439.

Laurance, W.F. et al., 2002. Predictors of deforestation in the Brazilian Amazon. *Journal of Biogeography*, 29(5-6), pp.737–748. Available at: http://doi.wiley.com/10.1046/j.1365-2699.2002.00721.x.

Levi, T. et al., 2011. Spatial tools for modeling the sustainability of subsistence hunting in tropical forests. *Ecological Applications*, 21(5), pp.1802–1818. Available at: http://doi.wiley.com/10.1890/10-0375.1.

Locke, H. & Dearden, P., 2005. Rethinking protected area categories and the new paradigm. *Environmental Conservation*, 32(1), pp.1–10. Available at: http://www.journals.cambridge.org/abstract_S0376892905001852 [Accessed March 20, 2016].

Loyola, R., 2014. Brazil cannot risk its environmental leadership D. Richardson, ed. *Diversity and Distributions*, 20(12), pp.1365–1367. Available at: http://doi.wiley.com/10.1111/ddi.12252.

M Goulding, Barthem, R. & Ferreira, E., 2003. *The Smithsonian atlas of the Amazon*,

Margules, C.R. & Pressey, R.L., 2000. Systematic conservation planning. *Nature*, 405(6783), pp.243–253. Available at: http://www.nature.com/doifinder/10.1038/35012251.

de Marques, A.A.B. & Peres, C. a., 2015. Pervasive legal threats to protected areas in Brazil. *Oryx*, 49(01), pp.25–29. Available at: http://www.journals.cambridge.org/abstract_S0030605314000726.

Master, L.L., 1991. Assessing Threats and Setting Priorities for Conservation.

Conservation Biology, 5(4), pp.559–563. Available at: http://doi.wiley.com/10.1111/j.1523-1739.1991.tb00370.x [Accessed March 20, 2016].

Mayr, E. & O'Hara, R.J., 1986. The biogeographic evidence supporting the Pleistocene forest refuge hypothesis. *Evolution*, pp.55–67.

McPherson, M. et al., 2008. GIS-based modeling of environmental risk surfaces (ERS) for conservation planning in Jamaica. *Journal of conservation planning*, 4, pp.60–89.

Naughton-Treves, L., Holland, M.B. & Brandon, K., 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Environment and Resources*, 30(1), pp.219–252. Available at: http://www.annualreviews.org/doi/abs/10.1146/annurev.energy.30.050504.164507.

Nelson, A. & Chomitz, K.M., 2011. Effectiveness of Strict vs. Multiple Use protected areas in reducing tropical forest fires: a global analysis using matching methods H. H. Bruun, ed. *PLoS ONE*, 6(8), p.e22722. Available at: http://dx.plos.org/10.1371/journal.pone.0022722.

Nepstad, D. et al., 2006. Inhibition of Amazon Deforestation and Fire by Parks and Indigenous Lands. *Conservation Biology*, 20(1), pp.65–73. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2006.00351.x [Accessed March 20, 2016].

Nepstad, D. et al., 2001. Road paving, fire regime feedbacks, and the future of Amazon forests. *Forest Ecology and Management*, 154(3), pp.395–407.

Olson, D.M. et al., 2001. Terrestrial Ecoregions of the World: A New Map of Life on Earth. *BioScience*, 51(11), p.933. Available at: http://bioscience.oxfordjournals.org/cgi/doi/10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2 [Accessed March 20, 2016].

Parry, L. et al., 2010. Drivers of rural exodus from Amazonian headwaters. *Population and Environment*, 32(2-3), pp.137–176. Available at: http://link.springer.com/10.1007/s11111-010-0127-8 [Accessed March 20, 2016].

Peres, C. A, Barlow, J. & Laurance, W.F., 2006. Detecting anthropogenic disturbance in tropical forests. *Trends in ecology & evolution (Personal edition)*, 21(5), pp.227–9. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16697907.

Peres, C. A., 2000. Effects of Subsistence Hunting on Vertebrate Community

Structure in Amazonian Forests. *Conservation Biology*, 14(1), pp.240–253. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2000.98485.x.

Peres, C. A., 2001a. Paving the way to the future of Amazonia. *Trends in Ecology & Evolution*, 16(5), pp.217–219. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0169534701021322.

Peres, C. a., 2001b. Synergistic Effects of Subsistence Hunting and Habitat Fragmentation on Amazonian Forest Vertebrates. *Conservation Biology*, 15(6), pp.1490–1505. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2001.01089.x.

Peres, C. a., 2005. Why We Need Megareserves in Amazonia. *Conservation Biology*, 19(3), pp.728–733. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00691.x.

Peres, C.A., 2011. Conservation in Sustainable-Use Tropical Forest Reserves. *Conservation Biology*, 25(6), pp.1124–1129. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2011.01770.x [Accessed March 20, 2016].

Pfaff, A. et al., 2015. Protected Areas' Impacts on Brazilian Amazon Deforestation: Examining Conservation – Development Interactions to Inform Planning K. Bawa, ed. *PLOS ONE*, 10(7), p.e0129460. Available at: http://dx.plos.org/10.1371/journal.pone.0129460.

Pickett, S.T., Parker, V.T. & Fiedler, P.L., 1992. The new paradigm in ecology: implications for conservation biology above the species level. In *Conservation biology*. Springer US, pp. 65–78.

Pimm, S.L. & Lawton, J.H., 1998. Planning for biodiversity. *Science*, p.2068.

Plantinga, A.J., Lubowski, R.N. & Stavins, R.N., 2002. The effects of potential land development on agricultural land prices. *Journal of Urban Economics*, 52(3), pp.561–581. Available at: <Go to ISI>://WOS:000180018100009.

Prendergast, J.R., Quinn, R.M. & Lawton, J.H., 1999. The Gaps between Theory and Practice in Selecting Nature Reserves. *Conservation Biology*, 13(3), pp.484–492. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1999.97428.x.

R Core Team, 2015. R: A language and environment for statistical computing. Available at: http://www.r-project.org/.

Rodrigues, A.S.L. et al., 2004. Global Gap Analysis: Priority Regions for Expanding the Global Protected-Area Network. *BioScience*, 54(12), p.1092. Available at:

http://bioscience.oxfordjournals.org/cgi/doi/10.1641/0006-3568(2004)054[1092:GGAPRF]2.0.CO;2 [Accessed March 20, 2016].

Roldán Ortega, R., 2004. *Models for recognizing indigenous land rights in Latin America*,

Rylands, A.B. & Brandon, K., 2005. Brazilian Protected Areas. *Conservation Biology*, 19(3), pp.612–618. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00711.x.

Rylands, A.B. & Brandon, K., 2005. Brazilian Protected Areas. *Conservation Biology*, 19(3), pp.612–618. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00711.x.

Schill, S. & Raber, G., 2012. Protected Area Tools (PAT) for ArcGIS software. Available at: http://www.gispatools.org.

Schwartzman, S. & Zimmerman, B., 2005. Conservation Alliances with Indigenous Peoples of the Amazon. *Conservation Biology*, 19(3), pp.721–727. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00695.x.

Silva Dias, M.A.F. et al., 2002. Cloud and rain processes in a biosphere-atmosphere interaction context in the Amazon Region. *Journal of Geophysical Research*, 107(D20), p.8072. Available at: http://onlinelibrary.wiley.com/doi/10.1029/2001JD000335/abstract.

Soares-Filho, B. et al., 2014. Cracking Brazil's forest code. *Science*, 344(6182), pp.363–364.

Soares-Filho, B.S. et al., 2006. Modelling conservation in the Amazon basin. *Nature*, 440(7083), pp.520–523. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16554817.

Soulé, M.E. & Sanjayan, M.A., 1998. ECOLOGY: Conservation Targets: Do They Help? *Science*, 279(5359), pp.2060–2061. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.279.5359.2060 [Accessed March 20, 2016].

Ter Steege, H. et al., 2003. A spatial model of tree α-diversity and tree density for the Amazon. *Biodiversity & Conservation*, 12(11), pp.2255–2277.

Ter Steege, H. et al., 2015. Estimating the global conservation status of more than 15,000 Amazonian tree species. *Science Advances*, 1(10), pp.e1500936–e1500936. Available at: http://advances.sciencemag.org/cgi/doi/10.1126/sciadv.1500936.

Terborgh, J., 1999. *Requiem for nature*, Island press.

Terborgh, J. & Schaik, C. van, 2002. Why the world needs parks. In *Making parks work: strategies for preserving tropical nature*. Washington, D.C.: Island press.

Valsecchi, J., El Bizri, H. & Figueira, J., 2014. Subsistence hunting of Cuniculus paca in the middle of the Solimões River, Amazonas, Brazil. *Brazilian Journal of Biology*, 74(3), pp.560–568. Available at: http://www.scielo.br/scielo.php?script=sci_arttext&pid=S1519-69842014000300560&lng=en&nrm=iso&tlng=en.

Walker, B.H., 1992. Biodiversity and Ecological Redundancy. *Conservation Biology*, 6(1), pp.18–23. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1992.610018.x [Accessed March 20, 2016].

Wilson, K. a., Carwardine, J. & Possingham, H.P., 2009. Setting Conservation Priorities. *Annals of the New York Academy of Sciences*, 1162(1), pp.237–264. Available at: http://doi.wiley.com/10.1111/j.1749-6632.2009.04149.x.

World Database on Protected Areas (WDPA), 2015. Statistics. UNEP/WCMC/IUCN/WCPA. Available at: http://www.protectedplanet.net/ [Accessed February 11, 2015].

Zhouri, A., 2010. “Adverse Forces” in the Brazilian Amazon: Developmentalism Versus Environmentalism and Indigenous Rights. *The Journal of Environment & Development*, 19(3), pp.252–273. Available at: http://jed.sagepub.com/cgi/doi/10.1177/1070496510378097.

Zimmerman, B. et al., 2001. Conservation and development alliances with the Kayapó of south-eastern Amazonia, a tropical forest indigenous people. *Environmental Conservation*, 28(1), pp.10–22.

Chapter 5

Reserve size vs positioning tradeoffs in conservation planning for regions facing markedly divergent anthropogenic threats.

To be submitted as:

Teles, D., Peres, C. A., Reserve size vs positioning tradeoffs in conservation planning for regions facing markedly divergent anthropogenic threats.

5.1 Introduction

The accelerated rate of biodiversity loss in many tropical biomes calls for effective strategies in protecting the Earth's remaining wildland areas (Butchart et al. 2010). Given the often severely limited resources to advance biodiversity conservation (Bruner et al. 2004), prioritization of which areas should be set aside for protection is a critical decision process in any systematic conservation planning of any given region (Brooks 2006). High biodiversity areas frequently co-occur with high levels of anthropogenic disturbance (Pimm & Lawton 1998; Luck et al. 2004), aggravating conflicts of interest between conservation imperatives and economic sectors of society in terms of where best to site viable protected areas (PAs) (Balmford et al. 2001). Allocating PAs to areas of high human development pressure can effectively contain biodiversity loss and confront the spread of human related threats (Bruner et al. 2001). However, compared to remote wilderness regions, high-threat areas typically involve higher opportunity costs (Lourival et al. 2008), ultimately leading to small PAs. Yet reserve size is one of the most important assets in reserve design, ensuring the viability of ecological processes and sufficiently large animal and plant populations in the long run (Peres 2005). A major challenge in PA design is therefore to both maximize the effective reserve size within the boundaries of local socio-political and financial feasibility, and allocate reserves to the most relevant strategic locations to minimize impending biodiversity loss.

Priority-setting of conservation areas at regional scales has been conducted under different approaches involving expert advice (Rylands & Brandon 2005), software of decision-making support (Pressey et al. 1996), or *ad hoc* determination (Pressey 1994). Despite clear differences in these processes, in general, all methods use the presence and/or distribution of biodiversity surrogate(s) and area vulnerability to human-related threats to propose the best available configuration of areas to be protected (or a rank of conservation relevance). However, regardless of the prioritization method used, areas facing high levels of threat are frequently avoided as candidate PAs (Joppa & Pfaff 2009; Chapter 4). This strategy often results in larger and supposedly better protected PAs than if high-threat areas were to be targeted on the basis of the with the same level of investment. However, by systematically evading present or predicted threats, important biodiversity features of the landscape may receive little or no desirable attention in planning PA networks (Luck et al. 2004).

Site protection is widely considered the most relevant conservation action to ensure long-term survival of several species (Lovejoy 2006; Boyd et al. 2008). The often-bleak future scenarios of many tropical biomes provide a resounding endorsement for the importance of PAs in safeguarding pristine forest areas (Laurance et al. 2001; Soares-Filho et al. 2006) where PAs are required to ensure a robust representation of the landscapes in which they have been positioned. While vulnerability and threats must be mapped and explicitly considered in any area prioritization process, a clear focus of the biodiversity outcomes is critical when ranking the conservation relevance of competing candidate sites (Tulloch et al. 2015) to maximize the feasibility of the solutions generated (Wilson et al. 2005).

In all Amazonian countries, PAs are selected and created in uncoordinated haphazard way, most frequently to avoid areas of high human pressure (Chapter 4). Here, we use the entire watershed of a major river of Central Amazonia as a case study to demonstrate the importance of considering the variation in site vulnerability in conservation planning and the perils of indefinitely penalizing areas of high human population pressure when defining the positioning of PAs. This watershed presents a strong longitudinal gradient of human disturbance, with most land use changes located in the lower portion of the river basin, which is typical of all roadless river basins of Amazonia (Chapter 3). This analysis uses sub-watersheds as effective planning units and a decision support algorithm to generate two conservation priority scenarios — with and without consideration of human threats. We therefore compare both of these outcomes and discuss how to best integrate basin scale information on biodiversity and human-related threats to enhance biodiversity conservation planning in similar data-poor regions facing severe limitations in financial resources. We also evaluate the efficacy and efficiency of the only existing protected area in this watershed, raising questions over the *de facto* protection of this watershed.

5.2 Methods

5.2.1 Study area

This analysis focuses on the Tefé River basin, which encompasses a total catchment area of ~24,050 km^2 (an area ~10% the size of the UK) spanning the municipal counties of Tefé and Alvarães within the Brazilian state of Amazonas. The Tefé is a black-water tributary (M Goulding et al. 2003) of the right bank of the Solimões (= Amazon) River and its watershed is characterised by a large lake (mean width ≈ 5 km) in its lower portion, which is very important for commercial fishing activities in the region (Fig. 5.1). In Amazonia, black-water rivers are generally associated with white or brown sandy soils (Sioli 1984), which tend to be very nutrient-poor. This can partly explain why the Tefé River watershed is so sparsely populated, with vast upriver areas completely uninhabited. The highest human population densities is concentrated in the lower part of the watershed, near the nutrient-rich Solimões River and the urban centre of Tefé, the largest city (~61,453 inhabitants) of the middle section of the Solimões River (IBGE 2010). The 3-4 month dry season (<50 mm/month; August to November) is alternated with a longer wet season (December to July) (Ayres 2006). The region exhibits a humid tropical climate (Afi according to Köppen) with average monthly temperatures over 23°C and annual temperature oscillation below 10°C (Lima et al. 1999). Like many other Amazonian river basins, the Tefé watershed remains roadless and can only be accessed by air (via Tefé airport) or waterways, with a fast-boat travel time >10 hours from Manaus, the largest Amazonian city.

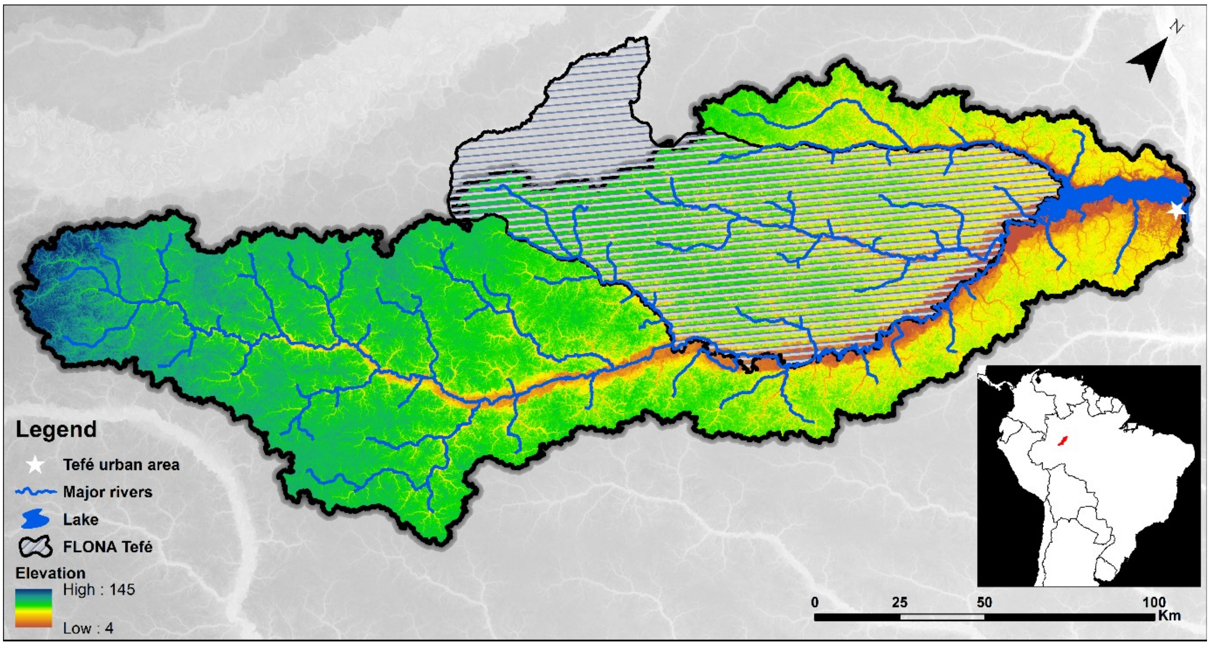


Fig. 5.1 – Delimitation of the Tefé River watershed. The star indicates the location of Tefé, the sixth largest city of the Brazilian State of Amazonas. The colour gradient (from yellow to dark green) indicates higher elevation from sea-level and the only protected area in the watershed is represented by the hatched area. Tributaries of 6th and higher orders (Horton-Strahler classification) are represented by blue lines. The red polygon inside the inset map shows the position of the watershed within South America.

5.2.2 Sub-watersheds

Conservation planning in Amazonia should be undertaken considering the geographic partition across major watersheds and their subdivisions (Chapters 3 and 4). Watersheds are nested hierarchical units and it is important to define which scale (basin size and order) is best fits a working template of conservation planning units according to management objectives (Thieme et al. 2007). However, there is a trade-off between the resolution and representativeness in selecting the spatial scale of sub-watersheds used as planning units. Large sub-watershed divisions may lack the necessary degree of detail required for the analysis, while small sub-basin units may lack adequate representation of their biogeographic features.

We used the program RiverTools© (Rivix LL 2012) to segment the Tefé watershed into sub-divisions using SRTM images with a 1-arc-sec spatial resolution and adopting 0.05 km^2 as an area threshold to prune the 1st-order watersheds. We selected as planning units all watersheds corresponding to 6th-order tributaries (Horton-Strahler classification) and to 5th-order streams/rivers that do not flow into 6th-order tributaries. Areas adjacent to major river channels left outside the above-mentioned categories were manually segmented based on the delimitation of higher-order watersheds (7th and 8th order).

5.2.3 Vulnerability map

Using the ARCGIS extension Protected Area Tools (Schill & Raber 2012), we created an Environmental Risk Surface (hereafter, ERS; see McPherson et al. 2008) as a measurement of vulnerability to anthropogenic threats throughout the entire study area. We used the following variables as measures of human threat: recent deforestation (between 2000 and 2013), Human Population Density - HPD, physical accessibility, and proximity to settlements. Deforestation was modelled as a fixed positional value (100) based on a map of 30-m spatial resolution generated from a time-series analysis of Landsat images (Hansen et al. 2013). HPD values were derived from population data from the year 2010 based on the Brazilian decadal census conducted by the Brazilian Institute of Geography and Statistics (IBGE 2010). HPD was assigned to six bins as ordinal (rank) categories (0, 0.01-1, 1.01-10, 10.01-25, 25.01-50, >50 persons/km^2), which were modelled with the values of 0, 20, 40, 60, 80 and 100, respectively. Accessibility was calculated considering both the overland and fluvial distance along 5th-order tributaries and rivers from the city of Tefé. In modelling accessibility we considered a fluvial travel velocity three times faster than overland travel, and the final raster represented the shortest distance to the urban centre considering both means of locomotion. Distance from settlements was derived from a shapefile containing all individual households within the

Tefé basin plus a 10-km surrounding buffer area. We then interpolated the household positional data using a Inverse Distance Weighting (IDW) technique considering in the final solution a 10-km radius of spatial influence from each household. The derived basin-wide raster interpolation was then used as a measure of proximity from any given settlement. Raster images describing both accessibility and proximity to settlements were normalised on a scale of 0 to 100, which correspond to the values for those variables used to model the ERS. The final ERS value for the entire river basin was then defined as the overall sum of the values of all overlapping threat layers. With the exception of deforestation, which is based on ~30-m pixels, all the other raster images had a spatial resolution of 30 arc-seconds (~1km).

5.2.4 Biodiversity surrogates

Biodiversity data is scarce or non-existent for all major Amazonian watersheds, and the Tefé basin is no exception. However, with a combination of remote sensing and field data we were able to generate a broad and meaningful set of biodiversity surrogates for the selection of conservation priority areas within this watershed. We defined the prioritization of areas in the Tefé watershed based on (1) the spatial distribution of all threatened bird and mammal species across Amazonia, (2) density of tree species (or genera) under risk of recruitment bottleneck induced by hunting pressure, (3) vegetation type, and (4) extent of floodplains.

The Tefé watershed contains seven threatened and one endemic (*Pyrrhura lucianii*) species of birds and nine species of threatened mammals, one of which endemic (*Pithecia albicans*). Protected areas (PAs) are of crucial importance to ensure the long-term survival of threatened species (Boyd et al. 2008) and the distribution of all abovementioned species was taken into consideration in the prioritization of areas. The geographic range polygons of mammals and bird species was obtained from two sources (IUCN 2015; BirdLife International and NatureServe 2014).

Several Amazonian plant genera are considered vulnerable to recruitment limitation generated by chronic overhunting of their seed dispersers (Peres & Roosmalen 2002; Peres et al. 2016). These species are generally large-seeded trees whose dispersal depends on a very limited number of large mammals that can ingest and pass those seeds intact (hereafter, "undispersed" trees). We modelled the density of trees of those genera (Appendix 5.1) for the watershed based on 36 forest inventories sourced from two datasets: 13 forest inventories from the RADAMBRASIL project (DNPM 1973-1983) and 23 forest inventories carried out by the HRT Oil & Gas company aiming to fulfil environmental licensing requirements as stipulated by the Amazonas State Environmental Agency (see distribution of tree plots in Appendix 5.2). The RADAMBRASIL dataset is based on forest inventories performed within 1-ha plots (500m x 20m), which were laid out and sampled by botanists and para-botanists hired by the Brazilian government. Within those plots, all trees ≥100 cm CBH (circumference at breast height) or ≥31.8 cm dbh (diameter at breast height) other than arborescent palms were measured and reliably identified to the level of genus. HRT Oil & Gas forest inventories were carried out by BRASA (BRA Soluções Ambientais), an environmental consultancy enterprise specializing in floristic inventories. All trees ≥15 cm DBH (diameter at breast height) were measured and identified to the highest possible level of taxonomic resolution, and/or at least to the level of genus. To standardize those datasets, only large trees from the HRT dataset above the minimum size cut-off in the RADAMBRASIL dataset (≥31.8 cm DBH) were used in the analyses. We interpolated the densities of the "undispersed" trees encountered in these assessments to the entire Tefé basin using ordinary krigging.

Broad classifications of vegetation types have long been used as a coarse-filter strategy to design conservation units in Amazonia (Ferreira et al. 2001). The vegetation map used in this priority-setting analysis was created following a physiognomic-ecological classification (Ellenberg & Mueller-Dombois 1967) performed by the Brazilian National Institute for Space Research (INPE). The map was based on the information from RADAMBRASIL forest inventories, that were subsequently updated by the Brazilian government (Veloso et al. 1991).

The width of the Amazon River and all its main tributaries are subject to marked seasonal variation, generating areas continuously flooded for periods of up to 210 days (Junk et al. 1989). These seasonally flooded areas are of great importance to fish and other animals (Haugaasen & Peres 2005; Hurd et al. 2016) and floodplain forests are well adapted to seasonal switches between terrestrial and aquatic phases, with many plant species entirely restricted to inundated forests (Parolin et al. 2004). Although floodplains vary according to climatic, hydrological, hydrochemical, and botanical features (Junk et al. 2011), they can be broadly classified into three different types (Assis et al. 2015). Várzeas are flooded forests under the influence of white-water rivers; igapós are restricted to black- or clear-water rivers; and Paleo-varzeas are ancient areas of varzeas (Mid-Late Pleistocene) that may be presently flooded by black-water rivers. We mapped the extent of those three types of flooded forests in the Tefé basin using satellite images provided by ALOS PALSAR (Rosenqvist et al. 2007). ALOS PALSAR is a L-band synthetic aperture radar (SAR) sensor that is widely considered as adequate for mapping flood areas (Hess et al. 2003). Watersheds of 5th-order tributaries comprised the smallest scale of analysis used to define these floodplain areas.

Because the software used to rank the priority areas does not process vector files, all maps of mammal and bird distributions, vegetation types, and floodplain areas were converted into rasters of 30 arc-second of spatial resolution prior to analysis.

5.2.5 Vertebrate biomass density

We carried out diurnal line-transect surveys of mid-sized to large-bodied mammals at nine forest sites distributed throughout the Tefé watershed (Appendix 5.2). Considering these nine sites, we walked a total census distance of ~1,853 km along 44 forest transects of at least 4000m each. The standardised census protocol follows the general line-transect survey guidelines described in Peres & Cunha (2011) and species, group size and perpendicular distance from the transect to the centre of the group were recorded. Surveys at BATE 1, BATE 2, Cacau, Acacia and Alto Tefé were based on a geometric 16-km^2 forest grid (4km x 4km) in which five parallel transects of 4 km each were walked simultaneously by several trained observers over a single 20-day period. Surveys at Clareira 6, Curimatá, Vila Moura, Cianê, Ponta da Castanha 1, Ponta da Castanha 2, and Açaituba were similar but included only three transects of 4-5km each.

We estimated the mammal species diversity and biomass density for those nine sites in addition to three more sites in the lower portion of the watershed, which had been surveyed by Johns (1986). Given the relative paucity of sites, we were unable to generate animal mammal biomass surface for the entire watershed. However, the longitudinal distribution of these sites allow us to understand the general pattern of abundance of medium to large sized mammals from the mouth to the headwater region of the Tefé river.

In particular, our estimates of overall mammal biomass density are based on the aggregate density ($\hat{D}s$) of caviomorph rodents (genus Dasyprocta), ungulates and primates (Appendix 5.3) based on distance sampling theory (Burnham et al. 1980) using the program Distance 6.2® (Thomas et al. 2010). In all cases, $\hat{D}s$ was estimated based on pooled (global) models considering all sites that we censused and subsequently post-stratified at the site level. The model best describing the detectability of each species, considering different key functions, was selected based on their respective ΔAIC values.

5.2.6 Selection of priority areas

We used the Zonation prioritization software to generate the relative ranking of conservation priorities across the entire Tefé watershed. The software algorithm prioritises the structural habitat connectivity and in doing so iteratively removes from the full landscape the locations of lowest conservation value, retaining the most important areas (Moilanen et al. 2005). Candidate cells were removed by Additive Benefit Function (Moilanen 2007) in which the sum of values for all biodiversity surrogates is used to define the significance of each cell (Moilanen et al. 2014). We used sub-watersheds (mean size = 95,8 km^2; sd = 95,7 km^2) as planning units and all biodiversity surrogates received the same weighting in the analysis. To avoid a fragmented final solution, we set a boundary length penalty (BLP) to the process of priority areas selection. BLP interferes in the hierarchy with which different areas are removed, prioritizing removal of areas that decrease the boundary length of the next iterative step (Moilanen & Wintle 2007).

In particular, we compared the results obtained considering only different biodiversity surrogates across the entire basin (optimal solution) with the results obtained when both biodiversity surrogates and measures of vulnerability were incorporated as a cost surface (low resistance solution). An additional prioritization scenario was also generated to enforce the maintenance of existing protected areas (within the Tefé National Forest). This last scenario was created to examine how efficient is the existing configuration of protected areas across the watershed to preserve all high-biodiversity sub-watersheds (discussed below).

5.2.7 Assessing efficiency and effectivity of the Tefé National Forest

The ~875,000-ha Tefé National Forest (FLONA Tefé) was created in 1989, and is the only protected area (83% of its area) located within the Tefé Basin (Fig 5.1). Brazilian National Forests are classified as IUCN category VI reserves, and are therefore legally inhabited. Around 500 families live inside the FLONA Tefé (Brianezi 2007).

We analysed how efficiently this PA can protect biodiversity by calculating the replacement cost of the prioritization solution, whereby the areas encompassed by the FLONA were the last to be removed in the prioritization process, with an unconstrained (optimal) prioritization solution. To define the FLONA effectiveness, we used the PA's performance index (P') as described by Spracklen et al. (2015). The index is the ratio of forest loss inside (r_{in}) and outside (r_{out}) the PA ($P' = r_{in}/r_{out}$). Forest loss was calculated as the ratio between forest cover in 2013 and in 2000 (Hansen et al. 2013) for 5-km buffers inside and outside the FLONA. Forest is defined as pixels containing >10% of tree canopy cover (FAO 2007). P' values < 1 indicate that internal forest loss is lower than what would be expected in the absence of protection; PAs are considered as effective if $P' < 0.75$ and highly effective if $P' < 0.5$ (Spracklen et al. 2015).

Unless explicitly stated, all GIS processing and statistical analyses were performed using ARCGIS 10.1 (ESRI 2011) and R version 3.1.3 (R Core Team 2015), respectively.

5.3 Results

We segmented the Tefé watershed into 251 sub-watersheds ranging from 11.31 to 790.36 km^2 (mean 95.81 km^2). Of those sub-watersheds, 115 correspond to

5th-order rivers, 79 to 6th-order rivers, and 57 are considered "pass-through" watersheds (as defined by Thieme et al. 2007), located in the main channel of the highest order rivers (Fig. 5.2). Watersheds of 5th and 6th order (headwater watersheds) contains all the upstream catchment areas contributing to the basin which by themselves represent an independent management unit. Meanwhile, pass-through watersheds are strategically positioned to control the accessibility to headwater watersheds. In practice, this means that a single headwater watershed can be selected as a PA, whereas the same does not apply to pass-through watersheds. However, if a PA is allocated to more than one headwater watershed, then it should also contain one or more pass-through watersheds in order to increase the PA defensibility and create a cohesive, interconnected cluster of areas.

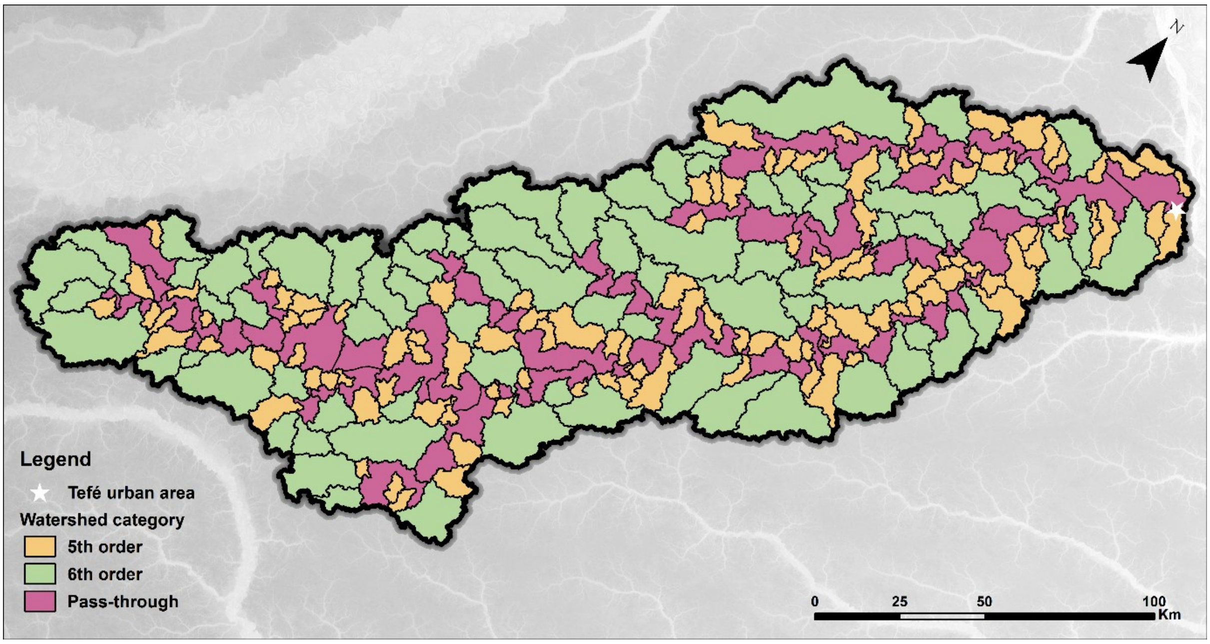


Fig. 5.2 –Sub-watersheds of the Tefé River Basin following the Horton-Strahler classification of main rivers. Pass-through watersheds are present on both sides of the main river channel and receive discharge from other sub-watersheds. The Tefé urban centre is indicated in the map by a white star.

5.3.1 Vulnerability map

The Tefé basin presents a well-defined longitudinal gradient of vulnerability (Fig. 5.3). The areas most prone to human impacts, defined by the ERS metric, are located in the lower part of the watershed around the Tefé lake and alongside the three largest tributaries of the Tefé river, with vulnerability values decreasing towards more remote areas. Small isolated areas of high vulnerability in the central to upper sections of the watershed were deforested to support oil and natural gas exploration activities. Vulnerability scores across the entire study region ranged from 2 to 400 (mean =67.9; SD = 43.3). In particular, most high-vulnerability areas are located inside the only existing protected area in the entire watershed (Fig. 5.1).

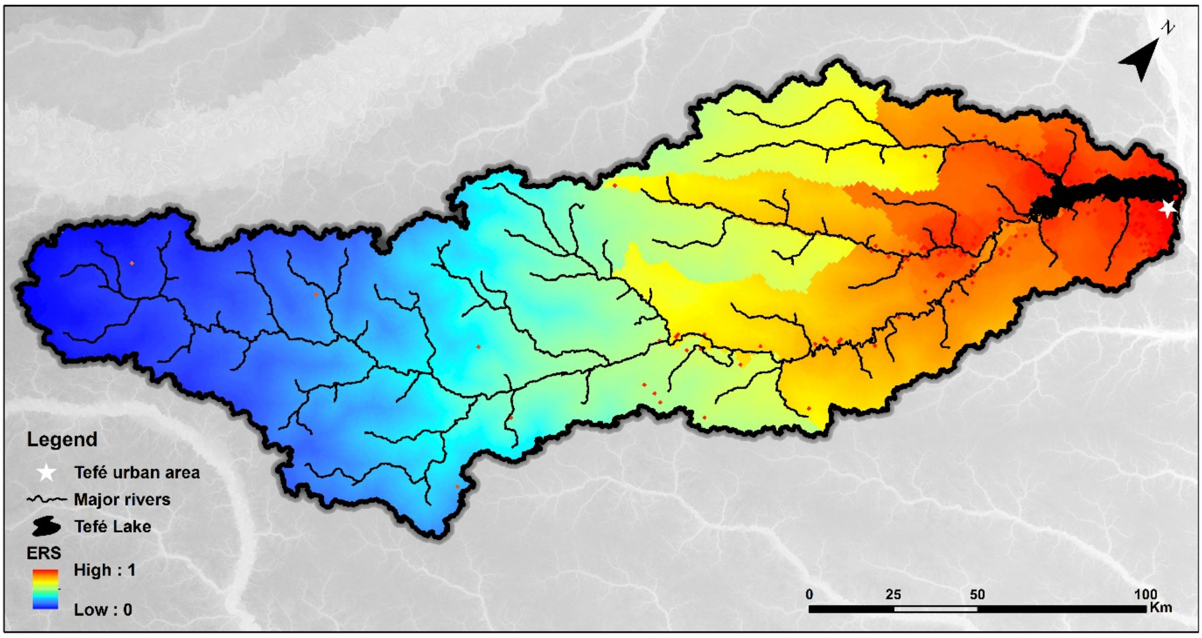


Fig. 5.3 – Geographic distribution of vulnerability to anthropogenic habitat disturbance throughout the study region. Vulnerability values are colour-coded from higher (red) to lower (blue) values associated with each ~1-km^2 pixel. The map present the vulnerability scores normalised (0 to 1 scale) where 1 represents the highest level of vulnerability encountered in the watershed. Note that the highest vulnerability scores are associated with the Tefé Lake (black area in the lowest portion of the watershed) in the neighbourhood of the city of Tefé (white star). Black lines indicate major rivers.

5.3.2 Biodiversity surrogates distribution

While vulnerability presents a clear distribution pattern, the same cannot be assumed for biodiversity surrogates. We encountered a large asymmetry in the distribution of all surrogates along the watershed (Fig. 5.4).

- Mammals and birds

The relevance of areas to threatened and endemic species of mammals and birds is represented in the map on the basis of a normalized Relative Biodiversity Index (nRBI) (Fig. 5.4A). The index was calculated using an area-weighted function for which the distribution of each species within each sample unit is related to the global distribution of the surrogate across the entire study region (Schill & Raber 2012). In other words, the index represents a measure of relative uniqueness of the watersheds in relation to mammal and bird distributions. Across the watershed, areas with highest uniqueness to birds and mammals are located in the vicinity of the Solimões River (near the mouth of the Tefé watershed) and in the Tefé-Purus interfluve (right bank of the Tefé River).

- Vegetation

Although the vegetation in most of the watershed is classified as Type 6 (Tall closed-canopy lowland evergreen forest) there are seven different major phyto-physiognomies across the entire Tefé basin (Fig.5.4 B):

- White sand forest “Campinarana” (Type 1)
- Open-canopy alluvial evergreen forest with arborescent palms (Type 2)
- Open-canopy evergreen lowland forest with arborescent palms (Type 3)
- Tall closed-canopy alluvial evergreen forest (Type 4)
- Medium-stature closed-canopy alluvial evergreen forest (Type 5)
- Tall closed-canopy lowland evergreen forest (Type 6)
- Formations with fluvial influence without palm trees (Type 7)

Areas of Campinarana (Type 1), a highly vulnerable and rare white-sand ecosystem (Adeney et al. 2016), are concentrated in the lower portions of the watershed. Conversely, areas of open-canopy evergreen lowland forest with arborescent palms (Type 3) can only be found in the upper part of the watershed.

- Threatened trees

Large-seeded trees threatened by dispersal limitation have a bimodal distribution with high-density areas in both the upper and lower parts of the watershed. The density values ranged from 53.77 to 73.3 (mean = 65.38, SD = 4.69) (Fig 5.4 C). These trees are dispersed by large mammals that are highly sensitive to hunting and the depletion of mammal population will directly affect the dispersal ability of the trees creating an indirect link between human pressure and long-term tree survival. Some of the highest densities encountered for those genera overlaps areas of high vulnerability (Fig. 5.3), posing concerns over their population dynamics in the near future.

- Floodplain areas

All three broadly defined types of floodplain areas occurring in Amazonia (Assis et al. 2015) are encountered within the study area. There are a total of 308,525 ha of seasonally flooded forests including 252,290 ha of Igapó (81.7%), 54,601ha of Paleo-varzea (17.7%) and 1,634 ha of Varzea (0.6%) (Fig. 5.4 D). While Igapó is well distributed across the watershed, all Paleo-varzeas are associated with the mainstem of the Tefé river. True varzeas are restricted to a small extension of the Solimões Varzea but were not included in the area prioritization process.

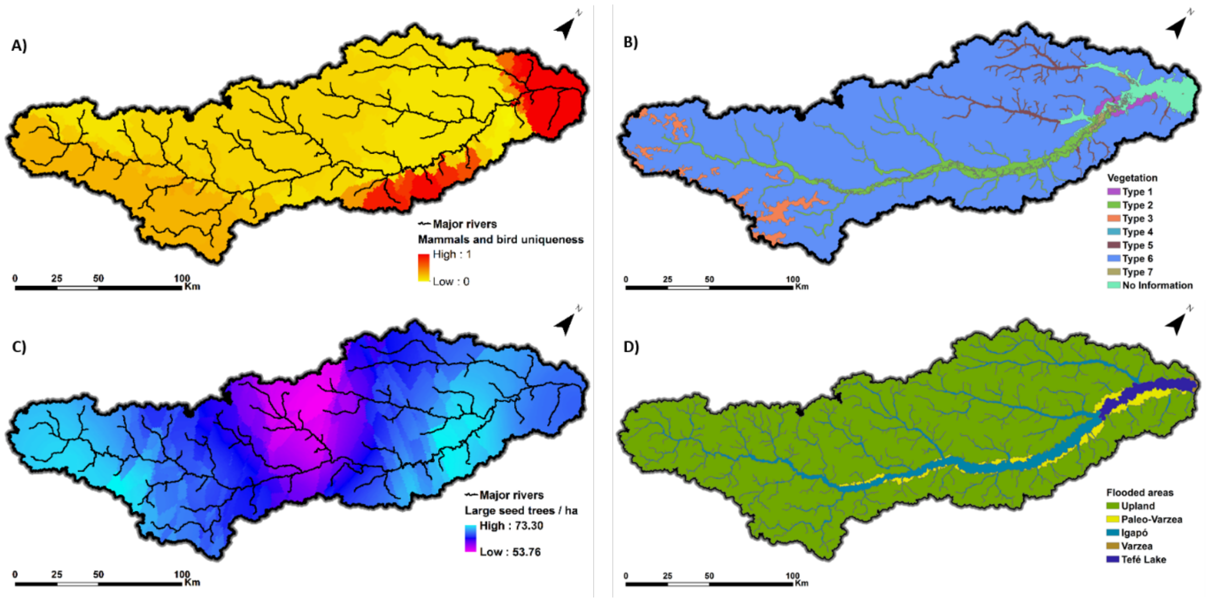


Fig. 5.4 – Distribution of biodiversity surrogates in the Tefé watershed: (A) Mammal and bird uniqueness defined on the basis of a normalized Relative Biodiversity Index (see main text); (B) Vegetation map following physiognomic-ecological classification, where Type 1 – White sand forest "Campinarana", Type 2 – Open-canopy alluvial evergreen forest with arborescent palms, Type 3 – Open-canopy evergreen lowland forest with arborescent palms, Type 4 – Tall closed-canopy alluvial evergreen forest Type 5 – Medium-stature closed-canopy alluvial evergreen forest, Type 6 – Tall closed-canopy lowland evergreen forest, Type 7 – Formations with fluvial influence without palm trees; (C) Density of large-seeded tree species sensitive to overhunting. The map was generate by ordinal krigging of data from forest inventories (see distribution of tree plots in Appendix 5.2); (D) Floodplain forest type and extent.

5.3.3 Mammal biomass

Our biomass estimates were based in 17 species of medium and large size mammals and ranged from 504 to 1192.7 kg/km2 (mean = 849.2, SD = 193.8) (Appendix 5.3). These values indicate the watershed present a high biomass of mammals compared with other sites in Amazonia (Peres 2000, Table 1). The highest values of biomass are encountered in the bottom part of the watershed in areas nearby the Tefé Lake (Fig 5.5). Similarly with threatened trees (Fig 5.4 C) the middle section of the watershed present the smallest values for the parameter assessed. Oleoduto (Fig 5.5, location 8) is placed outside the study area, but was kept in the analysis because its proximity with the watershed water dividers (less than five kilometres).

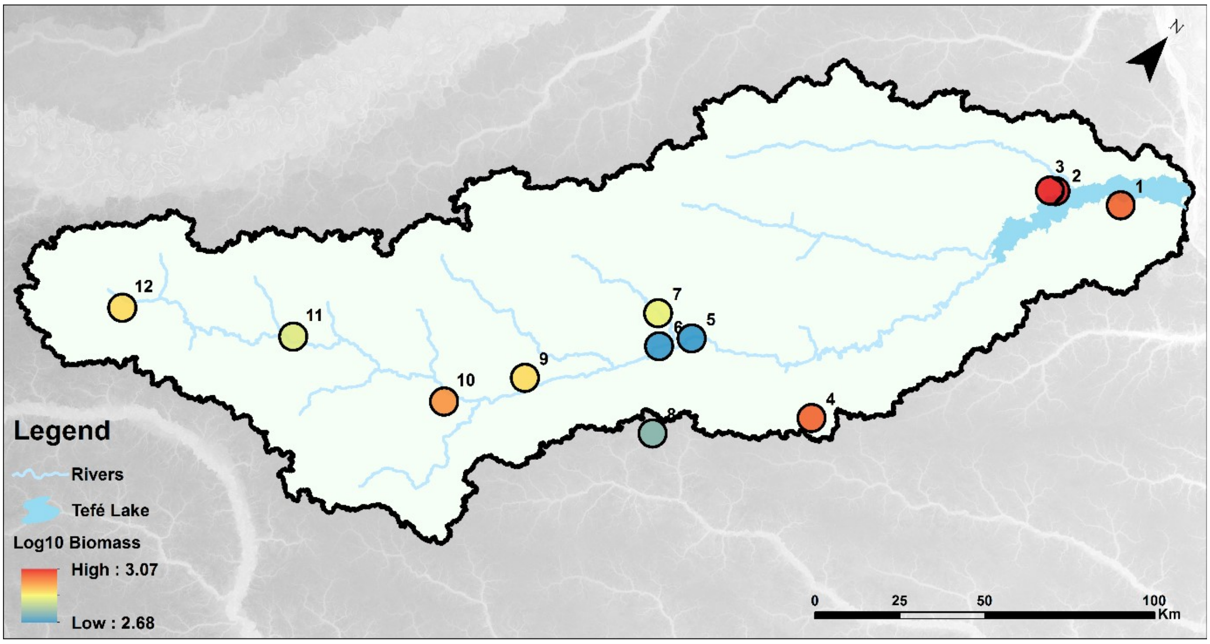


Fig. 5.5 – Log 10 (from kg/km2) biomass estimates for 12 sites in Tefé watershed, Central Amazônia. The numbers represent the localities were the data was collected, being: 1 – Acaituba[1], 2 – Ponta da Castanha logged[1], 3- Ponta da Castanha unlogged[1], 4 – Acácia, 5 - Vila Moura, 6 – Bate 1, 7 – Curimatá, 8 – Oleoduto, 9 – Bate 2, 10 – Cacau, 11 – Ciane, 12- AltoTefé (see Appendix 5.3 for details about biomass of each location). Biomass values are colour-coded from higher (red) to lower (blue) values in each of the sample locations. Major rivers and the Tefé lake are represented in light blue.

[1] Data collected for Andrew Jones (1986)

5.3.4 Priority areas

Figure 5.6 shows the prioritization ranking of sub-watershed in the Tefé basin, based on distribution of biodiversity surrogates. We generate two different solutions for the prioritization, one considering just the distribution of the surrogates, what we called optimal solution (Fig 5.6 A), and another one considering the vulnerability map (Fig 5.3) as cost surface, called low resistance solution (Fig 5.6 B). As expected both solution present divergent outcomes with few sub-watersheds (13 in total) considered top 20% ranked in both solutions (Fig 5.6 C). The optimal solution has the majority of its priority areas situated in the lower portion of the watershed and in the interfluve Tefé-Purus. Meanwhile the low resistance solution has the majority of its high ranked sub-watersheds located in the headwaters of the Tefé watershed.

Analysing the prioritization rankings (Fig 5.6) simultaneously with the threat map (Fig 5.3) two areas stands out as critical for PAs siting in the watershed (Fig 5.6, yellow circles). Some watersheds present a high relevance for biodiversity while located in areas of small anthropic pressure (Fig 5.6, yellow circle 1). Protecting this area present small opportunity cost, either biological or socio-political. Despite the high level of vulnerability encountered around the Tefé Lake indicates protecting the sub-watersheds located in the lower part of the watershed may not be feasible, the protection of some high ranked sub-watersheds located nearby the most vulnerable areas (Fig 5.6, yellow circle 2) would be beneficial containing the spread of the human frontiers while protecting relevant areas for biodiversity.

5.3.5 FLONA Tefé conservation value

The FLONA Tefé cannot be considered as either efficient or effectively protecting biodiversity within the Tefé watershed. The inclusion of the FLONA in the prioritization process generates an inefficient solution in terms of protecting all biodiversity surrogates (Fig. 5.7). The distribution of currently protected sub-watersheds underperform by 24.3% in terms of potential biodiversity protection if the most relevant sub-watersheds were protected. This poor allocation of conservation resources affects more than just the current scenario. If all future investments in the creation of PAs across the watershed were allocated in the most efficient fashion, which is rarely the case (Butchart et al. 2012), it would be necessary to protect almost 75% of the entire Tefé watershed to achieve the most efficient PA network configuration. However, the inefficient positioning of the FLONA is not its only design problem. Despite a low rate of forest loss (~3%) within a 5-km buffer area inside and outside the PA boundaries, deforestation is slightly higher inside the FLONA than outside (P′ = 1.002), clearly showing that this PA has little or no effect in containing the spread of human threats.

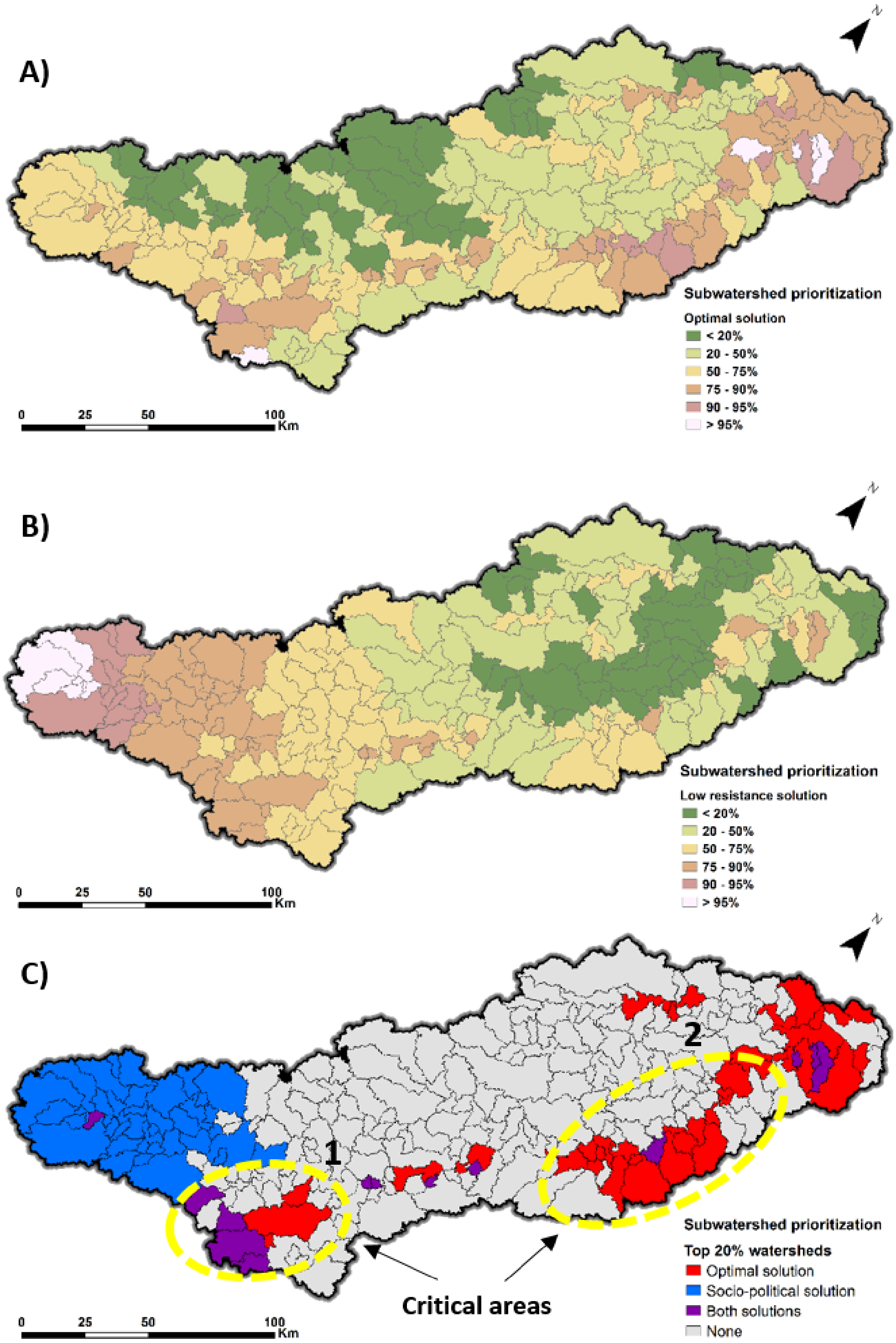


Fig. 5.6 – Sub-watershed biodiversity conservation prioritization in Tefé basin. (A) Priority ranking generated exclusively from biodiversity surrogates (optimal solution); (B) Priority ranking when using vulnerability map (Fig 5.3 this chapter) as cost surface (Low resistance solution); (C) Top 20% ranked areas for both solutions indicating where they overlap. The percentages presented in (A) and (B) show cumulative percentage of area already removed when this watershed is removed in the prioritization process, thus the higher the percentage more relevant is the sub-watershed. The yellow circles highlight the more relevant area to PA siting in the Tefé watershed considering defensibility, protection of biodiversity and low opportunity costs (circle 1) or threat contention (circle 2).

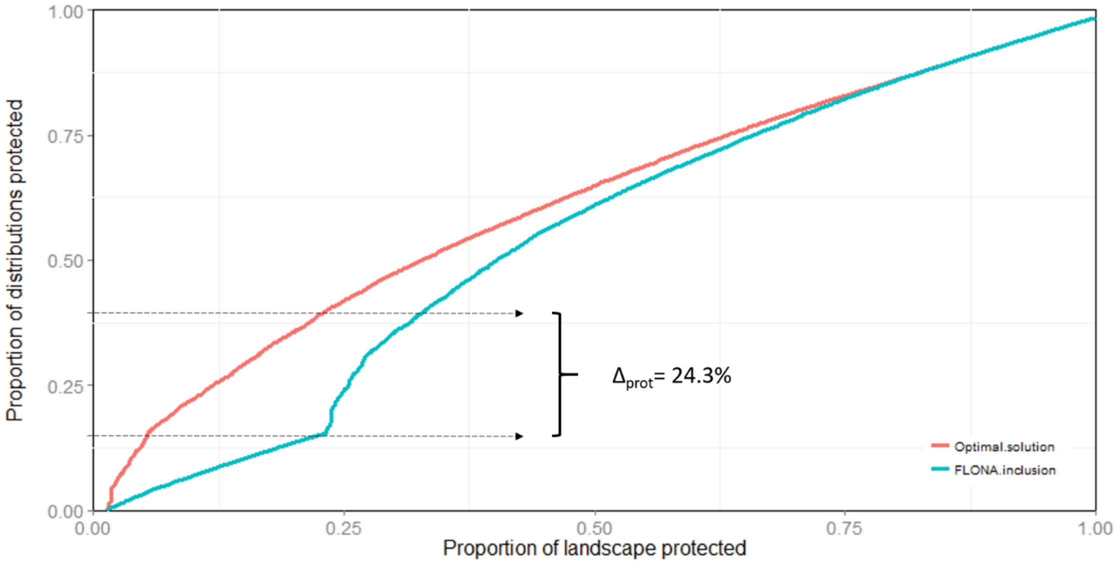


Fig. 5.7 – Performance curves of biodiversity conservation prioritization within the Tefé watershed. The red line represents the optimal solution where only biodiversity surrogates are considered in the prioritization. The blue line represents the solution when areas within the FLONA Tefé are the last to be removed from the landscape. The difference between both curves represent the replacement cost, indicating how the solution with the inclusion of the FLONA undermines the efficiency of prioritization areas. The X and Y axes represent the proportion of the landscape under protection and the the proportion of all biodiversity surrogate distributions protected (on average), resectively. Δ_{prot} indicates the difference between the actual degree of protection across the watershed and what would be achieved if an area equivalent in size to the FLONA Tefé were protected following the optimal solution.

5.4 Discussion

Our results clearly show that conservation action that implicitly or explicitly ignores the spatial distribution of anthropogenic threats can have a strong negative impact on the outcome of biodiversity protection. Systematically avoiding areas with high levels of human stressors, as in this case study, would generate a poor final configuration of PAs in both protecting key biodiversity assets and deterring the expansion of human-related threats. Threat maps are indispensable in any systematic conservation planning (Wilson et al. 2005) and they must be taken into account as at least subsidiary information to select areas to be set aside for protection.

In deciding priorities for the Rio Tefé Basin, selection algorithms could have been manipulated to give a higher weight to rare biodiversity features (Moilanen et al. 2005), which would have yielded a different, and likely less conservative, outcome than the low-resistance solution, thereby further penalizing remote sub-watersheds far from the main sources of threats. However, we did not do so for two reasons: (1) allow trade-offs between the biodiversity surrogates used, favouring sub-watersheds with a higher number of features rather than a few rare species; and (2) our main goal with the low-resistance solution was to realistically emulate the way in which PAs are actually sited in roadless areas of Amazonia (Chapter 4), which was successfully achieved.

Although this priority-setting analysis was carried out within a single Amazonian river basin, this process is representative of any other roadless region lacking biodiversity data. Our approach effectively used sub-watersheds as conservation planning units, multiple human threats representing a composite vulnerability surface, and combining field data and remote sensing to generate key biodiversity surrogates. Collectively, these amount to a feasible solution to generate heuristic conservation recommendations, which is particularly useful in areas facing rapid expansion of human colonization frontiers but lacking resources to either plan or implement biodiversity conservation.

5.4.1 Threat and biodiversity assessments in conservation planning for data-poor regions

The Tefé watershed present the typical pattern of vulnerability encountered in many other Amazonian roadless watersheds (Chapter 3). This is characterized by a well-defined hydrological gradient, whereby areas of highest vulnerability values are concentrated near the mouth of major tributaries and distance to the nearest urban centre exerts a strong influence on the spatial spread of human-related activities (Parry et al. 2010) (Fig. 5.3). For conservation planning purposes, it is vital to understand threat boundaries as a representation of present reality. Developing a prioritization rank considering only a future expansion in the threat intensity but not in the current extent of vulnerability would be risky. In this case, a fluid socioeconomic dynamic characterised by technological advances that facilitate predatory activities (e.g. faster and more fuel-efficient boats, more affordable chainsaws and fire weapons), growing market demand for forest products (Hamilton 2002), and expansion of established settlements, all of which can make longer upriver boat trips worthwhile, hence considerably expanding the range of high vulnerability areas. Therefore, vulnerability maps must be as up-to-date as possible and ideally present the most likely vectors of expansion. The extent and intensity of mapped vulnerability can ensure the recognition of opportunities and set constraints for real-world conservation planning (Wilson et al. 2005), thereby increasing the feasibility of solution for PAs creation and siting.

The present state of knowledge on Amazonian species distribution is widely considered as inadequate to justify recommendations for conservation planning (Schulman et al. 2007), wherein in an ideal world a large sum of money and time would have to be spent to enable biodiversity assessments across the entire biome. However, the potentially prolonged delay associated in acquiring sufficient knowledge on species distribution and composition will likely be detrimental to pre-emptive biodiversity protection (Hermoso et al. 2013). Brazilian Amazonia and most other tropical forest domains are experiencing rapid rates of habitat loss (Fearnside 2005), human population growth (Cincotta et al. 2000), and changes in per capita purchase power, all of which render the creation or implementation of PAs in the biome more urgent. In this exercise, we combined remote sensing, field survey data, and different scenarios of biodiversity hierarchy (land-classification and species distribution and abundance) to achieve a set of surrogates that satisfactorily represent the biodiversity relevance of each subregion as understood today (Fig. 5.4 and 5.5). This concept, coalescing fine and coarse filter approaches, has the advantage of sidestepping the shortcomings of using only one of those methods (Maddock & Plessis 1999), thereby producing a safer set of solutions for areas where biodiversity data is highly deficient.

The basic premise of coarse-filter conservation is to protect biotic communities of a given region, thereby ensuring the persistence of their species and ecological processes (Hunter Jr. 1991). This approach is especially relevant in areas lacking reliable species-level information, the typical reality in all Amazonian countries. Our broad-scale surrogates (physiognomic-ecological vegetation classes and extent of seasonally flooded forests) are appropriate in this context.

Amazonia hosts one of the most diverse plant communities on Earth including an estimated >15,000 tree species, 25% of which can be currently defined as threatened (ter Steege et al. 2015). The forest structure represented by the vegetation classes used here (Fig. 5.4B) is designed based on plant anatomy and physiology, climatic domains and levels of soil fertility (Veloso et al. 1992), all of which are good broad scale surrogates for tree species distributions. Floodplain areas (Fig. 5.4D) cover three sets of species: terrestrial, aquatic, and those in both environments (Naiman et al. 2010). Amazonian flooded forests capture an elevated number of endemic species (Parolin et al. 2004) and their patterns of plant composition change according to the geologic history and geochemistry of rivers (Assis et al. 2015). Yet the importance of seasonally flooded areas go far beyond their species composition. The natural mosaic containing both upland and floodplain forest ensures a marked temporal dynamic in plant phenology, with important positive demographic effects on vertebrate abundance (Haugaasen & Peres 2007; Hawes & Peres 2016) with many species typically more abundant in higher productivity floodplain forests (Peres 1999; Haugaasen & Peres 2005).

Despite the comprehensive coarse filter analysis undertaken here, some relevant species may have been neglected (Hunter Jr. 2005). Yet rare, threatened or endemic species should be additionally contemplated in any conservation priority setting (Su et al. 2004). However, by considering the variation in the density of trees potentially threatened by frugivore defaunation and the distribution of threatened and endemic mammal and bird species, we believe to have filled this gap.

The putative IUCN distribution data for all endemic and threatened species used here (Fig. 5.4A) appear to be adequate to our study area but can be very coarse for a finer scale analysis. However, information from the comprehensive and scientifically rigorous IUCN Red List (Rodrigues et al. 2006) can be of high relevance in conservation planning for areas lacking more detailed information on species distributions. The density of threatened trees was estimated based on real field data from tree plots distributed along the whole watershed, which were used to derive an interpolated density surface (Fig. 5.4C). For many other regions, detailed floristic data may not be available at this scale. However, ground data provides invaluable information about local biodiversity and must be incorporated in prioritization solutions even if indirectly. In this case, data from our terrestrial and arboreal wildlife surveys (Fig. 5.5) were not used to generate a priority area map but were considered in the definition of critical areas for conservation across the watershed.

5.4.2 Optimal reserve allocation: swallowing a bitter pill?

Before discussing PA siting and creation it is necessary to step back and realise the essential roles of PAs. Margules & Pressey (2000) provide a brief but precise definition of PAs, stating that they should be created: "… to separate elements of biodiversity from processes that threaten their existence in the wild…”. Although both biodiversity and threats are present in this definition, the main focus of conservation prioritization should be placed on biodiversity itself, rather than their threats.

Prioritization of conservation areas often concentrate on either minimizing biodiversity loss (focusing on biodiversity in more vulnerable areas) or maximizing biodiversity gain (focusing on overall biodiversity values) (Wilson et al. 2006), with the most adequate approach determined by the correlation between biodiversity value and vulnerability. Prioritization aiming to minimize biodiversity loss is more efficient when there is a positive correlation between vulnerability and biodiversity value, while strategies maximizing biodiversity gain perform better when such correlation is negative (Visconti et al. 2010). The sub-watersheds in the lower portion of Tefé Basin exhibit the top-ranking values for biodiversity (Fig. 5.6A), yet are the most vulnerable (Fig. 5.3). These areas should therefore receive a large part of the conservation investments allocated to this watershed, regardless of considerations of political or economic constraints. However, conservation typically agonises under severe funding shortages (Bruner et al. 2004), not least in Amazonia, and human development often takes political precedent over biodiversity protection (Lourival et al. 2008), resulting in high-vulnerability areas being ultimately avoided in most PA identification processes (Joppa & Pfaff 2009).

At first sight, the notion of avoiding high-vulnerability areas in creating new PAs is compelling. Beyond local socio-political resistance, areas far from colonization frontiers are rare candidates for short-term land-use, and command far lower land prices, thereby enabling larger land purchases for any given level of investment. However, by taking the path of least resistance, important biodiversity attributes can be lost (Pressey & Tully 1994), creating a false impression that the long-term viability of new PAs has been ensured. Our low-resistance solution (Fig. 5.6B) is in fact highly representative of how PAs have been created in roadless parts of Amazonia over the last two decades (Chapter 4). By applying this priority area ranking some important biodiversity features — notably Paleo-varzeas, Campinaranas and the watershed's highest mammal biomass density, which is likely due to higher productivity and higher soil fertility in the lowest parts of the watershed (Fig. 5.4 and 5.5) — would be severely underrepresented below the 25% threshold of watershed protection. In contrast, considering the prioritization accounting based entirely on biodiversity values (Fig. 5.6A), headwater areas presented mid to low-ranking relevance, posing serious questions over the relevance of large PAs in low-biodiversity areas (Pressey et al. 1993). This situation is particularly pertinent right across Brazilian Amazonia, where different regions compete with one another for conservation investments. In addition to draining resources that could be better invested in severely underprotected areas, poorly allocated PAs can also affect the efficiency of a regional scale PA network in the long run (Fig. 5.7). Moreover, the long-term viability of PAs created in remote areas confronting little or no human pressure can also be deceptive, since growing economic interests will eventually threaten them with the three D's of contemporary protected areas (downgrading, downsizing or degazzetement) (Mascia et al. 2014; Pack et al. 2016).

In particular, the lowest portion of the Tefé river basin consists of a vast ~392.5-km^2 relatively shallow black-water ria lake that once hosted outstanding natural phenomena including mass breeding migrations and large aggregations of key species of threatened aquatic mega-vertebrates, including Giant River Turtle (*Podocnemis expansa*), Black Caiman (*Melanosulchus niger*) and Amazonian Manatee (*Trichechus inunguis*). Although largely extinct at present, these phenomena were witnessed first-hand by Henry Walter Bates (1892) when he lived intermittently at Tefé (then the village of Ega) for 5 years (1850-1859). Bates describes Lago Tefé as the staple source of animal protein for the Ega settlement (then 1200 people in 107 households), with annual slaughters of some 400,000 turtles harvested for meat and ~48 million turtle eggs collected for their oil (Bates 1892, p. 314). As a key geographic component of the watershed, the Tefé Lake and its sandy beaches are in fact irreplaceable in that equally suitable breeding sites for freshwater turtles and other vertebrate species are entirely unavailable anywhere else in the Tefé and neighbouring river basins. A failure to protect this lake and neighbouring areas therefore forestalls any option to at least partly restore those spectacular ecological processes by fostering the population recovery of large aquatic vertebrates.

Unfortunately, the window of opportunity to create intact PAs protecting areas abutting the lake has already passed. The lake is important for the local economy and is one of the most vulnerable areas within the watershed (Fig. 5.3), which makes land acquisition for PA creation both prohibitive and unpopular. In this case, the conservation effort must shift to management actions (Thieme et al. 2007), and it is still possible to ensure the maintenance of important biodiversity features and ecological processes within the Tefé watershed. Considering both prioritization scenarios (Fig. 5.6A and B) and the distribution of threats (Fig. 5.3), we suggested some critical areas for protection across the watershed (Fig. 5.6C).

Some headwater sub-watersheds (Circle 1 in Fig. 5.6C) may provide a straightforward option to set aside new PAs. Protected areas in watershed regions could be large, relatively inexpensive, and relevant for biodiversity. Areas associated with both high biodiversity indicators and low opportunity costs should be prioritised. PA allocation to the other critical area (Circle 2 in Fig 5.6C) would be more controversial since it is a costly decision from a political and financial perspective. Those sub-watersheds are still within a manageable distance of the city of Tefé (<1 day round-trip in a motorized canoe) and already present some small farms and settlements. Land acquisition would be expensive, not to mention the costs of relocating dwellers and paying compensation fees for any lost revenues. In addition, this is by no means an easy task and would stir resentment with local political leaders and economic sectors that count on this area as part of a development vector. However, this area contain a large fraction of the top 20% priority ranked sub-watersheds (Fig. 5.6A) and is strategically positioned to deter or slow down the gradual upriver spread of human threats. These watersheds also contain all biodiversity surrogates examined here, including rare features as *Campinarana* and *Paleo-varzea* habitats (Fig. 5.4) and, in light of their passive defensibility, could be effectively protected by two strategically placed guard posts (Peres & Terborgh 1995). A protected area in this section of the river would also make upriver colonisation difficult by increasing travel costs to development areas farther afield, slowing down the expansion of human activities, and reducing any potential 'leakage' in human pressure and habitat loss elsewhere induced by PA creation (Ewers & Rodrigues 2008). Unfortunately, this approach has been consistently circumvented in identifying and creating new Amazonian PAs (Chapter 4). However, decisions favouring biodiversity targets over socio-political expedience is a "bitter pill" that must be swallowed while this strategy is still feasible. Simply maximising the total area protected in the least accessible regions will lead to biodiversity loss, which cannot be easily reversed even by expensive restoration programs in the future. Economic assessments, including land prices (Ando 1998) and future land use (Newburn et al. 2006), are crucial for this approach to be successful. These evaluations should be analysed concomitantly with the available budget to set a limit on how aggressive conservation actions should be. Nonetheless, habitat conversion generates economic losses over the time (Balmford et al. 2002) so that money spent in conservation now can be considered as a profitable investment in the long term.

5.4.3 The raw number's trap

The large fraction of existing protected areas within the Tefé Basin, which is well above recommended international targets, gives a false sense of security. A large area under protection does not necessarily translate into adequately safeguarded viable populations (Fig. 5.7) or strategic pre-emption of human related threats. The Tefé National Forest is severely understaffed and underfunded, counting on only three absentee reserve officials allocated to manage this unit, one of whom an administrative assistant. Considering the total area of this forest reserve (~875,000 ha), over 437,500 ha would have to be overseen by each reserve manager. Parkguard density is likely strongly correlated with park effectiveness, and the staffing level of this reserve (0.0002 manager per km^2) is 15,000 fold lower that the parkguard density of highly effective parks (Bruner et al. 2001). It is also important to note that local surveillance is not the only role of PA managers. This National Forest is also in direct conflict with a haphazard farmer settlement project (Brianezi 2007) and its intrinsic design (delimited by rivers) greatly reduces the efficiency of any enforcement action against illegal activities. In sum, this reserve is a "paper park" (*sensu* Peres & Terborgh 1995; Bruner et al. 2001) that risks becoming irrelevant in the long term in protecting regional scale, a reality typical of many other Amazonian PAs.

Total reserve acreage alone is not a robust measure of biodiversity protection (Butchart et al. 2012). Global biodiversity conservation targets through area protection are set based on political feasibility (Larsen et al. 2015), often merely serving to justify political objectives. Over 30% of the Tefé watershed is already formally protected. However, even if the FLONA Tefé worked properly, a large portion of key biodiversity assets within the watershed would remain unrepresented (Fig. 5.7). The same result would be obtained if our low resistance solution were used to prioritize ranking and promote the creation of a large PA in the headwaters of the Tefé Basin. Blindly equating larger PAs to better protection can in fact work in the opposite direction if a given PA is poorly congruent with biodiversity assets. Indeed, a large but poorly positioned PA often deflects conservation funds from more relevant areas, yet may give a deceptive impression of meeting declared conservation goals, thus paying lip service to both international donors and public opinion.

5.5 References

Adeney, J.M. et al., 2016. White-sand Ecosystems in Amazonia. *Biotropica*, 48(1), pp.7–23. Available at: http://doi.wiley.com/10.1111/btp.12293.

Ando, A., 1998. Species Distributions, Land Values, and Efficient Conservation. *Science*, 279(5359), pp.2126–2128. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.279.5359.2126.

Assis, R.L. et al., 2015. Patterns of tree diversity and composition in Amazonian floodplain paleo-várzea forest K. Woods, ed. *Journal of Vegetation Science*, 26(2), pp.312–322. Available at: http://doi.wiley.com/10.1111/jvs.12229.

Ayres, J.M., 2006. *As matas de várzea do Mamirauá: Médio Rio Solimões*, Belém: Sociedade Civil Mamirauá.

Balmford, A. et al., 2001. Conservation conflicts across Africa. *Science*, 291(March).

Balmford, A. et al., 2002. Economic reasons for conserving wild nature. *Science (New York, N.Y.)*, 297(2002), pp.950–953.

Bates, H.W., 1892. *The Naturalist on the River Amazons: A Record of Adventures, Habits of Animals, Sketches of Brazilian and Indian Life, and Aspects of Nature Under the Equator, During Eleven Years of Travel. With a Memoir of the Author, by Edward Clodd*, J. Murray.

BirdLife International and NatureServe, 2014. Bird species distribution maps of the world.

Boyd, C. et al., 2008. Spatial scale and the conservation of threatened species. *Conservation Letters*, 1(1), pp.37–43. Available at: http://doi.wiley.com/10.1111/j.1755-263X.2008.00002.x.

Brianezi, T., 2007. A reforma agrária ecológica na Floresta Nacional de Tefé. *Revista de Estudos Amazônicos*, 153(February).

Brooks, T.M., 2006. Global Biodiversity Conservation Priorities. *Science*, 313(5783), pp.58–61. Available at: http://www.sciencemag.org/content/313/5783/58.abstract.

Bruner, A.G. et al., 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, 291(5501), pp.125–128. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.291.5501.125.

Bruner, A.G., Gullison, R.E. & Balmford, A., 2004. Financial Costs and Shortfalls of Managing and Expanding Protected-Area Systems in Developing Countries. *BioScience*, 54(12), p.1119.

Burnham, K.P., Anderson, D.J. & Laake, J.L., 1980. Estimation of density from line transect sampling of biological populations. In *Wildlife Monographs*. Washington, D.C.: The Wildlife Society.

Butchart, S.H.M. et al., 2010. Global Biodiversity: Indicators of Recent Declines. *Science*, 328(5982), pp.1164–1168. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1187512 [Accessed March 20, 2016].

Butchart, S.H.M. et al., 2012. Protecting Important Sites for Biodiversity Contributes to Meeting Global Conservation Targets P. M. Bennett, ed. *PLoS ONE*, 7(3), p.e32529. Available at: http://dx.plos.org/10.1371/journal.pone.0032529.

Cincotta, R.P., Wisnewski, J. & Engelman, R., 2000. Human population in the biodiversity hotspots. *Nature*, 404(6781), pp.990–992.

DNPM, 1973. *Projeto RADAMBRASIL: Levantamento de Recursos Naturais*, Rio de Janeiro, Brazil.

Ellenberg, H. & Mueller-Dombois, D., 1967. Tentative physiognomic-ecological classification of plant formations of the Earth. In *Berichte des Geobotanischen Institutes der Eidg. Techn. Hochshule Stiftung Rübel*. Zurich, pp. 21–55.

ESRI, 2011. ArcGIS Desktop: Release 10.

Ewers, R.M. & Rodrigues, A.S.L., 2008. Estimates of reserve effectiveness are confounded by leakage. *Trends in Ecology & Evolution*, 23(3), pp.113–116.

FAO, 2007. *State of the World's Forests*, Rome, Italy.

Fearnside, P.M., 2005. Deforestation in Brazilian Amazonia: History, Rates, and Consequences. *Conservation Biology*, 19(3), pp.680–688. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00697.x.

Ferreira, L. et al., 2001. Identificação de áreas prioritárias para a conservação da biodiversidade por meio da representatividade das unidades de conservação e tipos de vegetação nas ecorregiões da Amazônia brasileira. In *Biodiversidade na Amazônia brasileira: avaliação e ações prioritárias para a conservação, uso sustentável e repartição de benefícios*. São Paulo: Instituto Socioambiental, pp. 268–86.

Hamilton, S.K., 2002. Human impacts on hydrology in the Pantanal wetland of South America. *Water Science and Technology*, 45(11), pp.35–44.

Hansen, M.C. et al., 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*, 342(6160), pp.850–853. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.1244693 [Accessed March 20, 2016].

Haugaasen, T. & Peres, C.A., 2005. Mammal assemblage structure in Amazonian flooded and unflooded forests. *Journal of Tropical Ecology*, 21, pp.133–145. Available at: http://journals.cambridge.org/abstract_S026646740400207X [Accessed November 2, 2010].

Haugaasen, T. & Peres, C.A., 2007. Vertebrate responses to fruit production in Amazonian flooded and unflooded forests. *Biodiversity and Conservation*, 16(14), pp.4165–4190. Available at: http://link.springer.com/10.1007/s10531-007-9217-z [Accessed March 25, 2016].

Hawes, J.E. & Peres, C.A., 2016. Patterns of plant phenology in Amazonian seasonally flooded and unflooded forests. *Biotropica*, p.n/a–n/a. Available at: http://doi.wiley.com/10.1111/btp.12315 [Accessed March 22, 2016].

Hermoso, V., Kennard, M.J. & Linke, S., 2013. Data Acquisition for Conservation Assessments: Is the Effort Worth It? C. G. de Leaniz, ed. *PLoS ONE*, 8(3), p.e59662. Available at: http://dx.plos.org/10.1371/journal.pone.0059662.

Hess, L.L. et al., 2003. Dual-season mapping of wetland inundation and vegetation for the central Amazon basin. *Remote Sensing of Environment*, 87(4), pp.404–428.

Hunter Jr., M.L., 2005. A Mesofilter Conservation Strategy to Complement Fine and Coarse Filters. *Conservation Biology*, 19(4), pp.1025–1029. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00172.x.

Hunter Jr., M.L., 1991. Coping with ignorance: the coarse-filter strategy for maintaining biological diversity. In K. Kohm, ed. *Balancing on the brink of extinction*. Washington, D.C.: Island press, pp. 266–281.

Hurd, L.E. et al., 2016. Amazon floodplain fish communities: Habitat connectivity and conservation in a rapidly deteriorating environment. *Biological Conservation*, 195, pp.118–127.

IBGE, 2010. Censo Demográfico 2010 - Resultados do universo. Available at: www.ibge.gov.br [Accessed June 14, 2015].

IUCN, 2015. The IUCN Red List of Threatened Species - Version 2015-4. Available at: http://www.iucnredlist.org [Accessed July 7, 2015].

Johns, A.D., 1986. *Effects of Habitat Disturbance on Rainforest Wildlife in Brazilian Amazonia, Final Report*,

Joppa, L.N. & Pfaff, A., 2009. High and Far: Biases in the Location of Protected Areas J. Moen, ed. *PLoS ONE*, 4(12), p.e8273. Available at: http://dx.plos.org/10.1371/journal.pone.0008273.

Junk, W.J. et al., 2011. A Classification of Major Naturally-Occurring Amazonian Lowland Wetlands. *Wetlands*, 31(4), pp.623–640. Available at: http://link.springer.com/10.1007/s13157-011-0190-7.

Junk, W.J., Bayley, P.B. & Sparks, R.E., 1989. The flood pulse concept in river-floodplain systems. T. Mailund, ed. *Canadian special publication of fisheries and aquatic sciences*, 106(1), pp.110–127. Available at: http://dx.plos.org/10.1371/journal.pone.0028909.

Larsen, F.W., Turner, W.R. & Mittermeier, R. a., 2015. Will protection of 17% of land by 2020 be enough to safeguard biodiversity and critical ecosystem services? *Oryx*, 49(01), pp.74–79. Available at: http://www.journals.cambridge.org/abstract_S0030605313001348.

Laurance, W. et al., 2001. The future of the Brazilian Amazon. *Science (Washington)*, 291(5503), pp.438–439.

Lima, A.A.C. et al., 1999. Levantamento de reconhecimento de média intensidade dos solos e avaliação da aptidão agrícola das terras do pólo Juruá-Solimões, Amazonas. *Embrapa Solos. Boletim de Pesquisa.*

Lourival, R. et al., 2008. Getting fourteen for the price of one! Understanding the factors that influence land value and how they affect biodiversity conservation in central Brazil. *Ecological Economics*, 67(1), pp.20–31. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0921800908001547.

Lovejoy, T.E., 2006. Protected areas: a prism for a changing world. *Trends in Ecology & Evolution*, 21(6), pp.329–333.

Luck, G.W. et al., 2004. Alleviating spatial conflict between people and biodiversity. *Proceedings of the National Academy of Sciences*, 101(1), pp.182–186. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.2237148100.

M Goulding, Barthem, R. & Ferreira, E., 2003. *The Smithsonian atlas of the Amazon*,

Maddock, A. & Plessis, M.A. du, 1999. Can species data only be appropriately used to conserve biodiversity? *Biodiversity and Conservation*, 8(5), pp.603–615. Available at: http://link.springer.com/10.1023/A:1008819015819.

Margules, C.R. & Pressey, R.L., 2000. Systematic conservation planning. *Nature*, 405(6783), pp.243–253. Available at: http://www.nature.com/doifinder/10.1038/35012251.

Mascia, M.B. et al., 2014. Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900–2010. *Biological Conservation*, 169, pp.355–361.

McPherson, M. et al., 2008. GIS-based modeling of environmental risk surfaces (ERS) for conservation planning in Jamaica. *Journal of conservation planning*, 4, pp.60–89.

Moilanen, A., 2007. Landscape Zonation, benefit functions and target-based planning: Unifying reserve selection strategies. *Biological Conservation*, 134(4), pp.571–579.

Moilanen, A. et al., 2005. Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. *Proceedings of the Royal Society B: Biological Sciences*, 272(1575), pp.1885–1891. Available at: http://rspb.royalsocietypublishing.org/cgi/doi/10.1098/rspb.2005.3164.

Moilanen, A. et al., 2014. Spatial conservation planning methods and software ZONATION. User Manual. , p.288.

MOILANEN, A. & WINTLE, B.A., 2007. The Boundary-Quality Penalty: a Quantitative Method for Approximating Species Responses to Fragmentation in Reserve Selection. *Conservation Biology*, 21(2), pp.355–364. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2006.00625.x [Accessed March 25, 2016].

Naiman, R.J., Decamps, H. & McClain, M.E., 2010. *Riparia: ecology, conservation, and management of streamside communities*, Academic Press.

Newburn, D.A., Berck, P. & Merenlender, A.M., 2006. Habitat and Open Space at Risk of Land-Use Conversion: Targeting Strategies for Land Conservation. *American Journal of Agricultural Economics*, 88(1), pp.28–42. Available at: http://ajae.oxfordjournals.org/cgi/doi/10.1111/j.1467-8276.2006.00837.x [Accessed March 25, 2016].

Pack, S.M. et al., 2016. Protected Area Downgrading, Downsizing, and Degazettement (PADDD) in the Amazon. *Biological Conservation*, 197, pp.32–39. Available at:

http://linkinghub.elsevier.com/retrieve/pii/S0006320716300386.

Parolin, P. et al., 2004. Central Amazonian Floodplain Forests: Tree Adaptations in a Pulsing System. *The Botanical Review*, 70(3), pp.357–380. Available at: http://link.springer.com/10.1663/0006-8101(2004)070[0357:CAFFTA]2.0.CO;2.

Parry, L. et al., 2010. Rural-urban migration brings conservation threats and opportunities to Amazonian watersheds. *Conservation Letters*, 3(4), pp.251–259. Available at: http://doi.wiley.com/10.1111/j.1755-263X.2010.00106.x.

Peres, C., 1999. The structure of nonvolant mammal communities in different Amazonian forest types. *Mammals of the Neotropics*, 3, pp.564–581.

Peres, C. a & Cunha, A. a, 2011. Manual para censo e monitoramento de vertebrados de médio e grande porte por transecção linear em florestas tropicais. *Wildife Conservation Society*, p.32. Available at:.

Peres, C. a., 2000. Effects of Subsistence Hunting on Vertebrate Community Structure in Amazonian Forests. *Conservation Biology*, 14(1), pp.240–253. Available at: http://doi.wiley.com/10.1046/j.1523-1739.2000.98485.x.

Peres, C. a., 2005. Why We Need Megareserves in Amazonia. *Conservation Biology*, 19(3), pp.728–733. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00691.x.

Peres, C. a. & Terborgh, J.W., 1995. Amazonian Nature Reserves: An Analysis of the Defensibility Status of Existing Conservation Units and Design Criteria for the Future. *Conservation Biology*, 9(1), pp.34–46. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1995.09010034.x.

Peres, C.A. et al., 2016. Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *Proceedings of the National Academy of Sciences*, 113(4), pp.892–897. Available at: http://www.pnas.org/lookup/doi/10.1073/pnas.1516525113 [Accessed March 21, 2016].

Peres, C.A. & Roosmalen, M. van, 2002. Primate frugivory in two species-rich neotropical forests: implications for the demography of large-seeded plants in overhunted areas. In *Seed dispersal and frugivory: ecology, evolution and conservation. Third International Symposium-Workshop on Frugivores and Seed Dispersal, São Pedro, Brazil, 6-11 August 2000*. Wallingford: CABI, pp. 407–421. Available at: http://books.google.com/books?hl=en&lr=&id=sU7213gPmDMC&oi=fnd&pg=PA407&dq=Primate+Frugivory+in+Two+Species-rich+Neotropical+Forests+:+Implications+for+the+Demography+of+Large-seeded+Plants+in+Overhunted+Areas&ots=7WiG-YqkDP&sig=n71A-azICuKj8tR6eDUO0YMG8Q.

Pimm, S.L. & Lawton, J.H., 1998. Planning for biodiversity. *Science*, p.2068.

Pressey, R.L., 1994. Ad Hoc Reservations: Forward or Backward Steps in Developing Representative Reserve Systems? *Conservation Biology*, 8(3), pp.662–668. Available at: http://doi.wiley.com/10.1046/j.1523-1739.1994.08030662.x [Accessed March 25, 2016].

Pressey, R.L. et al., 1993. Beyond opportunism: Key principles for systematic reserve selection. *Trends in Ecology & Evolution*, 8(4), pp.124–128.

Pressey, R.L., Possingham, H.P. & Margules, C.R., 1996. Optimality in reserve selection algorithms: When does it matter and how much? *Biological Conservation*, 76(3), pp.259–267.

Pressey, R.L. & Tully, S.L., 1994. The cost of ad hoc reservation: A case study in western New South Wales. *Austral Ecology*, 19(4), pp.375–384. Available at: http://doi.wiley.com/10.1111/j.1442-9993.1994.tb00503.x.

R Core Team, 2015. R: A language and environment for statistical computing. Available at: http://www.r-project.org/.

Rivix LL, 2012. RiverTools topographic and river network analysis.

Rodrigues, A. et al., 2006. The value of the IUCN Red List for conservation. *Trends in Ecology & Evolution*, 21(2), pp.71–76. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0169534705003320.

Rosenqvist, A. et al., 2007. ALOS PALSAR: A Pathfinder Mission for Global-Scale Monitoring of the Environment. *IEEE Transactions on Geoscience and Remote Sensing*, 45(11), pp.3307–3316. Available at: http://ieeexplore.ieee.org/lpdocs/epic03/wrapper.htm?arnumber=4358862 [Accessed March 25, 2016].

Rylands, A.B. & Brandon, K., 2005. Brazilian Protected Areas. *Conservation Biology*, 19(3), pp.612–618. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00711.x.

Schill, S. & Raber, G., 2012. Protected Area Tools (PAT) for ArcGIS software. Available at: http://www.gispatools.org.

Schulman, L. et al., 2007. Amazonian biodiversity and protected areas: do they meet? *Biodiversity and Conservation*, 16(11), pp.3011–3051. Available at: http://link.springer.com/10.1007/s10531-007-9158-6.

Sioli, H., 1984. The Amazon and its main affluents: hydrography, morphology of the river courses, and river types. In *The Amazon*. Springer Netherlands, pp. 127–165.

Soares-Filho, B.S. et al., 2006. Modelling conservation in the Amazon basin. *Nature*, 440(7083), pp.520–523. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16554817.

Spracklen, B.D. et al., 2015. A Global Analysis of Deforestation in Moist Tropical Forest Protected Areas R. Zang, ed. *PLOS ONE*, 10(12), p.e0143886. Available at: http://dx.plos.org/10.1371/journal.pone.0143886.

ter Steege, H. et al., 2015. Estimating the global conservation status of more than 15,000 Amazonian tree species. *Science Advances*, 1(10), pp.e1500936–e1500936. Available at: http://advances.sciencemag.org/cgi/doi/10.1126/sciadv.1500936.

Su, J.C. et al., 2004. Beyond Species Richness: Community Similarity as a Measure of Cross-Taxon Congruence for Coarse-Filter Conservation. *Conservation Biology*, 18(1), pp.167–173. Available at:

http://doi.wiley.com/10.1111/j.1523-1739.2004.00337.x.

Thieme, M. et al., 2007. Freshwater conservation planning in data-poor areas: An example from a remote Amazonian basin (Madre de Dios River, Peru and Bolivia). *Biological Conservation*, 135(4), pp.484–501. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0006320706004733.

Thomas, L. et al., 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology*, 47(1), pp.5–14. Available at: http://doi.wiley.com/10.1111/j.1365-2664.2009.01737.x [Accessed March 25, 2016].

Tulloch, V.J. et al., 2015. Why do we map threats? Linking threat mapping with actions to make better conservation decisions. *Frontiers in Ecology and the Environment*, 13(2), pp.91–99. Available at: http://doi.wiley.com/10.1890/140022.

Veloso, H. et al., 1992. *Manual técnico da vegetação brasileira*,

Veloso, H.P., Rangel Filho, A.L.R. & Lima, J.C.A., 1991. *Classificação da vegetação brasileira, adaptada a um sistema universal*,

Visconti, P. et al., 2010. Habitat vulnerability in conservation planning-when it matters and how much. *Conservation Letters*, 3(6), pp.404–414. Available at: http://doi.wiley.com/10.1111/j.1755-263X.2010.00130.x.

Wilson, K. et al., 2005. Measuring and Incorporating Vulnerability into Conservation Planning. *Environmental Management*, 35(5), pp.527–543. Available at: http://link.springer.com/10.1007/s00267-004-0095-9.

Wilson, K.A. et al., 2006. Prioritizing global conservation efforts. *Nature*, 440(7082), pp.337–340.

Chapter 6

Concluding remarks

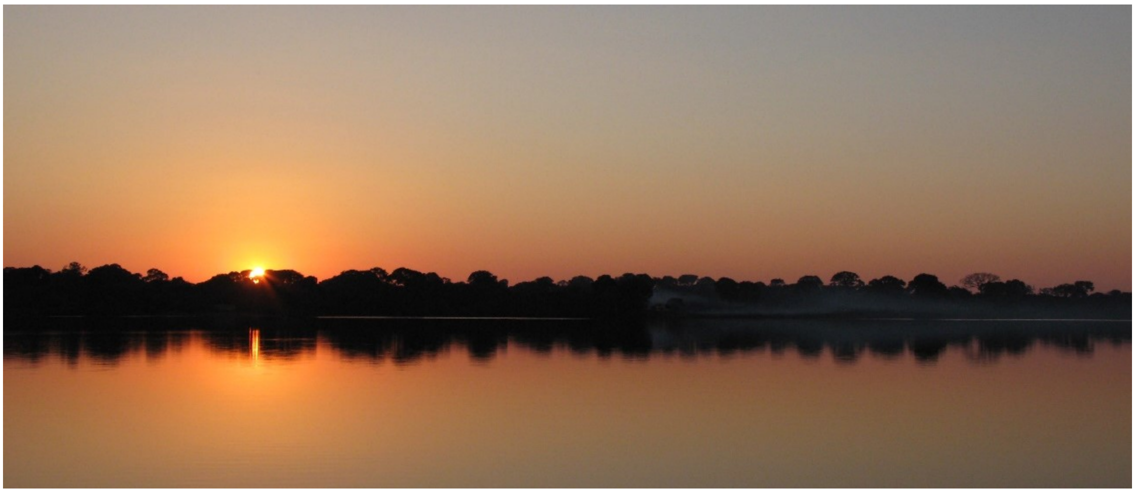


Photo: *Sunset at the Uauaçú Lake, Amazonas, Brazil. Photograph: Davi Teles*

6.1 Key considerations

6.1.1 Vulnerability and de facto protection in Amazonia

Amazonia is entering a crucial moment for the conservation of its biodiversity. The biome has continental proportions, 8.12 million km^2 given its broadest phytogeographic definition (Eva et al. 2005), and the biodiversity assessments performed in its territory to date are either severely biased in their distribution (Kress et al. 1998; Peres 2005; Hopkins 2007) or have sample limitations that reduce their application (Houghton 2005, Chapter 2). Indeed, Amazonian biodiversity, despite the large number of studies realised across the biome, remains poorly known (Peres 2005; Schulman et al. 2007). Conversely, human-induced threats to the biome are relatively well documented and mapped (Laurance et al. 2002; Swenson et al. 2011; Nepstad et al. 2001; Lees et al. 2016, Chapter 3), revealing an imperative urgency for conservation actions while this is still possible.

Watersheds located in the south-eastern portion of Amazonia (Tocantins, Xingu, Tapajós and Madeira watersheds) have a dense and well established road network that connects this region with more economically developed urban centres outside the biome (Fig. 3.3, Chapter 3). These watersheds comprise the most likely region to be the focus of future intensive deforestation (Laurance et al. 2002) and Chapter 3 warns about the distinct possibility of large areas in the Madeira headwaters merging with the Brazilian 'Arc of Deforestation' (Nepstad et al 1999, Laurance et al 2001, Morton et al 2006) despite their political boundaries. Cattle ranching and soybean monoculture are the major sources of large-scale anthropogenic disturbances in both of these areas and, due to the economic importance of these activities (Bowman et al. 2012; Gasparri et al. 2013), there is the potential for even higher forest conversion and degradation rates than those generally anticipated for the region (Soares-Filho et al. 2006). While supply-chain governance actions (Gibbs et al. 2015; Assunção & Rocha 2014) will be of pivotal importance in avoiding the advance of cattle ranching and soybean frontiers, safeguarding strategic areas within well enforced PAs remains the most reliable tool to combat the seemingly relentless expansion of high vulnerability areas (Bruner et al. 2001).

Although rural Amazonia remains sparsely populated, there are several burgeoning urban centres surrounding the biome (Fig. 6.1). The economy of these often fast-growing large cities relies heavily on free-for-all forest and aquatic timber and nontimber resources from a large surrounding catchment area, thereby increasing pressure within the biome.

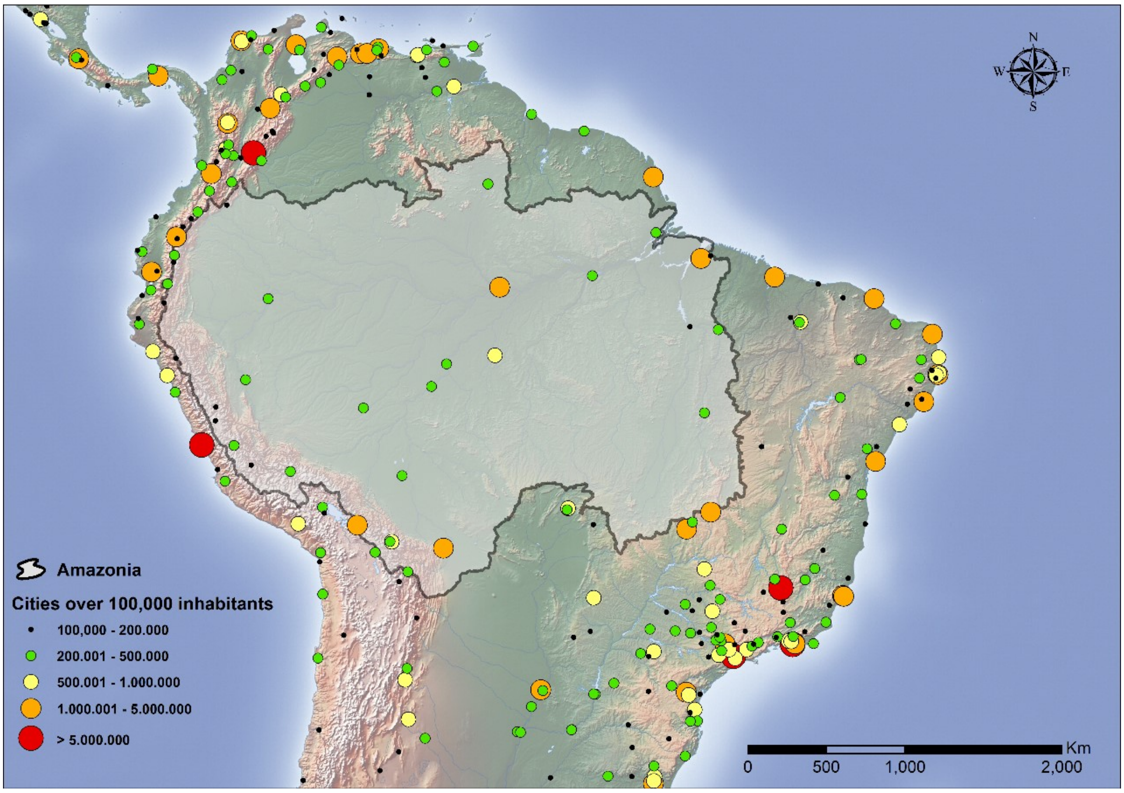


Fig. 6.1. Location of the most highly populated cities in South America with regard to the hydrologic domain of Amazonia (represented by the shaded area). Symbol size and colour indicate urban population size (source: http://www.naturalearthdata.com/).

Amazonia is facing high rates of human population growth (Cincotta et al. 2000) and it is only a matter of time until new major infrastructure projects – sworn by the Brazilian, Peruvian and Colombian governments – paves the way to densely populated urban nuclei develop inside the biome boundaries, amplifying the number of high-vulnerability areas. Within Brazil, a country that comprises almost two-thirds of Amazonia, all states within the boundaries of Amazonia have a population growth rate much higher than the rest of the country average over the last 40 years (Table 6.1). The state of Roraima currently presents the second highest growth rate of those states, matching the high vulnerability hotspot identified within this state (Chapter 3). This region is the focus of rampant agricultural settlement projects instigated by the Brazilian government and paid by Brazilian taxpayers (Diniz & Santos 2005; Schneider & Peres 2015), and could become a new hub for vulnerability expansion in the watersheds located north of the Amazon River channel. Given this scenario, a conservation strategy aiming to prioritize high-biodiversity areas under heavy human development pressure can be beneficial to both containing biodiversity loss and confronting the spread of human-related threats (cf. Bruner et al. 2001). However, as demonstrated in Chapter 4, the stark reality of protected area deployment in Amazonia has sadly been diametrically opposite to this strategic approach.

Table 6.1 – Annual population growth (%) over four decades (1970 – 2010) for the nine states of within Amazonia boundaries in Brazil. The population growth for the whole country is also presented for comparative purposes.

State	**1970-1980**	**1980-1990**	**1990-2000**	**2000-2010**
Acre	3.42	3.01	3.29	3.16
Amapá	4.36	4.67	5.77	4.04
Amazonas	4.12	3.57	3.31	2.39
Mato Grosso	2.99	1.45	0.98	2.12
Maranhão	3.09	2.1	1.63	1.63
Pará	4.62	3.46	2.54	2.24
Rondônia	16.03	7.89	2.24	1.32
Roraima	6.83	9.63	4.58	3.89
Tocantins	3.56	2.01	2.61	1.96
BRASIL	**2.48**	**1.93**	**1.64**	**1.23**

Source: IBGE: Brazilian National Census data for 1970, 1980, 1991, 2000 and 2010 (http://www.ibge.gov.br/estadosat/)

Regardless of the considerable political and financial investments allocated to a large number (and total area) of forest reserves in Amazonia (Fig. 4.2), those areas set aside for protection are currently positioned in a sub-optimal fashion to contain the spread of human threats (Chapter 4). Amazonian PAs have been created by deliberately avoiding socio-political confrontation and human related threats (Fig. 4.9), with the juxtaposition of high vulnerability areas and PAs being observed both within and between watersheds (Fig. 4.5 and 4.6).

High vulnerability areas tend to be avoided in reserve design and allocation processes all over the world (Joppa & Pfaff 2009) and there are some undeniable advantages in setting aside PAs under this logic. In addition to lower socio-political resistance, speeding the creation and implementation process, areas far from the colonization frontier have a lower probability of use and subsequent land price inflation, enabling the creation of much larger PAs. However, the perils associated with this strategy far outweigh their advantages. On a landscape scale, concentrating PA creation into watersheds with low levels of vulnerability can create a redundancy problem (Walker 1992). For example, the Amazonian PA network is biased towards overprotected low-productivity regions with similar geological history, neglecting the nutrient-rich watersheds of the biome’s southern and south-eastern regions (Fig. 4.6). Furthermore, reserve creation, like any other conservation action, must be planned with clear biodiversity outcomes as goals (Tulloch et al. 2015) and, by prioritising areas with reduced political resistance, important biodiversity assets can be left entirely unprotected (Chapter 5). Poor allocation of PAs is even more detrimental in an environment facing severe resource shortages since it withdraws investments that could alternatively have been used in areas of more urgent need. In short, the allocation of PAs to areas of low vulnerability is of great importance to justify lofty political objectives and meet declared conservation goals, but is of limited relevance in protecting the (known or unknown) spatial turnover of biodiversity over a continental scale region such as Amazonia.

It is also noteworthy that PAs created in areas far from sources of human pressure will eventually be threatened by the menace of downgrading, downsizing or degazzetement actions (Pack et al. 2016), so that we cannot assume their long-term viability and land security. The inefficiency with which areas to be set aside for protection are chosen in Amazonia is not the only concern regarding the existing PA network within the biome. In Chapter 5, I explore the case of a ~875,000-ha National Forest that has little or no effect in containing the spread of human threats which, in my own personal experience while working for a state agency that trebled the PA network of Amazonas, can be considered a representative example of several PAs created across the entire biome.

6.1.2 Using watersheds as conservation planning units

The present state of knowledge on Amazonian species distributions is wholly inadequate to justify recommendations for reserve design (Schulman et al. 2007), creating an important dilemma for conservation planners. Resources are limited (Bruner et al. 2004) and declining (Campos-Silva et al. 2015), and threats are rapidly expanding (Fearnside 2005), so investing in long drawn biodiversity assessments can potentially jeopardize the effective protection of biodiversity in the biome (Hermoso et al. 2013). In Chapter 5, I showed that an analysis using sub-watersheds as conservation planning units, mapping human threats and combining field surveys with remote sensing to generate biodiversity surrogates, is an effective solution to generate heuristic conservation recommendations. Watersheds are ecological units encompassing rivers, floodplains and uplands (Naiman et al. 2010), and consequently contain the biota of most terrestrial realms across landform gradients. When associated with an analysis coalescing fine and coarse filter approaches (Chapter 5), the use of watersheds as planning units can be a strong tool for the combined protection of both freshwater and terrestrial ecosystems.

One of the biggest advantages of using watersheds as conservation planning units is their hierarchical segmentation (Thieme et al. 2007). Larger watersheds can be segmented into smaller units that do not lose their connection with the larger scale, thereby functioning as a fractal dimension. Planning conservation through watersheds permits the navigation across the different spatial scales, by using the most appropriate resolution for any intended conservation action. In this thesis, I used three of those scales: (i) the complete hydrologic domain of the pan-Amazon region (Chapters 3 and 4), (ii) major Amazonian watersheds (Chapters 3, 4 and 5) and (iii) meso-watersheds within a single major river basin (Chapter 5). The largest scale was used for a global understanding of the degree of protection and the distribution of vulnerability in the biome. Amazonia was formed by different geologic process and presents marked geochemical differences between its rivers (Sioli 1984), heterogeneous patterns of biodiversity abundance and distribution (Gascon et al. 2000; Ter Steege et al. 2003; Bass et al. 2010), and primary productivity (Aragão et al. 2009). An analysis at this widest scale permits the evaluation of how biodiversity assets are under- or over-protected, and this information can help to redirect conservation investments aiming to optimise resource use, making that the scale at which conservation policies should be defined. However, it is at the major watershed scale (Chapter 3 and 4) that a diverse set of conservation actions (such as PA siting, reforestation programs and land use ordination) should be planned and built/consolidated in the long term.

Meso-watersheds, as sub-divisions of mega-watersheds, can be effectively used as building blocks for protected areas, as stated above and in Chapter 5. At this scale, two main types of watersheds can be classified: (1) headwater watersheds, containing all upstream catchment areas contributing to the basin; and (2) pass-through watersheds, located along the main channel of major rivers. Headwater watersheds represent independent management unit in themselves, while pass-through watersheds cannot be selected alone for the implementation of a PA. However, if a PA is supposed to encompass more than one headwater watershed it should also contain one or more pass-through watersheds, increasing its defensibility and cohesiveness. In defining meso-watersheds it is vital to consider the trade-off between resolution and representativeness that exists between different scales. Large meso-watersheds may not present the required level of detail for planning, while small meso-watersheds may lack adequate representation of desirable biodiversity targets. The most appropriate scale depends on the size of the major watershed and on the goals of conservation planning. Micro-watersheds are also relevant as PA management units, and there is a successful example using this approach in Amazonia (Marinelli et al. 2007) although this scale was not assessed in this thesis. .

Systematic conservation faces an implementation crossroad, where despite the large amount of published research on how to spend conservation investments, very few of them actually amount to tangible conservation actions (Knight et al. 2008). This is another step where biodiversity conservation is conducted in an inefficient manner and which must be avoided. The approach presented in this thesis and its practical recommendations have high potential to be applied to conservation planning in Amazonia and other poorly known tropical forest regions. Beyond the reasons mentioned previously, government environmental agencies in Amazonia already employ, to a certain extent, internal divisions which give due consideration to rivers, and in remote parts of the biome watersheds are the only meaningful geographic segmentation available for conservation planners (Chapter 3). Fortunately, despite the large size of the major Amazonian watersheds, most of them are contained within a single country (Table 3.2), allowing the biodiversity conservation planning at the watershed scale to be conducted as a national policy. The approach is feasible and will represent a considerable advance in the way biodiversity is protected in Amazonia.

Although the main focus of this thesis was Amazonia, the concepts discussed here can be applied in any other roadless region lacking biodiversity data. Planning conservation using watersheds as management units generate heuristic conservation recommendations, particularly useful in areas facing rapid biodiversity degradation and lacking resources to implement biodiversity conservation policies as it is the case of several regions in Africa and Southeast of Asia.

6.2 Future steps

6.2.1 Applying the concept

The expansion of anthropogenic threats in Amazonia will almost certainly not slow down in the near future (Chapter 1). Our best chance of protecting important elements of the biodiversity of the largest rainforest in the world is through systematic conservation planning (Margules & Pressey 2000) that maximizes effectiveness and the efficient use of resources invested considering existing limitations. In 2008, Norway and Brazil signed a letter of interest where Norway agreed to contribute US$1 billion to the Brazilian government to decrease deforestation rates in Amazonia. This agreement finished at the end of 2015 but was renewed until 2020 at this year’s climate summit in Paris. This resource, together with other international and multilateral sources, finances the Amazon Protected Areas Program (ARPA), the most ambitious PA program in the world (Soares-Filho et al. 2010). For the second phase of ARPA, a new map of Amazonian priority areas for conservation will be elaborated and hopefully the results of this thesis, following publication in peer reviewed journals, can influence the way these priority areas will be defined. Ideally, priority areas should be defined using watersheds as a planning unit and considering not just biodiversity assets but also the dynamic landscape of anthropogenic pressures that threaten them.

The process of converting watersheds into conservation planning units in Amazonia, if it ever happens, will be long but priority area assessments based on the geography of river catchment areas is an important first step. However, despite the difficulties in changing the process of area selection and PA creation in different countries, it is urgent to consider the border of watersheds in PA design to avoid replicating examples such as the Xingu National Park. This

2,642,003-hectare park may not be viable in the long-run as a result of fatal design flaws at the onset of the park creation process. The headwaters of the Xingu River were left outside of the PA in its creation and by 2003 about one-third of its original forest cover had been removed (Schwartzman & Zimmerman 2005). The water of the Xingu River inside the PA has become murky and indigenous communities living inside the park are experiencing problems with their fisheries, which provide their main source of animal protein. Impending catastrophes such as Xingu Park can potentially be avoided if the approach proposed in this thesis is adopted in the design of new PAs.

6.2.2 Financial assessment

Biodiversity conservation can be compared with trauma rooms, where decisions about where to invest limited resources need to be taken quickly while aiming for maximum returns from the investments made (Bottrill et al. 2008). This "triage" in biodiversity conservation needs to consider four aspects: (1) Politics: National development plans that limit the use of each area; (2) Biodiversity value: How relevant the area is in protecting a given set of biodiversity assets; (3) Threat level: Degree of human pressure each PA candidate will face; and (4) Cost: Total investment required to create any given PA. Political factors are generally beyond the influence of conservation planners and depend heavily on swaying popular approval. This is the aspect with least flexibility where, in general, the creation of a new PA is decided as either feasible or not. The other three aspects present a spectrum of possibilities and conservation planners must make decisions about where to position new PAs based on the values presented by those variables. Aspects 2 and 3 have been covered in detail in this thesis. However, disregarding the costs associated with the creation of PAs, in the proposition of new areas to be set aside for protection, can lead to inefficient or even unfeasible solutions (Ando 1998). To date there is no land price modelling for Amazonia, hindering the use of any conservation planning strategies already developed. Cost estimates allow a clear evaluation of the trade-off between reserve size and the most appropriate positioning, which is critical in fine-tuning reserve design. The creation of a land cost surface for Amazonia would be beneficial for the application of the approach from this thesis and I intend to pursue this goal as the next step towards establishing watersheds as conservation units in Amazonia.

6.3 References

Ando, A., 1998. Species Distributions, Land Values, and Efficient Conservation. *Science*, 279(5359), pp.2126–2128. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.279.5359.2126.

Aragão, L.E.O.C. et al., 2009. Above- and below-ground net primary productivity across ten Amazonian forests on contrasting soils. *Biogeosciences*, 6(12), pp.2759–2778. Available at: http://www.biogeosciences.net/6/2759/2009/ [Accessed March 24, 2016].

Assunção, J. & Rocha, R., 2014. *Getting Greener by Going Black : The Priority Municipalities in Brazil*, Rio de Janeiro, Brazil.

Bass, M.S. et al., 2010. Global Conservation Significance of Ecuador's Yasuní National Park A. Hector, ed. *PLoS ONE*, 5(1), p.e8767. Available at: http://dx.plos.org/10.1371/journal.pone.0008767.

Bottrill, M.C. et al., 2008. Is conservation triage just smart decision making? *Trends in Ecology & Evolution*, 23(12), pp.649–654. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0169534708002814.

Bowman, M.S. et al., 2012. Persistence of cattle ranching in the Brazilian Amazon: A spatial analysis of the rationale for beef production. *Land Use Policy*, 29(3), pp.558–568.

Bruner, A.G. et al., 2001. Effectiveness of Parks in Protecting Tropical Biodiversity. *Science*, 291(5501), pp.125–128. Available at: http://www.sciencemag.org/cgi/doi/10.1126/science.291.5501.125.

Bruner, A.G., Gullison, R.E. & Balmford, A., 2004. Financial Costs and Shortfalls of Managing and Expanding Protected-Area Systems in Developing Countries. *BioScience*, 54(12), p.1119. Available at: http://bioscience.oxfordjournals.org/cgi/doi/10.1641/0006-3568(2004)054[1119:FCASOM]2.0.CO;2 [Accessed March 24, 2016].

Campos-Silva, J.V., da Fonseca Junior, S.F. & da Silva Peres, C.A., 2015. Policy reversals do not bode well for conservation in Brazilian Amazonia. *Natureza & Conservação*, 13(2), pp.193–195. Available at: http://linkinghub.elsevier.com/retrieve/pii/S1679007315000444.

Cincotta, R.P., Wisnewski, J. & Engelman, R., 2000. Human population in the biodiversity hotspots. *Nature*, 404(6781), pp.990–992. Available at: http://www.nature.com/doifinder/10.1038/35010105.

Claude Gascon et al., 2000. Riverine barriers and the geographic distribution of Amazonian species. *Proceedings of the National Academy of Sciences*, 97(25), pp.13672–13677.

Diniz, A.M.A. & Santos, R. dos, 2005. O vertiginoso crescimento populacional de Roraima e seus impactos socioambientais. *Cad Geografia*, 15(25), pp.23–44.

Eva, H. et al., 2005. *a Proposal for Defining the Geographical*, Office for Official Publications of the European Communities.

Fearnside, P.M., 2005. Deforestation in Brazilian Amazonia: History, Rates, and Consequences. *Conservation Biology*, 19(3), pp.680–688. Available at: http://doi.wiley.com/10.1111/j.1523-1739.2005.00697.x.

Gasparri, N.I., Grau, H.R. & Gutiérrez Angonese, J., 2013. Linkages between soybean and neotropical deforestation: Coupling and transient decoupling dynamics in a multi-decadal analysis. *Global Environmental Change*, 23(6), pp.1605–1614.

Gibbs, H.K. et al., 2015. Brazil's Soy Moratorium. *Science*, 347(6220), pp.377–378. Available at: http://www.sciencemag.org/content/347/6220/377.summary.

Hermoso, V., Kennard, M.J. & Linke, S., 2013. Data Acquisition for Conservation Assessments: Is the Effort Worth It? C. G. de Leaniz, ed. *PLoS ONE*, 8(3), p.e59662. Available at: http://dx.plos.org/10.1371/journal.pone.0059662.

Hopkins, M.J.G., 2007. Modelling the known and unknown plant biodiversity of the Amazon Basin. *Journal of Biogeography*, 34(8), pp.1400–1411. Available at: http://doi.wiley.com/10.1111/j.1365-2699.2007.01737.x [Accessed March 19, 2016].

Houghton, R.A., 2005. Aboveground Forest Biomass and the Global Carbon Balance. *Global Change Biology*, 11(6), pp.945–958. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2005.00955.x [Accessed March 24, 2016].

Joppa, L.N. & Pfaff, A., 2009. High and Far: Biases in the Location of Protected Areas J. Moen, ed. *PLoS ONE*, 4(12), p.e8273. Available at: http://dx.plos.org/10.1371/journal.pone.0008273.

Knight, A.T. et al., 2008. Knowing but not doing: Selecting priority conservation areas and the research-implementation gap. *Conservation Biology*, 22(3), pp.610–617.

Kress, W. et al., 1998. Amazonian biodiversity: assessing conservation priorities with taxonomic data. *Biodiversity & Conservation*, 7(12), pp.1577–1587.

Laurance, W. et al., 2001. The future of the Brazilian Amazon. *Science (Washington)*, 291(5503), pp.438–439.

Laurance, W.F. et al., 2002. Predictors of deforestation in the Brazilian Amazon. *Journal of Biogeography*, 29(5-6), pp.737–748. Available at: http://doi.wiley.com/10.1046/j.1365-2699.2002.00721.x.

Lees, A.C. et al., 2016. Hydropower and the future of Amazonian biodiversity. *Biodiversity and Conservation*, 25(3), pp.451–466. Available at: http://link.springer.com/10.1007/s10531-016-1072-3.

Margules, C.R. & Pressey, R.L., 2000. Systematic conservation planning. *Nature*, 405(6783), pp.243–253. Available at: http://www.nature.com/doifinder/10.1038/35012251.

Morton, D.C. et al., 2006. Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences*, 103(39), pp.14637–14641. Available at: http://www.pnas.org/cgi/doi/10.1073/pnas.0606377103 [Accessed March 25, 2016].

Naiman, R.J., Decamps, H. & McClain, M.E., 2010. *Riparia: ecology, conservation, and management of streamside communities*, Academic Press.

Nepstad, D.C. et al., 1999. Large-scale impoverishment of Amazonian forests by logging and fire. Nature, 398(6727), pp.505–508. Available at: http://www.nature.com/doifinder/10.1038/19066.

Nepstad, D. et al., 2001. Road paving, fire regime feedbacks, and the future of Amazon forests. *Forest Ecology and Management*, 154(3), pp.395–407.

Pack, S.M. et al., 2016. Protected Area Downgrading, Downsizing, and Degazettement (PADDD) in the Amazon. *Biological Conservation*, 197, pp.32–39. Available at: http://linkinghub.elsevier.com/retrieve/pii/S0006320716300386.

Peres, C. A., 2005. Why We Need Megareserves in Amazonia. *Conservation Biology*, 19(3), pp.728–733.

Schneider, M. & Peres, C.A., 2015. Environmental Costs of Government-Sponsored Agrarian Settlements in Brazilian Amazonia R. Zang, ed. *PLOS ONE*, 10(8). Available at: http://dx.plos.org/10.1371/journal.pone.0134016.

Schulman, L. et al., 2007. Amazonian biodiversity and protected areas: do they meet? *Biodiversity and Conservation*, 16(11), pp.3011–3051.

Schwartzman, S. & Zimmerman, B., 2005. Conservation Alliances with Indigenous Peoples of the Amazon. *Conservation Biology*, 19(3), pp.721–727.

Sioli, H., 1984. The Amazon and its main affluents: hydrography, morphology of the river courses, and river types. In *The Amazon*. Springer Netherlands, pp. 127–165.

Soares-Filho, B. et al., 2010. Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences*, 107(24), pp.10821–10826.

Soares-Filho, B.S. et al., 2006. Modelling conservation in the Amazon basin. *Nature*, 440(7083), pp.520–523. Available at: http://www.ncbi.nlm.nih.gov/pubmed/16554817.

Ter Steege, H. et al., 2003. A spatial model of tree α-diversity and tree density for the Amazon. *Biodiversity & Conservation*, 12(11), pp.2255–2277.

Swenson, J.J. et al., 2011. Gold Mining in the Peruvian Amazon: Global Prices, Deforestation, and Mercury Imports G. J.-P. Schumann, ed. *PLoS ONE*, 6(4), p.e18875. Available at: http://dx.plos.org/10.1371/journal.pone.0018875.

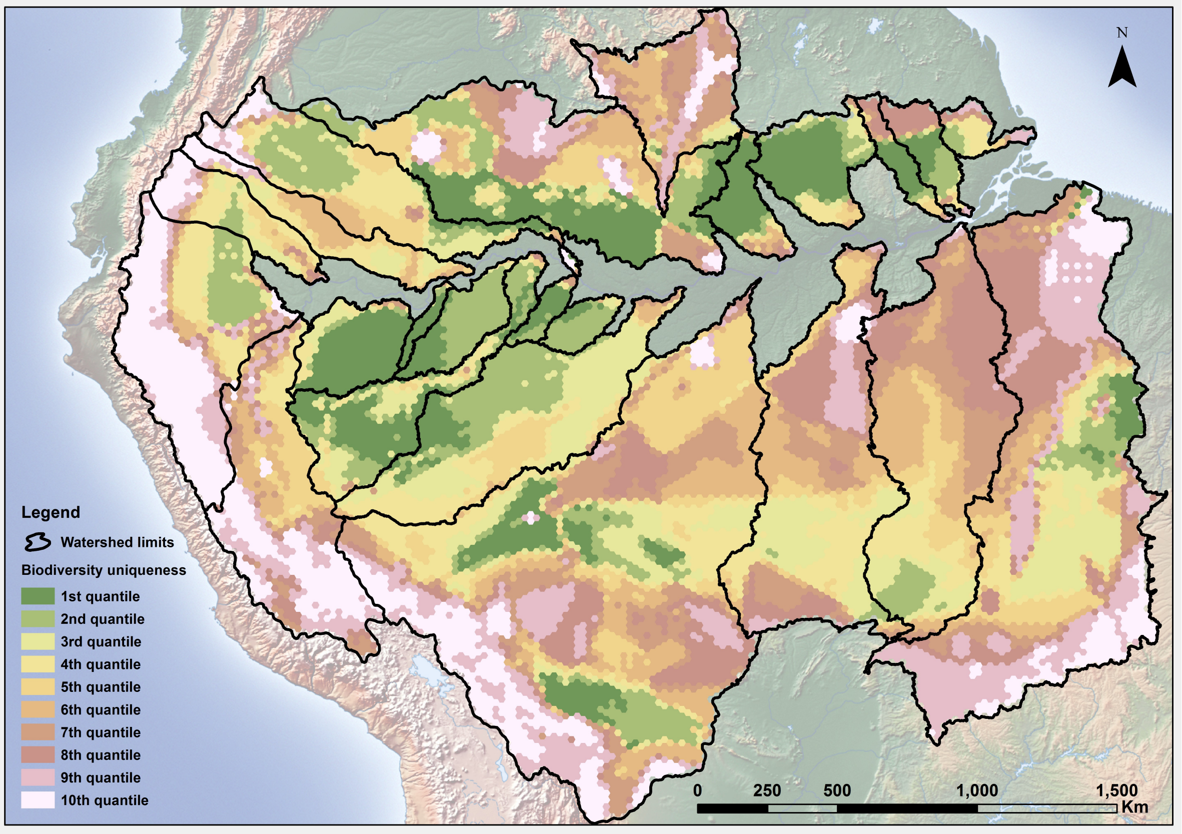
Thieme, M. et al., 2007. Freshwater conservation planning in data-poor areas: An example from a remote Amazonian basin (Madre de Dios River, Peru and Bolivia). *Biological Conservation*, 135(4), pp.484–501.

Tulloch, V.J. et al., 2015. Why do we map threats? Linking threat mapping with actions to make better conservation decisions. *Frontiers in Ecology and the Environment*, 13(2), pp.91–99.

Walker, B.H., 1992. Biodiversity and Ecological Redundancy. *Conservation Biology*, 6(1), pp.18–23.

Appendices

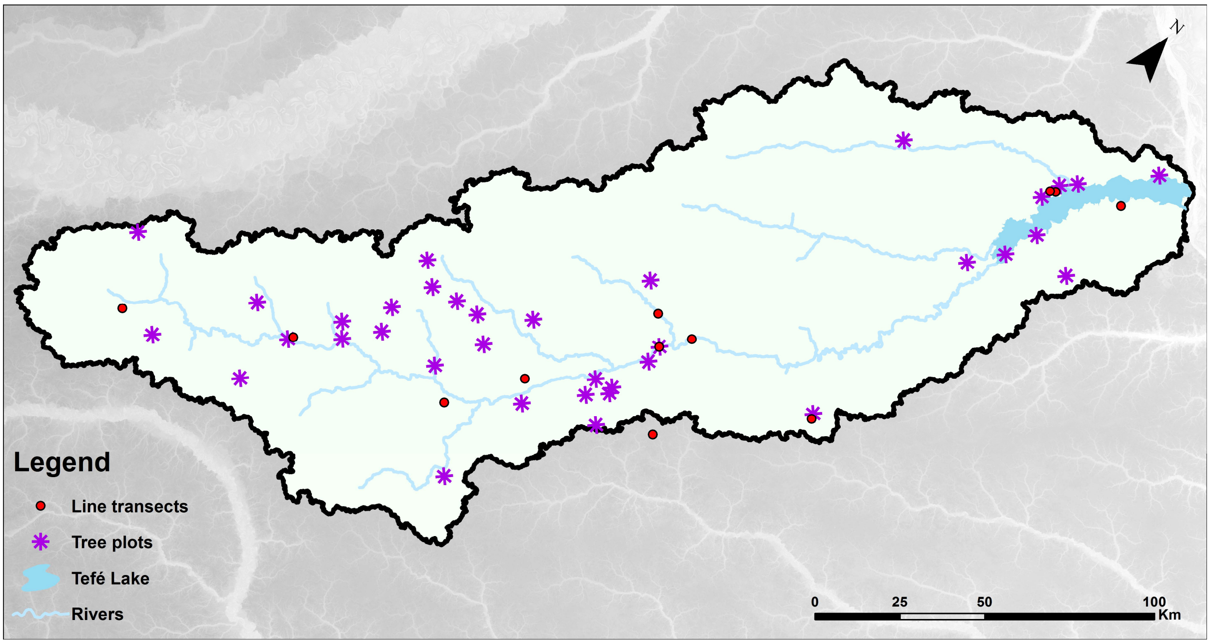
Apendix 4.1 - Geographic distribution of biodiversity uniqueness throughout the 23 major Amazonian watersheds. Biodiversity uniqueness scores are colour-coded according their distribution quantiles from green (lower) to pink (higher). The values are associated with 500 km^2 hexagonal cells.



Apendix 5.1 – Amazonian plant genera considered vulnerable to recruitment limitation generated by chronic overhunting of their seed dispersers (*sensu* Peres & Roosmalen 2002)

Abarema
Abuta
Ampelocera
Anacardium
Aniba
Annona
Anomospermum
Apeiba
Brosinum
Calophyllum
Caryomene
Casearia
Cassia
Catostemma
Cayaponia
Chelloclinum
Chrysophyllum
Clarisia
Couepia
Coussapoa
Dialium
Dicranostyles
Dilkea
Dimorphandra
Diospyros
Discophora
Duguetia
Duroia
Ecclinusa
Emmotum
Enterolobium
Ephedrantus
Eugenia
Fusaea
Gnetum
Guarea
Guazuma
Gustavia
Helicostylis
Heteropsis
Hymenaea
Inga
Iryanthera
Lacunaria
Leonia
Leretia
Licania
Licaria
Lysiostyles
Manilkara
Maquira
Maripa
Matisia
Micropholis
Minquartia
Monstera
Mora
Moronobea
Mouriri
Moutabea
Naucleopsis
Nectandra
Ocotea
Omphalea
Orthomene
Osteophloem
Parinari
Passiflora
Paullinia
Peltogyne
Perebea
Peritassa
Philodendron
Platonia
Poraqueiba
Pouteria
Pradosia
Ptychopetalum
Quiina
Rheedia
Rhodostemonodaphne
Rollinia
Sacoglottis
Salacia
Schistostemon
Senna
Simaba
Simarouba
Socratea
Spondias
Strycnos
Stryphnodendron
Swartzia
Syagrus
Symphonia
Talisia
Theobroma
Tontelea
Tovomita
Trymatococcus
Vantanea
Zygia

Apendix 5.2 – Distribution of tree plots (purple asterisks) and mammals' census (red dots) performed in the Tefé watershed.



Apendix 5.3 – Biomass estimates (kg/km2) of caviomorph rodents, ungulates and primates for 12 sites in Tefé watershed, Central Amazônia.

Site	Total biomass (kg/km2)	Biomass estimates (kg/km2)								
		Alouatta seniculus	Ateles	Callicebus cupreus	Callicebus torquatus	Cebus albifrons	Cebus apella	Dasyprocta	Lagothrix	Mazama americana
1	983.5	27.6	63.5	0.1	2.2	18.1	25.1	13.0	118.5	30.1
2	1192.7	17.2	63.5	0.2	3.6	67.0	74.5	13.0	118.5	30.1
3	1049.9	17.2	63.5	0.3	2.4	30.2	26.8	13.0	118.5	30.1
4	985.7	2.1	0.9	0.0	2.7	0.0	18.7	13.6	123.6	70.3
5	504.7	2.6	6.2	0.8	1.0	2.2	18.6	21.9	120.2	80.5
6	611.5	0.5	10.8	0.0	4.5	0.0	18.7	10.1	200.0	28.2
7	809.9	54.6	11.1	0.9	2.8	1.5	24.0	26.6	121.2	23.7
8	681.2	1.0	0.0	0.9	4.9	17.4	31.0	18.2	170.7	8.2
9	848.4	2.5	39.0	0.3	0.1	0.6	13.7	17.1	185.0	24.1
10	921.4	2.1	42.5	0.0	1.9	6.9	15.0	19.6	105.2	10.9
11	759.8	23.9	62.0	0.1	5.1	5.8	15.4	6.2	114.0	14.9
12	841.3	1.0	27.4	0.6	0.9	15.8	29.1	1.6	244.7	12.7

Site	Mazama gouazoupira	Pecari tajacu	Pithecia	Saguinus fuscicolis	Saguinus mystax	Saimiri	Tapirus terrestris	Tayassu pecari
1	29.1	170.0	21.1	3.1	4.6	4.2	345.6	107.5
2	29.1	170.0	31.7	12.5	21.1	62.2	371.2	107.5
3	29.1	170.0	15.8	11.0	18.7	24.6	371.2	107.5
4	46.1	77.0	2.5	12.4	21.0	3.3	486.4	105.0
5	18.7	4.1	11.1	4.5	7.0	4.6	145.9	54.8
6	15.1	172.0	12.8	8.5	16.1	0.0	114.1	0.0
7	25.5	73.2	9.8	3.1	5.6	14.9	340.9	70.4
8	10.5	176.0	5.3	6.6	11.1	0.0	206.5	12.8
9	82.3	99.6	8.0	5.9	11.7	0.0	358.4	0.0
10	8.7	47.0	3.9	8.2	17.2	0.1	422.4	209.9
11	5.9	44.0	3.3	2.3	4.1	8.5	347.1	97.3
12	14.8	70.2	7.2	4.3	7.1	7.0	396.8	0.0

Sites: 1 – Acaituba , 2 – Ponta da Castanha logged , 3- Ponta da Castanha unlogged , 4 – Acácia, 5 - Vila Moura, 6 – Bate 1, 7 – Curimatá, 8 – Oleoduto, 9 – Bate 2, 10 – Cacau, 11 – Ciane, 12- AltoTefé