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Global mammal conservation:

past trends, current challenges and optimal strategies for intervention

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PhD Thesis

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In 2012 the global human population has surpassed 7 billion people, more than twice the value registered 50 years before. The Caribbean Monk Seal, the Saudi Gazelle and (probably) the Yangtze River Dolphin went extinct during that 50 years, together with tens of other mammals, birds and amphibians. Meanwhile, the Black-footed Ferret, the Asian Wild Horse and a few more species were rescued from the brink of extinction. This PhD thesis attempts to describe how human choices determine mammal species' fate, and how conservation science can influence such choices.

The Scimitar-horned Oryx (*Oryx dammah*) is today extinct in the wild, while the Asian Wild Horse (*Equus ferus*; on the background) has been reintroduced to its original habitat. Image property of Moreno Di Marco.

The Indian Rhino population (*Rhinoceros unicornis*) is today increasing in number, and is an example of successful conservation intervention. Image courtesy of Federica Chiozza.

GLOBAL MAMMAL CONSERVATION

Past trends, current challenges and optimal strategies for intervention

Contents

Structure of the thesis

This thesis reports on the advances and findings of a 42-months PhD research project within the Global Mammal Assessment lab, Department Biology and Biotechnology, Sapienza University of Rome. After an inital summary (both in English and in Italian), the text is divided into 6 chapters and is based on 4 research papers that represent the key scientific output of the PhD project and 4 additional PhD-related papers that are discussed, together with the others, in the final chapter (see "List of papers" below, for details).

The first chapter provides a general background of the PhD project. The Ecological context that qualifies the PhD project is described in this chapter, with particular reference to the science of Conservation Biology and the disciplines of Systematic Conservation Planning and Comparative Extinction Risk Analysis. The necessity for global scale conservation planning is presented, as well as the opportunity that global scale planning provides toward the implementation of cooperative conservation strategies for mammals. A description of the Global Mammal Assessment program (GMA) is also provided, together with the specific role that the PhD project has had within the initiative. The gap in scientific knowledge that justified the rationale of this research is described in detail. The focus, the goal and the objectives of the PhD project are described in this chapter.

Four research chapters follow the introduction section, and are the core scientific output of the PhD project. Each of those chapters takes inspiration from a scientific paper written during the PhD activity (referred to in each of the chapter headings).

Chapter II is the first research chapter. This chapter provides a detailed description of one of the core GMA products, the habitat suitability models for terrestrial mammal species. The process of creating the models is described, together with the validation of model accuracy. Global patterns of species distribution and richness are investigated and compared to biogeographical patterns, species taxonomy, species habitat requirements and current species risk of extinction. The main aim of this chapter is to describe the distribution of species' suitable habitat at a global scale, and to characterise ecological processes determining it. The key output of this research chapter is the definition of species-specific habitat relationships that refine information on global mammal species distribution as well as the distribution of species assemblages.

Chapter III is the second research chapter. This chapter provides the detailed description of a 2 years project that is the result of a collaboration with experts of the IUCN Red List belonging to several research institution worldwide. This chapter presents a 40 year story of global conservation success and failure for the world's carnivores and ungulates (two groups that include many iconic species). A systematic analysis of past literature sources, as well as a collection of contextual information provided by experts were performed to obtain information on past species conservation status. Synthetic indicators of species extinction risk were then calculated to track changes in the global trend of species decline. The main aim of this chapter is to characterise the spatial and temporal trends in the extinction risk of the world's carnivores and ungulates from the 1970s up to date. The key finding of this chapter is that a negative trend in global species conservation status was already ongoing 40 years ago and has exacerbate recently due to geopolitical changes and unsustainable grow of resources consumption by humans, especially in species-rich areas.

Chapter IV is the third research chapter. This chapter provides the detailed description of a multidisciplinary research project, where techniques of spatial modelling, extinction risk analysis and spatial conservation prioritization were combined to identify global conservation priorities for an effective reduction of mammal species extinction risk. The main aim of this chapter is to show that species' biological traits (such as body mass or reproductive parameters) could be used to determine a species' ability to recover from a declining condition. The main finding of this chapter is the identification of priority species that are currently threatened with extinction yet likely recoverable, due to their biological characteristics. Key conservation areas, where priority species are found, are only partially protected despite their conservation relevance has been identified also under other conservation prioritization schemes.

Chapter V is the last research chapter. This chapter provides a detailed description of a research project whose relevance lies between the topics of conservation planning and extinction risk analysis. This chapter describes the methodological challenges and the potential shortfalls of measuring the level of threat to which a species is exposed. The main aim of this chapter is to demonstrate that the variability characterising the process of measuring the threat level to which a species is exposed has critical knock-on effects for conservation. The main finding of this chapter is that common measures of threat level for species may poorly represent the real threat effect in determining a species' risk of extinction.

A detailed analytical process demonstrates how to deal with threat variability in a way that maximise the reliability of a threat quantification process.

Chapter VI is the last chapter, and provides a general discussion on the contribution of the PhD project to the improvement of knowledge and methodologies for global mammal species conservation. A critical evaluation of the objective and scope presented in the introduction is provided. A description of the project outcomes and their relevance for conservation is given. Finally future research directions are discussed.

Summary

The conservation of the world's mammals has been one of the main goals for international agencies and NGOs in the past decades, as well as a core challenge for conservation scientists. Nonetheless, several mammal species went extinct in the past century and many others declined seriously in recent decades (a total of 76 species disappeared after 1500). Today 25% of mammal species are threatened with extinction, and the necessity of a globally coordinated conservation strategy has been proclaimed. At a global scale, factors such as habitat loss and direct kill, are the key drivers of mammal species decline and extinction. The current biodiversity crisis, and in particular the current decline of mammals, needs operational tools to be tackled. Such tools include both on-the-ground implementation of conservation actions and a well framed conservation decision making process. Yet, a poor integration of macroecological studies and conservation planning studies has characterised past strategies for global species conservation. The scope of this PhD thesis is to advance the knowledge on global mammal conservation, by integrating existing data and by combining multi-disciplinary methodologies to provide innovative approaches to conservation decision making. The objectives of this thesis are: (i) to characterise global patterns in the distribution of suitable habitat for mammals; (ii) to describe patterns and processes characterising the recent decline of mammal species; (iii) to provide insights into the characterisation of internal sources of mammal species extinction risk and to use them in conservation planning; (iv) to address the methodological challenges that affect the quantification of threats to biodiversity.

(i) A key aspect determining the effectiveness of mammal species conservation is the knowledge of species distribution. However, large-scale and fine-resolution information on mammal distribution has often been lacking. We filled this gap by developing habitat suitability models for over 90% of existing terrestrial mammal species, based on their habitat preferences with a 300m pixel base. Based on the developed models, we conducted a global, fine-resolution analysis of patterns of species richness. We found that the richness of mammal species estimated by the overlap of their suitable habitat is on average one-third less than the one estimated by the overlap of their geographic ranges. Differences in the proportion of suitable habitat within mammal geographic ranges were correlated to species IUCN Red List category, with threatened species having smaller values than non-threatened ones.

(ii) Assessing temporal changes in species' extinction risks is necessary for measuring conservation success or failure at large spatial and taxonomical scales. Yet such a knowledge is limited even in well-studied group, such as mammals. We addressed this gap by consulting past literature sources, and assigning retrospective Red List categories to the world's carnivores and ungulates (n=550) for the past 40 years. We found that 23% of species moved one or more categories closer to extinction, with Southeast Asia showing the steepest declines (where the figure is 45%). We described a declining trend in mammal conservation status that was already underway 40 years ago, and has now accelerated, uncovering alarming future scenarios of global species losses. This declining trend is the ultimate result of geopolitical events, international regulations, shifting cultural values and natural resource exploitation. Studying trends in global species decline can help conservationists to recognize which conservation policies and strategies are (or are not) contributing to biodiversity protection.

(iii) With one fourth of the world's mammals threatened with extinction and limited budget to save them, adopting an efficient conservation strategy is crucial. Previous approaches to setting global conservation priorities have assumed all species to have equal conservation value, or have focused on species with high extinction risk, species that may be hard to save. We propose a novel approach and focus on threatened species having the greatest recovery opportunity, using a new conservation benefit metric. We discover that 65-87% of all threatened and potentially recoverable mammal species are overlooked by existing prioritization approaches. Our innovative metric has the potential to integrate with every strategy that aims to maximize the likelihood of conservation success.

(iv) Anthropogenic threats drive species to extinction and are the focus of extinction risk analysis and conservation planning. However, the effects that multiple threats quantification methods have on threat measurement remain untested. We addressed this gap by quantifying the magnitude of human footprint for 901 Southeast Asian mammals according to several methods. We demonstrate that adopting different threat quantification techniques causes significant disagreement in threat measurements. We found that biases were non-linear and dependent on distal factors, such as the proportion of suitable habitat within species' range and species' habitat specialism. Threatened species were best predicted when measuring their proportion of range exposed to high levels of human footprint (a proxy of threat extent). Improving techniques to quantify biodiversity threats will enhance the effectiveness of extinction risk analyses and conservation decisions.

Based on the outcomes of the presented PhD research, we propose that three main factors influence the global extinction risk of mammal species: human threats, species biology and conservation actions. The role of these factors in determining species decline or recovery, has been explored both theoretically and analytically. However, our understanding of how they interact to determine a species' final condition of endangerment is still limited. A key research challenge in the next future would be the exploration of such an interaction. In this thesis, we set a basis for the realization of such an analysis and the next step will be the investigation of the combinatory effects of all the mentioned factors. The urgency to properly address the current biodiversity crisis calls for a more coordinative effort in defining shared global strategies for intervention. Having a global coordination of conservation plan has a great potential to be more cost-effective than having several scattered plans, without contrasting the need for local scale practical interventions. We believe that increasing the biological inputs in conservation planning, through the consideration of species' biological characteristics, represents a promising field of future research expansion where expertise from multiple backgrounds can be integrated to define innovative strategies to address global scale extinction risk. The research findings presented in this thesis will contribute to improve future mammal conservation by: guiding the definition of more biologically-informed conservation strategies, improving our ability to analyze evidence of conservation success, providing general guidelines to address methodological uncertainty in conservation.

Riassunto [*In Italian*]

La conservazione dei mammiferi a scala globale è stato uno degli obiettivi principali per le agenzie di conservazione internazionali e per quelle non governative nei decenni passati, oltre che una delle sfide principali per i biologi della conservazione. Ciononostante, numerose specie di mammiferi si sono estinte nell'ultimo secolo e molte altre hanno subito un forte declino (in totale 76 specie si sono estinte dopo il 1500). Oggi il 25% delle specie di mammiferi è minacciato di estinzione, e la necessità di una strategia di conservazione a scala globale è stata proclamata. A scala globale, fattori come la perdita di habitat e l'uccisione diretta delle specie sono tra le cause principali del declino e dell'estinzione delle specie. L'attuale crisi di biodiversità, e in particolare l'attuale declino dei mammiferi, necessitano di strumenti operativi per essere affrontati. Questi strumenti includono sia l'implementazione sul territorio di azioni di conservazione, che la definizione di un processo decisionale efficiente. Purtroppo però, una scarsa integrazione di studi macroecologici e di pianificazione della conservazione ha caratterizzato le strategie di conservazione a scala globale in passato. Questa tesi di dottorato si propone di avanzare lo stato delle conoscenze sulla conservazione globale dei mammiferi, attraverso l'integrazione dei dati esistenti e la combinazione di metodi multidisciplinari per fornire approcci innovativi al processo decisionale in conservazione. Gli obiettivi sono: (i) caratterizzare i patterns globali di distribuzione dell'habitat idoneo delle specie di mammiferi; (ii) descrivere i patterns e i processi che hanno caratterizzano il recente declino delle specie di mammiferi, (iii) fornire approfondimenti nella caratterizzazione delle fonti di variabilità interna (biologica) che influiscono sul rischio di estinzione delle specie di mammiferi e mostrare metodi per utilizzarle in ambito di conservazione; (iv) affrontare il problema dell'incertezza nella quantificazione delle minacce alle specie.

(i) Un aspetto centrale nel determinare l'efficacia delle strategie di conservazione delle specie è la conoscenza della loro distribuzione. Ad oggi però, mancano informazioni su vasta scala e ad alta risoluzione sulla distribuzione delle specie. Noi abbiamo colmato questa lacuna sviluppando modelli di idoneità ambientale per oltre il 90% dei mammiferi terrestri, sulla base delle loro preferenze di habitat. Sulla base dei modelli sviluppati, abbiamo condotto un'analisi globale sui pattern di ricchezza specifica. Abbiamo trovato che la ricchezza di specie di mammiferi, stimata sulla base della distribuzione dell'habitat idoneo, è in media minore (di

circa un terzo) rispetto alla ricchezza sulla base del loro areale di distribuzione. Le differenze nella proporzione di habitat all'interno dell'areale delle specie è correlata alla loro categoria di minaccia IUCN, con le specie minacciate che hanno valori minori rispetto a quelle non minacciate.

(ii) Valutare i cambiamenti temporali nel rischio di estinzione delle specie è necessario per misurare il successo o il fallimento delle azioni di conservazione, ma tale informazione è spesso limitata, anche in gruppi ben studiati come i mammiferi. Noi abbiamo colmato questa lacuna consultando fonti bibliografiche passate,e valutando retrospettivamente lo status di minaccia dei carnivori e degli ungulati a scala globale (n=550). Abbiamo trovato che il 23% delle specie in questi gruppi si è avvicinato all'estinzione tra il 1975 e il 2008, con il Sudest Asiatico che ha mostrato il declino più marcato (con una percentuale che sale al 45%). Abbiamo descritto un trend di declino nello status di conservazione delle specie che era già in corso 40 anni fa ed è accelerato recentemente, lasciando intravedere allarmanti scenari di estinzione futura. Questo trend di declino è il risultato ultimo di eventi geopolitici, regolamenti internazionali, cambiamento dei valori culturali e livelli di sfruttamento delle risorse negli ultimi 40 anni. Studiare i trends globali di declino delle specie può aiutare i biologi della conservazione a riconoscere quali strategie stanno (o non stanno) contribuendo alla salvaguardia della biodiversità.

(iii) Con un quarto delle specie di mammiferi minacciate di estinzione e alla luce delle limitate risorse per salvaguardarle tutte, adottare una strategia di conservazione efficiente è cruciale. I precedenti approcci di definizione delle priorità di conservazione hanno assunto che tutte le specie avessero un eguale valore di conservazione, o in alternativa si sono concentrati su specie ad alto rischio di estinzione, specie che potrebbe essere difficile recuperare. Noi proponiamo un nuovo approccio e ci concentriamo sul quelle specie minacciate che hanno la più alta probabilità di poter essere recuperate (grazie alle loro caratteristiche biologiche), attraverso la definizione di una nuova metrica di conservazione. Abbiamo scoperto che circa il 65-87% delle specie di mammiferi minacciate e potenzialmente recuperabili non sono oggi considerate come prioritarie dalle principali strategie di conservazione. La nostra metrica innovativa ha la potenzialità di integrarsi con tali strategie al fine di massimizzare la probabilità di successo delle azioni di conservazione.

(iv) Le minacce antropiche determinano l'estinzione delle specie e sono il focus principale sia delle analisi del rischio di estinzione che della pianificazione delle azioni di conservazione. Ad oggi però, gli effetti che molteplici metodi per quantificare le minacce hanno nel misurare l' impatto sulle specie non sono stati verificati. Noi abbiamo colmato tale lacuna quantificando su base spaziale il livello di impatto antropico per 901 mammiferi del Sudest Asiatico, attraverso molteplici metodi. Abbiamo dimostrato che l' adozione di varie tecniche di quantificazione causa un disaccordo nella misurazione dei livelli di minaccia per le specie. Le differenze nelle misurazioni sono tipicamente non lineari e sono influenzate da fattori come la proporzione di habitat idoneo all'interno dell'areale delle specie o il loro livello di specificità nella scelta dell'habitat. La misurazione della porzione di areale delle specie esposto ad alti livelli di impatto umano (cioè una rappresentazione dell'estensione della minaccia) è risultata essere il metodo migliore per predire il rischio di estinzione delle specie. Migliorare le tecniche di quantificazione delle minacce per la biodiversità può incrementare l'efficienza delle analisi di rischio di estinzione e l'efficacia delle azioni di conservazione.

Sulla base dei risultati delle ricerche qui presentate, abbiamo individuato 3 fattori principali che influenzano il rischio di estinzione globale delle specie di mammiferi: le minacce antropiche, la biologia delle specie e le azioni di conservazione. Il ruolo di tali fattori nel determinare il declino o il recupero delle specie è stato esplorato sia teoricamente che analiticamente. Per ora, la nostra comprensione del modo in cui tali fattori interagiscono per determinare la condizione di rischio di una specie è ancora limitata. Un importante fronte di ricerca per il prossimo futuro sarà l'esplorazione di tale interazione. In questa tesi abbiamo definito le basi analitiche per la realizzazione di questo tipo di ricerca e i prossimi passi consisteranno nell'investigazione dell'effetto combinato dei fattori sopra menzionati. L'urgenza di affrontare l'attuale crisi della biodiversità porta alla luce la necessità di definire azioni e strategie di conservazione più coordinate. Avere una maggiore coordinazione a scala globale per i piani di conservazione ha il potenziale di essere più efficace economicamente rispetto ad avere tante strategie indipendenti, senza per questo contrastare il bisogno di avere interventi coordinati a scala locale. Incrementare l'uso di dati biologici nella definizione di strategie di conservazione rappresenta un promettente settore di espansione per le future ricerche, attraverso l'integrazione di conoscenze multidisciplinari e la definizione di metriche innovative. I risultati della presente ricerca contribuiranno a migliorare le future strategie di conservazione dei mammiferi attraverso: la definizione di priorità basate su informazioni biologiche sempre più accurate, il miglioramento della nostra capacità di analizzare le evidenze del successo di conservazione, la generazione di linee guida per risolvere l'incertezza metodologica che caratterizza le scelte di conservazione.

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List of papers associated with the PhD Thesis

This thesis is based on 8 scientific papers that have been written during a 42 months PhD research (between November 2009 and July 2013). Four papers represent the core scientific output of this PhD, while the remaining 4 represent PhD-related research outputs.

Key Research papers

Rondinini, C., M. Di Marco, F. Chiozza, G. Santulli, D. Baisero, P. Visconti, M. Hoffmann, J. Schipper, S.N. Stuart, M.F. Tognelli, G. Amori, A. Falcucci, L. Maiorano and L. Boitani. (2011) Global habitat suitability models for terrestrial mammals. Philosophical Transactions of the Royal Society B, 366 (1578): 2633-2641 .

Di Marco, M., L. Boitani, D. Mallon, A. Iacucci, P. Visconti, E. Meijaard, J. Schipper, M. Hoffmann, C. Rondinini. Lessons from the past:a retrospective evaluationof the global decline of carnivores and ungulates. *Submitted.*

Di Marco, M., C. Rondinini, L. Boitani, K. A. Murray. (2013) Comparing multiple species distribution proxies and different quantifications of the Human Footprint map, implications for conservation. Biological Conservation, 165: 203–211.

Di Marco, M., M. Cardillo, H.P. Possingham, K.A. Wilson, S.P. Blomberg, L. Boitani and C. Rondinini. (2012). A Novel Approach for Global Mammal Extinction Risk Reduction. Conservation Letters, 5 (2): 134-141.

Other PhD-related research papers

Rondinini, C, M. Di Marco (corr. author), P. Visconti, S.H.M. Butchart, L. Boitani. (2013) Update or outdate: long-term viability of the IUCN Red List. Conservation Letters *doi: 10.1111/conl.12040.*

Visconti, P., M. Di Marco, M. (corr. author), J.G. Álvarez-Romero, S.R. Januchowski-Hartley, R. L. Pressey, R. Weeks, C. Rondinini. (2013) Effects of errors and gaps in spatial data sets on assessment of conservation progress. Conservation Biology, *In press.*

Rondinini, C., L. Boitani, A.S.L. Rodrigues, T.M. Brooks, R.L. Pressey, P. Visconti, J.E.M. Baillie, D. Baisero, M. Cabeza, K.R. Crooks, M. Di Marco, K.H. Redford, S.A. Andelman, M. Hoffmann, L. Maiorano, S.N. Stuart and K.A. Wilson. (2011) Reconciling global mammal prioritisation schemes into a strategy. Philosophical Transactions of the Royal Society B, 366 (1578): 2722-2728.

Wilson, K.A., M.C. Evans, M. Di Marco, D.C. Green, L. Boitani, H.P. Possingham, F. Chiozza, and C. Rondinini. (2011). Prioritising conservation investments for mammal species globally. Philosophical Transactions of the Royal Society B, 366 (1578): 2670-2680.

List of conferences contributions associated with the PhD Thesis

The research arguments included in this PhD thesis have been presented as contributions in various international conferences in the form of oral presentations (or poster in one case). A selection of the most relevant contributions is provided below.

Key research contributions

Di Marco, M., L. Boitani, D. Mallon, M. Hoffmann, A. Iacucci, E. Meijaard, P. Visconti, J. Schipper, C. Rondinini. Lessons from the past: a retrospective evaluation of the global decline of carnivores and ungulates. Internation Conference on Conservation Biology (Baltimore, USA, 21-25 July 2013; **Student awards, selected finalist**) [*Forthcoming*]

Di Marco, M., Rondinini C., Boitani L, Murray KA. Implications of using different methods for quantifying threats to biodiversity. Student Conference on Conservation Science (Cambridge, 19-21 March 2013) [*Poster*]

Di Marco, M. Mammal species decline: past trends, current challenges, and possible conservation strategies. **Invited Seminar** at Zoological Society of London. (London, 10th December 2012)

Di Marco, M., C. Rondinini, F. Chiozza, G. Amori, R. Young and L. Boitani. 2012. Detecting priority species for reassessment improves the Red List efficiency in estimating extinction risk. European Congress of Conservation Biology (Glasgow, UK, 28-Aug 1-Sep 2012; **Student awards finalist, 3rd place**)

Di Marco, M., M. Cardillo, H.P. Possingham, K.A. Wilson, S.P. Blomberg, L. Boitani and C. Rondinini. 2011. A Novel Approach for Global Mammal Extinction Risk Reduction. International conference on Conservation Biology (Auckland, New Zealand, 5-9- Dec. 2011)

Rondinini, C., M. Di Marco (presenting), P. Visconti, L. Boitani. 2011. The current and future distribution of mammalian habitat. International conference on Conservation Biology (Auckland, New Zealand, 5-9- Dec. 2011; **Invited**)

Other PhD-related research contributions

Rondinini, C., Visconti P., Di Marco M., Brooks T.M. 2012. The Role of Biodiversity Scenarios in the Development of Global Conservation Strategies. European Congress of Conservation Biology (Glasgow, UK, 28-Aug 1-Sep 2012)

Chiozza F. Boitani L., Santulli G., Baisero D., Visconti P., Di Marco M., Rondinini C. 2010 Global habitat suitability models for mammal conservation. International Conference on Conservation Biology (Edmonton, Alberta, Canada 2010).

Rondinini, C, Chiozza, F, Di Marco, M, Visconti, P, Baisero, D, Boitani, L. 2010. The Global Distribution of Mammals. International Conference on Conservation Biology (Edmonton, Alberta, Canada 2010).

Wilson, K.A, Boitani, L, Chiozza, F, Evans, M.C, Di Marco, M, Possingham, H.P, Rondinini, C. 2010. Prioritising conservation investments for mammal species globally. International Conference on Conservation Biology (Edmonton, Alberta, Canada 2010).

CHAPTER I

General introduction

1. General introduction

1.1. The global decline of mammalian biodiversity: processes, threats and current knowledge

1.1.1. Mammal species loss

After the end-Cretaceous mass extinction, an evolutionary radiation has led to the diversification of forms that characterises the present-day mammals (Bininda-Emonds et al., 2007). The existing 5,500 mammal species (IUCN, 2012) show a variety of characteristics that relates both to their physiological adaptations to multiple environment and to their diversified ecological role. Mammals have colonized all of the earth's major habitats, from oceans to mountains' top (with the exception of the internal portions of the Arctic and the Antarctic). The variety of forms characterising mammal species encompasses 8 orders of body mass magnitude, from the few grams of the Bumble-bee Bat, *Craseonycteris thonglongyai*, and the Etruscan Pygmy Shrew, *Suncus etruscus*, to the 150 tons of the Blue Whale, *Balaenoptera musculus* (Wilson & Reeder, 2005). Mammals play key ecological roles (Hoffmann et al., 2011), e.g. for regulating vegetation cycles and seed dispersal (Fragoso et al., 2013; Pringle et al., 2007), and provide important human benefits such as food (Fa et al., 2003) and economic income from tourism (Di Minin et al., 2012).

The conservation of the world's mammals, has been one of the main goals for international agencies and NGOs in the past decades, as well as a core challenge for conservation scientists (Schipper et al., 2008; Hoffmann et al., 2010, 2011;). Together with birds, mammals receive a disproportionate amount of conservation attention respect to other groups (Clark & May, 2002). Nonetheless, some 255 species of mammals have gone extinct in the last 10,000 years (Turvey, 2009) of which one-third were lost in the last 500 years (Schipper et al., 2008).

The past 5 decades were characterised by the loss of small mammals, such as the Guam Flying Fox, *Pteropus tokudae* (last seen in the early 1970s), as well as the loss of large-bodied mammal species, such as the Caribbean Monk Seal, *Monachus tropicalis* (last seen in 1952), the Saudi Gazelle, *Gazella saudiya* (last seen in the 1980s), and probably the Yangtzee River Dolphin, *Lipotes vexillifer* (last confirmed report in 2002). Such events depict a figure of mammalian extinction that is progressing at a much faster pace than that registered from fossil records (1.8 extinctions/million species years), in a global context where the rate of species loss is three order of magnitude higher than normal (Barnosky et al., 2011).

1.1.2. Mammal species conservation

Today, 25% of mammal species are threatened with extinction (Hoffmann et al., 2010), and the necessity of a globally coordinated conservation strategy has been proclaimed (Rondinini et al., 2011a; Rondinini et al., 2011b). At a global scale, factors such as habitat loss (due mainly to expanding agriculture and logging) and overexploitation, are the key drivers of mammal species decline and extinction (Hoffmann et al., 2011). Nonetheless, future projections of land use change (Visconti et al., 2011) and climate change (Maiorano et al., 2011) uncover even more alarming scenarios of expected species loss.

Site-level conservation, whose main outcome is the definition of protected areas, has not always been successful in stopping species decline, for example in Southeast Asia (Craigie et al., 2010) and Africa (Gaveau et al., 2012). Moreover, in areas such as Australia, threats are often represented by the presence of introduced species (i.e. predators or competitors; Evans et al., 2011), making the creation of protected areas ineffective for species protection.

A recent meta-analysis of global biodiversity conservation indicators shows that an overall deteriorating trend has characterised global biodiversity, in spite of global commitments to halt biodiversity loss (Butchart et al., 2010). Conservation scientists argued that such a declining condition would have been even more accentuated without the implementation of conservation actions at a global scale (Hoffmann et al., 2010). The global conservation status of mammal species has deteriorated between 1996 and 2008 (Hoffmann et al., 2011), and many areas characterised by high species richness and endemism are today facing alarmingly high levels of habitat loss and overhunting, especially in the tropics (Schipper et al., 2008).

The current biodiversity crisis, and in particular the current decline of mammal species needs operational tools to be tackled. Such tools include both on-the-ground implementation of conservation actions and a well-framed conservation decision making process. Analytical tools for supporting conservation of mammals, such as Systematic Conservation Planning and Extinction Risk Analysis, are introduced below and the (poorly explored) potential for their integration provides a background for the scope of the presented PhD research (see section 1.5).

1.2. Mammals as a model group in conservation

Mammals include 5500 species divided into 27 orders, they represent 8.82% of all currently described vertebrates and 0.27% of all described eukaryotic taxa, where 85% of all predicted-to-exist eukaryotic species (8.7M in total) have not been described yet (Hoffmann et al. 2010; Mora et al. 2011). Mammals are, together with birds, the most studied group in conservation biology (Clark and May 2002). Mammals (Schipper et al. 2008), Birds (Butchart et al. 2004) and Amphibians (Stuart et al. 2004) are fully covered with assessments of extinction risk in the IUCN Red List. On opposite, other groups of vertebrates, invertebrates plants and fungi, have been only partially (sometime marginally) assessed in the Red List so far, yet a proposal to better represent those groups in future assessments has been launched (Stuart et al. 2010).

As already mentioned, mammals play key ecological roles in ecosystems (e.g., grazing, predation, and seed dispersal) and provide important benefits to humans (e.g., food, recreation, and income), many comprehensive books on various aspects of natural history, zoology and ecology of mammals have been published, including the Encyclopedia of Mammals (McDonald 2009). Many of the research progresses in conservation biology are based on the use of mammal species as a model group (e.g. Ceballos et al. 2005), and biological and ecological information is generally better available for mammals than for other groups (e.g. Jones et al. 2009). Mammals include a bigger proportion of threatened species than birds and a lower proportion of threatened species than amphibians (Hoffmann et al. 2010). The number of known historically (>1500) extinct mammal species is higher respect to amphibians and lower respect to birds (IUCN 2012). Importantly, continental-scale studies demonstrated the potential synergies between conservation of mammals and conservation of other groups (specifically amphibians) and discussed the "umbrella effect" of mammals conservation (Rondinini & Boitani 2006).

In this thesis, mammals are used as model group to study the dynamics and processes of extinction risk and to investigate the definition of optimal strategies for conservation intervention. The analytical results presented in this thesis are thus related to mammals and depend on the settings and assumptions presented in each of the research chapters, yet both the key findings and the main conclusions reported have broader relevance. I expect that the techniques developed during my PhD and presented in this thesis, as well as the interpretation of the presented results, will be of general relevance for conservation and will have the potential to contribute to the definition of improved biodiversity conservation strategies.

In brief, the choice of working on mammals can be justified by the following factors: high bio-geographical representativeness, demonstrated umbrella effect respect to other groups, high proportion of threatened species, relatively wide data-availability, relatively good knowledge of on-going threatening processes, conspicuous body of literature sources available.

1.3. Conservation biology, Systematic Conservation Planning and Conservation Prioritization, an overview

1.3.1. Conservation Biology

In December 1985, a paper by Michael E. Soulé (Soulé, 1985) clarified that:

Conservation biology differs from most other biological sciences in one important way: it is often a crisis discipline. Its relation to biology, particularly ecology, is analogous to that of surgery to physiology and war to political science. In crisis disciplines, one must act before knowing all the facts; crisis disciplines are thus a mixture of science and art, and their pursuit requires intuition as well as information.

This description is still valid, in its general sense, after 27 years of expansion, improvements and applications of the conservation biology science. This is perhaps related to the fact that the state of the global biodiversity crisis is today more alarming than ever before, forcing countries' governments to agree on the necessity for an immediate coordinated intervention (CBD, 2010). What has clearly emerged, during the past 27 years of scientific investigation, is that the one way to control the balance between "intuition" and "information" as well as to disentangle the mixture between "art" and "science" (and between social and natural science) is to develop and implement quantitative analytical tools, often borrowed from other disciplines such as ecology, mathematics or economics. Many authors presented an overview of the history and advances of conservation biology, including: Pullin (2002), Primack (2010), Sodhi & Ehrlich (2010).

1.3.2. Systematic Conservation Planning

One milestone in the quantitative development of conservation biology as a scientific discipline was the formal description of Systematic Conservation Planning (SCP). SCP emerged as discipline to objectively regulate the creation of new protected areas, in an historical context where the establishment of protected areas was not directly aimed at representing and preserving biodiversity. In fact, PAs creation was initially driven by factors such as the former delineation of hunting reserves or the presence of large-bodied charismatic species, and the selected sites where often located in remote areas, with a low economic value (Pressey, 1994). The need for a better identification of global areas for conservation of biodiversity led scientists to the description of a theoretical framework divided into six sequential stages (Margules & Pressey, 2000). Through these stages, overall goals are set, together with quantitative targets, for the conservation of biodiversity (represented through the distribution of selected features, such as species) and a final set of priority interventions is designed to ensure that an adequate level of protection is achieved, while the overall conflict with economic activities and the costs of plan implementation are minimised (Margules & Pressey, 2000). "Conservation planning is the process of locating, configuring, implementing and maintaining areas that are managed to promote the persistence of biodiversity and other natural values" (Pressey et al., 2007).

1.3.3. Spatial Conservation Prioritization

The rise and development of SCP theories and techniques have been paralleled by the definition of quantitative tools aimed at supporting scientists in the process of "configuring" a protected area system. Back in 1980, Jamie Kirkpatrick applied the first heuristic algorithm to select a set of areas for the protection of endemic and endangered plant species, in order to complement the existing protected area network in Tasmania (Kirkpatrick, 1983). From that point on, a multitude of mathematical techniques have been implemented, and further developed, to solve what has been defined as the problem of "Spatial Conservation Prioritization" (Moilanen et al., 2009). Spatial conservation prioritization aims at identifying optimal, or near-optimal, solutions to two broad types of conservation problems: the "Minimum Set" problem and the "Maximal Coverage" problem (Wilson et al., 2009). The objective of the minimum set problem is to minimize the resources expended while meeting a given set of conservation targets (Pressey, 2002). In this case, a minimum level of protection (i.e. the conservation target) is ensured for each of the analysed biodiversity features (e.g. the species), at the minimum possible cost. The objective of the maximal coverage problem is to maximize the number of "conservation targets" met for the analysed biodiversity features, given that a limited (and likely insufficient) budget can be spent (Church & ReVelle, 1974). The maximal coverage problem thus resembles a real-world situation in which limited conservation funding is available and the efficiency of funding allocation must be maximised. Many algorithms and software have been developed to solve both these problems (Moilanen et al., 2009). What is common among several definitions of the prioritization problem is the consideration of three key measures driving the process of decision making in conservation: vulnerability, irreplaceability and complementarity. Vulnerability represents the risk of a given natural area being degraded due to a threatening process (Wilson et al., 2005). If a natural area, and the species occurring within it, is expected to be lost due to a concurring threatening process, such as deforestation or natural resources extraction, that area is highly vulnerable. Irreplaceability measures the extent to which the loss of an area will compromise regional conservation targets (Ferrier et al., 2000). In other words, this measure indicates what is the relative importance of a given site in terms of its global representation of biodiversity, e.g. the only site where an endemic species occurs is not replaceable with any other site. Complementarity is the principle of selecting areas that complement one another in terms of the assets conserved (Justus & Sarkar, 2002). Two areas highly rich in species may not necessarily represent the two best conservation options if they share common and widespread species; to avoid duplication of conservation efforts, after a species-rich area is selected, the next best option would be to select an area that add the highest number of non-yet-represented species. Complementarity of an area, as well as its irreplaceability, are thus evaluated in relation to the whole planning region.

In this thesis techniques of SCP and spatial conservation prioritization are used and refined to include the latest up-to-date information of mammal species distribution and biological characteristics to provide insights into the definition of global conservation strategies.

1.3.4. Prioritizing species conservation

The problem of allocating limited conservation funds across a number of threatened species is known as the problem of prioritizing species conservation (Rondinini et al. 2011) and was also referred to in the past as the "Noah's Ark problem" (Weitzman et al. 1998). In fact, selecting which species to consider relevant when defining priorities for short term conservation intervention implies choices that necessary will result in conservation trade-offs (or "conservation triage"; Bottrill et al. 2008). A variety of methods were proposed to identify priority species of conservation concern, and/or to quantitatively rank species according to particular metrics of conservation relevance.

Such techiques were often based oin relative species' vulnerability and rarity (e.g. Ricketts et al. 2005) or species evolutionary distinctiveness (e.g. Isaac et al. 2007). Other prioritization approaches were instead based on the consideration of expected probability of success of a set of proposed conservation projects for threatened species' recovery (Joseph et al. 2009), to maximise conservation cost-effectiveness.

In this thesis the utility of species prioritization for conservation purposes will be explored, and the potential expansion of this front will be investigate by using species' biological characteristics to approximate species' recovery probability. In Chapter IV I will combine information on species' biology, in association with their vulnerability, to determine short-term priorities for mammal extinction risk reduction at a global scale.

1.4. Extinction Risk Analysis, a tool to describe species extinction risk

In a recent paper Cardillo & Meijaard describe Extinction risk analysis as the discipline whose aim is "to discover and describe generalizations about patterns and processes in the decline or threat status of species" (Cardillo & Meijaard, 2012). Extinction risk analysis has emerged as a scientific base supporting conservation practice, by providing information on the factors that influences species risk of extinction. In the past 15 years in particular, biologists have been investigating the predictability of extinction risk from species' life-history traits as well as from the level of threats to which they are exposed (see Bielby et al., 2010; Cardillo & Meijaard, 2012). Mammals (Cardillo et al., 2005; Davidson et al., 2009) and, at a minor level, birds (Lee & Jetz, 2011) have been typically selected as model groups for this kind of studies.

A number of techniques were used to model the effect of factors influencing extinction risk, including: phylogenetic comparative methods (Purvis, 2008), where the influence of predictors in determining the extinction risk of phylogenetically-related species is mediated by species' shared evolutive history; machine learning methods, such as classification of trees and random forests, where the extinction risk condition of species is modelled as a combination of multiple variables' states, and species are recursively partitioned into groups that are increasingly homogeneous respect to the tested variables (Cutler et al., 2007). Recently, taxonomically-informed generalised linear mixed models have also been employed as an alternative to classical phylogenetic methods for mammals (González-Suárez & Revilla, 2012).

Many of the extinction risk models mentioned before used categorical response variables derived from the IUCN Red List to represent species' extinction risk. Such variables generally come in the form of either species extinction risk status or global species' population trend. In the former case, threat status categories from the IUCN Red List (IUCN, 2001) are converted into ranked values representing level of extinction risk (e.g. see Cardillo et al., 2005). In the latter case, species are classified as "deteriorating", "stable" or "improving" depending on the global situation of their populations (e.g. see Murray et al., 2011), again registered by IUCN Red List (www.iucnredlist.org).

The utility of extinction risk analysis for practical conservation purpose has been recently argued (Cardillo & Meijaard, 2012). In fact despite several scientific developments in that field, a poor connection with conservation action implementation was observed. According to Cardillo & Meijaard, reasons why this has happened include the fact that conservation is generally a reactive discipline (while the outcomes of extinction risk studies fit better for proactive approaches) and the fact that sometimes the outcomes of these studies are vague and of difficult interpretation for conservationists. Nonetheless, the same authors envisage that comparative studies of extinction risk can become more relevant to conservation practice, provided that a few conditions are met, namely: trying to focalise the analytical scope (e.g. by restricting the taxonomic focus); providing conservation recommendations that are clearly interpretable (e.g. by including predicted extinction risk in area prioritization algorithms); implementing predictive approaches to conservation planning; rethinking the general conservation expectation of extinction risk studies.

In this thesis, extinction risk analysis and SCP techniques are combined at various spatial (from regional to global), temporal (both present and past data) and taxonomic (from carnivores and ungulates to all mammals) scales, with the main goal of meeting the above mentioned conditions.

1.5. Red List as a tool to monitor extinction risk

In the face of an accelerating rate of global biodiversity loss (Butchart et al. 2010), and with a limited budget to mitigate negative human impacts on species persistence (James et al. 2001), conservation scientists need systems to identify species at risk of extinction: the Red List of the International Union for Conservation of Nature (IUCN) provides the most widely used system. Many conservation plans for species protection and/or recovery, especially among vertebrates, account for species' Red List status (Hoffmann et al. 2010), and the use and misuse of Red List categories in conservation planning has also been widely discussed (Lamoreaux et al. 2002; Possingham et al. 2003; Rodrigues et al. 2006). The IUCN Red List is the most updated source on species' threat status, yet it is incomplete in terms of coverage within biota (Stuart et al. 2010). For example, only 5% of the bony fishes has been evaluated by the Red List (therefore the rest is Not Evaluated, NE), and for 26% of all amphibians there was insufficient information to assess the extinction risk, therefore they were classified as Data Deficient (DD), meaning that there was insufficient information on population size, trends, distribution and/or threats to assess them against the Red List criteria (IUCN 2001). Overall, among vertebrates 41.35% of all species have been assessed against the Red List criteria, including all mammals birds and amphibians, and 13.84% of them are listed as DD (IUCN 2012).

Red List categories provide a descriptive tool to assess species risk of extinction, from low (Least Concern) to very high (Critically Endangered), based on a set of quantitative criteria (IUCN 2001) broadly developed according to Caughley's paradigms of "small population" and "declining populations" (Caughley 1994). Under the first paradigm a population may go extinct when consisting of a few individuals on the basis of stochastic dynamics. Under the second paradigm a population declines due to one or more external deterministic factors (e.g. a threatening factor). Red List categories have been used in the past 20 years to monitor progresses toward biodiversity conservation (Hoffmann et al. 2010), both at the level of single species and through synthetic indexes of relative group's distance from extinction, such as the Red List Index (RLI; Butchart et al. 2004, 2007). Red List categories and RLI are among the set of predictors that have been used to monitor the achievements of the 2010 target of the Convention on Biological Diversity (Butchart et al. 2010), and will be used to monitor progresses toward the achievement of the 2020 CBD Aichi targets (Pereira et al. 2013), notably Target 12 on "preventing the extinction of known threatened species and improving and sustaining their conservation status".

Red List categories, and RLI, are coarse indicators of species extinction risk, given the coarse nature of the Red List criteria themselves (also considering that the same set of criteria is applied to fungi, plants and animals). Other, more refined, indicators to measure decline at the population level do exist and the Living Planet Index is one of the most known among them (LPI; Collen et al. 2009). Population-level indicators such as the LPI have the advantage of being more sensible than RLI even to small changes in species' population abundance. Additionally, LPI does not suffer from problems of extinction risk categorization (that partly relies on a set of non-tested assumptions on expected probability of species persistence, e.g. see also criterion E of the IUCN Red List; IUCN 2001). Yet a key limitation affects LPI and other population-based indicators of extinction risk trend, which is data-requirement. In fact under no mean, LPI provides a an extensive taxonomic or spatial representation of species status, even for vertebrates. As an example, the 2009 calculation of LPI was based on data for 739 vertebrate species (Collen et al. 2009), while the 2010 Red List classification system was based on data for 25,780 vertebrate species including more than 85% of all existing mammals, birds and amphibians (Hoffmann et al. 2010).

It is not among the scope of this thesis to discuss the relative appropriateness of many (and generaly correlated) existing systems and indices for monitoring biodiversity status and trends (e.g. see Brooks et al. 2006; Butchart et al. 2010; Pereira et al. 2013). Our rationale for using the Red List classification system is related to the fact that the Red List is, by far, the most complete and widely used conservation tool to measure (and monitor) extinction risk and extinction risk change over time, thus allowing a spatial, temporal and taxonomical comparison of our results with those reported in other studies.

1.6. Sapienza University, IUCN Red List and the Global Mammal Assessment program

This PhD project is part of the activities of the Global Mammal Assessment program (GMA). The GMA program is carried out at the Department of Biology and Biotechnologies, Sapienza University of Rome, a member of the IUCN Red List Partnership (*www.globalmammal.org*). Under the GMA program, the global extinction risk of mammal species is assessed and periodically updated. Additionally, information on species' distribution, conservation actions, ecology and occurring threats are maintained according to the standards of the IUCN Red List. The GMA builds on a network of mammal experts, coordinated under the IUCN Species Survival Commission and organised in 35 Specialist Groups (each responsible for one or more taxonomic groups).

Building on the most up to date information on mammal species distribution and status, the GMA research team performs multi-scale analyses aimed at describing macroecological patterns in mammal species diversity and distribution, as well as supporting conservation decision-making through global scale prioritization exercises, projection of future species distribution and threats, historical investigations of the decline process.

This PhD research has contributed to the scope of GMA by exploring connections between applied branches of Ecology, such as extinction risk analysis, and quantitative conservation applications, such as SCP. Biological traits characterising mammal species, distribution of threatening processes, distribution of species' suitable habitat, historical records of species decline and extinction are here collected, modelled and combined for multiple conservation-related purposes.

1.7. Scope and objectives of the presented research

1.7.1. Scope and Justification

A poor integration of macroecological studies and conservation studies has characterised past strategies for global species conservation. Threshold-based approaches at setting conservation priorities, such as the "biodiversity hotspots" (Myers et al., 2000), have proven to be successful in raising conservation awareness and funding (Myers, 2003). Yet the limited availability of resources calls for a need to better rationalise conservation interventions at a global scale (Wilson et al., 2006). Mammal species have often served as a model group for analysing patterns of extinction risk (Cardillo et al., 2005), for promoting large scale conservation plans (Carwardine et al., 2008) and, overall, for raising public awareness toward conservation issues, e.g. throughout campaign for the protection of "flagship" species (Smith et al., 2012). Today, a better integration of available knowledge has the potential to value-add global conservation strategies, especially throughout multidisciplinary approaches.

This PhD thesis arises from the need to better explore the existing knowledge on mammal species biology and conservation-related issues. A variety of biological, spatial and historical information is available for mammal species globally in a number of scattered forms that have not yet been adequately integrated and synthesized. Additionally several quantitative techniques have been explored to support the science of conservation biology, especially in the past 15 years. The scope of this PhD thesis is to advance mammal conservation both in its theoretical aspects and in its practical application, by integrating existing data and by combining multi-disciplinary methodologies to provide innovative approaches to conservation decision.

1.7.2. Objectives

A detailed description of the aims and scope of each presented research projects is provided in each research chapters (Chapters II to V, "Introduction" sections). Following the same order of the presented research Chapters, the key objectives of the presented thesis are:

Objective 1. To characterise the global distribution of suitable habitat for mammals, and to identify high resolution global patterns of species richness, ecology and biogeography;

Objective 2. To describe patterns and processes characterising the recent decline of mammal species with particular reference to data-rich mammal groups;

Objective 3. To provide insights into the characterisation of internal sources of mammal species extinction risk and to use them to propose a global conservation prioritization scheme;

Objective 4. To address the methodological challenges that affect the quantification of threats to mammal species and to provide simple guidlines for conservationists.

CHAPTER II

Global habitat suitability models of terrestrial mammals

2. Global habitat suitability models of terrestrial mammals[1]

2. 1. Abstract

Detailed large-scale information on mammal distribution has often been lacking, hindering conservation efforts. We used the information from the 2009 IUCN Red List of Threatened Species as a baseline for developing habitat suitability models for 5027 out of 5330 known terrestrial mammal species, based on their habitat relationships. We focused on the following environmental variables: land cover, elevation, and hydrological features. Models were developed at 300 m resolution and limited to within species' known geographic ranges. A subset of the models was validated using points of known species occurrence. We conducted a global, fine-scale analysis of patterns of species richness. The richness of mammal species estimated by the overlap of their suitable habitat is on average one-third less than that estimated by the overlap of their geographic ranges. The highest absolute difference is found in tropical and subtropical regions in South America, Africa and Southeast Asia that are not covered by dense forest. The proportion of suitable habitat within mammal geographic ranges correlates to the IUCN Red List category to which they have been assigned, decreasing monotonically from Least Concern to Endangered. These results demonstrate the importance of fine-resolution distribution data for the development of global conservation strategies for mammals.

2.2. Introduction

Despite mammals being among the most intensively studied taxa, detailed large-scale information on their distribution has been lacking, hindering conservation efforts. Broad maps

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[1] Based on *Rondinini, Di Marco, Chiozza, et al. (2011) Phil Trans Roy Soc Lond B*

of the extent of occurrence (i.e. the limit of the geographic range; Gaston, 1991) of terrestrial mammals have been produced in the past and have been used for elucidating general biogeographic patterns (Kreft, H. & Jetz, 2010; Lamoreux, J. F. & Lacher, 2010) and in global priority-setting analyses (Rodrigues et al. 2004; Ceballos et al. 2005). Although these latter studies have provided insightful hints on global conservation priorities for mammals, criticism to global biodiversity analyses has been raised because the resolution of the geographic ranges used is too coarse to be useful on the local scale, which is the scale of conservation action (Smith et al. 2009). The reason is that species are not homogeneously distributed across their ranges, either because some habitats are not suitable for the species or because they have no access (Rondinini et al. 2006). On the other hand, the variable quality and incomplete coverage of available point locality data of mammal distributions have meant that their utility for global analyses has been quite limited (Boitani et al. 2011).

To overcome the limitations of existing species datasets, it has been suggested that deductive habitat suitability models should be used (da Fonseca et al. 2000). These models, combining the geographic ranges and habitat preferences of species with environmental data such as vegetation and elevation, increase the resolution of the geographic range by helping to identify the unsuitable habitat within it (Rondinini et al., 2006; Gaston & Fuller 2009). Deductive models of this kind on a continental or sub-continental scale have been so far developed on a 1 km^2 resolution for Africa (Rondinini et al., 2005, 2006a,b), Southeast Asia (Catullo et al., 2008), Central America (Jenkins & Giri, 2008), and Europe (Maiorano et al. unpublished data). The use of these models for the identification of mammal conservation priorities demonstrated that the conservation of mammals requires more effort than an analysis of their geographic ranges would.

The completion of the Global Mammal Assessment by the Species Survival Commission of the International Union for the Conservation of Nature (IUCN-SSC) at the end of 2008 (Schipper et al., 2008) has provided an unprecedented opportunity to develop global distribution models for the entire mammalian fauna. The dataset, which received the input of thousands of mammal experts belonging to more than 30 Specialist Groups of the IUCN-SSC, is broadly based on the taxonomy published in Wilson & Reeder (2005), updated with species described afterwards, and contains the geographic range and information on the habitat preferences of each mammalian species, in addition to the conservation status of each species. The entire dataset is publicly available through the IUCN Red List (IUCN, 2010).

We developed high-resolution, species-specific habitat suitability models for the terrestrial mammals based on the IUCN Red List, to estimate the extent of suitable habitat for each species and provide an improved baseline for the development of global conservation strategies for mammals. We validated the models of a subset of species, for which reliable point locality data were available. We conducted a global, fine-scale analysis of patterns of species richness, and investigated the biogeographic and ecological patterns underlying the observed difference between the extent of geographic ranges and the extent of suitable habitat for mammals, to demonstrate the importance of fine-resolution mammal distribution data and identify potential conservation bias in previous global prioritisation exercises.

2.3. Material and methods

2.3.1 Model development

We used the geographic range (in ESRI shapefile format) and habitat preferences information obtained from the IUCN Red List of Threatened Species as a baseline for developing habitat suitability models for 5027 out of 5330 (94.3%) extant terrestrial (including coastal and flooded habitat) mammals. For 286 species we did not develop a habitat suitability model because the information on their habitat preferences was missing, or because they have a very small geographic range (<100 Km^2), in which case we preferred to use the entire range for the analysis. This threshold was chosen heuristically based on the mismatch (up to several kilometres) between the coastline boundaries used by the IUCN Red List for the species geographic ranges and those of the satellite maps used here to develop the habitat suitability models. For a further 17 species there was no information available on the geographic range (Supplementary Table 2.S1).

For each species, we developed a habitat suitability model at 300 m resolution (ca. 1.5 billion grid cells of size 0.09 km^2) and limited to within the species geographic range, to avoid extrapolating species presence beyond their distribution limits. We focused on the following environmental variables: type of land cover, elevation, and hydrological features. The type of land cover was mapped using Globcover ver. 2.1 (IONIA 2009), a global, 300 m resolution map containing a legend of 63 classes based on the standard UN Land Cover Classification

System (Di Gregorio & Jansen, 2000). The advantage of using an LCCS-based land cover map is that the habitat preferences assessed against its legend can be easily applied to other similar maps in the future. The elevation map was produced by resampling (averaging) to 300 m the SRTM elevation (USFS 2006) at 1 arc second resolution (approximately 90 m at the equator). The map of water bodies was produced by merging two sources: a 300 m wide buffer around the 210 class (water) of the Globcover for polygonal water bodies (lakes and large rivers), and the Vmap0 (NIMA 1997) linear permanent water map (converted to raster at 300 m resolution) for linear water bodies.

When known and recorded in the IUCN Red List, the information on the elevation range within which a species occurs is expressed as a minimum-maximum range in meters and as such used in the analysis. The rest of the information on habitat preferences, including the preferred habitat types, tolerance to human impact on natural habitat types, and close relationship with water bodies are in the form of a textual description, and were extracted in two steps. First, we assigned each species to one or more broad habitat types (forest, shrubland, grassland, bare, artificial) and intersected this information with the suitability of flooded habitat and to the level of tolerance to human-impacted (degraded or mosaic) natural habitat types, to generate an automated classification of the classes of the land cover map (Supplementary Table 2.S2). In the second step, applied only when detailed information on the habitat preference was available, if and where appropriate, we modified manually the suitability of individual land cover classes. In addition, we recorded whether the species' distribution should be restricted to within a small distance to water bodies.

We defined three levels of suitability for the land cover: high, corresponding to the primary habitat of a species, i.e. the preferred habitat where the species can persist; medium, corresponding to secondary habitat, i.e. habitat where a species can be found, but not persist in the absence of primary habitat; and unsuitable, where species are expected to be seldom or never found. All cells in the model inside the elevation range of the species retained the suitability score assigned to the land cover map, while the other cells were classified unsuitable. In addition, for species whose distribution is restricted to within a small distance to water bodies, all cells farther than 1 km from water bodies were classified unsuitable (the threshold was chosen because of the potential spatial mismatch between the map of hydrological features and the maps of the other environmental variables). Models were developed in a Mollweide equal area projection (to represent and calculate areas accurately) using the free-software GRASS GIS (GRASS Development Team 2010).

2.3.2. Model evaluation

For a subset of 263 species, point locality data were available to evaluate the habitat suitability models. These data were derived from four different datasets. (1) The occurrence data of the African Mammals Databank (Boitani et al., 2008), which were collected in 100 random localities in each of four countries (Morocco, Cameroon, Uganda, Botswana), for a total of 400 points, and consist of lists of species known to be present in a 1 km radius around the point (either by direct observation or by interview with residents and local wildlife professionals). (2) The occurrence data of the Southeast Asian Mammals Databank (Catullo et al. 2008), which were collected in 100 random localities in each of four countries (Thailand, Vietnam, Borneo, and the Philippines) and have otherwise similar characteristics as the African Mammal Databank point data. (3) Further occurrence data that were collected for the Southeast Asian Mammals Databank and consist of a set of occasional (non-random) occurrences derived from various sources (Catullo et al. 2008). (4) Further occurrence data that were extracted from the Global Biodiversity Inventory Facility (GBIF 2010). The latter two datasets contain occasional data of various provenance and age. The subset chosen for the evaluation of mammal models included data collected in the last 20 years (1989-2009) and with a nominal positional error < 1 km^2 (in the subsequent analysis the positional error for these points was degraded to 1 km^2). Only species with at least five separate occurrences (i.e. in different 1 km^2 cells) were considered for model evaluation.

To validate the habitat suitability models, we computed for each of the 263 species the proportion of 1 km^2 cells containing one or more occurrences and one or more 300 m cells with high and/or medium suitability (proportion of correctly predicted occurrences, or point prevalence). We compared this value to the proportion of cells with high or medium suitability within the range (the model prevalence across the range). If predicted suitable cells were distributed randomly with respect to species occurrences, the expectation would be that the point prevalence equals the model prevalence. When the point prevalence is higher than the model prevalence, the model is better than the geographic range at predicting occurrences, because it tends to correctly flag as unsuitable the cells that do not contain occurrences.

2.3.3. Biogeographic, ecological, and threat patterns

We used the habitat suitability models to map the global spatial pattern of terrestrial mammal species richness. In each cell we summed all the species for which the cell was inside the geographic range and had medium or high suitability according to the model. We compared this value to the richness of terrestrial mammals calculated by overlaying the geographic ranges.

Species assigned to only one broad habitat type (forest, shrubland, grassland, bare, flooded) were considered habitat specialists, while the remaining species were considered habitat generalists. This classification, and the list of biogeographic realms where the species are found (extracted from the IUCN Red List), were used to analyse biogeographic and ecological patterns of range size and model prevalence. In addition, we analysed how the proportion of suitable habitat inside geographic ranges varied with the conservation status of species (IUCN Red List category). We performed all the analyses with the free-software POSTGRESQL database (PostgreSQL Development Core Team 2010) and R environment for statistical computing (R Development Core Team 2010).

2.4. Results

2.4.1. Model evaluation

A mean ± SD of 33.8 ± 43.2 occurrences for each species were available to evaluate the models. On average, models predicted as suitable 54.8% ± 21.5% of a species' geographic range, and predicted correctly 77.1% ± 16.8% of the species occurrences. Models predicted species occurrences better than the geographic ranges for 241 out of 263 species (91.6%) (Fig. 2.1), with no significant correlation between number of occurrences and model performance (Spearman rank correlation, $p = 0.19$). Point occurrences were available mainly for Afrotropical (n = 83), Indomalayan (n = 75), Nearctic (n = 25), and Palaearctic (n = 19) species.

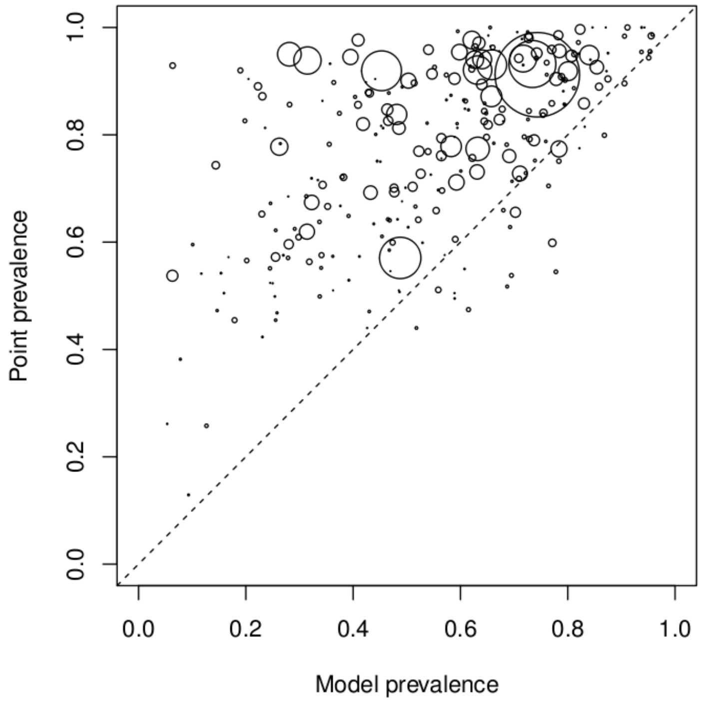


Fig 2.1. Validation of habitat suitability models for the 263 mammals with occurrence data. Each circle represents a species, and the size of the circle is proportional to the square root of the number of available occurrences for the species. Model prevalence: proportion of habitat with medium or high suitability within the species' range. Point prevalence: proportion of points correctly predicted (i.e. falling in cells containing habitat with medium and/or high suitability for the species).

The other realms were marginally or not represented (Neotropical, n = 3; Australasian and Oceanian, n = 0). The remaining point occurrences corresponded to species occurring in more than one realm. Species with point occurrences were mainly habitat generalists (n = 192, 73.0%) or forest specialists (n = 68, 25.9%). The proportion of models that predicted occurrences better than ranges was non-significantly higher in forest specialists (n = 65, 95.6%) than in habitat generalists (n = 173, 90.1%).

2.4.2. Biogeographic patterns of suitable habitat

The mean terrestrial mammal richness (300 m resolution) was 56.67 ± 41.75 species based on the overlap of the whole geographic ranges (Fig. 2.2a), and 37.70 ± 33.24 species based on the overlap of the suitable areas inside species geographic ranges (Fig. 2.2b). The peak of mammal richness (based on the overlap of suitable areas) was found in the Amazon basin (191 species estimated per cell), while other tropical regions in Africa and Southeast Asia appeared less rich in species. In comparison, the richness estimated by geographic ranges smoothed differences within and among tropical regions (Fig. 2.2).

The mean difference per cell between the two estimators of species richness was 18.78 ± 20.78, after excluding water basins from the calculation to avoid an overestimation (due to the fact that the models classify water basins as unsuitable for most terrestrial species) (Fig. 2.3a). Regions with the highest difference among the two estimators were found north and south of the Amazon basin, particularly in the Brazilian Cerrado and Atlantic Forest; in Central and Eastern Africa; and in Indochina.

The map of proportional difference among the two estimators of species richness (difference divided by richness estimated by geographic ranges) showed that in South China, Madagascar, and part of the Brazilian Cerrado habitat suitability models estimate that species richness is one-fourth or less than that predicted by geographic ranges (Fig. 2.3b). For many other regions of the world, including the tropical Andes, the rest of the Cerrado, the African savannas, most of Central, South and Southeast Asia, part of the coastal areas in Australia, and scattered areas in Europe and North America, mammal richness predicted by habitat suitability models was approximately half of that predicted by geographic ranges. The difference was relatively smaller in regions that are highly species-rich (tropical forests in the Amazon, Africa, and Borneo) or species-poor (cold and hot deserts).

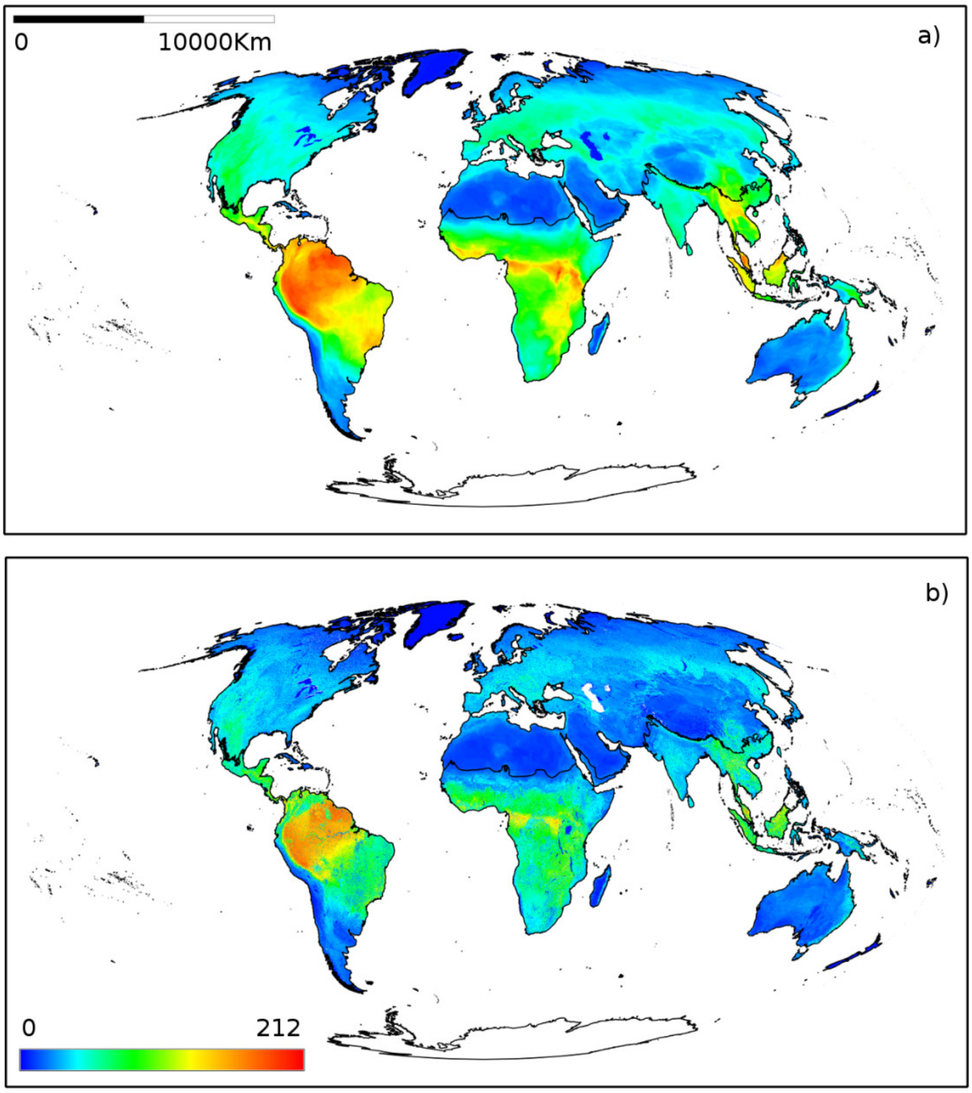


Fig 2.2. Mammal richness based on geographic ranges (a) and habitat suitability models (high and medium suitability combined) (b). Black lines on maps indicate biogeographic realms. The two upper limits on the scale bar indicate maximum richness based on habitat suitability models and geographic ranges respectively.

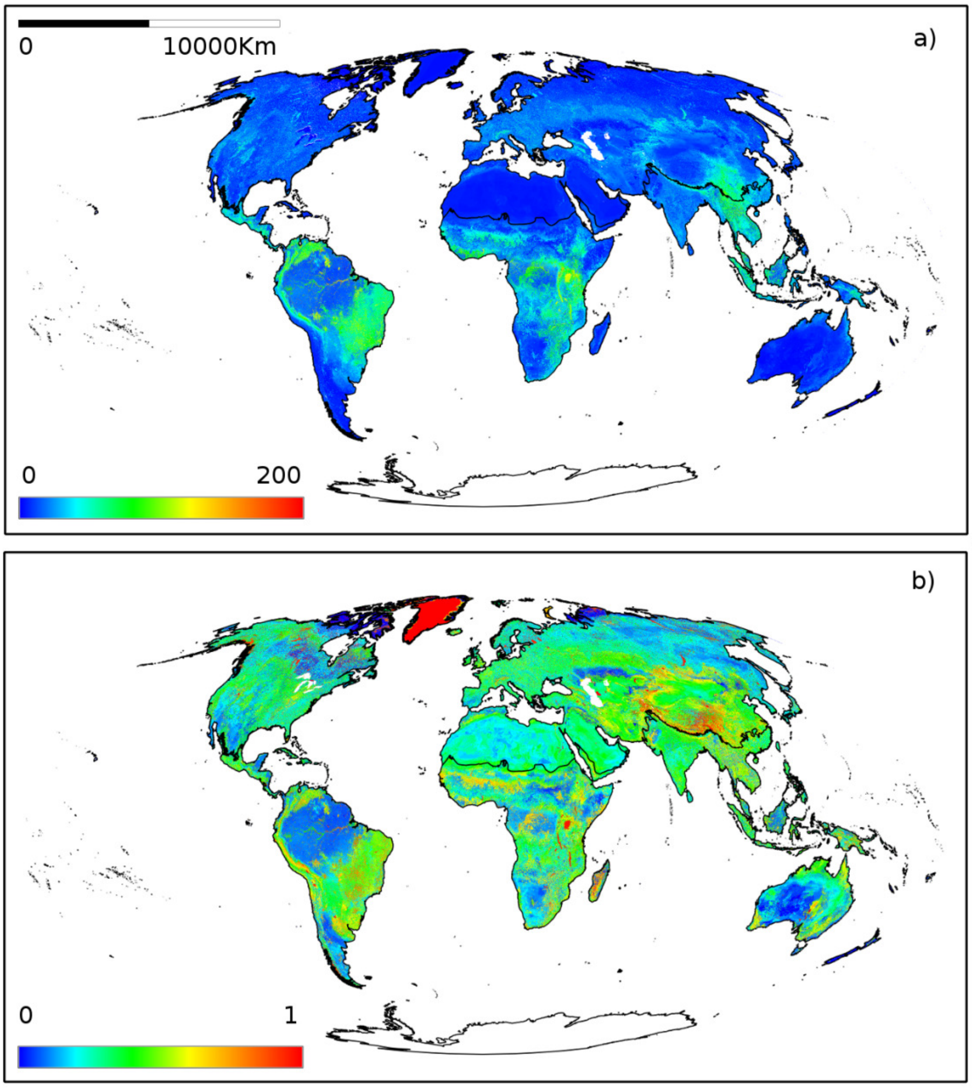


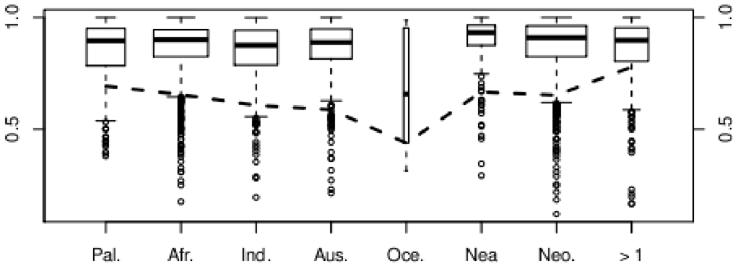
Fig 2.3. Absolute (a) and relative (b) difference of mammal richness estimated by geographic ranges and habitat suitability models. Absolute difference is represented as number of species. Relative difference is equal to absolute difference divided by the species richness estimated by geographic ranges. Black lines on maps indicate biogeographic realms.

The proportion of suitable habitat inside species ranges changed depending on the biogeographic realm where they are found (Kruskal-Wallis χ^2 =7 8.9, d.f. = 7, $p < 0.0001$). Nearctic mammals (n = 293) had the highest proportion of suitable habitat within their geographic range (0.65 ± 0.24), and Oceanian mammals (n = 11) had the lowest (0.39 ± 0.41). Indomalayan mammals had the second lowest ratio with 0.51 ± 0.27 (Fig. 2.4a).

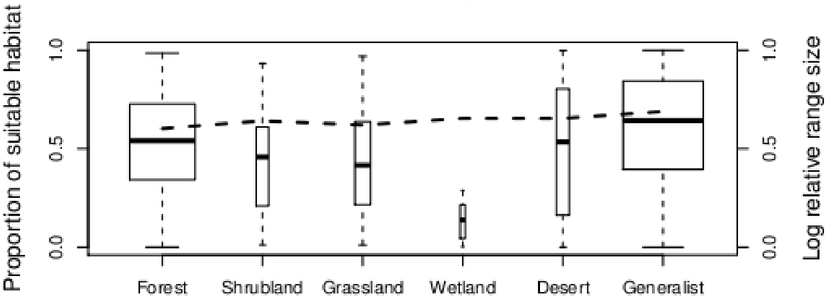
2.4.3. Ecological and threat patterns of suitable habitat

The proportion of suitable habitat inside mammal ranges varied depending on species habitat (Kruskal-Wallis $\chi^2 = 180.2$, d.f. = 5, $p < 0.0001$), from 59.8% ± 27.6% for habitat generalists (n = 2910) to 0.13 ± 0.10 for flooded habitat specialists (n = 12). Grassland specialists (n = 112) had the second lowest proportion of suitable habitat within the range (0.42 ± 0.26) (Fig. 2.4b).

(a) Realm



(b) Habitat type



(c) Red List category

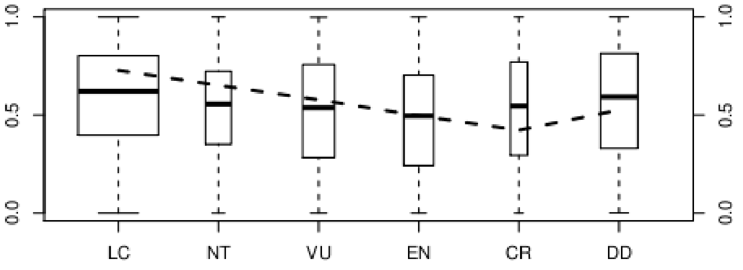


Fig 2.4. Tuckey boxplots of the proportion of suitable habitat in mammal geographic ranges by (a) biogeographic realms (with '>1' indicating taxa with a multi-realm distribution), (b) preferred habitat type, and (c) IUCN Red List category. The dotted line represents the log size of species geographic ranges.

The proportion of suitable habitat inside mammal ranges varied with the conservation status as assessed by the IUCN Red List (Kruskal-Wallis $\chi^2 = 81.7$, d.f. = 6, $p < 0.0001$), decreasing from 0.59 ± 0.26 for Least Concern species (n = 3072) to 0.48 ± 0.26 for Endangered species (n = 429) (Fig. 2.4c). The only exception were Critically Endangered species with a proportion of suitable habitat within the range of 0.52 ± 0.30 (n = 142). On the other hand, the decreasing trend of geographic range size was monotonic from Least Concern to Critically Endangered species (Fig. 2.4c).

2.5. Discussion

Habitat suitability models greatly enhanced the resolution of species distribution maps, which in previous global studies on mammals ranged between 10,000 and 20,000 km^2 (Ceballos et al., 2005; Schipper et al., 2008). This will benefit future global prioritization and conservation planning analyses. While our current resolution of 0.09 km^2 is limited by the resolution of maps of land cover and by the amount of computational effort required for map processing, these are likely to improve in the future due to the increasing availability of high-precision satellite remote sensors and fast computational resources.

The habitat suitability models that we developed have a number of limitations. They may be affected by the variable, and potentially biased (Tyre et al., 2001; Clevenger et al., 2002; Gu & Swihart, 2004; Vaugan & Ormerod, 2005), knowledge of the species–habitat relationships, and of the geographic ranges that have been used to limit the extent of suitable habitat. It is very likely that the ranges of different species have been drawn at different resolution. For example, the ermine *Mustela erminea* is the only species whose range extends into interior Greenland: while this type of inaccuracy was corrected by our model (which indicated interior Greenland as unsuitable) other inaccuracies at the margins of species ranges may have been overlooked if the habitat outside ranges is suitable. Our habitat suitability models are based on a standard and restricted set of ecological variables equal for all mammals. They are therefore unlikely to take into account all the most relevant variables affecting species distributions, and likely to still overestimate real distributions.

The availability of maps of environmental variables and the resolution at which environmental variables are mapped are limiting factors for the predictive power of habitat suitability models (Austin, 2002; Huston, 2002). If limiting variables are not mapped, or some variables are not mapped at the resolution at which species use their habitat, false presences and false absences (respectively) can be introduced in the models. These omission errors may be biased to particular types of land cover (e.g. fragmented habitat might be omitted as good habitat if existing maps are too coarse to identify small fragments), and can in turn result in geographic biases in the models (Rondinini et al., 2006; Jetz et al., 2008; Rondinini, C. Boitani 2012). When maps at higher resolution will be available in the future, the higher level of detail will reveal small patches that are not resolved at the current analysis resolution of 300 m. Higher-resolution maps of environmental variables, when available, will continue to refine the global picture of the distribution of suitable habitat for mammals. In addition, with increasing resolution of baseline maps, better consideration of the species-specific response to fragmentation (Crooks et al., 2011) will be necessary.

Habitat suitability models are influenced by subjective methodological decisions, including on the baseline maps chosen to represent the ecological variables. Different maps (e.g. GLC2000, JRC 2000, or the anthropogenic biomes, Ellis & Ramankutty 2008, for land cover) may produce different outcomes in terms of extent of suitable habitat. Here the choice of Globcover was driven primarily by its resolution, higher than in the other available land cover maps.

Even with the acknowledged limitations, our models as a whole improve the knowledge on species distribution provided by geographic ranges. Point occurrences were available to evaluate the habitat suitability models for samples of species in two boreal and two austral biogeographic realms. Overall, habitat suitability models predicted known species occurrences better than geographic ranges, even if on average the suitable habitat according to our models occupied half of the range. The result of the validation was probably negatively affected by the resolution of point occurrences, which is lower than that of the models (1 km^2 vs. 0.09 km^2). This can potentially obscure some fine-grained patterns of distribution of the suitable habitat. Also, most species for which point occurrences were available are habitat generalists. These tend to have a higher proportion of suitable habitat inside the range than habitat specialists, and the models for these species can be less sensitive in detecting habitat suitability than for habitat specialists (Crooks et al. 2006). If point occurrences for more habitat specialists were available, we would expect a further improvement of the model validation results.

The richness of mammal species estimated by the overlap of their suitable habitat is on average one-third less than that estimated by the overlap of their geographic ranges. Similar results are reported at the regional or continental level (Rondinini et al. 2005; Catullo et al., 2008; Jenkins &, Giri 2008). This has at least two general consequences. First, conservation assessments based on geographic ranges overestimate the effectiveness of current protected areas, and underestimate the amount of conservation effort that is needed to fill their gaps. Such commission errors, which come with high conservation costs, occur when species may be listed as present in protected areas that overlap their mapped extent of occurrence but where they actually do not occur (Rodrigues et al., 2004). The consequences of this error can be substantial even on a large scale (e.g. the area to be protected to conserve mammals and amphibians across Africa is underestimated by 50%, Rondinini et al. 2005). Second, they overestimate the number of different spatial options available for species conservation, because species distributions are more restricted and fragmented than geographic ranges represent.

The difference in mammal richness estimated through models or ranges is not even across the globe. The highest absolute difference is found in tropical and subtropical regions in South America, Africa and Southeast Asia that are not covered by dense forest. These regions would be expected to have high species richness based on the overlap of geographic ranges, but have been extensively converted to non-natural land cover. The relative difference in species richness is consistently high throughout the globe, with the exceptions of tropical forests and cold and hot deserts. This in turn means that the overestimation of the effectiveness of conservation areas, and of the number of spatial options for conservation, is higher in regions with higher level of habitat conversion than in relatively more intact regions. This conclusion is reinforced by the evidence that the regions with the highest difference between richness estimated by models and by geographic ranges are also those where higher loss of mammal habitat is expected in the future (Visconti et al. 2011).

The proportion of suitable habitat inside mammal geographic ranges is variable, and this variation is reflected in differences among biogeographic realms and habitat types. Oceanian (and to a lesser extent, Indomalayan) mammals, whose ranges are smaller than those of the other mammals, have also disproportionately less suitable habitat within ranges, while Nearctic species have a much higher proportion of suitable habitat within range. This disproportion may cause conservation assessments to overestimate the distribution and underestimate the extinction risk of Oceanian and Indomalayan mammals as compared with mammals in other biogeographic realms.

The proportion of suitable habitat within mammal geographic ranges correlates to the IUCN Red List category to which they have been assigned, decreasing monotonically from Least Concern to Endangered. This is the case even if, as expected, the geographic range of mammals also decreases with increasing category of threat. Therefore, more threatened mammals have disproportionately less suitable habitat inside their range than less threatened mammals. A similar result was found by Jetz et al. (Jetz et al., 2008) when comparing species range maps and well-studied survey locations of birds in Australia, North America and southern Africa. The only exception is with Critically Endangered mammals, whose small geographic ranges appear on average more suitable than those of the other mammals. This may be an effect of two factors: the geographic ranges of Critically Endangered species being very small, it is possible that the resolution of maps of environmental variables is not sufficient to identify the suitable habitat inside them; in addition, the geographic ranges of Critically Endangered species are likely to have received higher attention than those of the other species, and to have been drawn at a higher level of detail, including a more accurate exclusion of unsuitable habitat.

2.5.1. Conservation implications

Habitat suitability models are far from being a perfect representation of species distributions (Rondinini et al., 2006). Yet, the availability of point occurrence data is currently very limited, and this is unlikely to change substantially any soon (Boitani et al., 2011). Geographic ranges based solely on expert knowledge, which involves implicit interpolations and, to some degree, extrapolations, are highly subjective and their level of detail is variable according to the geographic region where a species lives, the size of its range, and its main habitat. Habitat suitability models improve on geographic ranges because they make inference explicit, easy to evaluate quantitatively, and more uniform across species, thereby reducing bias in mapping species distribution. Furthermore habitat suitability models allow an explicit validation, which should always be performed whenever they are used for conservation purposes.

The extent of suitable habitat estimated by habitat suitability models could be potentially used for re-assessing the conservation status of mammals under the criterion B of the IUCN Red List, but the current thresholds for criterion B are applicable either to the extent of occurrence (sub-criterion B1) or to the area of occupancy (sub-criterion B2). Therefore, further research is required to disentangle the relationship between extent of suitable habitat and the area of occupancy, to use the criterion B appropriately.

Habitat suitability models can track changes in the extent of suitable habitat over time as maps of land cover change are updated. Furthermore, they can be used to make explicit predictions on future habitat loss (Visconti et al. 2011), which can make conservation assessments proactive and more effective in preventing future threats.

CHAPTER III

Lessons from the past: a retrospective evaluation of the global decline of carnivores and ungulates

3. Lessons from the past: a retrospective evaluation of the global decline of carnivores and ungulates[2]

3.1. Abstract

Assessing temporal changes in species extinction risks is necessary for measuring conservation success or failure, and directing conservation resources towards species or regions that would benefit most. By consulting past literature sources, we assigned retrospective Red List categories to the world's carnivores and ungulates, to understand how their extinction risk changed since the 1970s. We found that 23% of species moved one or more categories closer to extinction, with Southeast Asia showing the steepest declines (where the figure is 45%). Some species recoveries were registered but, for each species undergoing an improvement in conservation status, eight deteriorated. Large-bodied species, including many iconic species, deteriorated significantly more than small-bodied ones. A declining trend in mammal conservation status was already underway 40 years ago and has now accelerated, uncovering alarming future scenarios of global species losses. Global human population, per-capita GDP and resources use have doubled since the 1970s. The trends that we describe are the ultimate result of geopolitical events, international regulations, shifting cultural values and natural resource exploitation. Studying trends in global species decline could help conservationists to recognize which conservation policies and strategies are (or are not) contributing to safeguarding biodiversity, thus informing the designation of better future strategies.

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[2] *Based on Di Marco et al. Submitted*

3.2. Introduction

Global human population has doubled since the 1970s and has now surpassed 7 billion people (www.unfpa.org), while both GDP per capita and global materials use have more than doubled since then (Krausmann et al. 2009). CO_2 emissions have also more than doubled in recent years (Boden et al. 2010), also in relation to the fast economic growth of developing countries such as Brazil, Russia, India, China and South Africa (so called "BRICS"), that have now emerged as global economic leaders (all of them being also invaluably rich in biodiversity). Changes at global geopolitical level have occurred, and events such as the collapse of the Soviet Union brought about massive changes in natural resources exploitation of the world's largest continent.

That same period was characterised by a growing public and political awareness of conservation problems. For example, in 1970 protected areas covered less than 2% of the global land area, but they have steadily expanded since then, and today cover over 12% of land (UNEP-WCMC 2012). Nonetheless, this significant improvement is not considered to be enough, to win the race against habitat conversion (Butchart et al. 2010). Concurrently, international conventions, such as the CITES (CITES 1973) and the European Bird Directive (e.g. see Donald et al. 2007), were launched to stop unsustainable wildlife trade and to protect important biodiversity areas. However, local levels of wildlife consumption have often risen to unsustainable levels due to increases in human population size and affluence (Bennett et al. 2002).

Overall, global biodiversity loss is accelerating, and global indicators reveal a growing mismatch between increasing pressures and slowing responses (Butchart et al. 2010). Today about 40% of the world’s surface has been converted to cropland and pasture (Foley et al. 2005), contributing to the loss of about 35-40% of the world's forests (FAO 2012). These pressures are reflected in global species extinction rates which now exceed natural background rates (measured in fossils records), by two to three orders of magnitude (Barnosky et al. 2011). Reducing such losses is high on global political agendas, but these are not always translated into effective action as they competes with other global objectives such as increasing economic growth. The search for optimal trade-off solutions is driving both the definition of novel and ambitious international conservation targets, such as those in the 2012-

2020 CBD Strategic Plan for Biodiversity (e.g. Target 12; CBD 2010), and the call for consolidation and upscaling of effective conservation strategies (e.g. Rondinini et al. 2011).

Global species declines are particularly well studied among mammals (Hoffmann et al. 2011; Schipper et al. 2008). Mammals include roughly 5500 existing species (IUCN 2012; Wilson & Reeder 2005) and new species are constantly discovered (Reeder et al. 2007). The conservation of mammals is both a major goal for international agencies and NGOs, and a core challenge for conservation scientists (Schipper et al. 2008; Hoffmann et al. 2010, 2011; Rondinini et al. 2011). Together with birds, mammals receive a disproportionate amount of conservation attention respective to other groups (Clark & May 2002). International conservation efforts have succeeded in recovering several mammal species from the brink of extinction, but have failed in reverting the global declining trend of mammalian biodiversity: one quarter of mammal species are now threatened with extinction (Schipper et al. 2008). While over 60% of mammal species are either bats or rodents, conservation marketing campaigns generally raise public awareness for the protection of large-bodied mammals, often belonging to carnivores or ungulates (so called "flagship” species; Smith et al. 2012). Nonetheless, 36% of the world's carnivores and ungulates are today threatened with extinction according to the IUCN Red List (IUCN 2012), a value significantly higher compared with other mammals.

The IUCN Red List is the most comprehensive resource detailing the global conservation status of species (Mace et al. 2008; Rodrigues et al. 2006). It is also a fundamental tool to raise awareness and to help direct conservation actions for species (Fitter & Fitter 1987). The global conservation status of all mammals was evaluated in the Red List, through the Global Mammal Assessment, in 2008 (Schipper et al. 2008), and a global retrospective assessment involving corrections to earlier listings (Baillie & Groombridge 1996) was subsequently done for 1996 (Hoffmann et al. 2010, 2011). From these assessments, an indicator of the changing conservation status of species, the Red List Index (RLI defined below; Butchart et al. 2007), was calculated to measure a trend in global mammal extinction risk. However, the RLI is a relatively coarse measure of extinction risk (as a consequence of the broad nature of Red List categories), and changes in the RLI may only be manifested over multi-year time-frames. This is especially true for long-lived species, such as ungulates and carnivores, for which the 3 generations time-frame applies to certain Red List criteria (specifically, A and C), which implies that their status may change only several years after a decline has started (IUCN SSC 2001).

Where data are sufficient, retrospectively assigning species Red List categories at a previous point in time (using currently available classification criteria; IUCN SSC 2001) is a valid method to infer past trends in extinction risk. Retrospective Red List assessments have been employed on amphibians (Stuart et al. 2004), mammals (Hoffmann et al. 2010, 2011) and corals (Carpenter et al. 2008), and are indeed implicit in the process for conducting reassessments as part of determining genuine changes (Butchart et al. 2004). Yet, perhaps because of the amount of data gathering required and potential levels of uncertainty, they have been limited to one or two decades in the past. This limits the ability to understand the impact of slow, long-term anthropogenic drivers of species' extinction risk as well as the ability to evaluate the effect of conservation efforts on long-lived species for which re-bounding from previous decline is typically slow, and the legacy of past declines is carried forward for many years.

While evidence exists for the long-term decline of several mammal populations in the past 4 decades (Collen et al. 2009), these have never been comprehensively evaluated at the species level. We have some understanding of changes in extinction risk over time, but we have no longer-term picture of genuine change that enables us to associate extinction risk changes with long-term drivers of change or conservation actions. Yet, this understanding has important management implications in the future.

Here, we perform a longer-term retrospective assessment of the Red List status of the world's carnivores and ungulates, extending our analysis two further decades back in time to the 1970s. We focus our analysis on carnivore and ungulate species for 4 main reasons: (i) the increased availability of historical base-line data on species conservation status; (ii) many species in these groups have long generation lengths (e.g. 10yrs for bears, 20yrs for rhinos; IUCN 2012) and thus need extended monitoring; (iii) they are significantly more threatened with extinction than other mammals (thus being more likely to be sensitive to status changes); and (iv) several iconic and well-studied species belong to these groups (that represent 10% of all mammals).

We calculate extinction risk trends for the world's carnivores and ungulates from the 1970s to 2008, and interpret the observed global trends with respect to regional and global socio-economic changes and conservation policies. We discuss the importance of long-term retrospective analyses to interpret past trends and guide future conservation policies.

3.3. Methods

3.3.1. Data sources

We included in our analyses all species of carnivores (Carnivora, n=284), and ungulates (Perissodactyla, n=16, and terrestrial Cetartiodactyla, n=244) currently described in the IUCN Red List (www.iucnredlist.org). For the purpose of our analyses, we also include elephants together with ungulates (Proboscidea, n=2). We focused our attention on IUCN-related sources of information of species threat status, in particular Red Data Books and Action Plans (full list in APPENDIX 3.S1). Contextual information was also used in a few instances, in the form of personal communications by experts of a particular species group and/or a particular area, in a way similar to the current Red Listing process. We looked for all sorts of information potentially related to species' threat status (as described in IUCN Red List criteria; IUCN SSC 2001), including data on: population size and structure, global species' trend, trends in known threats, trends in habitat availability (e.g., local deforestation rates), and geographic range. When quantitative information on a particular feature was missing (e.g. population reduction was not quantifiable), we interpreted available information (e.g. "a severe population decline through the whole species' range") to assign a range of possible Red List categories to the species (see section below).

3.3.2. Retrospective assignment of Red List categories

We started by collecting the latest version of the Red List categories available for all species in our groups (IUCN 2012). We then assigned each species a Red List category for 1996 by consulting Hoffmann et al. (Hoffmann et al. 2010), who listed all mammal species that underwent a genuine change in status after 1996 (i.e. those species not listed retained the same status between the two time periods). Since our aim was to evaluate a longer (> 12 years) temporal trend in species status change, we focused our data collection on assessing species Red List categories for the 1970s and 1980s. We assumed the mid-years of those decades (i.e. 1975 and 1985) as reference points for the assessments. As with (Hoffmann et al. 2011), we took a conservative approach in retrospectively assigning categories by assuming no change unless there was adequate evidence to the contrary. We identified all species for which a change in status has occurred between 1975 and 2008.We applied current Red List criteria (IUCN SSC 2001) to verify which species had undergone a genuine change in status and which had not (see Fig. 3.S1 for a representation of our methodology).

When available information was contrasting or unclear, and in general, when data uncertainty had the potential to result in the assignment of multiple threat categories, we adopted one "best guess" (i.e. most likely) category and one or two "alternative" (i.e. less likely yet still plausible) categories to which we assigned a probability score (Fig. 3.S1). By doing so, we could statistically deal with uncertainty present in old data sources (see section on "Definition of trend").

We had three types of information for species in our dataset: assessed, assumed, no data. "Assessed" refers to species for which available data made it possible to assign a given Red List category. "Assumed" refers to species where no evidence was available to support a genuine change in species Red List status between decades. This particularly referred to those Least Concern (LC) species that were absent from past Red Data Books (which contained information on threatened species only), but were LC in all recent assessments and were not considered to face any significant decline according to the consulted experts. "No data" refers to species for which it was not possible to make an assessment or an assumption (i.e. Data Deficient species), and which were subsequently excluded from analysis.

All the retrospective assessments followed a sequential check process (Fig. 3.S1). An initial evaluation of all available information was made by MDM and AI under the coordination of CR, and resulted in a first assignment of each individual species' threat category. All the assigned categories were then double checked for potential mistakes and inconsistencies by experts on particular species groups (LB, DM, EM and JS); as a third step, MH verified the double-checked categories. Finally, a consensus (i.e. a category or a set of possible alternative categories) was reached on all species for which contrasting opinions emerged originally (see also McBride et al. 2012).

We restricted our panel to a few experts, all of them familiar with the Red List Categories and Criteria and with a vast experience in the assessment process, to ensure that our interpretation of past information was both consistent across species, and comparable with the current Red List assessment procedure.

3.3.3. Definition of trend

We defined a global trend in species' threat status over the past 40 years by calculating a Red List Index (RLI; (Butchart et al. 2007) for each species group in each time step. The RLI is a metric that aggregates the Red List categories of a given group of species, after converting them into ranked values: 0 Least Concern (LC), 1 Near Threatened (NT), 2 Vulnerable (VU), 3 Endangered (EN), 4 Critically Endangered (CR) and 5 Critically Endangered (Possibly Extinct) (CRPe), Extinct (EX) and Extinct in the Wild (EW). RLI represents the relative overall distance of a group of species from extinction, ranging from 0 (all species are extinct) to 1 (all species are of least concern).

We calculated the global RLI trend for ungulates and carnivores and we also calculated separate trends for large- and small-bodied species of those groups (body mass data sources were the same as in Di Marco et al. 2012). We classified small-bodied species for each group as those species having a body mass equal to, or smaller than, the median group value, while species larger than the median value were considered large-bodied. The median body mass values were 3.8 kg for carnivores and 58 kg for ungulates. In order to account for the uncertainty in the assignment of retrospective Red List categories, we employed random probabilistic permutations based on the alternative categories assignment (see previous paragraph). We repeated the RLI calculation 1,000 times for 1975 and 1985, by randomly extracting a Red List category for each species with uncertain assessments (accounting for the probability associated to each alternative category associated to the species). We obtained a distribution of possible RLI values for 1975 and 1985. We used the median RLI value as a reference point and used the 2.5th and 97.5th percentiles of the distributions to represent a confidence interval around the median RLI values.

3.3.4. Spatial representation of trend

We calculated spatially explicit changes in Red List categories and RLI values by aggregating species' distributions in low-resolution squared cells (Fig. 3.S2). Since the detailed extent of past distributions is unknown for almost all species in our groups, we superimposed current species range to a grid of 250*250 km. For each species, we considered as part of the low-resolution range all cells containing a portion of current species range. By doing so, we aimed to minimize the possibility of mismatching species distributions between years. We calculated separate Red List changes and RLI values for each cell, based on the Red List categories of species occurring in the cell. We also calculated an overall RLI trend divided by countries (based on the global Red List status of species included in each country) and by biogeographic realms. Changes in this spatial RLI reflect improvements or deteriorations in the aggregate risk of extinctions occurring anywhere in the range of the species that is present in a given cell/country. This spatial RLI, therefore, is an index of the aggregate global risk of extinction of the species the cell/country hosts, and is not necessarily related to localized threats or conservation actions.

3.4. Results

We retrospectively assessed the Red List status of 261 carnivores and 237 ungulates; 48 species, historically Extinct (<1970) or having a status of Data Deficient (DD), were excluded from the analyses. We report a complete list of the assessments (including categories, criteria and justifications) in Table 3.S1.

A general deterioration in Red List status was observed through the decades (Fig. 3.S3), with a decrease in the total number of LC species and a general increase in the proportion of threatened species (VU, EN, CR). Between 1975 and 2008, 370 (74.30%) species did not show any change in status while 114 (22.89%) experienced a deterioration and 14 (2.81%) an improvement (Table 3.1).

		RL '75					
		LC	**NT**	**VU**	**EN**	**CR**	**EX+EW**
RL '08	**LC**	262	3	1	1	0	0
	NT	24	20	4	0	0	0
	VU	15	18	52	2	1	0
	EN	6	4	28	27	2	0
	CR	3	1	3	10	9	0
	EX+EW	0	0	1	0	1	0

Table 3.1 Transition matrix of the Red List categories between 1975 and 2008. Each cell reports the total number of species (carnivores + ungulates) that had an original Red List category in 1975 (column) and final category in 2008 (row). The diagonal reports the number of species that maintained their original Red List category (light grey background); the upper part reports the number of species that improved their status (white background); the bottom part reports the number of species that deteriorated their status (dark grey).

Overall, improvements in Red List categories were mainly observed in South and North America, while deteriorations were mainly registered in Southeast and Central Asia (Fig. 3.1).The total percentage of threatened species increased from 28.5% in 1975, to 29.1% in 1985, to 33.1% in 1996, to 36.3% in 2008. The general deterioration in overall status was reflected in a negative trend of the RLI both for ungulates and carnivores (Fig. 3.2). Ungulates have a worse conservation status and a steeper decline than carnivores and the conservation status of small-bodied species is generally better compared with large-species, through all decades. The probability of undergoing a status deterioration after 1975 was significantly smaller for species below 10 kg (16.5%) than for species above 100 kg (31.6%) ($p<0.05$; binomial test for proportions). In pair-wise comparisons, the proportion of declining species in a given body mass interval did not differ significantly between carnivores and ungulates (Fig. 3.S4). The aggregated RLI (carnivores and ungulates together) showed a deterioration from 0.84 in 1975 to 0.78 in 2008 (which equates to 30% of species deteriorating by one Red List category).

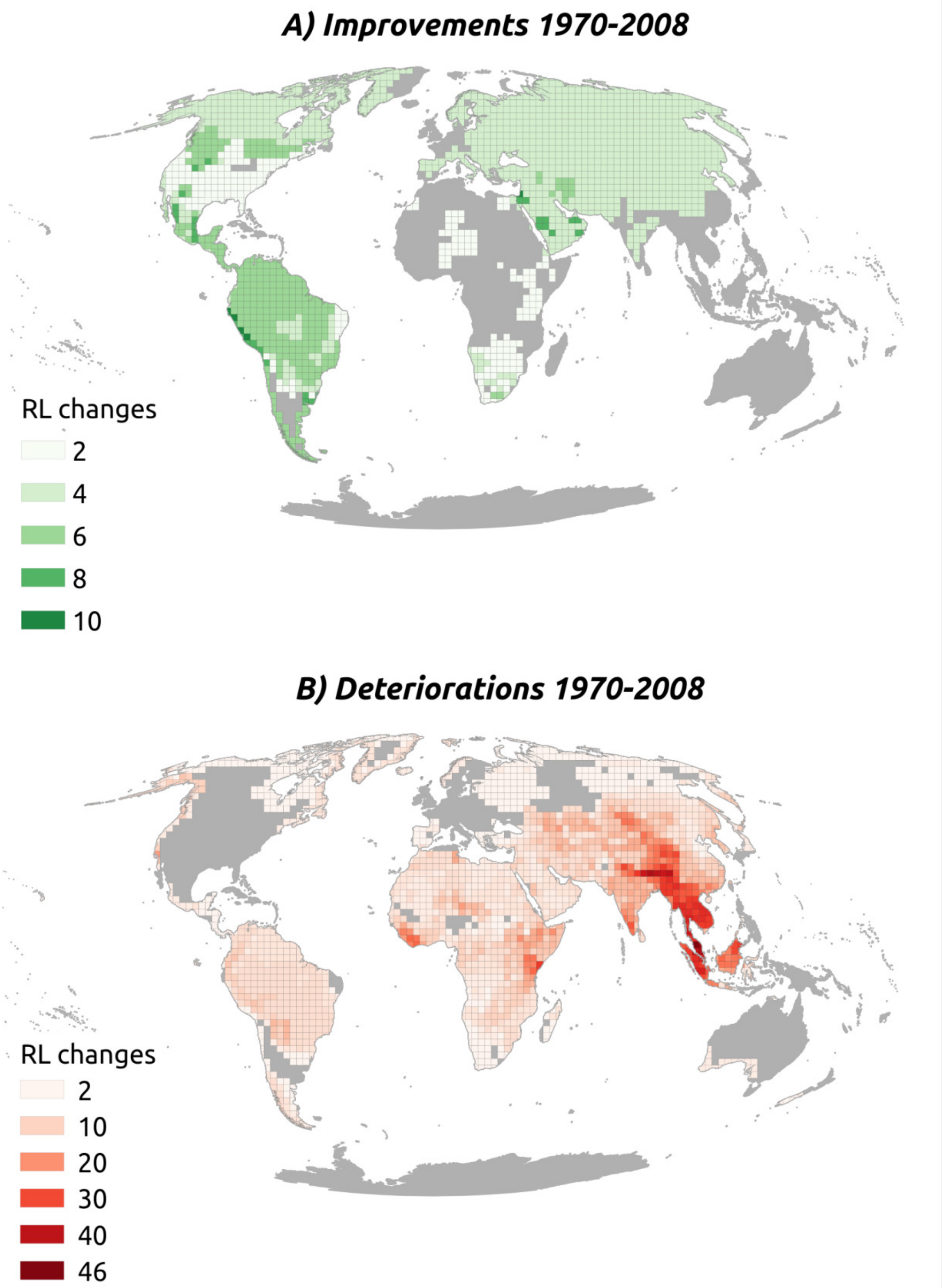

Fig. 3.1 Total sum of Red List category changes in each 250km cell. The sum of Red List category changes is reported for all species that underwent (a) an improvement or (b) a deterioration (b) in their conservation status between 1975 and 2008.

Between 1975 and 1985, a general improvement or stability in extinction risk was observed in Europe, North America and Northeast Asia, while a negative trend characterized Southeast Asia and large portions of Africa and South America (Fig. 3.S5a). Between 1985 and 1996, an inversion of the trend (from negative to stable/positive) was observed in South America and portions of Africa, while Madagascar and Australia (characterised by the presence of pinnipeds) were characterised by high levels of deterioration (Fig. 3.S5b). Between 1996 and 2008, a stable trend was observed in North America, central and northern Europe while different levels of deterioration were observed elsewhere, particularly in Southeast Asia and Australia (Fig. 3.S5c). Overall, an expansion of the deteriorating areas can be observed throughout the decades (Fig. 3.S5).

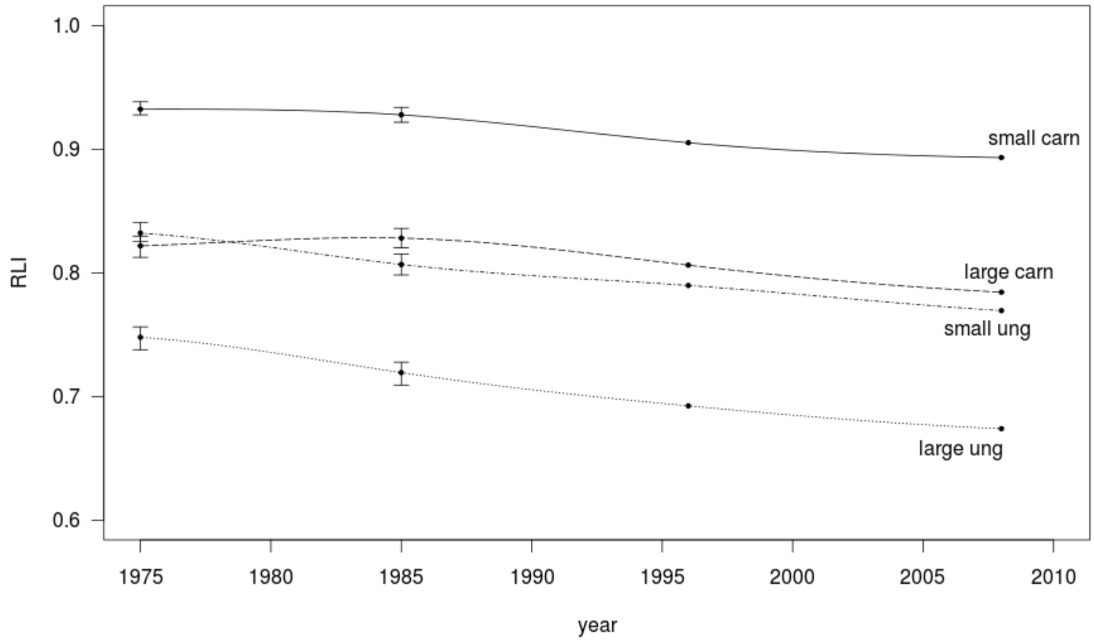

Fig. 3.2 Red List Index (RLI) calculated for carnivores and ungulates between 1975 and 2008. Species are divided by group and weight, small species are those being below the median for their group. Confidence interval bars (95%) were calculated for the retrospective RLI (1975 and 1985) to account for uncertainty in retrospective assessments (see methods for details).

When measuring RLI trend for countries, Southeast Asia and Australia showed the steepest deterioration (Fig. 3.S6). Among those countries containing at least 10 species of carnivores or ungulates, Southeast Asian countries were the most impacted by RLI deterioration (Table 3.2; Table 3.S2). Indonesia and Malaysia, both characterised by high species richness, showed a 0.1 decline in RLI values between 1975 and 2008, roughly corresponding to 50% of species moving one step closer to extinction. Overall, an average 0.08 difference was registered in all decades, between the RLI measured in Southeast Asia and that measured globally (Fig. 3.S7). Correspondingly, Indomalaya and Australasia were the realms with the lowest RLI values across the four decades (Fig. 3.S8).

Name	**RLI'75**	**RLI'85**	**RLI'96**	**RLI'08**	**deltaRLI ('08-'75)**	**N. species (endemics)**
Indonesia	0.78	0.76	0.73	0.67	-0.11	59 (10)
Malaysia	0.82	0.80	0.78	0.72	-0.10	50 (0)
Philippines	0.72	0.71	0.68	0.62	-0.09	17 (10)
Brunei Darussalam	0.88	0.87	0.85	0.79	-0.08	27 (0)
Tunisia	0.86	0.82	0.80	0.78	-0.08	19 (0)
Thailand	0.82	0.80	0.78	0.74	-0.08	58 (0)
Cambodia	0.80	0.79	0.76	0.72	-0.08	44 (0)
Mongolia	0.90	0.89	0.87	0.83	-0.08	37 (1)
Lao PDR	0.82	0.81	0.79	0.75	-0.08	49 (0)
Viet Nam	0.80	0.79	0.77	0.73	-0.07	60 (0)

Table 3.2 List of the top 10 countries that underwent a deterioration in RLI between 1975 and 2008. The total number of species and the number of endemic species are also reported. The full list of countries is reported in Tab S2. Note: only countries with at least 10 species of carnivores or ungulates are included in this table. In 2008, the Philippines showed the lowest RLI value across all countries.

3.5. Discussion

A global deterioration in the conservation status of the world's carnivores and ungulates has taken place since the 1970s and has been exacerbated recently. Hoffmann et al. (Hoffmann et al. 2010) depicted a decline in global mammal conservation status between 1996 and 2008, with a negative trend that was less marked than that observed in amphibians but more severe than birds. We show here that a declining trend in mammal conservation status was already underway 40 years ago and is now accelerating, uncovering alarming future scenarios of global species losses. For each species which showed an improvement in conservation status, eight species showed a deterioration, with a general "up-listing" of species from Least Concern to threatened. This trend also characterised iconic species that received substantial conservation attention in past years, such as the Giant Panda, *Ailuropoda melanoleuca*.

In agreement with earlier studies (Cardillo et al. 2005; Hoffmann et al. 2011), small-bodied carnivores and ungulates had a lower probability of experiencing a decline than large-bodied species. Since ungulates are on average larger than carnivores (body mass medians for the two groups are in a ratio of ≈ 1:15), the former show a higher proportion of declining species (albeit not significantly). Also, larger bodied ungulates are more attractive to predators, and declining natural prey base affects large carnivores (also preferentially persecuted as potential or actual enemies of livestock).

Geographical distribution also influenced the probability of a species undergoing a genuine deterioration in extinction risk. Measured both in terms of the RLI and the total sum of Red List category deteriorations, the Indomalayan realm (and Southeast Asia in particular) showed the worst conservation status. Enhancing conservation actions in that region has the potential to provide immediate effects for the recovery of threatened mammalian biodiversity, considering that many threatened mammals living there have a high "Extinction Risk reduction Opportunity" associated to their biological traits (Di Marco et al. 2012). On the other hand, a few improvements in species' conservation status were scattered throughout the globe and more commonly observed in South America.

A growth in human population density (with unequal rates, e.g. higher in Asian countries than elsewhere), an increase in the rate of natural resources exploitation and an expansion of intensive agriculture practices (also in mega-diverse tropical areas), have characterised the global economic growth in the recent past. Global political and economic equilibriums have changed with the growth of BRICS countries. Meanwhile, conservation policies, protected areas expansion and international treaties were implemented to address the alarmingly growing biodiversity crisis. Anthropogenic impacts on the landscape and global conservation interventions have framed the global trends of carnivores and ungulates populations in the past 40 years. The trends that we have described here are the ultimate result of geopolitical events, international regulations, shifting cultural values and natural resource exploitation. This is well demonstrated with three examples, discussed below (Fig. 3.3).

(i) South American felids, like the Jaguar *Panthera onca*, have historically suffered from a widespread decline due to global trade in their pelt. The Convention on International Trade in Endangered Species of wild fauna and flora (CITES 1973) became operational in 1975, and as a direct effect the fur trade of felid pelts was banned and South American felid species recovered substantially (Fig. 3.3a). The possibility for these species to maintain their current status will depend on the ability of international conservation bodies to deal with emerging threats, such as habitat loss, that are predicted to exacerbate under future scenarios (Visconti et al. 2011). (ii) Habitat loss, and especially forest conversion to oil plant plantations, has exacerbated in Southeast Asia in the past 30 years (Sodhi et al. 2004), while hunting and unregulated harvest of species are rampant (Corlett 2007). As a result, an acceleration in the decline of species living in that region was observed (Fig. 3.3b), a result that has been evidenced in bird species in the same region in the same time period (Butchart et al. 2005). (iii) The collapse of the Soviet Union had a noticeable effect on the conservation status of the species living in the countries of the former USSR due to simultaneous collapse of protected area systems, withdrawal of subsidies, and an abrupt transition to a free-market economy (among other factors). Many species such as the Saiga Antelope, *Saiga tatarica,* have been drastically impacted by such a change (Milner-Gulland et al. 2001). In fact several species living in that region were already declining in 1970s, yet their rate of deterioration increased dramatically in the 1990s (with a loss of RLI values that increased from 0.05 in the period 1975-1985 to 0.15 in the period 1996-2008; Fig. 3.3c).

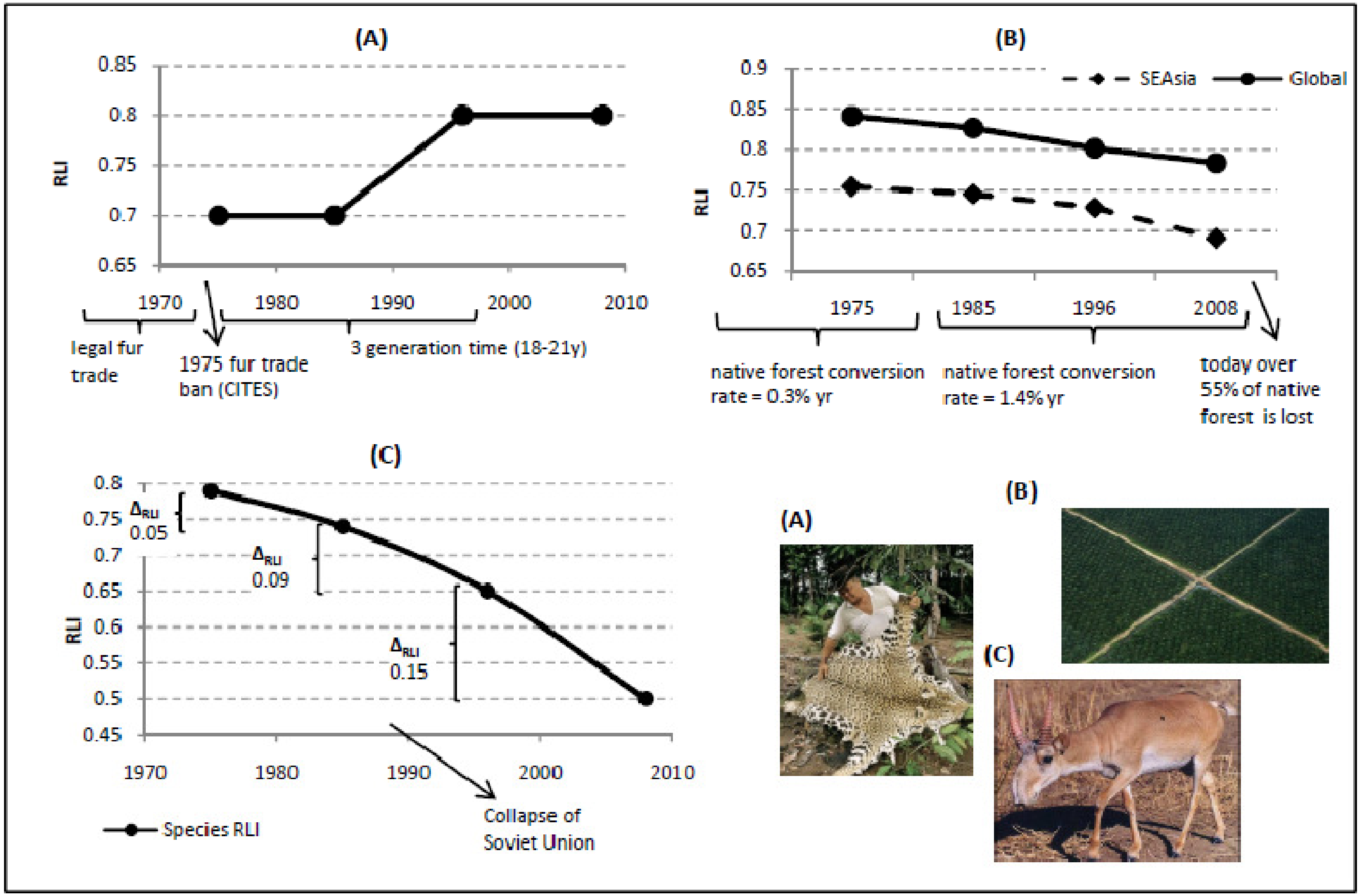
(A)
0.85
0.8
0.75
0.7
0.65
RLI
1970
1980
1990
2000
2010
legal fur trade
1975 fur trade ban (CITES)
3 generation time (18-21y)
(B)
0.9
0.85
0.8
0.75
0.7
0.65
RLI
SEAsia
Global
1975
1985
1996
2008
native forest conversion rate = 0.3% yr
native forest conversion rate = 1.4% yr
today over 55% of native forest is lost
(C)
0.8
0.75
0.7
0.65
0.6
0.55
0.5
0.45
RLI
Δ_RLI 0.05
Δ_RLI 0.09
Δ_RLI 0.15
1970
1980
1990
2000
2010
Species RLI
Collapse of Soviet Union
(A)
(B)
(C)

Fig. 3.3 Recent drivers of change in the extinction risk of carnivores and ungulates. A) Global Red List Index (RLI) trend for South American felids affected by the fur trade ban of the CITES convention. This analysis is based on the Red List status of: *Leopardus pardalis*, *L.tigrinus*, *L. wiedii*, *L.colocolo*, *L.geoffroyi* and *Panthera onca*.B) RLI trend for Southeast Asian carnivores and ungulates (n=110) in relation to forest conversion (e.g. to oil palm plantation; (Sodhi et al. 2004)) compared to global trend. (C) Accelerating rate of decline of species after the collapse of Soviet Union. The RLI has been calculated based on the Red List status of all declining species (n=20) occurring in the former Soviet Union (i.e. those up-listed between 1970 and 2008, e.g. Saiga Antelope, *Saiga tatarica*). The bottom right photographs represent: Jaguar pelt (A), oil palm plantation in Sumatra (B), Saiga Antelope (C).

3.5.1. Conclusions

Retrospective assessments provide novel insights into the impacts of longer term threat processes (as well as changing political circumstances and conservation interventions) on trends in species extinction risk, and how these affect different species groups in different regions. These assessments reduce the risk of "shifting baseline syndrome" (or "generational amnesia"), potentially affecting judgment on the desirable conservation status of species (e.g. see Turvey et al. 2010). Retrospective assessments can also shed light on our interpretation of future scenarios of emerging threats (such as agriculture intensification or climate change). We suggest that careful application of the methodology used here has tremendous potential for use in other taxa (where the data allow), both for short and longer-term estimation of trends in extinction risk. This underlines the importance of ensuring that supporting information for assessments, as required through the Red List assessment process, be properly archived over time to inform the retrospective assessment process in the future.

The present study shows that 23% of all carnivore and ungulate species moved one or more Red List categories closer to extinction over a four-decadal period, while human population density, global resources consumption and per-capita GDP have doubled. Some species are now likely extinct (e.g. the Kouprey, *Bos sauveli*), at least one species went Extinct in the Wild (the Scimitar-horned Oryx, *Oryx dammah*), and at least one was definitively lost (the Saudi Gazelle, *Gazella saudiya*). Many species in these groups are conservation flagships (Smith et al. 2012), and have continued to deteriorate despite conservation attention (though likely at a lower rate). Conservation efforts often are local in scope and/or targeted towards a limited number of species, and/or insufficient to mitigate against particular threats. Further, protected areas are not always successful in preventing habitat loss or alteration (e.g., on the island of Sumatra, recent annual deforestation rates in protected areas are as high as those in commercial logging concessions; (Gaveau et al. 2012)), and many threatened species are not sufficiently protected anyway (Fuller et al. 2010; Rodrigues et al. 2004). Notwithstanding, we also find evidence for conservation successes (e.g. for the recovery of the European bison *Bison bonasus,* Arabian Oryx *Oryx leucoryx* and various South American felids), particularly where actions are characterized by a high degree of global coordination and national-level enforcement. Indeed, efficient implementation of national and international regulations has been demonstrated to play a major role in promoting species recovery for vertebrates (Hoffmann et al. 2010) .

Studying recent trends in global species decline could help conservationists to recognize which conservation policies and strategies are (or are not) making a contribution towards improving the status of biodiversity, thus informing the designation of better future strategies. Conservation science is now facing two key challenges (Rondinini, Boitani, et al. 2011): one is preventing the extinction of species under future scenarios of threats expansion (Visconti et al. 2011), the other is minimising the extinction risk of species which are currently threatened (Di Marco et al. 2012). The success of global conservation efforts in performing these two tasks will determine our ability to preserve the world's biodiversity for future generations.

CHAPTER IV

A Novel Approach for Global Mammal Extinction Risk Reduction

4. A Novel Approach for Global Mammal Extinction Risk Reduction[3]

4.1 Abstract

With one fourth of the world's mammals threatened with extinction and limited budget to save them, adopting an efficient conservation strategy is crucial. Previous approaches to setting global conservation priorities have assumed all species to have equal conservation value, or have focused on species with high extinction risk, species that may be hard to save. Here, we identify priority species for optimizing the reduction in overall extinction risk of the world's threatened terrestrial mammals. We take a novel approach and focus on species having the greatest recovery opportunity using a new conservation benefit metric: the Extinction risk Reduction Opportunity (*ERO*). We discover that 65-87% of all threatened and potentially recoverable species are overlooked by existing prioritization approaches. We use the ERO metric to prioritize threatened species, but the potential applications are broader; ERO has the potential to integrate with every strategy that aims to maximize the likelihood of conservation success.

4.2. Introduction

Developing global conservation plans for vertebrate species has been a primary focus for conservation scientists in recent years (Brooks et al. 2006; Grenyer et al. 2006; Hoffmann et al. 2010). Mammals are often selected as a model taxon for defining spatial conservation priorities at a global scale (Schipper et al. 2008; Rondinini et al. 2011b), and recent research efforts have concentrated on defining economically and socially compatible mammal conservation strategies (Carwardine et al. 2008). Simultaneously, but independently,

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[3] *Based on Di Marco, et al. (2012) Conserv Lett*

biologists have been investigating the predictability of extinction risk from biological traits and phylogeny, mammals again often being the model taxon (Cardillo et al. 2005, 2008; Davidson et al. 2009; Fritz et al. 2009). We combine information on species' current and intrinsic extinction risk to define a new conservation metric, the *Extinction risk Reduction Opportunity* (hereafter: *ERO*; see details in *Materials and Methods*), that detects threatened species with a high biological potential for recovery. This metric builds upon the concept of "latent extinction risk", which identifies species with the greatest potential for future decline, based on the negative discrepancy between current threat status and the extinction risk predicted from biological traits (Cardillo et al. 2006). The ERO approach, on the other hand, uses current and intrinsic threat status to identify threatened species with the greatest potential for recovery from an imminent risk of extinction. It focuses on species that are likely to be easiest to save, thereby maximizing the cost efficiency of conservation projects.

Our analyses focus on threatened terrestrial mammals, representing one fourth of all non-extinct, data-sufficient terrestrial mammal species. We use information reported in the Red List of Threatened Species from the International Union for the Conservation of Nature (IUCN) as a source for species current risk of extinction. We model species' intrinsic extinction risk following an established approach by Cardillo et al. (2008; see details in APPENDIX 4.S1). We use a recently released database of mammals' life-history traits (PanTHERIA; Jones et al. 2009) as a source for our extinction risk models; we use Multiple Imputation (MI, Rubin 1987; see also Fisher et al. 2003) to impute the missing values in the database's fields (see details in APPENDIX 4.S1). With the use of MI in our extinction risk models, we avoid many of the problems related to the presence of missing data encountered in previous studies (Cardillo et al. 2005, 2006, 2008) so that our models are likely to be more stable and robust. We use a recently updated source of mammals' phylogeny (Bininda-Emonds et al. 2007) to correct for phylogenetical non-independence in our models.

We test the performance of our approach in terms of priority species definition and compare it to existing and previously proposed mammal conservation strategies. We evaluate how Critically Endangered (CR) species in the Red List (those having an extremely high risk of extinction; IUCN 2001; IUCN 2010), Alliance for Zero Extinction (AZE) species (those confined within "centers of imminent extinction"; Ricketts et al. 2005; AZE 2010) and Evolutionarily Distinct and Globally Endangered (EDGE) species (Collen et al. 2011) perform in terms of ERO value. We then use habitat suitability models from the Global

Mammal Assessment program (GMA; Rondinini et al. 2011a) to run a global spatial prioritization analysis to define the top 5% of areas for the conservation of the top-ranked ERO species. We compare our results to those found with an analysis oriented toward the detection of top priority areas for an equal number of CR species. We show that existing conservation strategies for mammals are not efficiently addressing species' extinction risk reduction. We finally define taxonomic and spatial priorities for minimizing the risk of extinction in threatened terrestrial mammal species.

4.3. Methods

4.3.1. Data sources

Our analysis was focused on threatened species according to IUCN Red List (IUCN 2010), which represent 21.2% of terrestrial non-extinct mammals. As a source for mammal phylogeny, we used a recently updated version (Fritz et al. 2009) of the Bininda-Emonds et al. (2007) 's supertree. We excluded 163 species from our analysis (14% of the total threatened mammals) due to a lack of phylogenetic information in the updated supertree. We analyzed the freely available PanTHERIA database (Jones et al. 2009), recently used to compare global pattern of functional and phylogenetic diversity (Safi et al 2011), to derive the species' biological traits that potentially correlated with extinction risk. We compiled the missing data fields of PanTHERIA using a Multiple Imputation procedure (Rubin 1987) applicable to phylogenetically structured datasets (Fisher et al. 2003; Fisher & Blomberg 2011; see details in APPENDIX 4.S1). We used species' current and (statistically) predicted extinction risk as a currency for assigning a conservation value to each species defining a new conservation metric. The ERO metric accounts for i) the total possible reduction in species extinction risk, that depends on species' current threat status and ii) the opportunity of having such a reduction, that is related to species' intrinsic risk of extinction. The ERO metric is defined as:

$$\mathrm{ERO} = (\mathrm{ER}_{\mathrm{IUCN}} \cdot (5-\mathrm{ER}_{\mathrm{INTR}}))/5, \qquad [1]$$

where $\mathrm{ER}_{\mathrm{IUCN}}$ is the current extinction risk based on the species' IUCN category (IUCN 2010) and $\mathrm{ER}_{\mathrm{INTR}}$ is the species' intrinsic extinction risk (see example in Fig. 4.1).

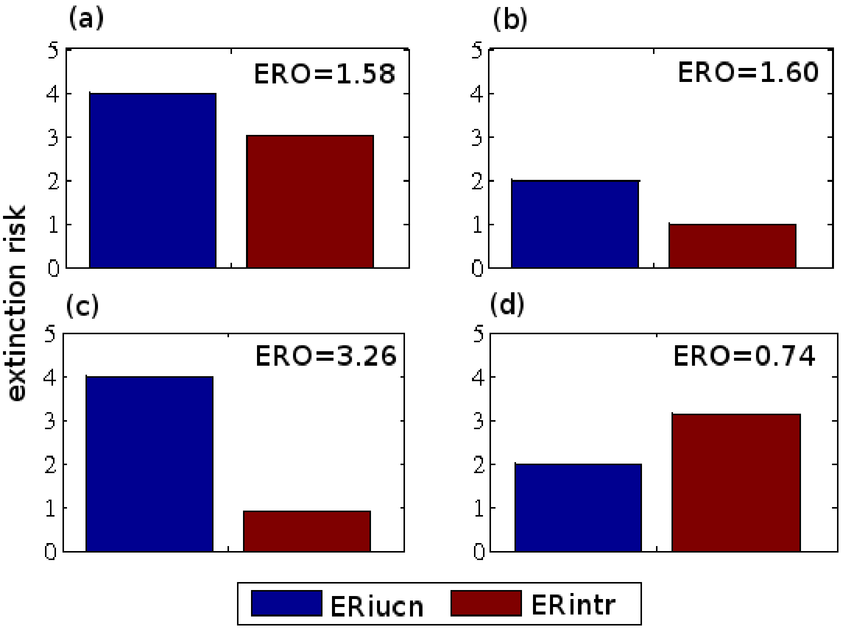

Fig. 4.1 The ERO (Extinction risk Reduction Opportunity) components. The graph shows the current (ER_{IUCN}) and intrinsic (ER_{INTR}) extinction risk of four example species and their associated ERO values (derived from *[1]* in *Materials and Methods*; see Table 4.S2 for a complete species' list). *a)* The Aders' Duiker (*Cephalophus adersi*) has both a high ER_{IUCN} and a high ER_{INTR} values, resulting in a medium ERO value. *b)* The Clouded Leopard (*Neofelis nebulosa*), has a both a low ER_{IUCN} and a low ER_{INTR} values, resulting in a medium ERO value. *c)* The Black-spotted Cuscus (*Spilocuscus rufoniger*) has a high ER_{IUCN} value and a low ER_{INTR} value, resulting a high ERO value. *d)* The Andaman Spiny Shrew (*Crocidura hispida*) has a low ER_{IUCN} value and a high ER_{INTR} value, resulting a low ERO value.

For each species, ER_{IUCN} was defined by converting the species' IUCN category into a ranked index from 0 to 5 (Purvis et al. 2000; Cardillo et al. 2005), while ER_{INTR} was the fitted value from the extinction risk models (in a 0-5 scale for consistency; see details in APPENDIX 4.S1). In defining the ERO metric, we assumed that the higher the current extinction risk for a species, the faster we should act to preserve that species, given that the higher the intrinsic extinction risk for a species the smaller is our probability of reducing its current extinction risk. Habitat suitability models from GMA (Rondinini et al. 2011a) were finally used as a measure of habitat quality for the species included in the prioritization analysis (see below). The total suitable habitat (in km^2) within each Planning Unit (PU) in the analysis represented the PU absolute value for the species.

4.3.2. Prioritization analysis

We considered a global grid of square PUs with a 10 km resolution and used Zonation (Moilanen et al. 2005; Moilanen 2007) for ranking the PUs according to their species content. We have excluded Antarctica from our analysis as only marine mammals live there. For each species, the geographic range and the amount of suitable habitat (from GMA models) within each relevant PU were considered. The Zonation algorithm produces '*a hierarchical prioritization of the conservation value of the landscape*'. It assigns each cell to a landscape fraction based on its priority level; the top 10% of selected cells are part of the top 20%, the top 20% of selected cells are part of the top 30% and so on (Moilanen & Kujala 2008). We assigned a representation target for each species based on their distribution range dimension, following Rodrigues et al. (2004). The target was 10% of the distribution range for species with a range dimension bigger than 250,000 km^2 and 100% of the distribution range for species with a range dimension smaller than 1,000 km^2; for all intermediate-range species, we adopted a log-interpolated value between 10% and 100%. We used the ERO values together with the species representation targets to formulate a generalized benefit function (Moilanen & Kujala 2008) and then used it to run a spatial prioritization analysis in order to find the top 5% terrestrial area for preserving the top-ranked species according to ERO (see APPENDIX 4.S2 for details on benefit function formulation). We then repeated the analysis on Red List's CR species (IUCN 2001; 2010) assuming an equal initial conservation value among them, in order to verify the spatial difference occurring with the use of the ERO metric versus the classical Red List categories approach in conservation priorities setting. Both analyses were run on the same number of species, calculated as the number of CR species for which an ERO value was available ($n=139$, $n \approx 75\%$ of terrestrial CR mammal species; IUCN 2010). We overlapped the final priority area maps to species' fine-scale (300m) distribution maps (Rondinini et al. 2011a), in order to check the representation level of the taxa within the selected cells and the performance of the selection algorithm at the analysis' resolution level (10km). We then investigated the current protection status of the resulting priority areas in relation to the existing protected areas system. We selected protected areas in IUCN categories I-IV from the World Database of Protected Areas (WDPA 2010). All missing-shape sites were included as a buffer area centred in the WDPA point coordinates, having the same area as declared in the database. We also calculated the level of spatial overlap between our priority areas and the Earth's biodiversity hotspots (Mittermeier et al. 2005).

4.4. Results

Through the imputation of missing data we statistically approximated the completion of PanTHERIA database (Jones et al. 2009); the rate of missing information for parameter estimation (i.e. the variation in results across the imputed data set that reflects the statistical uncertainty due to missing data; Rubin 1987) appeared to be small (i.e. equal or smaller than 1%) in all orders, with only a few exceptions for some of the intercept parameters (see Table 4.S1 for extinction risk models and Table 4.S2 for a complete list of species and associated ERO values). Species prioritized using EDGE, AZE and CR measures had a significantly lower ERO value (i.e. a lower potential conservation relevance) with respect to a corresponding number of top-rank ERO species (Table 4.1). Even if EDGE metric was not a good predictor of the potential ERO value for threatened mammals, it performed better than a random choice of species in terms of median ERO. On the other hand, the AZE's species selection procedure resulted in an underrepresentation of the potential ERO value for mammals when compared to randomly selected species sets (Table 4.1). Unsurprisingly, the use of ERO instead of latent extinction risk (Cardillo et al. 2006) gave very different results in term of threatened species ranking (we considered the absolute difference among the percentage ranks of each species according to the two metrics); within threatened species, there was an average difference of 49.57% (se = 0.92%) in the ranking of species according to the two metrics. The average change in rank for a species when using ERO metric instead of Red List categories was 24.70% (se=0.71%).

	IUCN CR	**AZE**	**EDGE**	**IUCN threatened**
species pool median (95% range)	1.79 (0.77 - 3.10)	1.63 (0.53-2.62)	1.91 (1.02 - 2.88)	1.60 (0.93 - 2.68)
top ERO ranked median (95% range)	2.35 (2.09 - 3.11)	2.41 (2.12 - 3.11)	2.57 (2.32 - 3.15)	†
WRS test	p <<0.01	p <<0.01	p <<0.01	†
Random test	*	0%	97.88%	†
Species sets difference	65.24%	87.4%	75%	†
n‡	**139**	**126**	**80**	**964**

Table 4.1. Comparison of the ERO values of priority species detected according to existing conservation schemes. Comparison of ERO values among top ERO ranked species and species detected using other

conservation metrics (species pool), using the same species sample dimension (n). Species pools came from: CR species (IUCN CR) (IUCN 2010), AZE species (AZE 2010) and top ranked EDGE species (Collen et al. 2011) (see text for details). IUCN threatened species were included as a general reference. WRS test= Wilcoxon rank sum test for significance of difference in ERO values among top ERO ranked species and selected species pool; Random test = percentage of cases (out of 10,000 comparisons) where median ERO values of selected species pool resulted bigger than that of an equally sized random species sample (stratified by Red List categories composition of species pool); Species sets difference = percentage of taxa in the species pool not included in the corresponding top ERO ranked species set. * Species pool exactly corresponds to an equally sized stratified sample. † Species pool coincides with top ERO ranked species sample. ‡ Species without a defined ERO value (due to a lack of phylogenetic information) have been excluded from this analysis (see *Materials and Methods* and APPENDIX 4.S1 for details).

Mammals' CR species have on average a higher intrinsic extinction risk than Endangered species (EN; IUCN 2001), and EN species have a higher intrinsic extinction risk than Vulnerable species (VU; IUCN 2001); there is however large overlap among the categories (Fig. 4.2). This was reflected in an almost complete overlap of the ERO probability density function for EN and CR species, which also had a similar median value (Wilcoxon rank sum test, p=0.67). Conversely VU species had a significantly smaller median value (Wilcoxon rank sum test, p <<0.01 for CR vs VU and EN vs VU). Due to their high intrinsic extinction risk value, CR species show the biggest overall loss in conservation value when using the ERO metric instead of IUCN category (Fig. 4.2c) and this also influenced our prioritization analysis.

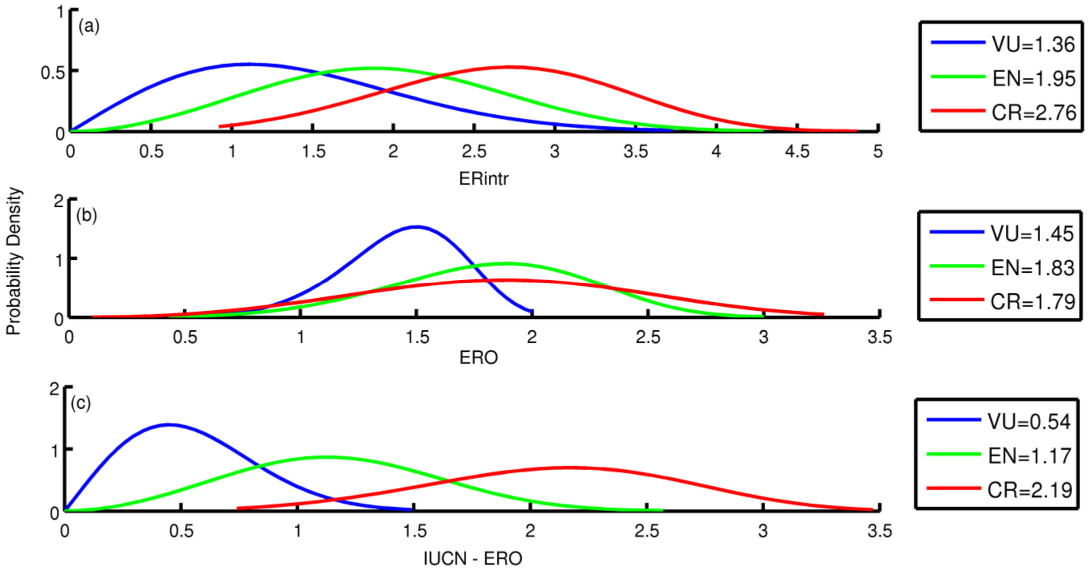

Fig. 4.2 Approximated probability density functions for extinction risk descriptors. Weibull distributions were fitted to the data, data are sorted by Red List categories (IUCN 2001). *a)* ERintr = intrinsic Extinction Risk; *b)* ERO = Extinction risk Reduction Opportunity; *c)* IUCN - ERO = "IUCN minus ERO", representing the difference in species value when adopting the ERO metric instead of Red List categories. In each graph the median value of the metric for each Red List category is reported in the legend.

An average 59.8% (sd = 36.9%) of the species' range was included within the priority area for top-ranked ERO species, and 40 out of 139 species (28.8%) were underrepresented with respect to their target (mean proportion of covered representation target = 45.6%, sd=36.5%). Only 2 out of 139 top-rank ERO species (1.4%) were excluded from the top 5% ERO priority area (both them were part of the top 10% area). The priority area selected for CR species conservation, included on average a bigger portion of CR species range (mean proportion of included range was 87.4%, sd=20.9%) and 51 out of 139 CR species (36.7%) were underrepresented with respect to their target (mean proportion of covered target =74.8%, sd=23.2). We discovered that our approach, focusing on opportunity rather than likely loss, significantly alters spatial conservation priorities for mammals. Priority areas detected according to CR species distribution overlaps only partially with ERO priority sites (Fig. 4.3; see also example in Fig. 4.S1 for a detailed interpretation), with 61.85% of the area being selected only under one or the other criterion. Only 7.04% of the ERO priority areas fall into the current protected area network (WDPA 2010), even though half of them (48.02%) have already been included into the earth's biodiversity hotspots (Mittermeier et al. 2005).

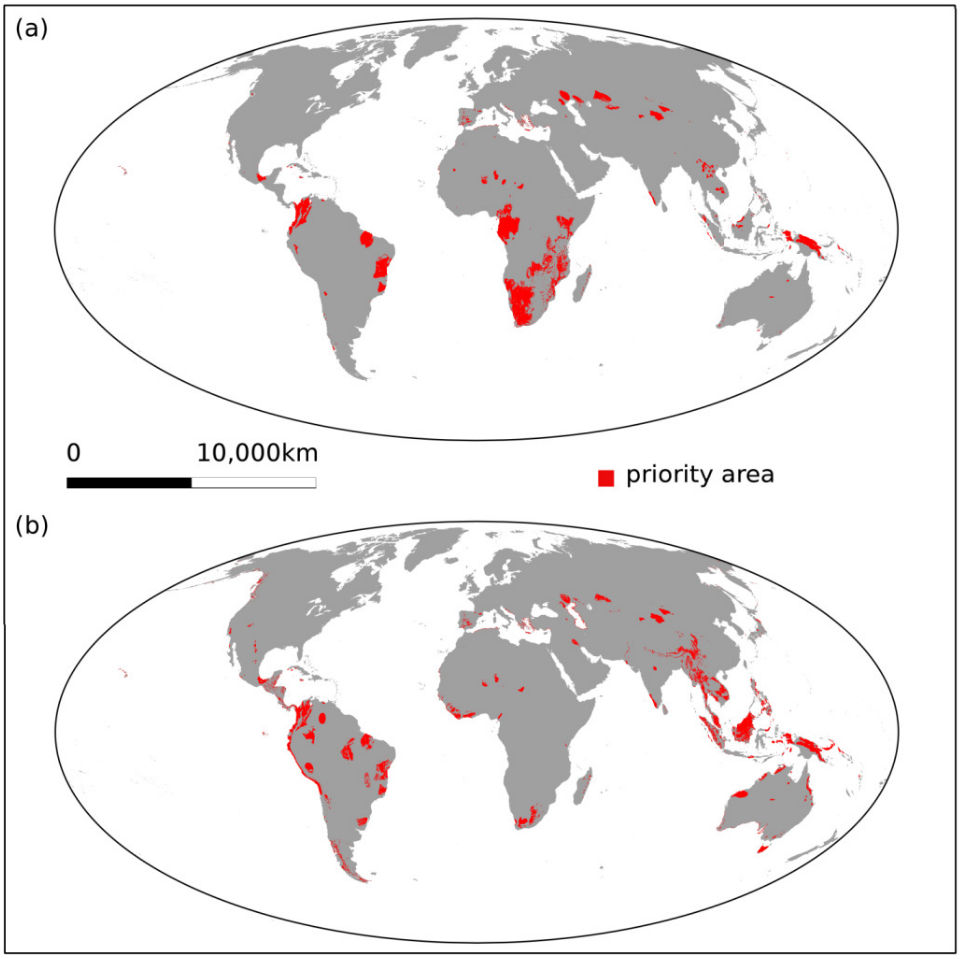

Fig. 4.3 Top priority areas detected for conserving threatened species. Priority areas include the highest ranked 5% of cells. **a)** Priority areas for CR species (IUCN 2010); **b)** priority areas for top-rank ERO species. Scale-bar and colors are the same in both maps, cell size is 100 km^2 (Antarctica was excluded from analysis).

4.5. Discussion

ERO and latent extinction risk (Cardillo et al. 2006) describe different (almost opposite) aspects of species' extinction risk. While latent risk identifies species with a potentially high future risk of decline despite being currently non-threatened (a proactive approach that aims to anticipate future species declines), ERO identifies species that are facing an imminent risk of decline and have the biological potential for recovery (a reactive approach to potentially solvable problems). Detecting species with a high ERO value thus allows conservation planners to optimize the short-term efforts for minimizing species declines. Our results suggest that more conservation emphasis should be placed on areas that are important for extinction risk reduction, especially (but not exclusively) in South America and South-East Asia, where the biggest proportion of top-ranked ERO species is found (Fig. 4.3 and Fig 4.S1). Currently these areas are largely unprotected, and only partially included in biodiversity hotspots; this is undesirable, considering the limited expenditure in biodiversity conservation and the rate of on-going biodiversity loss (Butchart et al. 2010).

Species with a high extinction risk deserve immediate conservation attention, but not all threatened species have the same intrinsic extinction risk value, hence the same opportunity of being recovered (Table 4.S2). For example, both Saiga Antelope (*Saiga tatarica*) and Javan Rhinoceros (*Rhinoceros sondaicus*) are CR species in the Red List (IUCN 2010); by calculating an intrinsic extinction risk of 1.12 for the former and 4.34 for the latter, the ERO approach suggests that investing in conservation of Saiga (ERO = 3.1) will provide a greater contribution to reducing overall mammal extinction risk compared to Javan Rhinoceros (ERO=0.53). We did not account for information on conservation investment for threatened species, which is a factor affecting species recovery (i.e. spending more conservation money on Javan Rhinoceros might increase its chances of recovery, despite its low ERO value). The ERO metric does not preclude the possibility of greater investment in lower-ranked species; it simply provides a way of ranking threatened species while accounting for their recovery potential.

We recognize that there may be considerable uncertainty involved in ranking one species over another for 3 main reasons: i) species considered to have an equal threat status (i.e. the same Red List category) do not necessarily have the same current probability of extinction; ii) some species are Data Deficient and cannot be assigned to a threat status category; iii) extinction risk models vary in their predictive power among clades. None of these points affect the overall advantages of the ERO framework if compared to previous approaches; shifts in species ranking are possible in response to both future conservation status changes and updated life-history information.

It has been stressed in the past that spending money on the most threatened species is not an efficient way of allocating limited conservation funds (Possingham et al. 2002). Our results support this idea, showing that CR species have on average a higher intrinsic extinction risk than EN or VU species. In this study we have used ERO to rank threatened species under the IUCN Red List, but the basic principle of prioritizing the most easily-recoverable threatened species could be extended beyond this by combining ERO with other prioritization schemes. For example, AZE (Ricketts et al. 2005) focuses on species restricted to small and isolated sites (and generally having a small ERO value). ERO values could provide a way of ranking AZE species by accounting for their recovery potential. Other recent approaches to conservation prioritization incorporate phylogenetic information in the definition of species conservation value (Redding & Mooers 2006; Isaac et al. 2007; Collen et al. 2011), yet without detecting the most biologically profitable opportunities for conservation. Incorporating ERO into metrics such as EDGE (Isaac et al. 2007; Collen et al. 2011) could allow fine tuning of phylogenetically-based conservation priorities. Moreover, using ERO will provide recommendation that are clearly interpretable in conservation terms, as desirable in order to augment the relevance of comparative studies of extinction risk to conservation practice (Cardillo & Meijaard 2012).

In our analyses we explored a portion of the extinction risk reduction problem with a new emphasis on recovery opportunity, but we did not consider all the factors affecting species recovery. In particular, we did not account for specific conservation actions in our prioritization analysis (Wilson et al. 2011), and we did not consider conservation costs related to the actions (i.e. what is the cost of preserving one or more viable Saiga populations in the next n years?). We do not claim that our metric will provide the final solution to the global conservation prioritization problem, yet it will add a necessary (and currently disregarded) piece of information. Economic and social factors must be considered when defining a conservation strategy and we believe that ERO would be a valid component of a comprehensive prioritization framework that takes these factors into account, as suitable data become available.

Future risk projections for species (Visconti et al. 2011) could also be integrated into the analyses, in order to take into account species' potential for recovery under different extinction risk scenarios, or to account for predicted changes in the primary threatening processes (e.g. climate change). Even though different conservation metrics are designed to address different conservation objectives, ERO has the potential to integrate with every metric that aims to maximize the likelihood of conservation success. Several conservation programs such as the IUCN's "Save Our Species" program (www.sospecies.org) and the "Mohamed bin Zayed" species conservation fund (www.mbzspeciesconservation.org), orient their call to the conservation of a particular group of species or areas; these programs may benefit of a metric such as ERO to evaluate the expected efficacy of several proposed research projects in terms of potential extinction risk reduction. Moreover, conservation agencies such as IUCN, Wildlife Conservation Society (www.wcs.org) and Conservation International (www.conservation.org) may explicitly include status recovery as a requirement for the definition of a global conservation strategy. Future research should focus on the definition of a combined (and comprehensive) species' conservation metric that accounts for ERO value. Defining a combined metric could directly affect the adoption of a joint conservation strategy that could in turn raise the chance of having a more cooperative effort among several existing conservation agencies.

CHAPTER V

Comparing multiple species distribution proxies and different quantifications of the Human Footprint map, implications for conservation

5. Comparing multiple species distribution proxies and different quantifications of the Human Footprint map, implications for conservation[4]

5.1. Abstract

Anthropogenic threats drive species to extinction and are the focus of extinction risk analyses and conservation planning. Threats are often quantified through higher level proxies, such as the human footprint (HF). We tested the effects that multiple methods of representing species' distribution and different quantifications of a HF map have on threat measurement, and how these influence conservation decisions. We quantified the magnitude of HF for 901 Southeast Asian mammals according to several methods. We ranked the species according to the measured HF value, and produced priority lists of top-impacted species. The different representations of species' distribution caused significant disagreement in HF calculations. HF values were on average lower when calculated in species' suitable habitat or occurrence points in comparison to the whole geographic range. Biases were non-linear and dependent on distal factors, such as the proportion of suitable habitat within species' range and species' habitat specialism. Using different HF quantifications also yielded disagreement, with 2-56% difference observed in species membership among priority lists. Threatened species were best predicted, and significantly placed in the top-ranking, when measuring their proportion of range exposed to high levels of HF. We thus show that the HF extent, not only its average value, influences species extinction risk. A well-framed global conservation strategy must address the quantification of human impact on biodiversity. The selection of quantification methods has implications for how such impact is evaluated. Improving techniques to quantify

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[4] *Based on Di Marco et al., Biological Conservation, 2013.*

biodiversity threats will enhance the effectiveness of extinction risk analyses and conservation decisions.

5.2. Introduction

Extrinsic threatening factors, such as habitat degradation, invasive species and climate change, are proximate drivers of species' declines and extinctions (Fisher et al., 2003; Purvis et al., 2005; Cardillo et al., 2005). Intrinsic life-history and ecological traits often mediate the risk posed by threats (Cardillo *et al.*, 2008), and can therefore serve as powerful predictors of decline or extinction (Davidson *et al.*, 2009; Di Marco *et al.*, 2012). However, quantifying the potential influence that human impact has on species, is fundamental to extinction risk analyses (Murray et al., 2011; Cardillo & Meijaard, 2012; Davidson et al., 2012), i.e. the broad category of studies whose aims are to "discover and describe generalizations about patterns and processes in the decline or threat status of species" (Cardillo & Meijaard, 2012). Moreover, threat quantification is a key component of conservation planning (Pressey et al., 2007) and it informs prioritization of conservation investments (Wilson et al., 2011).

Numerous methods have been used to evaluate and quantify exposure of species to threats (Salafsky *et al.*, 2008), yet the effects of adopting one quantification method over another have not been systematically explored. Additionally, a number of methods are available for spatially representing a species' distribution (reviewed in Rondinini *et al.*, 2006; Franklin, 2009; Boitani *et al.*, 2011), including extent of occurrence (EOO, often represented by distribution range polygons) (IUCN, 2012), habitat suitability models (HSM; e.g. Rondinini *et al.*, 2011b), species distribution models (SDM; Elith *et al.*, 2006), and points of confirmed species occurrence (PO, often contained in integrative databases such as the Global Biodiversity Information Facility; GBIF, 2012).

The diversity of threat quantification methods currently available, in addition to the diversity of species' distribution proxies, results in the combination of a large number of conceivable and measurable threat values for species. For example, the mean human population density (HPD) within a species' EOO is often used as a proxy of human impact on species (e.g., mammals: Cardillo *et al.*, 2005, 2008; amphibians: Bielby et al., 2008 and Sodhi *et al.*, 2008). Other approximated measures of human impact, including various measures of human influence and human footprint, are used in extinction risk analysis as well (e.g. mammals: Fritz et al. 2009; birds: Lee & Jetz, 2011; plants: Davies et al., 2011). Indirect proxies of human activity intensity, such as per-capita GNP, have proven to correlate well with the proportion of threatened mammal species per country (Kerr & Currie 1995). Other ways (i.e. different from mean value) of measuring threat-species overlap are also encountered (e.g., 5th percentile of HPD within a species range; Cardillo *et al.*, 2008). The median value, instead of the mean, has been used to quantify levels of other extrinsic factors (e.g. median level of evapotranspiration; Cardillo *et al.*, 2008), and an average over PO has been used instead of the mean across the EOO in some cases (e.g., mean environmental suitability for a pathogen; Murray *et al.*, 2011).

Here we investigate the effect of adopting multiple proxies of a species distribution and using different quantifications of a map representing human impact, on species threat assessment. We focus on a widely used global map of anthropogenic impact on landscapes, the Human Footprint (HF; Sanderson *et al.*, 2002). The HF metric, a biogeographically normalised version of the Human Impact Index (HII), combines information on human population density and levels of land use (Sanderson *et al.*, 2002), and is a general measure of human impact on biodiversity. The HF and HII, have been used as proxies of threat in recent analyses of extinction risk for mammals (Safi & Pettorelli, 2010; Yackulic et al. 2011) and birds (Lee & Jetz, 2011). While HF does not include information on the presence of specific operating threats (e.g. direct kill or presence of invasive species), it includes variables (e.g. human population density) that have proven to approximate this information fairly well (Spear et al., 2013). Its importance as an indirect proxy of threat to species (representing human encroachment) and its relevance in structuring global extinction risk have been demonstrated (Lee & Jetz, 2011). Despite we consider HF to be an appropriate proxy of human impact on species in Southeast Asia, we do not claim its use as a general proxy for threat levels everywhere. For example, we recognise that the distribution of invasive species would be a more indicated proxy in other places, like Australia (Evans et al. 2012).

As a model species group, we chose terrestrial Southeast Asian mammals because detailed information on their distributions is publicly available (Catullo *et al.*, 2008) in a number of forms that are broadly representative of available data for various other taxa. Mammals in Southeast Asia are a relevant model group for exploring the effect of variability among HF quantification methods given their exposure to generally high levels of human influence on the landscape (Sodhi *et al.*, 2004). Focusing our analyses on one comprehensive (and composite) proxy of anthropogenic impact, one (relatively well-studied) taxonomic group and one geographic region afforded greater control over our ability to demonstrate the general complexity affecting the quantification of human impact. Our general findings are nevertheless relevant to all situations in which threat levels are measured for species, a feature common to many global and regional analyses (e.g., Hof *et al.*, 2011)

We systematically test the effect of combining different species' distribution proxies and HF quantification methods, on the measurement of human impact on species. We also compare the results of our threat quantifications with an independent measure of extinction risk, the Red List categories of the International Union for the Conservation of Nature (hereafter IUCN) to verify which method is most effective for detecting threatened species from HF information alone. Red List categories have been widely used as a response variable in extinction risk analysis (e.g. Cardillo et al. 2005, 2008; Davidson et al., 2009, 2012; Di Marco et al., 2012) and their correlation with HF values has been demonstrated in mammals (Yackulic et al., 2011).

5.3. Methods

5.3.1 Description of data sources

We selected an updated version of the Human Footprint map (WCS & CIESIN, 2005) as a spatial proxy of anthropogenic impact on species. The HF map was described as "the sum total of ecological footprints of the human population" (Sanderson *et al.*, 2002), it was derived from several different data sources divided into four main types: population density, land transformation, accessibility, and electrical power infrastructure. We used a raster map at 1 km resolution at the equator, with values ranging from 0 (no human impact) to 100 (maximum human impact). Since our analyses were done at a regional scale, we preferred to use human footprint over human impact index (the two maps were highly correlated in the study region; R=0.9997). Our spatial scale also minimised possible sources of heterogeneity in HF map sources definition, potentially affecting analyses at a larger (e.g. global) scale.

We used the Southeast Asian Mammal Databank (SAMD; Catullo *et al.*, 2008) to obtain information on the distribution of 901 terrestrial mammal species in South East Asia. We included all the species for which a habitat suitability model was developed, according to the species-habitat relationships in terms of land cover, elevation range, and water presence collected from the scientific literature (see Catullo *et al.*, 2008 for further details). We obtained three proxies of species' distribution (Fig. 5.1): extent of occurrence (EOO), extent of suitable habitat (ESH) and points of occurrence (PO). The EOO represents the current species' distribution range, while the ESH represents the suitable area within the species' distribution range when considering species' habitat preferences. We classified as "suitable" as all the area of medium and high habitat suitability according to SAMD habitat models (raster maps with 1 km resolution at the equator), and "unsuitable" as all the areas of low and no suitability. Species' POs were collected with various methods during the SAMD project, including field surveys and analyses of literature. We analysed PO for 84 species (out of 901), selecting only those species for which at least 10 points were available.

Finally, we created two additional datasets of simulated occurrence points to evaluate the effect of different sampling techniques on threat quantification. The first dataset contained random points (30 points per species) generated within each species' ESH. The second dataset contained random points (30 points per species) generated within the portion of species' ESH more easily accessible to humans. We defined an area as "accessible" if it was potentially reachable within one day’s travel from a major city, according to Nelson' s (2008) global accessibility map. The rationale here was to simulate a biased dataset where species were more likely sampled in easily reachable areas, which is a common scenario for poorly studied species (Rondinini *et al.*, 2006). In this way, we could measure the effect of having unbiased (i.e. the first simulated dataset) or biased (i.e. the second one) points when measuring the level of a threat, and compare them with actual points.

Our dataset contained different distribution proxies for each species, where only one of these (typically EOO) is usually available in other studies; in this way, we could provide a full comparison of different quantification methods for the same set of species. We used data published between 2002 and 2008, all of them based on sources mainly collected during the 1990s and early 2000s.

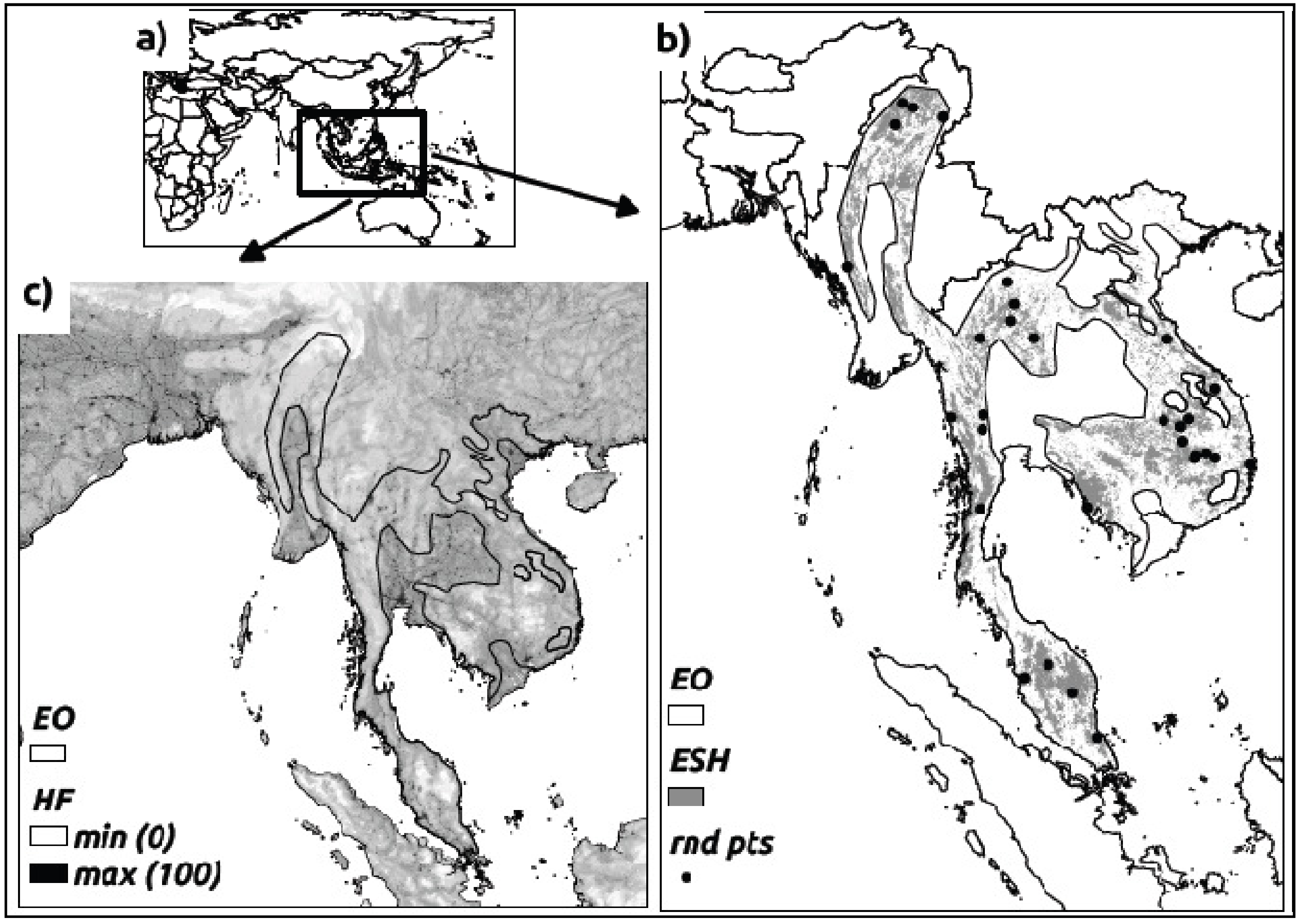


Figure 5.1. Distribution of the large-spotted civet, Viverra megaspila, and associated map of HF. (a) Global view showing the study region. (b) Regional view showing the species' distribution according to EOO (black line), ESH (grey shade) and random occurrence points (black points) within ESH (see text for details of rnd pts). (c) Regional view showing HF (in grey scale) overlaid with species' EOO (black line). For this example species, suitable habitat covers 34.2% of the EOO and mean HF value is 26.9, 20.3 and 19.6 for EOO, ESH and random points respectively.

5.3.2. Spatial and statistical analyses

We tested whether different methods of quantifying a particular threat for a species yielded different results in terms of the measured threat level. We also tested what effect any variability in threat quantification might have on a simple conservation prioritisation analysis based on threat level ranking. We organised our analyses according to six research questions, each related to one analytical step (see also Table 5.1 for a summary). We performed sensitivity tests on the thresholds used for threat quantification in steps Va, Vb, VIb, VIc (see APPENDIX 5.A, for a summary of the sensitivity settings).

(I) Do the mean, median and modal values of HF differ?

We tested the effect of using different HF quantification methods (representing central tendency) within the same species' distribution proxy (EOO), by evaluating the significance of differences in the values obtained according to different methods (paired two-tailed t-test), and their level of correlation (adjusted R^2).

(II) Do mean HF values calculated across three different proxies of species distribution (EOO, ESH and PO) correlate well with each other?

We tested, with an approach similar to that in point *(I)*, the effect of calculating the mean HF value when different ways of representing species' distributions are adopted. In this case we were interested in comparing the level of HF in suitable habitat (i.e. where a lower human presence is expected) with that in the whole species range, to see how these related to HF values measured across points of occurrence.

(III) How similar are the mean HF values calculated in three different datasets of PO (one real and two simulated), and how do they relate to the value calculated within species' ESH?

We tested, with an approach similar to that in point *(I)*, the effect that different sampling techniques used to obtain species PO have on HF quantification, when using PO to approximate species' distributions.

(IV) What is the role of habitat prevalence, measured as the proportion of suitable habitat within a species range (ESH/EOO), in generating different mean values of HF for ESH and EOO?

We verified the type of relationship between habitat prevalence and the difference in mean HF values calculated in ESH and EOO, by fitting different curves to the data (linear regression model, logistic function model, polynomial models and generalized additive model; Wood, 2006) and comparing their performance with AIC. This allowed us to identify whether the disagreement between quantification methods deviated significantly from uniform (linear) and quantify how predictable the biases were. Additionally we verified whether the difference in mean HF calculated in EOO and ESH was equivalent among specialist species (those occurring only in one major habitat type) and generalist species (occurring in more than one major habitat type; Rondinini *et al.*, 2011b).

(V) What is the relationship between HF prevalence and mean HF value?

(V.a) We measured the proportion of species' EOO overlapping with areas of high human impact (i.e., the "prevalence" of areas with high HF within EOO) and evaluated its relationship with the mean HF within EOO. In this context, we identified areas of high human impact as the areas where values of HF were larger than a given threshold. We initially set a threshold of HF=25 since this value was close to the average HF value in our study region (mean=24.5). To investigate the influence of setting this arbitrary threshold, we then performed a sensitivity analysis by varying the threshold from 10 to 90 (APPENDIX A). We associated the obtained proportional EOO values to the mean HF within EOO, to investigate the strength of correlation between the two measures and to test whether one measure was a good predictor of the other. As for point *(IV)*, we used various models to quantify the relationship between the two variables.

(V.b) As an additional comparison, we calculated the deciles of the statistical distribution of HF values within the species' EOO (i.e. 10th, 20th, ..., 90th percentiles) and compared them with that the values obtained by using the HF prevalence method. We calculated the relationship among the quantifications obtained for each species when using these two methods according to different thresholds and deciles (81 combinations of 9 thresholds and 9 deciles were tested through linear regression; APPENDIX A).

(VI) What is the effect of adopting different HF quantification methods when defining priority species of conservation concern?

(VI.a) We ranked the species in our dataset according to their descending order of measured HF value, for each method of quantification. We then selected, for each method, a list

containing the top 10% of ranked species and conduced pair-wise comparisons to determine differences in species membership among lists.

(VI.b) Additionally, in order to evaluate the relation between measured HF values and species' extinction risk, we calculated the proportion of IUCN Red List threatened species in each of the top 10% priority lists. For this analysis, we classified the species in our dataset as either threatened (Critically Endangered, Endangered, Vulnerable) or non-threatened (Least Concern and Near Threatened), excluding species that were Data Deficient and Not Evaluated (IUCN SSC, 2001). All IUCN threatened species in our dataset are impacted by anthropogenic threats, including habitat loss due to agriculture development and/or biological resources use in the form of direct kill (e.g., hunting) or logging (i.e., habitat degradation) (Salafsky *et al.*, 2008). Our study region included over 95% of the global distribution for 64% of IUCN threatened species in the dataset, and over half of the global distribution for 90% of them (Catullo *et al.*, 2008; Rondinini *et al.*, 2011b).

(VI.c) Finally, to verify the effect of selecting a fixed proportion of top-ranked species when identifying conservation priorities, we ran a sensitivity analysis by increasing the proportion of top-ranked species selected (APPENDIX A); we then compared the proportion of IUCN threatened species in each of the priority lists defined, with the proportion of threatened species in the full dataset (binomial test for proportions).

Spatial analyses were carried out in GRASS GIS (GRASS Development Team, 2012), statistical analyses were performed in R environment (R Development Core Team, 2011).

Analytical step	Short description	Distribution proxies	Quantification method	Related results
I	Comparison of central tendency	EOO	Mean HF, Median HF, Modal HF	Fig. 5.2a + Tab. 5.2
II	Comparison of distribution proxies	EOO, ESH, PO	Mean HF	Fig. 5.2b + Tab. 5.2
III	Comparison of points of occurrence	PO, rnd pts, rnd acc pts	Mean HF	Fig. 5.2c + Tab. 5.2
IV	Role of habitat prevalence	EOO, ESH	Difference in mean HF values	Fig. 5.2d
Va	HF prevalence vs mean HF value	EOO	% areas where HF> 25 (threat prevalence)	Fig. 5.3 + APPENDIX 5.B
Vb	HF prevalence vs quantiles of HF distribution	EOO	Quantiles of HF distribution vs threat prevalence	Fig. 5.4
VIa	Comparison of species priority lists	all above proxies	all above methods	Tab. 5.3
VIb	Detection of threatened species	all above proxies	all above methods	Tab. 5.4 + APPENDIX 5.C
VIc	Sensitivity in the detection of threatened species	EOO	% area where HF>10	APPENDIX 5.D

Table 5.1 Summary of the analytical settings related to the research questions and to the results. Acronyms: EOO, extent of occurrence; ESH, extent of suitable habitat; PO, points of occurrence; rnd pts, randomly generated points,; rnd acc pts, random accessible points; HF, human footprint. A detailed description of the analytical steps is provided in section 5.3.2.

5.4. Results

In pair-wise comparisons, the results of the different quantification methods were moderately well correlated with each other (R^2>0.5), with the exception of PO (Table 5.2).

However, when performing linear regressions among different methods, high levels of variance influenced the residual standard errors (Table 5.2).

						R^2-adj (se residuals)
EOO mean	0.953 (1.529)	0.698 (3.869)	0.834 (2.849)	0.315 (3.154)	0.793 (3.189)	0.640 (4.177)
0.719 (1.548)*	**EOO median**	0.743 (3.609)	0.805 (3.121)	0.328 (3.373)	0.765 (3.438)	0.614 (4.372)
2.019 (4.223)*	1.300 (3.886)*	**EOO mode**	0.602 (4.749)	0.253 (2.716)	0.563 (4.990)	0.466 (5.480)
2.124 (2.859)*	1.404 (3.134)*	0.0113 (4.813)	**ESH mean**	0.144 (4.482)	0.940 (1.617)	0.794 (3.000)
0.551 (3.694)	-0.124 (3.775)	-1.970 (3.687)*	-1.655 (4.974)*	**PO mean**	0.119 (3.866)	0.081 (3.949)
2.112 (3.232)*	1.396 (3.483)*	0.096 (5.108)	-0.033 (1.653)	1.513 (5.106)*	**Rnd pts mean**	0.741 (3.426)
-0.046 (4.183)	-0.768 (4.381)*	-2.069 (5.524)*	-2.187 (2.999)*	-0.937 (4.953)*	-2.156 (3.424)*	**Rnd acc pts mean**
Mean diff (sd diff)						

Table 5.2. Descriptive statistics of the relationships among different methods of representing Human Footprint (HF) (reported along the diagonal, indicating both row and column headings). Values above the diagonal represent the adjusted R^2 values and the residual standard errors (in parenthesis) of a variable predicting the variables on its left. Values below the diagonal represent the mean and standard deviation (in parenthesis) of the differences between HF values calculated with different methods. EOO mean, median, and mode: the mean, median and modal HF value within species' extent of occurrence. ESH mean: mean HF value within species' extent of suitable habitat. PO mean: mean HF value across species' points of occurrence. Rnd pts mean: mean HF value across species' random points. Rnd acc pts mean: mean HF value across species' random accessible points (see methods for details). * Asterisks denote that two variables have a significant mean difference (paired two-tailed t-test).

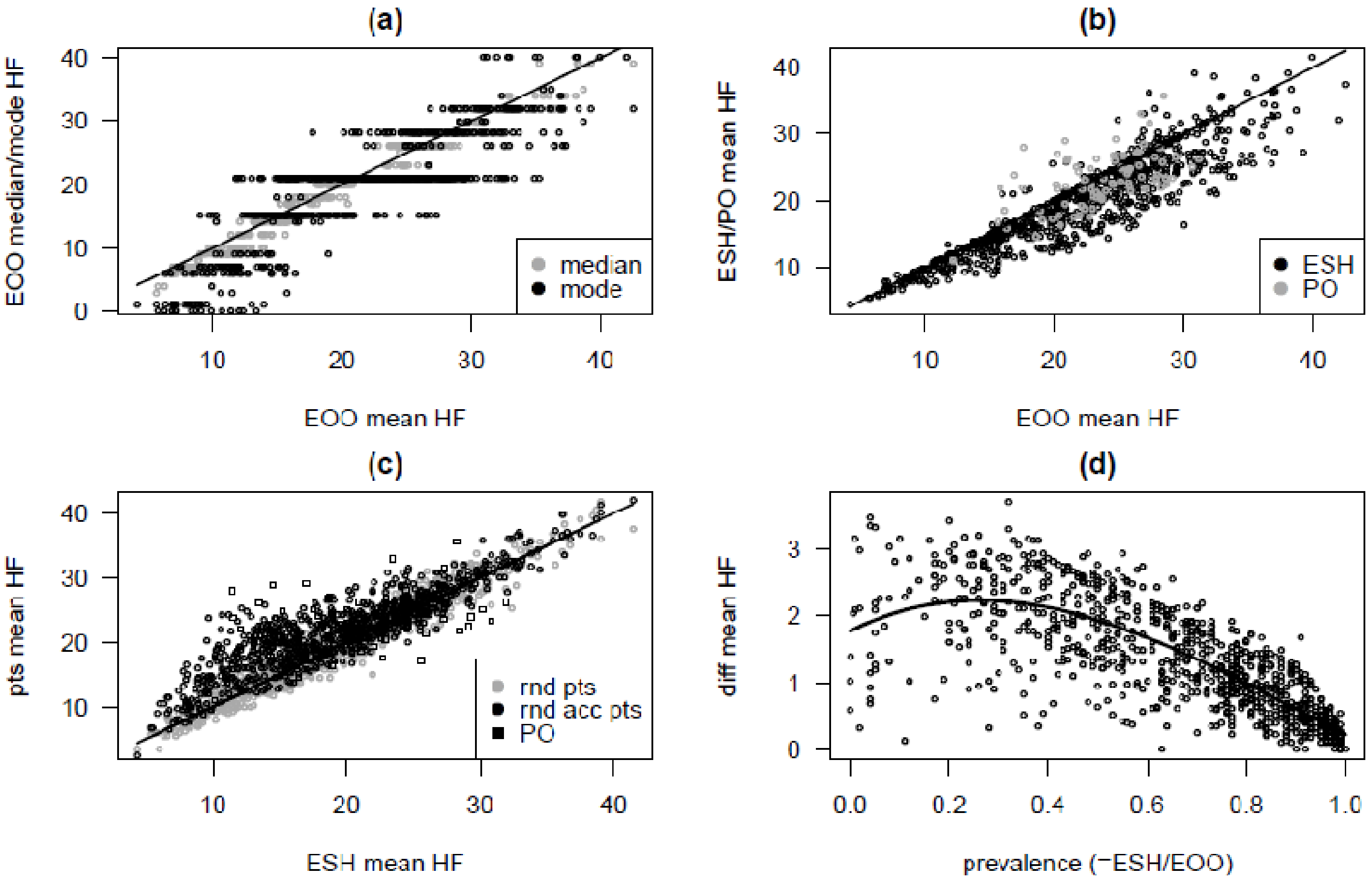
Despite being well-correlated, different quantification methods resulted in significantly different values of HF in most cases, with 15 out of 21 pair-wise comparisons showing a significant difference (paired two-sided t-tests) in the value of HF as calculated with various methods (Table 5.2). We address the questions reported in the methods section sequentially below.

(I) Despite being well correlated with the mean value, the modal and median values of HF within EOO presented high levels of variation, including both over and under estimation (Fig. 5.2a). For example, species with a median HF value of 21 had a mean HF ranging from 10 to 35 (i.e. half to almost twice the median value).

(II) ESH and, to a lesser extent, PO had on average a smaller mean HF value than EOO (Fig. 5.2b). An average difference of $\Delta_{HF}=2$ is found when measuring mean HF in species' ESH instead of EOO (Table 5.2).

(III) While random points appeared to be a good proxy for ESH when calculating mean HF, both PO and random accessible points tended to overestimate HF with respect to ESH, especially for low values of mean HF (Fig. 5.2c).

(IV) The prevalence of suitable habitat influenced the differences in mean HF values calculated within EOO and ESH (Fig. 5.2d). Species with a lower proportion of suitable habitat within EOO were more likely to have very different mean HF values for EOO and ESH. On opposite, the difference decreased for species with higher habitat prevalence values (difference=0 when habitat prevalence=1, by definition). We found a moderate level of correlation (adjusted $R^2=0.61$) between the habitat prevalence and the square-root of the absolute difference in mean HF within EOO and ESH, yet the relationship was non-linear (a 3rd degree polynomial curve was the best fitting model). Specialist species showed on average significantly higher differences in mean HF (mean difference=2.66, sd=2.71) than generalist species (mean= 2.18, sd=3.1; $p<0.05$, one tailed t-test). This is partly related to the significantly higher level of habitat prevalence that characterises generalist species (mean prevalence=0.69, sd=0.26) with respect to specialist species (mean prevalence=0.64, sd=0.26; $p<0.01$, one tailed t-test).


(a)
ECO median/mode HF
0
10
20
30
40
10
20
30
40
EOO mean HF
median
mode
(b)
ESH/PO mean HF
10
20
30
40
10
20
30
40
EOO mean HF
ESH
PO
(c)
pts mean HF
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20
30
40
10
20
30
40
ESH mean HF
rnd pts
rnd acc pts
PO
(d)
diff mean HF
0
1
2
3
0.0
0.2
0.4
0.6
0.8
1.0
prevalence (~ESH/EOO)

Figure 5.2. Comparison of different methods for quantifying HF for 901 South-east Asian mammal species. Species are represented as open dots in all the graphs. (a) Mean HF (x axis), median HF and modal HF (y axis) within species' EOO. (b) Mean HF within species' EOO (x axis), mean HF within species ESH and across species' PO (y axis). (c) Mean HF within species' ESH (x axis), random points (rnd pts), random accessible points (rnd acc pts) and PO (y axis). (d) Relation between species' habitat prevalence (i.e., proportion of suitable habitat within species' range; x axis) and difference in mean HF value within EOO and ESH (calculated as the square root of the absolute difference; y axis). The lines in plots (a)-(c) represent an exact 1:1 correspondence, while the line in plot (d) represents the fitted 3rd degree polynomial model (best supported model). See Table 5.2 for a description of the variables.

(V.a) A high level of correlation (GAM model; adjusted R^2=0.93; p<0.01) was found between the mean HF within EOO and the proportion of EOO areas with HF>25 (Fig. 5.3), showing a predictable but non-linear relationship between the results of these two methods. A clear pattern was present when changing the HF threshold of 25, with lower thresholds performing better than higher thresholds in correlating with mean HF (APPENDIX B). The proportion of EOO areas with HF>20 was the metric that best correlated with mean HF values (adjusted R^2=0.94), while using higher thresholds resulted in poorer correlation.

(V.b) Using a fixed HF threshold to measure HF prevalence, or calculating a given percentile of the HF distribution within species' EOO gave results that correlated only partially with each other. In fact, only when using low HF thresholds (≤30) was the resulting quantification well correlated with that obtained by measuring HF distribution percentiles (adjusted R^2 >0.75) (Fig. 5.4).

(VI.a) The derivation of priority species' lists was influenced by the HF quantification methods adopted, with the identification of lists that differed by 2-56% in terms of species membership (Table 5.3). This could result in having up to half of the species being included or excluded from a conservation prioritisation plan depending on the HF quantification method adopted.

(VI.b) The proportion of EOO overlapping with areas where HF>25 was the best proxy of species' IUCN threat status with 39% threatened species included in the top priority list (Table 5.4). However its performance was not significantly different from that of the mean and median HF within EOO. Under no methods, among those listed in Table 5.4, did the top 10% priority species list contain a proportion of IUCN threatened species that differed significantly from that of the full species dataset (mean=28%), indicating a performance not significantly

better than random. However, in subsequent sensitivity analyses, using lower HF thresholds improved predictive ability significantly. When using a threshold of HF>10, the priority species list contained a proportion of IUCN threatened species significantly higher than random (43%; $p < 0.05$, binomial test for proportions; APPENDIX C). As a reference note, 175 species (20% of the total) have 100% of their EOO overlapping with areas where HF>5, and 46% of these are threatened.

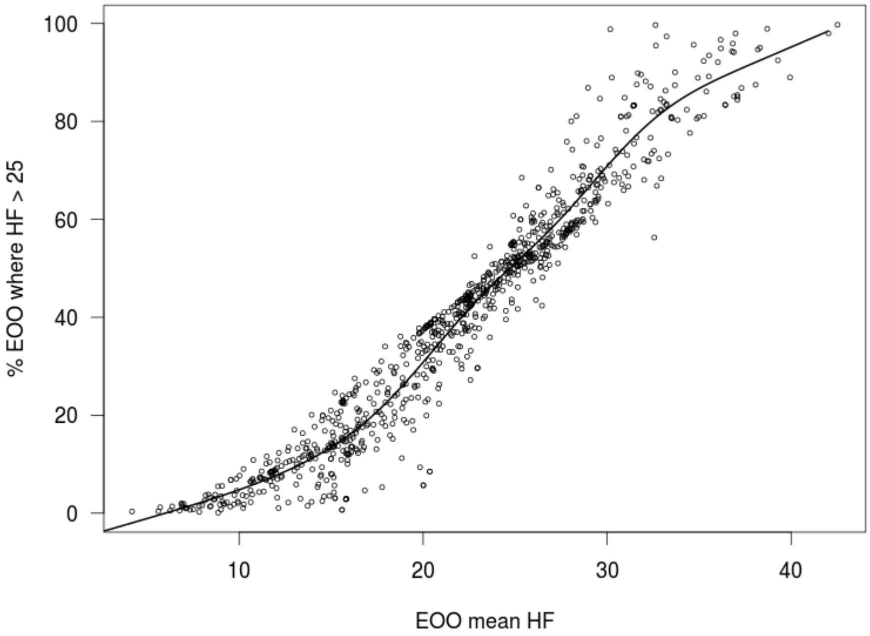

Figure 5.3. Comparison of the mean HF values within species' EOO (x axis) and proportion of species' EOO overlapping with areas of HF higher than 25 (y axis). Each dot represents a species, the line represents the fitted GAM model (the best supported model). Sensitivity analyses using varying HF thresholds are reported in APPENDIX B.

(VI.c) The selection of a 10% rank threshold to identify priority species lists resulted in the identification of a significantly higher proportion of IUCN threatened species than that contained in the full dataset. No other rank threshold performed better in detecting IUCN threatened species (APPENDIX D), when ranking species according to their proportion of EOO overlapping with areas where HF>10 (i.e. the most supported method for predicting IUCN threatened species).

Finally, the prevalence of suitable habitat was also a good predictor of IUCN threatened species, with 41% of the species prioritised by this method being threatened (significantly higher than random, $p<0.05$).

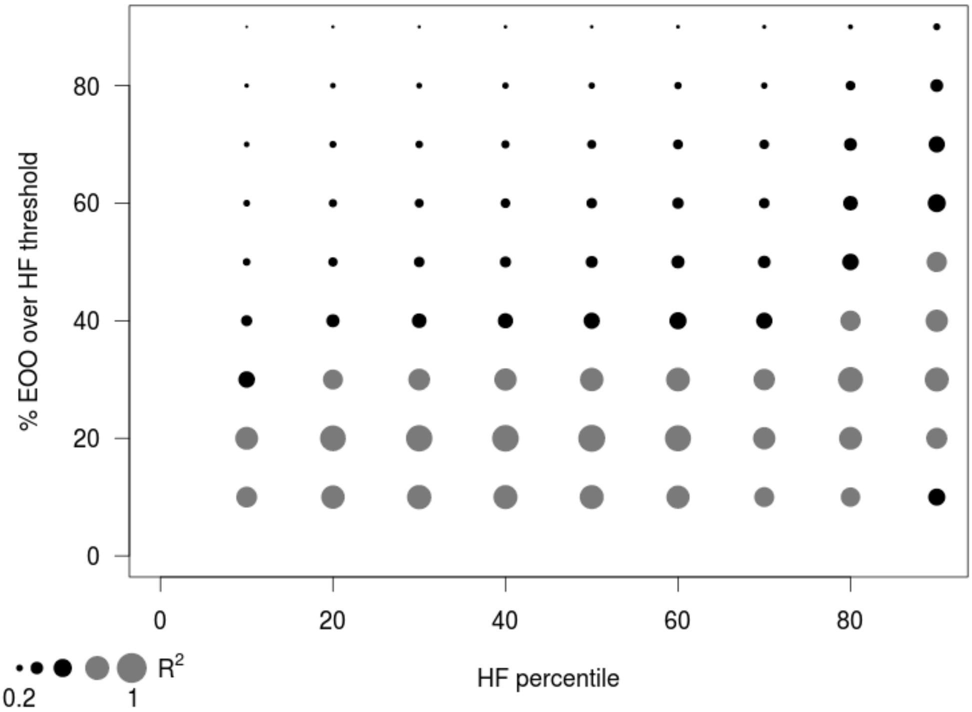

Figure. 5.4 Comparison of the distribution percentile quantification method (x axis) *vs* the threat prevalence quantification method (y axis). The distribution percentile method calculates a given percentile (i.e. 10th , 20th, ..., 90th) of the distribution of HF within the species' EOO. The prevalence quantification method calculates the proportion of species' EOO overlapping with areas of HF larger than a given threshold (i.e. 10, 20, ..., 90). The correlation between threat values measured according to different methods (and thresholds) was calculated through all the 901 species. The size of dots is proportional to the correlation level (adjusted R^2 value) for each combination of threat prevalence and threat distribution percentile (gray dots indicate $R^2 > 0.6$).

EOO mean	2.22	21.11	43.33	43.18	44.32	14.44
	EOO median†	11.76	42.86	46.34	46.34	15.29
		EOO mode†	51.35	55.56	51.39	33.33
			ESH mean	17.05	14.77	45.56
				Rnd pts mean	30.00	47.78
					Rnd acc pts mean	50.00
						%EOO where HF>25%

Table 5.3. Pairwise comparisons (% difference in species list membership) among threat quantification methods (reported along the diagonal, indicating both row and column headings) when creating priority species lists on the basis of threat impact. Lists were defined by selecting the top 10% of ranked species for each method. See Table 5.2 for a description of the variables. Occurrence points were excluded from this analysis because they were only available for a subset of species. † NB Using "EOO median" and "EOO mode", several species in the bottom part of the list had the same rank; the priority lists were thus cut at thresholds of 8.3% for median and 9.4% for mode, to avoid an arbitrary selection among equally ranked species.

Quantification method	N. threatened species	Difference significance
EOO mean	25 (74)	none
EOO median†	20 (71)	none
EOO mode†	12 (62)	none
ESH mean	17 (76)	none
Rnd pts mean	15 (76)	none
Rnd acc pts mean	15 (77)	none
%EOO where HF>25%	30 (76)	significantly higher than: EOO mode, ESH mean, Rnd pts mean, and Rnd acc pts mean.

Table 5.4. Number of IUCN threatened species in high-priority lists defined with various threat quantification methods. Lists were defined by selecting the top 10% ranked species, data deficient species were then removed from the lists. See Table 5.2 for a description of the variables reported in the first column. The second column reports the number of threatened species and the number of total species (in parentheses) in the lists. The third column reports details on the significance ($p \leq 0.05$) of difference in the proportions of threatened species found according to various methods (binomial test for proportions). Occurrence points were excluded from this analysis since they were only available for a subset of species. † NB Using "EOO median" and "EOO mode", several species in the bottom part of the list had the same rank; the priority lists were thus cut at thresholds of 8.3% for median and 9.4% for mode, to avoid an arbitrary selection among equally ranked species.

5.5. Discussion

We showed how the method for representing species' distributions, and the method by which human footprint is quantified across different distribution proxies, have a significant effect on threat levels measurement. This could greatly affect decisions on priority conservation actions based on threat levels, and the ability to identify species threatened with extinction.

Even where different HF quantification methods yielded values that were moderately well correlated across species, these correlations were not typically linear indicating that biases may be complex to account for. Similarly, important levels of uncertainty may be present when using one method to predict the results of the others, suggesting that even subtle differences among methods could make the measurement of a species' threat value inconsistent. For example, differences were evident when using various measures of central tendency, with the mean value of HF being poorly representative of the median or modal value. Similarly, the method for representing species' distributions influenced HF value calculations, with the mean HF value being generally higher if measured in EOO compared to ESH or PO. This result is not surprising considering that habitat suitability models usually consider as "unsuitable" those portion of a species range characterised by high levels of human population density and intensive land use. Nonetheless it allowed us to show how the mean HF values measured in PO are intermediate between those measured in EOO and those measured in ESH. Additionally we showed that unbiased (random) points yielded non-significant differences in calculating mean HF compared to ESH while biased points (sampled in easily reachable areas) were characterised by higher values both respect to ESH and to real (collected) points, but not significantly higher respect to EOO (Table 5.2).

These differences could in turn result in the identification of differing levels of anthropogenic impacts to a species, depending on the HF quantification method adopted. When identifying priority species lists on the basis of measured HF value, this result translated to over half of the species selected according to one method (e.g. EOO mode) being disregarded according to another method (e.g. ESH mean). Similar effects should be expected for other analyses which employ quantitative information on threats, such as extinction risk analyses or conservation planning exercises. This highlights the importance of using sensitivity analyses wherever methodological variability affects threat calculations.

Despite our results are quantitatively dependent on our analytical settings (i.e., measurement of Sanderson' s human footprint for mammals in South East Asia), our general findings (and analytical approach) have broader relevance, especially for situations where spatial data availability may be more limited than ours. We assume that uncertainty related to threat quantification methods characterises every situation where a threatening process must be measured, no matter which particular threat or analysis resolution are considered.

Habitat suitability models are currently available for mammals (Rondinini *et al.*, 2011b) and birds (Jetz *et al.*, 2007) globally, and for vertebrates and plants regionally (Araújo *et al.*, 2005). EOO polygons are available for most terrestrial vertebrates (IUCN, 2012) but not commonly for invertebrates (but see Clausnitzer *et al.*, 2009). PO are often available for species across a range of taxa, although spatial sampling bias can be problematic for inference (Phillips *et al.*, 2009). Here we showed that distribution data availability can influence threat value calculations and subsequent analyses. For example, a (non-significantly) greater proportion of IUCN threatened species were prioritised when calculating mean HF within EOO compared to ESH.

Our results (Table 5.4) suggest that EOO is, on average, a better choice for measuring the influence of human impact on threatened species than ESH or PO. Even though some portion of a species' EOO may not contain suitable habitat, human presence in these areas could indirectly affect the level of impact on suitable areas trough contagion (a mechanism also driving habitat clearance; Boakes *et al.*, 2010). Ecological traits, such as habitat prevalence and habitat specialism influenced the disagreement in threat values calculated among methods. Measuring HF within ESH was more problematic when only a small proportion of a species' EOO contains suitable habitat and when the species was a habitat specialist (e.g., only occurring in forest). Rondinini, *et al.* (2011b) showed that IUCN threatened species have, on average, lower values of habitat prevalence compared to non-threatened species. Accordingly, we found that the prevalence of suitable habitat is itself a good predictor of IUCN threatened species in our dataset. This recalls that the proportion of suitable habitat remaining within a species' range is a direct consequence of the habitat loss that has affected the species. When available, the prevalence of suitable habitat (proxy for the remaining natural habitat) is a useful complement to classical tests of anthropogenic impact on species.

The prevalence of high HF values within species' distributions correlated well with mean HF values. However, the relationship between the results of these two quantification methods was again non-linear (Fig. 5.3). Species marginally affected by human impact (i.e., those characterised by intermediate values of mean HF) tended to have relatively higher proportion of their EOO falling in areas of high HF in comparison to species having high or low HF values. Similarly, the prediction error was lower for those species facing average HF values. This means that for species facing an unusually high or low level of human impact (compared to the regional average), threat prevalence is not necessarily well correlated to mean threat value. Additionally, when using a low threshold, values of HF prevalence were well correlated with the low quantiles of the statistical distribution of HF within species range. For example, the 20th percentile of the HF distribution was highly correlation with the proportion of species range in areas having HF bigger than 20 (Fig. 5.4). This makes the results of the two quantification methods (both related to threat extent) comparable, provided that appropriate thresholds are used.

Through our analysis we did not aim to describe the causal relationship between threats and extinction risk (as measured in the Red List), yet our purpose was to demonstrate how the determination of the role of a given threat in predicting extinction risk is affected by the particular quantification method adopted. In fact measuring the proportion of EOO overlapping with areas of HF>10 was the most effective method for identifying IUCN threatened species in the top ranking of HF impact. At this threshold, the proportion of top-ranked threatened species was significantly higher than that present in the whole dataset. In post-hoc tests, we showed that a HF threshold of 5 was even better in identifying threatened species (albeit not significantly better); however, in this case the selection of a priority species list (e.g. containing the top 10% ranked species) may be problematic because 20% of the species had the same measured threat value and thus the same rank (a problem also encountered, at minor levels, when using median and modal measures). This suggests that, when setting a threshold to measure threat prevalence, a balance must be reached in the trade-off between prediction ability and classification ability.

Measuring threat prevalence has an analogy with the comparative philosophy of the IUCN Red List criteria (IUCN SSC, 2001), which are used to assign an extinction risk status to species by comparing their attributes (e.g., population size, distribution range size, decline rate) with various reference values. Similarly, quantifying the proportion of high threat areas within species' ranges allows to compare the threat extent with the average condition of the region where species occur. For example two species, one living in areas where high levels of HF are concentrated in a few sites (e.g., cities) and the other living in areas where moderate levels of HF occur more extensively (e.g. rural areas), could have the same mean HF values even though their levels of perceived threat would be different. Measuring threat prevalence would be better indicated than measuring a mean threat value, to emphasise this difference. This highlights the importance that threat distribution, not only its magnitude, has in determining a species' extinction risk and adds to calls for better and more realistic threat mapping.

5.5.1. Conclusions

The quantification of threats to biodiversity is an important component for conservation-oriented exercises, such as extinction risk modelling (Cardillo & Meijaard, 2012) and conservation planning (Pressey *et al.*, 2007), and can act as a link between them. Our results demonstrated that changing the methodological settings by which a given threat is quantified, could directly affect conservation decisions, such as the identification of priority species to be included in conservation planning analyses. We can derive a set of general recommendations from our results to standardise the process of threat quantification:

- various methods of threat quantification (e.g. mean threat value, extent of high threat areas, statistical distribution of threat values, etc.) should be tested to identify the best proxy of extinction risk for a given analysis;

- the impact of quantification variability on results (e.g. due to threshold setting) should be accounted for, through sensitivity analyses;
- measuring threat values in both suitable and unsuitable portions of a species' range is preferable when the threat has potential contagious effects (like human footprint);
- the proportion of suitable habitat in a species range should be measured and accounted for in extinction risk modelling when possible;
- sampling biases should be considered and accounted for (e.g., by using informed species distribution models) when using species' occurrence points to measure threat levels;
- the human footprint prevalence (its extent) within species' EOO appears to be a better proxy for species extinction risk respect to mean human footprint value;
- the level of threat in the study region should be measured and used as a benchmark to set sensible thresholds when measuring threat prevalence;
- in general, lower thresholds are preferred when measuring threat prevalence but the possibility of having equally valued species must be accounted for (again through sensitivity analyses).

The identification of priority species for conservation is a key component in the definition of global conservation strategies (Rondinini *et al.*, 2011a). Accounting for well quantified threats, together with species' ecological traits, will help in improving the effectiveness of species prioritisation protocols. We expect that improving techniques to quantify biodiversity threats and understanding their biases and limitations will have important flow-on effects for planning effective conservation actions and structuring informative extinction risk models, both now and under future forecast scenarios. A poor threats quantification can alter the evaluation of the threats role in extinction risk analysis, often reducing the conservation relevance of the analysis itself. An improved threats quantification can instead inform conservation decisions, thus contributing to reducing the gap between science, policy and practice.

CHAPTER VI

General discussion

6. General discussion

6.1 Evaluation of the PhD objectives achievement

Four key objectives of this PhD research were defined in the first chapter (section 1.5.2.), in addition to various research aims that have been detailed in each of the research chapters presented (Chapter II, III, IV and V). These key objectives provided a rationale for the definition of a research framework that resulted in the preparation of 8 research papers (4 of them being the core of this thesis and described in the research chapters). This section provides a short description of the achievement of the aforementioned objectives.

Objective 1. Through the characterisation of species' habitat preferences (in terms of land cover, elevation, human tolerance and water needs) we were able to refine the spatial information on global terrestrial mammal species distribution (over 90% species modelled). With our models, we detected that an average 59% (sd=28%) of suitable areas characterise the global distribution range of a mammal species. We described differences in habitat availability according to species taxonomic, biogeography, threat status and land cover preferences. We also derived high-resolution synthetic maps of global species richness both within species ranges and within species suitable habitats. We found that global representation of biodiversity, through species richness maps, is affected by the way in which species distribution is mapped, and a global average of 19 species per pixel are only represented by their geographic range (i.e. not by the presence of suitable habitat).

Objective 2. We collected past information on the threat status of the world's carnivores and ungulates, two groups than include many long-lived species and some of the most iconic (and flagship) mammals, such as the Giant Panda, *Ailuropoda melanoleuca*. We performed retrospective assessments on the extinction risk of species in those two groups and built a database synthesizing all the collected information. We were able to quantitatively characterise the recent decline of these groups (from the 1970s up to date) and to show the temporal and spatial patterns in this declining process. We were also able to interpret the described decline with respect to geopolitical events, international regulations, shifting cultural values and natural resource exploitation that have characterised the human population in the past 40 years.

Objective 3. We used a published dataset that includes information on multiple life history traits of terrestrial mammals. We statistically addressed the missing information present in the database and combined it with a recently updated source of species phylogeny. We then calculated for each species an intrinsic risk of extinction, a measure that characterises a species' intrinsic ability to recover from a declining condition (provided that causal threatening factors are stopped). By combining this information with information on current species' threat status, we were able to identify those species that are currently threatened with extinction and have a high potential to be recovered. We discovered that many of the species that we identified are not currently prioritised by global conservation strategies, and live in areas that are largely unprotected. We propose that the protection of key areas where these species occur represents a short term priority for minimising the extinction risk of terrestrial mammals.

Objective 4. We performed a comprehensive analysis of the human footprint on Southeast Asian mammals, a group that has recently suffered from an exacerbation of habitat loss and direct kill and that is facing a steady increase of the human population density. We compared multiple quantification techniques to measure the level of human footprint on species and showed how different the results could be between various quantifications. We showed that conservation related information, such as the level of species endangerment, can vary substantially, depending on the quantification method adopted. Sensitivity testing allowed us to provide a few general guidelines that can improve the way in which conservation scientists address the issue of quantifying threats to biodiversity under a wide range of available methods.

6.2. Conservation relevance of the key research outcomes

6.2.1. The conservation value of habitat suitability models

A better representation of the spatial distribution of species has the potential to improve our definition of global priorities for conservation interventions. Using species geographic ranges to represent a species distribution implies the assumption that all the areas within the range have an equal chance to be actually occupied by the species. Through the use of habitat suitability models, we were able to identify those portions of a species' range that are more likely to be actually occupied by the species. These models allowed us to perform subsequent analyses of spatial conservation prioritization, as well as to test the impact of anthropogenic threats on species (by verifying the overlap of species' habitat with threat processes).

The potential conservation relevance of habitat suitability models, such as those presented in Chapter II for mammals and those presented by Jetz et al. (2007) for birds, has still to be fully explored at a global scale. As discussed in Chapter II, habitat suitability models can refine the representation of species distribution, thus serving as a basis for conservation planning purposes as in the case of Chapter IV (i.e. Step I in Margules and Pressey 2000: "Compile data on the biodiversity of the planning region"). Additionally their use for monitoring the status of species has a great potential. In fact models can be constantly updated on the basis of newly available base maps, particularly land cover maps. In this way the models can be used to track changes in the extent of suitable habitat over time. This monitoring process could be used to evaluate rates of loss of suitable habitat for the species, a characteristic than can inform IUCN Red List criteria A and B (IUCN 2001) in the definition of species threat status categories.

The proportion of suitable habitat remaining within a species' range is a direct consequence of the habitat loss that has affected the species. The use of habitat models in extinction risk analysis is promising too. We showed, both in Chapter II and in Chapter V, that habitat suitability models can be used to predict species extinction risk (as defined by IUCN Red List category). In fact, both by using global scale models (Rondinini et al. 2011) and regional scale models (Catullo et al. 2008), the amount of "habitat prevalence" (i.e. the proportion of suitable habitat within a species' range) was a good predictor of species extinction risk.

6.2.2. The importance of historical information in providing evidence for conservation scientists

Evaluating historical information on species threat status allows conservation scientists to evaluate the long-term effect of external drivers of species' extinction risk change, including the diffusion of threatening processes and the implementation of conservation strategies. Data on past species status are often available, but generally scattered across multiple data sources and datasets. The availability of such sources is increasing, and today many IUCN-related literature sources are available for online download (e.g. all IUCN action plans are available from: *www.iucn.org/about/work/programmes/species/publications/species_actions_plans/*). Yet, in order to be used, such information not only needs to be stored, but also needs to be synthesized, classified and checked by experts: it needs to be included into a database.

A successful example of database creation is the PanTHERIA database "a species-level database of life history, ecology, and geography of extant and recently extinct mammals" (Jones et al. 2009). In that datasets, the authors have collected information on the biological traits characterising terrestrial mammal species (e.g. body mass, litter size, gestation length, etc.), and made it available in a form that is easily (and freely) accessible by others. Similarly, we have collected, classified and checked past information on species threat status (see details in Chapter III) and we included it into a database (see Table 3.S1, section 8.2.3). The analysis of this database, allowed us to quantitatively characterise the recent global decline in the conservation status of carnivores and ungulates, and its relation to external drivers. Additionally, we envisage that many additional researches may be based on our dataset, and a clear potential for combining it with biological datasets (such as Pantheria) does exist (see sections 6.3. below).

We showed that collecting historical information on species threat status is doable with a reasonable amount of research effort (2 operators and 6 experts consulted), and the data can be classified while accounting for potential uncertainty (that anyway had little impact on our general description of a 4-decadal deteriorating trend). The methodology that we have used to create our datasets, can be easily transferred to other mammal orders, as well as other species groups. The limiting factor for such an exercise would be the availability of historical information, yet the possibility of using expert consultation and the collection of contextual and anecdotal information (as sometimes done in the current Red List assessment process) could be explored, given that a rigorous check protocol is enforced. We stress the importance of providing and properly storing information on species characteristics (such as their threat status). We showed how this information can be used to provide direct evidence of conservation strategies effectiveness as well as to inform future conservation decisions.

6.2.3. Defining short term priorities for interventions

Conservation science faces two key challenges (Rondinini, et al. 2011): one is preventing the extinction of species under future scenarios of threats expansion (Visconti et al. 2011), the other is minimising the extinction risk of species which are currently threatened (Di Marco et al. 2012). The success of global conservation efforts in performing these two crucial tasks will determine our ability to preserve the world's biodiversity for future generations. In a recent paper (Rondinini et al. 2011), we discussed how these two tasks can be classified under the general topics of "proactive" and "reactive" conservation strategies.

Under the proactive strategy, conservation interventions are planned to minimize the possibility of having future species' decline, in a way that is "preventive" and aims at protecting species while they are still abundant, rather than acting when they are highly threatened with extinction (and interventions may arrive too late or they can be too expensive to be implemented). Under the second strategy, conservation actions are oriented toward species that are currently threatened with extinction and are already facing a real risk of being lost unless threatening processes are stopped.

Both the aforementioned strategies present potential disadvantages. The proactive strategy may suffer from uncertainty in the prediction of future threat distribution, and both species and areas identified as being potentially impacted by future threats expansion does not necessarily correspond across different definitions of proactive priorities. For example, South America was identified as a future hotspot of mammal species loss by Visconti et al. (2011) while Southeast Asia was identified as a basin of "latent extinction risk" by Cardillo et al. (2006), and we identified sites within both these two areas to be important future battlegrounds for minimising the impact of agriculture and logging on mammal species (Wilson et al. 2011).

On the other hand, reactive strategies may focus on species that are already facing a too-high risk of going extinct. Those species may thus represent a "risky" investment for conservation for at least two reasons (Bottrill et al. 2008). First, they may require a disproportionate amount of conservation resources, and this would necessarily impact the availability of funds for other species, under a condition of limited funds availability. Second, they may go extinct despite of conservation interventions, in case factors such as infectious diseases or inbreeding depression (often related to the small population size characterising highly threatened species) are manifested.

In Chapter IV, we presented a novel approach to define reactive conservation strategies. In fact, we proposed that the risk of implementing unsuccessful short term conservation intervention could be minimised by accounting for species' intrinsic ability to recover from a declining condition. Species' biology could be used as a proxy for informing conservation scientists on intrinsic species' recovery probability. We showed that species within the same class of extinction risk (as assessed in the IUCN Red List; IUCN 2011) have very different biological characteristics, and therefore show a different response to external pressures. Species that are currently threatened with extinction, but are likely able to recover from such a condition (provided that threatening factors are stopped) represent a valuable short term conservation priority. Many of the priority species that we identified are distributed in Southeast Asia. Such an area has been characterised by a recent expansion of intensive agriculture and plantations (Sodhi et al. 2004) as well as rampant levels of hunting (Corlett 2007). In Chapter III, we also showed that this area (more than any other) has been characterised, in recent years, by a deterioration in the conservation status of carnivores and ungulates. Additionally, further expansions of the threats in that area are predicted to occur in the future (Wilson et al. 2011). The combination of these findings supports the necessity of defining short term priorities for conservation intervention, to avoid losing (several) species that have recently become threatened, have a potential to be recovered and face a serious risk of suffering from increasing levels of human pressures.

6.2.4. Dealing with methodological uncertainty in conservation

Methodological uncertainty has the potential to affect conservation decisions. In Chapter III, we accounted for uncertainty in the assessment of retrospective Red List categories by assigning a set of potential Red List categories to each species for which contrasting information was available. We then used such information to derive a range of potential extinction risk trends (through a Monte Carlo resampling procedure). In Chapter IV, we showed how the definition on global priorities for the conservation of mammals depends on the way in which we assign a specific conservation value to species (i.e. a proxy of conservation relevance). In Chapter V, we showed how the way in which we quantify a given threat may affect the evaluation of the impact of that threat on mammal species. Additional sources of uncertainty include the methods with which we represent the distribution of a species. In a recent paper (Visconti et al. *In press*), we showed how the way in which we represent the spatial overlap between protected areas and species ranges may alter substantially our definition of gaps in the conservation status of South American mammals. For example, missing information on the boundaries of protected areas causes both an over- or an under-estimation of the level of species protection (Visconti et al. *In press*). This in turn can affect our monitoring of conservation progress as well as achievement of international targets, such as those established by the Convention on Biological Diversity (CBD 2010).

While methodological uncertainty should not prevent the implementation of conservation actions, in front of an accelerating global biodiversity crisis, the potential shortfalls of not properly accounting for uncertainty in conservation have been already presented (Regan et al. 2005). We showed that sensitivity analysis represents an important way to deal with uncertainty in the definition of conservation-relevant parameters values. Sensitivity testing of critical variables can help scientists to fully explore the level of uncertainty affecting the data and to properly report the results of conservation relevant analyses. Providing decision makers with a set of alternative scenarios will help the definition of stronger conservation strategies. This has been demonstrated with the comparison of scenarios of impact of land cover change on future mammal species distribution (Visconti et al. 2011). Additionally, we have recently projected uncertainty in future information associated with the Red List assessments (Rondinini et al. *In Press*), and showed that, under multiple scenarios, the current budget would be insufficient to sustain a growth in the number of assessed species and will result in having most of the Red List dataset being out of date (i.e. a condition where information on species' threat status is too old to be considered reliable for conservation purposes). In that paper, we have found that different strategies for maintaing the Red List database up to date would require an increase in budget, and propose the enhancement of web-based assessment tools as a relatively cheap strategy to constantly maintain up to date the assessments for 160,000 species (i.e. the figure proposed by Stuart et al. 2010).

6.3. Limitations and caveats of the presented research

In this thesis I presented a number of innovative approaches to support mammal conservation, based on the use of some of the most recently developed research techniques and datasets. Additionally, I proposed novel analytical approaches and defined new datasets to further advance mammalian conservation theory, and biodiversity conservation in general. Such an exercise was obviously prone to a number of assumptions, discussed in each of the research chapters, and was necessarily related to a number of potential caveats and limitations. In this section, I provide a critical discussion of the most relevant limitations that affect the described work (following the same order of the presented research chapters).

Defining high-resolution patterns of the distribution of over 5,000 species is clearly a data-intensive exercise. The identification of suitable habitat for each species was standardised as much as possible, yet many potential biases exists in the amount of information available, e.g. for large-bodied vs small-bodied species. Additionally the use of species geographical range as a proxy to limit the spatial distribution of species' suitable habitat brought additional uncertainty. Species ranges suffer from a variety of potential problems (see Rondinini et al. 2006), related to the fact that they are often derived on the basis of expert-knowledge, and available information may be partial or biased for poorly-known species and/or regions. This obviously has the potential to limit the accuracy of habitat suitability models definition. Nonetheless a different modelling approach, e.g. based on the consideration of species' points of occurrence, would potentially suffer from biases in available point datasets and sampling efforts (e.g. toward easily accessible areas; see also Boitani et al. 2011). Additionally, a point-based approach would not be applicable at a global taxonomical and spatial scale, thus exacerbating the taxonomic and spatial bias of the proposed analyses (i.e. by focusing on a non-representative portion of mammalian biodiversity). Further development of the proposed models, e.g. through a mixed approach of expert-based knowledge on species' habitat preferences and statistical adjustments based on occurrence points, would improve our understanding of global mammalian distribution. Nonetheless the potential of our original approach in refining the definition of species ranges and to identify discrepancies in global representation of mammalian biodiversity was presented (e.g. see Fig. 2.3 on mismatches in species richness calculations).

The use of Red List categories and Red List Index (in Chapter III) for quantitatively characterising the trend in mammalian extinction risk over the past decades is subject to the limitations of these metrics, as mentioned in the introduction (see Section 1.5.). In fact, Red List categories does not necessarily represent the absolute distance of a species from extinction, and the Red List Index may be unpredictably affected by relative differences in the scale of improvements or deteriorations of species assessed according to different Red List criteria (e.g. criterion A on population decline vs criterion D on small population size; IUCN 2001). As mentioned in Section 1.5., other indicators of extinction risk does exists, and are likely based on more proximal measures of species decline (e.g. population abundance). The choice of using the Red List approach was based on its clear advantage in being widely applicable to all of the species in our dataset (thus being representative of our study taxa). Additionally, the presented results proved to be robust to changes in the way in which biodiversity trends were measured. In fact, a clear declining trend, 1975-2008, in the conservation status of carnivores and ungulates was observed not only according to the RLI metric but also in terms of an increasing global proportion of threatened species; this confirm that our results were not dependent on the way in which species' Red List categories were converted into relative measures of extinction risk. Moreover, the relevance of the Red List and the Red List Index metric for monitoring progresses toward achievement of the Convention on Biological Diversity targets has been clearly demonstrated elsewere (Butchart et al. 2010, Pereira et al. 2013), and the work presented here has a potential to provide additional temporal insigths on biodiversity trends monitoring.

The Extinction risk Reduction Opportunity metric (ERO) is based on the statistical approximation of species' intrinsic risk of extinction. Modelling intrinsic risk of extinction is obviously subject to a series of assumptions on the model definition and accuracy (see Purvis et al. 2000; Cardillo et al. 2005). As discussed in Chapter IV, the ERO metric represents only a partial, yet innovative, solution to the species prioritization problem. In fact other information should be accounted for, when defining priorities for intervention, e.g. evolutionary distinctiveness (Isaac et al. 2007) or cost of conservation actions implementation (Wilson et al. 2011) among the others. I envisage that an integrated approach, including several aspects of species conservation-relevance (including their intrinsic risk of extinction), would provide an improved contribution to conservation strategy effectiveness.

In Chapter V, an approach to investigate the effect of variability in the quantification of species-threat overlap was proposed. Such an approach was necessarily limited to a specific case study, where the analytical settings were controlled and the effects of adopting a number of permutations on measured species threat levels were tested. Using human footprint to approximate the level of human impact within species distribution is not necessarily appropriate in every condition (despite such an exercise was performed in many cases, at a regional and global scale e.g. see Yackulic et al. 2011). Yet the scope of the presented analysis was to show how methodological uncertainty affect conservation choices (e.g. in terms of conservation prioritization and extinction risk calculation). Such source of uncertainty is often underlying conservation-relevant decisions and not accounting for it may have potentially unpredictable effects in terms of conservation actions effectiveness. The guidelines provided in Chapter V are potentially relevant for many situations in which a particular threat is measured for a given taxon, notwinstanding the assummptions underlying our analyses.

6.4. Future research developments: understanding the interaction between factors determining extinction risk in mammals

Based on the outcomes of the presented research program, we propose that three main factors influences the global extinction risk of mammal species: human threats, species' biology and conservation actions.

The first and most immediate factor is represented by human threats. Species' extinction is a component of evolution, as well as speciation, and both are well documented in the fossil record. What is not natural, is the exacerbation of species' extinction rates associated to the recent human history (Barnosky et al. 2011). Turvey (2009), documented 255 mammals extinctions occurred in the Holocene, and 76 of them have occurred in the past 500 years (Schipper et al. 2008). The causal relationship between vertebrate species decline and anthropogenic threatening factors has been discussed in Hoffmann et al. (2010). Unsustainable level of single threatening factors have driven recent mammalian extinctions, as in the the case for the Caribbean Monk seal (*Monachus tropicalis*) that was affected by direct kill (both in the form of hunting and persecution) since its discovery by Columbus in 1494 (Kovacs 2008) and went finally extinct in 1950s. In other cases, it has been a combination of multiple factors that led to a species extinction, as in the case of the Tasmanian Wolf (*Thylacinus cynocephalus*) that was directly persecuted as a sheep predator, suffered habitat modification, faced an increasing competition with introduced domestic dogs and was affected by a distemper-like disease until his extinction occurred, in the 1930s (McKnight 2008).

The second factor influencing species extinction risk is their biology. Species biology is not itself a threatening factor, yet it determines the way in which a given species responds to threatening factors. It is possibly the most quantitatively studied, among the factors that we mentioned earlier. In fact, the large scientific literature body that addresses extinction risk analysis is mostly focused on the evaluation of the relationship between species' biological traits and species extinction risk. Many studies have addressed that relationship in mammals, including: Cardillo et al. (2005, 2006, 2008), Safi & Pettorelli (2010), Jones & Safi (2011) Davidson et al. (2009, 2012), Di Marco (2012). The fact that species' biological traits have been much more widely explored in extinction risk analysis respect to anthropogenic threats, depends on several factors (reviewed in Murray et al. *Submitted*), perhaps the most evident are the general paucity of knowledge on potentially threatening processes for many species (especially among relatively poor studied groups) and the difficulty in providing quantitative measures of threats, that is also related to the variability in threat quantification methods (see Chapter V).

The third factor that we discuss here is represented by conservation actions. Conservation actions mitigate the impact of threats on biodiversity. Overall, global biodiversity loss is accelerating, and global indicators reveal a growing mismatch between increasing pressures and slowing responses (Butchart et al. 2010). Nonetheless, the role of conservation actions in reducing the global decline of vertebrate species in the recent years has been discussed in Hoffmann et al. (2010), as well as in Chapter III of this thesis. Conservation actions are represented by a variety of interventions including the delineation of protected areas (Margules and Pressey 2000), the control of invasive species (Evans et al. 2011), the reduction of forest logging (Wilson et al. 2011) and the reintroduction of species extinct in the wild.

The roles of all of the above mentioned factors in determining species decline or recovery have been explored both theoretically and analytically. However, our understanding of how they interact to determine a species' final condition of endangerment is still limited. We believe that a key research challenge in the short term would be the exploration of such an interaction. In this thesis, we set a basis for the realization of that analysis, by introducing all the necessary elements: a detailed characterization of species distribution, a background of past changes in species extinction risk, an investigation of relevant biological traits that mediate the effect of threatening factors and a general set of rules to measure threatening factors. The next step of this research will be the investigation of the combinatory effects of all the mentioned factors and the determination of their emergent properties.

6.5. Concluding remarks

The definition of strategies for the conservation of biodiversity must account for trade-offs in spatial scales. Conservation actions are generally implemented at a local scale and thus would benefit from local coordination (Smith et al. 2009). However a lack of global scale coordination may have undesirable effects in terms of: potential duplication of efforts, spatial inequality in the availability of resources, lack of spatial and temporal coordination of conservation interventions. Having a global coordination of conservation plan has a great potential to be more cost-effective than having several scattered plans, without contrasting the need for local scale practical interventions. Mammals could represent an excellent model group for multiscale conservation planning due to their: relatively high data availability (respect to other groups), high proportion of threatened species (25%), representation in all terrestrial biomes, ability to attract public awareness toward conservation problems.

As introduced in Chapter I, Conservation Biology is the discipline of making decisions within a limited time-frame and subject to some level of uncertainty on baseline data. For this reasons, conservation strategies must have good biological support, must be easily accessible both by funding agencies and managers, and must be of immediate application. A demonstration of success in this sense is represented by the Hotspots of Biodiversity (Myers et al. 2000). By presenting the idea of focusing conservation efforts in areas characterised by the presence of endemic and threatened taxa, that strategy has been able to attract 750 million dollars in the first 15 years since it was launched (Myers, 2003). A clear advantage of the hotspot strategy, respect to systematic conservation planning, is its simplicity, it is focused only on two conservation aspects (rarity and vulnerability), and it needs relatively few data to be applied.

On the other hand, systematic conservation planning has the support of a robust methodological background and requires information on species distribution, desired conservation targets for species, available budget and cost of implementing conservation actions. Additionally, conservation plans need constant monitoring and update to account for changing patterns in the distribution of biodiversity and threatening processes (Pressey et al. 2007). The use of complementarity-based algorithms make conservation planning less easily accessible to mangers without a specific background (e.g. unlike the hotspot strategy). To date, successful applications of the systematic conservation planning approach are mostly local or regional in their scope, like the conservation plan for the Cape Floristic Region of South Africa (Cowling et al. 2003).

Despite several conservation planning analyses were performed at a global scale (e.g. Carwardine et al. 2008, Wilson et al. 2011), a clear conservation outcome was not associated to these analyses. The urgency to properly address the current biodiversity crisis calls for a more coordinative effort in defining shared global strategies for intervention (Rondinini et al. 2011). Global scale conservation planning applications are promising, also thanks to an increased availability of data on species distribution (Chapter II), historical species threat status (Chapter III), threatening processes (Sanderson et al. 2002) and proxy of land economic value (Naidoo & Iwamura 2007). However a key aspect that would determine a global scale conservation planning scheme is its potential acquisition by policy makers, funding agencies and managers.

Systematic conservation planning is a multidisciplinary science, it requires input from a multitude of fields such as mathematics (for algorithms developments), spatial ecology (for spatial representation of biodiversity features) economy and social science (for verifying actual feasibility of conservation plans). We believe that increasing the biological inputs in conservation planning, through the consideration of species' biological characteristics, represents a promising field of future research expansion where expertise from multiple backgrounds can be put together to define innovative strategies to address global scale extinction risk (see Chapter IV).

In this PhD thesis, various data sources were analysed through a combination of analytical techniques to describe and advance the current knowledge on global mammal conservation. However, additional efforts are needed to properly understand the trade-offs between many existing approaches to species conservation (mammal species in particular). We envisage that a global scale reconciliation of existing conservation techniques and strategies for intervention, can only arise from a collective effort in determining shared priorities both for conservation research and for conservation action planning. The research findings presented in this thesis will contribute to future mammal conservation by: guiding the definition of more biologically-informed conservation strategies, improving our ability to use and analyze existing evidence of conservation success and failure and providing general guidelines to address the methodological uncertainty characterising conservation choices.

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Three and a half years ago, I was enrolled in a PhD project on "global mammal conservation planning". At that time, I had absolute no experience in "global conservation planning" (i.e. 75% of topic) and very little knowledge about "mammals" (i.e. the remaining 25% of the topic). Moreover, I started this PhD without any long-term idea about potential sources of income, but with a return ticket for Australia in my hands. Any single outcome of this PhD, and any single ability that I have acquired during it, depended on the full commitment of my supervisors in transmitting to me their knowledge and, most importantly, their passion toward conservation biology.

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8. APPENDICES AND SUPPORTING INFORMATION

8.1. Appendices to Chapter 2

Table 2.S1 List of mammal species included in the analysis.

Order	Family	Genus	Species	EOO	HMS	HS
AFROSORICIDA	CHRYSOCHLORIDAE	Amblysomus	corriae	50809.14	48155.92	45041.5
AFROSORICIDA	CHRYSOCHLORIDAE	Amblysomus	hottentotus	213157.8	200861.99	200861.99
AFROSORICIDA	CHRYSOCHLORIDAE	Amblysomus	marleyi	1406.61	1316.6	1307.51
AFROSORICIDA	CHRYSOCHLORIDAE	Amblysomus	robustus	1115.73	1071.99	1071.99
AFROSORICIDA	CHRYSOCHLORIDAE	Amblysomus	septentrionalis	32846.4	30902.48	30846.61
AFROSORICIDA	CHRYSOCHLORIDAE	Calcochloris	leucorhinus	262264.95	196826.17	196826.17
AFROSORICIDA	CHRYSOCHLORIDAE	Calcochloris	obtusirostris	116128.89	60069.05	33931.02
AFROSORICIDA	CHRYSOCHLORIDAE	Calcochloris	tytonis	314.55	109.34	109.34
AFROSORICIDA	CHRYSOCHLORIDAE	Carpitalpa	arendsi	9167.85	8944.83	4856.4
AFROSORICIDA	CHRYSOCHLORIDAE	Chlorotalpa	duthieae	10933.47	924.29	924.3
AFROSORICIDA	CHRYSOCHLORIDAE	Chlorotalpa	sclateri	92152.62	53155.93	53155.93
AFROSORICIDA	CHRYSOCHLORIDAE	Chrysochloris	asiatica	41801.94	28374.49	21186.81
AFROSORICIDA	CHRYSOCHLORIDAE	Chrysochloris	stuhlmanni	104543.73	60589.75	60589.75
AFROSORICIDA	CHRYSOCHLORIDAE	Chrysochloris	visagiei	368.37	357.38	357.38
AFROSORICIDA	CHRYSOCHLORIDAE	Chrysospalax	trevelyani	40414.68	19714.67	18665.73
AFROSORICIDA	CHRYSOCHLORIDAE	Chrysospalax	villosus	46049.94	10818.74	1680.56
<The rest of the table has been omitted due to space limitations, refer to electronic material for a complete version>						

List of mammal species included in the analysis (5311 out of 5330 mammals listed in the GMA database; for the remaining 19 species no information was available on the geographic range, thus they could not be mapped). EOO: extent of occurrence; HMS: extent of high and medium suitability habitat; HS: extent of high suitability habitat. All areas in km2. Asterisk indicate species for which the habitat suitability models was not developed.

Table 2.S2 Matrix of rules used to assign suitability scores to each Globcover land cover class for each mammal

Globcover code	Globcover description	Forest					
		Low tolerance to human disturbance	Medium tolerance to human disturbance	High tolerance to human disturbance	Flooded		
					Temporary	Permanent	Saline
10	Cultivated and Managed areas	3	3	2	3	3	3
11	Post-flooding or irrigated croplands (or aquatic)	3	3	2	3	3	3
12	Post-flooding or irrigated shrub or tree crops	3	3	2	3	3	3
13	Post-flooding or irrigated herbaceous crops	3	3	2	3	3	3
14	Rainfed croplands	3	3	2	3	3	3
15	Rainfed herbaceous crops	3	3	2	3	3	3
16	Rainfed shrub or tree crops (cashcrops, vineyards, olive tree, orchards,…)	3	3	2	3	3	3
20	Mosaic cropland (50-70%) / vegetation (grassland/shrubland/forest) (20-50%)	3	2	1	3	3	3
21	Mosaic cropland (50-70%) / grassland or shrubland (20-50%)	3	3	2	3	3	3
22	Mosaic cropland (50-70%) / forest (20-50%)	3	2	1	3	3	3
<The rest of the table has been omitted due to space limitations, refer to electronic material for a complete version>							

Matrix of rules used to assign suitability scores to each Globcover land cover class for each mammal, based on the preference for broad habitat types extracted from the textual information published on the IUCN Red List of Threatened Species (www.iucnredlist.org). Suitability values for land cover classes are: 1 = high; 2 = medium; 3 = low. For example, if a species is only found in forest and has a low tolerance to human disturbance, forest classes (40 to 102) are assigned high suitability, classes of mosaic of forest/shrubland with grassland (110, 120) are assigned medium suitability, all other classes are assigned low suitability. If a species is found in more than one broad habitat type (e.g. forest and shrubland), the suitability score assigned to each land cover class is the highest. For example, a species that is found in both forest (including flooded saline) and shrubland, and has low tolerance to human disturbance, gets the following habitat suitability scores: low suitability to cultivated classes (10 to 32); high suitability to forest classes (40 to 102) and to forest permanently flooded with saline or brackish water (170); high suitability to shrubland classes (130 to 136); medium suitability to classes of mosaic of forest/shrubland with grassland (110, 120); low suitability to all other classes.

8.2. Appendices to Chapter 3

8.2.1. APPENDIX 3.S1 - List of literature sources*

* For additional details and download of the action plans see also: *www.iucn.org/about/work/programmes/species/publications/species_actions_plans/*

- African Elephants, and Rhinos. Status survey and conservation action plan. 1990. Compiled by D.H.M. Cumming, R.F. du Toit and S.N. Stuart.
- African Rhino. Status survey and conservation action plan. 1999. Compiled by R. Emslie and M. Brooks.
- Antelopes. Global survey and regional action plan, parts I,II and III. 1988. Compiled by R. East.
- Antelopes. Global survey and regional action plan, part IV. 2001. Compiled by D.P. Mallon and S.C. Kingswood.
- Asian Elephant, an action plan for its conservation. 1990. Compiled by C Santiapilliai and P Jackson.
- Asian Rhinos, an Action Plan for their conservation , 1989. Compiled by Mohd. Khan bin Momin Khan Chairman.
- Asian Rhinos. Status survey and conservation action plan. 1997. Edited by T.J. Foose and N van Strien.
- Bears. Status survey and conservation action plan. 1999. Compiled by C. Servheen, S Herrero and B. Peyton.
- Canids: Foxes, Wolves, Jackals and Dogs. Status survey and conservation action plan, 2004. Edited by C. Sillero-Zubiri, M. Hoffmann and D. W. Macdonald.
- Deer. Status survey and conservation action plan. 1998. Compiled by C. Wemmer.

- Equids: Zebras, Asses and Horses. Status survey and conservation action plan. 2002. Edited by P. D. Moehlman.
- Foxes, Wolves, Jackals, and Dogs. An action plan for the conservation of canids. 1990. Compiled by J.R. Ginsberg and D.W. Macdonald.
- Hyaenas. Status survey and conservation action plan. 1998. Compiled by G. Mills and H. Hofer.
- Otters, an action plan for their conservation. 1990. Edited by Pat Foster-Turley, Sheila Macdonald, and Chris Mason.
- IUCN (2012) The IUCN Red List of Threatened Species. Version 2012.2. Available from *www.iucnredlist.org.*
- Part VII, terrestrial mammals (excluding bats): species report. In Conservation of species of wild flora and vertebrate fauna threatened in the community, 1982. Compiled by J. Thornback, M. Jenkias and J. Gilmour . lUCN Species Conservation Monitoring Unit Cambridge.
- Pigs, Peccaries, and Hippos. Status survey and conservation action plan. 1993. Compiled by D. M. Brooks, R. E. Bodmer and S. Matola.
- Red Panda, Olingos, Coatis, Raccoons, and their relatives. Status survey and conservation action plan, 1994. Compiled by A. R. Glaston.
- Seals, Fur Seals, Sea Lions and Walrus. Status survey and conservation action plan, 1993. Compiled by P. Reijnders, S. Brasseur, J, van deer Toorn et al.
- Tapirs. Status survey and conservation action plan.1997. Edited by D.M. Brooks, R.E. Bodmer and S. Matola.
- The 1986 lUCN Red List of threatened animals, 1986. Prepared by The lUCN Conservation Monitoring Centre, Cambridge U.K.

- The 1988 lUCN Red List of threatened animals, 1988. Prepared by The lUCN Conservation Monitoring Centre, Cambridge U.K.
- The 1990 lUCN Red List of Threatened Animals, 1990. Prepared by the World Conservation Monitoring Centre, Cambridge U.K.
- The 1994 lUCN Red List of Threatened Animals, 1994. Compiled by the World Conservation Monitoring Centre, Cambridge UK.
- The 1996 lUCN Red List of Threatened Animals, 1996. Compiled and Edited by Jonathan Baillie and Brian Groombridge, IUCN SSC.
- The IUCN Mammal Red Data Book, 1982. Compiled by Jane Thornback and Martin Jenkins, lUCN Conservation Monitoring Centre.
- The IUCN Red Data Book, Vol. 1 - Mammalia. Second Edition. Goodwin, H.A. and Holloway, C.W. (1972), lUCN. (Revisions: 1973 and 1974 by Goodwin, H.A. and Holloway, C.W.; 1976 by Fitter, R.S.R. and Holloway, C.W.; and 1978 by Thornback, J.).
- The SSC Conservation Communications Programme and the Deer Action Plan, 1998. Compiled by the Deer Specialist Group
- Wild cats, status survey and conservation action plan, 1996. Compiled and edited by K. Nowell and P. Jackson.
- Wild cattle, bison and buffaloes. Their status and potential value. 1983. Compiled by J. Thornback.
- Wild sheep and goats and their relatives. Status survey and conservation action plan, 1997. Compiled by D. M. Shackleton.
- Weasels, Civets, Mongooses, and their relatives. An action plan for the conservation of mustelids and viverrids, 1989. Compiled by A. Schreiber, R. Wirth, M. Riffel and H. Van Rompaey.

- Zebras, Asses, and Horses. An action plan for the conservation of wild equids, 1992. Edited by P. Duncan.

8.2.2. Supporting Figures for chapter 3

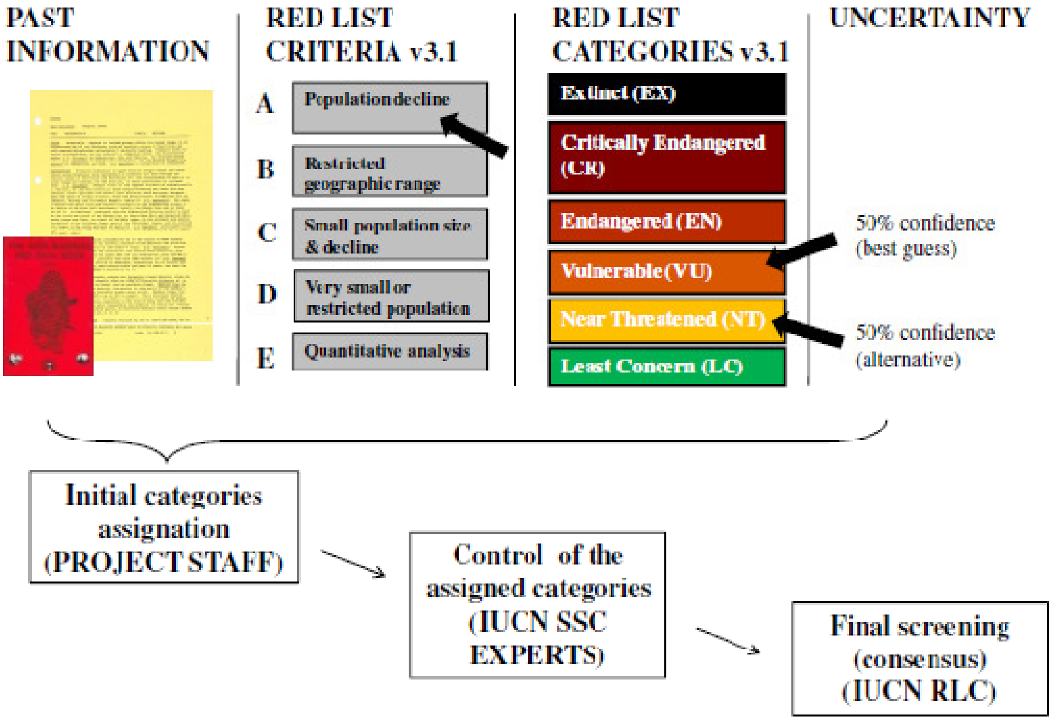

Fig. 3.S1 Work flow of the retrospective assessment process. See methods in the main text for a description.

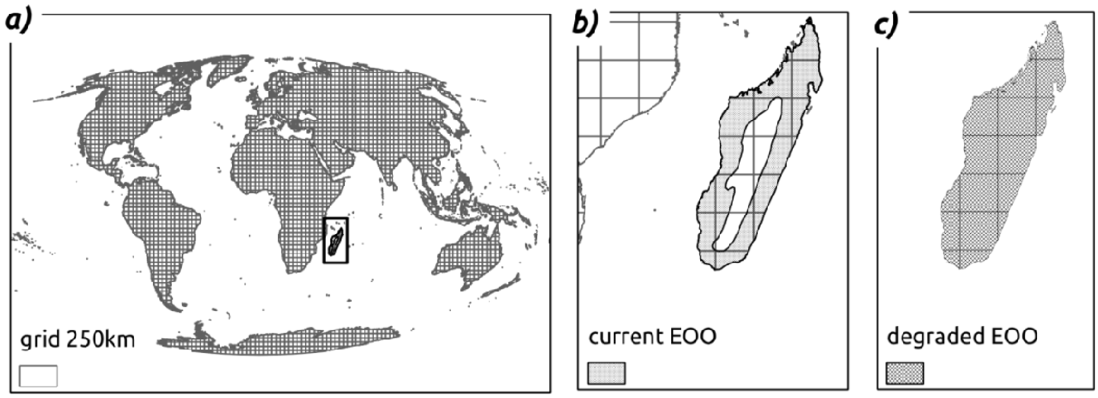

Fig. 3.S2 Definition of a spatially degraded species distribution, according to a 250km grid cell. The panel represents: a) the global placement of the example species, the Fossa (*Cryptoprocta ferox*), b) its current extent of occurrence (EOO) overlapped to the grid cells, c) the grid cells occupied by the species (degraded EOO).

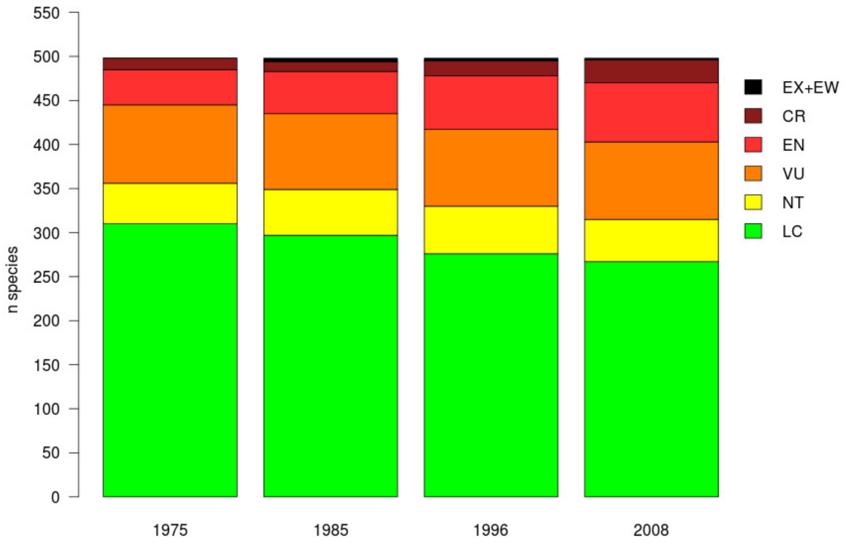

Fig. 3. S3 Number of species (carnivores and ungulates) in each Red List category from 1975 to 2008.

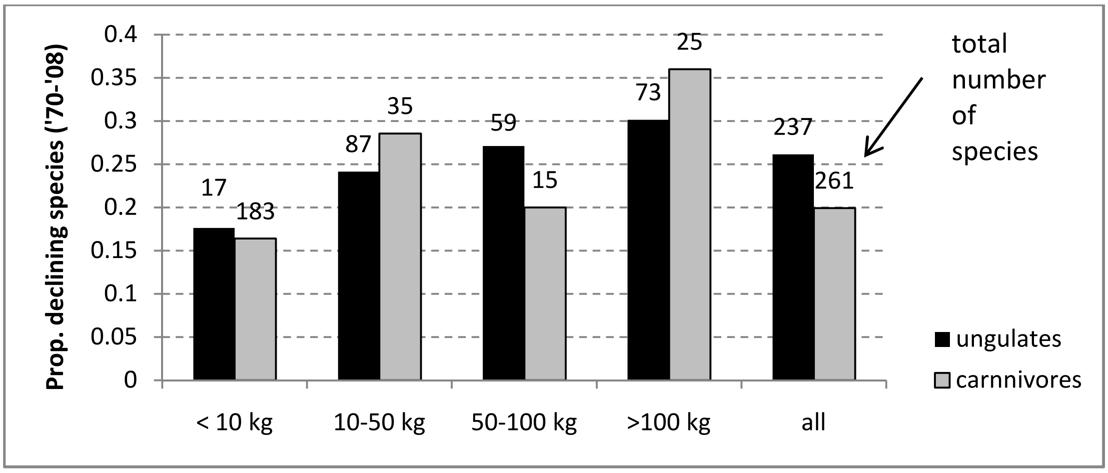

Fig. 3. S4 Proportion of declining species (1970-2008) in mammals and ungulates grouped by body mass intervals. Each bar reports the proportion of declining species for a given group in a given body mass interval; total numbers of species considered are reported on top of bars. NB The difference in proportion of declining species for a given body mass range is never significant among carnivores and ungulates (binomial test for proportions).

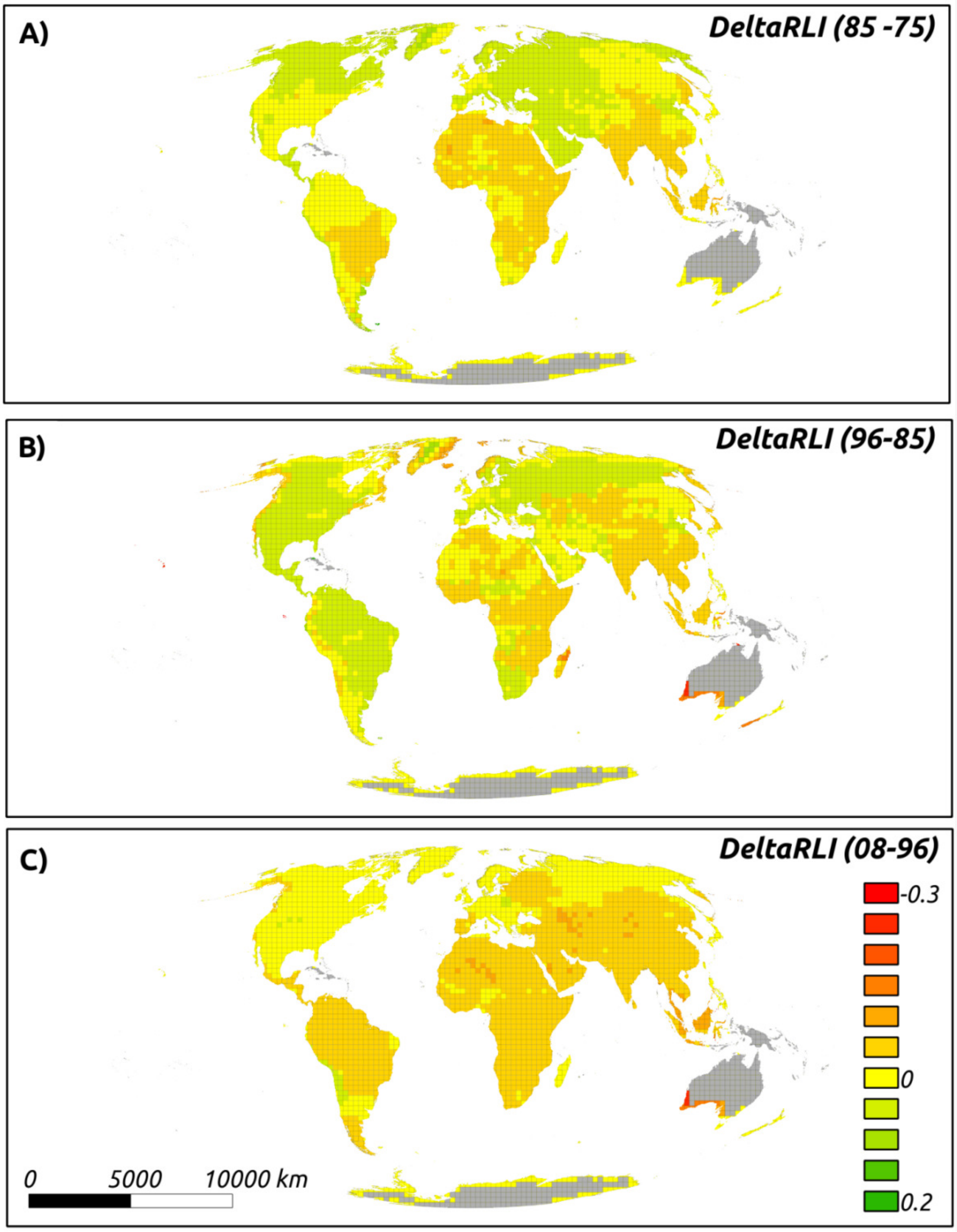

Fig. 3.S5 Spatial changes in Red List Index (RLI) values between consecutive species assessments. The maps represent, for each grid cell,the net difference between the RLI calculated in one assessment period and the previous one. Colour scale and scale bar are the same in all maps. Red scale indicates deterioration in the RLI, green scale indicates improvement, yellow indicates stability, grey indicates cells without ungulates or carnivores species.

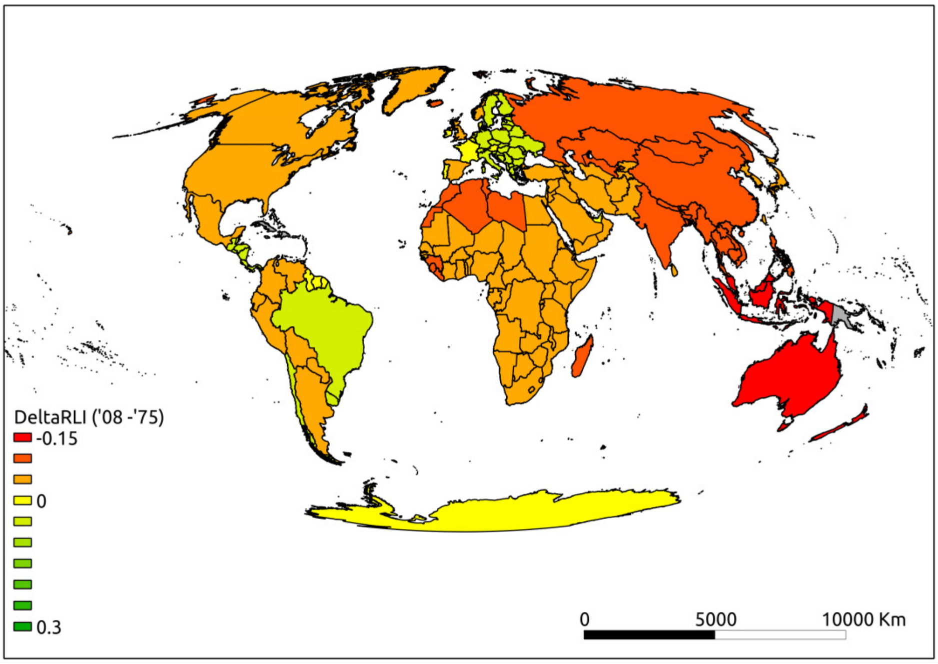

Fig. 3.S6 Changes in countries' Red List Index (RLI) values between 1975 and 2008. The maps represent, for each country, the net difference between the RLI calculated in 2008 and that calculated in 1975. Red scale indicates deterioration in the RLI, green scale indicates improvement, yellow indicates stability, grey indicates countries without ungulates or carnivores species. This analysis is based on the global Red List status of species included in each country, it is not based on national Red List assessments. See also Table S2 for further details.

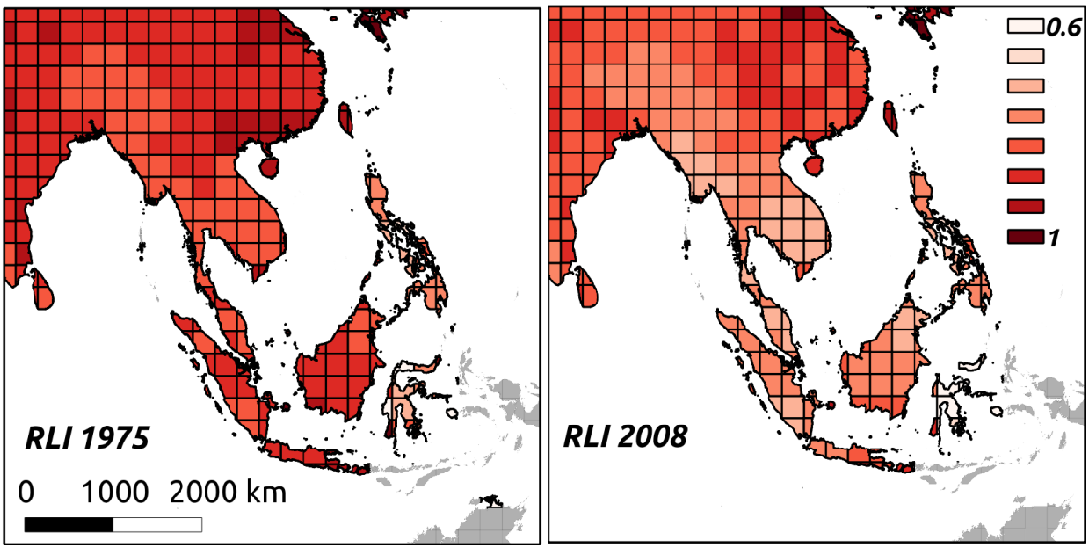

Fig. 3.S7 Differences in overall conservation status of Southeast Asia between 1975 and 2008. The Red List Index values are reported in red scale. Colour and bar scale are the same for both maps.

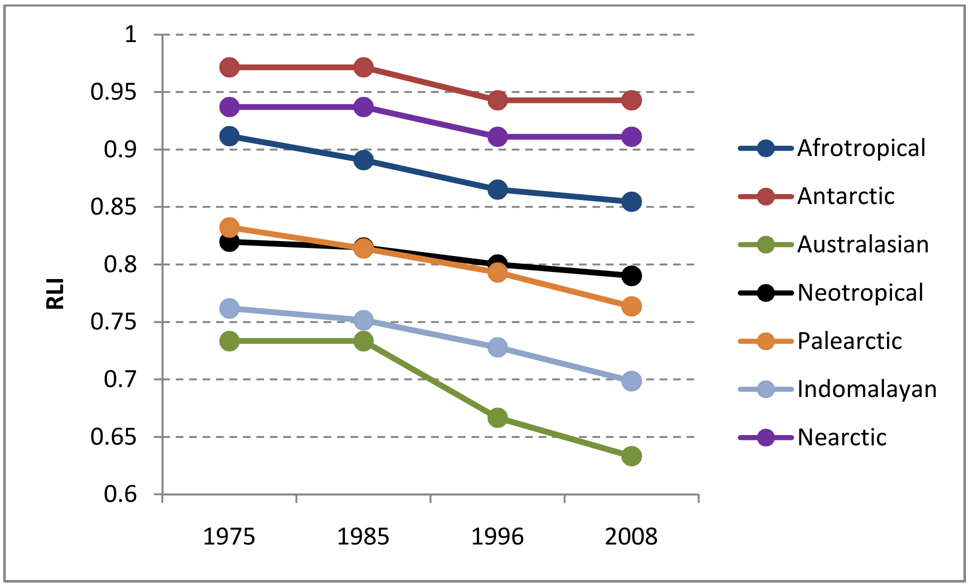

Fig. 3.S8 Red List Index trend divided by biogeographic realm. NB Oceanian realm was excluded from this analysis since it includes no ungulates and only one carnivore species (the Hawaiian Monk Seal *Monachus schauinslandi*).

8.2.3. Supporting Tables for Chapter 3

Table 3.S1 Examples of retrospective assessments for the 1970s.

<table>
<tr><th>Order</th><th>Family</th><th>Genus</th><th>Species</th><th>RL 2008</th><th>RL 1996</th><th>Assessment type '70s</th><th>Best guess category '70s</th><th>Alternative categories '70s</th><th>RL criteria '70s</th><th>Confidence '70s</th></tr>
<tr><td>CARNIVORA</td><td>FELIDAE</td><td><i>Acinonyx</i></td><td><i>jubatus</i></td><td>VU</td><td>VU</td><td>Assessed</td><td>EN</td><td>VU</td><td>A2cd</td><td>50/50</td></tr>
<tr><td>Justification '70s</td><td colspan="10">Total population size in early '70s is probably around 15000 individuals and almost certainly within 8000-25000. This figure is roughly half of the population present in 1960 and the rate of decline has not diminished. A 50% population reduction in less than 3 generations (Gen. Time is 6 years) has occurred for this species.</td></tr>
<tr><td>Order</td><td>Family</td><td>Genus</td><td>Species</td><td>RL 2008</td><td>RL 1996</td><td>Assessment type '70s</td><td>Best guess category '70s</td><td>Alternative categories '70s</td><td>RL criteria '70s</td><td>Confidence '70s</td></tr>
<tr><td>CETARTIODACTYLA</td><td>BOVIDAE</td><td><i>Addax</i></td><td><i>nasomaculatus</i></td><td>CR</td><td>CR</td><td>Assessed</td><td>VU</td><td>EN</td><td>C1</td><td>75/25</td></tr>
<tr><td>Justification '70s</td><td colspan="10">Total population very likely to be below 10,000 mature individuals and less likely to be smaller than 2,500. Newby (Mammals of Africa) cites one record of 11,000 animals at one site in 1960 and says they were present in fair numbers up to early 1970s, and estimates for 2 countries are missing. Hunting has been a negative factor since 1930s (Newby in press) and is still common, a decline bigger than 10% (possibly even >20%) has likely occurred in the past 15 years (Gen. length is 7 years).</td></tr>
</table>

Examples of retrospective assessments for the 1970s. For each species, the Red List status of 1996 and 2008 is reported together with the retrospectively assigned status for the 1970s (including categories, criteria and level of confidence in case of alternative categories assignation). Note: Only two example species are reported here, the rest of the table has been omitted due to space limitations; refer to electronic material for a complete version.

Table 3.S2 Net difference in RLI values for countries

Name	RLI'75	RLI'85	RLI'96	RLI'08	deltaRLI ('08-'75)	N. species (endemics)
Faroe Islands	1.00	1.00	0.87	0.87	-0.13	3(0)
Saint Pierre And Miquelon	0.96	0.96	0.84	0.84	-0.12	5(0)
Indonesia	0.78	0.76	0.73	0.67	-0.11	59(10)
Malaysia	0.82	0.80	0.78	0.72	-0.10	50(0)
Australia	0.90	0.90	0.83	0.80	-0.10	9(1)
New Zealand	0.90	0.90	0.80	0.80	-0.10	3(0)
Bahrain	1.00	1.00	1.00	0.90	-0.10	2(0)
Philippines	0.72	0.71	0.68	0.62	-0.09	17(10)
Brunei Darussalam	0.88	0.87	0.85	0.79	-0.08	27(0)
Tunisia	0.86	0.82	0.80	0.78	-0.08	19(0)
Thailand	0.82	0.80	0.78	0.74	-0.08	58(0)
Cambodia	0.80	0.79	0.76	0.72	-0.08	44(0)
Mongolia	0.90	0.89	0.87	0.83	-0.08	37(1)
Lao PDR	0.82	0.81	0.79	0.75	-0.08	49(0)
Madagascar	0.78	0.78	0.70	0.70	-0.08	8(8)
Svalbard And Jan Mayen	0.98	0.98	0.90	0.90	-0.08	9(0)
Viet Nam	0.80	0.79	0.77	0.73	-0.07	60(0)
Kazakhstan	0.91	0.91	0.89	0.84	-0.07	37(0)
Algeria	0.84	0.83	0.81	0.78	-0.07	24(0)
Uzbekistan	0.92	0.91	0.90	0.85	-0.07	33(0)
Singapore	0.92	0.92	0.90	0.85	-0.07	12(0)
Iceland	1.00	1.00	0.93	0.93	-0.07	6(0)
India	0.82	0.80	0.78	0.76	-0.07	101(4)
Bangladesh	0.88	0.87	0.85	0.81	-0.06	41(0)
Russian Federation	0.90	0.90	0.87	0.84	-0.06	68(2)
China	0.83	0.81	0.80	0.77	-0.06	108(6)
Morocco	0.86	0.86	0.83	0.80	-0.06	25(0)
Libyan Arab	0.89	0.86	0.85	0.83	-0.06	13(0)

Name	RLI'75	RLI'85	RLI'96	RLI'08	deltaRLI ('08-'75)	N. species (endemics)
Jamahiriya						
Hong Kong	0.95	0.95	0.92	0.89	-0.06	20(0)
Liberia	0.96	0.95	0.91	0.90	-0.06	39(0)
Bhutan	0.83	0.82	0.80	0.78	-0.06	61(0)
Western Sahara	0.87	0.86	0.84	0.81	-0.06	14(0)
Myanmar	0.81	0.81	0.79	0.76	-0.06	73(0)
Nepal	0.85	0.83	0.81	0.79	-0.05	64(0)
Sierra Leone	0.97	0.96	0.92	0.91	-0.05	42(0)
Guinea	0.96	0.94	0.92	0.91	-0.05	57(0)
Korea DPR	0.89	0.88	0.87	0.84	-0.05	25(0)
CÃ´te D'Ivoire	0.96	0.95	0.92	0.91	-0.05	56(0)
Somalia	0.93	0.91	0.89	0.88	-0.05	57(0)
Ethiopia	0.91	0.89	0.87	0.86	-0.05	79(3)
Japan	0.95	0.95	0.92	0.90	-0.04	25(4)
Kenya	0.95	0.93	0.92	0.91	-0.04	83(1)
Turkmenistan	0.91	0.91	0.91	0.87	-0.04	28(0)
Pakistan	0.89	0.88	0.88	0.85	-0.04	52(0)
Niger	0.93	0.92	0.90	0.89	-0.04	49(0)
Senegal	0.96	0.93	0.93	0.91	-0.04	49(0)
Korea, Republic of	0.89	0.89	0.87	0.85	-0.04	21(0)
Chad	0.93	0.91	0.90	0.89	-0.04	56(0)
Iran	0.89	0.89	0.89	0.85	-0.04	37(0)
Tajikistan	0.91	0.91	0.89	0.87	-0.04	32(0)
Mauritania	0.94	0.93	0.91	0.90	-0.04	32(0)
Afghanistan	0.90	0.90	0.89	0.86	-0.04	38(0)
Sri Lanka	0.85	0.84	0.83	0.81	-0.04	22(3)
Djibouti	0.96	0.96	0.94	0.93	-0.04	28(0)
Ecuador	0.85	0.85	0.83	0.82	-0.04	37(2)
Sudan	0.93	0.93	0.91	0.90	-0.03	87(1)
Tanzania, United Republic of	0.95	0.94	0.93	0.92	-0.03	82(1)
Paraguay	0.87	0.85	0.86	0.84	-0.03	28(0)
Iraq	0.93	0.94	0.93	0.90	-0.03	25(0)
Armenia	0.91	0.92	0.91	0.88	-0.03	26(0)
Bolivia	0.88	0.87	0.86	0.85	-0.03	43(0)
Mali	0.93	0.93	0.92	0.91	-0.03	49(0)
United Kingdom	0.99	0.99	0.96	0.96	-0.03	14(0)
Kyrgyzstan	0.91	0.92	0.90	0.88	-0.03	22(0)
Equatorial Guinea	0.96	0.96	0.95	0.93	-0.03	31(0)
Ghana	0.98	0.97	0.96	0.95	-0.03	54(0)
Argentina	0.83	0.82	0.82	0.81	-0.03	44(0)
Yemen	0.91	0.93	0.93	0.89	-0.03	16(0)

Name	RLI'75	RLI'85	RLI'96	RLI'08	deltaRLI ('08-'75)	N. species (endemics)
Spain	0.89	0.89	0.89	0.87	-0.02	24(0)
United States	0.92	0.92	0.90	0.90	-0.02	62(4)
Gabon	0.96	0.96	0.95	0.94	-0.02	43(0)
Eritrea	0.94	0.94	0.93	0.92	-0.02	43(0)
Burkina Faso	0.95	0.94	0.93	0.93	-0.02	46(0)
Canada	0.97	0.97	0.95	0.94	-0.02	48(0)
Egypt	0.86	0.87	0.86	0.84	-0.02	28(1)
Turkey	0.91	0.92	0.91	0.89	-0.02	28(0)
Congo	0.97	0.96	0.96	0.95	-0.02	51(0)
Saudi Arabia	0.86	0.86	0.89	0.84	-0.02	20(0)
Guinea-Bissau	0.98	0.98	0.97	0.96	-0.02	41(0)
Jordan	0.88	0.88	0.90	0.86	-0.02	21(0)
Central African Republic	0.96	0.95	0.95	0.94	-0.02	66(0)
Azerbaijan	0.92	0.93	0.93	0.90	-0.02	22(0)
Syrian Arab Republic	0.94	0.95	0.95	0.92	-0.02	22(0)
Greenland	0.95	0.96	0.93	0.93	-0.02	12(0)
Malawi	0.96	0.95	0.95	0.94	-0.02	55(0)
Congo, The Democratic Republic Of The	0.98	0.97	0.96	0.96	-0.02	81(1)
Cameroon	0.96	0.95	0.94	0.94	-0.02	67(0)
Togo	0.97	0.97	0.96	0.95	-0.02	47(0)
Colombia	0.86	0.86	0.85	0.84	-0.02	40(0)
Burundi	0.96	0.96	0.95	0.95	-0.02	48(0)
Uganda	0.97	0.96	0.96	0.95	-0.02	73(0)
Gambia	1.00	1.00	0.99	0.98	-0.02	36(0)
Kuwait	0.95	0.97	0.97	0.93	-0.02	12(0)
Benin	0.95	0.95	0.94	0.94	-0.02	50(0)
Rwanda	0.97	0.96	0.96	0.95	-0.01	54(0)
Angola	0.96	0.95	0.95	0.94	-0.01	80(0)
	0.87	0.87	0.88	0.86	-0.01	32(0)
Zambia	0.96	0.95	0.95	0.94	-0.01	64(0)
Taiwan, Province Of China	0.91	0.91	0.91	0.90	-0.01	16(1)
Nigeria	0.94	0.94	0.94	0.93	-0.01	65(0)
Botswana	0.95	0.94	0.94	0.94	-0.01	65(0)
Mozambique	0.95	0.94	0.94	0.94	-0.01	65(0)
Zimbabwe	0.95	0.94	0.94	0.94	-0.01	65(0)
Oman	0.86	0.87	0.88	0.84	-0.01	18(0)
Peru	0.84	0.84	0.83	0.83	-0.01	44(0)

Name	**RLI'75**	**RLI'85**	**RLI'96**	**RLI'08**	**deltaRLI ('08-'75)**	**N. species (endemics)**
Georgia	0.91	0.92	0.93	0.90	-0.01	23(0)
Norway	0.97	0.97	0.96	0.96	-0.01	24(0)
Namibia	0.94	0.94	0.94	0.93	-0.01	70(0)
Venezuela, Bolivarian Republic of	0.89	0.89	0.90	0.88	-0.01	29(0)
Swaziland	0.97	0.97	0.96	0.96	-0.01	56(0)
Mexico	0.92	0.93	0.93	0.91	0.00	44(2)
South Africa	0.94	0.94	0.94	0.94	0.00	76(1)
United Arab Emirates	0.85	0.87	0.88	0.85	0.00	15(0)
French Guiana	0.88	0.88	0.91	0.88	0.00	21(0)
Portugal	0.89	0.90	0.90	0.89	0.00	16(0)
Suriname	0.89	0.89	0.91	0.89	0.00	22(0)
Guyana	0.90	0.90	0.91	0.90	0.00	23(0)
France	0.92	0.93	0.94	0.92	0.00	22(0)
Qatar	0.93	0.95	0.95	0.93	0.00	8(0)
Lesotho	0.95	0.95	0.96	0.95	0.00	39(0)
Monaco	0.97	0.97	0.97	0.97	0.00	7(0)
Ireland	0.98	0.98	0.98	0.98	0.00	8(0)
Gibraltar	0.98	0.98	0.98	0.98	0.00	9(0)
San Marino	0.98	0.98	0.98	0.98	0.00	10(0)
Denmark	0.98	0.98	0.98	0.98	0.00	11(0)
Liechtenstein	0.98	0.98	0.98	0.98	0.00	12(0)
Luxembourg	0.98	0.98	0.98	0.98	0.00	12(0)
Netherlands	0.98	0.98	0.98	0.98	0.00	13(0)
Belgium	0.99	0.99	0.99	0.99	0.00	14(0)
Austria	0.99	0.99	0.99	0.99	0.00	19(0)
Antarctica	1.00	1.00	1.00	1.00	0.00	6(0)
Bouvet Island	1.00	1.00	1.00	1.00	0.00	4(0)
French Southern Territories	1.00	1.00	1.00	1.00	0.00	4(0)
Heard and McDonald Islands	1.00	1.00	1.00	1.00	0.00	4(0)
Saint Helena, Ascension and Tristan Da Cunha	1.00	1.00	1.00	1.00	0.00	2(0)
South Georgia & South Sandwich Islands	1.00	1.00	1.00	1.00	0.00	6(0)
Trinidad & Tobago	1.00	1.00	1.00	1.00	0.00	3(0)
Brazil	0.89	0.89	0.91	0.89	0.01	42(2)

Name	RLI'75	RLI'85	RLI'96	RLI'08	deltaRLI ('08-'75)	N. species (endemics)
Panama	0.92	0.93	0.93	0.93	0.01	27(0)
Chile	0.77	0.78	0.78	0.78	0.01	25(2)
Ukraine	0.90	0.91	0.91	0.91	0.01	21(0)
Romania	0.92	0.93	0.94	0.93	0.01	21(0)
Greece	0.92	0.93	0.93	0.93	0.01	20(0)
Belarus	0.92	0.93	0.93	0.93	0.01	19(0)
Estonia	0.93	0.94	0.96	0.94	0.01	18(0)
Latvia	0.93	0.94	0.96	0.94	0.01	18(0)
Albania	0.91	0.92	0.92	0.92	0.01	18(0)
Costa Rica	0.91	0.92	0.94	0.93	0.02	27(0)
Guatemala	0.91	0.92	0.94	0.93	0.02	25(0)
Nicaragua	0.93	0.94	0.96	0.95	0.02	25(0)
Honduras	0.93	0.94	0.95	0.95	0.02	24(0)
Belize	0.90	0.91	0.93	0.92	0.02	22(0)
Yugoslavia	0.95	0.96	0.97	0.97	0.02	20(0)
Poland	0.95	0.96	0.96	0.97	0.02	20(0)
Italy	0.97	0.98	0.99	0.99	0.02	20(0)
Croatia	0.97	0.98	0.99	0.99	0.02	19(0)
Germany	0.97	0.98	0.99	0.99	0.02	19(0)
Sweden	0.97	0.98	0.99	0.99	0.02	19(0)
El Salvador	0.93	0.94	0.96	0.96	0.02	20(0)
Bulgaria	0.94	0.96	0.97	0.97	0.02	18(0)
Macedonia	0.94	0.96	0.97	0.97	0.02	18(0)
Bosnia	0.97	0.98	0.99	0.99	0.02	18(0)
Finland	0.97	0.98	0.99	0.99	0.02	18(0)
Slovakia	0.97	0.98	0.99	0.99	0.02	18(0)
Czech Republic	0.96	0.98	0.99	0.99	0.02	17(0)
Hungary	0.96	0.98	0.99	0.99	0.02	17(0)
Slovenia	0.96	0.98	0.99	0.99	0.02	17(0)
Switzerland	0.96	0.98	0.99	0.99	0.02	17(0)
Lithuania	0.96	0.98	0.99	0.99	0.03	16(0)
Lebanon	0.91	0.92	0.93	0.93	0.03	15(0)
Moldova, Republic of	0.96	0.97	0.99	0.99	0.03	14(0)
Uruguay	0.89	0.90	0.92	0.92	0.03	19(0)
Falkland Islands (Malvinas)	0.70	0.90	1.00	1.00	0.30	3(0)

Net difference in RLI values for countries. The total number of species and the number of endemic species are also reported. Note: countries with less than 10 species of carnivores or ungulates are shaded with gray.

8.3. Appendices to Chapter 4

8.3.1. APPENDIX 4.S1 - Statistical Modelling and Multiple Imputation

The calculation of intrinsic extinction risk for species was based on previous works by Cardillo et al. (2004, 2005, 2006, 2008). Here we only considered the biological predictors of extinction risk (leaving out the extrinsic and the human-related variables) in order to define the species intrinsic extinction risk. We derived the biological traits of species from the freely available PanTHERIA database (Jones et al. 2009). All biological variables were considered to be possible predictors of extinction risk. Though PanTHERIA is one of the most comprehensive databases of mammal life-history traits (with more than 30 different biological variables collected for over 5400 species), it lacks information on one or more of the variables for most taxa. Moreover, we did not know *a priori* which variables would have been considered to be significant predictors of extinction risk when considering all species instead of a subset as in previous works (Cardillo et al. 2006, 2008). In order to overcome the problem of missing biological data for species and to avoid the risk of excluding predictors of potential interest in our analysis, we compiled the PanTHERIA database using a Multiple Imputation procedure (MI; Rubin 1987); MI has already been used as a supporting tool in extinction risk modeling (Fisher et al. 2003), and some examples of its application have recently been published (Fisher & Blomberg 2011; Murray et al. 2011).

In the absence of a robust approach that explicitly accounts for phylogeny when imputing a taxonomically related dataset, we chose to apply the ordinary predictive mean matching method (pmm; Little 1988; van Buuren & Groothuis-Oudshoorn 2010) within each separate mammalian order. Through the application of the pmm method, it is expected that the donor taxon is a close relative of the recipient taxon (Fisher et al. 2003). By imputing each order separately, we reduced the probability of having a donor out of the pool of the recipient's close relatives. For each order, we created 10 imputed datasets (running 1,000 iterations for each one) according to the following rules: i) in order to be used as a predictor in the imputation of variable a, the variable b must contain at least 10% of usable information (<90% missingness), and ii) it must show a correlation value with a that is finite (sd $\neq 0$) and bigger than $|0.01|$.

All continuous variables were log-transformed for the purpose of the MI and then transformed back before running the extinction risk models. Log-transformation allowed us to avoid having too high of a collinearity level among predictors (van Buuren & Groothuis-Oudshoorn 2010); moreover, it has previously been shown that some of the biological traits are more related on the log-linear scale than on the linear scale (e.g. home range and body mass; McNab 1963). We defined the intrinsic extinction risk prediction models for each mammal order using the IUCN Red List category (IUCN 2010) as the dependent variable, see also Cardillo et al. (2005, 2008) for details. We randomly chose 5 out of 10 imputed datasets for identifying the significant extinction risk predictors. For each of the five sampled datasets, we found a minimum adequate model (MAM) of extinction risk using multiple regression through the origin. We first fitted a full model using all predictors, then simplified to a model in which the marginal contribution of all predictors to variance in extinction risk was significant ($p \leq 0.05$). To account for non-independence of species arising from shared ancestry we fitted models using phylogenetically independent contrasts, calculated with the "crunch" algorithm in the R package CAIC (Purvis & Rambaut 1995). Predictors that were significant in a majority of the five MAMs were retained; we considered this to be the set of predictors that have a significant, independent influence on extinction risk. We chose to discard from the final model definition all the variables with a "missingness" level (i.e. proportion of missing-data cases) bigger than 80%, in order to avoid having unpredictable effects over the parameters estimates.

Once the final model structure was defined for each separate order (using the 5 selected imputed datasets), we applied it to the remaining 5 datasets to estimate the parameters values (betas and intercepts). Because the crunch algorithm does not estimate an intercept, we could not derive fitted values of extinction risk from these models. We therefore fitted the majority set of predictors to the data using non-phylogenetic, ordinary least-squares regression, and derived fitted values from this model. We then used the 5 estimates of the parameters in order to calculate the rate of missing information due to the missing data imputation (Rubin 1987; see Table 4.S1). The final step was calculating the fitted extinction risk values using the parameters estimates and the imputed variables values, and then aggregating them into a final fitted extinction risk value (the average from the 5 extinction risk values; Table 4.S2). The aggregated fitted extinction risk correspond to the species' intrinsic extinction risk value (ER_{INTR}) that was used to calculate the species' ERO value according to *[1]* (see Materials and Methods for details). Multiple Imputation and extinction risk analyses were carried out in R (R Development Core Team 2009) using packages MICE (van Buuren & Groothuis-Oudshoorn 2010) and CAIC (Purvis & Rambaut 1995).

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8.3.2. APPENDIX 4.S2 - Formulation of the generalized benefit function used for the prioritization analysis

We used the ERO values together with the species' representation targets (*Tj*) to customize a generalized benefit function (Moilanen & Kujala 2008) that relates the species' value during the progression of the prioritization analysis (*Vj*) to their remaining distribution range (*Rj*). We used the ‘ramp with linear over-representation’ function (Moilanen & Kujala 2008) as a template to construct our benefit function. For each j species: we assigned 100% of its ERO value to a species at the beginning of analysis ($Rj = 1$); once the species' remaining distribution range decreased and reached the target area ($Rj = Tj$), species value was 90% of the original ERO value ($Vj = 0.9 \cdot EROj$). When species' remaining distribution range went below the target area ($Rj < Tj$), species' value decreased and finally reached 0 ($Vj \rightarrow 0$). Our benefit function assumes the form of a linear function with 2 slopes (with the target as a split point) defined according to Moilanen & Kujala (2008):

$$Vj(Rj) = \begin{cases} (0.9 \cdot EROj) \cdot \left(\dfrac{Rj}{Tj} \right) & if \quad Rj \le Tj \\ (0.9 \cdot EROj) + \left((0.1 \cdot EROj) \cdot \left(\dfrac{Rj - Tj}{1 - Tj} \right) \right) & if \quad Rj > Tj \end{cases} , \qquad [S1].$$

SUPPORTING REFERENCES

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8.3.3 Supporting Tables for Chapter 4

Table 4.S1. Minimum adequate models of intrinsic extinction risk for terrestrial mammal orders.

Predictors	Afrotheria	Carnivora	Chiroptera	Lagomorpha	Marsupials	Minor clades	Primates	Rodentia	Ungulates
Intercept	2.35 (0.62) <0.01	5.37 (0.48) 0.01	3.13 (0.18) <0.01	3.84 (0.58) <0.01	5.72 (0.68) 0.04	3.38 (0.25) 0.04	-4.72 (2.63) 0.10	3.91 (0.16) 0.02	5.79 (0.50) 0.05
Adult Body Mass	0.21 (0.05) <0.01		0.10 (0.02) <0.01		0.05 (0.02) <0.01	0.16 (0.02) <0.01		0.09 (0.01) <0.01	
Diet Breadth		-0.06 (0.03) <0.01	0.14 (0.04) <0.01	0.29 (0.13) <0.01					
Gestation Length					-0.44 (0.12) <0.01		2.15 (0.26) 0.01		
Habitat Breadth	-0.35 (0.24) <0.01					0.15 (0.07) 0.01		-0.21 (0.03) <0.01	
Home Range Size				-0.04 (0.06) <0.01					0.14 (0.03) <0.01
Population Density				-0.10 (0.06) <0.01					
Population Group Size									-0.12 (0.03) <0.01
Trophic Level			0.15 (0.03) <0.01					0.10 (0.03) <0.01	
Weaning Age	-0.65 (0.15) <0.01								
Geographic Range Size		-0.30 (0.03) <0.01	-0.26 (0.01) <0.01	-0.28 (0.03) <0.01	-0.34 (0.02) <0.01	-0.30 (0.01) <0.01	-0.39 (0.02) <0.01	-0.30 (0.01) <0.01	-0.32 (0.03) <0.01

Minimum adequate models of intrinsic extinction risk for terrestrial mammal orders. Only biological predictors of extinction risk were considered; a variable was included in the model if it was significant in 3 or more of the 5 imputed datasets. The final estimates refer to the average value calculated across the 5 datasets. For each variable/order, the parameter estimate and parameter total variance (in parentheses) are in the first row while the rate of missing information (gamma) is in the second row. Parameter estimates, total variance and missing information rate were calculated following Rubin (1987). Only significant predictors are reported here.

Table 4.S2 List of terrestrial mammal species used for analysis and their extinction risk values (as defined in the text).

Species rank	**Order**	**Family**	**Genus**	**Species**	**ER_{IUCN}**	**ER_{INTR}**	**ERO**
1	DIPROTODONTIA	PHALANGERIDAE	Spilocuscus	rufoniger	4	0.920	3.264
2	CHIROPTERA	PTEROPODIDAE	Mirimiri	acrodonta	4	0.940	3.248
3	CARNIVORA	PHOCIDAE	Monachus	monachus	4	1.060	3.152
4	PERISSODACTYLA	RHINOCEROTIDAE	Diceros	bicornis	4	1.110	3.112
5	CETARTIODACTYLA	BOVIDAE	Saiga	tatarica	4	1.125	3.100
6	RODENTIA	DASYPROCTIDAE	Dasyprocta	mexicana	4	1.160	3.072
7	PRIMATES	PITHECIIDAE	Chiropotes	satanas	4	1.220	3.024
8	CARNIVORA	PHOCIDAE	Monachus	schauinslandi	4	1.230	3.016
9	RODENTIA	SCIURIDAE	Marmota	sibirica	3	0.000	3.000
9	CARNIVORA	OTARIIDAE	Eumetopias	jubatus	3	0.000	3.000
11	CHIROPTERA	VESPERTILIONIDAE	Myotis	sodalis	3	0.180	2.892
12	MONOTREMATA	TACHYGLOSSIDAE	Zaglossus	bartoni	4	1.400	2.880
13	RODENTIA	NESOMYIDAE	Mystromys	albicaudatus	3	0.250	2.850
14	RODENTIA	CHINCHILLIDAE	Chinchilla	chinchilla	4	1.500	2.800
15	RODENTIA	MURIDAE	Zyzomys	pedunculatus	4	1.520	2.784
16	PRIMATES	ATELIDAE	Ateles	belzebuth	3	0.370	2.778
17	RODENTIA	MURIDAE	Solomys	ponceleti	4	1.530	2.776
18	MONOTREMATA	TACHYGLOSSIDAE	Zaglossus	bruijnii	4	1.560	2.752
18	CETARTIODACTYLA	BOVIDAE	Nanger	dama	4	1.560	2.752
<The rest of the table has been omitted due to space limitations, refer to electronic material for a complete version>							

8.3.4. Supporting Analysis for Chapter 4

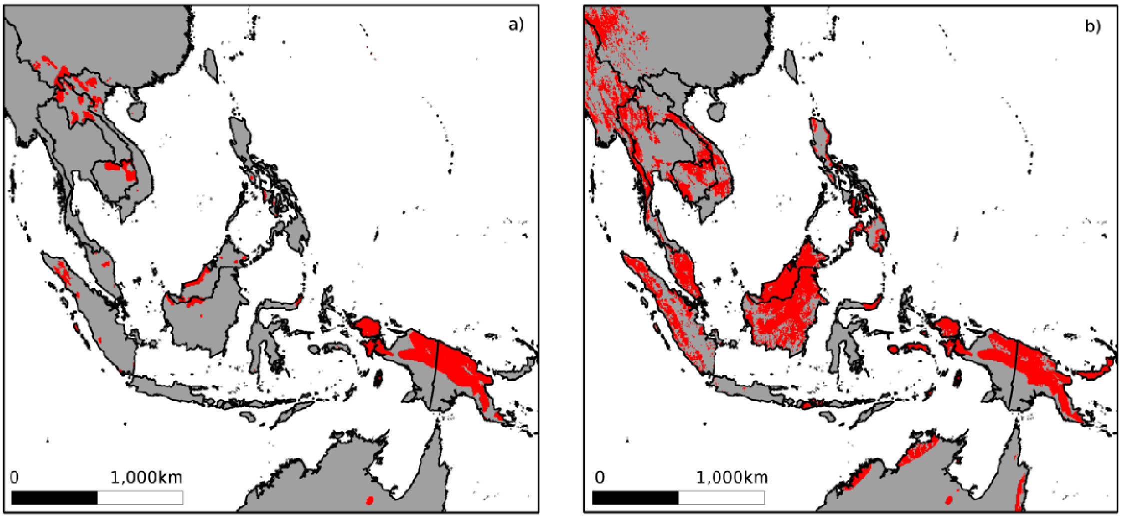

FIGURE 4.S1 Difference in priority area definition in south-east Asia and northern Australia, according to IUCN's critically endangered (CR) species distribution (a) and top-ranked ERO species distribution (b).

This maps represent a zoomed portion of the maps reported in Fig. 3 in the paper (refer to Fig. 3's legend for details on map definition). Some CR species, such as the Echidnas (*Zaglossus bartoni* and *Z. brujinii*) in Papua New Guinea, are included in both maps, due to their low intrinsic extinction risk level and their current high risk of extinction (i.e. high ERO value). Endangered species with a low intrinsic risk of extinction (i.e. medium-high ERO value) such as the Northern Quoll (*Dasyurus hallucatus*) in northern Australia and the Dhole (*Cuon aplinus*) in Indonesia, Thailand and Myanmar are excluded from CR priority areas (a) but included in ERO priority area (b). CR species with a high current risk of extinction and a high intrinsic risk of extinction (i.e. medium-low ERO value), such as the Tonkin Snub-nosed Monkey (*Rhinopithecus avunculus*) in Viet Nam and the Sumatran Rhinoceros (*Dicerorhinus sumatrensis*) in Malaysia and Indonesia, are included in CR priority map (a) but excluded from ERO priority map (b).

Overall using the ERO metric instead of CR status to detect priority areas for species' conservation, results in a bigger proportion of priority areas found in south-east Asia and a smaller proportion of areas found in Africa (see Fig. 3 for a global comparison). This is not surprising: considering that the extrinsic components (i.e. the human related threats) of extinction risk are currently playing a relatively bigger role in south-east Asia than in Africa, removing them from south-east Asia would result into a bigger conservation return in terms of overall species' extinction risk reduction.

8.4. Appendices to Chapter 5

8.4.1. APPENDIX 5.A

Summary of all the sensitivity analyses performed (Table 5.A1).

Quantification method tested	Description	Values tested	Relevant analytical step	Related result
HF percentile	Percentiles of the distribution of HF within species EOO	10th, 20th, 30th, ..., 90th HF percentiles	**Vb**. HF prevalence vs quantiles of HF distribution	Fig. 5.4
HF prevalence	Proportion of EOO in areas with HF bigger than a given threshold	HF threshold = 10, 20, 30, ..., 90	**Va.**, HF prevalence vs mean HF value, **Vb.**, HF prevalence vs quantiles of HF distribution **VIb.** Detection of threatened species	APPENDIX 5.B + APPENDIX 5.C + Fig. 5.4
Proportion of top-ranked species (priority lists)	Proportions of top-ranked species selected to define priority lists.	10%, 20%, 30%, ..., 90% of species	**VIc.** Sensitivity in the detection of threatened species	APPENDIX 5.D

Table 5.A1 Summary of the sensitivity analyses performed, with reference to the analytical settings described in the methods (section 5.3.2) and summarised in Table 5.1. A description of the threat quantification method, the sensitivity settings, and result obtained is provided. EOO, extent of occurrence; HF, human footprint.

8.4.2. APPENDIX 5.B

Sensitivity testing on the human footprint thresholds used to compare different quantification methods (Fig. 5.B1).

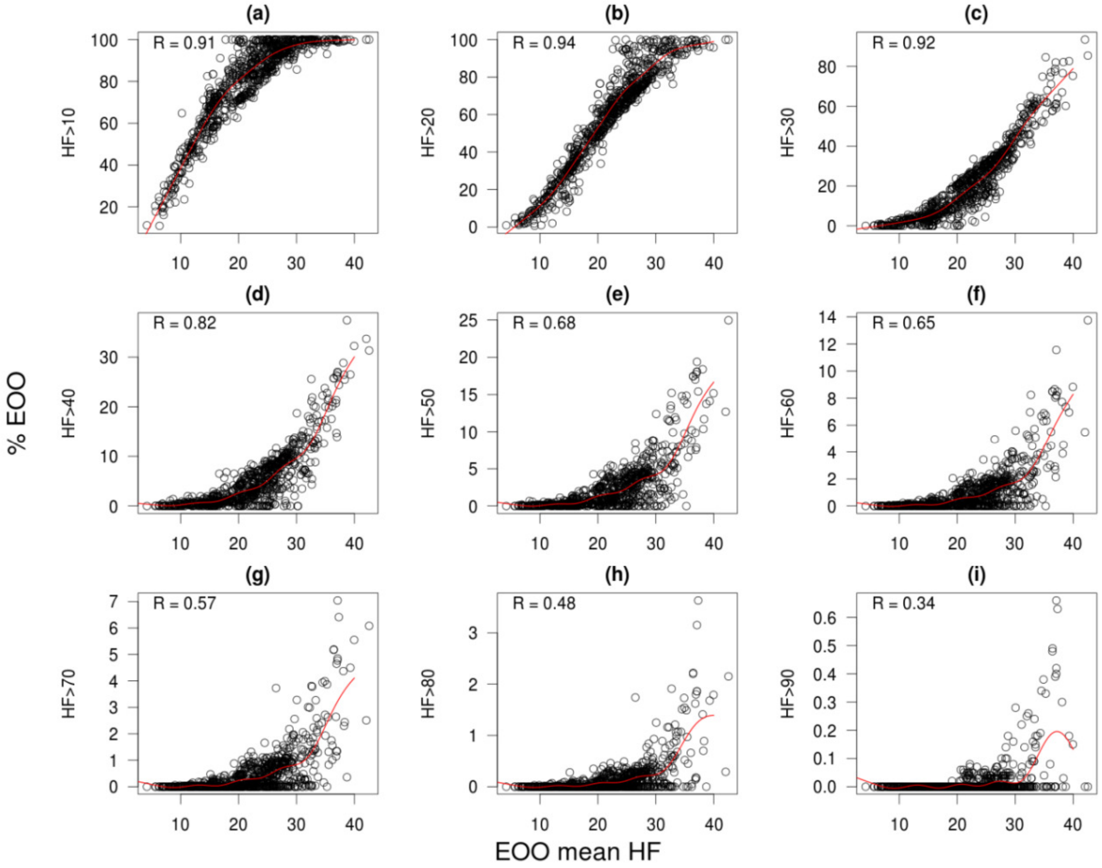

Fig. 5.B1 Comparison of the mean Human Footprint (HF) value within species' extent of occurrence (EOO; x axis) and HF prevalence within EOO (i.e., % EOO overlapping with areas of HF larger that a given threshold, y axis). A different HF threshold (from 10 to 90 with a step of 10) is reported in plots (a) to (i). Each dot represents a species; a fitted GAM curve (i.e. best performing model from AIC) is shown in red. NB In plot (b) a sigmoid curve was indistinguishable in support ($\Delta AIC < 2$) in comparison to the GAM shown here for consistency with the other plots.

8.4.3 APPENDIX 5.C

Sensitivity testing on the human footprint thresholds used detect threatened species (Fig. 5.C1).

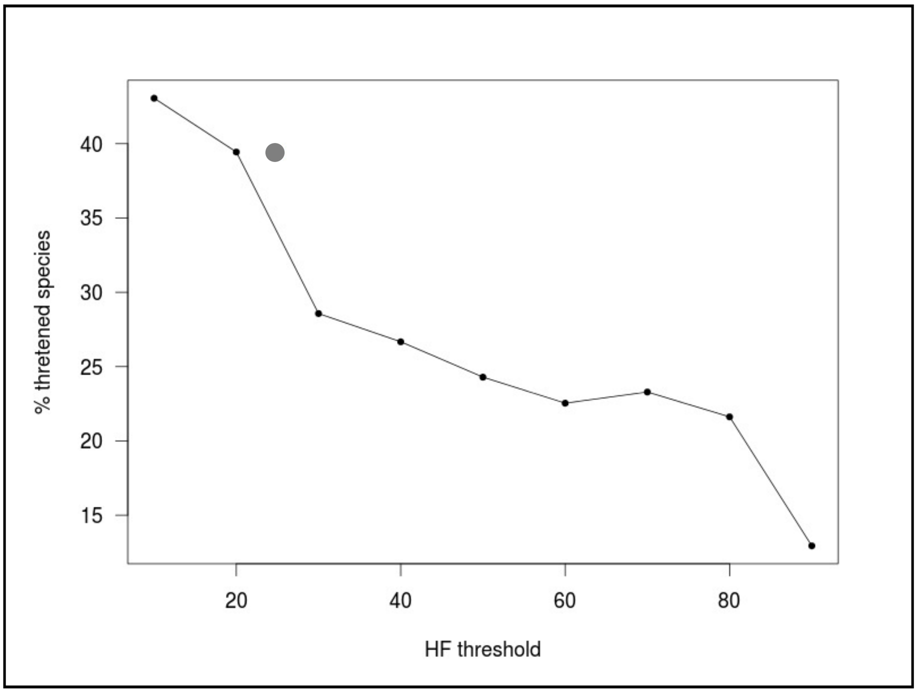

Fig. 5.C1 Proportion of IUCN threatened species included in a series of top 10% priority lists according to various thresholds of threat prevalence quantification. The proportions of species' extent of occurrence (EOO) overlapping with areas of Human Footprint (HF) larger than a given threshold (from 10 to 90) were calculated and used to rank species in descending order of threat level. For each list, the top 10% ranked species were extracted and the proportion of IUCN threatened species among them calculated. The grey dot represents the value corresponding to a threshold of HF = 25 (i.e., the same threshold reported in Table 5.4 and Fig. 5.3).

8.4.4. APPENDIX 5.D

Sensitivity analysis on the percentage threshold used to select priority species lists (Fig. 5.D1).

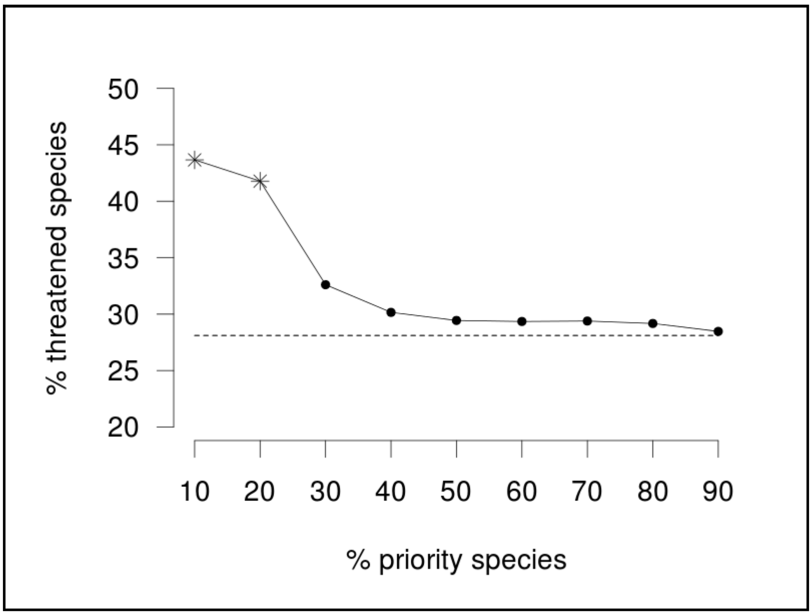

Fig. 5.D1 Sensitivity analysis on the threshold used to select priority species lists. Species are ranked according to their level of threat prevalence (i.e., proportion of extent of occurrence overlapping with areas where human footprint>10). The percentage of IUCN threatened species selected by setting different rank thresholds (from 10% to 90% top-ranked species selected) was compared with the percentage of threatened species in the full dataset (dashed line). Asterisks indicate that the proportion of threatened species in priority list is significantly higher than the background proportion (binomial test for proportions), while dots indicate a non-significant difference.