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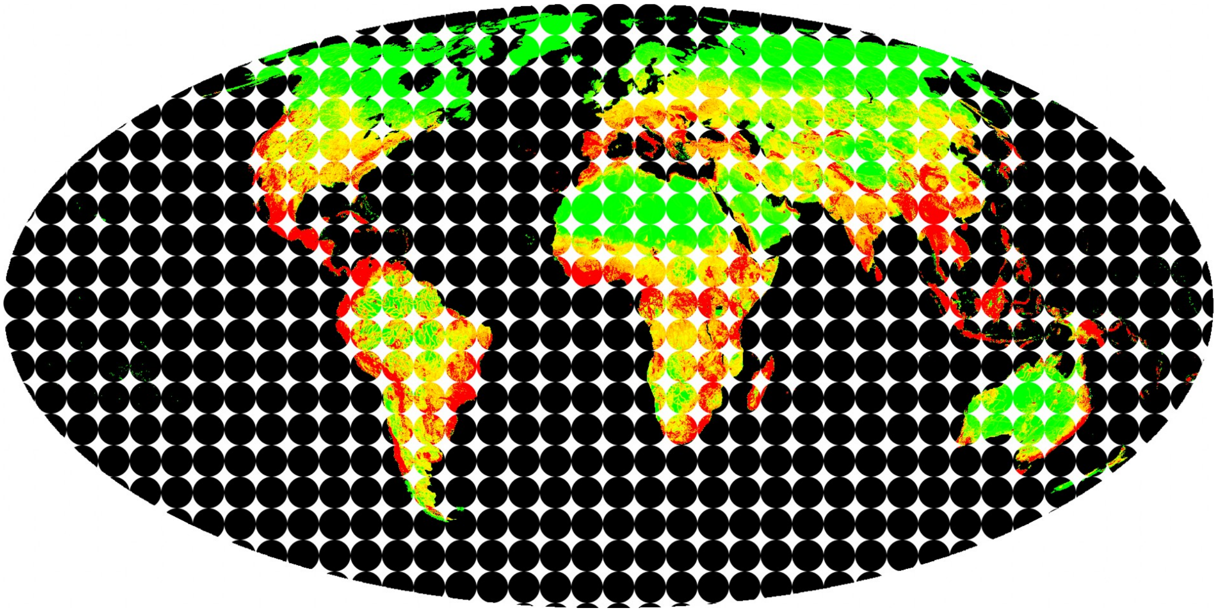
THE USE OF HABITAT SUITABILITY MODELS IN CONSERVATION PLANNING



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1 – INTRODUCTION

1.1 – Our World: Status and Challenges

It has now been well established that, as human beings, we have had a tremendous impact on our surroundings: the extent to which we influenced our environment and biodiversity in general has led many authors to start adopting the term “anthropocene” (Braje & Erlandson 2013). Modern extinction rates highly surpass historical figures (Diamond *et al*. 1989) and the number of species assigned by the International Union for the Conservation of Nature (IUCN) to higher Red List extinction threat categories increases every year (IUCN 2013). Currently, of the 71,576 species assessed in the IUCN Red List, 30% have been assigned to one of three threatened categories. The Mammalian taxon contains some of the most charismatic and widely recognised species, undoubtedly receiving a disproportionate amount of scientific and social attention (e.g. see Smith *et al*. 2012). Along with birds, mammals are among the only major taxa to have been fully assessed in the IUCN Red List. Despite this, it is still estimated that 25% of Mammalian species are under threat of extinction (IUCN 2013).

There are undoubtedly many causes that may contribute to a specie's status. Habitat loss, habitat fragmentation and degradation, hunting and collection, persecution, competition with invasive species, climate change and pollution are only some among the major direct threats that have so far been identified for mammals. All of these, however link to a common underlying driver: the rise in human population and the increased use of resources (Secretariat of the Convention on Biological Diversity 2010). Since the advent of industrialization in the 1800's, the human population has undergone an exponential increase: we recently touched the milestone of 7,000,000 individuals, and projections indicate there may be as many as 9,500,000 living individuals by the year 2050 (United Nations Department of Economic and Social Affairs. Population Division 2014). Sustaining such a large and increasing population will inevitably lead to both an increase in land conversion from natural habitat to agricultural and farm land, as well as an increase in pressures and threats such as climate change (Millennium Ecosystem Assessment 2005). While on one side society will need to confront itself with the need for more resources to sustain the increasing human population, on the other it faces the unarguable fact that human well-being is intrinsically tied to the continuous persistence of ecological systems (United Nations 2012).

While many authors agree that protected areas alone are not sufficient to provide a comprehensive long-term protection to biodiversity, protected areas remain an essential conservation tool (PBL - Netherlands Environmental Assessment Agency 2010). They are a key element of the United Nation's approach to achieve the Millennium Development Goals, and explicit targets for global land and marine protected surfaces been ratified in the Convention on Biological Diversity's (CBD) Aichi targets. One main concern with the institution of protected areas is that these represent fixed areas with permanent investment, and have historically been planned ignoring key biological processes, such that species distributions are dynamic and respond to a wide array of external stimuli and pressures. While protected areas are expected to explicitly prevent or limit habitat conversion by anthropic means, they are permeable to external influences such as pollution and climate change.

Conservation biology attempts to mitigate, find solutions to and reverse the modern biodiversity crisis, and *ad hoc* interventions are gradually being replaced by more efficient and comprehensive systematic approaches (Margules & Pressey 2000). While on the one hand conservation biology is a crisis discipline which needs to give fast and efficient answers, systematic conservation planning ideally relies on a comprehensive knowledge and understanding of the problems and conditions faced in order to identify the most efficient strategy. Resources for conservation are insufficient to tackle all possible areas of intervention, therefore elements such as intervention costs, susceptibility, irreplaceability and success probability of success are routinely integrated in order to prioritise interventions and to maximise effectiveness (Pimm *et al*. 2001; Naidoo *et al*. 2006; Carwardine *et al*. 2008). In one of its simplest interpretations, a prioritization approach tends to assign a higher conservation priority to areas or features that are both more threatened and more irreplaceable. The concept being that sites containing species exposed to high conservation pressure but with a wide distribution (high susceptibility, low irreplaceability) and sites containing species with a restricted distribution but not exposed to any conservation pressure (low susceptibility, high irreplaceability) should rank after those that contain species both under pressure and with restricted distributions (e.g. see Carwardine *et al*. 2006). In order to develop such a conservation strategy, a comprehensive knowledge base on species distribution, biology and ecology, distribution of threats and probability of success is needed (Rondinini *et al*. 2011c).

However, our knowledge of even elementary features such as species biology or distribution is far from complete (e.g. see Boitani *et al*. 2011; Rondinini *et al*. 2011a, 2011c). While local or regional analyses and interventions may rely on a data collection phase, this approach becomes infeasible when dealing with global analyses, or if the number of species involved is exceedingly high. In these circumstances, researchers need to rely on data collections provided by third party institutions.

1.2 – Knowledge: State of the Art

1.2.1 – Species Distribution

The IUCN, through its Red List of threatened species, arguably provides the single largest information base on species distribution, and the IUCN Red List Categories and Criteria (IUCN Species Survival Commission 2012) are a de facto standard for conservation status assessment and classification. Mammals have received a full assessment through IUCN's global mammal assessment initiative in 2008 (Schipper *et al*. 2008). As of the 2013.2 assessment, 5503 mammal species have been evaluated, and maps of distribution ranges compiled by species experts are freely available for download. These maps are, however, mostly functional to IUCN's classification criteria, and largely tend to approximate a specie's Extent Of Occurrence (EOO) which is defined as the minimum convex polygon around presence point locations or areas of known presence. Efforts are made to remove large areas which are clearly unsuitable in the drawn polygons (IUCN Species Survival Commission 2012) by, for example, removing lakes and seas for terrestrial species. The resulting polygons, however, have been shown to still contain a large proportion of areas that are effectively unsuitable or provide little sustain to the individual species (Rondinini *et al*. 2005, 2011b).

A potential solution to fill this information gap consists in the use of habitat suitability models (HSM) as a tool to refine coarse EOO data to more realistic surrogates of species presence. Recently published maps of environmental variables and the availability of information on species-habitat relationships from IUCN initiatives (Schipper *et al*. 2008) allow for the development of high resolution HSMs at global scale (Rondinini *et al*. 2011b). While HSMs provide a welcome addition to our knowledge base, it is still uncertain what the implications of using these models in global conservation approaches are, particularly in relation to previous works (e.g. Rodrigues *et al*. 2004).

A third, alternative, data source consists in the use of large scale collections of point-presence sources such as those offered by the Global Biodiversity Information Facility (GBIF, GBIF 2014) or species atlases to map species presence. These are, however, usually characterized by a number of limitations such as uneven sampling effort and resolution across regions, as well as complete lack of data for certain species or regions. These limitations make them a poorly suited data source when dealing with large scale conservation exercises.

1.2.2 – Current Pressures

The IUCN Red List provides a description of threats for individual species, where available. These range from habitat loss and degradation to persecution, but are generally compiled from a species-wide point of view, and no information on the geographical distribution of said threats is generally provided. Arguably, the ideal data sources of pressures to be used for a systematic conservation analysis would be fine tuned to individual species, with each pressure mapped at the appropriate perceptual and effect resolutions (Andrén *et al*. 1997). From a global analysis perspective, however, this approach becomes impractical if not impossible to implement. A more feasible approach would be to use global maps of general influence, which can be used as proxies of human pressure on wildlife. The Global Human Influence Index (HII) Dataset of the Last of the Wild Project (Sanderson *et al*. 2002; Wildlife Conservation Society & Center for International Earth Science Information Network 2005) is one of said maps. It estimates human influence on the entire globe at a nominal resolution of 1 km^2, and accounts for local population density, land transformation, accessibility via road and train network, and electrical power infra-structure. Although the authors agree that it is only a partial representation of anthropic pressure (for example lacking any information on effects of pollution, climate change and other global phenomena), it still directly or indirectly includes the major pressures identified for mammals: namely habitat loss and direct collection (Cardillo *et al*. 2004; IUCN 2013).

1.2.3 – Future Pressures

In order to achieve the objectives set by the Millennium Development Goals (MDG), the United Nations Environmental Programme (UNEP) has promoted a series of initiatives to explore the consequences of different scenarios of socio-economic development. The Intergovernmental Panel on Climate Change (IPCC) was founded by the UNEP and the World Meteorological Association to provide scientific insight on climate change and its potential environmental, social and economic consequences. Products of the IPCC such as the Special Report on Emission Scenarios (SRES, IPCC 2007) have already been used to evaluate the potential impacts of development scenarios (Millennium Ecosystem Assessment 2005), as well as to design strategic socioeconomic development pathways that would allow for the MDG to be achieved (van Vuuren et al. 2012).

The Netherlands Environmental Assessment Agency (PBL) *Roads from Rio+20* report (van Vuuren *et al*. 2012), specifically explores the consequences of four development pathways. Business As Usual explores the consequences of a future where trends in economic growth, energy mix and consumption patterns are fundamentally unaltered from those seen today. Consumption Change aims to achieve the 2050 MDG targets by focusing on changes to consumption patterns, such as by reducing per capita meat intake, reducing waste in the production chains and generally by adopting a less energy-intensive lifestyle. Global Technology focuses on large-scale advances in technologically optimal solutions, such as intensive agriculture and a high level of international coordination, namely trade liberalisation.

Decentralised Solutions puts emphasis on reducing local impacts by decentralising resource allocations, for example promoting local energy production, and an agricultural system that is interwoven with natural corridor, as well as promoting policies that regulate equitable access to food. PBL used an integrated modelling approach to design said pathways (Bouwman *et al*. 2006), and among the various internal products they developed and used there are forecasts for biodiversity (Alkemade *et al*. 2009). These biodiversity forecasts have been used to inform on the biological consequences of favouring specific development strategies, and have recently been included in species specific approaches to methodologically assess individual specie's conservation status (Visconti *et al*. submitted; but also see § 2.4.1).

1.3 – Models: Opportunity and Vision

There is increasing evidence that global and integrated approaches to conservation are the most efficient in identifying optimal solutions (Millennium Ecosystem Assessment 2005). Because of the difficulties in obtaining wide scale empirical data sources, in the past large scale studies mainly used species' EOO as an indication of specie presence. Models allow for the opportunity to fill the information gap faced when dealing with these large-scale multi-species analyses: they have been used to refine species distribution data (Rondinini *et al*. 2011b) and assess the effectiveness of current protected area networks (Rondinini *et al*. 2005; Catullo *et al*. 2008). They have been used to forecast impacts of climate change on species distributions (Maiorano *et al*. 2011), and identify global hotspots of forecasted species loss

(Visconti *et al*. 2011). Examples highlighting the potential benefit of integrating modelled data in conservation are far from rare. However, virtually all published literature is based on regional if not local or single-species studies, and the wide variety of methods and approaches employed often render results not directly comparable across studies. In order for models to become informative decision-making tools, efforts need to be made in order to centralise and uniform analyses.

1.4 – Approach and Objectives

The work of this thesis aims to highlight the contribution that suitability models can give to global scale conservation biology. Four main themes will be approached: Opportunity offered by model improvements; Information added to analyses on current protection status; contribution offered to priority setting; information added to long term efficiency in conservation plans.

The modelling framework described by Rondinini *et al*. (2011b) will be re designed: the way environmental layers are combined will be modified, reducing fine-scale information loss.

A gap analysis for terrestrial mammals will be run following the approach described in Rodrigues *et al*. (2004). Conservation targets will be set by assuming the species occupies the entire EOO (as per the original approach), and by accounting for unsuitable habitat within the specie's EOO (Rondinini *et al*. 2011b). The coverage offered and proportion of target reached with the current protected area network under both approaches will be analysed, and gap species (i.e. species that do not reach their conservation target) identified. The implications of not using suitability models to inform species presence will be evaluated.

Species distribution data and conservation category from IUCN (IUCN 2013) and habitat suitability models (Rondinini *et al*. 2011b) will be used in conjunction with the HII (Sanderson *et al*. 2002; Wildlife Conservation Society & Center for International Earth Science Information Network 2005) in order to create a global pressure priority map for terrestrial mammal conservation. The top 17% of land mass and top 17% of each biome will be identified. Results will be interpreted in terms of benefits from using global vs local priority rankings and benefits of using a globalised pressure-state-response versus more simple richness approaches.

Following the approach used by Visconti *et al*. (submitted; but also see § 2.4.1), climate envelope models for carnivores and ungulates will be used to project current and future climate suitability maps. The climate-induced habitat loss in the current protected area network will be assessed for two socio-economic scenarios from the PBL Roads From Rio+20 report (van Vuuren *et al*. 2012): Business As Usual (BAU) and Consumption Change (CCH). Trends of habitat loss in protected areas will be highlighted and interpreted in terms of spatial and taxonomic effects. A general trend between habitat loss in protected areas and EOO size will be highlighted and extrapolated to all mammals in order to identify areas globally with a higher propension for reduced long-term effectiveness of protected areas.

2 – MATERIALS AND METHODS

2.1 – Improving Habitat Suitability Models

The HSMs published by Rondinini *et al*. (2011b) used the species-habitat relationships obtained from the 2008 IUCN global mammal assessment (Schipper *et al*. 2008). Information on habitat preferences, tolerance to anthropic disturbance, altitudinal limits and water dependence were coerced in three separate suitability models layers at a nominal resolution of 300 m on a Mollweide equal-area projection.

The altitudinal range layer was a binary model indicating whether the species was within or outside its altitudinal limits, and was associated with the Shuttle Radio Topography Mission (SRTM) Digital Elevation Model (DEM) (United States Geological Survey 2006). Species-habitat relationships and tolerance to anthropic disturbance were used to develop species-specific suitability scores for the GlobCover land cover classification scheme (European Space Agency & Universite Catholeque de Louvain 2009): three levels of suitability were used: unsuitable, medium suitability and high suitability. Since the resolution the GlobCover land cover layer was provided in was too coarse to identify many water streams which might have been essential for aquatic or water dependant species, the Vmap0 linear permanent water map (National Imagery and Mapping Agency 1997) was rasterised at the same resolution and projection GlobCover originally came in, and originally used to add extra information on water presence by substituting the original GlobCover classification values with the value “210”, which codifies water. Information on water dependence was used to identify terrestrial species with a tight relationship with water: presence of these species was assumed to be only within 1 km from any one water source, so a 1 km buffer was applied to areas identified as water from the land cover layer (the union of the GlobCover land cover map and the Vmap0 linear permanent water map) and the resulting water buffer area coerced into a “water buffer layer”. The three environmental layers (altitude, land cover, water presence) were united into a single combined environmental layer map (Figure 1). Information on species-specific environmental layers were uploaded into a PostgreSQL (PSQL) database (The PostgreSQL Global Development Group 2010) at the time of the original publication), which was then used to extract all combinations of values from the combined environmental layer which would result in a medium or high suitability score. While the actual suitability score came from information on land cover, any land cover type outside of altitudinal limits and of the water buffer zone (for terrestrial water-dependant species only) was classified as unsuitable.

In order to eliminate information loss that originated by overwriting the GlobCover land cover classes with information from the rasterised Vmap0 map, a new approach was devised (Figure 1) and a new combined environmental layer map was developed. This and all following GIS analyses were made using the Geographic Resources Analysis Support System (GRASS) software version 6.4.2 (GRASS Development Team 2012). Information from the altitude environmental layer was not altered. The GlobCover layer was not overwritten with data from Vmap0: in alternative, the water buffer layer from the previous approach was modified so that it provided 3 information classes on water presence: water present (if either GlobCover or Vmap0 indicated water presence), water nearby (if the site contains no water but is within a 1 km buffer from known water sources), no water (if the site contains no water and is further than 1 km from known water sources). All species followed the same general rule that land cover and altitudinal limits indicated the suitability score. However, aquatic, water dependant and terrestrial species were modelled following slightly different criteria. Aquatic species (i.e. species for whom water was the only suitable land cover class) were assigned high suitability score to all areas intersected by water and within their altitudinal limits, even if the main land cover type was not water. Water dependant species were modelled so that suitability scores were determined by the prevalent land cover type (i.e. as indicated by GlobCover), but only within their altitudinal limits and a 1 km buffer from known water sources. Land species were modelled as water dependant species, but without the 1 km buffer threshold.

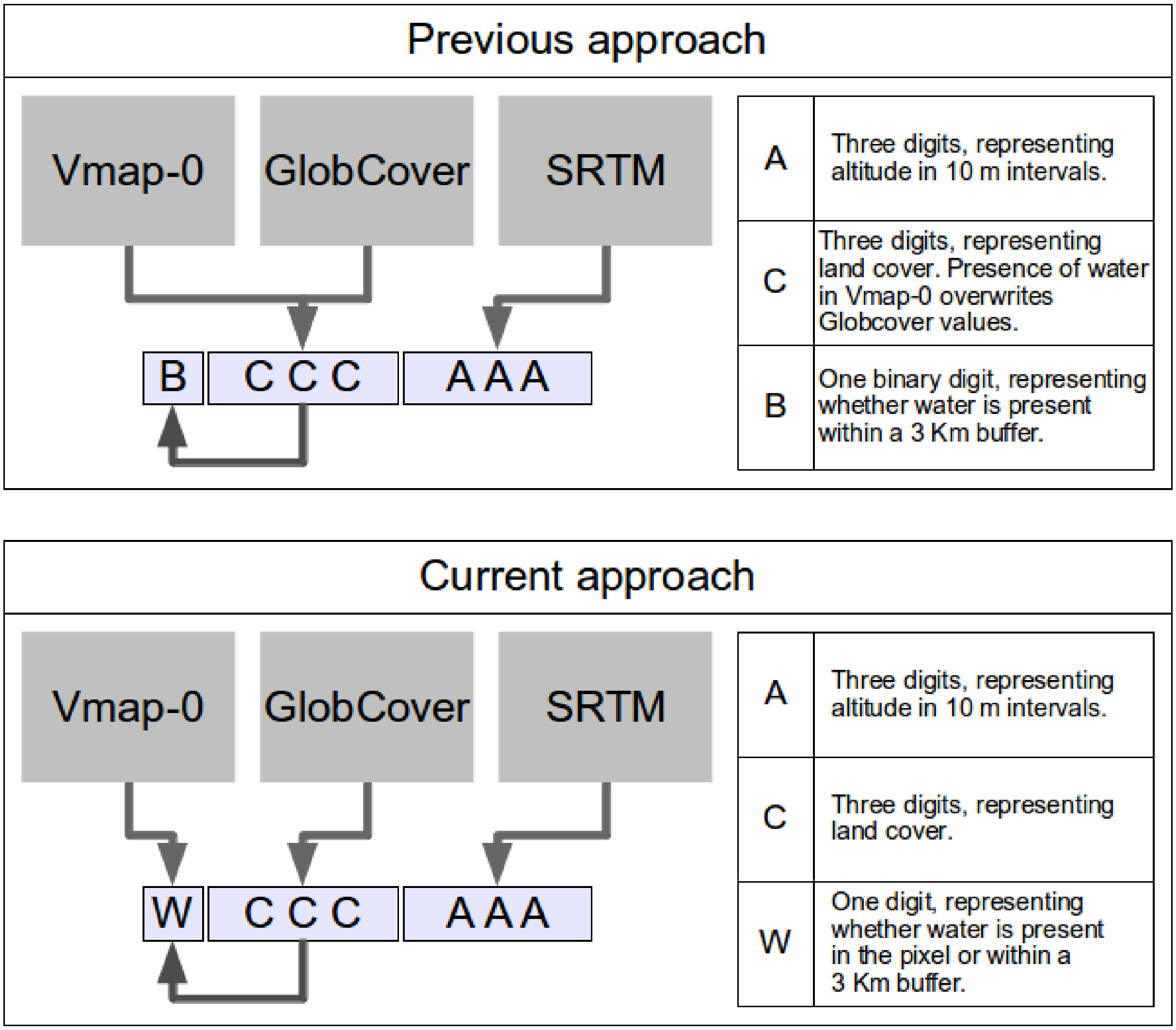

Figure 1: Previous and current approach in building the combined environmental map layer from which the habitat suitability models are obtained. Refer in text for a more detailed description and rationale of the approach. An altitude (Shuttle Radio Telegraphy Mission, SRTM), global land-use (GlobCover) and vectorised water-body layer are combined in a single map. In the previous approach, the vectorised water layer map (Vmap-0) would overwrite the digits that classified land-use, redefining the pixels as water in order to account for the fact that the majority of water bodies would not be picked up by GlobCover's relatively coarse (300 m) resolution. A single binary "water buffer digit" would define whether the pixel was within a 1 km buffer around mapped water sources, in order to refine models for water-dependant terrestrial species. In the new approach the species would a priori be classified as water species (only present in pixels where there is water), water-dependant species (only present within 1 km from known sources of water) or land-species (potentially present everywhere); and the water digit may be one of multiple values which define if the pixel contains water, is within a 1 km buffer from a pixel that does, or is outside of said threshold.

2.2 – Global Gap Analysis

The EOOs from the IUCN Red List 2013 assessment (IUCN 2013) and HSMs for 5097 terrestrial mammals (Rondinini *et al*. 2011b; but also see § 2.1) were used to calculate total EOO size and Extent of Suitable Habitat (ESH, i.e. the total suitable surface within a specie's EOO) for each species. Vectors of species EOO, originally in geographic projection, were reprojected to a Mollweide equal area projection. The ESH was calculated from the models accounting for both high and medium suitability habitat types. In order to obtain measurements of EOO and ESH within protected areas, the World Database of Protected Areas (WDPA, IUCN & UNEP 2011) was downloaded and used. Terrestrial protected areas falling into IUCN categories Ia to IV (IUCN 2008) were rasterised on a Mollweide projection at a resolution of 300 m. These specific categories were chosen as they explicitly incorporate definitions that give direct benefit to species protection.

In order to run the gap analysis, the approach described by Rodrigues *et al*. (Rodrigues *et al*. 2004) was used and adapted. Species specific conservation targets were set as a variable proportion of EOO that needed to be protected in function of absolute EOO size (EOO target). For species with EOO $\leq$ 1000 km^2, conservation target was set to 100%; for species with EOO $\geq$ 250.000 km^2, the target was set to 10%; for species with EOO contained between these two, the target was interpolated. A second target, based on available ESH (ESH target) was obtained by multiplying the EOO target with the proportion of suitable habitat within each specie's EOO. Species were considered gap if they failed to reach their representation target within protected areas. Three sets of gap analyses were performed: EOO-EOO based, HSM-HSM based and, in order to better interpret the discrepancy between these two main approaches, EOO-HSM based. For the EOO-EOO based gap analysis, EOO target was used and the entire EOO contributed to reaching the target; for the HSM-HSM based analysis HSM target was used, and only suitable area within the EOO was considered contributing; for the EOO-HSM based analysis, EOO target was used, but only suitable area contributed towards the target.

2.3 – Pressure-State-Response prioritisation approach

In order to identify areas with higher conservation priorities globally for terrestrial mammals, a pressure-state-response approach was used. One of the main concepts in conservation planning is that a candidate conservation feature, in order to receive benefit from any type of conservation action, should be both sensitive (i.e. exposed) to a conservation pressure and susceptible (i.e. potentially influenced) to it. The more a conservation feature is sensitive and the more it is susceptible, the higher it's conservation priority should be. In this approach, the HII score (Sanderson *et al*. 2002) was used as a proxy for sensitivity to general anthropic pressure, and each specie's Red List conservation status (IUCN 2013) and total available suitable habitat as a proxy for susceptibility.

Conservation priority maps were made globally at a 10 km resolution in Mollweide equal area projection. The HSMs for 5097 terrestrial mammals were used to refine IUCN range maps (Rondinini *et al*. 2011b; IUCN 2013; but also see § 2.1): true species presence was assumed to occur only in presence of at least medium suitability habitat, and the original HSMs at 300 m resolution were resampled so that each 10 km pixel represented the proportional contribution to the specie's total available suitable habitat within the EOO. Two priority indexes were developed (Figure 2). The first one represents extinction-risk weighted pressure state (PSW) which accounts for species' IUCN Red List category, and was obtained by summing the product of suitable habitat proportion, HII score (Sanderson *et al*. 2002) and a Red List category weight ranging from 1 for data deficient and least concerned species to 5 for critically endangered species. The second is a non-weighted priority index (PS), and was obtained by applying the same approach but excluding the Red List category weight. After both indexes were calculated and maps produced for the entire world's dry land, they were normalised by dividing each map for the highest index value recorded and multiplying the quotient by 100, so that the values for both PSW and PS maps ranged from 0 to 100. In order to understand the different effects of proportional priority score and priority rank, a complementary set of ranked priority maps with scores ranging from 0 to 100 were also developed. The global top 17% ranking areas of the world as well as the top 17% or each biome were selected as potential conservation candidates following the indications of Aichi target 11.

$$PS_a = \sum_{i=1}^{N_a} H_a \times P_{i,a}$$

$$PSW_a = \sum_{i=1}^{N_a} H_a \times P_{i,a} \times W_i$$

Figure 2: Two metrics used to calculate pressure-state maps in order to identify priority sites for conservation. H_a represents the Human influence index at site a. $P_{i,a}$ is the prevalence of species i in site a, or the contribution of site a to species i's extent of suitable habitat. W_i is the weight given to species i according to IUCN Red List categories: 1 for data deficient and least concerned species, 2 for near threatened, 3 for vulnerable, 4 for endangered and 5 for critically endangered.

In order to analyse the distribution of priority scores in protected areas, the WDPA ((IUCN & UNEP 2011)) was downloaded and used. Terrestrial protected areas falling into IUCN categories Ia to IV (IUCN 2008) were rasterised on a Mollweide projection at a resolution of 300 m. These specific categories were chosen as they explicitly incorporate definitions that give direct benefit to species protection. The 300 m map of protected areas was then resampled at 10 km in such a manner that the proportion of protection for each 10 km cell was recorded. An analogous approach was used to obtain maps of biogeographic realms and biomes (Olson *et al*. 2001; WWF 2001), with the difference that each 10km pixel was assigned to the realm or biome that covered the most surface (mode resampling). Statistical analyses were performed in the R statistical programming language suite (R Development Core Team 2011) with the addition of the “MASS” library (Venables, W. N. & Ripley 2002).

2.4 – Impact of climate change: effect on protected areas

While protected areas are expected to explicitly prevent virtually all forms of anthropic land-use change, they are still potentially influenced by climate change. In order to measure the extent to which current protected areas are influenced by climate change, climatic envelope models were developed for carnivores and ungulate (Visconti *et al*. submitted; but also see § 2.4.1) using 10 key bioclimatic variables (Table 1) and future climate envelopes were projected for the Business As Usual (BAU, associated with IPCC scenario A1B) and Consumption Change (CCH, associated with IPCC scenario B1) Rio+20 scenarios (IPCC 2007; van Vuuren *et al*. 2012), and the proportional change in coverage (i.e. climate-driven habitat loss) within protected areas was measured.

WorldClim Name	Variable type
Bio 1	Annual Mean Temperature
Bio 1	Mean Temperature of Wettest Quarter
Bio 9	Mean Temperature of Driest Quarter
Bio 10	Mean Temperature of Warmest Quarter
Bio 11	Mean Temperature of Coldest Quarter
Bio 12	Annual Precipitation
Bio 16	Precipitation of Wettest Quarter
Bio 17	Precipitation of Driest Quarter
Bio 18	Precipitation of Warmest Quarter
Bio 19	Precipitation of Coldest Quarter

Table 1: Ten bioclimatic variables used by Visconti *et al*. (submitted) to develop and forecast climatic envelope models for carnivores and ungulates. The variables were developed following the approach for the Anuclim software bioclim variables. Bioclimatic envelopes were developed using the bioclim2 R software library using 2010 climate, and projected for the climatic scenarios associated to the Roads from Rio+20 Business As Usual and Consumption Change strategic pathways (van Vuuren *et al*. 2012).

2.4.1 – Obtaining climate envelope models

The climatic envelope models used were originally developed for a publication that is currently under review (Visconti *et al*, submitted); the modelling approach is reported here for reference purposes. The biomod2 package for the R statistical programming suite (R Development Core Team 2011; Thuiller *et al*. 2013) was used to fit bioclimatic envelope models for 440 species belonging to the Orders Carnivora, Cetartiodactyla, Perissodactyla and Proboscidea, although only 418 of these contained protected areas within their ranges and were subsequently included in this study. Presence points for modelling were obtained by assuming species to be present in their entire EOO, and point coordinates were obtained by systematically sampling along a regular 30' grid (approximately 50 km at the equator). Pseudo-absences were obtained by randomly sampling 1000 points along the same grid, outside of each specie's EOO but within the intersection of each specie's continuous biogeographic realm and land-mass (island or continent) of origin. This allowed pseudo-absences to represent areas that were both potentially suitable and reachable for each species. Since for each species the number of presence points and pseudo-absences were in different numbers, these were weighted during the modelling process, so that presence and absence each contributed with a proportional weight of 0.5 in the model. Seven statistical models from the biomod2 software package were used for modelling (Generalized Linear Models, Generalized Boosted regression Models, Generalized Additive Models, Classification Tree Analyses, Artificial Neural Network, Multi-Adaptive Regression Splines, Random Forest), and for each species only models with a high predictive capacity (true skills statistics score >

0.7) were retained. The retained models were used to project climatic suitability for the BAU and CCH scenarios for the decadal intervals between 2010 and 2050; the outputs were binarised using the threshold that maximised TSS score, and subsequently combined into a single output by selecting the modal value. If among the models retained there was an equal number of presences and absences at any specific spatial location, presence was assumed.

2.4.2 – Habitat loss trends in protected areas

For each species, changes of available habitat in protected areas was measured as a combined effect of land-use, altitude and climate. Land-use, altitude (Rondinini et al. 2011b; but also see § 2.1) and protected areas (IUCN & UNEP 2011; but also see § 2.2) were assumed to remain constant, as altitude is virtually constant and protected areas should be both permanent and prevent all major forms or anthropic land-use change. The maps of protected areas, HSMs and climate envelope models were reprojected to Mollweide equal area projection, with a resolution of 100 km^2 (10 x 10 km) so that every pixel represented respectively the proportion of each site that was protected, of suitable habitat, and climatically suitable. By multiplying the proportions of suitable climate and land-use, a single map was obtained for every scenario and decade representing overall available habitat. An estimate of suitable habitat within protected areas was obtained for each species under both scenarios by multiplying these last maps with the proportion of protected areas. For every species, trends of change in available habitat within protected areas were measured as the proportional difference between habitat available in 2010 and habitat available for subsequent decades under both scenarios (Figure 3).

$$\Delta P_{sp,year,sc} = \frac{P_{sp,year,sc} - P_{sp,2010,sc}}{P_{sp,2010,sc}}$$

Figure 3: The proportional change in coverage (*ΔP*) offered by protected areas for the decadal interval 2010-2050 was calculated for 418 carnivore and ungulate species. Land-use was assumed not to change within protected areas, however, the total amount of habitat available (*P*) for each species (*sp*) within protected areas was assumed to vary in time (*year*) under two socio-economic development scenarios (*sc*) due to climate change.

A random forest modelling procedure was used to identify the intrinsic and extrinsic factors that appeared to have the highest impact on *ΔP*. The "randomForest" R statistical library (Liaw & Wiener 2002) was used for this objective, and 13 variables were tested in total: taxonomic Order and Family; Red List category and threatened vs non threatened status; species weight; biogeographic realm; central latitude and longitude in the specie's range; mean, median and 90th percentile of HII values in the specie's realm; specie's range size and protected surface area. Since no definitive predictor was found, analyses concentrated on the only clearly observable trend that could be extrapolated, between the maximum observed relative loss in protection and species range size: higher projected losses in suitable habitat were only observed among species with smaller ranges. In order to extrapolate this trend to all other terrestrial mammals, a generalised linear model (GLM) was fitted on the top 10% of species losing habitat by range size according to the BAU scenario in the year 2050. The 418 carnivore and ungulates were grouped by range size with a 2.000.000 km^2 wide bandwidth, and in each group the species within the 90th percentile of habitat loss were selected to fit a GLM using an inverse link function with gamma distribution.

2.4.3 – Extrapolating habitat loss trends to all mammals

Ideally, protected areas should aim to guarantee long term survival of the species they are designed for. If any single species has a high risk of losing suitable habitat through drivers that are unlikely to be mitigated by the institution of protected areas, there are grounds for the former to be excluded from any prioritization algorithm. This would lead to a potential shift in apparent conservation value for protected areas. In order to understand the spatial distribution of this shift in conservation value, a GLM was developed linking range size to potential range lost due to climate change. This GLM was then used to extrapolate potential habitat loss for all mammals. Species richness maps were developed using only species that risked losing more than predetermined thresholds of suitable habitat across their range.

3 – RESULTS

3.1 – Improvements to Habitat Suitability Models

On a Mollweide projection with a resolution of 300 m (0.09km^2), which is approximately the equatorial resolution the GlobCover land cover dataset comes at (European Space Agency & Universite Catholeque de Louvain 2009), the global land masses are subdivided in approximately 1,500,000,000 units. Of these, 0.3% are mainly covered in water and are recorded as such by the GlobCover map. A further 2.3% are intersected by minor water sources as indicated by the Vmap0 water layer (National Imagery and Mapping Agency 1997).

The previous approach used to register these minor water sources consisted in overwriting the original land cover values. At a resolution of 10 km (100 km^2), which is a relatively common scale for global conservation analyses, this approach influenced 56% of planning units (Figure 4). In the influenced land units, the proportion of reclassification error was strongly left-skewed (Shapiro Wilk normality test, W = 0.4026, p-value < 0.001), with a median reclassified surface of 3.6% and 95% of impacted cells having an affected proportion lower than 9%. The countries with an affected value higher than 9% were virtually all small islands (Table 2). Impacted land units were present in the entire world, and noticeable non impacted regions tended to concentrate only in desertic regions (Figure 5). Overall, the classification of

2.15% or the world's land surface was been improved in the current approach, with Europe and North America having a larger improvement prevalence (Table 3).

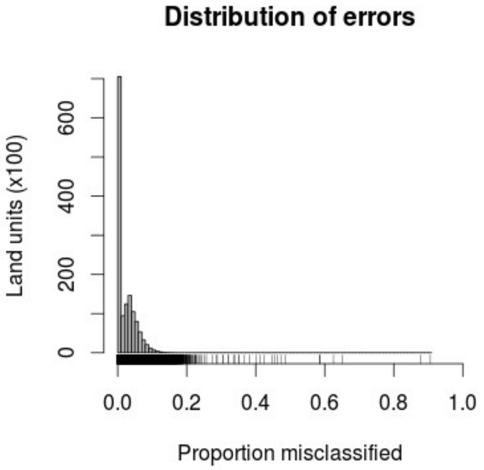

Figure 4: Distribution of land surface with improved land cover and water availability classification. Values refer to a comparison with previous global terrestrial mammal habitat suitability modelling, at a resolution of 10 x10 km. In order to account for species with strong water dependence, habitat suitability models in the previous approach (Rondinini *et al*. 2011b) used a GlobCover land cover layer (European Space Agency & Universite Catholeque de Louvain 2009) modified with additional information from the Vmap0 vector water layer (National Imagery and Mapping Agency 1997). land use was reclassified as water in accordance to the information in Vmap0 to account for minor water streams that were not registered by GlobCovers's spatial resolution.

Country	km²	Misclassification (%)	Continent
British Indian Ocean Territory	300	100	Asia
St. Helena	1300	100	Africa
Niue	400	100	Australian Area
Wallis and Futuna Islands	600	100	Australian Area
Samoa	4200	100	Australian Area
Seychelles	1700	82	Africa
French Polynesia	13300	82	Australian Area
Cook Islands	500	80	Australian Area
American Samoa	1100	79	Australian Area
Bermuda	700	56	North America
Tuvalu	300	56	Asia
Tonga	3600	43	Australian Area
Kiribati	5200	38	Australian Area
Virgin Islands (British)	1400	37	North America
Micronesia (Federated States of)	2700	34	Australian Area
Marshall Islands	1200	33	Asia
Anguilla	700	29	North America
Palau	1900	21	Asia
Saint Vincent and the Grenadines	1800	17	North America
Mauritius	3200	14	Africa
Fiji	37500	11	Australian Area
Belize	26600	10	North America
Vanuatu	27800	9	Australian Area
Solomon Islands	54100	9	Australian Area

Table 2: Countries impacted by more than 9% of their surface in the previous approach for global mammal habitat suitability modelling. The countries with higher proportional errors were mainly small islands.

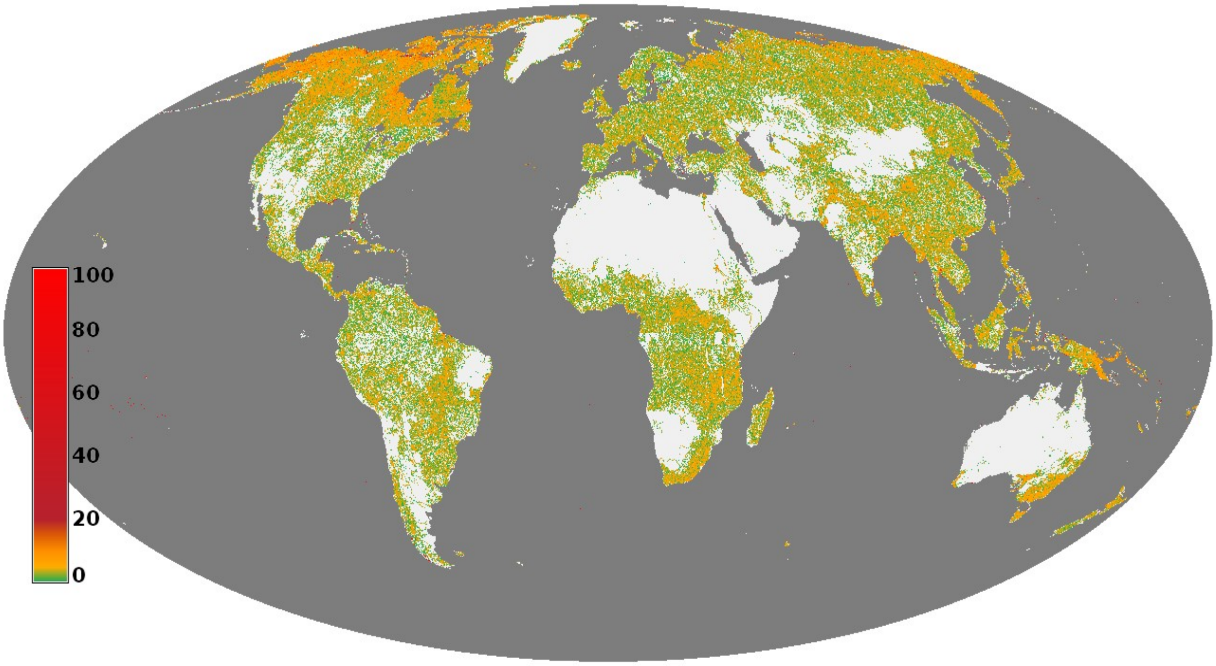

Figure 5: Proportion of reclassified surface at a resolution of 100 Km^2 (10 x 10 Km). Light grey indicates areas that were not affected. In the previous approach to global mammal habitat suitability modelling, detailed information from the Vmap0 vectorised water layer was used to overwrite land cover values to account for fine scale water sources.

Geographic region	Surface area Reclassified	Proportion of region Reclassified
World	2.980.869 Km^2	2.15%
Africa	399.162 Km^2	1.32%
Asia	601.307 Km^2	1.86%
Australia	98.419 Km^2	1.90%
Europe	752.330 Km^2	3.17%
North America	778.458 Km^2	3.06%
South America	350.886 Km^2	1.94%

Table 3: Total surface area and proportion of each geographic region that has been reclassified as water in previous global habitat suitability approaches for mammals.

3.2 – Global Gap Analysis for Terrestrial Mammals

By measuring each specie's proportional protected area cover we can observe that suitable habitat is significantly better protected than the entire species range, independently of threat level (Wilcoxon signed rank test: all species, $V = 3345816$, $p < 0.001$; threatened species, $V = 90467$, $p < 0.001$; non threatened species, $V = 2341009$, $p < 0.001$) (Figure 6). Threatened species are not, however, more covered than their non threatened counterparts, for both measures of EOO and ESH proportional protected area cover (Wilcoxon rank sum test: EOO, $W = 2073030$, $p = 0.145$; ESH, $W = 2087984$, $p = 0.268$) (Figure 7). On average, 9.7% ± 0.2 SE of the EOO and 10.8% ± 0.2 SE of the ESH were protected.

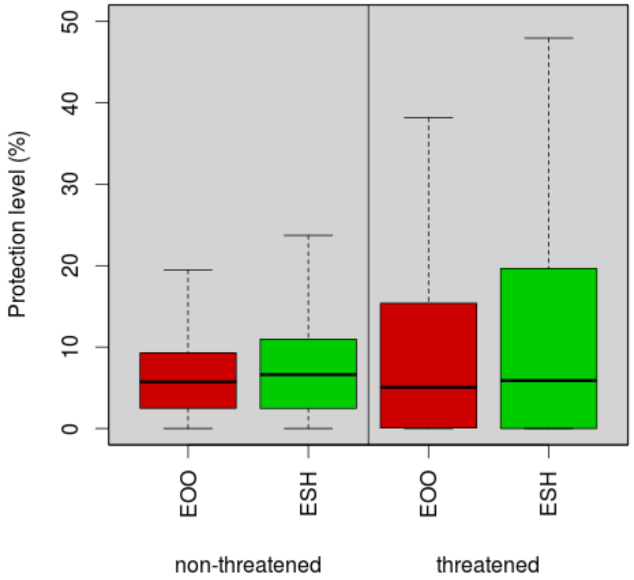

Figure 6: Protection level in the species range (EOO) and extent of suitable habitat (ESH), for threatened and non-threatened terrestrial mammals. Protected areas of IUCN category Ia-IV (IUCN 2008; IUCN & UNEP 2011) were used to measure coverage in species EOO (IUCN 2013) and ESH (Rondinini *et al*. 2011b).

The distribution of the EOO-ESH protected area cover residuals varied among taxonomic groups: of the 26 mammalian Orders analysed, 5 (Tubulidentata, Proboscidea, Dermoptera, Microbiotheria, Notoryctemorphia) had less than 3 species; a further 5 had non normal residual distributions (Shapiro test: Hyracoidea, $p = 0.35$; Pholidota, $p = 0.18$; Paucituberculata, $p = 0.93$; Pilosa, $p = 0.76$; Perissodactyla, $p = 0.83$); the remaining 16 Orders had normal residual distributions (Shapiro test: $p < 0.05$).

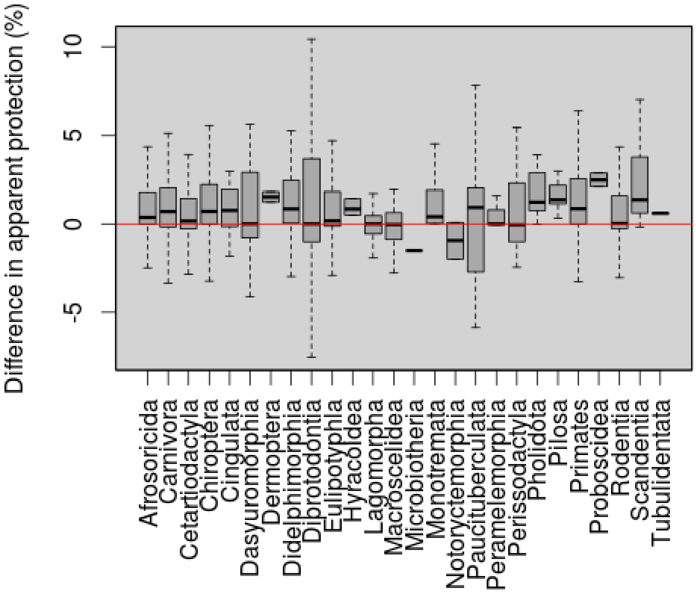

Figure 7: Difference in apparent protection between species range (EOO) and extent of suitable habitat (ESH) based measurements. Protected areas of IUCN category Ia-IV (IUCN 2008; IUCN & UNEP 2011) were used to measure coverage in species EOO (IUCN 2013) and ESH (Rondinini *et al*. 2011b). Positive values indicate that the ESH-based measurements register higher protection levels.

Applying the methods and thresholds described by (Rodrigues *et al*. 2004), 47% of all mammal species (n = 2417) were assigned a protected area cover target of 10% of the EOO; a further 8% (n = 416) a target of 100% of the EOO. The remaining species were assigned a variable target (n = 2264). On average 49% ± 26 SD of species EOO was suitable, and 95% of species had ESH values that fell between 6% and 92% of their EOO. There were also 1422 species (28% of total) for whom ESH was lower than the defined EOO based conservation target. For these species, non suitable habitat would need to be protected in order to reach non-gap status. Alternatively, these species would be gap by default if only suitable habitat is to be used to achieve EOO based conservation targets (Figure 8). There was a higher proportion of threatened species among the gap by default species (44%) than among the entire sample (20%); data deficient species also are over-represented in the gap by default sample (27% v.s. 13% of all species). Near threatened species were equally represented in the two groups (6%), and least concerned species was the only Red List category that was under-represented in the gap by default group (24% v.s. 60%) (Figure 9).

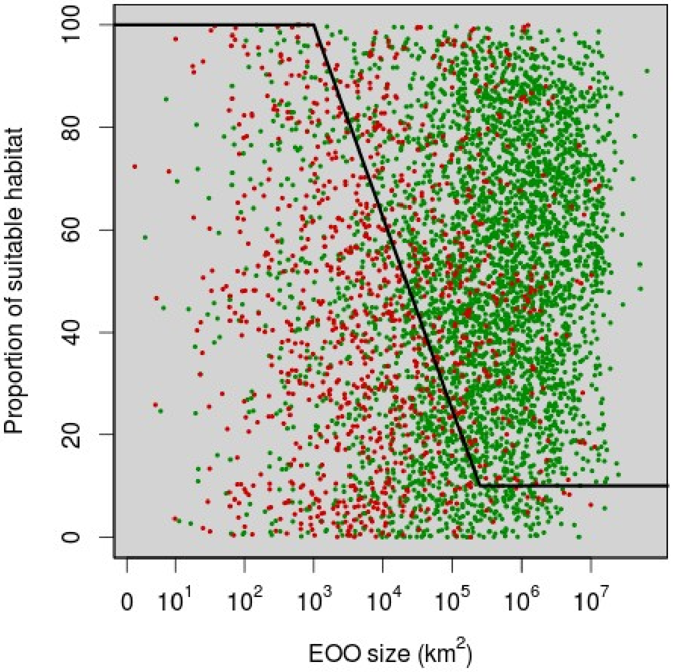

Figure 8: Extent Of Occurrence (EOO) versus proportion of suitable habitat for terrestrial mammals. Threatened (red) and non-threatened (green) species are shown. The black line represents the conservation target (a variable proportion of species EOO that needs to be protected) as used in (Rodrigues *et al*. 2004). For the species falling below the black line, non-suitable habitat would need to protected in order to achieve their conservation target.

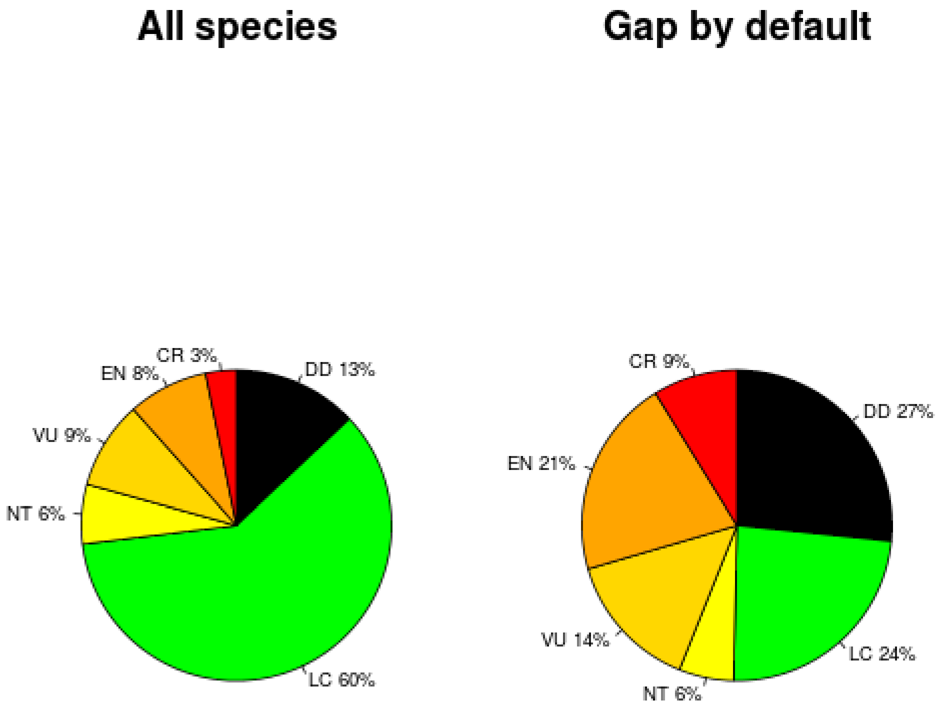

Figure 9: Proportion of species belonging to each IUCN Red List category for all terrestrial mammals and gap by default mammals. Gap by default species are those species for whom protected area coverage targets could not be achieved if targets were to be defined according to the extent of occurrence, but only achieved accounting for suitable habitat contribution. See in text for further details.

Accounting for ESH prevalence in EOO target setting (i.e. converting the proportional EOO target to km^2, multiplying this by ESH prevalence and allowing only suitable habitat to count towards the target goal, Figure 10) causes targets to systematically shift towards smaller values: 65% of species (n = 3335) have an ESH-adjusted target of less than 10% of their respective EOO. This means on top of the 2417 species with an EOO based target of 10%, a further 918 species had their target lowered to the 10% threshold. Only 2 species out of the original 416 (*Pipanacoctomys aureus* and *Ctenomys famosus*) maintained an ESH target of 100%.

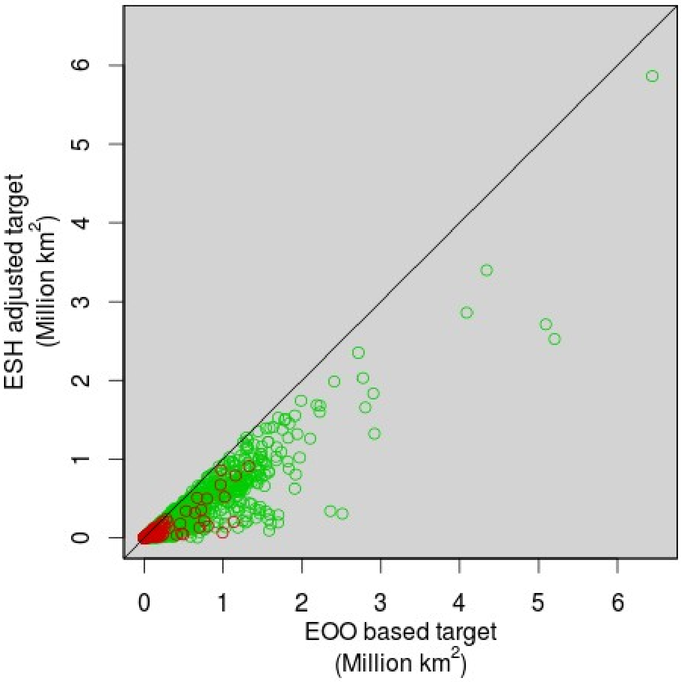

Figure 10: Ratio between Extent Of Occurrence (EOO) based and Extent of Suitable Habitat (ESH) adjusted conservation targets. Threatened (red) and non-threatened (green) terrestrial mammals are represented. The diagonal represents matching EOO and ESH targets. The formula used to calculate EOO conservation target is taken from (Rodrigues *et al*. 2004). ESH target was obtained by multiplying EOO targets with prevalence of highly suitable habitat within each specie's EOO.

EOO and ESH based approaches gave significantly different results when comparing percentage of target reached (paired Student's t: $t = -20.94$, $p < 0.001$): on average, species reached 45% ± 45 SD of their EOO based target, while this figure raises to 53% ± 55 SD when accounting for ESH prevalence. The percentage of target reached by the EOO or the ESH approach was strongly correlated (Pearson's $r = 0.89$). Out of 5097 species, 4690 (93%) were not impacted by target setting approach, with 4280 (84%) always failing to achieve their conservation target and 432 (9%) always reaching it. Assuming the ESH approach is an improved approach to conservation target setting, 1% of species ($n = 48$) were subject to omission error and 7% ($n = 337$) to commission error when failing to account for ESH prevalence (Figure 11).

The Orders Dasyuromorphia, Perissodactyla and Cingulata had higher proportions of omission errors: 6.9%, 6.3% and 4.8% respectively (Table 4). These are, however, taxa with a relatively small number of species (72, 16 and 21). For Perissodactyla, for example, there was only one species (*Tapirus indicus*) subject to omission errors. Numbers became more substantial for taxa subject to commission errors, despite the more modest percentual impact: 5.5% for Cetartiodactyla, 9% for Carnivora and 4.3% for Chiroptera (235, 245 and 1168 species respectively in each taxa).

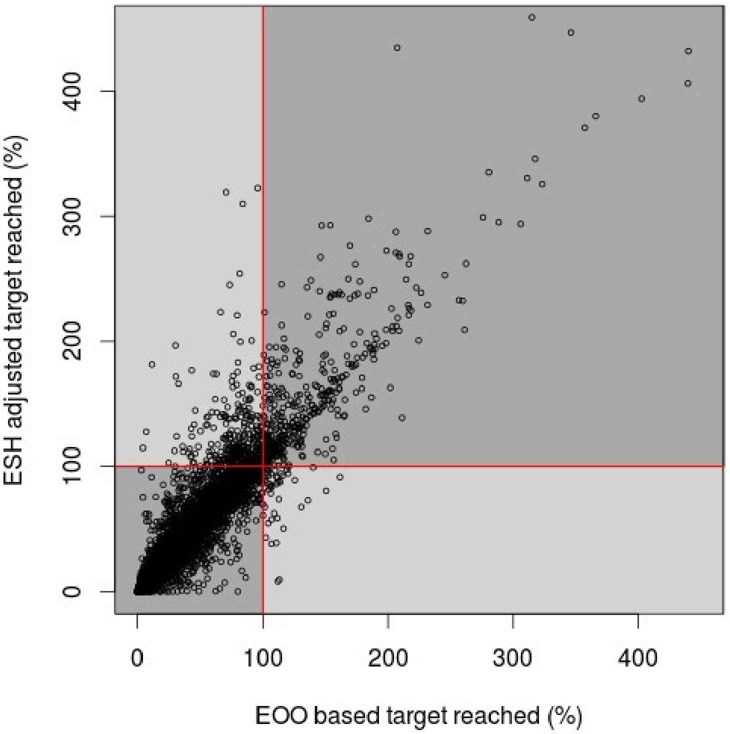

Figure 11: Proportion of conservation target reached using Extent Of Occurrence (EOO) versus Extent of Suitable Habitat (ESH) to calculate required coverage by protected areas and measure achievement. Species falling in the dark quadrants are classified as gap or non-gap under both approaches. Species in the top-left quadrant are commission species (n = 337), classified as gap according to the EOO approach, but not by the ESH approach. Species in the bottom-right quadrant are omission species (n=48), classified as non-gap according to the EOO approach, but as gap by the ESH approach.

Order	n.	Omission %	Commission %
Afrosoricida	53	0.0	0.0
Carnivora	245	0.8	9.0
Cetartiodactyla	235	1.3	5.5
Chiroptera	1068	0.7	11.9
Cingulata	21	4.8	9.5
Dasyuromorphia	72	6.9	6.9
Dermoptera	2	0.0	0.0
Didelphimorphia	94	0.0	6.4
Diprotodontia	138	2.9	10.9
Eulipotyphla	406	0.7	4.4
Hyracoidea	5	0.0	0.0
Lagomorpha	91	0.0	3.3
Macroscelidea	16	0.0	6.3
Microbiotheria	1	0.0	0.0
Monotremata	5	0.0	0.0
Notoryctemorphia	2	0.0	0.0
Paucituberculata	6	0.0	0.0
Peramelemorphia	19	0.0	5.3
Perissodactyla	16	6.3	0.0
Pholidota	8	0.0	25.0
Pilosa	9	0.0	0.0
Primates	407	0.0	4.2
Proboscidea	2	0.0	0.0
Rodentia	2156	1.0	4.3
Scandentia	19	0.0	21.1
Tubulidentata	1	0.0	0.0
total	5097	0.9	6.4

Table 4: Number of species and proportion of omission and commission errors in identifying gap species in terrestrial mammals. A species was considered gap if it failed to achieve the conservation target as calculated by (Rodrigues *et al*. 2004). Omission and commission errors refer to the discrepancy in classification when failing to account for the distribution and extent of suitable habitat present within the species range.

Measuring the proportion of target reached accounting for ESH, the distribution of values across taxonomic Orders varied, with only 6 groups of relatively small size having a normal distribution (Shapiro normality test: $p > 0.05$; Scandentia, Monotremata, Macroscelidea, Cingulata, Pilosa, Paucituberculata) Figure 12. With the exception of Proboscidea, Pholidota, Monotremata, Scandentia and Pilosa (respectively 100%, 50%, 40%, 37%, 33% of target reached), all species fare poorly, with at least two thirds of species failing to reach their conservation target. However, observations in these groups may at least in part be caused by stochasticity, as among them Cingulata is the largest order with 21 species, and the worst faring taxa (Dermoptera, Microbiotheria, Notoryctemorphia and Paucituberculata, Tubulidentata, for whom all species fail to reach their respective conservation targets) all have less than 2 species.

A more defined trend can be observed comparing the distribution of the proportional target reached among IUCN Red List categories, LC, NT, VU, EN and CR species having median ± median absolute deviation (MAD) of respectively 53% ± 47, 36% ± 45, 31% ± 43, 10% ± 15 and 4% ± 5 (Figure 13). Data deficient specie rank between VU and EN, with a median target reached of 14 ± 21 MAD. A significant difference between categories was registered (Kruskal-Wallis rank sum test: $\chi^2 = 508.5$, $p < 0.001$), however, Bonferroni corrected Wilcoxon rank sum tests between adjacent categories indicated that significant differences were only present between LC and NT species, and between VU and EN species (LC-NT $p < 0.001$; NT-VU $p = 1.000$; VU-EN $p < 0.001$; EN-CR $p = 0.860$).

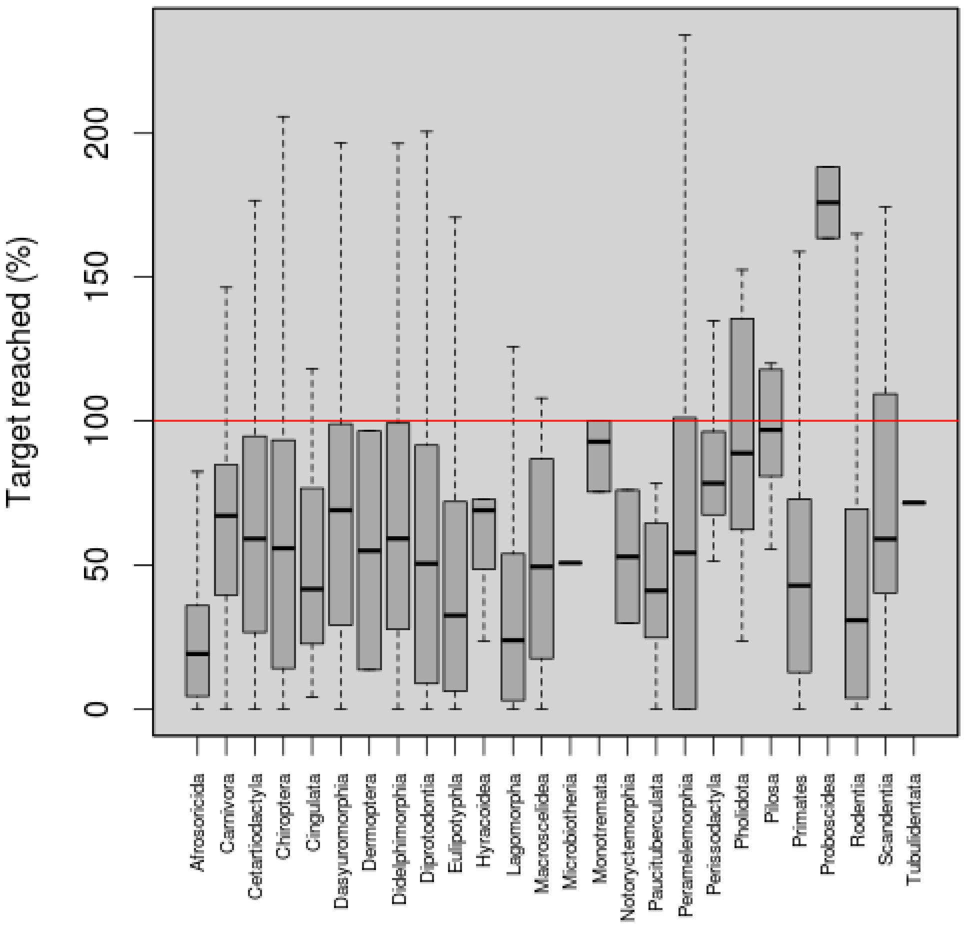

Figure 12: Proportion of conservation target reached by taxonomic Order for terrestrial mammals. The proportional conservation target is calculated according to the formula published in (Rodrigues *et al*. 2004) and adjusted accounting for each specie's available suitable habitat.

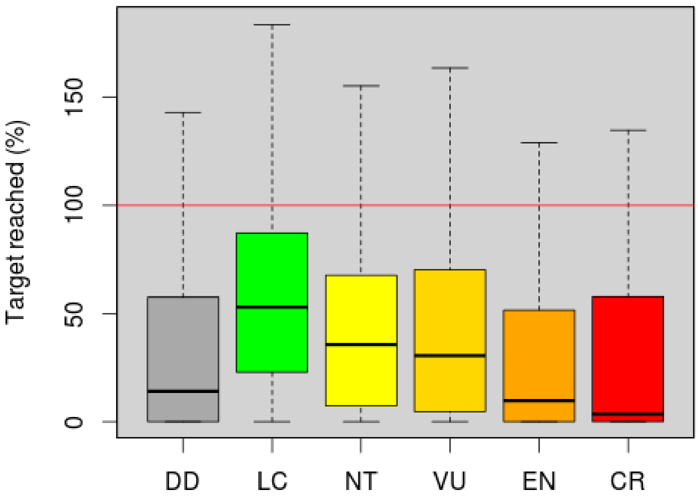

Figure 13: Proportion of conservation target reached by IUCN Red List category for terrestrial mammals. The proportional conservation target is calculated according to the formula published in (Rodrigues *et al*. 2004) and adjusted accounting for each specie's available suitable habitat. Bonferroni corrected Wilcoxon rank sum test between adjacent categories (excluding DD) indicates that significant differences were only present between LC and NT species ($p < 0.001$) and between VU and EN species ($p < 0.001$).

3.3 – Pressure-State-Response prioritisation approach

The choice to use or not IUCN Red List category when calculating a global cumulative pressure map for terrestrial mammals did not give rise to major differences in global patterns, and values between the two approaches were very strongly correlated (Pearson's $r = 0.936$); given this preliminary result it was decided to concentrate all further analyses using the PSW metric, which accounts for conservation status weight (Figure 14). Proportional scores were heavily weighted towards low values, with 11% of the global land surface having a priority score of zero and only a very small proportion of global land surface having extremely high priority. After normalising values to scores between 0 and 100, the 90th, 95th and 99th percentile of priority scores were respectively 0.08, 0.16 and 0.59; and the top 1% ranking sites held more than 19% of global cumulative conservation value.

PSW scores were not correlated with species richness (Pearson's $r = 0.060$), and top ranking sites were observed across the entire range of species richness values. However, once these two variables were ranked, a relatively weak correlation could be observed (Pearson's $r = 0.657$) (Figure 15).

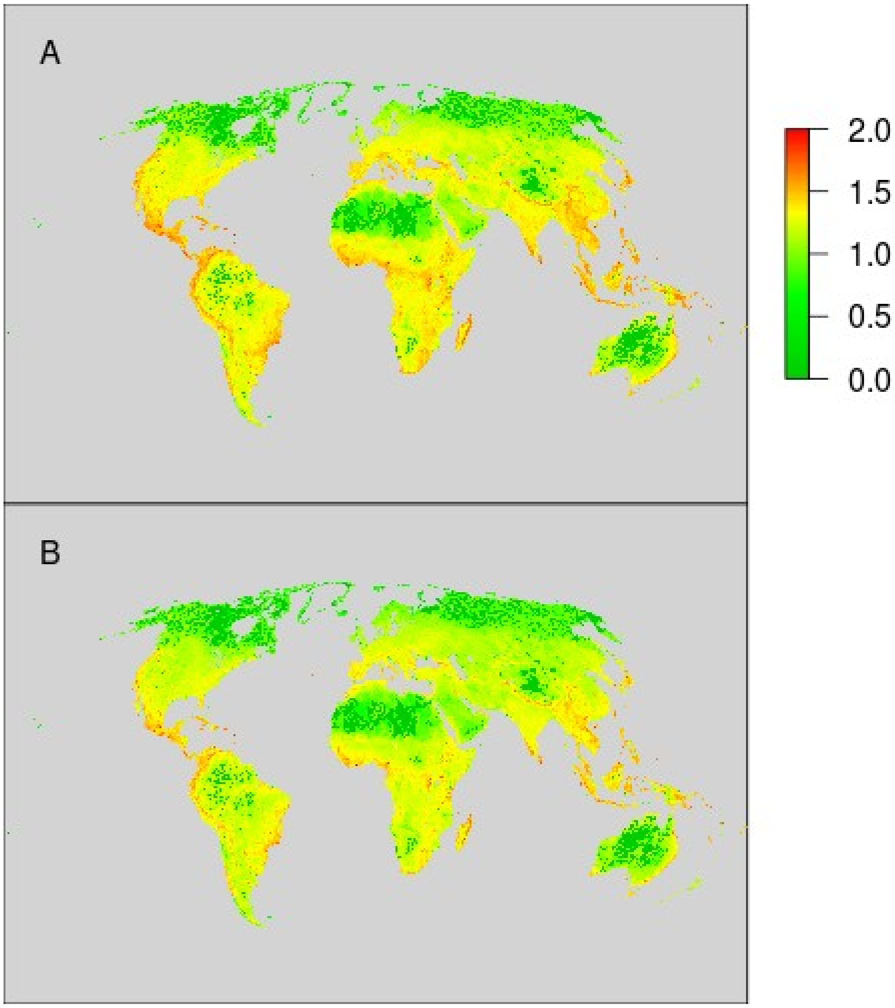

Figure 14: Conservation priority areas for terrestrial mammals. Map A accounts for anthropic pressure (Sanderson *et al*. 2002) and species specific prevalence (Rondinini *et al*. 2011b). Map B accounts for anthropic pressure and species specific prevalence as well as conservation status of species (IUCN 2013). The distribution of values was heavily weighted towards smaller values (see in text for a description of methods and results), therefore the data is presented with a base 10 log transformation for clarity.

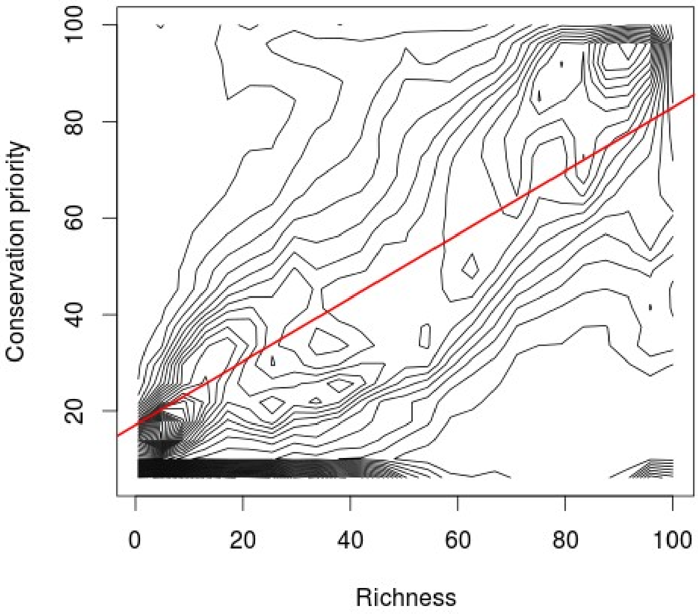

Figure 15: Ranked global richness versus Ranked conservation priority. Conservation priority accounts for anthropic pressure (Sanderson *et al*. 2002), species specific prevalence (Rondinini *et al*. 2011b) and species conservation status (IUCN Red List). Values for both axes were obtained sampling every pixel of two maps with a resolution of 10x10 Km; the values among the two axes were subsequently ordinally ranked and scaled from 1 to 100. The red line represents a linear model fitted on the values ($m = 0.66$, $c = 17.11$, $R^2 = 0.86$).

PSW values varied greatly both across biogeographic realms (Kruskal-Wallis rank sum test: $\chi^2 = 359933.1$, $p < 0.001$), and biomes (Kruskal-Wallis rank sum test: $\chi^2 = 625634.8$, $p < 0.001$). While the Oceanian and Indomalayan realms had the highest distribution of scores, with 75th PSW percentiles of 0.26 and 0.12 respectively these figures are not exceptionally informative from a conservation biology point of view, as they mostly represent the least valuable sites across the globe that would under ideal conditions not be considered for protection. (Figure 16).

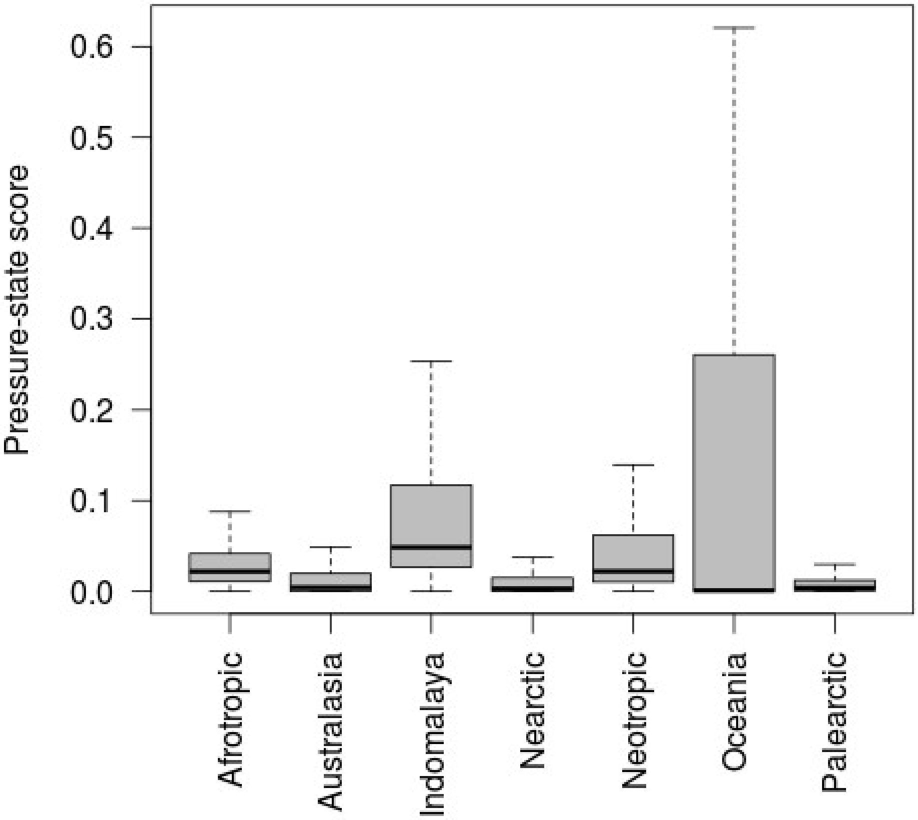

Figure 16: Distribution of global conservation priority scores for terrestrial mammals. A pressure-state approach accounting for anthropic pressure, extent of suitable habitat and conservation status of species was used to calculate conservation priority.

The CBD Aichi target 11 explicitly calls for 17% of the world's land and freshwater systems to be protected by 2020. Explicit recommendations have further been made that at least 10% of each ecoregion be protected. The top 17% scoring sites of each biome have been identified using both PS and PSW metrics (Figure 17). While the two approaches had an 91% concordance in identifying top priority sites, the overall distribution was significantly different

(Wilcoxon rank sum test: W = 17624341190, $p < 0.001$). The Nearctic was least impacted by approach type, with 97% of selected sites matching, while Oceania fared worst, with only a 71% match. All other realms had concordance values that ranged from 89% to 92%

Selecting sites imposing representation targets on biomes, however, strongly influenced potential maximum performance, with the global optimal solution covering 83% of the global cumulative conservation value and the biome representation solution covering 74%. The top 17% of sites using the PSW priority metric and imposing equal representation of biomes, for example, span across areas that rank from 22 to 100 on a priority scale from 0 to 100, and the lower quartile ranked less than 71. Selecting sites ignoring the equal representation requirement across biomes yielded a very different pattern, which is visible at a global scale (Figure 18), and for whom the distribution of values was significantly different (Wilcoxon rank sum test: W = 18111228918, $p < 0.001$). Biomes represent different proportions of the global land surface, ranging from mangroves that (excluding the Antarctic and inland Greenland) cover less than 0.2% of the world to deserts and xeric shrublands that cover 21% (Table 5). Protecting 17% of mangroves globally would equate to an area of less 39,000 km^2, a surface that would fit in a square of less than 200 km by side. When selecting the top 17% of global priority sites, on the other hand, we find that 42% of the mangrove biome is selected. Similarly, 81% of tropical and subtropical coniferous forests (0.5% of global land surface) were selected by the globally optimal approach. More specifically, there was an obvious trend with higher proportions of smaller biomes being present in the global top 17% (Figure 19).

At the resolution of these analyses (10 x 10 km), the distribution of proportion of protection in protected sites was strongly bimodal, with 27% of sites being less than 5% protected and a another 27% being more than 95% protected. However, 99% of the global protected surface was contained in sites with more than 5% protected area coverage. Statistical comparisons of the PSW values inside and outside protected areas would be of little significance for two reasons. Firstly the sample size of non protected areas is so large compared to that of protected areas and the distribution of values inside protected areas so wide (PSW values inside protected areas range from 0 to 100 and outside they range from 0 to 93.7, which corresponds to the 99.99993th percentile) that non significant differences would only be registered in the very unlikely event that protected areas were uniformly and almost methodically distributed. Secondly protected areas do provide forms of protection that tend to reduce anthropic pressure: a value that influences the PSW metric itself. This being said, if we define protected areas as sites with more than 5% protected surface, 19% of global priority sites are currently intersected by protected areas; these areas represent 11% of all currently existing protected areas. These figures however drop to 11% and 2% respectively if only sites with 100% protection are taken into consideration. This means there is moderate anthropic impact in close proximity to currently established conservation sites and that current protected areas give a substantial contribution to the reduction of these pressures.

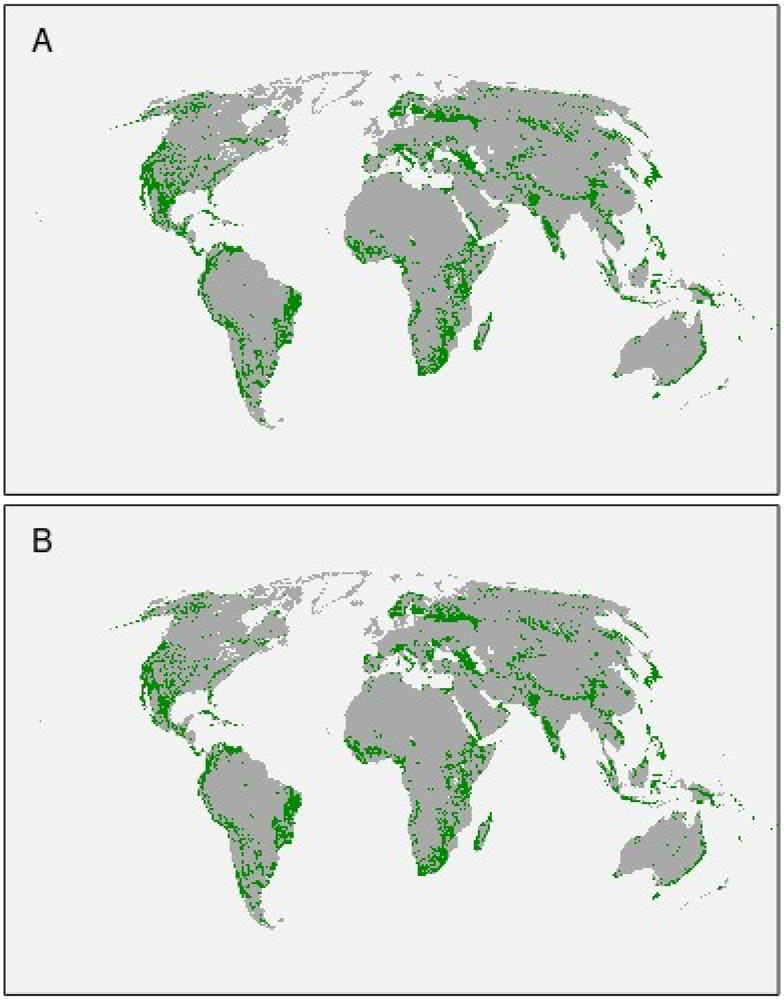

Figure 17: Highest priority conservation areas globally for terrestrial mammals. The top 17% scoring areas of each terrestrial biome (WWF 2001) was selected. Map A represents a priority selection accounting for anthropic pressure (Wildlife Conservation Society & Center for International Earth Science Information Network 2005) and species specific prevalence (Rondinini *et al*. 2011b). Map B represents a priority selection based on anthropic pressure and species specific prevalence as well as conservation status of species (IUCN 2013). The two approaches had a 9% mismatch in selected areas globally.

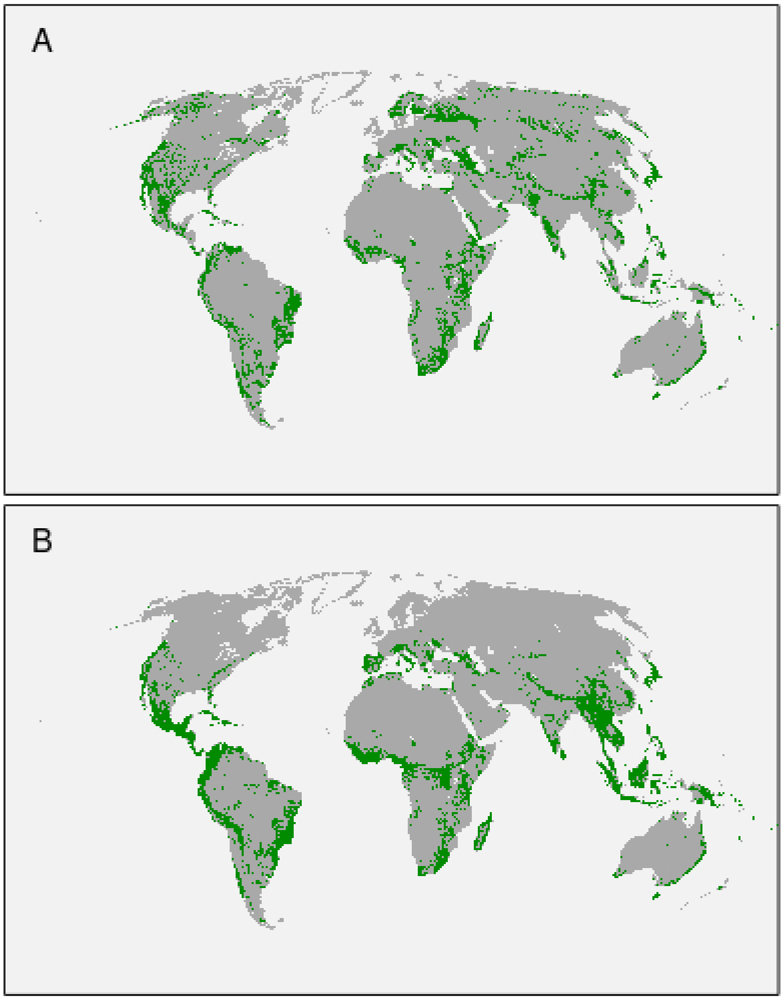

Figure 18: Highest priority conservation areas globally for terrestrial mammals. The top 17% priority areas were selected (A) accounting for and (B) disregarding the requirement to equally represent biomes in order to evaluate global impacts of equal representation targets. The two approaches had a 40% mismatch in selected areas globally.

Biome	Global surface (%)	Aichi target (%)	Global target (%)	Protection (%)
Boreal Forests/Taiga	11	17	0	7
Deserts & Xeric Shrublands	21	17	6	4
Flooded Grasslands & Savannas	1	17	18	10
Mangroves	< 1	17	42	6
Mediterranean Forests, Woodlands & Scrub	3	17	36	5
Montane Grasslands & Shrublands	4	17	23	4
Temperate Broadleaf & Mixed Forests	10	17	16	5
Temperate Conifer Forests	3	17	19	9
Temperate Grasslands, Savannas & Shrublands	7	17	5	2
Tropical & Subtropical Coniferous Forests	1	17	81	2
Tropical & Subtropical Dry Broadleaf Forests	2	17	52	5
Tropical & Subtropical Grasslands, Savannas & Shrublands	15	17	14	6
Tropical & Subtropical Moist Broadleaf Forests	15	17	49	7
Tundra	7	17	< 1	14

Table 5: Terrestrial biomes with: proportion of global land surface covered; proportion surface identified as priority following equal representation targets of 17% protection for each biome; proportion surface identified as priority ignoring equal representation targets; current protection level. Site priority was calculated using a pressure state approach accounting for anthropic pressure, species specific prevalence and conservation status. See in text for details on methods used to calculate the priority metric.

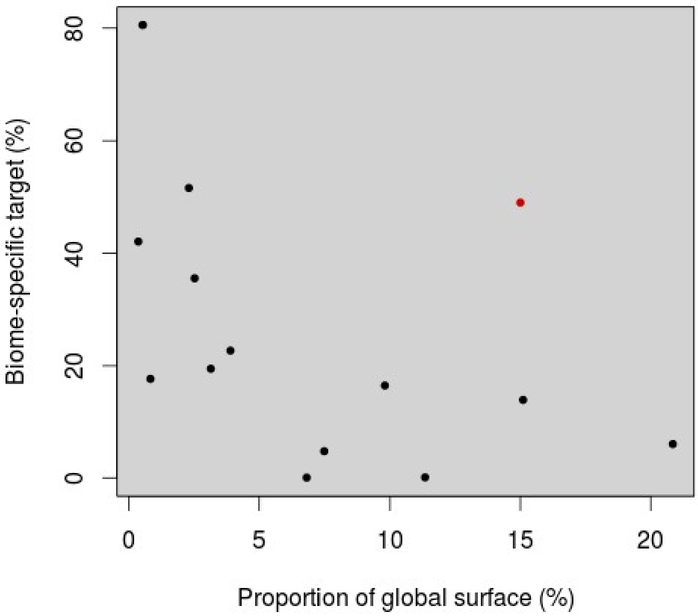

Figure 19: Proportion of global surface covered by each biome v.s. proportion of biome classified as conservation target under a global pressure-state ranking. The ranking accounts for anthropic pressure, species specific prevalence for terrestrial mammals and conservation status of species. The top 17% ranking sites globally were selected. With the exception of tropical & subtropical moist broadleaf forests (red), biomes with a smaller surface area generally were identified as having proportionally higher priorities.

3.4 – Impact of climate change on protected areas

3.4.1 – Assessing the impact on Carnivores and Ungulates

Of the 440 species for who projections of bioclimatic envelopes had been developed, only 418 (Table 6) had protected areas that intersect both their current range and climatic envelope. The climatic projections were also available outside of current ranges and species range expansion have been modelled in Visconti *et al*. (submitted; but also see § 2.4.1). Range expansion is not a statistical certainty, therefore under the precautionary principle would require for protected areas to be planned accounting for potential loss in efficiency but not for uncertain gain.

Forecasts indicate that climate driven habitat loss in protected areas is predominant under both the BAU and CCH scenario (Figure 20). By the year 2050, the median proportional loss of suitable habitat for all species was -4% ± 6 MAD under the BAU scenario, and -2% ± 2 MAD under the CCH scenario. Only a small number of species were forecasted to gain suitable habitat: 43 species under BAU and 48 species under CCH, (10% and 12%), with maximum gains of 7% and 9% respectively. The choice of development scenario had significant influence on the proportion of protected suitable habitat lost (paired Wilcoxon signed rank test: $p < 0.001$ for all tests after Bonferroni correction): this was seen for all years and taxonomic grouping, both when comparing proportional change against 2010 levels (Table 7) and when comparing adjacent decadal values.

Order	**Family**	**n**
Carnivora	Ailuridae	1
	Canidae	34
	Eupleridae	5
	Felidae	36
	Herpestidae	32
	Hyaenidae	4
	Mephitidae	9
	Mustelidae	48
	Nandiniidae	1
	Prionodontidae	2
	Procyonidae	12
	Ursidae	7
	Viverridae	30
Cetartiodactyla	Antilocapridae	1
	Bovidae	112
	Camelidae	3
	Cervidae	41
	Giraffidae	2
	Hippopotamidae	2
	Moschidae	5
	Suidae	7
	Tayassuidae	4
	Tragulidae	4
Perissodactyla	Equidae	7
	Rhinocerotidae	3
	Tapiridae	4
Proboscidea	Elephantidae	2

Table 6: taxonomy and number of species for whom impact of climate change driven habitat loss in protected areas was analysed. Only species that had suitable habitat in protected areas in their range were included in the analysis.

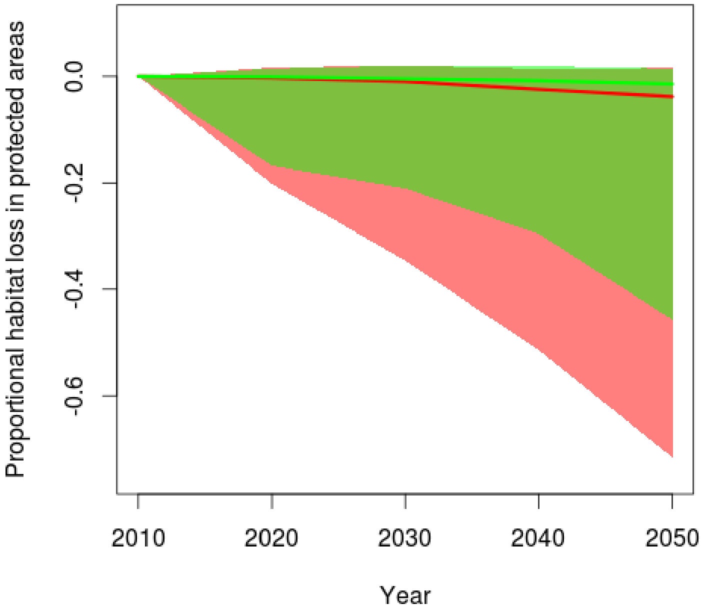

Figure 20: Forecasted habitat loss in protected areas for carnivores and ungulates under two socioeconomic development scenarios. Habitat loss was calculated as the climate driven proportional reduction in suitable area compared to 2010 levels: The development scenarios assumed no change in land use within protected areas and came to the Roads from Rio +20 report: Business As Usual (red); Consumption Change (green). The straight lines represent median change; the shaded cones represent the 95th percentile confidence interval range.

Wilcoxon signed rank test – BAU vs CCH				
	2020	2030	2040	2050
carn. & ung.	p < 0.001	p < 0.001	p < 0.001	p < 0.001
carnivores	p < 0.001	p < 0.001	p < 0.001	p < 0.001
ungulates	p < 0.001	p < 0.001	p < 0.001	p < 0.001

Wilcoxon rank sum test – Carnivores vs Ungulates				
	2020	2030	2040	2050
BAU	p = 0.035	p = 0.053	p = 0.012	p = 0.014
CCH	p = 0.040	p = 0.045	p = 0.054	p = 0.014

Table 7: Statistical tests comparing the distributions in the proportion of habitat loss within protected areas for two socioeconomic development scenarios of the Roads from Rio +20 report (van Vuuren *et al*. 2012): Business As Usual (BAU) and Consumption Change (CCH). Tests were run for the changes forecasted at the years 2020, 2030, 2040 and 2050; and stratified by taxonomy and scenario. Where data pairing was possible, the Wilcoxon signed rank test was applied, otherwise the Wilcoxon rank sum test was used.

Taxonomy generally influenced the amount of habitat lost in protected areas, but results were borderline and in two cases not significant when comparing carnivores vs ungulates. Ungulates lose more suitable habitat than carnivores, with a median habitat loss of 5% ± 8 MAD and 3% ± 4 MAD respectively under the BAU scenario, 2% ± 3 MAD and 1% ± 2 MAD under the CCH scenario (Table 7, Figure 21). Taxonomic Familiy appears to substantially influence habitat loss: The Eupleridae for example are expected to lose a median of 93% of their protected habitat under the BAU scenario and 59% under the CCH mitigation scenario; Elephanitdae a median of 42% and 21% respectively under the two scenarios and Rhinocerotidae 35% and 11% (Figure 22). A Kruskal Wallis rank sum test, however, revealed that differences were significant only under the CCH scenario (BAU: chi-squared = 37.4661, p = 0.068; CCH: chi-squared = 40.4362, p = 0.036).

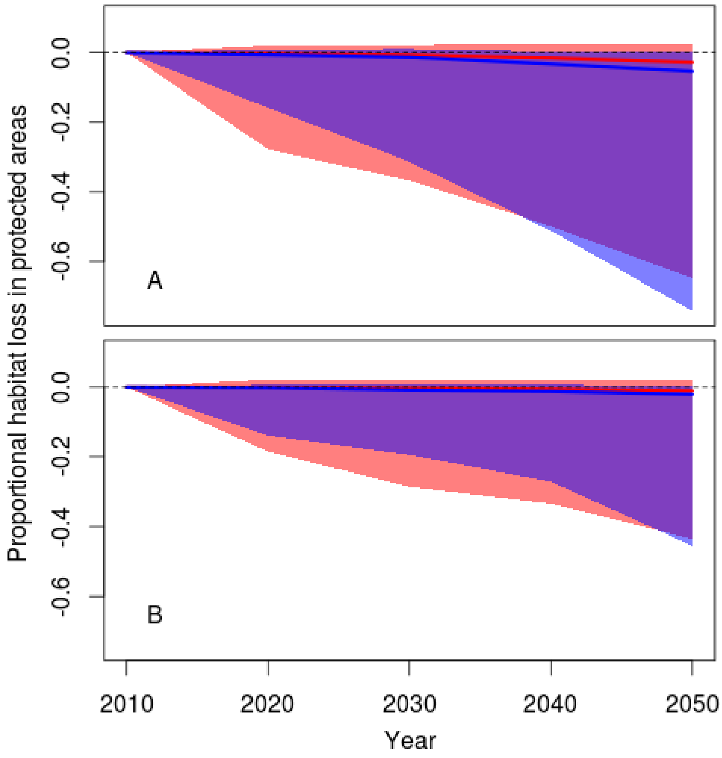

Figure 21: Forecasted habitat loss for carnivores (red) and ungulates (blue) under two socioeconomic development scenarios from the Roads from Rio +20 report (van Vuuren *et al*. 2012): Business As Usual (A); Consumption Change (B). The straight lines represent median change, the shaded cones represent the 95th percentile confidence interval range.

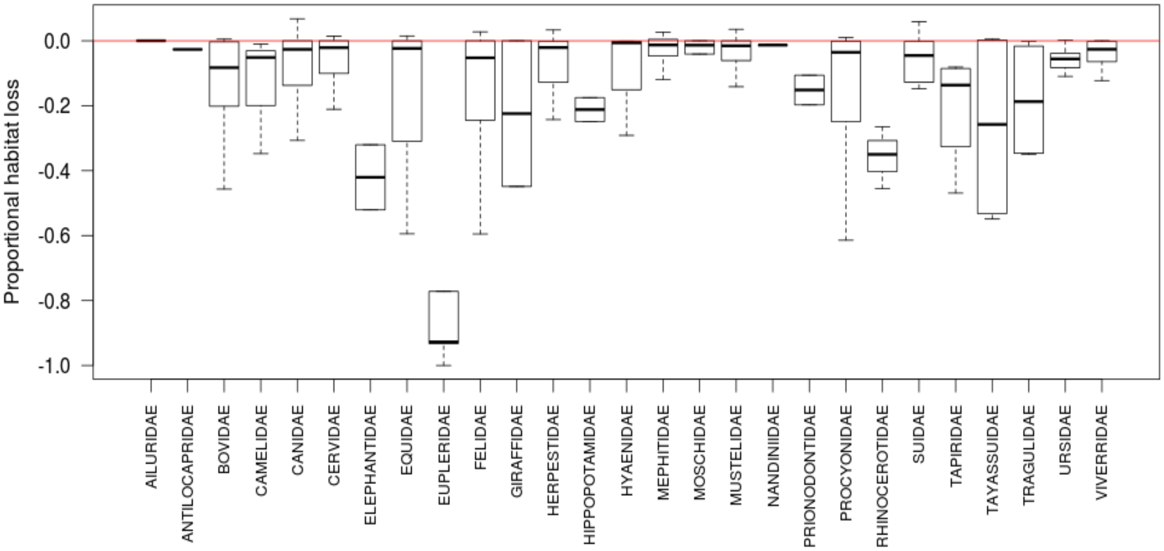

Figure 22: Proportional loss of habitat in protected areas for 27 mammalian Families by the year 2050. Bioclimatic envelope models and habitat suitability models were used to measure suitable surface area in protected areas with IUCN classification Ia-IV. Habitat loss was measured accounting for modelled changes in climate according to a Business As Usual development scenario.

When species were assigned to their predominant biogeographic realm, it was noted that geographic distribution greatly influenced impact (Kruskal Wallis rank sum test: chi-squared = 31.5208, $p < 0.001$) (Figure 23). The Nearctic and Palearctic did not differ significantly (Wilcoxon rank sum test: $W = 1546$, $p = 0.233$), and with a combined median loss of 1% ± 3% MAD of suitable habitat in protected areas were less impacted than other biogeographic regions. This difference is particularly evident when observing the higher quantiles, as 95% of the species analysed lose less than 31% of their protected range, while in the Afrotropic, Indomalayan and Neotropic realm this figure is of 62% 44% and 58% respectively.

Tropical regions had a high concentration of species that were highly impacted, and the decline in protected habitat by the year 2050 in the tropics was significantly higher than what vould be observed outside both for the BAU (Wilcoxon rank sum test W = 15236.5, p = 0.0001) and the CCH (Wilcoxon rank sum test W = 15121, p < 0.0001) scenario (Figure 24). Under the BAU scenario, median decline in the tropics was of 6% ± 8 MAD compared to 2% ± 3 MAD outside of the tropics. For the CCH scenario these declines were of 2% ± 3 MAD and 1% ± 2 MAD respectively.

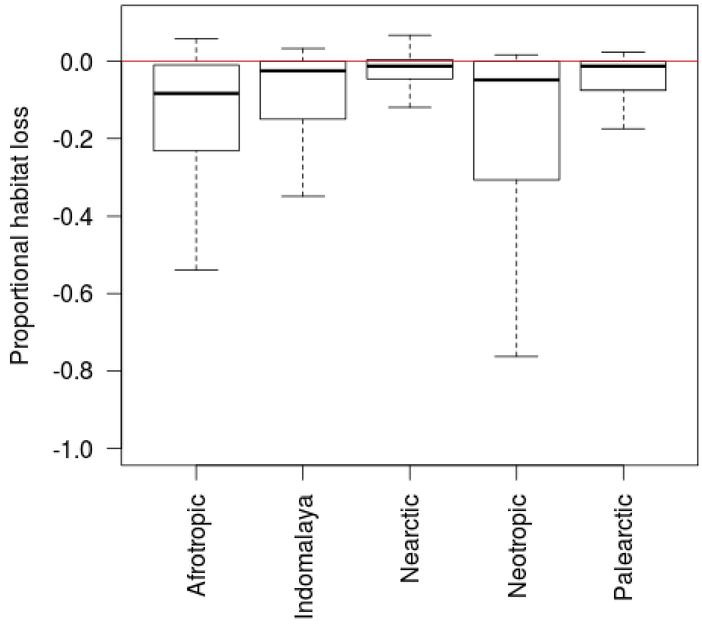

Figure 23: Proportional loss of habitat in protected areas within 5 biogeographic realms by the year 2050. Bioclimatic envelope models and habitat suitability models were used to measure suitable surface area in protected areas with IUCN classification Ia-IV for terrestrial carnivores and ungulates. Habitat loss was measured accounting for modelled changes in climate according to a Business As Usual socioeconomic development scenario (van Vuuren *et al*. 2012).

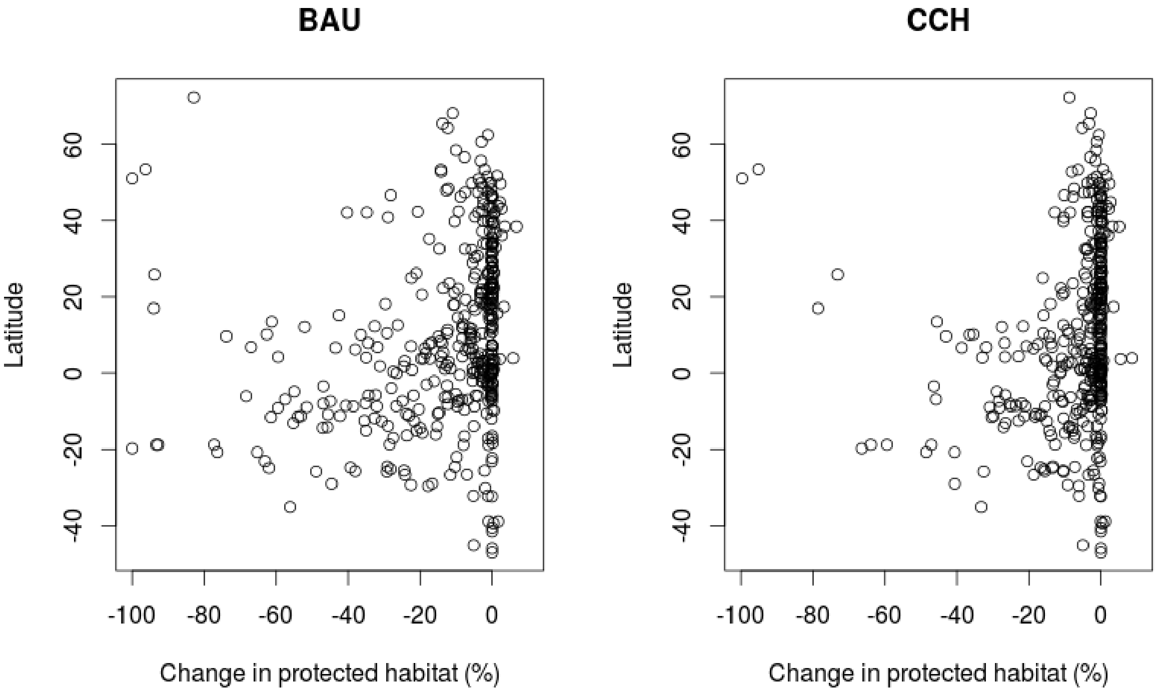

Figure 24: Proportional loss of suitable habitat in protected areas under a Business as Usual (BAU) and Climate Change (CCH) scenario by the year 2050 (van Vuuren *et al*. 2012). Bioclimatic envelope models and habitat suitability models were used to measure suitable surface area in protected areas with IUCN classification Ia-IV. See in text for further details.

Extinction risk, did not have a significantly impact on proportional habitat loss within protected areas when tested across all IUCN Red List categories (Kruskal Wallis rank sum test : chi-squared = 6.6125, p = 0.158). However, there was a significant difference between the distribution of values for least concerned species when comparing these to the cumulative distribution of all other classes (Wilcoxon rank sum test: W = 24189.5, p = 0.027) (Figure 25). No significance was found when shifting the threshold along the IUCN Red List category gradient (e.g. comparing LC and NT against VU, EN and CR species).

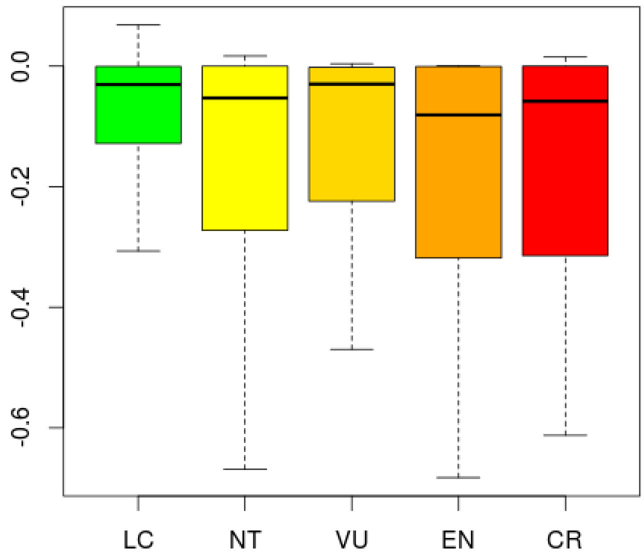

Figure 25: distribution of proportional loss of suitable habitat in protected areas for carnivores and ungulates by IUCN Red List category. Proportional loss of habitat in protected areas was measured comparing modelled 2050 suitable habitat availability to 2010 values and was exclusively climate driven, as land use was maintained unchanged in the models. Values refer to the Business As Usual socioeconomic development scenario from the Roads from Rio+20 report (van Vuuren *et al*. 2012), which is tied to IPCC SRES 4 scenario A1B.

3.4.2 – Extrapolating habitat loss trends to all mammals

Despite numerous factors were revealed to influence climate-induced habitat loss within protected areas, the attempt to model a response from a combination variables known to influence biodiversity revealed low predictive power. The random forest analysis could only explain 39% of the response when modelling habitat loss within protected areas, and 42% when modelling climatically-induced (i.e. ignoring land-use conversion) habitat loss within species range. Overall, no single variable contributed substantially more than the others (Table 8).

	% Inc. MSE	Inc. Node Purity
Order	4.2	0.1
Family	13.4	2.4
Red List Status	4.8	0.4
Threatened status	6.7	0.1
Weight	15.6	1.4
Realm	4.0	0.3
Latitude	16.3	1.8
Longitude	8.7	0.9
HII 90th percentile	12.1	0.7
HII 50th percentile	13.7	0.9
HII mean	11.6	0.6
Range size	9.4	1.1
Protected area size	11.2	0.7

Table 8: Thirteen variables used as determinants in a Random Forest analysis to infer climate-induced habitat decline within species range by 2050 for carnivores and ungulates, under a Business As Usual scenario. The variables represent a set or taxonomic (Order, Family), geographic (central Latitude and Longitude values within the specie's range, Realm), intrinsic (Weight), extrinsic (90th, 50th percentile and mean value of the Human Influence Index in the specie's range; Range size; Protected area size within the specie's range) and mixed (IUCN Red List status; Threat status: threatened v.s. non threatened) influences. Five-hundred trees were run with 4 variables randomly selected at each split. Percent increase in Mean Standard Error represents the mean percentual increase in prediction error across all trees if the variable is removed; it is a weight of variable importance. Increase Node Impurity represents the increase in residual sum of squares, if the variable is excluded.

The only clearly observable trend that could aid in a predictive approach was between species range size and climate-driven habitat loss (Figure 26): the highest values of proportional habitat loss within the entire species range were associated with the smaller species range sizes. While the heteroscedastic nature of this particular trend did not allow for an accurate prediction of habitat loss, it could be used to gain insight and estimate, for any specific range size, the highest loss a species could potentially be exposed to. The GLM fitted on the 90th percentile of loss in range size bins had higher residuals than an equivalent GLM fitted on all values (Paired Wilcoxon signed rank test: $V = 87571$, $p < 0.001$). Both approaches tended to overestimate proportional loss in carnivores and ungulates compared to the data from climate niche modelling. However, using the GLM fitted on all data resulted in predictions being 7% higher (median on the error: 6% ± 7 MAD), while using the 90th percentile gave predictions 46% higher (median on the error: 37% ± 25 MAD) (Figure 27). While the distribution of errors was too wide to make real predictions (Figure 28), using the 90th percentile GLM leads to underestimations of habitat loss for only 24 species (6% of the total). It was therefore possible to identify range size thresholds above which levels of habitat loss exceeding a maximum desired threshold were unlikely (Figure 29).

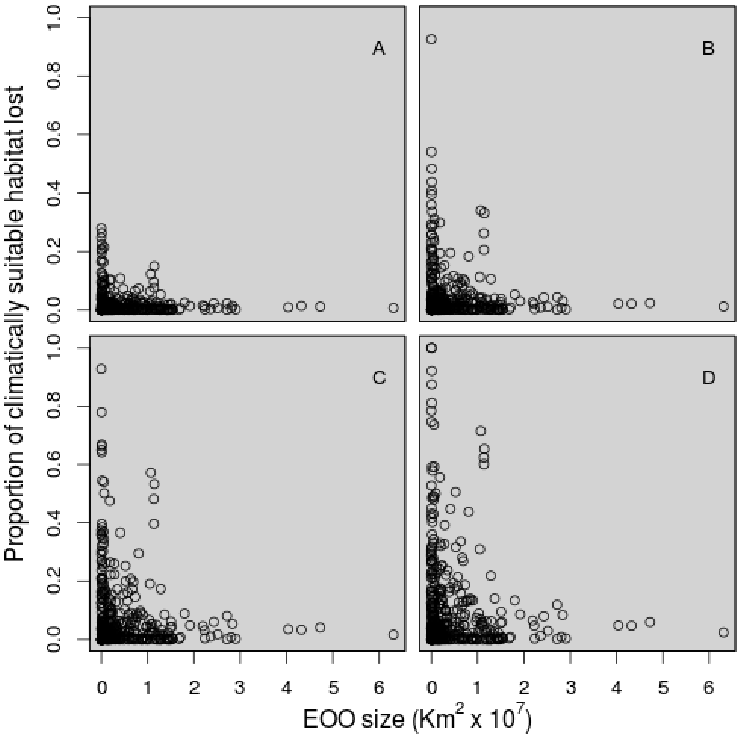

Figure 26: Range size versus proportion of climatically suitable habitat lost by the year 2020 (A), 2030 (B), 2040 (C) and 2050 (D) under the Rio +20 (van Vuuren *et al*. 2012) Business As Usual forecast scenario.

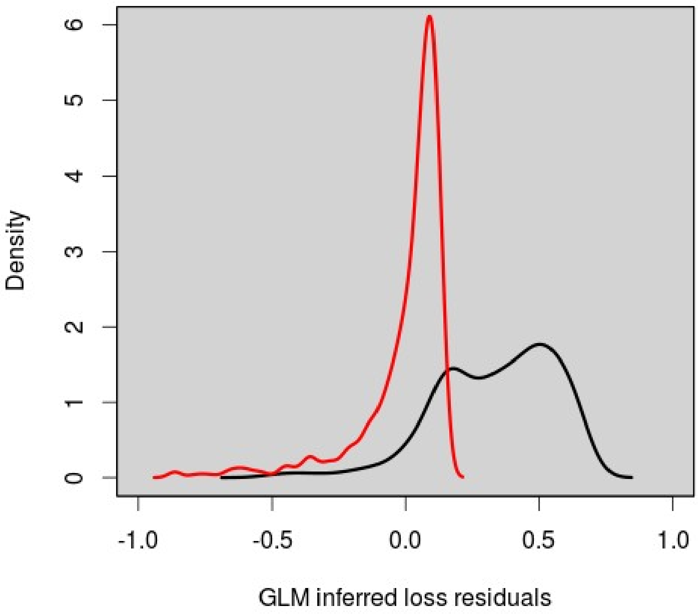

Figure 27: Distribution of residuals for gamma-link generalised linear models fitted on proportional loss of climatically suitable habitat within species range by the year 2050 and current range size. The red line represents a model fitted on 418 carnivore and ungulate species; the black line represents a model fitted on the 90th percentile of forecasted loss in range size bins with a bandwidth of 2 million km^2.

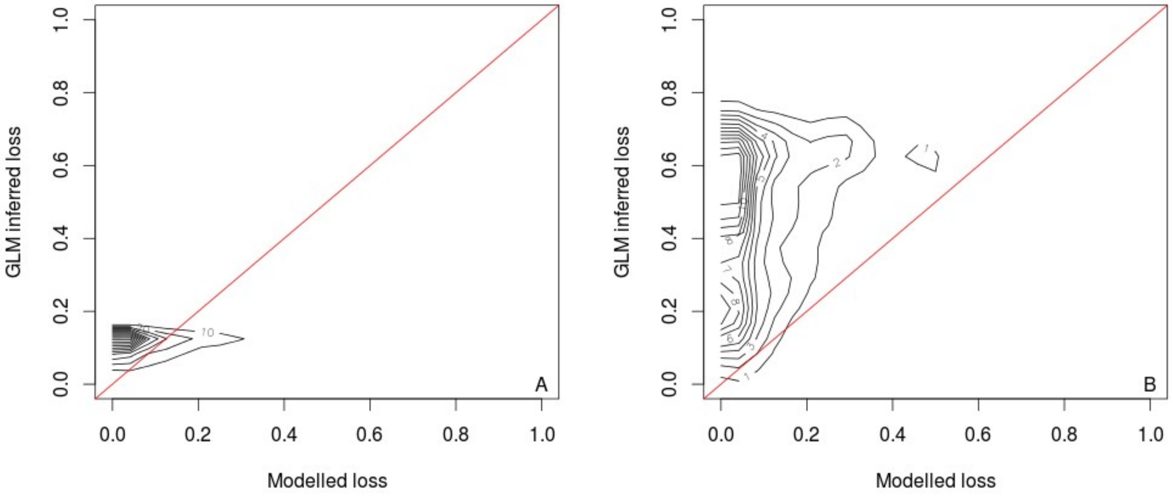

Figure 28: Two-dimensional density kernel of forecasted proportional loss in suitable protected habitat by the year 2050 under a Business As Usual versus glm-inferred values. Modelled loss was measured by observing the forecasted reduction in climatically suitable area within each specie's range. The GLM was fitted on the (A) all values and (B) the 90th percentile of the forecasted values across range size bins for 418 carnivore and ungulate species. The region under the diagonal line indicates cases where the GLM underestimates habitat loss.

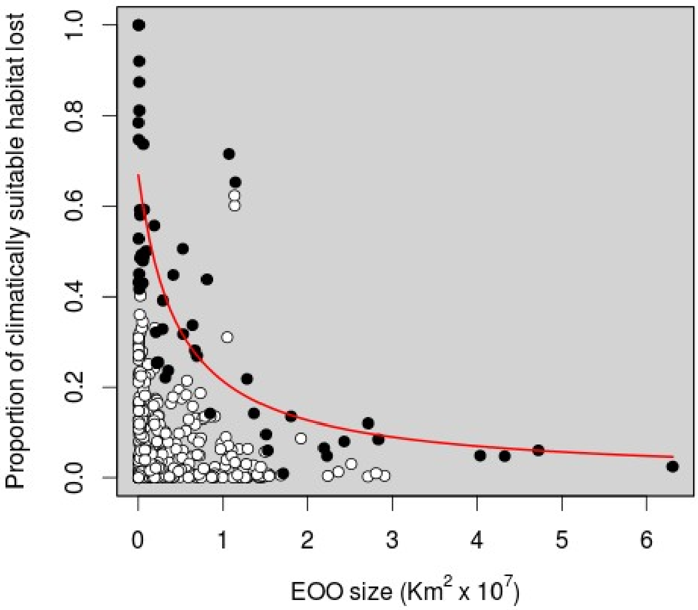

Figure 29: Range size versus the proportion of climatically suitable habitat lost by the year 2050 under a Business As Usual scenario. Climate envelop models were developed for 418 carnivore and ungulate species and used to calculate habitat loss in protected areas. The black points represent species falling in the 90th percentile of forecasted loss by range size with a bandwidth of 2 million km^2. The red line is a generalised linear model with a gamma link function fitted on the 90th percentile.

As the distribution of EOOs across all mammals is heavily skewed towards smaller sizes (Shapiro Wilk normality test: $W = 0.4771$, $p < 0.001$) and species with small EOOs tend to be heavily impacted by climate change, the vast majority of mammals have a predisposition to be at high potential risk of habitat loss. Half of all terrestrial mammals risk losing more than 64% of their habitat, and only 5% face a maximum risk of less than 25% loss. Both taxonomic order and biogeographic realm had a significant influence on the distribution of maximum potential loss (Kruskal Wallis rank sum test: biogeographic realm chi-squared = 302.0821, p <

0.001; taxonomic order chi-squared = 475.2294, p-value < 0.001) when the distribution of species was measured at the year 2050 under the BAU scenario.

When using maximum potential loss as a selection criteria to calculate robust species richness, it was observed that only a small portion of terrestrial mammals was highly robust to climate-induced habitat loss (Figure 30). The intercept of the GLM indicates that as species range approaches zero, the maximum expected loss rises up to 67%, meaning that on average 33% or more of each specie's ranges is safe from climate change effects. If losses up to 50% 30% and 10% are deemed acceptable for conservation planning purposes, median remaining global richness dropped to 90% ± 11% MAD, 61% ± 26% MAD and 8% ± 10% MAD respectively (Table 9) . Biogeographic regions overall responded differently to this effect (Figure 31). The Oceanian biogeographic region for example contains no species with moderate or high expected tolerance to climate impacts, while realms with wider surfaces and larger contiguous land masses (Nearctic, Palearctic, Neotropic and Paleotropic) appeared to on average contain species more robust to climate impacts. The Oceanian biogeographic region's response was however peculiar and non representative, probably influenced by its small extent (487 sites as this resolution), low mammalian richness (maximum richness = 6, median richness = 1) and presence of species with very reduced range sizes.. A similar, yet more moderate trend can also be observed in the Australasian realm, which is itself rich in small-ranged species.

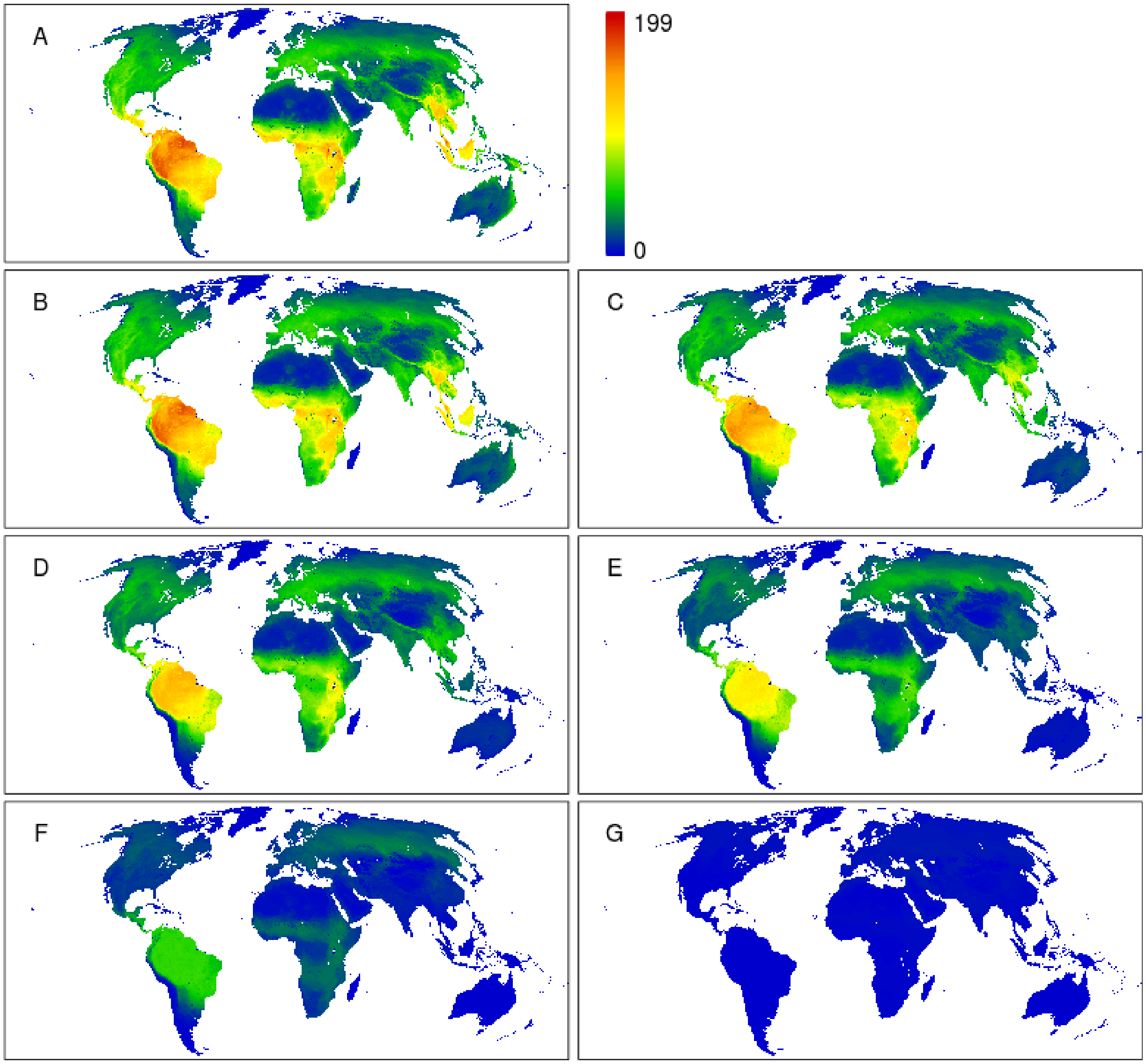

Figure 30: Robust richness, accounting for risk of habitat loss by the year 2050 due to climate change. Values range from 0 to 199. The first tile (A) represents mammalian richness at a 10 Km^2 resolution; the following tiles represent richness including only species that are expected to lose no more than (B) 60%, (C) 50%, (D) 40%, (E) 30%, (F) 20%, (G) 10% of climatically suitable habitat.

Median robust species richness (% ± MAD)

	minimim remaining habitat					
	60%	50%	60%	70%	80%	90%
World	99 ± 2	90 ± 11	79 ± 18	61 ± 26	35 ± 18	6 ± 10
Afrotropic	98 ± 3	90 ± 8	77 ± 13	53 ± 17	27 ± 10	3 ± 2
Australasia	94 ± 8	67 ± 26	48 ± 22	23 ± 11	0 ± 0	0 ± 0
Indomalaya	96 ± 6	83 ± 14	58 ± 17	31 ± 17	11 ± 9	4 ± 4
Nearctic	100 ± 0	92 ± 10	86 ± 16	66 ± 24	42 ± 16	8 ± 9
Neotropic	97 ± 3	89 ± 10	78 ± 15	65 ± 14	39 ± 9	0 ± 0
Oceania	0 ± 0	0 ± 0	0 ± 0	0 ± 0	0 ± 0	0 ± 0
Palearctic	100 ± 0	94 ± 9	86 ± 15	73 ± 23	40 ± 19	15 ± 5

Table 9: Species richness robust to habitat loss from climate change. Species richness was calculated at a resolution of 10 km^2 on a Mollweide equal area projection selecting only species that were expected to maintain more than a minimum remaining habitat threshold by the year 2050 under a Business As Usual scenario. Oceania appears to be extremely impacted due to the low number of sites (487 planning units), low mammalian richness (median = 1) and reduced species ranges.

Although the range of species richness within and outside of protected areas was very similar (0 to 190 within protected areas, 0 to 199 outside protected areas), the distribution of values in the two registered as significantly different to a Wilcoxon rank sum test (W = 108172613491, $p < 0.001$), with the former having a slightly higher median score than the latter (46 ± 28 MAD compared to 41 ± 28 MAD). Protected areas from the different biogeographic regions follow heterogeneous trends in safe richness lost at different thresholds of robustness (Figure 32); The Australasian and Indomalayan realms, for example, show a greater decrease in protected biodiversity at lower robustness requirements. In all regions, however there is a dramatic loss of protected richness well before robustness values of 80% are demanded. The response in the Oceanian region is strongly bimodal due to the low species richness, as only 12% of the protected area surface has a mammalian richness higher than 1. Comparing the ranked PSW priority of each site against the proportion of richness lost at increasing robustness levels, it was seen that richness was lost proportionally faster in low priority sites than in high priority ones (Figure 33).

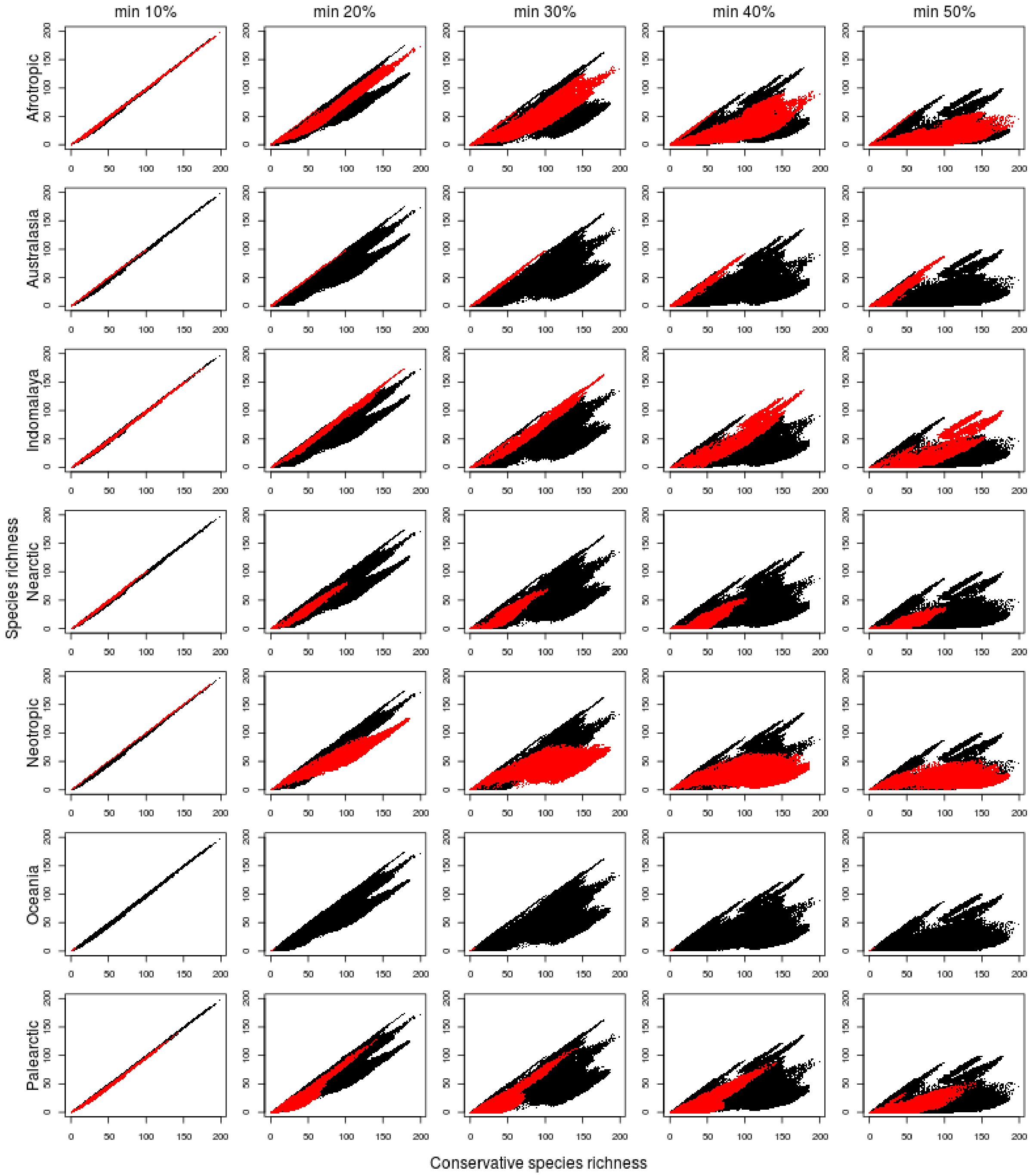

Figure 31: distribution of absolute richness v.s. robust richness of terrestrial mammals in biogeographic realms. Robust richness was calculated accounting for species that were expected to maintain at least a minimum proportion of their original suitable habitat in protected areas. Species specific robustness was calculated by fitting a GLM on the 90th percentile of modelled habitat loss for carnivores and ungulates; the GLM was then used to extrapolate values for all mammals. The black shadow represents the overall global response, the red shadow represents specific biogeographic response.

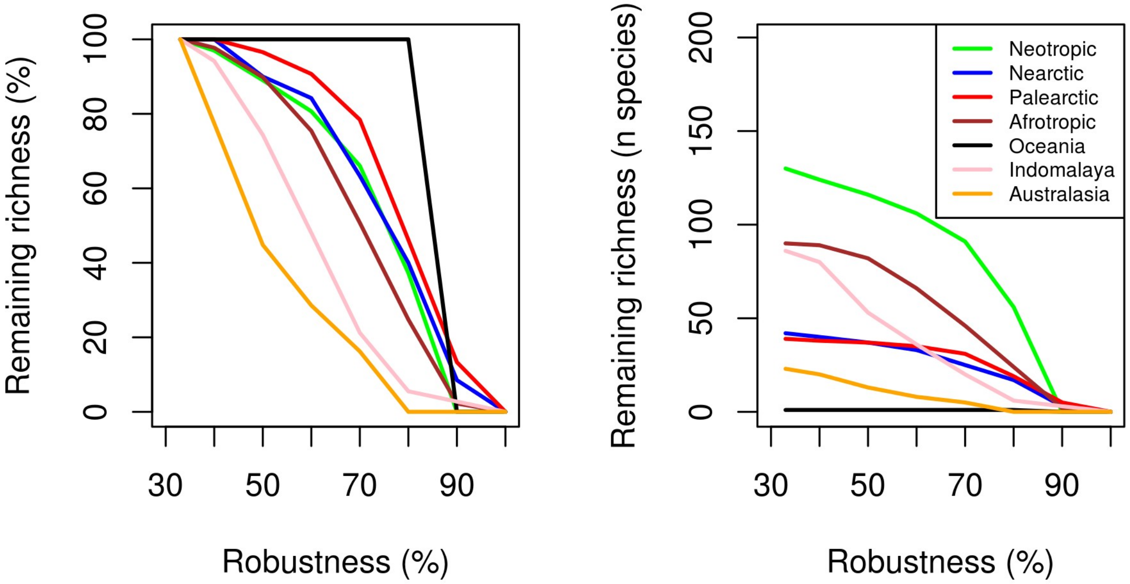

Figure 32: Proportional (left) and absolute (right) richness of terrestrial mammals in protected areas at different robustness thresholds. Robustness is the estimated minimum proportion of original suitable habitat remaining by the year 2050 under a Business As Usual socioeconomic development scenario. Protected areas are expected to prevent anthropic land use conversion, however suitable habitat may still decline due to climate change. A GLM linking Extent Of Occurrence and maximum expected loss was developed for carnivores and ungulates from forecasted climate envelope models; the GLM was then used to extrapolate expected loss for all terrestrial mammals. Robust richness was calculated by only including species with a minimum expected remaining habitat proportion corresponding to the desired robustness.

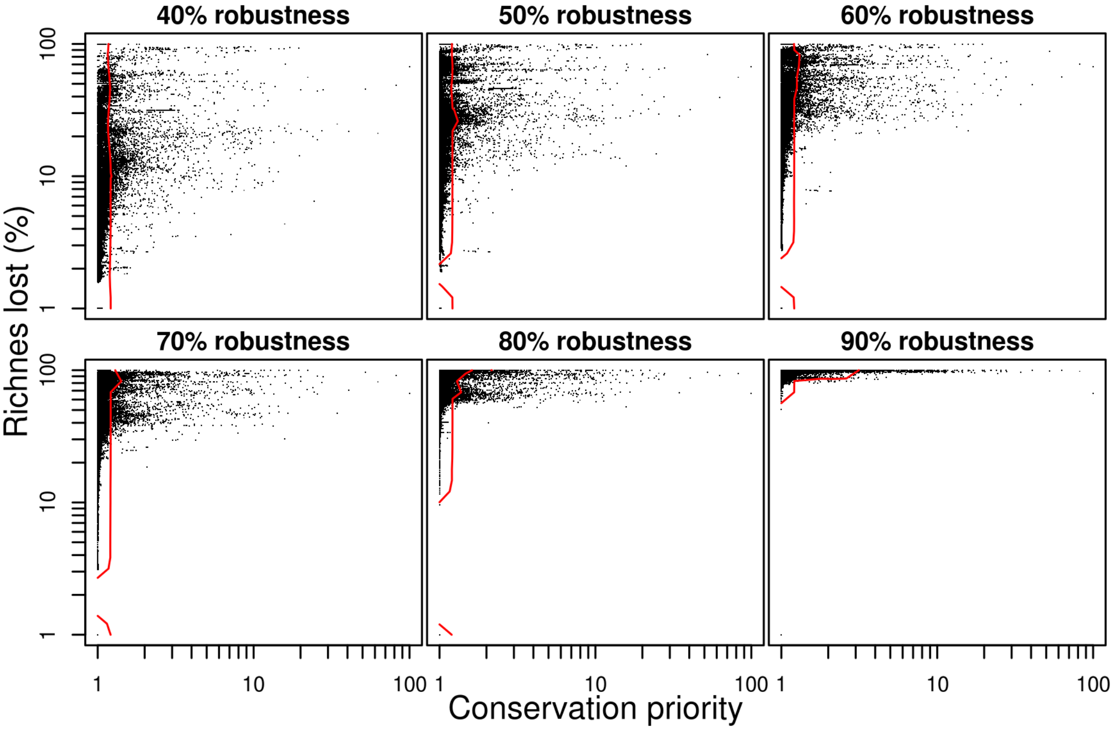

Figure 33: Proportional richness in protected areas for terrestrial mammals versus conservation priority at different robustness thresholds. Species robustness is the estimated by the minimum proportion of original suitable habitat remaining by the year 2050 under a Business As Usual socioeconomic development scenario. Protected areas are expected to prevent anthropic land use conversion, however suitable habitat may still decline due to climate change. A GLM linking Extent Of Occurrence and maximum expected habitat loss was developed for carnivores and ungulates from forecasted climate envelope models; the GLM was then used to extrapolate values for all terrestrial mammals. Robust richness was calculated by only including species with a minimum expected remaining habitat proportion equal or bigger than the desired robustness level.

4 – DISCUSSION

4.1 – General considerations

The practice of conservation biology has greatly evolved from its ancestral, first iterations dealing with spirituality or aesthetics (Primack 1998). While historically the focus has always been on the *ad hoc* protection of desired features or species, the modern iteration recognises wider benefits of natural systems to human society as a whole. Wide-spread impacts of desertification (D'Odorico *et al*. 2013) and deforestation (Laurance *et al*. 2001), for example, can already be observed globally and the impacts of biodiversity loss on ecosystem sustainability is a recognized threat (Johnson *et al*. 2007; Curtsdotter *et al*. 2011). Furthermore, all trends and forecasts indicate that in the near future the principal drivers of these events will likely exacerbate due to the increase in human population (van Vuuren *et al*. 2012). Conservation practitioners routinely have to deal with a number of issues that slow down hinder or downright prevent a complete and informed decision making process. Among the greatest issues faced we can identify the lack of resources, whether monetary (Balmford *et al*. 2000; Moore *et al*. 2004), time (Brooks *et al*. 2002) or knowledge (Funk & Richardson 2002).

As the scarcity of resources available for conservation interventions is a well established issue, multiple strategies and approaches have so far been devised to maximise the impact of partaken actions. These range from optimizations of the monetary cost to benefit ratio of conservation actions necessary to achieve the desired objectives (Moore *et al*. 2004; Naidoo *et al*. 2006; Naidoo & Iwamura 2007) to more complex considerations on the probability of success (Eklund *et al*. 2011), hidden cost rise and long-term viability (Araújo & Cabeza 2004; Williams *et al*. 2005). In many ways systematic conservation planning (Margules & Pressey 2000) has helped shape modern conservation science in this respect creating a distinct change in approach from previous *ad hoc* strategies. In order to take full advantage of all of these approaches, however, there is a strong need for information that is unbiased, free of errors, obtained with uniform efforts across all features of interest, and generally of adequate quality (Boitani *et al*. 2011). The main types of data used in conservation planning can be distinguished in three classes: Environmental (extrinsic) data, biological traits (intrinsic) and presence.

Significant progress has been made in the past years on improving the availability and quality of spatial data: advancements in computing, satellite imagery and international collaboration now allow for fine scaled representations of the entire world surface to be easily available. Global maps of land-use, climatic variables and anthropic presence, among the others, are now routinely used in numerous disciplines (e.g. Sanderson *et al*. 2002; European Space Agency & Universite Catholeque de Louvain 2009).

Databases on intrinsic data (e.g. Jones et al. 2009) are not technically challenging, but their development has been slow, possibly due to the relatively limited scientific research applied to taxonomically wide groups and sparse knowledge on species that are rare or from remote places. While some initiatives have been made and are under way to compile such databases, the quality of data is to a certain extent still relatively low, and the gaps in data quite wide spanning (Gonzalez-Suarez *et al*. 2012). This ultimately means that researchers often have to resort to imputing, interpolating or extrapolating data when this was necessary (e.g. Di Marco *et al*. 2012).

Data on presence has been approached slightly differently from the previous data sources. As species presence is the result of the dynamic interaction between the species and the environment, it can be used in two principal manners. Firstly, it can be used to determine a specie's status in response to natural or anthropogenic external pressures such as for example by using IUCN Red List criteria (IUCN Species Survival Commission 2012). Alternatively, it can be studied in order to determine the specie's ecological preferences and tolerances in relation to the environment, and therefore be used to assess the effect of potential conservation interventions. These two approaches often require data with widely different characteristics. While institutions such as the Global Biodiversity Information Facility (GBIF 2014) or citizen science initiatives (e.g. Sullivan *et al*. 2009) do attempt to compile and provide collections of species presence data, these strongly suffer from spatial, temporal and taxonomic bias (Boakes *et al*. 2010), and are far from a complete and comprehensive representation of multi-species distribution. Furthermore, the process of collecting empirical presence data is an activity that consumes both financial resources and time, and has to be balanced with the need to take hasteful conservation actions (Funk & Richardson 2002).

While past applications of large scale conservation analyses had little alternatives to the use of available coarse scale or generalised data (Rodrigues *et al*. 2004), the use of modelling approaches has been shown to be a valid option and hold great potential for data refinement (Funk & Richardson 2002). For example, (Rondinini *et al*. 2011b) found the mean proportion of suitable habitat in specie's range to be 56.67% and suggested ESH could be used as an alternative to EOO and area of occupancy in order to calculate conservation status under IUCN Red List criteria, while (Crooks *et al*. 2011) used HSMs to calculate habitat fragmentation and connectivity for carnivores.

One of the main drawbacks of empirical data is that it may have an expiration date. Species ranges and population sizes are dynamic, and land cover changes due to antropization, resource consumption, and naturalization (MacDonald *et al*. 2000; Falcucci *et al*. 2007). IUCN guidelines, for example, states that Red List assessments become outdated after 10 years, and just as reassessments are a necessity, so is the need to maintain up to date observation data. This, however may not be entirely possible given the large number of species, the limited funds and time.

A full understanding of a specie's conservation status and pressures are not possible through the use of models alone, as it's not currently possible to models many of the factors that influence a specie's presence and survival, such as for example hunting and competition from alien or invasive species. None the less, the use of a modelling approach to refine coarse scale data such as species EOO allows for parameters such as the ESH to be easily calculated. ESH is, for example, explicitly mentioned as a viable option to estimate population trends under IUCN Red List criterion A (IUCN Species Survival Commission 2012). When applied or developed in a systematic manner it may help to identify species of interest (such as through the IUCN Red List) or global sites of interest (Wilson *et al*. 2011). Modelling approaches are furthermore simple, fast and inexpensive to update when new environmental layers become available or if improved approaches are identified, as shown here with the update of the GMA 2011 HSMs (Rondinini *et al*. 2011b).

4.2 – Gap analysis

Failing or not being able to account for the prevalence and distribution of suitable habitat in a specie's range may strongly alter the perception of specie's status, potentially causing misinterpretations or leading the investigator to misjudged conclusions. This may be of particular concern when thresholds are used, as the interpretation tends to become more an issue of absolute numbers or inclusion/exclusion of species into categories of interest (e.g.

species that reach or fail to reach determined conservation targets).

Ideally, the aim of any conservation study should be to protect and sustain at least one viable population in order to guarantee long term species survival. However, many conservation targets, particularly when multiple species are concerned, are arbitrary (Jennings 2000; Rodrigues *et al*. 2004) and face little to no real biological significance or link to probability of survival. (Rodrigues *et al*. 2004) for example set two thresholds for their gap analysis: 100% protection for species with restricted distributions (less than 1000 km^2) and 10% for species with wide distributions (more than 250.000 km^2). The 100% target was based on the concept that some species distributions are so small that only complete protection can guarantee their survival and the fact that 1000 km^2 was the fundamental size of the planning units at the resolution their study was partaken at. In the present study, the base resolution of the land-use layer was at 300m (0.09 km^2), and species ranges were derived from vector maps. It is not unreasonable to assume that future land-use maps may come at an even finer resolutions, and vector data have a theoretically infinitely fine resolution (depending on the method used to draw or develop the vectors). It may be argued that the currently available resolution is already so fine that any single potentially identifiable land unit is individually insignificant towards any one specie's conservation. Finer scale base layers may however still be summarised at a coarser more practical planning unit size with added information such as proportional values. Although the issue wasn't covered in the present study, detailed information on the amount of suitable habitat contained in single planning units might allow or highlight the need to identify species specific minimum suitable habitat thresholds, as low suitable habitat prevalence might be an indicator of particularly isolated patches, ongoing habitat fragmentation or urbanization. Any of these causes might ultimately lead to species absence despite the presence of suitable habitat (Andren 1994), and ignoring such processes may lead to commission errors. The rationale for the upper threshold, on the other hand, was that species with wide distributions "blend" in the protected area matrix: as species ranges get bigger, the species is less in need of protected areas to ensure survival, and the proportion of protected EOO tends to converge to the global proportion of protected areas. The 10% cover threshold was selected as it was the approximate proportion of the world surface covered by protected areas at the time of their study. The 250.000 km^2 threshold was however arbitrary, and as specie's ranges get bigger it is only reasonable to argue that if conservation efforts are needed, these should be shifted from protecting habitat to ex-situ approaches. The grey wolf (*Canis lupus*), for example, has a species range of over 51,000,000 km^2, mostly in areas of low or moderate species richness and only 6% of this is protected; under the above mentioned approach, this species would remain gap unless an extra 2.000.000 km2 of its EOO was to be converted into protected areas.

As the focus of this specific analysis was to highlight the added benefit of HSMs in conservation analyses and protected area planning in particular rather than to develop functional products, the thresholds were not changed. The above considerations therefore all apply and results should be interpreted with caution. This being said, using HSMs offers a significant improvement in terms of interpretability compared to approaches strictly based on EOOs. The fact that 28% of species do not have enough suitable habitat to cover targets obtained with a traditional EOO based approach suggests two things. Firstly, even for species with very restricted ranges, protecting the entire range might not only not be sufficient to promote survival, but could also lead to a waste of resources on areas that may hold very limited biological value or that are already compromised. Secondly, among gap by default species there was both a higher concentration of threatened species and species with small ranges. Since habitat loss and fragmentation is the main threat so far identified for mammals, it is likely that in many instances special attention might need to be paid in order to identify when habitat restoration and *ex-situ* protection are essential in a successful integrated conservation strategy.

The previous global gap study (Rodrigues *et al*. 2004) identified that the ranges of 13% of the analysed species did not intersect any protected areas. For mammals, however, the figure was closer to 5%, denoting a generally higher attention to this taxon. Here, 11% of terrestrial mammals were identified with no protected EOO coverage, but it is not clear if the discrepancy is due to the extra species included in our study or the updates that have since undergone in the WDPA database. If we consider HSMs, however, the figure raised to 12% as an extra 54 mammals do not have any suitable habitat represented within protected areas. As on average species had a higher concentration of suitable habitat in protected areas, it is not surprising that accounting for ESH causes less species to fail in reaching their conservation targets: 91% partial gap species for EOO based assessment v.s. 84% partial gap species for the ESH based assessment. Both of these figures are however much higher than the figure of approximately 60% reported during previous study (Rodrigues *et al*. 2004). As mentioned these figures need to be interpreted with caution, as no form of population viability analysis has been performed, but it is beyond doubt that failing to account for habitat suitability in conservation planning may lead to strong commission errors when assessing current protection levels, which on the other hand may translate in a consequently strong potential for omission errors when planning further actions such as the expansion of the current protected area network.

While Scandentia and Pholidota have a higher proportion of species susceptible to omission error (identified as non gap by analysing their EOO exclusively) than other taxa, the small size of these groups means that there could be local stochastic effects in play. The number of rodents and bats species combined with the high incidence of omission errors observed and wide geographical distribution, on the other hand, suggests that these groups there could be particularly susceptible to this type of error. The gap classification errors in bats and rodents cannot be explicitly explained. However, it could be speculated that for bats it is a consequence of their locomotion which generally allows for greater distances to be covered during dispersal; while for rodents small home range size might allow for survival in very fragmented regions. Both of these factors could result in species with a greater portion of unsuitable habitat within their EOO. It might be possible that other species which have small habitat requirements, are highly mobile (e.g. birds) or which undergo strong dispersal events also suffer from these effects.

Species under higher threat levels reached on average a lower proportion of their conservation target. This is not surprising considering that habitat loss and degradation by definition occurs outside of protected areas. While habitat loss and degradation are not the only threats mammals are exposed to, they are the main contributors to mammal decline (IUCN 2013). More surprisingly, data deficient species, which traditionally have never been grouped with threatened species, show a response more compatible with the latter. This is particularly significant especially considering that data deficient species comprise 13% of all terrestrial mammals. While reaching a lower proportion of conservation target does not necessarily mean a species is more exposed to threats, it does however mean that it is less protected from them should they be present.

4.3 – Identification of priority sites

The map of anthropic pressure used in this study accounts for land transformation, which is recognised as the main threat to mammals (IUCN 2013). It does not, however, account for all possible threats to mammalian species. Threats such as direct killing (as in the case of subsistence hunting) are actually more likely to occur in rural areas, and may therefore be inversely correlated with human infrastructures. The use of a relatively wide resolution (10 km) may ultimately allow for a better evaluation of proximal threats caused my human presence alone. Threatened species are in many cases so due to the effects of anthropic pressures, therefore explicitly accounting for the IUCN Red List category in addition to anthropic pressure may introduce a certain degree of redundancy. The Red List category does however integrate information useful to identify both species who's main threat is not linked to habitat loss or correlated with anthropic infrastructures, and local populations which are not under direct threat but belong to a species that is, and therefore could be ideal conservation candidates. Explicitly accounting for conservation status is arguably more beneficial in the light of accounting for both local and global pressures to the species.

The areas identified as highest priority by the PS and PSW metrics are substantially different and do not match very well with mammal richness, although a very weak correlation between the two exists, this is non significant. The areas with highest priority virtually all fall in biodiversity hotspots (Underwood *et al*. 2008), particularly the PS metric. Only the Brazilian Cerrado and New Zeland are not represented. New Zeland's absence is most likely due to the small number of native terrestrial mammals, while the Brazilian Cerrado is largely under low anthropic pressure (Sanderson *et al*. 2002) and with only moderate richness in mammal species. In addition, the PS metric identifies as top priority a large portion of the sub-Saharan tropical strip, parts of southern China, Japan, Papua New Guinea and the eastern coast of Australia.

The strong resemblance between the priority maps developed in this study and endemic hotspots (Myers *et al*. 2000) is to a certain degree both expected and not surprising, since both metrics account for threat (habitat loss in Myers hotspots and anthropic pressure in the PS-PSW metrics) and restricted distributions (accounted for by endemism in Myers hotspots and species prevalence in the PS-PSW metrics). Perhaps the most interesting aspect of this metric is that using mammals alone resulted in patterns very similar to those from other studies that used multiple taxa (Myers *et al*. 2000; Rodrigues *et al*. 2004). The finer scale of pressure and presence maps in this approach does, however, offer two strategic benefits. Fine scale priorities are identified, and ranking allows for the most critical sites to be highlighted. Madagascar, for example was identified as high priority mainly in the coast, and specifically in the northern and eastern coast once species conservation status was accounted for. More importantly, the distribution of priority scores shows that there is a very small group of sites that hold extremely high priority scores. The top 1% of the global land surface accounts for 19% of the cumulative global conservation priority. That these sites should be considered as serious initial candidates to receive the limited conservation resources for protection or otherwise intervention.

The Indomalayan and Oceanic biogeographic regions showed a higher than average proportional concentration of high priority sites. While the theory of island biogeography explains both how small and isolated islands tend to have lower richness and how these are also more likely to have evolved endemic species (Chen & He 2009), continental and oceanic islands were also shown to be characterised by a much higher than usual endemic richness (the sum of the proportional endemism across all species present; (Kier *et al*. 2009), a factor which is intrinsic in the PS-PSW metrics. This also explains the high incidence of priority rich sites in other islands such as Japan and the Caribbean archipelago, but most importantly the high priority of islands such as Madagascar compared to continental regions characterised by both higher pressures and higher species richness such as central Europe or certain areas in continental North America.

The CBD Aichi target 11 explicitly states: "*By 2020, at least 17 per cent of terrestrial and inland water areas and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscape and seascape*". Furthermore, explicit recommendations have been made to maintain ecological representativeness, including the protection of at least 10% of each ecoregion in each country (Convention on Biological Diversity). This strategy ensures political equitability and binds adhering countries to not concentrate protected areas on ecoregions that are abundant or perhaps of low economic value. However it might also hinder the protection of high priority sites by redirecting national budgets in order to achieve the representation objectives. Biomes are larger biogeographic unit than ecoregions, yet already highlight some of the limitation of equitable targets. Biomes are not equally distributed across the globe with mangroves covering less than 1% of the land surface and Deserts & Xeric shrublands covering over 20%. Naturally, as smaller or isolated biomes tend to result in a certain degree of endemism, it is no surprise that more restricted biomes tend to be characterised by higher priority scores. Obvious benefits of cooperative efforts to conservation and sustainability have already been highlighted. The Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005), for example, highlights the benefits of globalized approaches compared to regional ones. The analysis presented here have shown how both approaches are capable of highlighting sites that cover the majority of cumulative priority values: imposing a constant representation target across all biomes results in a protected area network that has a 60% overlap with the best possible solution while maintaining the global cumulative priority value only 9% lower. This suggests that local representation targets may substantially influence efficiency, but detailed analyses on the effects of national representation targets may yield even more worrying results. There are far more nations than there are biomes, and for the latter, impacts might be of greater magnitude. Despite this, allowing for nationally defined targets may aid in reaching the objectives as they allow single nations to better evaluate local necessities, include shareholders in the area selection process and avoid a potentially detrimental aura of super-partes imposition.

The need to place protected areas in adequate locations needs, however, to be stressed. All biomes currently fall short of the 17% representation, yet the Boreal forests, Taigas and Tundra, which are arguably of lower economic interest, are the only biomes that have received more protected area cover than what identified as the optimal global solution. The Tundra in particular is currently 14 times more protected than what has been identified as priority under the optimal solution, while the tropical and subtropical coniferous forests, which are currently extreme rich in high priority sites, are grossly under-represented with current protected area covering forty times less than what identified as optimal.

4.4 – Impacts of Climate Change

Impacts of climate change have for more than two decades become one of the central focus points of conservation biology (Heller & Zavaleta 2009). In the light of niche theory (Hutchinson 1957; Soberón & Nakamura 2009), climate is clearly one of the factors that can limit a specie's distribution, and anthropic induced climate change is projected as being among the top drivers of biodiversity loss in the next century (Sala *et al*. 2000; Thomas *et al*. 2004). As one of the main aims of protected areas is the representation and long term persistence of biological elements, there has been growing concern on how climate change may negatively impact the efficiency of pre-existing or future parks or reserves (Heller & Zavaleta 2009). Rare climate types may for example disappear or novel climates may come in existence (Wiens *et al*. 2011), and species may disappear from certain protected areas altogether (Kharouba & Kerr 2010). Multiple recommendations have been made on how to approach this issue, such as improving inter-park connectivity either by strategically placing new parks (Halpin 1997) or generally recreating or protecting corridors (Donald & Evans 2006), locally adaptive management (Halpin 1997; Noss 2001) and proactive management (Buckland *et al*. 2001; Hulme 2005; Prato 2012). In more general terms, these strategies can be placed along an axis that ranges from risk-averse (i.e. improving resilience and protected area cover) to risk-tolerant (i.e. proactive approaches accounting for model forecasts) (Heller & Zavaleta 2009). Risk averse approaches do not require a robust prediction of climate change or future conditions in general, but because of this tend not to profit of all the latest research into ecological modelling and climate change, and might be detrimental in extreme scenarios as they could promote investment in sites and strategies that are destined to be unsuccessful (Heller & Zavaleta 2009). Risk tolerant approaches, on the other hand, need to avoid deterministic (i.e. single future) projections and need to account for uncertainty in order to develop a strategic plan that may be robust under multiple future scenarios (Millennium Ecosystem Assessment 2005; Heller & Zavaleta 2009). This issue is of particular concern when considering the extent of the current protected area network is still far from reaching Aichi target 11, and that new reserves can be selected accounting for both risk averse and risk tolerant approaches. Scenarios of climate change have become a staple in conservation research (e.g. IPCC 2007; Kujala *et al*. 2013) with the latest trend favouring the use of strategic development scenarios which may aid decision makers in achieving the millennium development goals (van Vuuren *et al*. 2012). It is only logical to analyse these in order to provide integrated reports that may aid in the decision making process.

Prudence must however be exercised, as global socio-economic development decisions and systematic conservation planning are not strictly integrated processes and international coordination cannot be guaranteed. Furthermore, while it is possible to develop climatic niche envelope projections outside of current ranges and model potential species range expansion (Visconti *et al*. submitted), species range shift is not a statistical certainty and under the precautionary principle protected areas should be planned accounting for potential loss in efficiency but not on uncertain gain.

Both business as usual and mitigation scenarios predict substantial changes in temperature patterns worldwide. Even the IPCC B1 family scenario, which assumes a global socially and environmentally responsible approach towards long term sustainability predicts mean temperature increases between 1.1 and 2.9 °C by the year 2100 compared to the year 2000 (IPCC 2007). It is therefore not surprising that under both BAU and CCH scenarios species presence in protected areas are in the vast majority of cases expected to show climate change driven decline. While for most carnivores and ungulates this decline by the year 2050 is expected to be modest, with 50% of species predicted to loose less than 4% of protected habitat under a BAU scenario, losses up to 100% are to be expected for a small portion of species. Adhering to a mitigation scenario has the potential to substantially reduce expected protection cover loss by effectively halving the median impact and greatly reducing the loss expected by the most susceptible species. As already seen by (Visconti *et al*. submitted), strong mitigation effect increases in the CCH scenario are to be expected after the year 2020, this however only translates in a reduced rate of species specific protected area loss.

Numerous factors contribute to the expected rate of protected habitat loss. The dichotomy between carnivores and ungulates does not appear exceedingly defined: despite ungulates being predicted to lose two and a half to three times more protected habitat than carnivores, the average figures were all very low, and in all cases half of the species analysed lost less than 5% of their protected suitable habitat; differences were mostly significant but at times only marginally so. Lower taxonomic levels showed responses that were clearly more distinct, however these are most likely also influenced by the smaller group size and potentially geographic distribution. Never the less, entire taxa may be destined to loose substantial protected area cover by the year 2050. The Eupleridae, for example are endemic to Madagascar and expected to loose between 59% and 93% of their protected area cover. Elephants were the only family that entirely reached their conservation target, but by facing a potential habitat loss of up to 52%, the African elephant (*Loxodonta africana*) risks dropping from 188% of its target protected area cover to just 90%.

While carnivores and ungulates are in no way representative of all biodiversity or even mammals, it is noteworthy that Palearctic and Nearctic species are much less exposed than Afrotropic, Neotropic and Indomalayan species. The biogeographic realm itself is not, however, a strong predictor; high impact species rather appeared to be concentrated around the equatorial region. It is furthermore noteworthy that the regions exposed to the highest impacts contain some of the greatest hotspots of biological diversity (Myers *et al*. 2000). The evident concentration of highly impacted species and higher overall loss in tropical regions could be explained in many ways, including the higher concentration of restricted range mammals and reduced seasonal climatic variability. Similar forecasts have also been reported for birds (Şekercioğlu *et al*. 2012).

Ultimately numerous factors appear to influence the actual risk of exposure, including range size, anthropic pressure, protected surface, weight, threat category and taxonomy. Many of these factors are however correlated to a certain degree. For example, not only do tropical species tend to have smaller ranges, but species with smaller ranges are also more likely to be threatened. Furthermore, below specific range size thresholds, these automatically fall in specific threat categories (IUCN Species Survival Commission 2012). Intrinsic factors alone are not sufficient to develop a robust predictive model, and such an approach is possibly not achievable due both to the strong autocorrelation between multiple single factors, as well as the inherent spatially explicit information that is lost in this manner.

Without directly accounting for these spatially explicit variables or using spatially explicit models, the best that can be achieved is to inform conservation interventions with species specific robustness values and calculate robust richness eliminating species that are too susceptible or for whom persistence cannot be guaranteed. This particular approach to obtain robust richness for all mammals indicates distinct patterns in individual biogeographic realms and a trend for low conservation priority areas to potentially be slightly more susceptible to climate change. However, the majority of species have restricted ranges and restricted range species are both more likely to lose protected area cover due to to climate change and tend to determine sites of high endemism and richness. Eliminating these species altogether therefore strongly or completely negates the benefits of concentrating limited conservation resources where they might contribute to a larger number of species. Global climate change is not a uniform process, and different regions of the world will experience different rates of warming or cooling (IPCC 2007). While intrinsic and extrinsic data have already been used to calculate innate extinction risk susceptibility and recovery opportunity, giving estimates of species specific risk and are clearly capable of being used to predict potential status (Di Marco *et al*. 2012), spatial information on the climates the species are currently and will in the future be exposed to cannot reliably be excluded from predictive models if the main aim is identifying conservation tools in the light of long-term sustainability.

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