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PROPUESTA DE UNA RED DE ÁREAS PRIORITARIAS PARA LA CONSERVACIÓN EN MESOAMÉRICA, CHOCÓ Y LOS ANDES TROPICALES: UN ENFOQUE USANDO DOMINIOS AMBIENTALES, MODELOS DE NICHO ECOLÓGICO DE ESPECIES Y ANÁLISIS MULTICRITERIO.

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P r e s e n t e

Me permito informar a usted que en la reunión ordinaria del Comité Académico del Posgrado en Ciencias Biológicas, celebrada el día 15 de junio de 2009, se aprobó el siguiente jurado para el examen de grado de **DOCTORA EN CIENCIAS** de la alumna **LONDOÑO MURCIA MARÍA CECILIA** con número de cuenta **505450931** con la tesis titulada: **"Propuesta de una red de áreas prioritarias para la conservación en Mesoamérica, Chocó y Los Andes Tropicales: un enfoque usando dominios ambientales, modelos de nicho ecológico de especies y análisis multicriterio"**, realizada bajo la dirección del **DR. VÍCTOR SÁNCHEZ CORDERO DÁVILA:**

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Índice

Resumen

El aumento de presiones antropogénicas sobre los recursos naturales ha conllevado a una extinción masiva de especies y a la pérdida de hábitats naturales e interacciones bióticas que sustentan los procesos ecosistémicos. Una de las metas de la biología de la conservación es identificar sitios prioritarios para la conservación y manejo de los recursos naturales. Mesoamérica, los Andes Tropicales y el Chocó son consideradas regiones únicas a nivel mundial por su gran número de endemismos y por las altas tasas de deforestación que ponen en riesgo su biodiversidad, también son reconocidas por que los países que las componen presentan problemas de inequidad social, subdesarrollo económico, pobreza, y conflictos sociales. Para que sea exitosa, la conservación no puede competir con las necesidades de las personas, por lo cual, la planeación de la conservación debe integrar criterios multidisciplinarios que conlleven a soluciones más integrales con mayores oportunidades de implementación.

El primer capítulo de este trabajo, a través de una revisión de estudios, demuestra la necesidad que existe de utilizar un rango más amplio de especies, parámetros ambientales, e incorporar el contexto social dentro de los ejercicios de priorización de áreas de conservación. El área de estudio del presente trabajo incluye la identificación de áreas prioritarias para la conservación en 10 países, que componen Mesoamérica, Chocó y los Andes Tropicales, usando un conjunto eficiente de herramientas. Se construyeron modelos de nicho para 313 especies de vertebrados y plantas amenazadas y se crearon clasificaciones ambientales para usar como sustitutos de biodiversidad. Los capítulos 3 y 4 demuestran la gran diversidad de estos componentes en la región de estudio y la magnitud por la que se han visto afectados debido a la pérdida de hábitat, finalmente se encuentra que Belize es el único país de la región que presenta áreas naturales protegidas que representen adecuadamente sus componentes de biodiversidad.

En los capítulos 2 y 5 se usaron programas computacionales (ResNet y ConsNet) para la selección de áreas prioritarias para la conservación, basándonos en la complementariedad de los componentes biológicos y ambientales, algunas de las áreas seleccionadas concuerdan y otras no con trabajos anteriores de priorización a niveles regionales y nacionales evidenciando el cambio en la metodología, los sustitutos utilizados y la importancia de las áreas que son seleccionadas conjuntamente pese a estas diferencias. Quizás uno de los resultados más llamativos es que para conseguir la

protección del 50% de los sustitutos de biodiversidad, los porcentajes de áreas por país que se deben dedicar a la conservación varían entre el 10% y el 80%, demostrando, que para algunos países es imposible dedicar tanta área para la conservación, por lo cual las estrategias de conservación deben ir más allá del establecimiento de ANP e incorporarse a otros usos y manejo del paisaje. Es por esto que el capítulo 6 ofrece un análisis multicriterio en la selección de áreas de conservación, usando el programa ConsNet e incorporando parámetros sociales tales como densidad de población, distancia a carreteras y poblados, productividad primaria y mortalidad infantil entre otros, seleccionando áreas de conservación en tres escenarios, 1) prioridades para reducir la amenaza por perturbaciones antrópicas a los componentes de biodiversidad, 2) prioridades que aseguren la representación eficiente de los componentes de la biodiversidad y 3) prioridades de conservación que no compitan con las necesidades de los personas dando más oportunidades para la implementación.

Este trabajo concluye que la eficiencia de la selección de áreas de conservación no pude estar limitada por la cantidad de área seleccionada por los diferentes algoritmos, pues valiosas oportunidades de implementación y manejo se están perdiendo al partir de esta premisa. Adicionalmente se demuestra el gran valor que tienen los agroecosistemas para la representación de la biodiversidad y de las diferencias que existen en las características de los sitios seleccionados a lo largo de los 10 países y sus diferentes ecoregiones. Es por esto que este trabajo concluye que los esfuerzos de conservación deben incorporarse al manejo del paisaje y usar diferentes estrategias. Dada la cantidad de hábitat transformado, la gran necesidad de conservación y los recursos limitados que se tienen destinados para la conservación, las selecciones de una red de áreas prioritarias para la conservación se debe realizar iniciando a escalas gruesas y finalizando a escalas finas del paisaje para garantizar un mayor éxito, y esta selección se debe hacer desde una multidisciplinariedad incluyendo diferentes componentes de la biodiversidad y teniendo en cuenta el uso del suelo y los aspectos sociales, para explorar las potencialidades de la implementación y reducir la amenaza a las comunidades naturales.

Abstract

The increase in anthropogenic pressure upon natural resources has lead to massive species extinction and lose of natural habitat and biotic interactions, by which environmental processes depends on. One of the goals of conservation biology is to identify priority areas where conservation and natural resource management should be focus. Mesoamerica, Chocó and the tropical Andes regions are recognised as biodiversity hotspots due to their high number of endemism, but giver their high deforestation rates, biodiversity in these regions is seriously threaten. Mesoamerica, Tropical Andes and Chocó regions are also recognized because their countries share social problems such as social inequality, poverty and social conflicts. Biological conservation can´t compete with people's needs; this is why conservation planning should incorporate multidisciplinary criteria that lead to solutions with greater implementation opportunities.

In the first chapter initiatives containing an explicit selection of areas for conservation were review, results showed that further studies should incorporate a broader set of species, environmental parameters and social context for identifying conservation areas. Our study uses a set of different tools in order to identify priority conservation areas within 10 countries in the Mesoamerica, Tropical Andes and Chocó regions. Niche models were constructed for 313 threatened vertebrate and plant species and environmental domains were also used as biodiversity surrogates. Chapter 3 and 4 show the diversity of species and environmental surrogates in the study region, and the effect that habitat transformation have had upon them. Results show that Belize is the only country in the region were natural protected areas represent its biodiversity components in a satisfactory manner.

Chapters 2 and 5 used the software ResNet and ConsNet, which based on the complementarity of biodiversity surrogates identify priority conservation areas. Some of the selected areas agree and some do not with areas chosen by other works at national and regional levels. This result implies a difference in the methodology and surrogates used to identify priority conservation areas and highlight the importance of regions selected by different approximations. One of the results that should be mention here is that for fulfilling 50% of surrogate's representation, countries should allocate between 10% and 80% of their areas for conservation. For some countries this is a huge amount of area that will be

impossible to achieve. That is why conservation strategies should go beyond the establishment of natural protected areas and try to incorporate other land use managements. Chapter 6 offers a multicriteria analysis including social parameters such as population density, distance to settlements and roads, primary productivity and infant mortality upon others, by which three scenarios are produce were conservation areas are selected: 1) priority regions for reducing anthropogenic threat to biodiversity, 2) priority regions for an efficient representation of biodiversity, and 3) priority regions were better implementation opportunities can be offer.

This work concludes that conservation area selection can´t be limited by efficiency in the amount of area selected by different algorithms, because doing so will reduce valuable opportunities for better implementation and management options. Additional importance is given to agricultural land showing high value for biodiversity representation along the 10 countries that encompass the study region. Finally this works discuss that conservation efforts should incorporate different strategies of natural resource use and management. Given the great amount of transformed habitat, the big need for conservation, and the limited amount of resources for conservation purposes, selection for conservation areas should be done at regional scales and then at local scales, and should have a multidisciplinary emphasis incorporating different components of biodiversity, land use and social aspects, in order to explore potential areas for better implementation opportunities and threat reduction to natural communities.

Introducción

La adecuada planeación de la conservación es una necesidad que enfrenta nuestro planeta (Margules & Pressey 2000), el cual está atravesando la más grande crisis de biodiversidad en su historia (Dirzo & Raven 2003). La principal causa de esta crisis es la transformación del hábitat para uso humano. El efecto de la pérdida de hábitat produce cambios físicos en los ecosistemas y en las interacciones bióticas, resultando en una influencia negativa fuerte sobre la biodiversidad (Fahrig 2003, Saunders et al. 1991).

Pese a esta evidente crisis, los recursos tanto de espacio como de dinero disponibles para la conservación son limitados, existiendo además restricciones sociales y culturales para el manejo e implementación de áreas de conservación (Possingham et al.2000). Los planes sistemáticos de conservación surgen como una respuesta a la necesidad de planear adecuadamente donde invertir los escasos recursos disponibles para la conservación (Margules & Pressey 2000, Possingham et al. 2000). Los planes sistemáticos de conservación se basan en una serie de pasos que van desde la participación e identificación de actores de conservación, hasta la apropiada evaluación y monitoreo de las estrategias de conservación implementadas (Margules and Pressey 2000); siendo el paso que trata la selección de áreas prioritarias para la conservación, el tema donde mayor trabajo se ha realizado.

Las áreas naturales protegidas (ANP) decretadas han sido en su mayoría establecidas sin una adecuada selección, ya que eran escogidas sin objetivos biológicos específicos, sino más bien por razones culturales, de belleza escénica, porque nadie habitaba ni usaba la tierra donde fueron establecidas o para representar especies carismáticas; por esta razón las áreas que se han destinado para la conservación tienen una mala representación de la biodiversidad (Possingham et al. 2000, Rodrigues et al. 2004). Ahora la selección de áreas se enfoca a ser eficiente, usando algoritmos que intentan resolver dos problemas principales: (1) encontrar el menor número de sitios donde se incluyan una cantidad definida de especies o características que se quieren representar, y (2) encontrar una solución que incluya el máximo número de especies o características que se quieren representar, teniendo un número de sitios a priori como limitante (Pressey et al. 1993, Church et al. 1996, Possingham et al. 2000, ReVelle et al. 2002).

Los métodos para la selección de áreas de conservación que usan el principio de complementariedad, entendido como la contribución cuantitativa de un sitio para representar las características o especies que aún no han sido representadas en los sitios seleccionados (Kirkpatrick 1983, Pressey et al. 1993, Margules & Pressey 2000, Possingham et al. 2000), han mostrado ser los más eficientes, resolviendo de manera óptima los dos problemas mencionados (Margules & Pressey 2000, Pressey et al. 1993).

La selección de áreas para la conservación se basa en el concepto de los sustitutos. Un sustituto de biodiversidad se refiere a un atributo o a una combinación de atributos bióticos y/o abióticos, que son usados para obtener información referente a la diversidad biológica en vez de medirla directamente (Sullivan & Chesson 1993). Dado que la conservación procura la persistencia y representación de toda la biodiversidad, el uso de sustitutos es imprescindible en la conservación, pues prácticamente es imposible medir todos los componentes de la biodiversidad (Austin & Margules 1986, Reyers et al. 2000). Para ser usado en la conservación biológica, un sustituto de biodiversidad debe ser cuantificable, se debe tener suficiente conocimiento sobre su biología y ecología, y se debe evaluar de manera apropiada; de manera que identifique claramente áreas importantes para la conservación y lineamientos de manejo en las mismas (Sarkar & Margules 2002, Cabeza et al. 2008).

Se han generado numerosos trabajos con el fin de evaluar que tanto representa un conjunto de sustitutos las característica de la biodiversidad que se desean conservar (ej. Andelman & Fagan 2000, Reyers et al. 2000, Fjeldsa 2007, Larsen et al. 2007, Rodrigues & Brooks 2007, Payet et al. 2009). De manera general, estos trabajos encuentran que las especies endemicas, amenazas y de distribución restringuida resultan ser los sustitutos mas eficientes. Sin embargo, la elección de sustitutos depende en gran medida de la disponibilidad de datos, por lo que en la mayoría de los casos la elección de un sustituto es subjetiva (Rondinini 2006).

Las especies has sido ampliamente usadas como sustitutos, sin embargo los datos de registros de especies presentan algunas limitaciones: son escasos los datos biológicos tomados de manera sistemática, muchas veces no se encuentran actualizados taxonómicamente, y casi siempre están sesgados hacia especies carismáticas, como mamíferos o aves, cerca de estaciones de investigación, sitios de fácil acceso, y altamente correlacionados con la distancia a carreteras (Possingham et al. 2000, Reddy & Davalos 2003). Adicionalmente los datos para muchas especies son limitados, menores a 25 registros geográficos por especie (Graham et al. 2004, Soberón & Peterson 2004).

Esta situación se presenta particularmente en áreas donde se tiene poco conocimiento de la biodiversidad, como es el caso de muchas áreas tropicales (Raxworthy et al. 2003).

Contar con un conocimiento adecuado sobre la distribución geográfica de las especies es crucial para la planeación de la conservación (Margules & Pressey 2000). Las aplicaciones de modelos de distribución de especies (MDE) permiten usar directamente información incompleta de los registros de las especies, para generar predicciones sobre los rangos de distribución de las mismas. Por ello son altamente prometedores y su uso en planes de conservación como sustitutos ha aumentado. Los MDE, convierten datos de localidades puntuales de distribución en rangos de distribución hipotéticos de especies, que resulta en evaluaciones más representativas que aquellas basadas en puntos de registros biológicos (Rondinini et al. 2006).

Otro conjunto de sustitutos que se ha usado ampliamente son las clasificaciones que combinan atributos biológicos con atributos abióticos (ej. Lombart et al. 2003). Quizá una de las clasificaciones más usadas son las ecoregiones terrestres de la WWF (Olson et al. 2001), usualmente empeladas en priorizaciones globales y regionales (Dinerstein et al. 1995, Loyola et al. 2007). La definición de categorías dentro de estas clasificaciones se hace de manera cualitativa, sin embargo hay otro tipo de clasificaciones ambientales que se realizan a partir de datos cuantitativos de variables ambientales, aprovechando información robusta que se ha generado a nivel mundial sobre estas variables (Hijmans et al. 2005). Dichas clasificaciones cuantitativas se basan en la identificación de espacios multidimensionales determinados por valores específicos de las diferentes variables ambientales, cada espacio multidimensional es llamado un dominio ambiental. Las clasificaciones de dominios ambientales se usan en planes de conservación partiendo de la hipótesis que las diferentes condiciones ambientales de cada dominio pueden representar comunidades de especies diferentes (Mackey et al. 1988, Belbin 1993, Trakhtenbrot & Kadmon 2005). Las clasificaciones cuantitativas basadas en características ambientales han sido usadas como sustitutos de biodiversidad (Faith & Walker 1996a, Pressey et al. 2000, Oliver et al. 2004, Sarkar et al. 2005), complementando la deficiencia de información detallada que tenemos sobre las distribuciones de muchas especies.

Una vez elegidos los sustitutos, los algoritmos para la identificación de áreas prioritarias de conservación seleccionan sitios complementarios de manera secuencial, hasta que se alcanza una meta cuantitativa de representación por sustituto. Las metas cuantitativas han sido criticadas por su falta de fundamento biológico, al ser valores impuestos arbitrariamente (Pressey et al. 2003, Faith et al. 2001, Svancarra et al. 2005). Algunas veces estos valores son mayores para, por ejemplo, especies en peligro, que para especies generalistas, o tienen valores en función al porcentaje del área a conservar, como el valor establecido por el Convenio sobre la Diversidad Biológica (2002), donde cada país debería proteger el 10% de cada uno de sus ecosistemas. El uso de estas metas cuantitativas sin bases biológicas puede tener consecuencias graves, pues se puede pensar que al cumplir las metas se está conservando adecuadamente la biodiversidad, cuando puede no ser cierto (Soulé & Sanjayan 1998). Idealmente estas metas de conservación deberían estar basadas en análisis de poblacionales viables, pero son pocas las especies que cuentan con este tipo de estudios.

La mayor crítica hacia los algoritmos para la selección de áreas de conservación se basa en que se ha enfatizado la representación, ignorando otros criterios para asegurar la persistencia de la biodiversidad (Cabeza & Moilanen 2001). Actualmente el uso de algoritmos sigue tratando de resolver los mismos problemas de representación de sustitutos y eficiencia en área seleccionada, pero ahora de una manera más compleja, integrando otros criterios que representen condiciones con mejores oportunidades para la persistencia de la biodiversidad e implementación de estrategias de conservación.

En un contexto global, las amenazas para la persistencia de biodiversidad están estrechamente relacionadas con las actividades humanas, las cuales a su vez están delimitadas por factores socio-económicos y culturales, por lo que la conservación es, sin duda, un problema social (Soulé 1991). Por lo tanto, la protección de la biodiversidad debe ser integrada a los planes de manejo y uso de los recursos naturales, estableciendo como objetivos no solo la representación y persistencia de la biodiversidad, sino también estrategias de estilos de vida sustentable que permitan que la conservación no compita con las necesidades humanas y pueda tener un mayor éxito en su implementación (Margules & Sarkar 2007).

Las áreas naturales protegidas han sido incluidas recientemente en las agendas internacionales como de gran importancia para los planes de desarrollo sustentable (United Nations 2008), y se espera que en países en desarrollo con altos niveles de pobreza, la conservación de la biodiversidad adquiera mayor importancia para contribuir al desarrollo local y la reducción de las necesidades básicas de las personas (Naughton-Treves et al. 2005).

El uso de criterios socio-económicos en la selección de áreas de conservación ha sido reconocido explícitamente en la literatura (Faith & Walker 1996b; Margules &

Pressey 2000; Margules & Sarkar 2007), y algunos trabajos ya los han implementado (ej. Faith et al. 1996, Faith & Walker 1996b, Cowling et al. 2003, Williams et al. 2003, Moffett et al. 2005). Usualmente los criterios socio-económicos utilizados están estrechamente relacionados con la persistencia de las especies, como la probabilidad en el cambio de uso de suelo relacionada con la distancia a fuetes de desarrollo, como asentamientos humanos y carreteras (Pressey et al. 1993, Wilson et al. 2005); sin embargo se pueden usar otros criterios, tales como valores culturales, recreacionales o educativos, densidad de población humana, niveles de pobreza, o probabilidad de desastres naturales; permitiendo que la selección de áreas de conservación pueda ser evaluada bajo diferentes criterios socio-económicos (Margules & Sarkar 2007).

Uno de los principales desafíos de la conservación es que resolver el problema de selección de áreas prioritarias de manera práctica es mucho más complejo que sólo encontrar la solución a la óptima representatividad de las características biológicas. Debe existir flexibilidad para proponer soluciones alternas, pues por condiciones sociales, culturales y económicas es altamente probable que la solución óptima no se pueda implementar (Possingham et al. 2000). El potencial de los algoritmos para la planeación de la conservación es amplio, pero se debe trabajar para hacerlos más efectivos al incorporarlos en escenarios reales de conservación.

El presente trabajo aborda el problema de la selección de áreas de conservación en un área de estudio que reúne tres regiones megadiversas: Mesoamérica, Chocó y Los Andes Tropicales. Estas regiones presentan una gran diversidad biológica, reflejada principalmente en el número de especies endémicas, pero también presentan altos niveles de pérdida de hábitat y amenaza para su persistencia (Myers et al. 2000). Adicionalmente no sólo comparten la riqueza biológica sino también riqueza racial y cultural, así como problemas sociales y económicos reflejados en altos niveles de pobreza, altas tasas de desempleo y bajos niveles de educación (Banco Mundial 2007).

La aproximación a la selección de áreas para conservación en este trabajo se hace por medio de alcanzar distintos porcentajes en la representación de dos grupos de sustitutos: 1) distribución de especies amenazadas, escogidas debido a la urgencia que representan para ser incorporadas en planes de conservación dado su alto riesgo de extinción (Mace et al. 2008) y 2) dominios ambientales, escogidos por la disponibilidad en los datos que permite compara homogéneamente la diversidad ambiental para toda la región de estudio. Así, se obtienen resultados no sólo sobre los sitios prioritarios para alcanzar una representatividad de manera eficiente , sino también sobre la representación de estos sustitutos en los sistemas de Áreas Naturales Protegidas establecidos para los diferentes países, se evalúa la riqueza y diversidad de estos sustitutos en las diferentes ecoregiones y países del área de estudio, y se realiza un análisis sobre la eficiencia de cada sustituto basado en el porcentaje de área sobrepuesta en las diferentes soluciones.

Finalmente este trabajo concluye con una selección de áreas de conservación que va más allá de la representación biológica, incorporado criterios sociales y variables no biológicas que permiten encontrar soluciones para diferentes escenarios de conservación. En conjunto este trabajo hace un aporte importante no sólo en la identificación de sitios prioritarios para la conservación en Mesoamérica, Chocó y Los Andes Tropicales, sino también en aspectos metodológicos relacionados con el mejor aprovechamiento de los datos disponibles para la priorización de áreas y las herramientas para el análisis de datos.

Hay un vacio importante en la planeación de la conservación donde es urgente demostrar que la conservación puede ir de la mano con mejores oportunidades sociales, y que al mismo tiempo se representen adecuadamente las características biológicas, con mejores oportunidades para su persistencia (Cameron and Williams 2008). Este trabajo es sin duda un valioso aporte para ir llenando ese vacío en regiones megadiversas.

CAPITULO I

REVISIÓN DE TRABAJOS REALIZADOS EN SELECCIÓN DE ÁREAS PARA LA CONSERVACIÓN EN MESOAMÉRICA, CHOCÓ Y LOS ANDES TROPICALES

Selection of areas for conservation in Mesoamerica, Tropical Andes, and Chocó hotspots: A review.

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ABSTRACT

Mesoamerica, Tropical Andes and Chocó regions are biodiversity hotspots where selection of conservation areas is urgent due to increasing deforestation. We reviewed 80 initiatives between 1999 and 2007, containing an explicit selection of areas for conservation in these regions, and evaluated their methodologies used for site prioritization. Mexico ranked top (N = 46) as the country with more studies of selection of areas for conservation throughout the region. Mammals, birds and certain groups of vascular plants were commonly used as biodiversity surrogates, and complementarity analyses were usually applied for conservation site selection. We observed geographic biases in conservation site selection, where Mexico ranked top and Ecuador, Panama, Honduras and Belize showed only few studies. Further studies should incorporate other parameters as well as environmental domains, environmental services and a social context.

Key Words: Conservation area selection, Mesoamerica, Tropical Andes, Chocó.

INTRODUCTION

Biodiversity conservation is under threat due to high rates of deforestation, which is particularly severe in tropical regions worldwide. It is estimated that only 5% of tropical forest will remain in the next 50 years, so immediate actions for protecting and conserving natural habitats holding high biodiversity is imperative (Dirzo and Raven 2003). For conservation to be implemented, priority areas need to be identified (Moore et al 2004). A central goal of systematic conservation planning is to identify conservation areas in the most efficient (maximum biodiversity represented) and economic (minimum area cost) way, achieving representation and persistence of biodiversity features (Margules and Pressey 2000). Selected areas for conservation should necessary have both representation, where all relevant features of biodiversity are adequately present, and persistence, where ecological and evolutionary processes are considered in the conservation planning exercise (Sarkar et al. 2006). Recent approaches developing methodological tools implementing algorithms for site selection promise quite useful for the conservation of biodiversity in high-species rich regions. These tools differ in various aspects as collection and treatment of biological and socioeconomic data, selection of features to represent biodiversity quantitatively (biodiversity surrogates), selection of individual conservation areas, vulnerability evaluation, and inclusion of multi-criteria analysis to satisfy biological and social goals, among others (Sarkar et al. 2006).

Mesoamerica, tropical Andes, and the Chocó are considered biodiversity hotspots worldwide (Myers et al. 2000, Calderón et al. 2004). Unfortunately, its exceptional biodiversity is at risk due to rampant deforestation rates and increasing human population growth. For example, mean annual deforestation rate was 1% from 1990 to 2005, and annual population growth increased 2% (World Bank 2007). Moreover, this region face great social inequality (32% of the children work), economic underdevelopment (unemployment from 2000 to 2005 was on average 7 %), poverty (on average, 42% of their population lives below the national poverty line), and social conflicts (nearly three million people emigrated from their country of origin between 2000 and 2005, and 69 700 were refugees in other countries in 2005; World Bank 2007). These regions hold a diverse range of distinct ethnic groups, including African descendants, indigenous and mestizo communities. Such cultural, social and economic complexity challenge biodiversity conservation in the region, and urgent actions, including selection of priority conservation areas, require immediate implementation.

Fortunately, there are important efforts devoting biodiversity conservation. Governmental international compromises involving biological conservation programs undertaken by Colombia, El Salvador, Guatemala, and Mexico date since 1988 when these countries joined the United Nation Environmental Program (World Bank 2007). Further, different international and national NGO`s have encouraged conservation programs aiming to identify, monitor and protect conservation areas. In addition, an impressive scientific literature documenting its biodiversity has served as a baseline for selecting areas for conservation. As Mesoamerica, Tropical Andes and Chocó hotspots have become a major focus of biodiversity conservation interest for several decades (Miller et al. 2001), we feel that efforts of conservation planning in these regions should be examined. Here, we surveyed the scientific literature to address (a) who is working in site prioritization?, (b) where has site prioritization been conducted?, and (c) which methods have been used? Our analyses intend to identify gaps in conservation planning, and summarize the effort of many people to identify priority areas for biological conservation.

METHODS

Study region:

The study region was defined based on the ecoregions classification proposed by Olson et al., (2001), consisting of 53 ecoregions located in Mexico, Belize, Guatemala, Honduras, El Salvador, Nicaragua, Costa Rica, Panama, Colombia, and Ecuador. The northern boundary was delimited in the transitional Neartic-Neotropical biogeography region of Mexico; it was included up to the Balsas depression which is the region with more influence of tropical Mesoamerican elements (Morrone 2005). The southern and south-eastern boundaries were set in Ecuador and Colombia, delimited by the ecoregions that intercepted the topographic transition were the three ranges, Cordillera Occidental, Cordillera Central, and Cordillera Oriental, emerge from the Andes (Figure 1).

Literature Review

We conducted a search in the World Wide Web (October 2007) retrieving published scientific literature containing the words "conservation areas" and Mesoamerica or Belize or Colombia or Costa Rica or Ecuador or El Salvador or Guatemala or Honduras or Mexico or Nicaragua or Panama in any part of the text; the following scientific search engines were consulted: Blackwell Synergy, J Store, ISI Web of Knowledge, Biological Abstracts, Academic Google and Redalyc (for Latin American publications). The title, abstracts, and literature of close to 250 scientific papers were examined to determine if a selection of conservation areas was conducted. We compiled a database modified from Leslie (2005), divided in five topics: General Information, Location and Scale, Aim and Surrogates, Identification Process, and Primary Outcomes (Table 1 in Supplementary material). As our study focused on the methods used to select conservation areas and their location, we decided to treat each investigation independently, when a different methodology or study area was identified.

RESULTS

Who is working in site prioritization in Mesoamerica, Tropical Andes and Chocó?

Seventy-six reviewed papers were published between 1990 and October 2007 (Figure 2), of which 16 (21%) were conducted by nine international organizations: Bird Life International (BLI), Comisión Centroamericana de Ambiente y Desarrollo (CCAD), Conservation International (CI), Plant Life International (PLI), Ramsar Convention, The Nature Conservancy (TNC), Wild Conservation Society (WCS), World Resource Institute (WRI), and World Wildlife Found (WWF). Mexican institutions published 27 papers, of which 21 were conducted by one institution: the Universidad Nacional Autónoma de México (UNAM), followed by US institutions with 13 papers; Colombia, Costa Rica, Ecuador and Guatemala added nine published papers. UK, Canada, Australia, Brazil, Denmark and Spain institutions contributed with 11 papers.

Peer reviewed journals and type of publication

Fifty nine (77%) papers were published in peer reviewed journals, with an impact factor (IF) ranging between 0.218 and 9.643. *Conservation Biology* with 12, *Biological Conservation* with 9, and *Biodiversity and Conservation* with 8 papers, were the main used scientific journals. Further, nine papers were published in Latin America; two were books and 15 were documents available on web sites.

Location and Scale

A total of nine papers dealt with a global scale, and 14 included more than one country (regional scale), 33 were conducted in one country (national scale), and 20 were considered only part of a country (local scale). Mexico ranked top at a national and local levels with 34 published works, followed by Colombia with 7 (Figure 3). The most common geographical unit used was grids, followed by natural divisions such as ecoregions, life zones, and sites. Occasionally, circles, political divisions and natural protected areas (NPA) were units used.

We observed no linear correlation between the average size of the geographical unit (grid, squares or circles) and the size of the study area, neither a concordance between the geographical unit size and the scale of the work (local, national, regional or global).

Aims and Surrogates.

The aim of 75% of the reviewed papers was the identification or prioritization of conservation areas, whereas 26% were intended to identify patterns of diversity, endemism or biogeography of certain taxonomic groups. Population assessments, NPA assessment and ecosystem evaluations were additional aims that lead to the selection of conservation areas. Species were the surrogates most used in 53 papers (69%). Eight papers used a combination of various taxonomic groups above the level of phylum, and four used a combination of different classes of vertebrates. Birds, mammals, and vascular plants were commonly used as biodiversity surrogates (Figure 4). Conversely, only seven papers used physical or environmental features instead of species as biodiversity surrogates; ten papers included both environmental features and species as biodiversity surrogates. Sociopolitical, biogeography or phylogeography parameters were used consistently in combination with species as surrogates in nine papers (11%). Species distribution models were applied before the site selection process in only 7% of the published works.

Species richness was the biological value most commonly used, included in 50% of papers, although in 86% of the cases was used in combination with other values, as presence of endemic, rare, or endangered species, or a diversity index. Other biological values used were occurrence (the fact that a species is present in a specific site), and population measurements (e.g. minimum viable populations). Other parameters taken into account for the site selection in 38% of the papers were presence of conserved habitat or measurements of the biological quality of the particular site, representation of habitat or ecoregional diversity, and consideration of threats to biodiversity. Area size, connectivity among sites and human population, were only used in 21% of the surveyed papers. Ecosystem services, ecosystem functioning, economic important species, land productivity, environmental variables or genetic distances were used in only 17% of the published papers

Selection process

Complementarity was the criteria most often used in site selection, followed by the single presence of characteristic species, such as migratory, charismatic, threatened or endemic species. Expert opinion was explicitly used in only 10% of works. Vulnerability of habitats to deforestation, urgency to protect threatened species, and conservation opportunities, such as willingness of local people or commitment of organizations were the criteria used in 32% of works. Species richness was used as a criterion in only two works; presence of characteristic species was commonly used altogether.

C-Plan, CPLEX, Focalize, Resnet, LQ graph and MultCSync were the specialized software packages used for site selection, but only in five works. PAE (Parsimony Analysis of Endemism), gap analysis, track analysis and statistical methods of ordination and classifications were used in 13% of surveyed papers. Fifteen papers included decreed NPA into the selection process, and 23 papers analyzed geographic concordance between decreed NPA and selected priority sites; only four investigations considered decreed NPA adequately protecting biodiversity.

DISCUSSION

There is an international interest to identify conservation areas in Mesoamerica, Tropical Andes, and Chocó hotspots reflected in a strong publication record of site prioritization. Presence of international organizations and academic institutions in these regions has played an important role in implementing conservation areas as part of both a national and international effort (Jepson 2005). Academic initiatives were numerous, showing an active and continuous research in site prioritization, especially since 2002. Mexico was the country with the largest number of published works, demonstrating its leadership in this field in Latin-America. The National Autonomous University of Mexico is the largest institution for higher education in Mexico and one of the oldest and most prestigious in Latin America. Nevertheless, the research output per scientist is still low and suggests an average production of only one international article every 2 or 3 years (Delgado y Russel 1992).

The small publication record in site prioritization in Latin America is of concern, despite being a high biodiversity and conservation priority area, work is being poorly divulgated. Leimu and Koricheva (2005) demonstrated that ecological works from Latin American institutions or authors with Latin American names are less cited than other works worldwide. Moreover, documents in Spanish are less likely to be cited than those written in English which reach a wider audience (Collazo-Reyes et al 2008). Consequently, few studies addressing this important topic are widely available worldwide, and most studies remain for local or national reference Gibbs (1995). Further, it has been estimated that approximately 55% of scientific papers from developing countries are publish in local journals with limited presence in international databases (Gaillard 1989).

The poor divulgation of Latin American research in selection of conservation areas has a strong impact limiting funding for research and implementation of conservation. Governments worldwide have adopted different strategies to stimulate research in their countries, where the emphasis is put on individual researchers as well as the number and citation of their publications (Vaughan 2008). Latin America scientists have to be more aware of the responsibility that publication means for the development of strategies for selection of conservation areas in their own countries. More publication effort on site prioritization is needed, particularly in Belize, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Honduras, Nicaragua and Panama, were site prioritization publications have been published mostly by international organizations and institutions that develop projects at regional scales, and the divulgation of works at national and local scales is almost lacking.

The lack of linear relationship between the grid size and the study area suggest that selection units do not tend to be larger in assessments of extensive regions and smaller in more localized studies as suggested by Pressey and Logan (1998). The choice of selection units has important implications for the process of area selection as well as the implementation of its results (Pressey and Logan 1998). Small selection units can represent targets much more efficiently than larger ones, important in determining the eventual costs and the likelihood of implementation, while large units lead to above-target representation of some targeted features, and have a relation with the improvement in the persistence of organisms in larger sized reserves (Janzen, 1986; Shafer, 1990; Saunders et al., 1991; Hobbs, 1993; Laurence 2002). Additionally, size, composition and organization of groups are likely to vary with geographical scale (Sarkar et al 2006), resulting in low congruence of selected sites and weak correlations of irreplaceability between selection units of different sizes (Warman et al 2004). As a consequence, recommendations from studies that have been applied at only one spatial scale must be considered cautiously (Warman et al 2004).

Conservation planning in Latin America must consider methods dealing with spatial heterogeneity of scale in the data available. More studies comparing the distribution of priority areas at different scales must be conducted (Sarkar et al 2006) and examine the consequences of their choice of selection units on site selection (Warman et al 2004). Despite that political units were used only occasionally, they have a significant effect on the sites selected by complementarity, resulting in a loss of overall efficiency. Geopolitical coordination may result in improved overall efficiency and in a better allocation of resources, giving priority to species that are rare across the entire region or to the sites where each species is expected to present a higher probability of long term persistence (Rodrigues y Gaston 2002). Eighteen percent of the published papers were conducted at a regional scale, showing that research integrating several countries in the region has been done, providing information that can be use to intent a geopolitical coordination, especially in the Mesoamerica region.

Biodiversity was the main aim of selection for conservation. Biodiversity surrogates included to represent biodiversity were restricted to some vertebrate and vascular plant groups. Birds and mammals were usually included as biodiversity surrogates because both of availability of data (Rodrigues y Brooks 2007) and that they are charismatic; other important faunistic and floristic groups that were usually excluded due to lack of species distribution data. Charismatic species have been traditionally used in conservation area selection, despite that they do not performed much better, for representing biodiversity, that selecting areas at random (Andelman & Fagan 2000). A balanced combination of surrogates, including well-sampled biological groups and even environmental variables should provide an adequate approach (Margules and Sarkar 2007).

Regions were conservation planning is urgent are regions with poor biological data availability (Pimm 2000), additionally biological inventories have geographical bias, implying that when delimitating species distribution areas, areas where species are potentially present are typically excluded (Stockwell & Peters, 1999; Dennis & Thomas, 2000; Peterson, 2004; Soberón & Peterson, 2004). The use of ecological niche modeling in conservation planning is a valuable tool for identifying habitat where a species has not been recorded but is likely to occur (Illoldi-Rangel et al 2008), and therefore are desirable as biodiversity surrogates in poor data situations and for avoiding geographical bias in data.

As conservation planning is necessarily based on those surrogates for biodiversity for which data can be obtained, and usually data on species are often not sufficiently widespread or consistent at the spatial scales required by conservation planners (Oliver et al 2004), not even enough to get appropriate niche models. Environmental surrogates, as well as vegetation types or other land classifications which are based on data mapped from remote sources, often provide surrogate data at the planning scale (Oliver et al 2004, Rodrigues y Brooks 2007). In addition, in many parts of the world they are the only spatially consistent data available for conservation planning (Noss 1987, Margules and Pressey 2000). As an alternative, environmental surrogates are commonly used for conservation planning, either alone or in combination with taxonomic surrogates (Oliver et al 2004).

Consequently, conservation site selection in Mesoamerica, Tropical Andes and Chocó needs to incorporate other biological groups, additionally to the most commonly used, and consider niche models and environmental variables in poor data situations.

Although ensuring representation is a clear first step in systematic conservation planning (Margules et al. 1988, Pressey et al. 1993), it does not necessarily ensure persistence in the long term (Margules et al. 1994, Rodrigues et al. 2000a, Virolainen et al. 1999). If spatial data on biodiversity representation are difficult to obtain, data on spatial patterns of expected persistence are even more difficult (Rodrigues y Brooks 2007). A fundamental research front in conservation planning is thus developing biodiversity surrogates for improving the probability of persistence (Rodrigues y Brooks 2007). Possible surrogates of long-term persistence within a particular taxon include past patterns of species persistence (Rodrigues et al. 2000a), species abundance (Rodrigues et al. 2000b), habitat quality, and connectivity (Araújo &Williams 2000); population viability analysis and habitat viability analysis are additional methods that can be used for addressing the persistence of populations (Margules and Sarkar 2007).

As shown by our results, data and criteria that could represent the persistence of represented special are very poorly used, showing that site selection for conservation in these regions is bias toward the representation of species and not their persistence. More work linking ecological studies with conservation studies to help critical evaluation of population persistence included in the site prioritization process is badly needed (Gerber et al., 2003; Pressey et al., 2003; Fleishman et al 2006).

Prioritization of conservation areas is Mesoamerica, Tropical Andes and Chocó appeared conducted economically and effectively as complementarity criteria for site selection was usually included. Conservation planning methods based on complementary representation have been developed over the past two decades (Kirkpatrick 1983, Margules & Pressey 2000, Pressey et al. 1993) and are more efficient in identifying networks of areas representing a diversity of biological features than methods that prioritize areas by ranking them based on criteria as rarity and complementarity of species diversity, as well as ecological integrity and resilience (Margules et al. 1988,Williams et al. 1996; Karr, 1981; Smogor and Angermeier, 2001; Davis and Slobodkin, 2004; Gunderson, 2000; Allison, 2004).

It is important to point out that continuing to rely exclusively on simple metrics such as species richness, which are known to be information-poor and subject to estimation errors, is unacceptable. The conceptual and methodological tools exist to make conservation planning a more exact science by implementing more standards for sampling ecological communities, estimating parameters, and using the information derived to develop conservation priorities that reflect the multiple values of natural ecosystems and facilitated implementation (Fleishman et al 2006, Pressey and Bottrill 2008). Recent examples are influential planning software (Possingham et al. 2006), statistical methods to promote real-time negotiation among stakeholders (Ferrier et al. 2000), and new analytical methods used by large organizations (Murdoch et al. 2007). But let us not forget that involving expert judgments complements the limitations of databases used for systematic planning, especially in the implementation phase (Knight and Cowling 2007, Pressey and Bottrill 2008).

Conservation planning needs to solve the complexities and uncertainties of implementation (Pressey and Bottrill 2008), and must incorporate the social-economic understanding of driving decisions of use of natural-resource (Margules and Sarkar 2007) addressing socioeconomic considerations at the outset (Cowling and Pressey 2003). The general need for biological priorities to be assessed in a sociopolitical context has long been recognized (Vane-Wright 1996; Pressey and Bottrill 2008), Studies in selection of conservation sites in Mesoamerica, Tropical Andes and Chocó regions seems precarious in integrating biological with socioeconomic data.

Further, scheduling conservation action is fundamental because it is typically impossible to protect all of the biodiversity features of interest in a region at the same time (Pressey & Taffs 2001). In order to minimize biodiversity loss, some features should be given protection first (Margules & Pressey 2000, Pressey & Taffs 2001): those of high vulnerability and of high irreplaceability (Rodrigues y Brooks 2007). Quantitative priorities must be tempered with consideration of opportunities for action, reflecting a balance between priorities based on the persistence of biodiversity and the need to consider the real-world opportunities and constraints that affect conservation actions (Pressey and Bottrill 2008).

Conservation policy that fails to take account of diverse relationships between conservation needs and the demands of poverty reduction and the related consumptive demands of the growing world economy, risk failure (Sanderson and Redford 2003). Variables like willingness of local communities and stakeholder opinion has to be involved, as well as variables that reflect the vulnerability of habitats and opportunity for conservation to ensure a successful implementation.

The ecosystem services framework (ESF) highlights the long-term role that healthy ecosystems play in the sustainable provision of human well-being, economic development and poverty alleviation worldwide (Turner and Daily 2008). Information of ecosystem services and economic important species must be incorporated into conservation area selection in Mesoamerica, Tropical Andes, and

Chocó, so that policymakers can have practical recommendations and biodiversity creates economical incentives for its conservation (Balvanera et al 2001; Pearce 2007).

Current works of site selection in Mesoamerica, Tropical Andes and Choco, treat both biodiversity and human economic systems as static, relying on a snapshot in time of the distribution and abundance of biodiversity (Meir et al 2004). In the real world, the process of identifying and implementing reserve networks violates these assumptions (Meir et al 2004). Conservation decisions might be improved through the addition of information that could be used to reduce future uncertainty in the site selection process, e.g. comprehensive knowledge of land ownership and land value (Ando et al. 1998), projections of future land conversion patterns (Theobald & Hobbs 1998; Pontius et al. 2001; Waddell et al. 2002; Li et al. 2003), and projections of future bioclimatic conditions (Midgley et al. 2002; Pyke 2004). Different methods for selecting areas for conservation had been used in these hotspots; a reasonable next step might be to do a consensus of the areas that had been identify by different methods and start on them a systematic conservation plan (Margules and Sarkar 2007).

Since 1990, there had been projects to identifying priority areas for conservation; 17 years later, trends show that the decreed NPA are not representing adequately the priority areas for conservation. This review showed that much work involving different surrogates and multi-criteria analysis have still to be done in conservation area selection in Mesoamerica, Tropical Andes and Chocó regions, in order to have a better representation of biodiversity and greater opportunities of implementation.

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FIGURE LEGENDS

Figure 1: Study region.

Figure 2. Number of published papers per year.

Figure 3: Number of published papers per country at regional, national, and local scales.

Figure 4: Number of papers that used different taxonomic groups as biodiversity surrogates

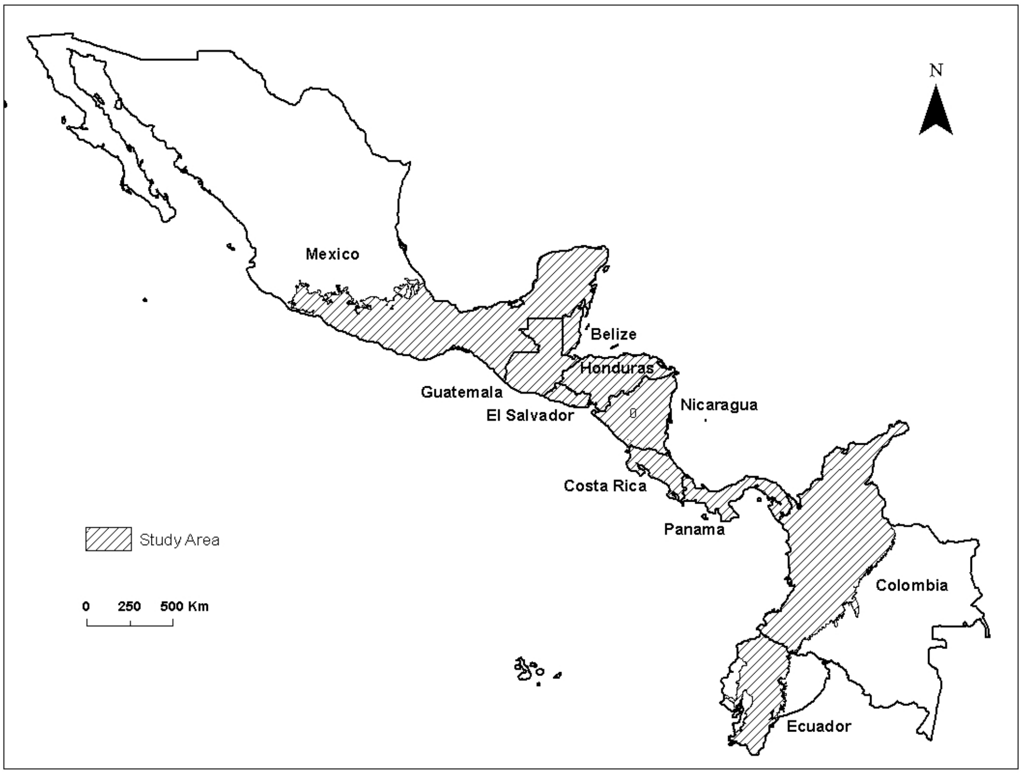

Figure 1: Study region.

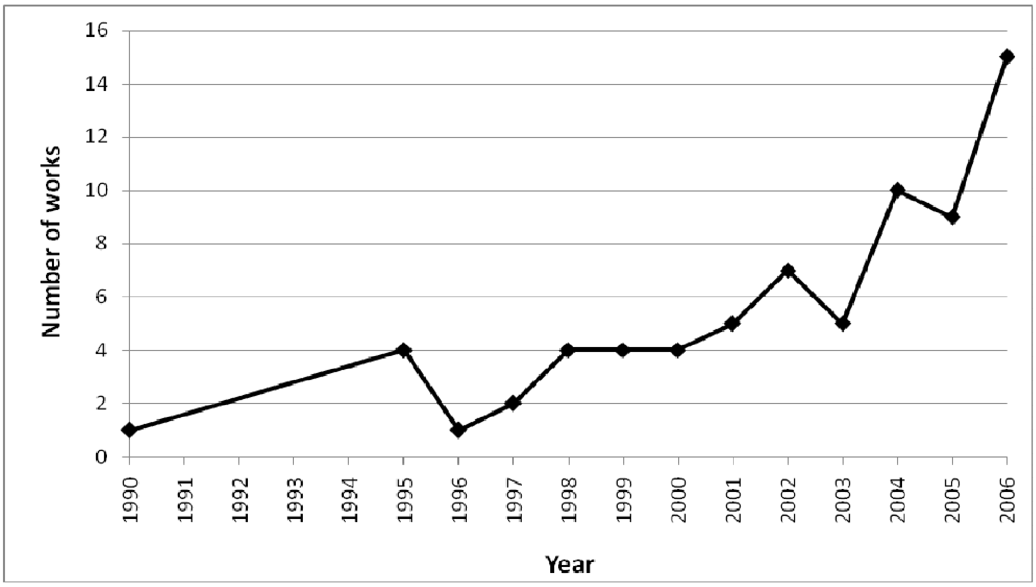

Figure 2. Number of published papers per year.

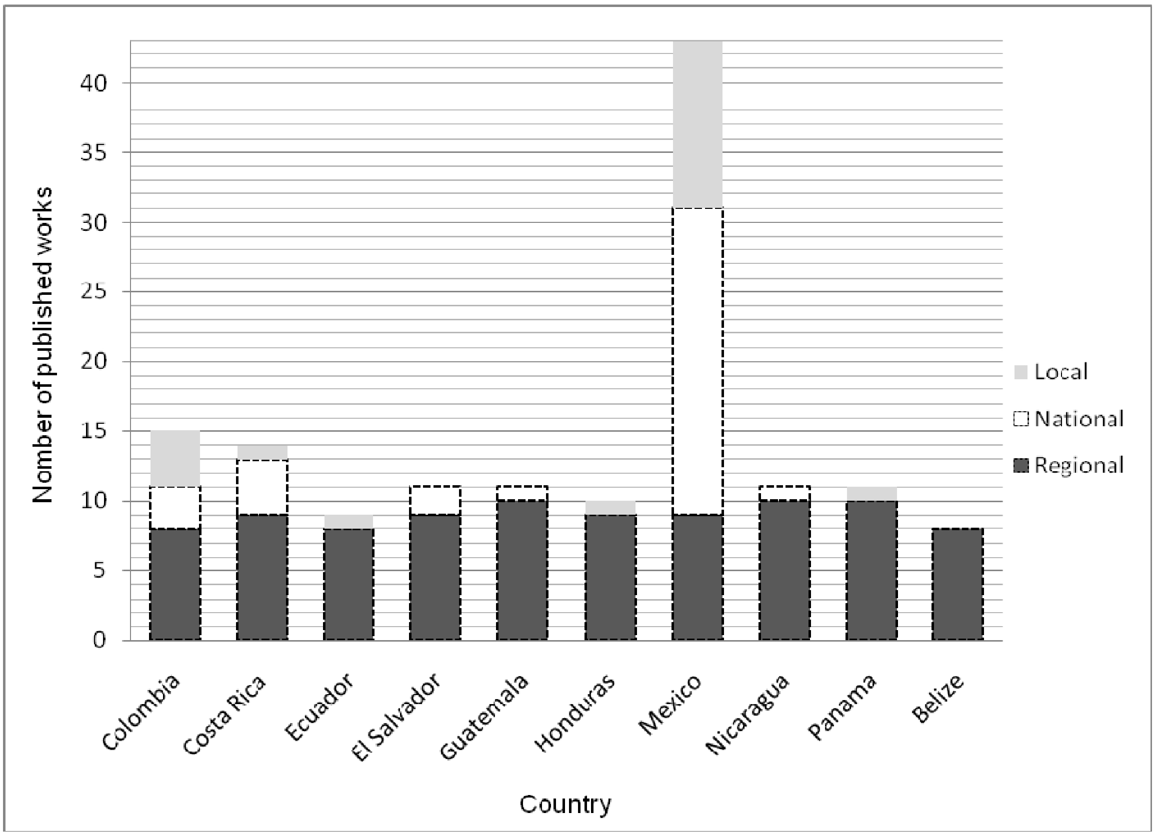

Figure 3: Number of published papers per country at regional, national, and local scales.

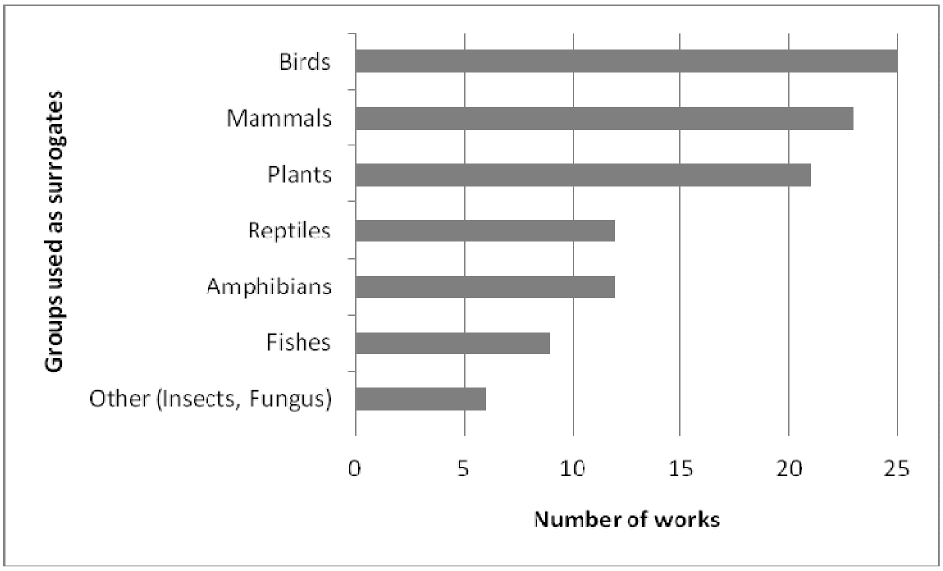

Figure 4: Number of papers that used different taxonomic groups as biodiversity surrogates.

Table 1(Supplementary Material): Questions of the six themes in the database, answered for each reviewed paper.

General Information
Authors and Title
Year and Publication
Publication impact
Institution of first author
Location and Scale
Location
Political Scale: (1=Local, 2=National, 3=Regional, 4=Global)
Size of planning region (in km2)
Type of units?
Average planning unit size (km2)
Aim and surrogates
Primary objective of the case (1=Conservation Area Identification, 2=other)
Is there evaluation of, or a comparisons with established NPA ?(1=yes, 0= no)
Key criteria (1=species, 2=physical/environmental, 3=sociopolitical, 4=biogeography, 5=phylogeography)
Key biological criteria (1=mammals, 2=fishes, 3=amphibians, 4=reptiles, 5=birds, 6=invertebrates, 7=plants, 8=others)
Other criteria: (1= Representation of habitats or Ecoregions, 2= Consideration of anthropogenic extraordinary sites, 3= Consideration of natural extraordinary sites, 4= Priority area size, 5= Connectivity among priority areas, 6= Efforts for protection,7= Human population, 8=Distance to roads or towns 9= Distance to rivers, 10= Environmental variables, 11=Economic important species, 12= Genetic distance, 13= Ecosystem functioning, 14= Ecosystem services, 15= Presence of organization that make the project viable, 16= Consideration of threats to biodiversity
What parameters of Biodiversity were used (1=richness, 2= Occurrence, 3=Abundance, 4=diversity, if diversity which measurement?, 5=complementarily, 6=rarity, 7=endemism, 8= endangered sp, 9= composition, 10= other, 11= Similarity 12= migratory species, 13= small range species)
Was distribution modeling used? (1=yes, 0=no)
Identification Process
Expert opinion used to help make decisions? (1=yes, 0=no)
Sitting tools (algorithms) used for decision support? Which?
Criteria used for site selection (1=irreplaceability, 2= Complementarity, 3=rarity, 4=richness, 5=endangered, 6=vulnerability, 7=abundance, 8= endemism, 9=conservation status, 10=panbiogeographic nodes, 11=phylogenetic diversity, 12=adjacency to NPA, 13=migratory species)
Primary outcomes

Were established NPA adequately representing or protecting biodiversity? (1=yes, 0= no) Did the established NPA overlap the selected areas? (1=yes, 0= no)

Papers and initiatives review.

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CAPITULO II

ANÁLISIS PRELIMINAR DE SELECCIÓN DE ÁREAS PARA CONSERVACIÓN EN MESOAMÉRICA, CHOCÓ Y ANDES TROPICALES, USANDO MODELOS DE NICHO Y ECOREGIONES

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ORIGINAL PAPER

Systematic conservation assessment for the Mesoamerica, Chocó, and Tropical Andes biodiversity hotspots: a preliminary analysis

Sahotra Sarkar · Víctor Sánchez-Cordero · Maria Cecilia Londoño · Trevon Fuller

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Abstract Using IUCN Red List species as biodiversity surrogates, supplemented with additional analyses based on ecoregional diversity, priority areas for conservation in Mesoamerica, Chocó, and the Tropical Andes were identified using the methods of systematic conservation planning. Species' ecological niches were modeled from occurrence records using a maximum entropy algorithm. Niche models for 78 species were refined to produce geographical distributions. Areas were prioritized for conservation attention using a complementarity-based algorithm implemented in the ResNet software package. Targets of representation for Red List species were explored from 10 to 90% of the modeled distributions at 10% increments; for the 53 ecoregions, the target was 10% for each ecoregion. Selected areas were widely dispersed across the region, reflecting the widespread distribution of Red List species in Mesoamerica, Chocó, and the Tropical Andes, which underscores the region's importance for biodiversity. In general, existing protected areas were no more representative of biodiversity than areas outside them. Among the countries in the region, the protected areas of Belize performed best and those of Colombia and Ecuador worst. A high representation target led to the selection of a very large proportion of each country except Colombia and Ecuador (for a 90% target, 83–95% of each country was selected). Since such large proportions of land cannot realistically be set aside as parks or reserves, biodiversity conservation in Mesoamerica, Chocó, and the Tropical Andes will require integrative landscape management which combines human use of the land with securing the persistence of biota.

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Keywords Area prioritization · Ecological niche models · Mesoamerica · Tropical Andes · Chocó · Reserve selection algorithms · ResNet · Systematic conservation planning

Introduction

Mesoamerica, Chocó, and the Tropical Andes, bounded in the north by the Isthmus of Tehuantepec (in southern Mexico), ending in the south and south–east in northern Colombia and Ecuador, and limited to the west and north–east by the Pacific Ocean and the Caribbean Sea, respectively, have long been recognized as being among the Earth's most important centers of species diversity and endemism (Myers et al. 2000; Calderón et al. 2004). Though this region occupies <0.5% of Earth's terrestrial surface, it is estimated to contain about 7–10% of its species (Miller et al. 2001). The area contains more than 20,000 plant species by most estimates, of which 20% are likely to be endemic to the region (CCAD 2002). Similarly, about a third of its >500 mammal species are endemic, and there are over 1,000 recorded bird species (CCAD 2002). There are several factors responsible for the high biodiversity of the region, starting with its geological origins as an archipelago near large species-rich continental landmasses (Hooghiemstra et al. 1992; Kappelle et al. 1992), and accentuated by the high environmental diversity within its boundaries. A large variation in precipitation (500–7,500 mm year^{-1}) is superimposed on discontinuous mountain chains (0–3,820 m; average annual temperature, 32.5–7.5°C) which separate the Pacific and Caribbean basins (Calderón et al. 2004). The soil is rich and volcanic, encouraging the proliferation of biota, and ever since Mesoamerica's Inter-Oceanic Channel was closed some 3–5 Mya with the formation of the Isthmus of Panama (Donnelly 1989), immigration from both the north and the south have led to the assembly of exceptional faunal and floral diversity in a relatively tiny amount of land (Raven and Axelrod 1974). This assembly was part of what has been called "the great American biotic interchange" (Stehli and Webb 1985).

Because of this high species richness and endemism, Mesoamerica, Chocó, and the Tropical Andes have been a major focus of biodiversity conservation interest for several decades (Jukofsky 1992; Miller et al. 2001). In spite of many well-publicized initiatives for conservation both nationally (Sarukhán et al. 1996; Sarukhán and Dirzo 2001; Evans 1999; Fandiño-Lozano and Wyngaarden 2005b) and trans-nationally (CCAD 1993; Illueca 1997; Miller et al. 2001; Calderón et al. 2004), the biodiversity of Mesoamerica is under continued threat from a variety of factors including a deforestation rate of about 1% year^{-1} from 2000 to 2005 (FAO 2005), a human population growth rate of over 2% year^{-1} (Miller et al. 2001), and the reliance of the majority of the human population on biological resources taken directly from the wild (Miller et al. 2001). According to some estimates, about 30% of the region's primary and secondary vegetation have been completely transformed to agricultural fields and urban settlements (Bryant et al. 1997). Other habitats, including coral reefs, mangroves, wetlands, and grasslands have suffered similar losses (Burke et al. 2000; Matthews et al. 2000; Revenga et al. 2000). Poverty is a major factor in maintaining threats to all habitats, with almost half the human population living below the poverty line, and much of it lacking access to basic health care, education, and safe water (Miller et al. 2001).

These threats, the high species richness, and the high level of endemism led to the designation of Mesoamerica, Chocó, and the Tropical Andes as three of the 25 global biodiversity "hotspots" (Myers et al. 2000). Conservation planning for this region as an integral whole has been a priority for several organizations for almost two decades. In 1990, the New York Zoological Society (which became the Wildlife Conservation Society in 1993) along with the Caribbean Conservation Corporation launched the Paseo Pantera initiative as a "cooperative agreement" with USAID (Jukofsky 1992). The aim of the project was to establish a forest corridor running from Panama to the Selva Maya at the intersection of Mexico, Guatemala, and Belize. This corridor was supposed to provide continuous habitat for potential dispersal of panthers throughout the region (Carr et al. 1994). More importantly, the initiative marked the return of peace to Mesoamerica and the potential for regional co-operation on environmental problems (Illueca 1997).

The Paseo Pantera initiative provided the background for the trans-governmental Mesoamerican Biological Corridor (MBC) project. As early as 1989, with the prospect of peace looming in the foreground, the heads-of-state of several countries in the region signed a "Charter Agreement for the Protection of the Environment" which established the Comisión Centroamericana de Ambiente y Desarrollo (CCAD 1989). Guatemala, Belize, El Salvador, Honduras, Nicaragua, Costa Rica, and Panama were full members, with Mexico having observer status (Miller et al. 2001). Following the adoption of the Rio Convention on Biodiversity in 1992, CCAD promoted several regional initiatives for forest management and establishment of priority areas for biodiversity conservation (CCAD 1993). Sustainable development was soon added to the agenda (CCAD 1994).

Between 1992 and 1997, through discussions primarily at the governmental level, the Paseo Pantera initiative was embedded into an agenda that integrated conservation and sustainable development in what was initially called the Central American Biological Corridor (Miller et al. 2001). Its geographical focus was eventually expanded to include the five southern states of Mexico (Campeche, Chiapas, Quintana Roo, Tabasco, and Yucatan) after which it was renamed as the Mesoamerican Biological Corridor (MBC). The MBC was endorsed by regional heads-of-state in 1997 and responsibility for planning and co-ordinating its implementation was assigned to the CCAD. By 1998 CCAD had prepared a proposal, "Program for the Consolidation of the Mesoamerican Biological Corridor," to submit for funding to the United Nations Development Program (UNDP)'s Global Environment Facility (GEF) and the German Gesellschaft für Technische Zusammenarbeit (GTZ) (Miller et al. 2001). Funding was approved in 1999, and an MBC Regional Office Coordinating Unit was opened in Managua; since then, additional funds have come from other sources, including the World Bank [see Global Transboundary Protected Area Network (http://www.tbpa.net/case_10.htm, last accessed 17 October 2008)].

Transnational non-governmental organizations (NGOs) have also begun developing regional conservation plans which are intended to complement the planning initiatives of the official MBC project. Conservation International (CI) has begun the process of identifying "Key Biodiversity Areas" (Conservation International 2004) on the basis of habitat requirements of critically endangered (CR) species and those with restricted ranges (Jaime Garcia-Moren, personal communication). In 2004, The Nature Conservancy (TNC) published a "portfolio" of both terrestrial and marine "action sites" for 27 terrestrial and five marine ecoregions (as defined by Olson et al. 2001) as well as coarser-scale "action areas" (Calderón et al. 2004). As biodiversity surrogates (which it calls "targets") TNC used 403 terrestrial, 25 freshwater, and 34 coastal-marine ecological communities and systems. The viability of the surrogates in each area was assessed by experts on the basis of size, condition, and landscape context (connectivity and intactness) following the methodology of Morris et al. (1999). Experts selected a network of priority sites so that the best occurrence and at least ten viable occurrences of each surrogate was achieved, resulting in a portfolio of 78 terrestrial, 50 freshwater, and 15 coastal-marine sites. These sites were then aggregated at a coarser spatial scale to identify 20 priority areas (Calderón et al. 2004).

The analysis reported here uses systematic conservation planning (SCP) methods (sensu, Margules and Pressey 2000; Cowling and Pressey 2003; Sarkar 2005; Margules and Sarkar 2007) to identify priority areas for conservation planning using IUCN Red List species and ecoregions as biodiversity surrogates. These priority areas are then compared to the existing protected areas of the region to assess the performance of the latter with respect to the inclusion of habitats of Red List species and the diverse ecoregions of Mesoamerica, Chocó, and the Tropical Andes. The analysis is limited to the terrestrial context, including freshwater and coastal but not marine habitats. The study region consists of 53 ecoregions (sensu, Olson et al. 2001). Species' distributions were modeled using a maximum entropy algorithm and the area prioritization used a complementarity-based algorithm. The use of such algorithms to supplement expert opinion and analysis is the main distinguishing feature of the SCP approach (Cowling et al. 2003; Margules and Sarkar 2007). This appears to be the first use of this approach in a trans-national context in Mesoamerica, Chocó, and the Tropical Andes. However, this analysis carries out a priority area identification only for a preliminary assessment of conservation goals and the performance of existing protected areas (PAs); it is not intended to prescribe management policies.

The SCP protocol envisions a set of stages to formulate a conservation plan for a region, including: stakeholder identification and involvement; data collection and assessment; modeling and corrective data treatment, when necessary; the choice of biodiversity surrogates, conservation targets, and goals; a review of existing conservation areas; prioritization of additional potential conservation areas; assessment of site viability; feasibility assessment of a conservation proposal weighed against competing demands for land; and the eventual implementation and periodic reassessment of a plan (Margules and Sarkar 2007). Since its purpose is assessment rather than the formulation of an implementation-oriented plan, this analysis does not include all stages of the SCP protocol. There is no explicit stakeholder involvement; however, preliminary identification of priority areas is important because it can guide the identification of potential stakeholders and help instigate their active involvement as more sophisticated implementation-oriented plans are developed. Feasibility assessment can only be performed with the full involvement of such stakeholders. Finally, biological viability analysis is only incorporated in this analysis in a rudimentary way by excluding anthropogenically transformed areas (see "Landscape GIS Models").

This analysis also differs from the past and ongoing transnational analyses of conservation in the region in including a larger study area than is customary, extending into Colombia and Ecuador. The delineation of the study area was based on the ecological definition of Mesoamerica, Chocó, and the Tropical Andes (as explained in "Study Region"). In contrast, the MBC project uses political boundaries because implementation must take place in political context. The World Resources Institute (WRI) also used the political delineation of the study area from the MBC project (Miller et al. 2001). While it is true that implementation depends on political boundaries, the presumption here is that ecological analyses should be based on ecological criteria and, after priority areas are identified in this way, the question of political implementation should be broached—this is the "feasibility assessment" stage of SCP. If political implementation of conservation plans is impractical in some areas, then the SCP process envisions the replacement of these areas with other (biologically) functionally equivalent areas with such areas also chosen using ecological criteria.

In contrast, TNC used ecoregions to delineate its study region but its analysis is restricted to a subset of the area analyzed here, ignoring parts of Mexico to the north and Colombia and Ecuador to the south. There are two additional and more important differences between

TNC's analysis and this one: (1) TNC chose to ignore species-level surrogates on the grounds that there were insufficient data on them. However, there is some good data available for a suite of Red List species and to ignore these altogether when identifying priority areas for conservation is unwarranted and TNC could have profitably supplemented their surrogate set with Red List species. Such a mixed strategy was followed by the Australian BioRap Project to develop a provisional conservation portfolio for Papua New Guinea when faced with sparse species-level data (Faith et al. 2001); and (2) even though TNC has advocated methods similar to SCP in other contexts (see Groves et al. 2002), its Mesoamerican analysis was based on expert judgment. Cowling et al. (2003) and others (Pressey 1994; Smith et al. 2006; Margules and Sarkar 2007) have pointed out the pitfalls of not using systematic (often algorithmic) methods to supplement expert opinion: systematic methods lead to repeatable analyses, permit detailed exploration of alternative choices of surrogates, targets, and goals, and enable explicit quantitative evaluation of results in meeting those targets and goals. Expert-based plans have also been known to produce plans that do not provide adequate representation of biodiversity surrogates and are often not optimal with respect to spatial economy (Sarakinos et al. 2001; Cowling et al. 2003). Good planning involves the use of both systematic methods and expert judgment; this analysis aims to provide some baseline results that may be evaluated by experts and then re-analyzed systematically. Expert judgment was used here to refine species' ecological niche models.

More specifically, the major aims of this analysis were (1) a broad identification of priority areas for Red List species in the Mesoamerica, Chocó, and Tropical Andes region, (2) an identification of priority areas that also represent the ecoregional diversity of Mesoamerica, Chocó, and the Tropical Andes; and (3) an analysis of the performance of existing protected areas (PAs) with respect to both these goals. A subsidiary goal was to analyze the portfolio produced by TNC. However, these results should not be interpreted as recommending individual areas for immediate protection. Rather, the analysis should form the basis for discussion with stakeholders about opportunities and constraints, and experts about appropriate surrogates, targets, and goals. Simultaneously, there must be significantly more data collection and assessment so that more sophisticated analyses geared towards producing implementation-oriented results can be performed.

Landscape features

Study region

For this study the Mesoamerica, Chocó, and Tropical Andes region was defined using the ecoregional classification of Olson et al. (2001), which is a refinement Dinerstein et al. (1995) and Ricketts et al. (1995). The study region consisted of 53 ecoregions in 10 countries (Fig. 1): Mexico, Belize, Guatemala, Honduras, El Salvador, Nicaragua, Costa Rica, Panama, Colombia, and Ecuador. Marine habitats were excluded from consideration. Major islands were included: Isla Cozumel and Isla del Carmen in Mexico; Ambergris Caye and the Turneffe Islands in Belize; Isla de Roatán in Honduras; and Isla de Coiba and Isla del Rey (in the Archipelago de Las Perlas) in Panama. The Pacific Ocean and the Caribbean Sea define the western and north-eastern boundaries of the study region.

The northern boundary was delineated within the Isthmus of Tehuantepec in Mexico, including the Balsas depression and ecoregions associated with it because of their high biogeographic affinity with Mesoamerican biota (Morrone 2005; Escalante et al. 2007). This boundary marks the southern range limits of several Nearctic taxa and northern range limits

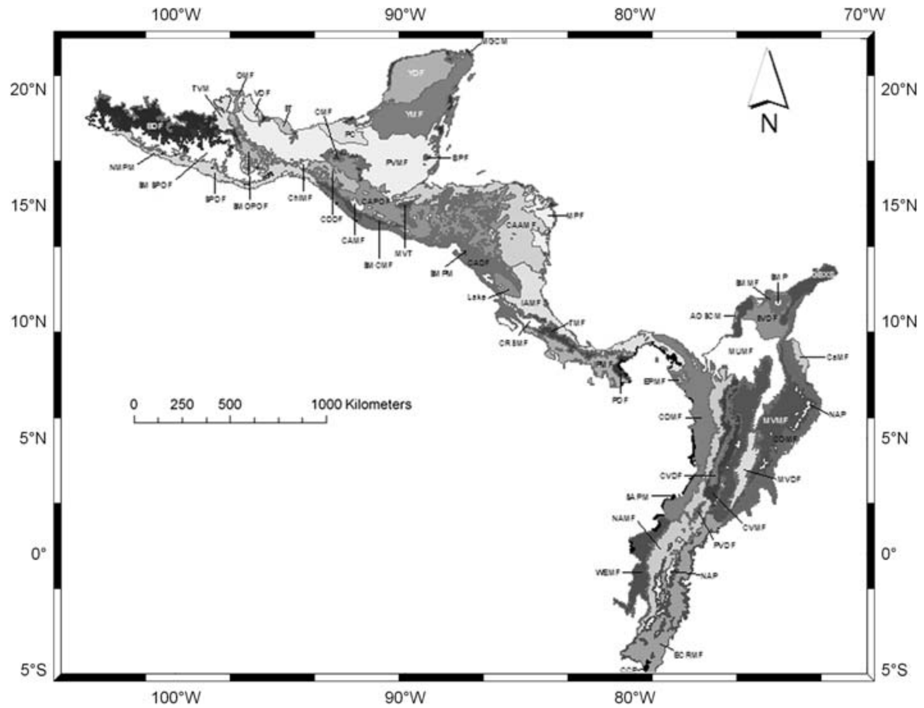


Fig. 1 Study region. AOSCM, Amazon-Orinoco-Southern Caribbean mangroves; BDF, Balsas dry forests; BPF, Belizian pine forests; CaMF, Catatumbo moist forests; CVDF, Cauca Valley dry forests; CVMF, Cauca Valley montane forests; CAAMF, Central American Atlantic moist forests; CADF, Central American dry forests; CAMF, Central American montane forests; CAPOF, Central American pine-oak forests; CDDF, Chiapas depression dry forests; CMF, Chiapas montane forests; ChiMF, Chimalapas montane forests; CDMF, Chocó-Darién moist forests; CCP, Cordillera Central páramo; COMF, Cordillera Oriental montane forests; CRSMF, Costa Rican seasonal moist forests; ECRMF, Eastern Cordillera real montane forests; EPMF, Eastern Panamanian montane forests; GBXS, Guajira-Barranquilla xeric scrub; IAMF, Isthmian-Atlantic moist forests; IPMF, Isthmian-Pacific moist forests; Lake, Lake; MVDF, Magdalena Valley dry forests; MVMF, Magdalena Valley montane forests; MUMF, Magdalena-Urabá moist forests; MGCM, Mesoamerican Gulf-Caribbean mangroves; MPF, Miskito pine forests; MVT, Motagua Valley thornscrub; NAP, Northern Andean páramo; NMPM, Northern Mesoamerican Pacific mangroves; NAMF, Northwestern Andean montane forests; OMF, Oaxacan montane forests; PDF, Panamanian dry forests; PC, Pantanos de Centla; PVDF, Patía Valley dry forests; PVMF, Petén-Veracruz moist forests; SMMF, Santa Marta montane forests; SMP, Santa Marta páramo; ST, Sierra de los Tuxtlas; SMCMF, Sierra Madre de Chiapas moist forests; SMOPOF, Sierra Madre de Oaxaca pine-oak forests; SMSPOF, Sierra Madre del Sur pine-oak forests; SVDF, Sinú Valley dry forests; SAPM, South American Pacific mangroves; SMPM, Southern Mesoamerican Pacific mangroves; SPDF, Southern Pacific dry forests; TMF, Talamancan montane forests; TVM, Tehuacán Valley matorral; VDF, Veracruz dry forests; WEMF, Western Ecuador moist forests; YDF, Yucatán dry forests; YMF, Yucatán moist forests

of several Neotropical taxa. Such Nearctic taxa consist of a wide range of floristic and faunistic groups including terrestrial vertebrates (Fa and Morales 1991; Escalante et al. 2007), lepidoptera (Peterson et al. 1999), cacti (Bravo et al. 1999), and composites and bryophytes (Delgadillo 2000; Delgadillo and Villaseñor 2002). The Neotropical taxa include mammals such as shrews, armadillo, muroid rodents (Briones and Sánchez-Cordero 2004), as well as amphibians, reptiles, and birds (Casas-Andreu et al. 1996; Peterson et al. 1993, 1999, 2004). From the west to the east, 11 ecoregions comprise the northern boundary: Balsas dry forests, Sierra Madre del Sur pine-oak forests, Oaxaca montane forests, Sierra

Madre de Oaxaca, Veracruz dry forests, Tehuacán Valley matorral, Mesoamerican Gulf-Caribbean mangroves, Sierra de los Tuxtlas, Petén-Veracruz moist forests, Pantanos de Centla, and Yucatan dry forests. Three other ecoregions could have potentially been included but were excluded because recent work, based on mammal distributions, has determined that these are part of a separate biogeographical region (Escalante et al. 2007): these are the Trans-Mexican volcanic belt, the Jalisco dry forests, and the Veracruz moist forest.

The southern and south-eastern boundaries were taken to lie in northern Ecuador and Colombia because the ranges of several Mesoamerican species, including endemic species of conservation concern, end there. These include mammal species, especially rodents such as heteromyids and peromiscines (Anderson et al. 2002a, b; Anderson 2003), bats (Albuja 1992, 1999), birds (Cracraft 1985; Best and Kessler 1995; Joseph and Stockwell 2002), and other terrestrial vertebrates (Albuja et al. 1980; Jarrín-V 2001). More importantly, this is where the Andes split into three separate ranges: Cordillera Occidental, Cordillera Central, and Cordillera Oriental. This topographical transition was used to define these boundaries. The ecoregions at the boundary are those that intersect with the three ranges emerging from the Andes. From west to east, the southern boundary ends with three ecoregions in Ecuador: the Eastern Cordillera Real montane forests, Northwestern Andean montane forests, and Western Ecuador moist forests. The south-eastern boundary continues in Ecuador and Colombia ending with three more ecoregions: Cordillera Oriental montane forests, Catatumbo moist forests, and Guajira-Barranquilla xeric scrub.

The study area includes 809 existing protected areas (PAs) as identified by the World Data Base on Protected Areas (http://www.unep-wcmc.org/wdpa/, last accessed 22 June 2007); 557 of these are classified by the World Conservation Union (IUCN) under one of their categories, I–VI, while 252 other PAs, although being so designated in the different countries, are yet to be placed under any of the IUCN categories. These other PAs and the PAs from all IUCN categories (I–VI) were included in the analysis with no distinction made between levels of protection since there was no information available at a regional scale on the relative performance of the different categories. While it is customary wisdom that strictly protected categories (I and II) perform better than the others, recent work in Mexico indicates that Biosphere Reserves (category VI) perform better than National Parks (Sánchez-Cordero and Figueroa 2007; Figueroa and Sánchez-Cordero 2008).

Table 1 shows the area and percentage area of each ecoregion in the total study area and the percentage of the ecoregion already in a protected area. Table 2 shows the area and percentage of each ecoregion present in each country, the percentage of each ecoregion represented within each country, and the percentage of that ecoregion protected within each country.

Biodiversity surrogates

SCP requires the identification of biodiversity surrogates, that is, quantifiable features of biodiversity on which data can be realistically obtained for use in planning protocols. In the terminology of Sarkar and Margules (2002) (see also, Sarkar 2002; Margules and Sarkar 2007), "true" surrogates are those such features that are themselves the components of biodiversity targeted for conservation; "estimator" surrogates are environmental or biological features which can be used to estimate the representation of true surrogates in conservation areas. In this study, species at risk of extinction, as identified by the IUCN Red List (http://www.iucnredlist.org, last accessed 21 June 2007), were taken as true surrogates which is uncontroversial. However, distributional data were available for only 9% of these species, and only 2% could be reliably modeled. Moreover, most of these were

Table 1 Ecoregions of Mesoamerica, relative prevalence, and representation in protected areas

Ecoregion	Area (km^2)	Area of Mesoamerica (%)	Ecoregion in protected areas (%)	Transformation (%)
Petén-Veracruz moist forests	148770.721	8.963	16.715	40.921
Central American pine-oak forests	111342.561	6.708	4.357	50.422
Magdalena Valley montane forests	105053.236	6.329	1.08	13.759
Central American Atlantic moist forests	89473.642	5.39	10.754	22.927
Northwestern Andean montane forests	81164.133	4.89	5.07	14.429
Magdalena-Urabá moist forests	76741.005	4.623	0.448	40.670
Eastern Cordillera real montane forests	76445.092	4.606	5.05	5.116
Chocó-Darién moist forests	72846.025	4.389	5.818	19.057
Yucatán moist forests	69533.519	4.189	4.011	8.916
Central American dry forests	67519.839	4.068	1.227	60.957
Balsas dry forests	62441.917	3.762	1.114	35.831
Sierra Madre del Sur pine-oak forests	61173.904	3.686	0.426	18.461
Cordillera Oriental montane forests	58681.511	3.535	4.676	5.753
Isthmian-Atlantic moist forests	57820.55	3.483	6.411	31.904
Yucatán dry forests	49723.833	2.996	0.366	16.653
Southern Pacific dry forests	41790.925	2.518	0.356	49.914
Western Ecuador moist forests	33886.716	2.042	0.661	17.801
Cauca Valley montane forests	32055.07	1.931	0.558	16.092
Northern Andean páramo	29635.932	1.785	3.988	2.729
Isthmian-Pacific moist forests	28955.537	1.744	1.556	49.568
Guajira-Barranquilla xeric scrub	27462.73	1.655	3.968	21.043
Sinú Valley dry forests	24980.499	1.505	0.388	44.107
Mesoamerican Gulf-Caribbean mangroves	21545.447	1.298	4.324	11.801
Magdalena Valley dry forests	19635.192	1.183	0.003	10.832
Miskito pine forests	18053.745	1.088	0.482	8.856
Pantanos de Centla	17082.886	1.029	1.865	58.292
Talamancan montane forests	16341.725	0.985	4.285	11.295
Sierra Madre de Oaxaca pine-oak forests	14345.454	0.864	0.323	26.423
Chiapas depression dry forests	14021.603	0.845	0.105	77.723
Central American montane forests	13299.105	0.801	1.368	42.736
Sierra Madre de Chiapas moist forests	11258.153	0.678	0.66	60.214
Costa Rican seasonal moist forests	10628.901	0.64	0.441	32.716
Tehuacán Valley matorral	9892.221	0.596	0.687	10.643
Lake	8014.42	0.483	0.121	1.183
Oaxacan montane forests	7600.825	0.458	0.001	47.172
Cauca Valley dry forests	7344.886	0.443	0	17.552
Catatumbo moist forests	6764.966	0.408	0.342	18.247
South American Pacific mangroves	6628.151	0.399	0.42	15.985
Veracruz dry forests	6610.969	0.398	0.001	60.874
Southern Mesoamerican Pacific mangroves	6281.579	0.378	0.881	24.155
Chiapas montane forests	5778.264	0.348	0.109	64.325

Table 1 continued

Ecoregion	Area (km^2)	Area of Mesoamerica (%)	Ecoregion in protected areas (%)	Transformation (%)
Panamanian dry forests	5070.623	0.305	0.065	84.418
Santa Marta montane forests	4784.658	0.288	1.467	0.381
Sierra de los Tuxtlas	3825.368	0.23	0.639	67.993
Eastern Panamanian montane forests	3044.012	0.183	0.914	59.566
Belizian pine forests	2830.694	0.171	0.423	16.648
Amazon-Orinoco-Southern Caribbean mangroves	2509.693	0.151	0.238	12.307
Motagua Valley thornscrub	2336.769	0.141	0.189	87.720
Patía Valley dry forests	2270.689	0.137	0	4.821
Chimalapas montane forests	2083.674	0.126	0.056	55.642
Santa Marta páramo	1243.629	0.075	0.53	0.000
Northern Mesoamerican Pacific mangroves	720.046	0.043	0.012	12.550
Cordillera Central páramo	500.16	0.03	0.051	0.219
Total	1659847.404	100	100	28.252

plants. To address this deficiency, the 53 ecoregions defined by Olson et al. (2001) were also added as estimator surrogates in a subsequent analysis. This assumes that representing the full variety of ecoregions in conservation areas will lead to the representation of the Red List species for which no modeled distributions were available (a more detailed classification into ecoregions—that is, one with more categories—which would be more desirable was not available at the regional scale).

For the 10 countries in the study region, the total number of species in the three categories of the IUCN Red List, CR, Endangered (EN), and Vulnerable (VU) are, for the four orders of vertebrates: Amphibians—CR 222, EN 286, VU, 184; Mammals—CR 19, EN 53 VU 67; Birds—CR 26, EN 62, VU 112; Reptiles—CR 13, EN 8, VU 22; and for Plants—CR 372, EN 916, VU 1271. In total there are 3,633 such species. Note that these numbers include areas outside the study region in the cases of Mexico, Colombia, and Ecuador. Table S1 lists the 333 species for which occurrence data are available in publicly accessible data bases (see "Acknowledgments"). It includes the number of geo-referenced data points available and the number of those points that are post-1980. Only post-1980 data were used to model species' distributions because the large-scale deforestation of the previous decades is known to have significantly altered the land cover of Mesoamerica (Utting 1997).

Methods of analysis

Landscape GIS models

Maps of ecoregions were obtained from the World Wide Fund for Nature (http://www.worldwildlife.org/science/data/terreco.cfm, last accessed 22 June 2007). The study region was divided into 0.02° × 0.02° longitude × latitude cells. This resulted in 343,383 cells with an average area of 4.818 km^2 (SD = 0.098; max = 4.946; min = 4.599). All point

Table 2 Percentage of country occupied by each ecoregion and the percentage of each ecoregion in a country

Ecoregion	Area (km^2)	Total country (%)	Total ecoregion in country (%)	Ecoregion in country's protected areas (%)	Transformation (%)
Belize	21829.006				18.022
Belizian pine forests	2825.374	13.114	99.812	12.158	16.525
Central American Atlantic moist forests	1.532	0.007	0.002	0.019	0.000
Mesoamerican Gulf-Caribbean mangroves	2434.81	9.886	11.301	5.4	9.105
Petén-Veracruz moist forests	16511.063	76.732	11.098	82.423	19.458
Yucatán moist forests	56.228	0.262	0.081	0	58.230
Colombia	510164.522				18.571
Amazon-Orinoco-Southern Caribbean mangroves	2509.693	0.492	100	0.994	12.307
Catatumbo moist forests	6764.966	1.326	100	1.428	18.247
Cauca Valley dry forests	7344.886	1.44	100	0	17.552
Cauca Valley montane forests	32055.07	6.283	100	2.328	16.092
Central American dry forests	8.683	0.002	0.013	0.005	0.000
Chocó-Darién moist forests	59479.251	11.659	81.651	15.753	14.669
Cordillera Oriental montane forests	58681.511	11.502	100	19.507	5.753
Eastern Cordillera real montane forests	10712.771	2.1	14.014	2.206	8.813
Eastern Panamanian montane forests	966.067	0.189	31.737	0.952	67.306
Guajira-Barranquilla xeric scrub	27462.73	5.383	100	16.553	21.043
Magdalena Valley dry forests	19635.193	3.849	100	0.013	10.832
Magdalena Valley montane forests	105053.236	20.592	100	4.503	13.759
Magdalena-Urabá moist forests	76741.005	15.042	100	1.869	40.670
Northern Andean páramo	14205.196	2.784	47.932	8.986	2.002
Northwestern Andean montane forests	48962.26	9.597	60.325	13.954	15.240
Patía Valley dry forests	2270.689	0.445	100	0	4.821
Santa Marta montane forests	4784.658	0.938	100	6.119	0.381
Santa Marta páramo	1243.629	0.244	100	2.21	0.000
Sinú Valley dry forests	24980.499	4.897	100	1.621	44.107
South American Pacific mangroves	3917.766	0.768	59.108	0.997	6.230
Western Ecuador moist forests	2384.764	0.467	7.037	0.002	14.726
Costa Rica	50531.604				24.963
Central American dry forests	5953.959	11.783	8.818	6.718	56.314
Costa Rican seasonal moist forests	8513.287	16.847	80.096	6.111	27.042
Isthmian-Atlantic moist forests	16687.546	33.024	28.861	25.566	21.071

Table 2 continued

Ecoregion	Area (km^2)	Total country (%)	Total ecoregion in country (%)	Ecoregion in country's protected areas (%)	Transformation (%)
Isthmian-Pacific moist forests	9166.819	18.141	31.658	16.425	26.206
Mesoamerican Gulf-Caribbean mangroves	271.807	0.538	1.262	1.61	3.919
Southern Mesoamerican Pacific mangroves	762.778	1.51	12.143	0.825	20.760
Talamancan montane forests	9175.409	18.158	56.147	42.745	9.499
Ecuador	146261.303				9.226
Cordillera Central páramo	500.16	0.342	100	0.575	0.219
Eastern Cordillera real montane forests	65732.321	44.942	85.986	50.815	4.513
Northern Andean páramo	15430.736	10.55	52.068	20.616	3.398
Northwestern Andean montane forests	32201.873	22.017	39.675	19.382	13.195
South American Pacific mangroves	894.262	0.611	13.492	1.191	8.139
Western Ecuador moist forests	31501.952	21.538	92.963	7.421	18.034
El Salvador	20601.449				56.337
Central American dry forests	8289.927	40.24	12.278	36.25	61.962
Central American montane forests	950.425	4.613	7.147	23.343	37.344
Central American pine-oak forests	10748.755	52.175	9.654	35.476	56.127
Sierra Madre de Chiapas moist forests	74.343	0.361	0.66	1.12	50.524
Southern Mesoamerican Pacific mangroves	537.998	2.611	8.565	3.811	8.203
Guatemala	109316.807				40.061
Central American Atlantic moist forests	8436.563	7.718	9.429	7.125	33.568
Central American dry forests	6610.744	6.047	9.791	0.262	66.542
Central American montane forests	5958.073	5.45	44.801	7.827	43.533
Central American pine-oak forests	29429.441	26.921	26.431	7.661	51.333
Chiapas depression dry forests	903.49	0.826	6.444	0	48.454
Chiapas montane forests	192.251	0.176	3.327	0.003	56.441
Mesoamerican Gulf-Caribbean mangroves	322.81	0.295	1.498	0.721	2.248
Motagua Valley thornscrub	2336.769	2.138	100	1.277	87.720
Petén-Veracruz moist forests	48160.639	44.056	32.372	74.229	23.493
Sierra Madre de Chiapas moist forests	5770.291	5.279	51.254	0.177	74.144
Southern Mesoamerican Pacific mangroves	1060.763	0.97	16.887	0.389	61.401

Table 2 continued

Ecoregion	Area (km^2)	Total country (%)	Total ecoregion in country (%)	Ecoregion in country's protected areas (%)	Transformation (%)
Yucatán moist forests	134.974	0.123	0.194	0.329	10.508
Honduras	111772.341				33.983
Central American Atlantic moist forests	33698.636	30.149	37.663	61.548	15.027
Central American dry forests	19081.171	17.071	28.26	1.354	58.241
Central American montane forests	5546.825	4.963	41.708	0	42.612
Central American pine-oak forests	44819.181	40.099	40.253	29.674	42.305
Mesoamerican Gulf-Caribbean mangroves	1914.746	1.713	8.887	2.686	5.152
Miskito pine forests	6038.202	5.402	33.446	3.444	4.419
Southern Mesoamerican Pacific mangroves	673.58	0.603	10.723	1.294	17.271
Mexico	488588.842				35.313
Balsas dry forests	62441.917	12.78	100	6.381	35.831
Belizian pine forests	5.321	0.001	0.188	0	81.657
Central American dry forests	3214.949	0.658	4.761	0.993	74.532
Central American pine-oak forests	15982.722	3.271	14.355	2.116	61.081
Chiapas depression dry forests	13118.113	2.685	93.556	0.598	79.739
Chiapas montane forests	5586.013	1.143	96.673	0.625	64.597
Chimalapas montane forests	2083.674	0.426	100	0.319	55.642
Mesoamerican Gulf-Caribbean mangroves	13103.022	2.682	60.816	17.814	15.536
Northern Mesoamerican Pacific mangroves	720.046	0.147	100	0.07	12.550
Oaxacan montane forests	7600.825	1.556	100	0.001	47.172
Pantanos de Centla	17082.886	3.496	100	10.685	58.292
Petén-Veracruz moist forests	84099.019	17.213	56.529	16.314	55.115
Sierra de los Tuxtlas	3825.368	0.783	100	3.662	67.993
Sierra Madre de Chiapas moist forests	5413.519	1.108	48.085	3.62	45.500
Sierra Madre de Oaxaca pine-oak forests	14345.454	2.936	100	1.853	26.423
Sierra Madre del Sur pine-oak forests	61173.904	12.521	100	2.441	18.461
Southern Mesoamerican Pacific mangroves	1431.826	0.293	22.794	1.728	16.317
Southern Pacific dry forests	41790.925	8.553	100	2.038	49.914
Tehuacán Valley matorral	9892.221	2.025	100	3.937	10.643
Veracruz dry forests	6610.969	1.353	100	0.002	60.874

Table 2 continued

Ecoregion	Area (km^2)	Total country (%)	Total ecoregion in country (%)	Ecoregion in country's protected areas (%)	Transformation (%)
Yucatán dry forests	49723.833	10.177	100	2.099	16.653
Yucatán moist forests	69342.317	14.192	99.725	22.704	8.873
Nicaragua	127729.399				32.076
Central American Atlantic moist forests	47336.911	37.06	52.906	47.455	26.655
Central American dry forests	24360.385	19.072	36.079	4.279	60.592
Central American montane forests	843.781	0.661	6.345	1.679	44.007
Central American pine-oak forests	10362.462	8.113	9.307	2.939	60.588
Costa Rican seasonal moist forests	2115.607	1.656	19.904	0.648	55.547
Isthmian-Atlantic moist forests	18590.544	14.555	32.152	31.321	22.219
Lake	8014.42	6.275	100	1.268	1.183
Mesoamerican Gulf-Caribbean mangroves	3179.198	2.489	14.756	5.679	4.824
Miskito pine forests	12015.543	9.407	66.554	2.019	11.086
Southern Mesoamerican Pacific mangroves	910.547	0.713	14.496	2.713	6.225
Panama	73052.102				48.378
Chocó-Darién moist forests	13366.774	18.298	18.349	29.061	38.584
Eastern Panamanian montane forests	2077.945	2.844	68.263	9.757	55.968
Isthmian-Atlantic moist forests	22542.46	30.858	38.987	26.246	47.911
Isthmian-Pacific moist forests	19788.718	27.088	68.342	7.624	60.390
Mesoamerican Gulf-Caribbean mangroves	319.054	0.437	1.481	0.765	4.835
Panamanian dry forests	5070.623	6.941	100	0.932	84.418
South American Pacific mangroves	1816.124	2.486	27.4	1.062	40.894
Southern Mesoamerican Pacific mangroves	904.087	1.238	14.393	1.366	28.416
Talamancan montane forests	7166.317	9.81	43.853	23.187	13.594

For Colombia, Ecuador, and Mexico, only that part which is within the study region is being considered (see "Methods")

occurrence data for the Red List species were resampled to this grid, reducing multiple records of a species in the same cell to one occurrence point. Land cover data for the study region was obtained from the Global Land Cover 2000 (Hansen et al. 2000). These data were classified into 60 categories (see Table 6 in the Supplementary Material). Cropland, intensive agriculture, and urban/built categories were interpreted as transformed areas while the rest of the categories were interpreted as being untransformed enough for potential inclusion within conservation areas as they are or for ecological restoration aimed at the persistence of biota. The Global Land Cover 2000 data had a resolution of $0.01° \times 0.01°$ which were resampled to a $0.02° \times 0.02°$ resolution. The percentage of transformation of each ecoregion is given in Table 1, and that for each ecoregion in each country is given in Table 2.

Data on the protected areas were obtained from the World Database of Protected Areas (WDPA; http://www.unep-wcmc.org/wdpa/, last accessed 22 June 2007) maintained by the United Nations Environment Programme and the World Conservation Monitoring Centre. Areas in Tables 1 and 2 were calculated using the equal area cylindrical projection (by using Projector! extension in ArcView 3.2).

Ecological niche models

Point occurrence data for animals and plants listed in the categories CR, EN and VU of the IUCN Red List were obtained from several scientific collections (see "Acknowledgments"). In order to reflect the current state of the habitat, only point occurrence data collected since 1980 were retained. Ecological niche models were then constructed for the 101 listed animal and plant species for which there were sufficient data (i.e., there were at least four occurrence records).

The Maxent software package (Version 2.2; Phillips et al. 2004, 2006) was used to construct ecological niche models. Maxent has been shown to be robust for modeling distributions from presence-only data (Elith et al. 2006). Following published recommendations, Maxent was run without the threshold feature and without duplicates so that there was at most one sample per pixel; linear, quadratic and product features were used (Phillips et al. 2004, 2006; Pawar et al. 2007). The convergence threshold was set to 1.0×10^{-5}, which is a conservative value based upon North American breeding bird survey data and small mammal data from Latin America (Phillips et al. 2004, 2006).

For climatic variables, 19 layers, each at a resolution of 30″ ($0.0083° \times 0.0083°$), were obtained from the WorldClim database (Hijmans et al. 2005; http://worldclim.org, last accessed 12 December 2006); These layers [Pawar et al. (2007) provide a complete list] were used along with elevation, slope, and aspect as the environmental variables. Elevation was obtained from the U. S. Geological Survey's Hydro-1K DEM data set (USGS 1998) and slope and aspect were derived from the DEM using the Spatial Analyst extension of ArcMap 9.0. All climate and spatial layers were clipped to an area bounded by 21° 30′ 30.27″ N by 5° 0′ 1.12″ S and 103° 30′ 52.49″ W by 71° 6′ 45.72″ W, a box containing the study region, and were resampled to a $0.02° \times 0.02°$ resolution in ArcGIS.

Niche model accuracy was evaluated by constructing the models using 75% of the available records, with the other 25% used for testing. At least four occurrence records are necessary to construct such a niche model. Model accuracy was determined using a receiver operating characteristic (ROC) analysis (Phillips et al. 2006). For all thresholds an ROC curve was produced with sensitivity plotted on the *y* axis and (1—specificity) plotted on the *x*-axis. The area under the curve (AUC) provides a measure of model performance. An optimal model would have an AUC of 1 while a model that predicted species occurrences at random would have an AUC of 0.5.

Following Pawar et al. (2007), only those niche models possessing an AUC >0.75 and a *P*-value of <0.05 (for the sensitivity and specificity tests) were retained for further use. Finally, the relative probabilities predicted by Maxent were converted into expected presences for a species by dividing all relative probabilities across the landscape by the highest relative probability achieved for that species. This normalization assumes that a species is at least present in the cell with the highest predicted relative probability of its occurrence.

Geographic projection of species' ecological niches must be modified to produce distributions because species may not occupy all regions ecologically suitable for them for a variety of reasons including barriers to dispersal and competition with other species (Soberón and Peterson 2005). Refinement removed from each model all cells with a predicted probability of <0.1. Stricter refinement protocols have sometimes been used; for instance, in many analyses by restricting to cells that are contiguous with those that have a reported occurrence of species (Soberón and Peterson 2005). However, because of the sparse available data for the study region, which is a result of inadequate sampling, such strict refinement is likely to exclude many areas in which a species is likely to be present. For the species used in this analysis, there is no suggestion of geographical barriers to their dispersal within Mesoamerica, Chocó, and the Tropical Andes. Expert knowledge was used to drop niche models which showed systematic over-prediction (see "Results").

Area prioritization and representation targets

Area prioritization was carried out with a heuristic complementarity-based algorithm implemented in the ResNet software package (Garson et al. 2002; Sarkar et al. 2002) because such algorithms are computationally fast while finding near-optimal solutions to problems (Csuti et al. 1997; Sarkar et al. 2004). The optimization problem to be solved is the minimum-area problem: finding the smallest set of areas in which all biodiversity surrogates meet their representation targets. ResNet selects areas to solve the minimum-area problem using a two-pass algorithm. In the version of ResNet that was used, the first pass uses complementarity to select the area with the largest total expected value for the presence of surrogates with unmet targets. Ties are broken by selecting one of the tied cells at random. Area selection terminates when the targeted representation for each species has been met. The second pass removes cells that may have become redundant with respect to achieving the representation targets due to later cell selection. Other versions of ResNet use both rarity and complementarity in the first pass; however, Sarkar et al. (2004) reported that using complementarity alone produces better results with probabilistic data.

The aim of this analysis was a broad identification of priority areas for Red List species and to analyze the performance of existing PAs, and not to recommend individual areas for immediate protection. Thus, a range of representation targets was used to identify priority areas without recommending a specific target. Those areas that are selected at low targets of representation of Red List species have higher priority values than those only selected at higher targets. Four prioritization scenarios were run in this analysis. Two different surrogate sets were used. In both, the Red List species were included, and targets of representation were set from 10 to 90% in 10% increments for each of the species. In the second, a uniform target of 10% was additionally set for the ecoregions. In general, targets of representation used in such prioritization protocols do not have full biological justification (Soulé and Sanjayan 1998; Margules and Sarkar 2007). Using a wide range of targets for the most important biodiversity surrogates in this analysis, the Red List species, avoids this problem. The 10% target for the ecoregions is also conventional (Margules and Sarkar 2007), though the Secretariat of the Convention on Biological Diversity (2002) set that target for each of the world's ecoregions. Hence the use of the 10% target here; it was assumed that higher targets for them would be unrealistic. Area prioritization was carried twice for each of these alternatives, once for the whole study area and once removing the transformed areas from the study region.

The effectiveness of each country's existing protected areas at representing biodiversity was estimated by calculating: (1) the proportion of the entire country selected by ResNet (hereafter p_c) and (2) the proportion of the country's protected areas so prioritized (hereafter p_{pa}). At each representation target, the following were computed: the point estimate of the difference in proportions ($p_c - p_{pa}$), the lower limit of the 95% confidence interval of the difference (hereafter L), and upper limit of the 95% confidence interval (hereafter U). If $p_c > p_{pa}$, $L > 0$, and $U > 0$, then the prioritized proportion of the entire country was significantly greater than the prioritized proportion of the country's protected areas. Conversely, if $p_c < p_{pa}$, $L < 0$, and $U < 0$, then the prioritized proportion of the country's protected areas was significantly greater than the prioritized proportion of the entire country.

The point estimate ($p_c - p_{pa}$) may be too liberal due to the non-independence of p_c and p_{pa} (Agresti 2002). To address this issue, separate contingency tables were constructed for the country and protected area data. In the former, cell$_{j1}$ represented the number of selected sites in country j and cell$_{j2}$ represented the number of unselected sites. In the latter, cell$_{j1}$ represented the number of selected sites in the protected areas of country j and cell$_{j2}$ represented the number of unselected sites in the protected areas. Pearson residuals >3 indicated that more sites were selected in a given category than expected (Simonoff 2003).

Results

Of the IUCN Red List species in the study region (that is, species that that are CR, EN, or VU), there was at least one occurrence record for 333 species in the accessible databases (see Table S1 of Supplementary Materials). However, at least four post-1980 occurrence records were only available for 101 species. Niche models were constructed for all of these; only 94 species had an AUC >0.75 (Table S1). Seven additional species were dropped because of large P-values. A further nine species were dropped from the analysis because of presumed overprediction, as described in "Ecological Niche Models". Whereas the modeled distribution predicted the species to be distributed across the entire study region, these species have restricted distributions as specified by IUCN Red List (http://www.iucnredlist.org, last accessed 21 June 2007): *Cyanolyca nana* is restricted in Mexico to Oaxaca, Queretaro, north Hidalgo, and central Veracruz; *Thorius narisovalis* is restricted in Mexico to north-central Oaxaca; *Lonchocarpus yoroensis* is restricted to Honduras, Mexico, and Nicaragua; *Mollinedia ruae* and *Lonchocarpus retiferus* are restricted to Honduras and Nicaragua; *Dendropanax sessiliflorus*, *Quararibea gomeziana*, and *Stenanona panamensis* are restricted to Costa Rica and Panama; and *Inga canonegrensis* is restricted to Costa Rica. For the other species, the two most common parameters producing the largest AUC when used independently (Explanatory Variable 1 in Table S1) were temperature seasonality (16 times) and precipitation of the wettest month (14 times) and the two most common parameters reducing the AUC most when excluded (Explanatory Variable 2 in Table S1) were altitude (24 times) and temperature (19 times). Table S1 shows these critical explanatory variables for all 78 species (5 CR; 21 EN; 52 VU) used in this analysis; there were 10 amphibian (3 CR; 5 EN; 2 VU), three bird (2 EN; 1 VU), three mammal (3 VU), and 62 plant (2 CR; 14 EN; 46 VU) species. This means 80% of the species used for the area prioritization were plants. We further refined our species list to 51 species that included 10 or more point locality records. The intersection between the set of sites prioritized to be put under a conservation plan when sites were selected to represent the 78 species with at least four records and the set of sites selected to represent the 51 species with at least 10 records was 81% on average (range: 74–92%;

Supplementary Material Table S5). In light of this, the use of species with at least 10 records does not appear to result in a significantly different conservation area network than the use of species with at least four records.

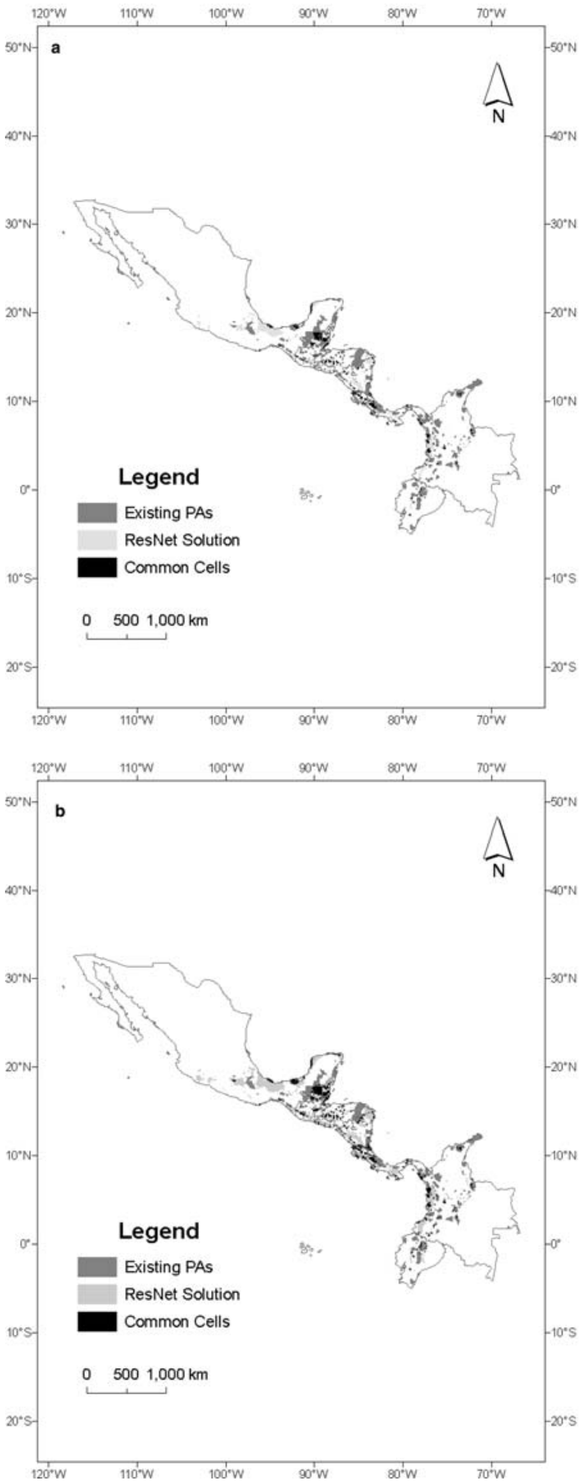
Figures 2 and 3 shows the ResNet solutions for the different targets; the total selected area increases as the representation targets increase. In Fig. 2, only Red List species are used as surrogates. The percentage of total area selected varied from 5.1% for the 10% target to 82.04% for the 90% target. The percentage of overlap between the selected area and the PAs ranged between 6.46 and 87.7% for these two targets. Figure 3 corresponds to Fig. 2 when the ecoregions are also included as surrogates (with a uniform representation target of 10%). The percentage of total area selected was now 9.96% for the 10% target and 82.03% for the 90% target. The percentage of overlap between the selected area and the PAs ranged between 10.97 and 87.63%. In both these figures selected cells are widely dispersed across the region and, as expected, the amount of land selected increases monotonically with the target.

Figure 4 shows the percentage of untransformed area in each country selected at different targets, and Fig. 5 shows the percentage of the country selected in untransformed areas, both under two scenarios: Red List species as surrogates, and both Red List species and ecoregions as surrogates. Figure 6 shows the percentage of the existing PAs selected as a function of the representation target under the two scenarios. Throughout the analyses, the results for Mexico, Colombia, and Ecuador only refer to areas within the study region. What is striking is that similar patterns are seen for the percentage of the country′s untransformed area and that of the existing PAs that is selected at different targets. When Red List species alone are used as biodiversity surrogates, the PA network of Belize performs better than those of all other countries at all representation targets in the sense that a larger fraction of it was selected in the ResNet runs. When ecoregions are also included as surrogates, Costa Rica and El Salvador performs better than Belize (Fig. 6). The existing PA network of Ecuador performs worst using Red List species as surrogates, followed by Mexico, Honduras and Colombia. At the higher targets the same conclusion once again holds when ecoregions are included as surrogates (Fig. 6). However, for all countries except Colombia, Ecuador, and Mexico, protecting 90% of the modeled distribution for just these 78 species takes up 82.3–99.7% of each country's area. For Ecuador, Colombia, and Mexico, 65.8–79.7% is selected. (Figs. 3, 4).

The most important priority areas are those selected at the lowest representation targets for biodiversity surrogates because these areas are needed even to maintain minimal representation of species at risk. This fact can also be used to assess the performance of the existing PAs. Areas within 189 of the existing PAs were selected even at the lowest target used (10%). However, in most cases very few cells were selected (Figs. 2a, 3a). For each country in the study area, Table S2 of the Supplementary Materials lists the PAs which have at least one cell represented in a solution at the 10% species representation target. In addition, Table S2 lists the PAs that have some cells represented at the 10% species and ecoregion representation target, the 50% species representation target, and so on.

For prioritizations using only species, as well as for prioritizations that included both the species and the ecoregions, the prioritized proportion of Guatemala's protected areas was significantly greater than the prioritized proportion of the entire country at all representation targets [range of ($p_{\mathrm{pa}} - p_{\mathrm{c}}$): 4.08–26.29%]. Conversely, for Panama, the prioritized proportion of the entire country selected was significantly greater than the prioritized proportion of the protected areas [range of ($p_{\mathrm{c}} - p_{\mathrm{pa}}$): 2.4–14.75%]. For the species only prioritizations at each representation target, the null hypothesis that equal proportions of each country were selected was rejected ($\chi^2 \geq 4704.88$), as was the hypothesis that equal

Fig. 2 Selected areas, Red List species as biodiversity surrogates: **a** target of representation for Red List species = 10%; **b** target = 20%; **c** target = 30%; **d** target = 40%; **e** target = 50%; **f** target = 60%; **g** target = 70%; **h** target = 80%; **i** target = 90%. *Black* areas correspond to cells that are common to the existing PAs and the ResNet solutions; the *dark grey* cells are the additional ones in the existing PAs; and the *light grey* cells are those in the ResNet solutions



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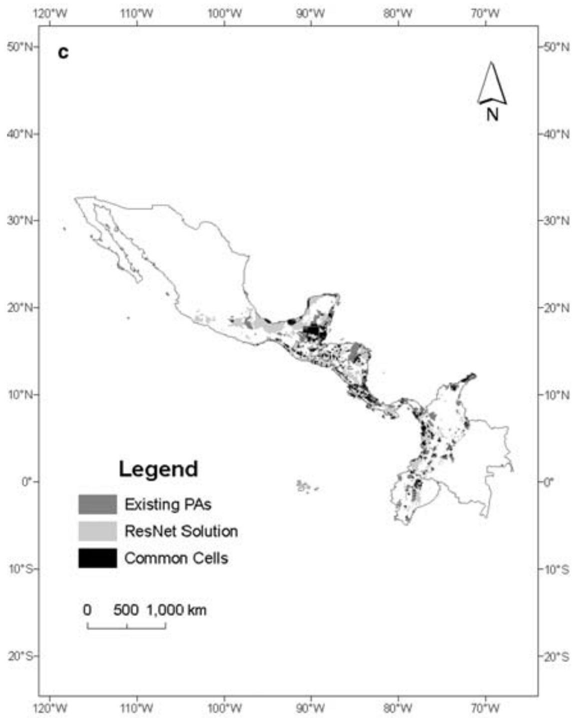

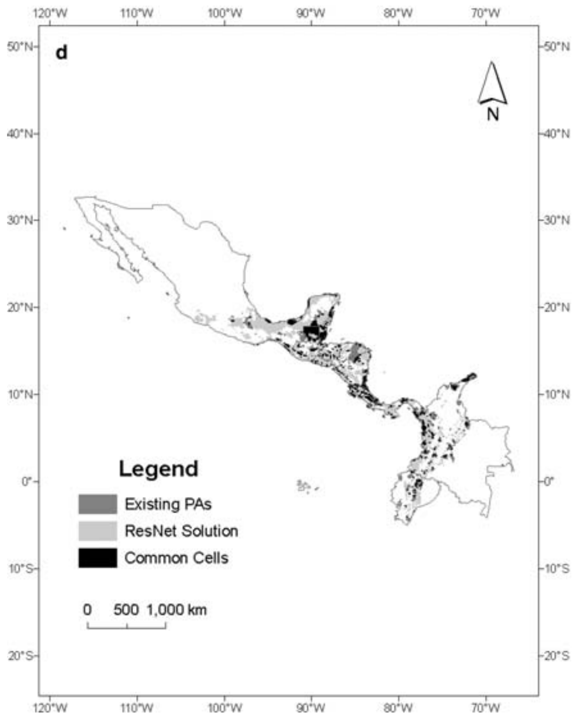

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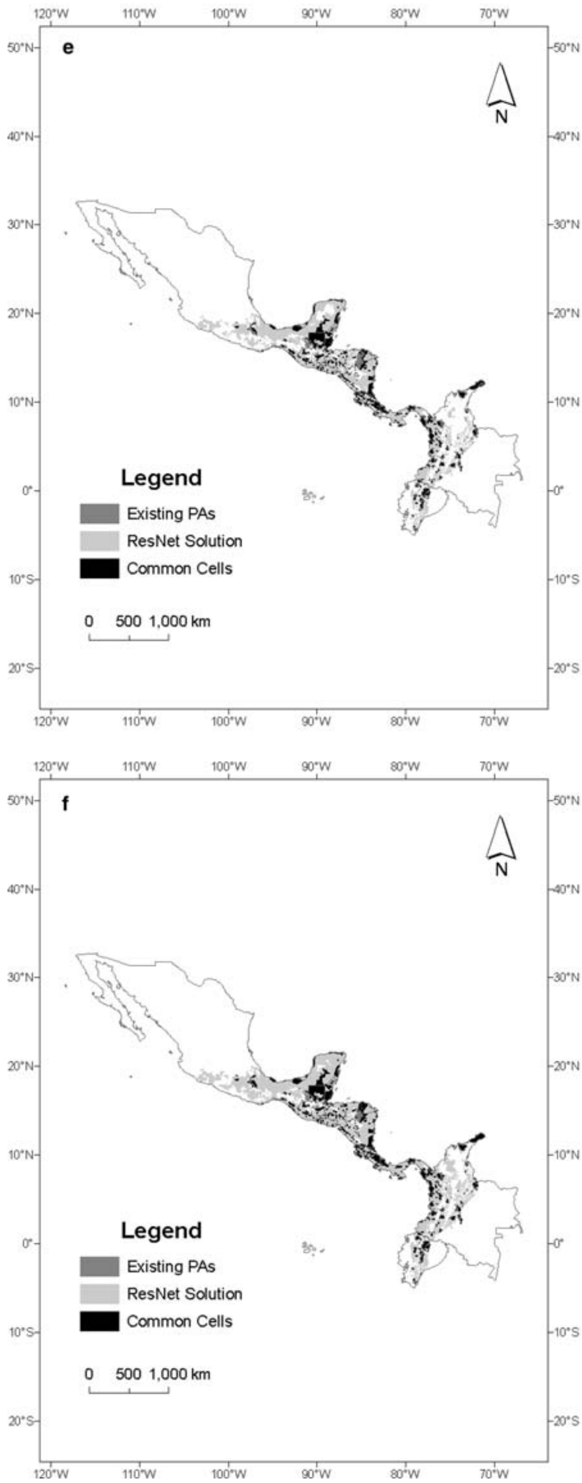

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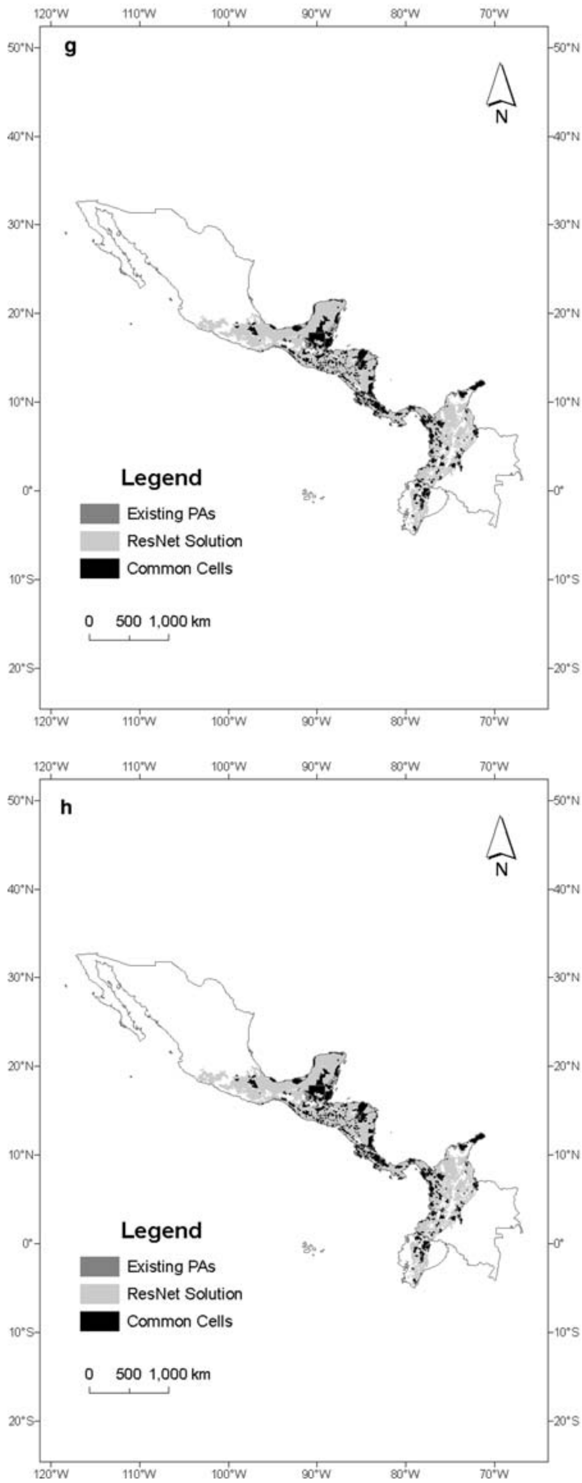

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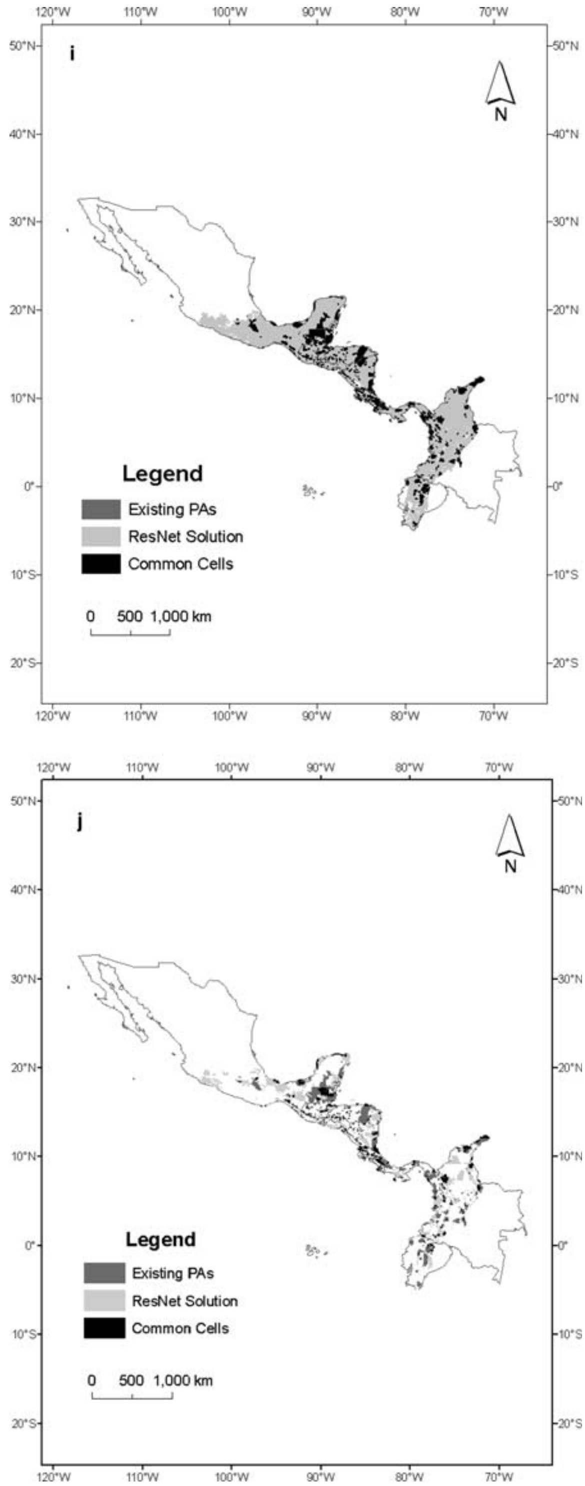
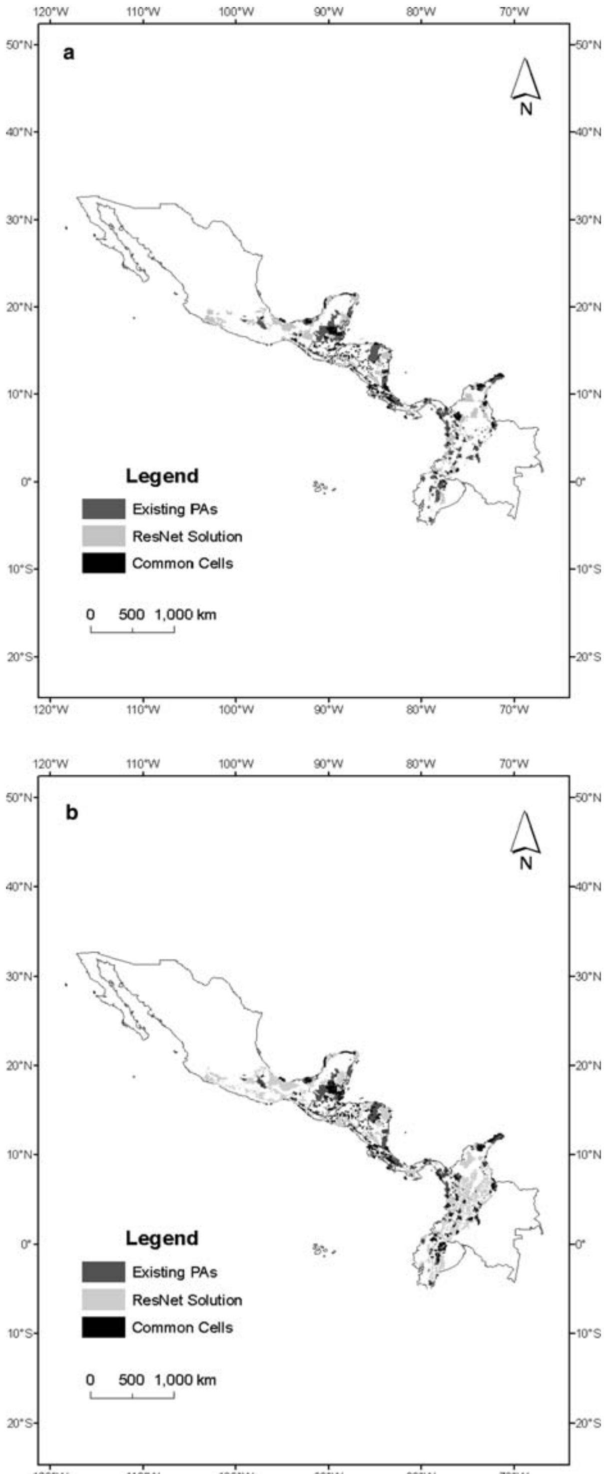


Fig. 3 Selected areas, Red List species and ecoregions as biodiversity surrogates: The target of representation was uniformly 10% of each ecoregion. For Red List species' targets and the interpretation of the cells, see the caption of Fig. 2



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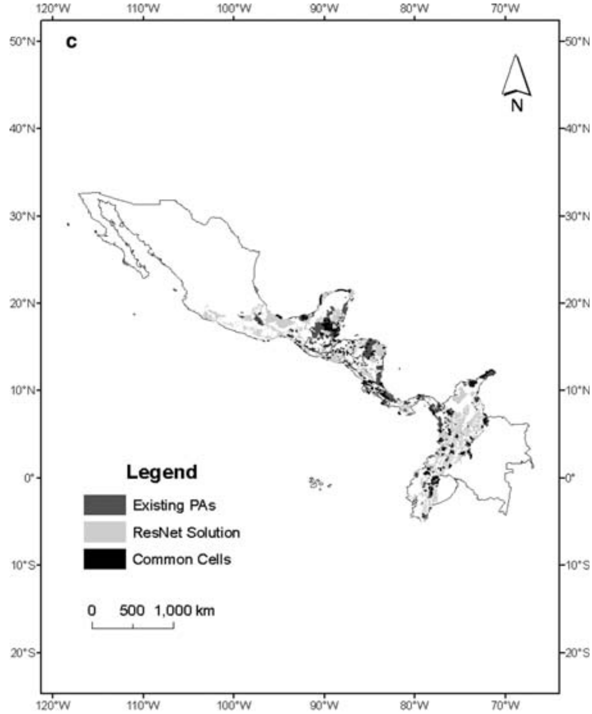

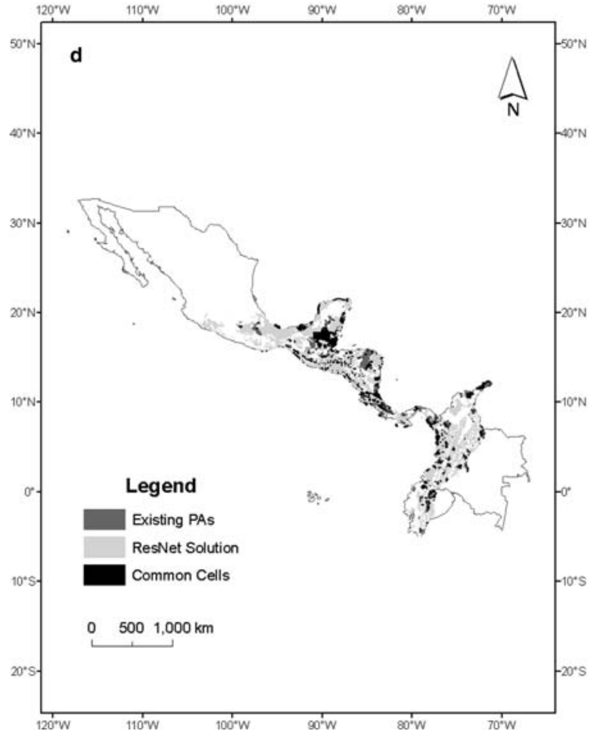

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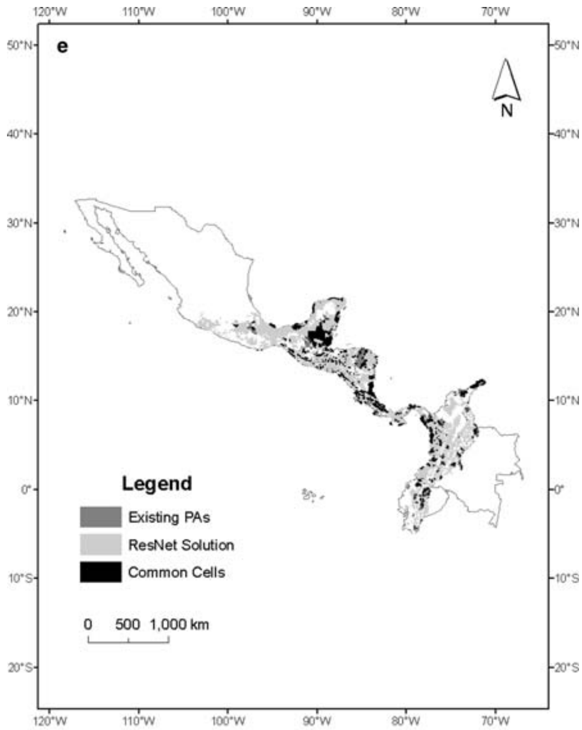

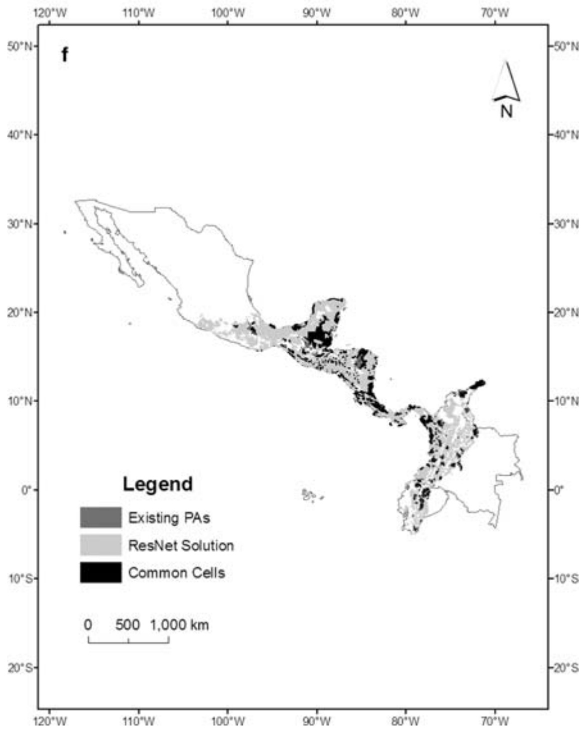

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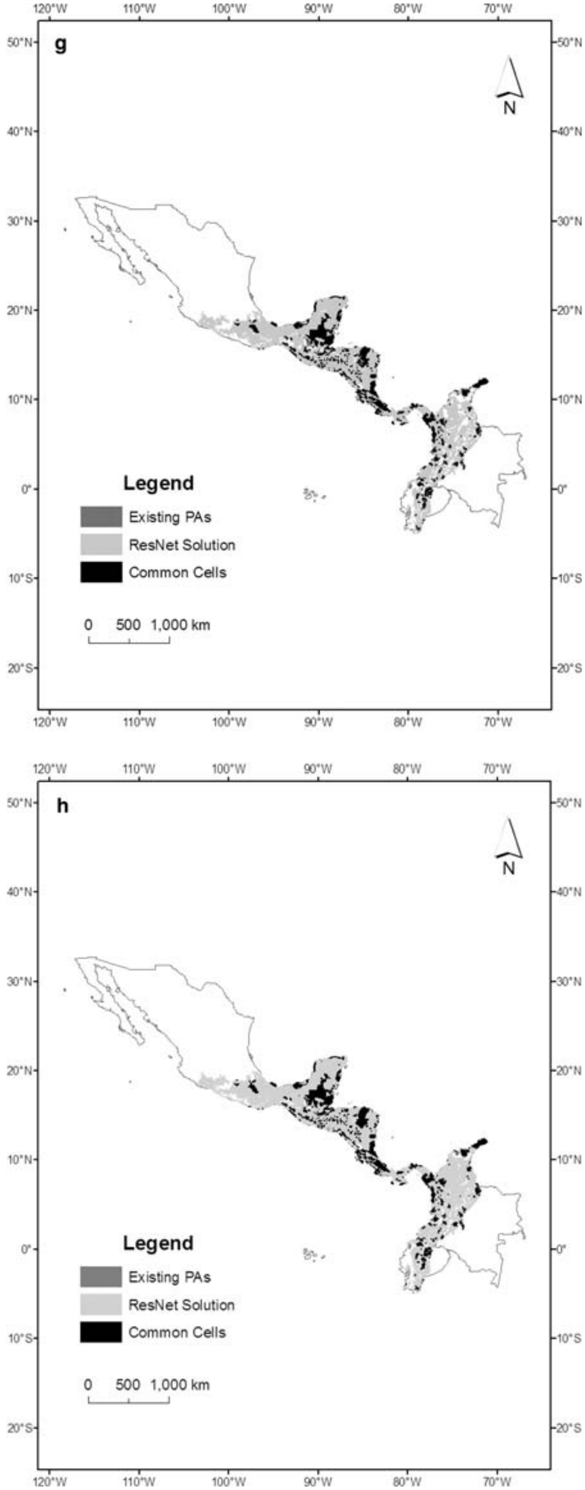

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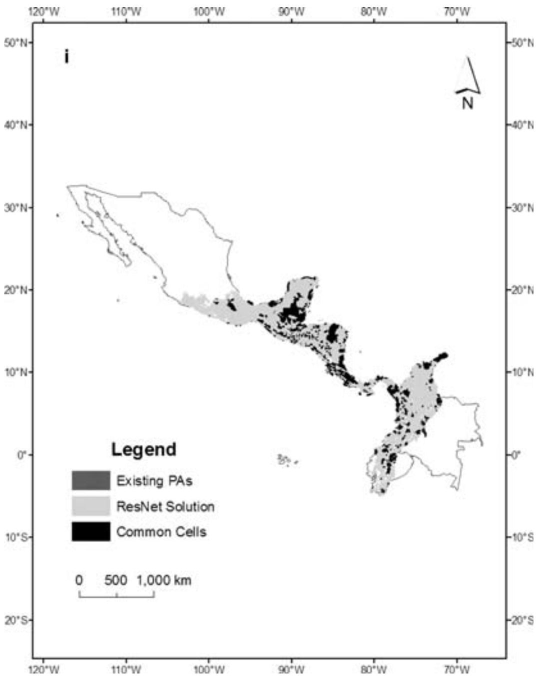

proportions of the protected areas of each country were selected ($\chi^2 \geq 1941.07$); in each case, $df = 9$ and $P < 2.2 \times 10^{-16}$. For all representation targets >10%, the surplus of selected sites in Guatemala was significantly greater than that of any other country (range of Pearson residuals: 19.59–48.36).

Discussion

The results obtained here are of relevance to other regional planning exercises for Mesoamerica, Chocó, and the Tropical Andes that are under way, for instance, as part of the MBC project even though those exercises are limited to a subset of the study region analyzed here. The larger landscape ecological context—which was adopted here on biogeographical grounds (see "Study Region")—helps place more restricted analyses in their proper regional context. Even at this larger scale, the most salient result obtained here is that, for all countries except Colombia, Mexico and Ecuador, protection of a large proportion of the distribution of only 78 Red List species takes up a very large proportion of the untransformed land: if the distribution representation target is 90%, the untransformed land needed amounts to 85–98% of each country. Including more species will only increase this area. The size of this area is a result of species at risk being widely distributed throughout Mesoamerica, Chocó, and the Tropical Andes which, in turn, underscores the region's importance for biodiversity. In the case of Colombia, Mexico and Ecuador, less area is probably being selected only because there were many fewer species' records (including those of endemic species) in the data set.

High representation targets are inevitable to ensure the persistence of species at risk (Sarakinos et al. 2001; Margules and Sarkar 2007). However, it is unreasonable to expect

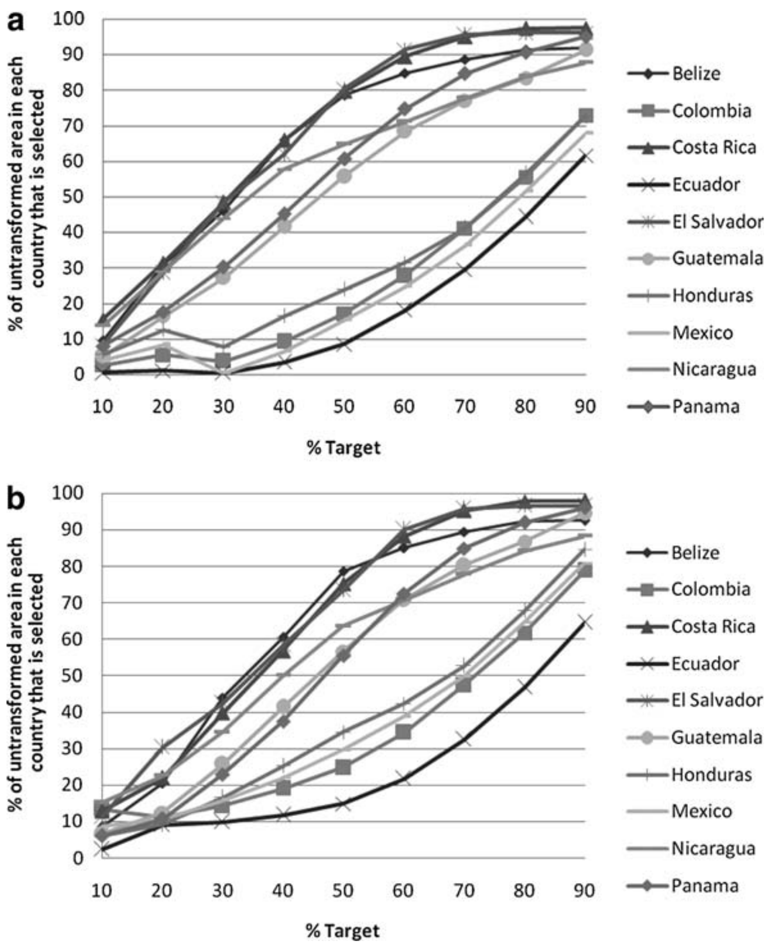

Fig. 4 Percentage of untransformed area selected in each country as a function of the representation target for Red List species: **a** only Red List species used as biodiversity surrogates; **b** both Red List species and ecoregions used as biodiversity surrogates

such large proportions of the land of any country to be set aside for conservation if that involves no human activity or use of the land. What this analysis shows is that, for biodiversity conservation to work in Mesoamerica, Chocó, and the Tropical Andes, an integrative approach to land use over the entire landscape must be developed. Exclusionary policies such as setting up National Parks or "absolute" biological reserves may have far less a role to play than land management through ecologically sound practices as, for instance, encouraged in Biosphere Reserves (Figueroa and Sánchez-Cordero 2008). This means that planning must involve human stakeholders from the beginning as envisioned in the SCP protocol.

It is encouraging that 189 of the 809 existing PAs (see Table S1) are selected when Red List species are surrogates, and the target is only 10%, which is a very modest target for such species. This seems to suggest that, unlike the ad hoc selection of existing PAs in many areas of the world (Pressey 1994; Pressey et al. 1996), those in Mesoamerica, Chocó, and the Tropical Andes were better selected to represent its biodiversity. However, what Figs. 4, 5, and 6 shows is that only a slightly higher fraction of the area within existing PAs

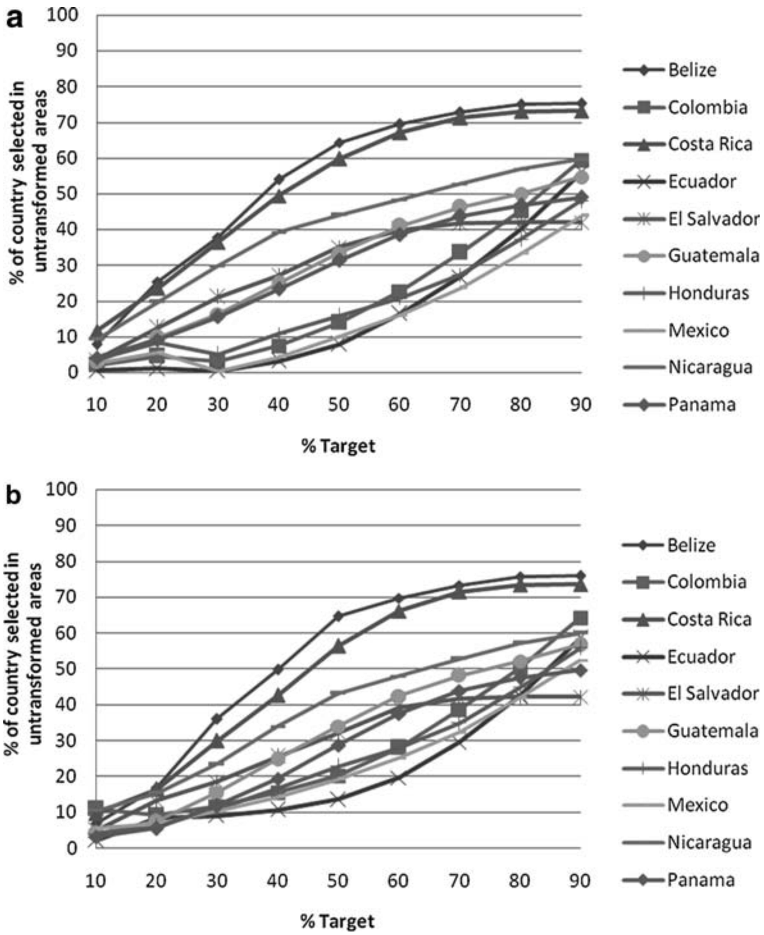

Fig. 5 Percentage of country's untransformed area selected as a function of the representation target for Red List species: **a** only Red List species used as biodiversity surrogates; **b** both Red List species and ecoregions used as biodiversity surrogates

is selected at any target compared to the fraction of the area of each country that is selected at that target. Thus, areas within the existing PAs are doing little better than areas outside with respect to representing biodiversity. In Figs. 4, 5, and 6 this conclusion is particularly obvious in the case of Colombia, Mexico, and Ecuador. The goal of SCP is to avoid this problem of inappropriate land allocation by prioritizing the most representative areas for conservation action.

Turning to individual countries, it is striking that all analyses show the existing PA network of Guatemala outperforming those of the other Mesoamerican countries. However, this is only true because those regions of Guatemala that were considered consisted of the remaining anthropogenically untransformed areas. Similarly, this analysis shows that the protected areas of Panama performing poorly compared to the other countries, once again when we restrict attention to the untransformed areas. Though there was no uniform statistical trend across all representations targets, Colombia and Ecuador also perform poorly in terms of the representativeness of their existing PAs which confirms

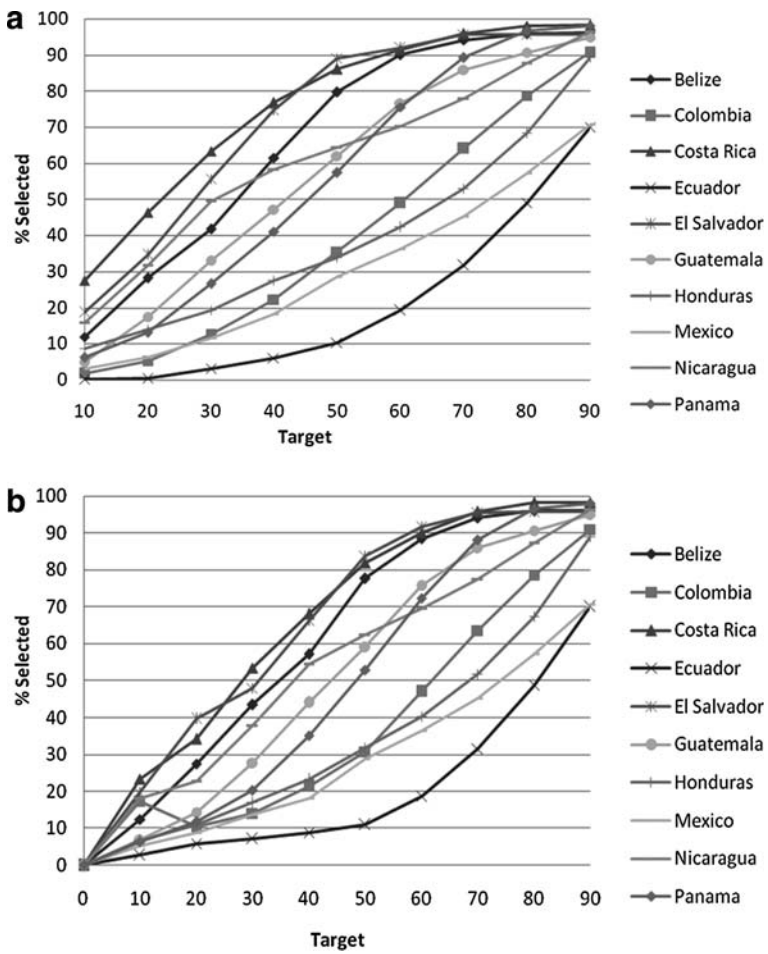

Fig. 6 Percentage of protected areas selected as a function of the representation target for Red List species. The entire area were available for selection. **a** Only Red List species used as biodiversity surrogates; **b** both Red List species and ecoregions used as biodiversity surrogates

the results of earlier analyses. Sierra et al. (2002) noted that, though more than 14% of Ecuador's terrestrial area falls within existing PAs, some habitat types, especially ecosystems on the coast and in the western Andes were under-represented. In this analysis, these ecosystems, when they fall within the study area, were selected at even low targets, using either the Red List species or both species and ecoregions as surrogates (Figs. 2a, 3a). Similarly, Fandiño-Lozano (1996) noted that though almost 10% of Colombia's habitat falls within PAs, 47.2% of the habitat types were unprotected. In both countries it is likely that excessive attention to tropical wet forests has led to their over-representation in the existing PA network at the expense of other habitats.

The results obtained here should be compared to those from other area prioritization efforts in the region both in order to identify areas that are assigned high priority by all methods (so that these areas receive special attention) and to see whether SCP techniques make any unique contribution. The methodology followed by CI in designating priority areas is very similar to the one used here; indeed in some regions of the world such as Melanesia, CI is using SCP techniques (Chris Margules 2007, personal communication). It is therefore quite likely that CI's results will be similar to those obtained here when they are made public. However, it is particularly instructive to compare these results to the portfolio published by the Nature Conservancy because the methodology used by Calderón et al. (2004), with its reliance only on environmental (non-taxonomic) surrogates and expert judgment is very different from the one used here. TNC distinguished its "portfolio" which consisted of 143 nominal conservation areas from 20 more coarse-grained "conservation action areas" (Calderón et al. 2004).

At a coarse spatial resolution TNC's portfolio and this analysis identify similar priority areas, for instance, the Lacandon-Maya forest in Mexico, the Maya Mountains of Belize, the Cordillera Central of Costa Rica, and the Bosque San Blas Darién of Panama. For Costa Rica there is very good concordance between TNC's portfolio and the results of this exercise, probably reflecting the high level of expertise readily available on Costa Rica's biodiversity (Evans 1999). At finer spatial resolutions, there are important differences: uniformly, this analysis selects a small fraction of the areas in TNC's portfolio even in the same region (especially for the 10 and 20% targets), thus providing a more fine-tuned identification of priority areas. This suggests that SCP methods can be usefully deployed to refine that portfolio. Moreover, there were some major differences: (1) in Honduras, TNC's portfolio prioritizes the Bosawar—Río Plátano area which does not emerge as important in this analysis for targets <70%; (2) in Nicaragua, TNC prioritized the Mahogany area which this analysis does not; and (3) this analysis identified some of the tropical deciduous forests of Panama as priority areas even at 10 and 20% targets, whereas TNC's portfolio ignores them. These forests should probably be part of any conservation portfolio, but the differences between TNC's portfolio and these SCP results merit further detailed analysis.

Turning to TNC's conservation action areas, these are concentrated to the north–east of the region (at the Mexico-Guatemala border and in Guatemala and Honduras) and to the south (Costa Rica and Panama). Each such area is large and an SCP analysis could be used to specify conservation areas within them more exactly. However, some important habitat types are not included in any TNC conservation action area. For instance, tropical dry forests of Costa Rica and Nicaragua (the Central American Dry Forest ecoregion) are not in an action area, in spite of being one of the most threatened ecotypes in the region (see that percentages of protection and transformation in Table 1). Similarly the Sierra Madre de Oaxaca pine-oak forests are not targeted even though they are not adequately protected (Table 1). These habitats were selected in this analysis at targets as low as 10 and 20% with either surrogate set. TNC's conservation action areas should be treated with caution.

Colombia and Ecuador were not part of TNC's analysis. For Colombia, at the national level, Fandiño-Lozano and van Wyngaarden (2005b) have recently carried out an SCP exercise using the Focalize (Fandiño-Lozano and van Wyngaarden 2005a) and C-Plan (Pressey 1999; Ferrier et al. 2000) software packages. As surrogates they used 337 topological and 62 chorological types that were established using remote-sensed data (Landsat images). Representation targets were set individually for each topological type based on the estimated area needed for minimal viable populations of four mammal species, *Panthera onca*, *Puma concolor*, *Tapirus terrestris* and *Tapirus pinchaque*. When the target here was 10 or 20%, areas selected here were similar to those prioritized by Fandiño-Lozano and van Wyngaarden (2005b) in much of the area within the study region. However, they prioritize areas between the Sanquianga and the Farallones de Cali PAs as well as an area between the Muchique and Galeras PAs in the southwest part of the country. These do not emerge as priority areas in this analysis. Moreover, this analysis does prioritize areas in the extreme west of the Cordillera de los Picachos PA as well as an area between the Laguna de Cocha, the Purace, and the Alto Fragua Indi Wasi PAs, an area south of Los Colorados and areas in the extreme north of the Western Ecuador Moist Forests which Fandiño-Lozano and van Wyngaarden (2005b) analysis excludes.

Finally, six limitations of this analysis should also be noted: (1) the most important limitation is that it was based on modeled distributions of only 78 species which is a mere 2% of the Red List species of the Mesoamerica, Chocó, Tropical Andes region. Moreover, 62 or 80% of the species used were plants. Efforts are now under way to collate and systematize data from many regional museums, universities, and other repositories to create a comprehensive public database for the region. It is hoped that the publication of this preliminary analysis will encourage local and regional scientists to share data by showing how useful these data can be in generating an adequate conservation plan for the region. This analysis will be repeated every year with additional data so that results become increasingly relevant towards the design of an implementation-oriented plan; (2) this analysis only used species deemed to be at risk by IUCN. These should be supplemented by subregional—for instance, national—priority lists of species at risk. However, other species are also of strong conservation concern, for instance, species that are endemic to the region even if they are not at present at risk. Future analyses will try to include as many of these as possible; (3) the land cover data set used was coarse and may not indicate all anthropogenically transformed areas that should not be regarded as candidate areas for conservation. Efforts are also under way to use remote-sensed data to generate a finer-resolution and more accurate land cover map so that all biologically unviable land can be excluded when developing a conservation plan; (4) the classification of the study area into only 53 ecoregions was also coarse. Future work will have to use a finer classification which will have to be created for the region. While such classifications exist for many of the countries individually, for instance, Mexico (CONABIO 1998), Costa Rica (Holdridge 1967), and Colombia (Fandiño-Lozano and van Wyngaarden 2005b), a regional classification is still lacking; (5) there was no attempt at all to incorporate spatial criteria such as size, shape, connectivity, dispersion, replication, and alignment into the priority area network designed here. These are obviously important for the persistence of biota (Sarkar et al. 2006; Margules and Sarkar 2007) and should be included in future analysis; and (6) the analysis performed here is still quite far from one that can produce an implementation-oriented plan. For that purpose stakeholders must be brought in as decision-makers and the analysis must take into full account sociopolitical opportunities and constraints.

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CAPITULO III

DISTRIBUCIÓN Y CONSERVACIÓN DE ESPECIES AMENAZADAS EN MESOAMÉRICA, CHOCÓ Y LOS ANDES TROPICALES

Distribución y Conservación de Especies Amenazadas en Mesoamérica, Chocó y Andes Tropicales.

Distribution and conservation of endangered species in Mesoamerica, Chocó and Tropical Andes.

Enviado a la Revista Mexicana de Biodiversidad.

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Resumen. Este estudio presenta modelos de distribución potencial para 313 especies en Mesoamérica, Chocó y los Andes Tropicales, incluyendo 285 plantas y 28 vertebrados terrestres en las categorías de amenaza: CR, EN y VU, de la UICN. Las distribuciones potenciales fueron refinadas considerando el hábitat natural remanente y el hábitat transformado (agricultura y asentamientos humanos). Los resultados indican que las especies incluidas en las categorías de amenaza de la lista roja de la UICN no muestran el mismo grado de amenaza por pérdida de hábitat, o porcentaje de representatividad en áreas protegidas para los diferentes países donde se distribuyen. Finalmente, se identifica la cordillera de Talamanca en Panamá y Costa Rica, y los páramos y cordilleras andinas, y selva húmeda del oeste de Ecuador, como sitios con alta riqueza de especies amenazadas y presencia de condiciones ambientales idóneas para la presencia potencial de especies amenazadas. Dichos sitios pueden representar oportunidades de investigación valiosas, donde se deben enfocar los apoyos de financiamiento para incrementar las opciones viables de conservación y conocimiento de estas especies amenazadas.

Palabras Clave: Modelos de nicho ecológico, Áreas Protegidas, Vegetación natural remanente, Área transformada, Investigación, Conservación

Abstract. This study produced ecological niche modeling projected as potential distributions of species occurring in Mesoamerica, Chocó and Tropical Andes, including 285 vascular plants, and 28 terrestrial vertebrates listed in the IUCN Red List. The distribution models were further refined by quantifying remnant natural habitat and transformed areas (agriculture and human settlements) within the potential disitrbutions. Our results show that included IUCN endangered species do not show similar level of threat within countries of this region, highlighting the importance of local habitat transformation and representativeness in protected areas to define species threats in a geographic context. We identified the Talamanca region in Panama and Costa Rica, the Andean Cordillera, Andean Páramos, and western moist forest in Ecuador as places holding high endangered species richness suitable for establishing programs supporting research and conservation of these species.

Key words: Ecological Niche Models, Natural Protected Areas, Natural Vegetation Cover, Transformed habitat, Research, Conservation.

Introducción

La transformación del hábitat es una de las principales causas de pérdida de biodiversidad y de la crisis de extinciones de especies a nivel mundial (Dirzo y Raven, 2003). En particular, las regiones de Mesoamérica, Andes Tropicales y Chocó son reconocidas como un centro de alta diversidad biológica (*hot spot*) por su alta riqueza de especies y endemismos. No obstante, altas tasas de deforestación ponen en riesgo la persistencia de esta excepcional biodiversidad (Myers et al., 2000).

Desde 1950, la Unión Internacional para la Conservación de la Naturaleza (UICN) ha compilado listas de especies en riesgo de extinción (listas rojas). Los objetivos de las listas rojas son proveer un índice global del grado de degradación de la biodiversidad, e identificar y documentar aquellas especies con mayor necesidad de conservación, con el fin de reducir tasas de extinción de especies (Mace et al., 2008). Las listas rojas pretenden también elevar el grado de conciencia conservacionista, así como coadyuvar acciones de conservación en beneficio de especies en peligro de extinción (Fitter y Fitter, 1987). Las tres categorías de amenaza, críticamente amenazada (CR), amenazada (EN) y vulnerable (VU), están definidas cualitativamente por la probabilidad de extinción en un periodo de tiempo. Las especies clasificadas dentro de alguna de estas categorías, deben cumplir al menos uno de cinco criterios: (a) alta tasa de reducción de la población, (b) área de distribución reducida con riesgo de transformación del hábitat, (c) tamaño poblacional reducido y en declive, (d) tamaño poblacional muy pequeño y/o, (e) análisis cuantitativo de probabilidades de extinción desfavorables para una especie (Mace et al., 2008).

Sin embargo, existe una diferencia entre medidas de riesgo y prioridades de conservación (Mace et al., 2008). No obstante que la UICN clasifica a las especies en términos de sus riesgos de extinción, esto no equivale a determinar las prioridades de conservación, que incluyen otros factores como costos, beneficios, logística, probabilidades de éxito y otras características biológicas de las especies (Possingham et al., 2002; Mace y Baillie, 2007). La lista roja es una herramienta para identificar casos urgentes de conservación de especies, donde se debe evaluar su situación y diseñar e implementar acciones de conservación efectivas, mas no es una herramienta para identificar prioridades de conservación *per se* (Mace et al., 2008).

Una crítica recurrente dentro de la comunidad conservacionista es el invertir altos recursos financieros en la conservación de especies en la categoría CR, en virtud de la supuesta baja efectividad de esta estrategia; algunas de estas especies necesitan una gran inversión de recursos y esfuerzo y los resultados muestran bajas tasas de éxito. En contraste, otras especies ubicadas en otras categorías de riesgo, pueden ser conservadas a un menor costo (Mace y Baillie, 2007). La manera óptima de invertir los recursos demuestra que no se trata de que las especies más amenazadas reciban mayores fondos, sino que los recursos sean repartidos de manera que las acciones de conservación aseguren la viabilidad de la mayor cantidad de especies amenazadas (Possingham et al., 2002; Mace y Baillie, 2007).

En este sentido, el manejo y conservación de la biodiversidad requiere tener un conocimiento razonable de la distribución geográfica de las especies (Margules y Pressey, 2000). Una limitante a esta demanda de información, es que no existe conocimiento en detalle sobre la distribución geográfica de la mayoría de las especies (Graham et al., 2004; Soberón y Peterson, 2004). Recientemente, se han desarrollado herramientas metodológicas para modelar el nicho ecológico de las especies (MNE), proyectado como su distribución potencial (Peterson et al., 1999; Raxworthy et al., 2003). Los MNE se basan, explícita o implícitamente, en el concepto de nicho ecológico de Hutchinson (Hutchinson, 1957). Esta hipótesis asume que las especies se encuentran en un equilibrio con su ambiente (generalmente delimitado por características climáticas), y que los rangos de distribución se pueden predecir basados en las características ambientales de los sitios donde se han observado y/o colectado; es decir, las localidades de colecta (Guisan y Thuiller, 2005). Los diferentes métodos para generar MNE usan distintas reglas o algoritmos matemáticos de cómputo para definir el nicho ecológico de la especie en un espacio ambiental multidimensional (espacio ecológico), basado en los registros de colecta y variables ambientales. Una vez que el nicho ecológico ha sido definido en un espacio ecológico, se proyecta a un espacio geográfico, produciendo un mapa de distribución predictivo (Tsoar et al., 2007).

Este marco conceptual ha adquirido gran relevancia en estudios de biogeografía (Fleishman et al., 2003; Raxworthy et al., 2003; Bourg et al., 2005; Tsoar et al., 2007; Peterson et al., 2009). Los modelos de nicho ecológico se han aplicado para múltiples propósitos, incluyendo (1) detección de nuevas especies y expansión de áreas de distribución (Raxworthy et al., 2003; Bourg et al., 2005), (2) predicción de cambios en los rangos de distribución bajo escenarios de cambio climático, predicción geográfica de una especie invasora (Higgins et al., 1999; Rouget et al., 2004), (3) identificación de áreas potenciales para una reintroducción exitosa de especies en peligro de extinción (Engler et al., 2004; Bourg et al., 2005), (4) entendimiento de cómo la pérdida de hábitat impacta negativamente en la distribución de especies (Sánchez-Cordero et al., 2005) y, (5) identificación de áreas prioritarias de conservación (Araújo y Williams, 2000; Ferrier et al., 2002; Sarkar et al., 2009). El poder predictivo de estos modelos se ha evaluado para diferentes técnicas y conjuntos de datos, mostrando de manera general resultados positivos (Elith et al., 2006), que hacen de ellos una gran herramienta para aplicaciones de conservación (Margules y Sarkar, 2007).

En virtud de que la pérdida de hábitat natural impacta negativamente en la distribución de especies (Sánchez-Cordero et al., 2005), es necesario evaluar la representatividad de las especies dentro de las áreas protegidas (AP). Las AP constituyen una de las principales estrategias de conservación, por lo que es deseable que las AP representen adecuadamente la diversidad de una región y, de esta manera, aseguren los procesos que garanticen su persistencia (Margules y Pressey, 2000). Aunque las AP cumplen su papel en reducir las tasas de pérdida de hábitat (Aaron et al., 2001; Figueroa y Sánchez-Cordero, 2008), generalmente no son representativas de la biodiversidad (Rodrigues et al., 2004b). En particular, estudios en Mesoamérica, Andes Tropicales y Chocó han demostrado que las AP no representan adecuadamente la diversidad biológica regional (Andelman y Willig, 2003; Armenteras et al., 2003; Cue-Bar et al., 2006; Fuller et al., 2006; Sarkar et al., 2009).

Este trabajo (1) cuantifica el hábitat natural remanente dentro de la distribución potencial de una lista selecta de especies amenazadas, (2) identifica las áreas donde se encuentra una mayor riqueza de especies amenazadas y, dónde se deberían invertir recursos financieros para fomentar la investigación y establecer planes de conservación para estas especies y, (3) evalúa la representación de las especies amenazadas en las AP de la región de Mesoamérica, Chocó y los Andes Tropicales.

Métodos

Área de estudio. El área de estudio comprende la región de Mesoamérica, Chocó y los Andes tropicales, usando las ecoregiones terrestres como criterio para delimitación (Olson et al., 2001), consta de 53 ecoregiones en México, Belice, Guatemala, Honduras, El Salvador, Nicaragua, Costa Rica, Panamá , Colombia y Ecuador (Figura 1). El límite norte está delineado por las ecoregiones asociadas con la depresión del Balsas en México, dada su alta afinidad biogeográfica con la biota Mesoamericana (Morrone, 2005). El límite sur está definido por la transición geográfica en que la cordillera de Los Andes se separa en tres ramas: cordillera occidental, cordillera central y cordillera oriental; las ecoregiones que limitan son aquellas que se interceptan con esta transición. La región de estudio se dividió en celdas de 0.02° X 0.02°, resultando en 343383 celdas, con un área promedio de 4.818 km^2.

Obtención de datos. Los registros de especies enlistadas en la UICN en las categorías de amenaza (CR, EN y VU) fueron obtenidos de colecciones científicas disponibles en las siguientes direcciones de internet: MaNIS (http://manisnet.org, último acceso mayo 2008), HerpNET (http://www.herpnet.org/, último acceso mayo 2008), ORNIS (http://olla.berkeley.edu/ornisnet/, último acceso mayo 2008), REMIB (Red Mundial de Información sobre Biodiversidad; http://www.conabio.gob.mx/remib/doctos/remib_esp.html, último acceso abril 2007), Smithsonian National Museum of Natural History (http://www.mnh.si.edu/rc/, último acceso Abril 2007) y University of Missouri Botanical Garden, W^3TROPICOS (http://mobot.mobot.org/W3T/Search/vast.html, último acceso mayo 2008). Los datos climáticos se obtuvieron de WorldClim (Hijmans et al., 2005), y consistieron en 19 variables bioclimáticas a una resolución de 1km^2. Las variables bioclimáticas son derivadas de valores mensuales de temperatura y precipitación, representando promedios anuales, estacionalidad y factores extremos, generando variables climáticas correlacionadas estrechamente con la biología de las especies (http://www.worldclim.org/bioclim.htm, último acceso mayo 2007). Los datos de altitud fueron tomados de U.S. Geological Survey's Hydro-1K DEM (USGS, 1998) y, a partir de estos se calculó la pendiente y el aspecto, usando la extensión de análisis espacial en ArcMap 9.2.(ESRI, 2006). Todas las variables climáticas y topográficas fueron re-muestreadas a 0.02° x 0.02°.

Modelos de nicho ecológico. Se generaron MNE proyectados como distribuciones potenciales de 581 especies en las categorías de riesgo de la UICN, de las cuales 509 fueron plantas y 72 vertebrados terrestres, usando el programa MaxEnt versión 3.2.1 (Phillips et al., 2006). Maxent ha mostrado ser un algoritmo adecuado, resultando en modelos altamente predictivos, incluso en casos donde se cuenta únicamente con pocos registros de localidades de colecta (< de 10) (Pearson et al., 2007). Los modelos se realizaron en formato logarítmico y usando un 25% de los datos como set de prueba.

Del total de 581 especies, sólo se incluyeron 313 especies en los análisis (285 plantas y 28 vertebrados terrestres), en virtud de que mostraron los MNE más robustos, con valores de AUC > a 0.75 (*Area under the curve*; Phillips et al. 2006) y P < a 0.05 (ver cuadro suplementario CS1); estos valores indican distribuciones con alto poder predictivo (Pawar et al., 2007). Las hipótesis de distribución de las 313 especies se corroboraron con las distribuciones reportadas por la UICN. Los MNE incluidos en los análisis se reclasificaron en presencias y ausencias, usando como valor umbral el valor mínimo del set de entrenamiento, siendo apropiado para modelos que se han generado usando pocos registros de colecta y, teniendo un significado ecológico claro, al identificar sitios que son ambientalmente tan apropiados, como los que se han encontrado en dichos registros de colecta (Pearson et al., 2007).

Análisis geográfico de las distribuciones potenciales. Las distribuciones potenciales de las 313 especies se sobrepusieron usando el programa ArcMap 9.2 (ESRI, 2006), y se clasificaron las áreas de coincidencia de distribución de especies en las siguientes categorías arbitrarias: de 1 a 5, de 6 a 10, de 11 a 20, de 21 a 50 y, de 51 a 116, resultando en 5 categorías de áreas de densidad de riqueza de especies; este ejercicio también se realizó considerando grupos taxonómicos que contaron con más de 10 especies (Cuadro 1 y 2). Usando el programa ArcView, con la extensión Projector! y la proyección "Equal Area Cylindrical", se calculó el área de distribución potencial, el área de hábitat transformado dentro de la distribución potencial y, el área dentro de las AP. Se consideró como criterio de hábitat transformado las áreas de cultivo, agricultura intensiva y urbanas del Global Land Cover 2000 (Eva et al., 2003; Latifovic et al., 2003) (Ver cuadro suplementario CS2). Las AP fueron tomadas de World Data Base on Protected Areas del 2007 (WDPA, 2007) siendo incluidas todas las categorías.

Resultados

La UICN reporta un total de 1074 especies de vertebrados terrestres y 2559 especies de plantas en las listas rojas incluidas las tres categorías de amenaza, para los países del área de estudio ([http://www.iucnredlist.org/search/search-basic último acceso Mayo 2007](http://www.iucnredlist.org/search/search-basic)). Sin embargo, solo fue posible obtener MNE robustos para 313 especies, de las cuales 285 fueron plantas y 28 vertebrados terrestres (Cuadro 1).

Al superponer las áreas de distribución de las 313 especies se observó que se distribuyen en prácticamente la totalidad del área de estudio (Figura 2); solo el 0.5% de la región de estudio, correspondiente a 8218 km^2, no mostró presencia de la distribución potencial de alguna especie. Ecuador mostró cerca del 30% del área, con una alta superposición de especies (> 51 especies). Colombia, Costa Rica, Guatemala, Nicaragua y Panamá también presentaron áreas con superposición de más de 51 especies, pero en porcentajes menores al 0.5% de cada país (Figura 3).

Los países que presentaron más del 50% de su área, con superposición de especies en la categoría 21 – 50 especies, fueron Costa Rica, Ecuador, Nicaragua y Panamá. El Salvador y Honduras presentaron más del 50% de su área en la categoría de 11 a 20 especies y, México más del 50% de su área en la categoría de 1 a 5 especies (Figura 3).

En El Salvador y Panamá, más del 50% de las áreas que contienen una riqueza, entre 11 y 20 especies, se encontraron en áreas transformadas; en Guatemala, se observó el 50% de las áreas con una riqueza entre 21 a 50 especies también en áreas transformadas (Cuadro 3). Belice y Ecuador son los países que mostraron menos porcentaje de las distribuciones en áreas transformadas, siendo estas menores al 25% (Cuadro 3), seguidos por Colombia, Honduras y México, que mostraron porcentajes inferiores al 40% de las distribuciones de especies en áreas transformadas (Cuadro 3). En cuanto a representación de las especies en AP, Belice mostró los mayores valores de superposición de las distribuciones potenciales de especies amenazadas con las AP; áreas que mostraron una riqueza mayor de 11 especies, tuvieron valores de representación en AP superiores al 60%; Panamá, Nicaragua y Costa Rica mostraron las áreas con mayor riqueza de especies, con más del 50% de su distribución en sus respectivas AP. Colombia, Ecuador y El Salvador presentaron los valores más bajos de representación de especies en sus respectivas AP, ninguna de las categorías de riqueza de especies presentó valores de representación en AP superiores al 20% (Cuadro 4).

La representación taxonómica de los diferentes grupos, en los países, se muestra para aquellos grupos taxonómicos que contaron con más de 10 especies: las Clases Amphibia, Polypodiopsida y Liliopsida y, los Ordenes Asterales, Campanulales, Fabales, Laurales, Myrtales, Rosales, Rubiales y Scrophulariales, de la Clase Magnoliopsida. En general, el área de distribución potencial por grupo taxonómico, no se correlacionó positivamente con el número de especies, pero sí con el porcentaje de área transformada ($r = 0.86$ $P < 0.05$). Los grupos taxonómicos que mostraron una mayor áreas en su distribución potencial fueron los órdenes Fabales, Laurales, Myrtales y Rubiales; los de menor áreas de distribución fueron los órdenes Campanulales, Asterales y la Clase Liliopsida. Los porcentajes de distribución potencial en hábitat transformado variaron de 11.39 - 29.71% y, los porcentajes de distribución potencial representados en AP variaron de 11.92 - 18.72% (Cuadro 5). No se observó una correlación significativa entre el área de distribución potencial y el porcentaje de área transformada en cada país, para cada grupo taxonómico.

Guatemala, El Salvador, Honduras, Panamá, Nicaragua y México presentaron los mayores porcentajes de distribuciones potenciales en áreas transformadas, para las clases Amphibia, Liliopsida, Polipodiopsida y, los órdenes Asterales, Fabales, Laurales, Myrtales, Scrophulariales y Rubiales (Figuras 4 a, b, c, d, f, g, h, j, k). Los mayores porcentajes de representación de especies en AP, se ubicaron en Belice y Costa Rica, para las clases Amphibia, Liliopsida, Polipodiopsida, y los órdenes Asterales, Fabales, Laurales, Rosales (Figura 4a, b, c, d, f, g, i). En El Salvador, para la Clase Liliopsoda y, en Guatemala, para el Orden Rosales, se presentaron también valores altos en la representación de especies en las AP (Figuras 4b, i). En todos los países, los ordenes Campanulales y Rosales, mostraron porcentajes inferiores al 35% en áreas transformadas (Figuras 4e, i) y, los órdenes Campanulales, Myrtales, Scrophulariales y Rubiales, porcentajes inferiores al 31% en las AP (Figura 4b, h, j, k). Para el Orden Rubiales, esta última tendencia se excluye en Belice, para el cual, se observó un 99% de representación en las AP, presentando una distribución potencial de sólo 24km^2 (Figura 4k).

Para cada grupo taxonómico se identificó el sitio con la mayor riqueza de especies a nivel regional y que coincidiera con áreas naturales remanentes, considerándolos sitios donde se pueden enfocar esfuerzos de investigación que generen mas conocimiento para la conservación de un mayor número de especies amenazadas simultáneamente. De acuerdo a lo anterior para los anfibios la Cordillera de Talamanca entre Panamá y Costa Rica, para las Asterales, Campanulales, Myrtales, Polypodiophytas, Rosales y Scrophulariales los páramos andinos en el Ecuador, para las Fabales las selvas húmedas del noroeste de Ecuador, para las Laurales, Liliopsidas el oeste de la cordillera de los andes en el centro de Ecuador y para las Rubiales el Este de la Cordillera Real en el Ecuador, representan estos sitios de importancia.

Discusión

Es importante resaltar que se lograron generar MNE proyectados como distribuciones potenciales, para el 2.6 % de los vertebrados terrestres y 11 % de las plantas incluidas en las listas rojas de la UICN para esta región (Cuadros 1 y 2). Este trabajo es el producto de una búsqueda exhaustiva de registros en las colecciones internacionales de libre acceso en internet y demuestra el limitado número de especies que pueden ser utilizadas, a través de estas bases de datos, para los países del área de estudio. Es posible que existan registros adicionales para especies amenazadas en colecciones locales, pero la falta de acceso a estos registros dificulta su análisis. Consecuentemente, una prioridad es el establecer mecanismos de acceso abierto a colecciones locales y regionales para incrementar el número de registros de muchas especies amenazadas. Es indispensable apoyar con recursos económicos estas iniciativas (Graham et al., 2004).

Adicionalmente, una de las principales limitantes para aplicar MNE a especies amenazadas es el número reducido de registros de localidades de colecta que presentan dichas especies a nivel global (Graham et al., 2004). Es obvio que esta limitante debe ser una prioridad en delinear prioridades de investigación en los inventarios biológicos en esta región; se debe enfatizar el registro de especies de flora y fauna incluidas en las listas rojas de la UICN para obtener información más confiable de su distribución y, por ende, de su conservación (Sánchez-Cordero et al., 2001; Graham et al., 2004).

No obstante, este trabajo identificó la ubicación de sitios con características ambientales idóneas para la presencia de una alta concentración de especies de flora y fauna incluidas en las listas rojas de la UICN, en una región biogeográfica que es de gran importancia en conservación (Myers et al. 2000).

Las evaluaciones de riesgo de amenaza de especies en listas rojas deben efectuarse a diferentes escalas- global, regional y local- en virtud de que a estas escalas se puede presentar diferente grado de amenaza, ya sea extirpación poblacional o incluso la extinción (Mace et al., 2008). Por ejemplo, una especie incluida en listas rojas puede estar en buen estado de conservación, a nivel de un país, región o sitio, pero puede estar bajo una amenaza a una escala mundial; el caso opuesto sería que una especie no puede estar en un grado de amenaza a nivel global o de una país, pero estar en riesgo de amenaza o extinción a nivel regional o local (Sánchez-Cordero et al., 2005; Mace et al., 2008). Es decir, el grado de amenaza o riesgo de extinción se da en un contexto geográfico (Sánchez-Cordero et al., 2005).

Los resultados del estudio indican que las especies incluidas en las listas rojas de la UICN, no muestran un grado de amenaza equivalente en los diferentes países en donde se distribuyen, es decir, los grupos taxonómicos no se comportan de la misma manera en los diferentes países. El enfoque de este estudio permitió determinar, por un lado, el contexto geográfico de la amenaza y, por otro, las áreas que presentan potencialidad para una alta riqueza de especies en diferentes categorías de riesgo, dadas sus características ambientales (Cuadro 3; Figura 2). Más aún, para las especies con amenaza global de extinción, se identifican áreas donde además de contar con las características ambientales idóneas para la presencia de estas especies, (1) existen mejores condiciones de vegetación natural y (2) existe presencia de áreas protegidas y, por ende, son áreas donde se deben enfocar los apoyos de financiamiento, ampliando el área o reforzando el manejo de las AP para incrementar las opciones viables de conservación e investigación (Figura 4).

Por ejemplo, este estudio demuestra que una proporción importante de la distribución potencial de especies en categorías de amenaza, dentro de los grupos taxonómicos analizados, coincide con áreas transformadas (Cuadro 3; Figura 4). Esto indica que existe una proporción menor de hábitat natural remanente en dichas distribuciones potenciales (Cuadro 5). Recientemente, se ha propuesto que las especies conservan su nicho ecológico en tiempos evolutivos, lo que supone una lenta adaptación a hábitat nuevos (Peterson et al., 1999). Si esta tesis es correcta, esto sugiere que las especies no pueden mantener poblaciones viables reproductivas en áreas transformadas ("hábitat nuevo" para la especie), sin inmigración de individuos de hábitat naturales remanentes (Peterson y Holt, 2003). Por tanto, los grupos taxonómicos que mostraron esta tendencia, requieren de una alta prioridad de conservación en aquellas áreas de su distribución, que muestran aún hábitat natural remanente (Sánchez-Cordero et al., 2005).

Se observó una baja representación de los grupos taxonómicos en las AP de la región; los valores de coincidencia geográfica, entre la distribución potencial de especies amenazadas en las listas rojas de la UICN y las AP, fluctuó entre el 10 – 20%. Esta tendencia puede reflejar que en los criterios de selección y decreto de varias AP de la región, no se consideró, de manera prioritaria, la presencia de especies amenazadas incluidas en las listas rojas de la UICN. Estos resultados coinciden con algunos ejercicios en donde se demuestra la baja representatividad del contenido de biodiversidad en las AP en países de la región (Cuadros 4-5; Figura 4); esto sugiere que muchas son AP *ad hoc*, es decir, cuyos criterios de selección fueron políticos, escénicos, etc., (Fuller et al., 2007; Sarkar et al., 2009). Una alternativa a esta deficiencia es el generar redes o sistemas de áreas de conservación, que incluyan AP y áreas enfocadas a manejo de recursos y restauración ecológica (Ervin, 2003; Rodrigues et al., 2004a; Fuller et al., 2006; Margules y Sarkar, 2007; Chan y Daily, 2008; Sarkar et al., 2009). Esto es particularmente relevante, en virtud de que se ha observado una tasa de cambio de uso de suelo y vegetación más alta en sitios fuera de las AP (Sánchez-Azofeifa et al., 2003). Incluso, existen ejemplos de una deficiente protección de especies dentro de las AP en la región. Tal es el caso del primate amenazado, *Sanguinus oedipus*, en Colombia; se establecieron AP que, entre los años 1990 y 2000, tuvieron una pérdida del hábitat natural de poco más del 70%, afectando la conservación de esta especie (Miller et al., 2004). Este escenario confirma que es necesario establecer alternativas adicionales de áreas de conservación que, integradas a las AP, formen redes de áreas de conservación interconectadas entre sí (Margules y Sarkar, 2007).

La representatividad de las especies amenazadas en las AP contrastó en los países de la región. En Belice y Costa Rica, se observaron los valores más altos de representatividad, en tanto, México, Honduras, El Salvador y Colombia, mostraron los valores más bajos (Cuadro 4). Esto indica que últimos son países que necesitan establecer programas más ambiciosos de conservación que se enfoquen a tener una representación más adecuada de especies amenazadas en las áreas avocadas a la conservación. Estos resultados coinciden con análisis previos de la marginal representatividad de la biodiversidad en las AP de estos países (Calderón et al., 2004; Sarkar et al., 2009). Consecuentemente, la planeación sistemática de la conservación debe entonces hacerse desde múltiples escalas, desde un nivel local o paisajístico, hasta global, e integrar diferentes estrategias de conservación que conformen redes y sistemas de áreas de conservación interconectados entre sí (Chan y Daily, 2008).

No obstante el bajo porcentaje de especies amenazadas incluidas en el análisis, existe un sustento de que las tendencias de este estudio coinciden con estudios previos que incluyen la biodiversidad de ciertos grupos faunísticos y florísticos. Por ejemplo, Colombia, Ecuador, Panamá y México están dentro de los 20 países, a nivel mundial, con más especies de plantas amenazadas (Hilton-Taylor y Mittermeier, 2000). Luna Vega et al (2007) encuentran que, para 25 especies de plantas vasculares amenazadas de México, su representación en AP es muy baja, dado que la mayoría de la distribución se encuentra en los bosques de niebla, ecosistema muy amenazado y poco protegido en México. De manera similar, pese a que México es el país con mayor diversidad de pinos, donde el 55% son endémicos y, por lo menos 20 especies están en alguna categoría de riesgo, su representación en AP no es adecuada y no se cuenta con programas de conservación y uso sostenible eficientes para estas especies (Sanchez-Gonzalez, 2008). Bernal y Galeno (2006) predicen que cualquier especie nueva de palmas (Arecaceae) que se encuentre en los andes de Colombia estará amenazada. Por ende, estos países requieren particular atención a las especies amenazadas, dando prioridad de investigación a programas de conservación y proyectos enfocados a ubicar registros y poblaciones en regiones donde las distribuciones potenciales predicen su presencia (Figura 4). Un estudio en esta dirección es el de Granados-Tochoy et al. (2007), quienes redescubrieron en Colombia, después de 200 años, la especie amenazada *Solanum humboldtianum*, y que ahora cuenta con un programa específico de conservación.

En otros grupos biológicos, la situación es similar; en mamíferos, al menos un 25% de las especies endémicas han perdido más del 50% de su hábitat natural, especialmente en los estados de Veracruz y en la Faja Transvolcánica Mexicana (Sánchez-Cordero et al., 2005). Para las aves, el caso del quetzal (*Pharomachrus mocinno*) es emblemático; Solórzano et al (2003), basados en un estudio en México, muestran que su distribución ha disminuido en un 82% en los últimos 30 años , por pérdida de hábitat natural, y concluyen que se necesitan esfuerzos de conservación en toda la región Mesoamericana, pues la situación de pérdida de hábitat es similar en Nicaragua, El Salvador, Guatemala y Honduras. Otra especie de ave amenazada afectada por la transformación de los bosques de pino y encino en áreas de cultivo y ganadería en Honduras y Guatemala, es *Dendroica chrysoparia* (Rappole et al., 2000). Los ecosistemas de pino-encino en la región Mesoamericana contienen composición de especies amenazadas que los hacen prioritarios en la selección de áreas de conservación (Sarkar et al., 2009).

El trabajo de Solórzano et al (2003) coincide con las tendencias observadas en este estudio; Belice, Costa Rica y Panamá fueron los países Centroamericanos, que presentaron una mayor proporción de distribuciones potenciales dentro de las AP. Sin embargo, aún en estos países hay especies que se encuentran bajo una amenaza, tal como el anfibio *Craugastor punctariolus*, en Panamá, donde en 2 meses se extinguieron 3 sub-poblaciones debido a la presencia del hongo *Batrachochytrium dendrobatidis* (Ryan et al., 2008). Por otro lado, no obstante que el porcentaje de áreas transformada en las distribuciones potenciales de los grupos taxonómicos fue bajo en Colombia y Ecuador, algunas especies amenazadas presentan condiciones adversas. Por ejemplo, para el Paujil (*Crax alberti*), en el departamento de Antioquia, se había perdido el 96% de su hábitat natural hasta el 2002, ubicando a la especie en peligro de extinción (Melo-Vasquez et al., 2008).

Los fondos asignados para la conservación parecen insuficientes para atender la complicada situación de muchas especies amenazadas. Las organizaciones gubernamentales y ONG de conservación encargadas demandan estrategias simples y directas para la asignación de recursos limitados. Tomando en cuenta un enfoque regional o global como punto de referencia, se deben identificar aquellas áreas, a escala nacional y local que conlleven acciones concretas de conservación (Margules y Sarkar, 2007). Este estudio identificó áreas que, dado su porcentaje de transformación y representatividad en AP de distribuciones de especies amenazadas de la UICN, se deberían priorizar para establecer investigación y programas viables de conservación y manejo de especies particulares. Los protocolos de priorización de proyectos (PPP) e implementación de teorías de toma de decisiones, se han usado para optimizar la asignación de recursos a especies amenazadas; los PPP se han diseñado para que se consideren simultáneamente los costos, los beneficios (incluyendo valor de las especies) y la probabilidad de éxito del manejo (Margules y Sarkar, 2007). El uso de PPP puede mejorar sustancialmente los resultados de la conservación de especies amenazadas, al incrementar la eficiencia y asegurar la transparencia de las decisiones de manejo (McCarthy et al., 2008; Joseph et al., 2009).

Asimismo, las áreas de alta densidad de especies amenazadas que contienen aún hábitat natural remanente en sus distribuciones potenciales (Cuadros 3-5, Figura 4) deben ser consideradas prioritarias para establecer programas de investigación. La ampliación de distribución y el descubrimiento de especies nuevas, a partir de modelos de nicho ecológico de especies taxonómicamente relacionadas, es un reto de investigación (Raxworthy et al., 2003; Bourg et al., 2005). En Ecuador, se generaron modelos de nicho ecológico proyectado como distribución potencial del colibrí amenazado, *Metallura baroni*, y se encontraron registros de poblaciones de la especie en sitios predichos por dicha distribución (Tinoco et al., 2009). Asimismo, la distribución potencial de especies amenazadas pueden servir como sitios adecuados para su reintroducción de poblaciones y manejo (Engler et al., 2004; Bourg et al., 2005). Finalmente, las iniciativas de conservación para especies amenazadas globalmente presentes en las regiones de Mesoamérica, Chocó y Andes Tropicales, deben enfocarse en especies que muestran altos porcentajes de su distribución potencial en áreas transformadas para algunos países, pero que, a su vez, cuenten con suficiente área en sitios conservados y en AP en otros países, que ofrezcan mejores oportunidades para el éxito de los planes de conservación. De esta manera se pueden alcanzar esfuerzos regionales en la conservación de especies globalmente amenazadas.

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Cuadro 1. Número de especies incluidas en este estudio (Análisis) y número de especies reportadas en la lista roja de la UICN (Lista Roja), por grupo taxonómico, en las diferentes categorías de riesgo. Los mapas por grupo taxonómico sólo se realizaron para aquellos grupos que presentaron más de 10 especies. CR = críticamente amenazada; EN = amenazada; VU = vulnerable.

	CR		**EN**		**VU**		**Total**	
	Análisis	**Lista Roja**	**Análisis**	**Lista Roja**	**Análisis**	**Lista Roja**	**Análisis**	**Lista Roja**
Animalia	**4**	**280**	**18**	**409**	**6**	**385**	**28**	**1074**
Amphibia	4	222	18	286	2	184	24	692
Aves		26		62	1	112	1	200
Mammalia		19		53	3	67	3	139
Reptiles		13		8		22		43
Plantae	**7**	**372**	**73**	**916**	**205**	**1271**	**285**	**2559**
BRYOPHYTA								
Bryopsida		3		2				5
Marchantiopsida		4		1		2		7
LYCOPODIOPHYTA								
Isoetopsida						1		1
Lycopodiopsida			1	2	3	8	4	10
Sellaginellopsida						1		1
POLYPODIOPHYTA								
Polypodiopsida		24	3	22	11	50	14	96
TRACHEOPHYTA								
Coniferopsida		1		6	1	9	1	16
Cycadopsida		19		18		17		54
Liliopsida	1	34	11	114	24	191	36	339
Magnoliopsida	6	287	58	751	166	992	230	2030
Total	**11**	**652**	**91**	**1325**	**211**	**1656**	**313**	**3633**

Cuadro 2. Ordenes de la Clase Magnoliopsida, con el número de especies en que modelaron los nichos ecológicos, proyectados como distribuciones potenciales. Los órdenes, en negritas, indican aquellos en los que se presentan mapas de distribucion potencial y, análisis por países, en la región de estudio.

Orden	No Especies	Orden	No	Orden	No
Apiales	8	Fagales	4	Polygalales	3
Asterales	12	Gentianales	7	Proteales	1
Campanulales	11	Geraniales	4	**Rosales**	12
Capparales	6	Juglandales	2	**Rubiales**	11
Celastrales	3	Lamiales	5	Santalales	1
Cornales	1	**Laurales**	11	Sapindales	9
Dipsacales	2	Lecythidales	7	**Scrophulariales**	22
Ebenales	8	Magnoliales	4	Solanales	4
Ericales	1	Malvales	5	Theales	5
Euphorbiales	3	**Myrtales**	32	Urticales	2
Fabales	17	Piperales	3	Violales	4

Cuadro 3. Porcentaje de la distribución potencial para los diferentes grupos de riqueza de especies que coinciden con áreas transformadas de cada país.

	0	**1- 5**	**6- 10**	**11-20**	**21 a 50**	**51 a 115**
Belice	3.75	15.28	18.69	5.60	1.14	
Colombia	16.55	34.28	25.69	16.22	11.28	26.42
Costa Rica	53.18	0.09	19.81	48.00	18.31	
Ecuador	25.96	15.11	4.72	7.42	9.67	7.09
El Salvador		6.52	24.11	60.14	47.58	
Guatemala	2.06	15.65	21.33	43.90	55.40	41.27
Honduras	2.14	10.64	36.95	31.41	28.75	
México	25.97	30.16	38.53	37.99	33.68	
Nicaragua	4.71	0.86	32.65	48.54	26.25	
Panamá	36.95	25.43	24.38	60.87	38.80	8.24

Cuadro 4. Porcentaje de las distribuciones potenciales para los diferentes grupos de riqueza de especies que coincide con las AP de cada país de la región de estudio.

	0	**1 a 5**	**6 a 10**	**11 a 20**	**21 a 50**	**51 a 111**
Belice	10.01	25.09	33.46	65.20	96.89	
Colombia	1.58	6.63	8.01	10.62	14.40	9.90
Costa Rica	17.61	90.9	27.07	9.60	31.67	84.67
Ecuador		8.60	0.02	0.64	14.37	17.75
El Salvador		2.75	13.16	1.94	4.30	
Guatemala	26.48	65.06	45.78	18.06	14.26	
Honduras	50.33	51.98	26.57	16.94	25.21	
México	31.05	11.33	6.91	10.13	1.82	
Nicaragua	35.83	4.86	17.50	15.15	19.27	50.06
Panamá	0.00	4.15	2.06	3.52	26.16	70.02

Cuadro 5. Area total (km^2), porcentaje en área transformada (% AT) y porcentaje en AP (% AP) para las distribuciones potenciales de los diferentes grupos taxonómicos en la región de estudio.

	Area total	% AT	% AP
Campanulales	274075	11.95	11.92
Liliopsida	319738	11.39	13.91
Asterales	320367	14.68	16.05
Rosales	399428	13.81	16.56
Polypodiophyta	406268	11.96	12.94
Amphibia	434762	20.43	18.72
Scrophulariales	495023	15.55	14.92
Rubiales	742785	17.36	16.57
Myrtales	808800	23.10	13.66
Laurales	857474	29.71	17.86
Fabales	1139650	27.01	14.30

Figura **1**. Mapa del área de estudio que comprende la región biogeográfica de Mesoamérica, El Chocó y los Andes Tropicales.

Figura **2**. Mapa que muestra la concentración del número de especies en áreas transformadas (Transformadas, en rojo) y en las áreas protegidas (AP, en verde), superponiendo las distribuciones potenciales de las 313 especies: 28 vertebrados terrestres y 285 plantas. Ver métodos para mayores detalles. El mapa también indica la riqueza de especies amenazadas incluidas en el análisis en las áreas con vegetación natural remanenete (negro, alto numero de especies; gris tenue, bajo número de especies). Ver texto para mayores detalles.

Figura **3.** Porcentaje del área total del país que presenta distribución potencial de especies amenazadas para los diferentes grupos de riqueza de especies.

Figura **4**. Mapa que indica la distribución potencial de los diferentes grupos taxonómicos, mostrando la superposición de las distribuciones por número de especies en áreas con vegetación natural remanente (negro, alto numero de especies; gris tenue, bajo número de especies), y distribucion potencial en áreas transformadas (rojo) y AP (verde), en la tabla se muestra para cada grupo taxonómico el áreas en Km^2 de la distribución potencial, porcentaje que coinciden en áreas transformadas(% Trasnformado) y porcentaje en AP (%AP) en los países de la región de estudio. (a) Amphibia, (b) Liliopsida, (c) Polipodiopsida, (d) Asterales, (e) Campanulales, (f) Fabales, (g) Laurales, (h) Myrtales, (i) Rosales, (j) Scrophulariales y, (k) Rubiales.

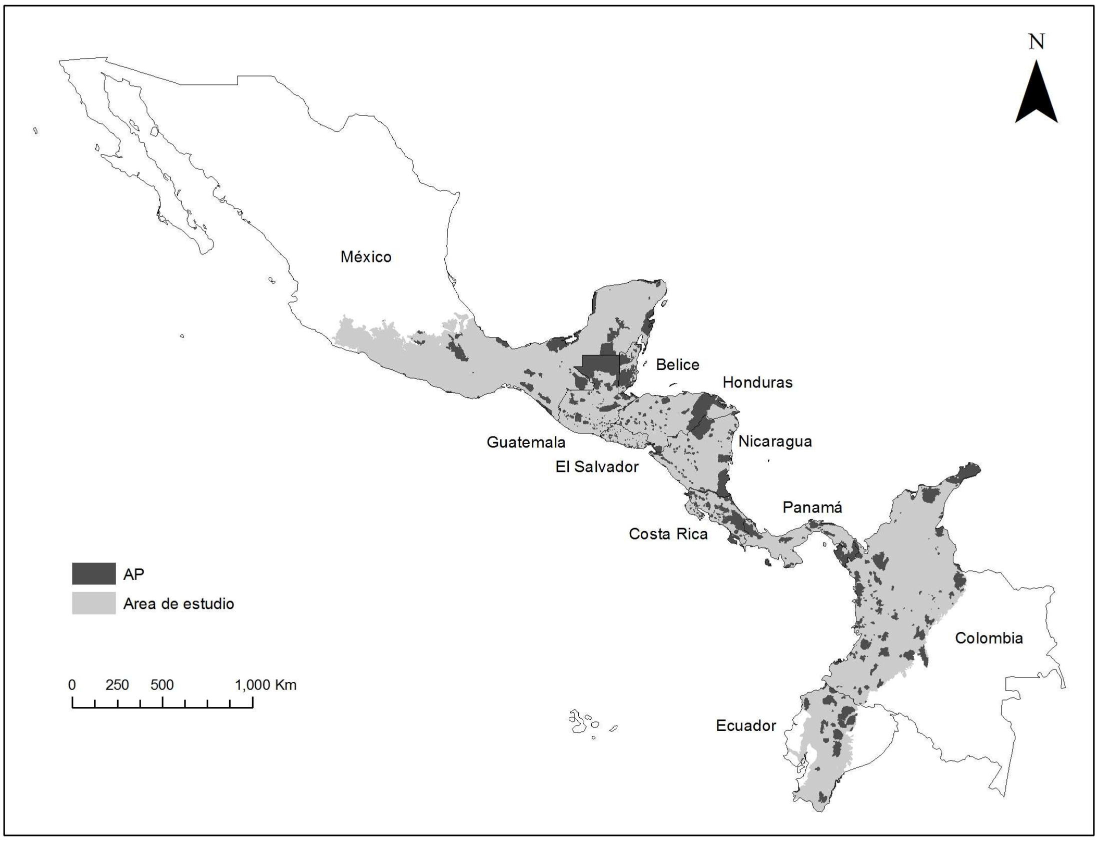

N
México
Belice
Honduras
Guatemala
Nicaragua
El Salvador
Panamá
Costa Rica
AP
Area de estudio
0 250 500 1,000 Km
Colombia
Ecuador

Figura 1.

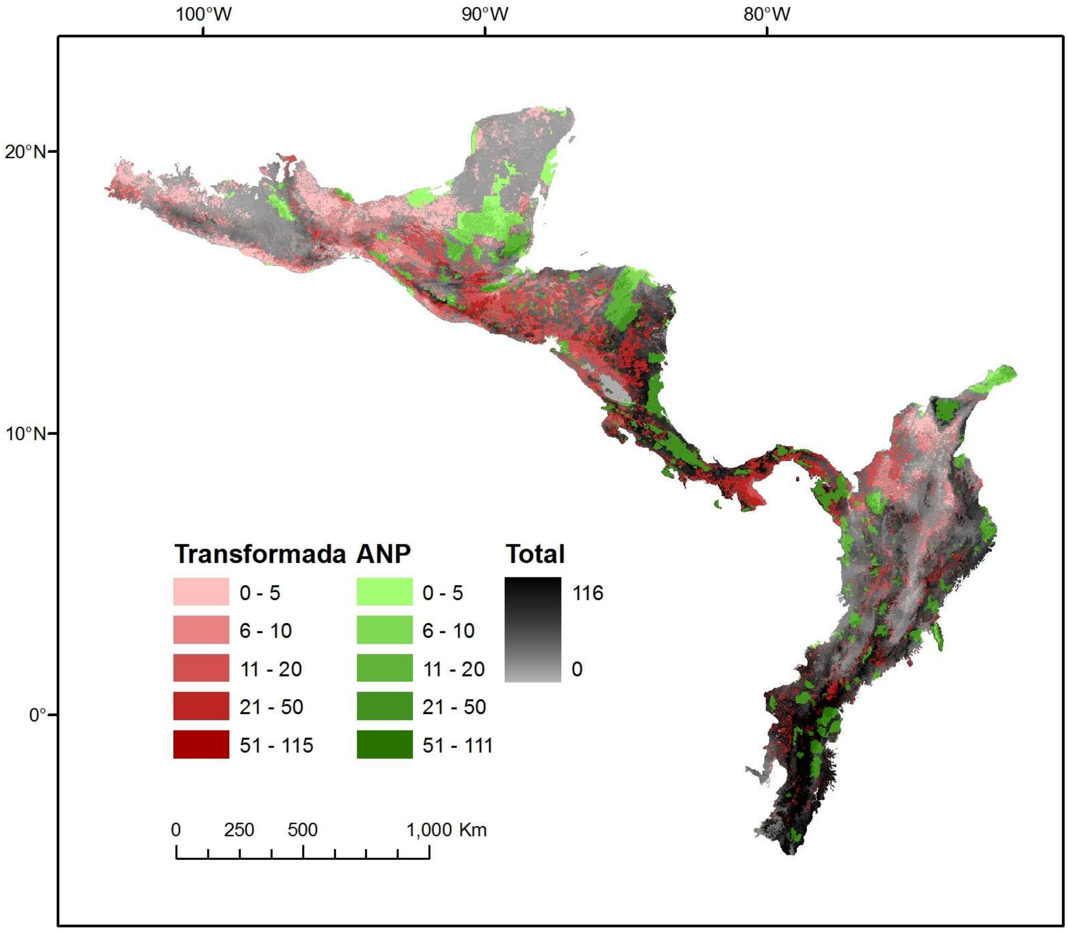

100°W
90°W
80°W
20°N
10°N
0°
Transformada
0 - 5
6 - 10
11 - 20
21 - 50
51 - 115
ANP
0 - 5
6 - 10
11 - 20
21 - 50
51 - 111
Total
116
0
0
250
500
1,000 Km

Figura 2.

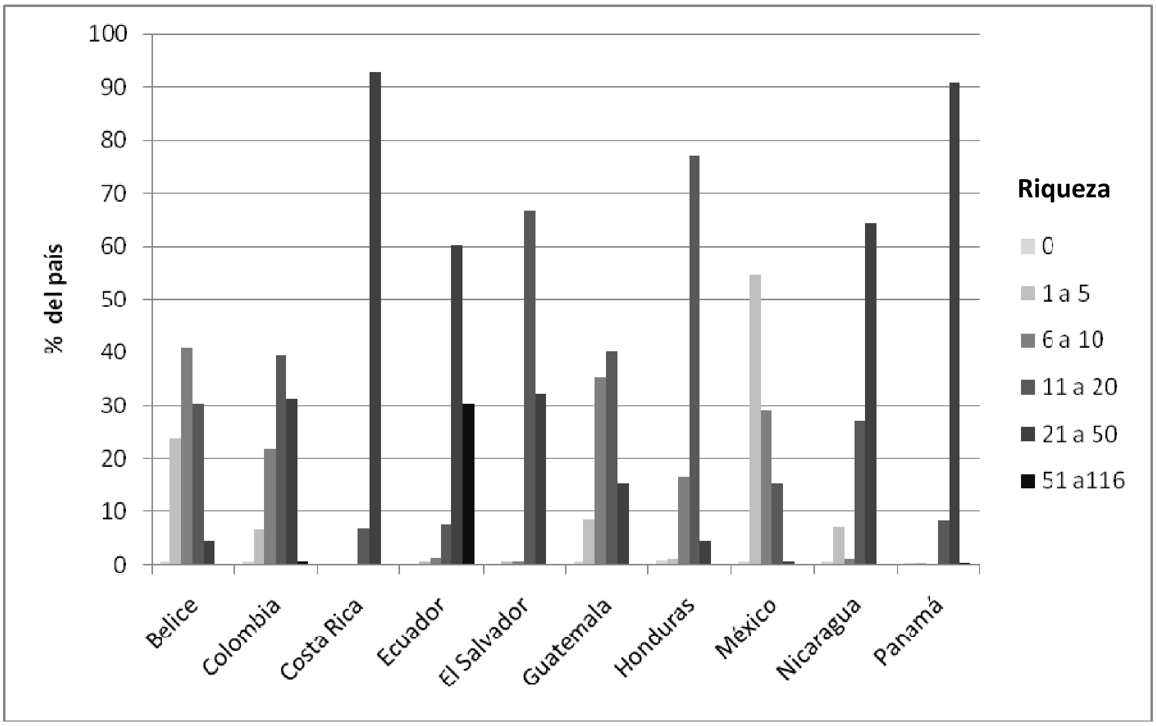
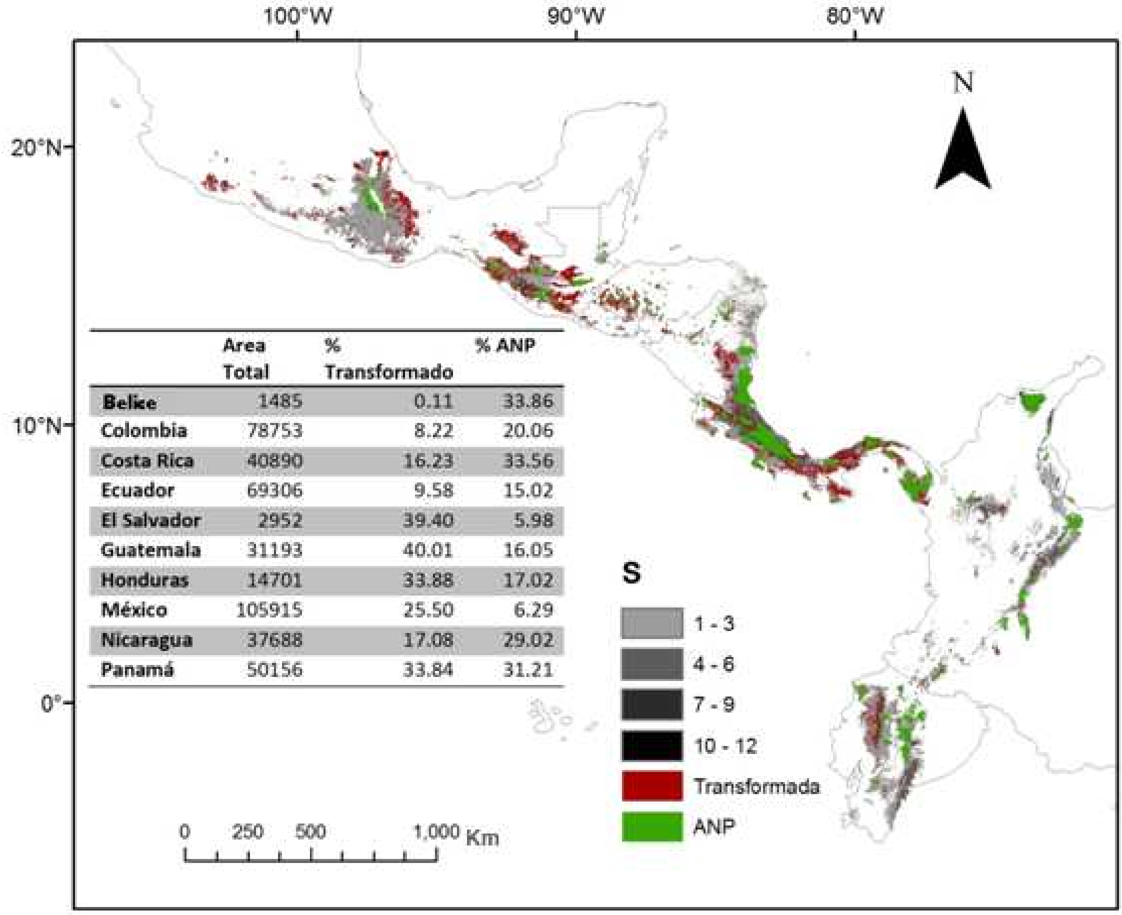
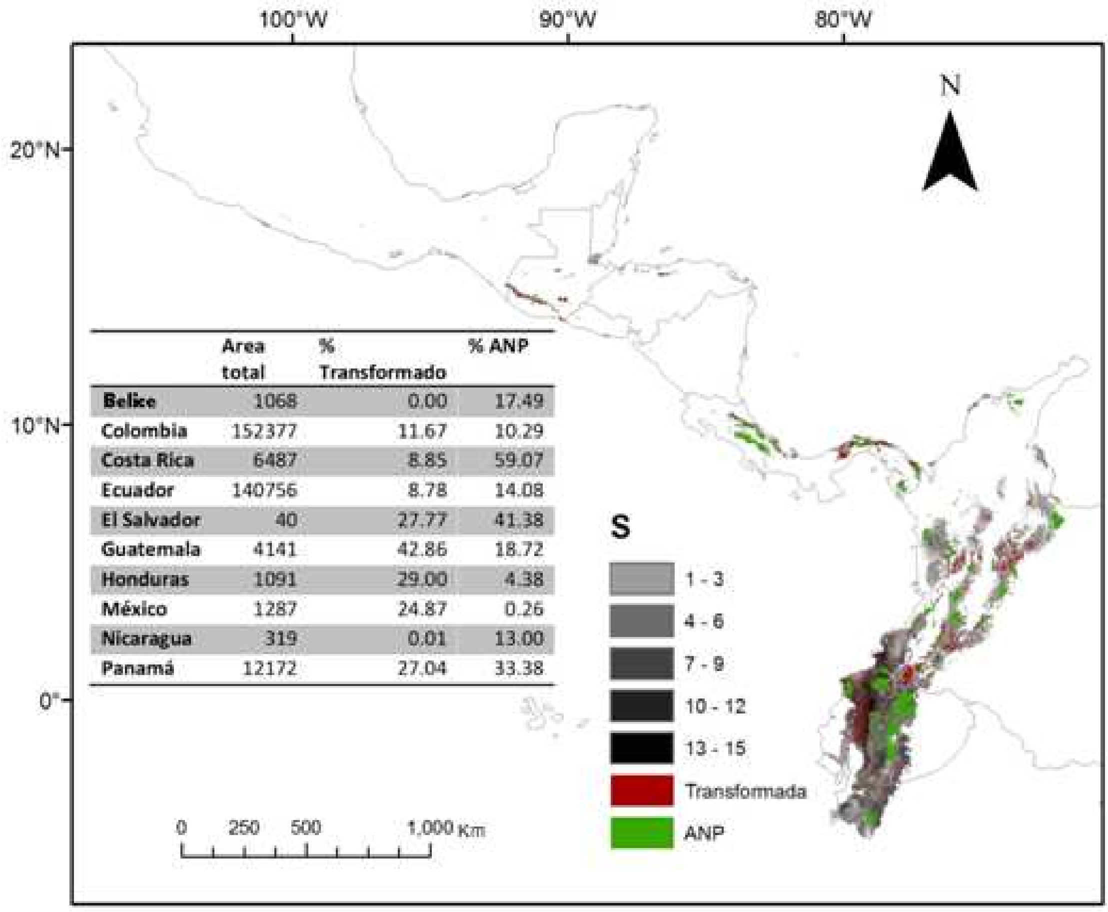

% del país
100
90
80
70
60
50
40
30
20
10
0
Belice
Colombia
Costa Rica
Ecuador
El Salvador
Guatemala
Honduras
México
Nicaragua
Panamá
Riqueza
0
1 a 5
6 a 10
11 a 20
21 a 50
51 a116

Figura 3.



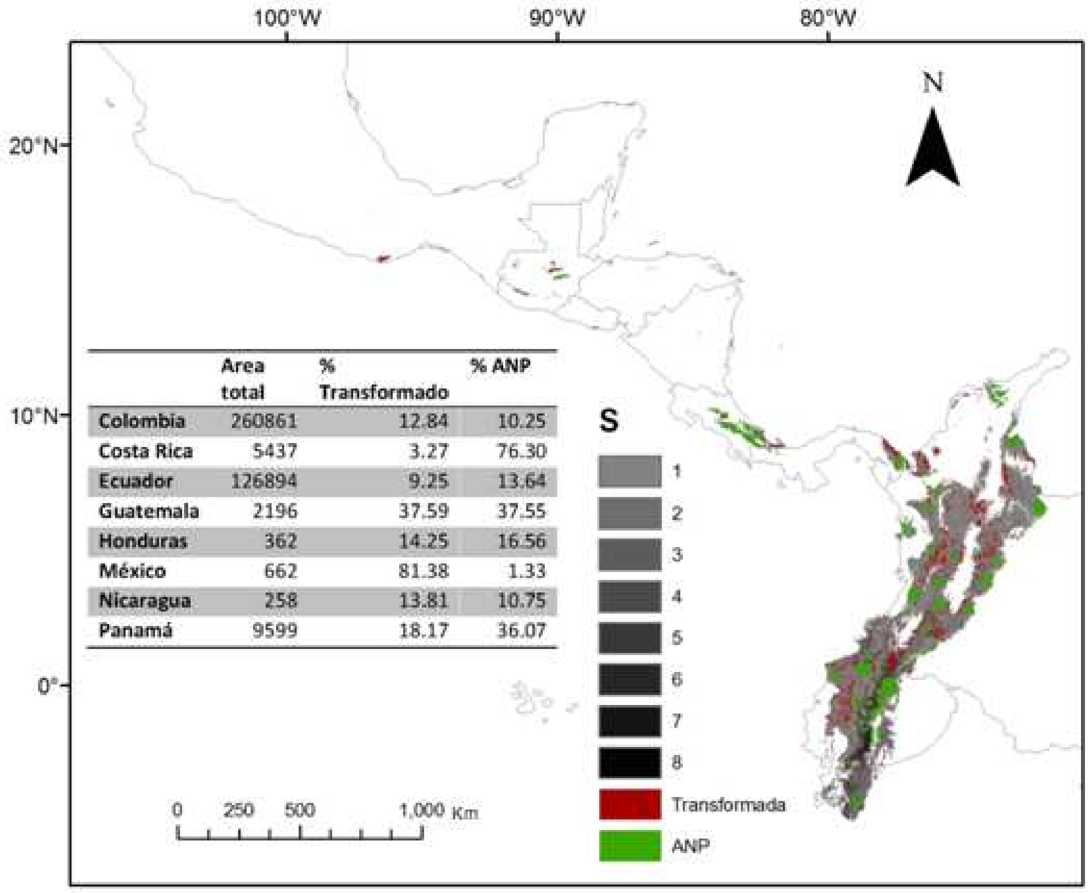
	Area Total	% Transformado	% ANP
Belice	1485	0.11	33.86
Colombia	78753	8.22	20.06
Costa Rica	40890	16.23	33.56
Ecuador	69306	9.58	15.02
El Salvador	2952	39.40	5.98
Guatemala	31193	40.01	16.05
Honduras	14701	33.88	17.02
México	105915	25.50	6.29
Nicaragua	37688	17.08	29.02
Panamá	50156	33.84	31.21

Figura 4 (a)



	Area total	% Transformado	% ANP
Belice	1068	0.00	17.49
Colombia	152377	11.67	10.29
Costa Rica	6487	8.85	59.07
Ecuador	140756	8.78	14.08
El Salvador	40	27.77	41.38
Guatemala	4141	42.86	18.72
Honduras	1091	29.00	4.38
México	1287	24.87	0.26
Nicaragua	319	0.01	13.00
Panamá	12172	27.04	33.38

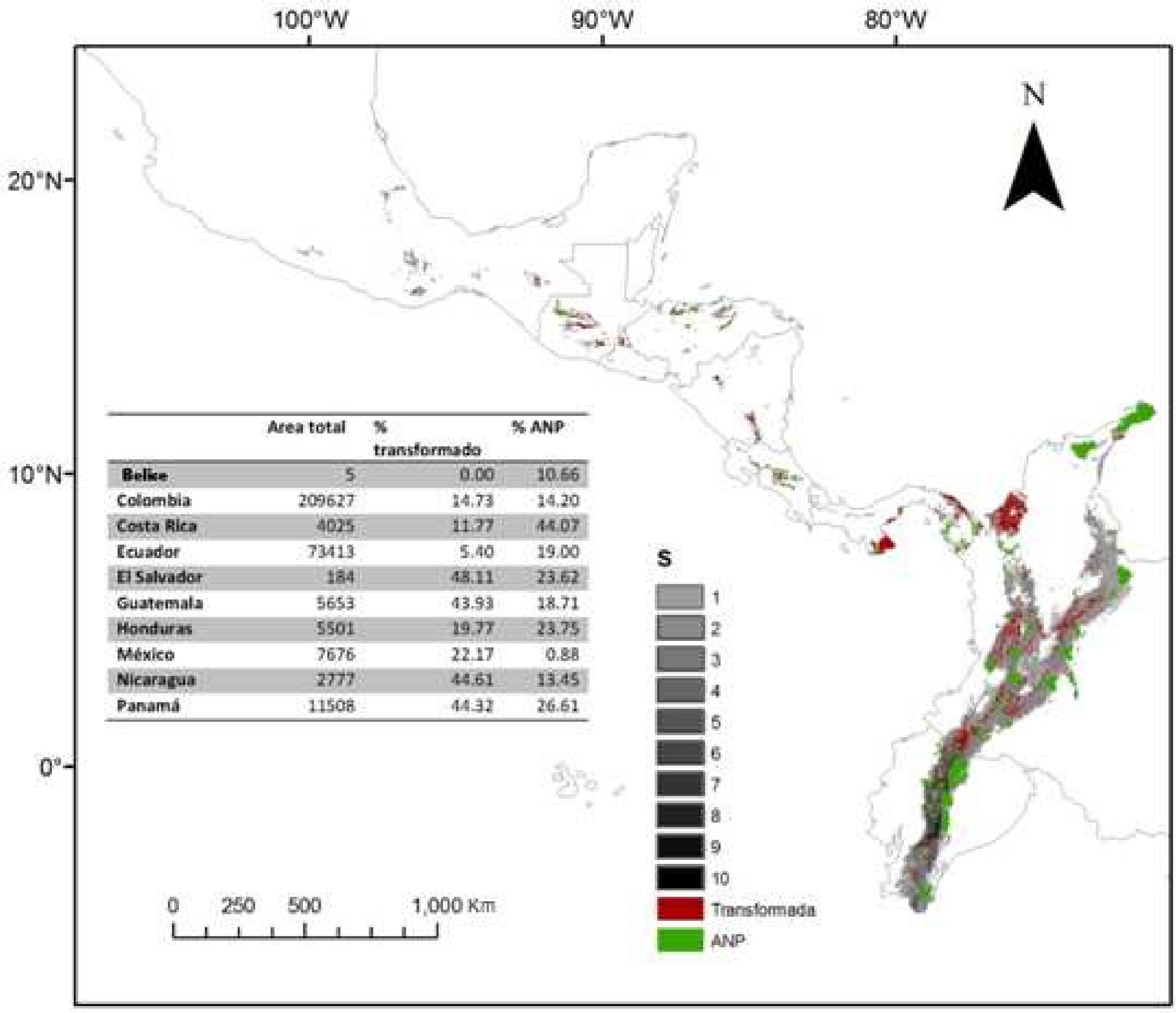
Figura 4 (b)


100°W
90°W
80°W
20°N
10°N
0°
N

	Area total	% Transformado	% ANP
Colombia	260861	12.84	10.25
Costa Rica	5437	3.27	76.30
Ecuador	126894	9.25	13.64
Guatemala	2196	37.59	37.55
Honduras	362	14.25	16.56
México	662	81.38	1.33
Nicaragua	258	13.81	10.75
Panamá	9599	18.17	36.07

S
1
2
3
4
5
6
7
8
Transformada
ANP
0 250 500 1,000 Km

Figura 4 (c)



	Area total	% transformado	% ANP
Belice	5	0.00	10.66
Colombia	209627	14.73	14.20
Costa Rica	4025	11.77	44.07
Ecuador	73413	5.40	19.00
El Salvador	184	48.11	23.62
Guatemala	5653	43.93	18.71
Honduras	5501	19.77	23.75
México	7676	22.17	0.88
Nicaragua	2777	44.61	13.45
Panamá	11508	44.32	26.61

Figura 4 (d)

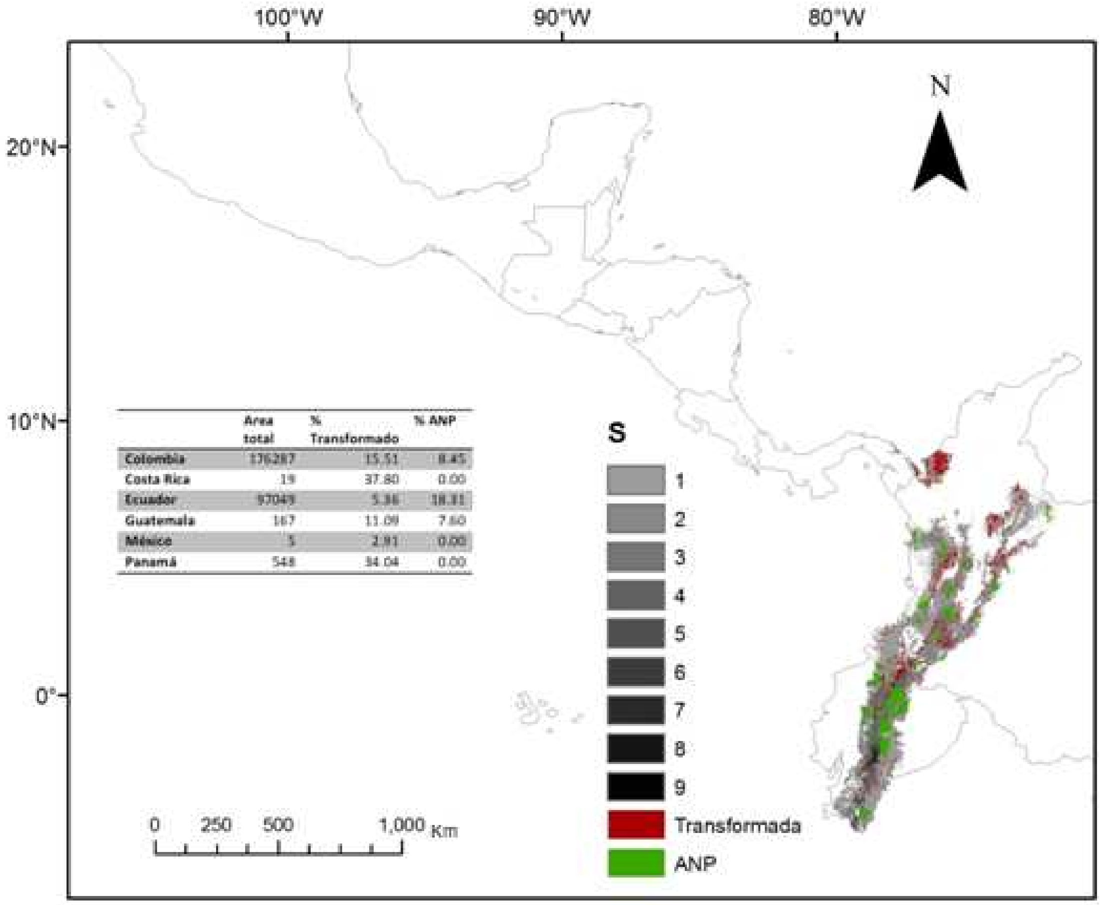
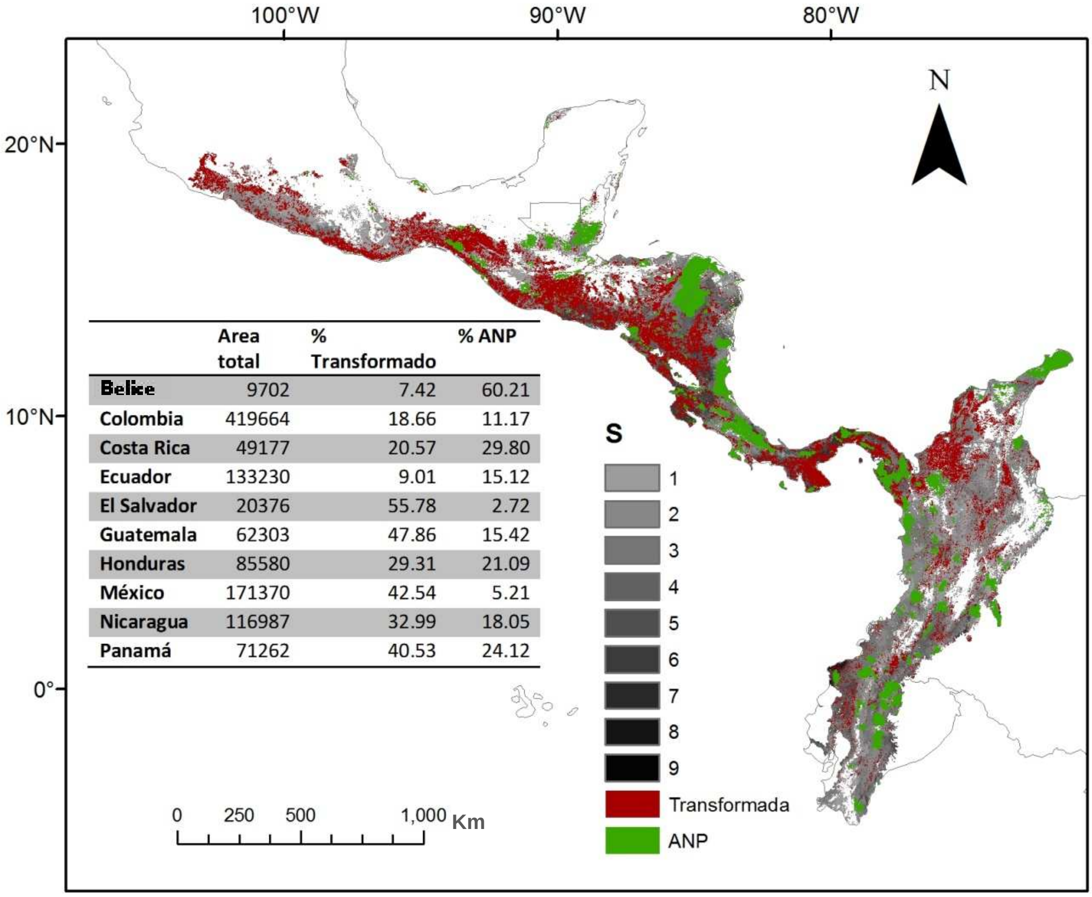
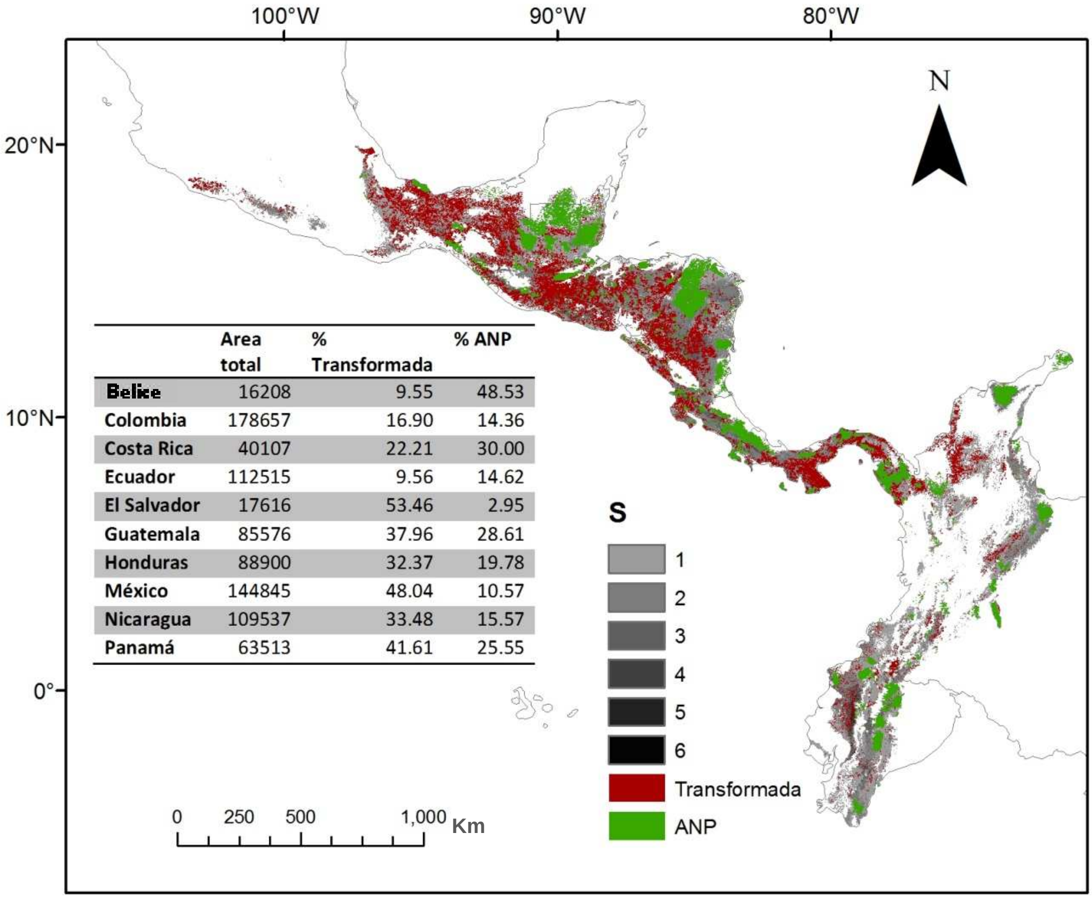

100°W
90°W
80°W
20°N
10°N
0°
N
Area total
% Transformado
% ANP
Colombia 176287 15.51 8.45
Costa Rica 19 37.80 0.00
Ecuador 97049 5.36 18.31
Guatemala 167 11.09 7.60
México 5 2.91 0.00
Panamá 548 34.04 0.00
S
1
2
3
4
5
6
7
8
9
Transformada
ANP
0 250 500 1,000 Km

Figura 4 (e)



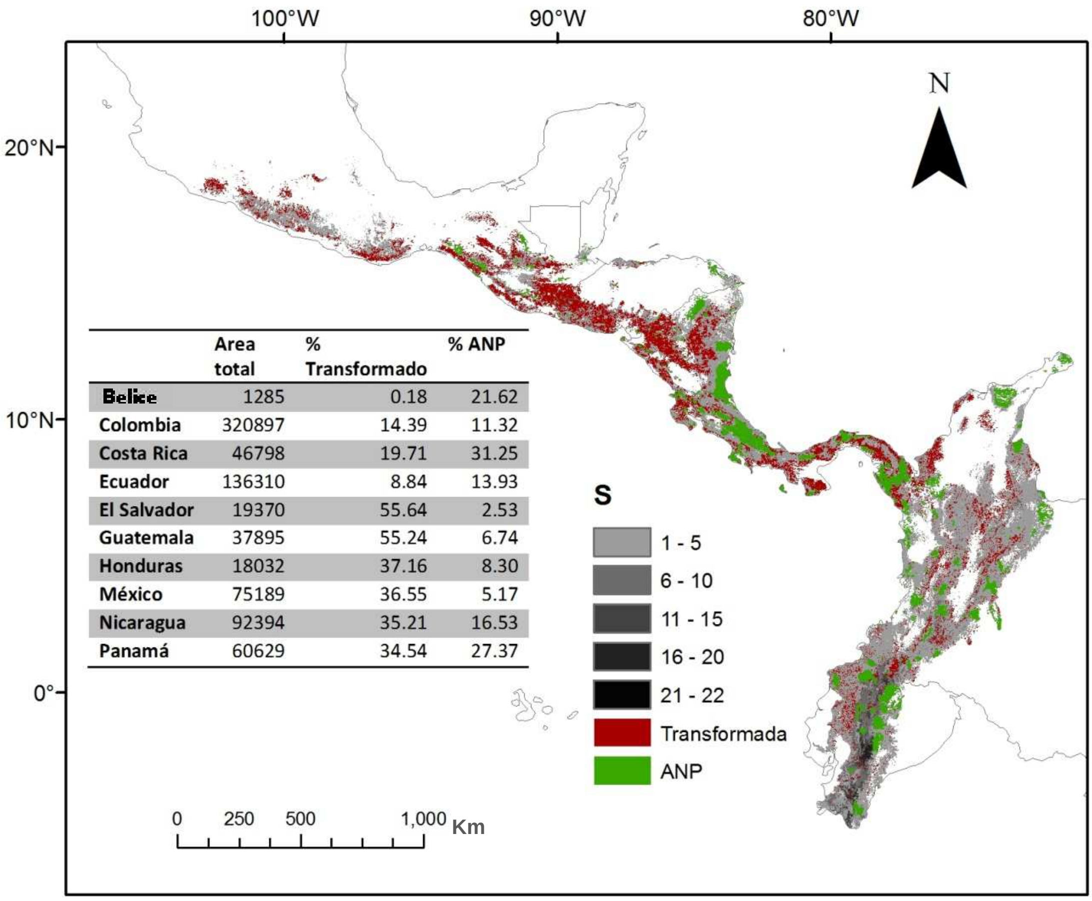
	Area total	% Transformado	% ANP
Belice	9702	7.42	60.21
Colombia	419664	18.66	11.17
Costa Rica	49177	20.57	29.80
Ecuador	133230	9.01	15.12
El Salvador	20376	55.78	2.72
Guatemala	62303	47.86	15.42
Honduras	85580	29.31	21.09
México	171370	42.54	5.21
Nicaragua	116987	32.99	18.05
Panamá	71262	40.53	24.12

Figura 4 (f)



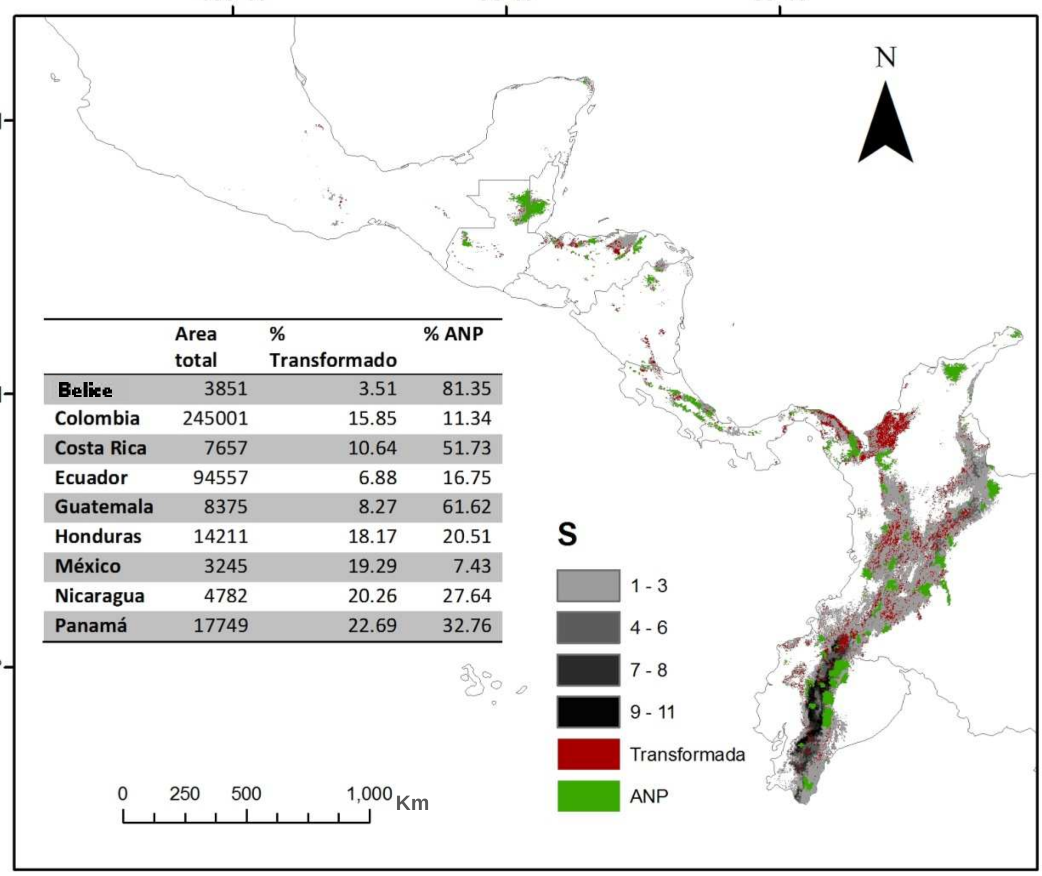
	Area total	% Transformada	% ANP
Belice	16208	9.55	48.53
Colombia	178657	16.90	14.36
Costa Rica	40107	22.21	30.00
Ecuador	112515	9.56	14.62
El Salvador	17616	53.46	2.95
Guatemala	85576	37.96	28.61
Honduras	88900	32.37	19.78
México	144845	48.04	10.57
Nicaragua	109537	33.48	15.57
Panamá	63513	41.61	25.55

Figura 4 (g)



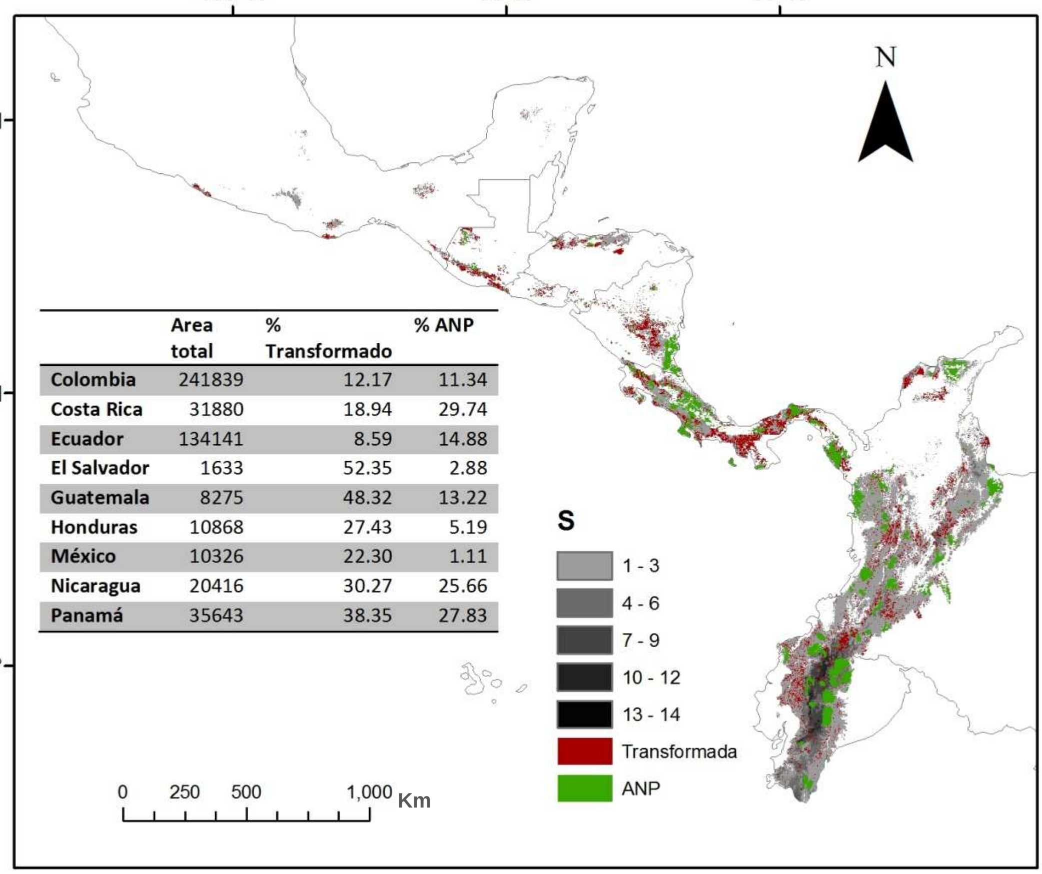
	Area total	% Transformado	% ANP
Belice	1285	0.18	21.62
Colombia	320897	14.39	11.32
Costa Rica	46798	19.71	31.25
Ecuador	136310	8.84	13.93
El Salvador	19370	55.64	2.53
Guatemala	37895	55.24	6.74
Honduras	18032	37.16	8.30
México	75189	36.55	5.17
Nicaragua	92394	35.21	16.53
Panamá	60629	34.54	27.37

Figura 4 (h)



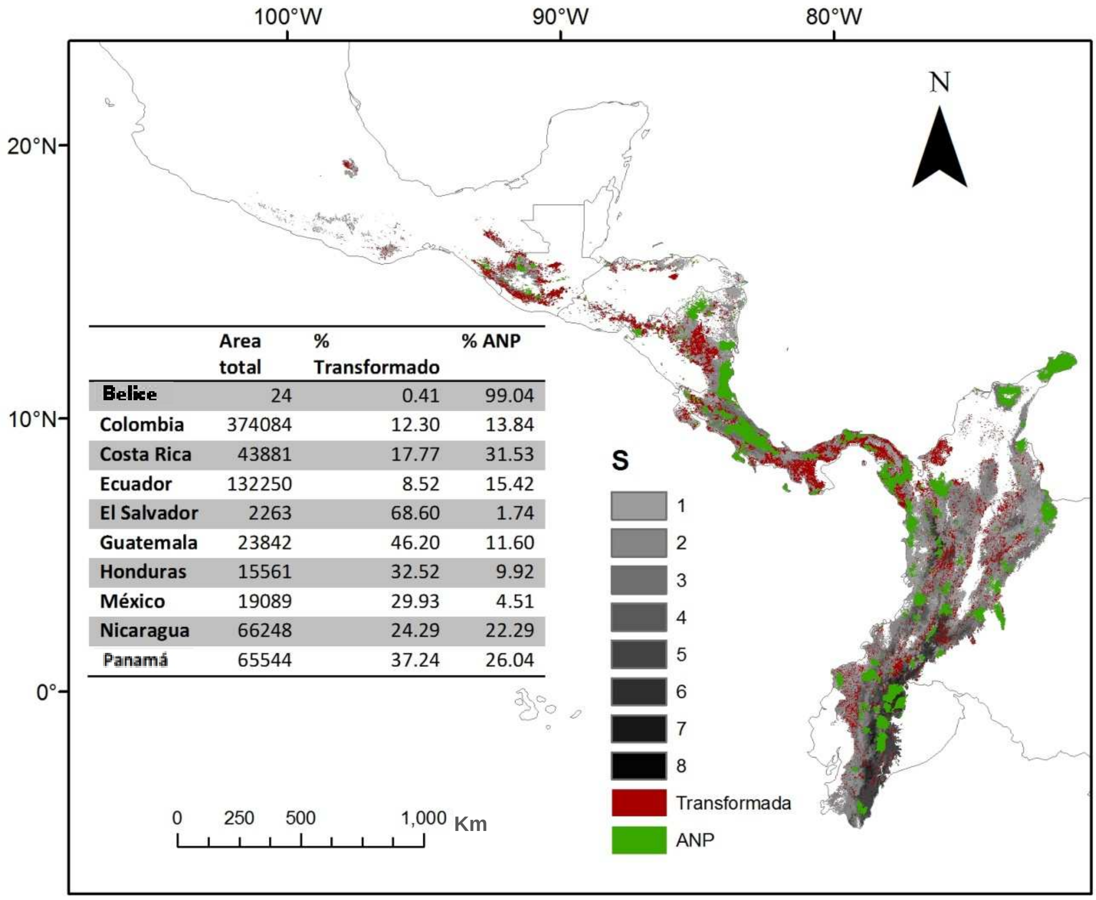
	Area total	% Transformado	% ANP
Belice	3851	3.51	81.35
Colombia	245001	15.85	11.34
Costa Rica	7657	10.64	51.73
Ecuador	94557	6.88	16.75
Guatemala	8375	8.27	61.62
Honduras	14211	18.17	20.51
México	3245	19.29	7.43
Nicaragua	4782	20.26	27.64
Panamá	17749	22.69	32.76

Figura 4 (i)



	Area total	% Transformado	% ANP
Colombia	241839	12.17	11.34
Costa Rica	31880	18.94	29.74
Ecuador	134141	8.59	14.88
El Salvador	1633	52.35	2.88
Guatemala	8275	48.32	13.22
Honduras	10868	27.43	5.19
México	10326	22.30	1.11
Nicaragua	20416	30.27	25.66
Panamá	35643	38.35	27.83

Figura 4 (j)



	Area total	% Transformado	% ANP
Belice	24	0.41	99.04
Colombia	374084	12.30	13.84
Costa Rica	43881	17.77	31.53
Ecuador	132250	8.52	15.42
El Salvador	2263	68.60	1.74
Guatemala	23842	46.20	11.60
Honduras	15561	32.52	9.92
México	19089	29.93	4.51
Nicaragua	66248	24.29	22.29
Panamá	65544	37.24	26.04

Figura 4 (k)

Cuadro suplementario CS1: Clasificación taxonómica de especies utilizadas en el análisis, número de registros utilizados para hacer los modelos y principales resultados de los modelos de distribución potencial. Las columnas variable y % hacen referencia a la variable que más contribuyó al modelo y el porcentaje de su contribución.

Clase	Orden	Especie	Categoría UICN	Registros usados	AUC prueba	Valor umbral	Variable	%
Amphibia	Anura	*Atelopus varius*	CR	16	0.965	0.158	Bio19	27.20
Amphibia	Anura	*Bufo aucoinae*	Vu	15	0.999	0.848	Bio13	33.90
Amphibia	Anura	*Phyllobates vitattus*	En	5	1	0.939	Bio13	36.00
Amphibia	Anura	*Plectrohyla glandulosa*	En	13	1	0.005	Bio6	44.90
Amphibia	Anura	*Ptychohyla salvadorensis*	En	4	0.999	0.59	Bio19	32.80
Amphibia	Anura	*Eleutherodactylus rubrimaculatus*	Vu	4	0.999	0.864	Bio13	62.40
Amphibia	Anura	*Rana vibicaria*	CR	18	0.998	0.151	Bio12	25.30
Amphibia	Caudata	*Bolitoglossa engelhardti*	En	21	0.996	0.475	Bio18	46.80
Amphibia	Caudata	*Bolitoglossa flavimembris*	En	16	1	0.825	Bio18	47.40
Amphibia	Caudata	*Bolitoglossa pesrubra*	En	23	1	0.13	Bio16	24.80
Amphibia	Caudata	*Bolitoglossa riletti*	En	6	0.998	0.578	Bio2	72.20
Amphibia	Caudata	*Bolitoglossa subpalmata*	En	24	0.997	0.127	Altitud	22.50
Amphibia	Caudata	*Dendrotriton bromeliacius*	En	27	1	0.904	Bio18	47.20
Amphibia	Caudata	*Lineatriton lineolus*	En	7	0.996	0.416	Bio4	56.10
Amphibia	Caudata	*Oedipina poelzi*	En	15	0.997	0.129	Bio19	20.60
Amphibia	Caudata	*Oedipina pseudouniformis*	En	14	0.97	0.285	Bio15	36.20
Amphibia	Caudata	*Oedipina uniformis*	En	41	0.995	0.099	Bio4	20.90
Amphibia	Caudata	*Parvimolge townsendi*	En	6	0.999	0.8	Bio4	44.30
Amphibia	Caudata	*Pseudoeurycea brunnata*	En	18	1	0.907	Bio18	40.80
Amphibia	Caudata	*Pseudoeurycea cochranae*	En	12	0.995	0.068	Bio6	56.60
Amphibia	Caudata	*Pseudoeurycea goebeli*	En	23	1	0.056	Bio18	40.90
Amphibia	Caudata	*Pseudoeurycea juarezi*	En	15	0.961	0.734	Altitud	31.30
Amphibia	Caudata	*Thorius magnipes*	CR	6	0.997	0.553	Bio6	44.60
Amphibia	Caudata	*Thorius narisovalis*	Cr	14	1	0.435	Bio6	57.60

Aves	Psittaciformes	*Amazona finschi*	Vu	6	0.999	0.584	Bio15	38.50
Mammalia	Chiroptera	*Artibeus inopinatus*	Vu	3	0.998	0.661	Bio17	35.80
Mammalia	Didelphimorphia	*Caluromys derbianus*	Vu	28	0.804	0.077	Bio19	23.80
Mammalia	Soricomorpha	*Cryptotis gracilis*	Vu	11	1	0.264	Bio1	27.10
Lycopodiopsida	Lycopodiales	*Huperzia austroecuadorica*	Vu	6	0.995	0.548	Altitud	41.00
Lycopodiopsida	Lycopodiales	*Huperzia compacta*	Vu	8	0.99	0.264	Altitud	54.00
Lycopodiopsida	Lycopodiales	*Huperzia llanganatensis*	Vu	6	0.993	0.371	Altitud	50.50
Lycopodiopsida	Lycopodiales	*Huperzia loxensis*	En	4	0.997	0.485	Altitud	33.30
Polypodiopsida	Blechnales	*Elaphoglossum antisanae*	Vu	10	0.993	0.375	Altitud	38.20
Polypodiopsida	Blechnales	*Elaphoglossum yatesii*	Vu	7	0.998	0.22	Altitud	52.40
Polypodiopsida	Blechnales	*Diplazium divisissimum*	Vu	12	0.991	0.05	Bio18	37.70
Polypodiopsida	Blechnales	*Diplazium oellgaardii*	Vu	11	0.85	0.562	Bio3	31.80
Polypodiopsida	Cyatheales	*Alsophila esmeraldensis*	En	6	1	0.884	Bio18	29.00
Polypodiopsida	Cyatheales	*Cyathea palaciosii*	En	5	0.994	0.366	Bio15	69.00
Polypodiopsida	Cyperales	*Calamagrostis aurea*	Vu	10	0.997	0.353	Altitud	50.40
Polypodiopsida	Cyperales	*Calamagrostis llanganatensis*	Vu	4	1	0.812	Altitud	45.10
Polypodiopsida	Cyperales	*Chusquea falcata*	Vu	7	0.994	0.595	Altitud	43.20
Polypodiopsida	Cyperales	*Neurolepis elata*	En	13	0.958	0.258	Altitud	22.20
Polypodiopsida	Cyperales	*Neurolepis laegaardii*	Vu	10	0.994	0.58	Altitud	22.6
Polypodiopsida	Cyperales	*Neurolepis villosa*	Vu	4	0.998	0.475	Altitud	41.20
Polypodiopsida	Hymenophyllales	*Trichomanes paucisorum*	Vu	6	0.997	0.671	Bio3	36.60
Polypodiopsida	Polypodiales	*Polypodium latissimum*	Vu	6	0.998	0.375	Bio15	84.20
Coniferopsida	Coniferales	*Pinus tecunumanii*	Vu	39	0.952	0.092	Altitud	31.80
Liliopsida	Arales	*Anthurium balslevii*	Vu	10	0.99	0.264	Bio18	56.60
Liliopsida	Arales	*Anthurium cutucuense*	En	9	0.933	0.331	Bio15	84.40
Liliopsida	Arales	*Anthurium esmeraldense*	Vu	4	0.999	0.7	Bio18	27.40
Liliopsida	Arales	*Anthurium gualeanum*	Vu	7	0.999	0.414	Bio19	30.90
Liliopsida	Arales	*Anthurium magnifolium*	Vu	4	0.974	0.479	Bio18	46.80
Liliopsida	Arales	*Anthurium pedunculare*	Vu	13	0.999	0.118	Bio18	23.50

Liliopsida	Arales	*Anthurium rimbachii*	Vu	18	0.999	0.533	Bio18	60.30
Liliopsida	Arales	*Anthurium subcoerulescens*	Vu	7	0.997	0.169	Bio18	59.70
Liliopsida	Arales	*Anthurium tenuifolium*	En	5	1	0.707	Bio4	68.50
Liliopsida	Arales	*Chlorospatha ilensis*	Vu	9	0.992	0.477	Bio15	38.20
Liliopsida	Arales	*Philodendron hooveri*	Vu	15	0.971	0.43	Bio18	55.90
Liliopsida	Arales	*Philodendron musifolium*	Vu	14	0.994	0.366	Bio4	50.30
Liliopsida	Arales	*Philodendron pogonocaule*	CR	18	0.948	0.177	Bio18	49.90
Liliopsida	Arales	*Stenospermation arborescens*	En	6	0.997	0.746	Bio15	40.60
Liliopsida	Arales	*Xanthosoma eggersii*	En	7	0.998	0.555	Bio18	44.50
Liliopsida	Arecales	*Aiphanes chiribogensis*	Vu	10	0.984	0.332	Bio4	55.20
Liliopsida	Arecales	*Aiphanes grandis*	En	4	0.998	0.501	Bio4	42.80
Liliopsida	Arecales	*Aiphanes verrucosa*	En	5	0.997	0.639	Altitud	24.90
Liliopsida	Arecales	*Bactris coloniata*	Vu	16	0.998	0.539	Bio2	46.20
Liliopsida	Bromeliales	*Aechmea aculeatosepala*	Vu	12	0.973	0.221	Bio15	60.40
Liliopsida	Bromeliales	*Guzmania alborosea*	Vu	11	0.999	0.496	Bio18	37.10
Liliopsida	Bromeliales	*Guzmania fusispica*	Vu	9	0.993	0.487	Bio4	42.20
Liliopsida	Bromeliales	*Guzmania hollinensis*	Vu	9	0.966	0.538	Bio15	88.40
Liliopsida	Bromeliales	*Guzmania madisonii*	Vu	11	0.997	0.291	Bio15	58.80
Liliopsida	Bromeliales	*Mezobromelia fulgens*	En	5	0.994	0.596	Altitud	26.80
Liliopsida	Bromeliales	*Pitcairnia ferrell-ingramiae*	Vu	18	0.999	0.303	Bio18	38.60
Liliopsida	Bromeliales	*Pitcairnia hirtzii*	Vu	9	0.993	0.291	Bio15	65.30
Liliopsida	Bromeliales	*Puya obconica*	Vu	5	0.993	0.619	Bio19	26.30
Liliopsida	Bromeliales	*Puya pygmaea*	Vu	11	0.983	0.499	Bio9	28.50
Liliopsida	Bromeliales	*Racinaea tandapiana*	Vu	7	0.999	0.467	Bio4	56.10
Liliopsida	Bromeliales	*Racinaea tripinnata*	En	8	0.996	0.407	Altitud	62.70
Liliopsida	Bromeliales	*Ronnbergia campanulata*	En	6	0.999	0.36	Bio15	44.80
Liliopsida	Bromeliales	*Tillandsia emergens*	Vu	9	0.973	0.323	Altitud	74.70
Liliopsida	Bromeliales	*Tillandsia rhodosticta*	Vu	7	0.986	0.494	Bio15	75.50
Liliopsida	Liliales	*Bomarea chimborazensis*	En	9	0.986	0.357	Altitud	75.30

Liliopsida	Liliales	*Bomarea uncifolia*	En	11	0.98	0.406	Altitud	56.80
Magnoliopsida	Apiales	*Dendropanax sessiliflorus*	Vu	15	0.962	0.109	Bio19	30.00
Magnoliopsida	Apiales	*Oreopanax hedraeostrobilus*	Vu	4	0.995	0.457	Bio15	39.70
Magnoliopsida	Apiales	*Oreopanax obscurus*	Vu	6	1	0.553	Bio15	40.90
Magnoliopsida	Apiales	*Oreopanax rosei*	Vu	7	0.99	0.305	Bio18	32.70
Magnoliopsida	Apiales	*Oreopanax sessiliflorus*	Vu	21	0.995	0.057	Altitud	37.20
Magnoliopsida	Apiales	*Schefflera brenesii*	Vu	17	0.939	0.064	Bio4	19.50
Magnoliopsida	Apiales	*Schefflera diplodactyla*	Vu	17	0.993	0.303	Bio14	59.70
Magnoliopsida	Apiales	*Hydrocotyle yanghuangensis*	Vu	6	0.98	0.523	Altitud	47.30
Magnoliopsida	Asterales	*Aequatorium jamesonii*	Vu	5	0.996	0.545	Altitud	81.00
Magnoliopsida	Asterales	*Ageratina dendroides*	Vu	10	0.999	0.452	Altitud	27.60
Magnoliopsida	Asterales	*Baccharis hieronymi*	Vu	4	0.99	0.605	Altitud	37.70
Magnoliopsida	Asterales	*Cronquistianthus niveus*	Vu	11	0.994	0.182	Bio4	28.50
Magnoliopsida	Asterales	*Diplostephium barclayanum*	Vu	4	0.999	0.515	Altitud	31.00
Magnoliopsida	Asterales	*Grosvenoria rimbachii*	Vu	11	0.971	0.103	Altitud	39.00
Magnoliopsida	Asterales	*Gynoxys baccharoides*	Vu	5	0.998	0.354	Altitud	89.70
Magnoliopsida	Asterales	*Gynoxys miniphylla*	Vu	12	0.994	0.01	Altitud	58.30
Magnoliopsida	Asterales	*Gynoxys reinaldii*	Vu	9	0.992	0.606	Bio4	31.50
Magnoliopsida	Asterales	*Loricaria scolopendra*	Vu	4	0.999	0.762	Bio15	36.70
Magnoliopsida	Asterales	*Mutisia microcephala*	Vu	6	0.998	0.579	Altitud	54.40
Magnoliopsida	Asterales	*Mutisia microphylla*	Vu	6	0.999	0.547	Altitud	57.10
Magnoliopsida	Campanulales	*Burmeistera holm-nielsenii*	En	7	0.991	0.073	Bio4	94.60
Magnoliopsida	Campanulales	*Centropogon aequatorialis*	En	6	0.997	0.35	Bio4	75.20
Magnoliopsida	Campanulales	*Centropogon baezanus*	Vu	24	0.989	0.3	Bio14	32.00
Magnoliopsida	Campanulales	*Centropogon comosus*	En	5	0.985	0.555	Altitud	24.20
Magnoliopsida	Campanulales	*Centropogon erythraeus*	En	9	0.979	0.642	Altitud	26.30
Magnoliopsida	Campanulales	*Centropogon medusa*	En	5	0.999	0.283	Altitud	57.40
Magnoliopsida	Campanulales	*Centropogon*	En	6	0.989	0.586	Bio4	42.70

		rimbachii						
Magnoliopsida	Campanulales	*Centropogon steyermarkii*	En	10	0.997	0.614	Altitud	22.50
Magnoliopsida	Campanulales	*Centropogon ursinus*	En	10	0.986	0.465	Bio1	19.60
Magnoliopsida	Campanulales	*Siphocampylus affinis*	Vu	9	0.983	0.265	Bio15	62.90
Magnoliopsida	Campanulales	*Siphocampylus ecuadoriensis*	En	9	0.997	0.421	Altitud	59.20
Magnoliopsida	Capparales	*Podandrogyne brevipedunculata*	En	18	0.98	0.159	Bio18	47.40
Magnoliopsida	Capparales	*Cardamine lojanensis*	Vu	4	0.996	0.651	Altitud	35.60
Magnoliopsida	Capparales	*Draba aretioides*	En	15	0.996	0.11	Bio5	67.90
Magnoliopsida	Capparales	*Draba splendens*	Vu	4	0.996	0.347	Altitud	60.10
Magnoliopsida	Capparales	*Draba spruceana*	Vu	6	0.993	0.622	Altitud	39.50
Magnoliopsida	Capparales	*Eudema nubigena*	En	11	0.998	0.345	Altitud	65.50
Magnoliopsida	Celastrales	*Ilex costaricensis*	Vu	22	0.997	0.039	Altitud	23.30
Magnoliopsida	Celastrales	*Ilex pallida*	Vu	32	0.973	0.015	Bio8	22.50
Magnoliopsida	Celastrales	*Ilex vulcanicola*	Vu	12	0.89	0.32	Bio13	18.00
Magnoliopsida	Cornales	*Cornus disciflora*	Vu	62	0.931	0.026	Altitud	38.50
Magnoliopsida	Dipsacales	*Viburnum divaricatum*	Cr	5	1	0.655	Bio18	43.90
Magnoliopsida	Dipsacales	*Valeriana cernua*	Vu	4	0.994	0.494	Altitud	68.00
Magnoliopsida	Ebenales	*Pouteria calistophylla*	Vu	12	0.98	0.441	Bio2	38.50
Magnoliopsida	Ebenales	*Pouteria capacifolia*	CR	7	0.968	0.588	Bio18	72.30
Magnoliopsida	Ebenales	*Pouteria collina*	Vu	14	0.989	0.083	Bio19	28.80
Magnoliopsida	Ebenales	*Pouteria congestifolia*	Vu	18	0.957	0.146	Bio19	35.50
Magnoliopsida	Ebenales	*Pouteria fossicola*	Vu	22	0.929	0.212	Bio19	33.10
Magnoliopsida	Ebenales	*Pouteria juruana*	En	20	0.98	0.218	Bio19	28.90
Magnoliopsida	Ebenales	*Pradosia montana*	Vu	4	1	0.647	Bio15	54.10
Magnoliopsida	Ebenales	*Symplocos fuscata*	Vu	15	0.996	0.507	Bio8	39.30
Magnoliopsida	Ericales	*Macleania loeseneriana*	Vu	16	0.987	0.109	Bio4	32.20
Magnoliopsida	Euphorbiales	*Croton coriaceus*	Vu	10	0.985	0.203	Altitud	44.90
Magnoliopsida	Euphorbiales	*Croton elegans*	Vu	9	0.986	0.476	Bio4	27.30
Magnoliopsida	Euphorbiales	*Garcia nutans*	En	48	0.914	0.024	Bio4	19.90
Magnoliopsida	Fabales	*Abarema killipii*	Vu	14	0.991	0.198	Bio15	57.40
Magnoliopsida	Fabales	*Astragalus sprucei*	Vu	7	0.994	0.429	Altitud	59.60
Magnoliopsida	Fabales	*Bauhinia pichinchensis*	Vu	11	0.991	0.079	Bio18	49.90
Magnoliopsida	Fabales	*Coursetia gracilis*	Vu	10	1	0.371	Bio4	43.90
Magnoliopsida	Fabales	*Dalbergia retusa*	Vu	18	0.955	0.383	Bio14	32.80
Magnoliopsida	Fabales	*Inga carinata*	En	6	0.994	0.613	Bio19	34.40
Magnoliopsida	Fabales	*Inga extra-nodis*	Vu	8	0.995	0.26	Bio15	55.60
Magnoliopsida	Fabales	*Inga litoralis*	En	12	0.998	0.157	Bio13	22.80

Magnoliopsida	Fabales	*Inga mortoniana*	En	19	0.926	0.24	Bio19	37.00
Magnoliopsida	Fabales	*Inga mucuna*	Vu	18	0.999	0.069	Bio2	32.60
Magnoliopsida	Fabales	*Lennea viridiflora*	Vu	12	0.978	0.213	Altitud	32.50
Magnoliopsida	Fabales	*Lonchocarpus phaseolifolius*	CR	45	0.97	0.008	Bio14	55.50
Magnoliopsida	Fabales	*Lonchocarpus phlebophyllus*	En	33	0.979	0.051	Bio14	36.90
Magnoliopsida	Fabales	*Lonchocarpus retiferus*	En	22	0.942	0.04	Bio4	48.50
Magnoliopsida	Fabales	*Lupinus nubigenus*	En	6	1	0.698	Altitud	48.30
Magnoliopsida	Fabales	*Swartzia haughtii*	Vu	26	0.851	0.031	Bio18	53.10
Magnoliopsida	Fabales	*Swartzia oraria*	CR	13	0.958	0.229	Bio19	24.30
Magnoliopsida	Fagales	*Quercus bumelioides*	Vu	59	0.994	0.004	Altitud	21.00
Magnoliopsida	Fagales	*Quercus costaricensis*	Vu	37	0.991	0.231	Altitud	22.60
Magnoliopsida	Fagales	*Quercus purulhana*	Vu	26	0.933	0.076	Bio4	29.00
Magnoliopsida	Fagales	*Ticodendron incognitum*	Vu	46	0.977	0.02	Altitud	26.40
Magnoliopsida	Gentianales	*Stemmadenia pauli*	Vu	16	1	0.719	Bio19	43.50
Magnoliopsida	Gentianales	*Gentianella hirculus*	En	8	0.998	0.53	Altitud	50.20
Magnoliopsida	Gentianales	*Gentianella hypericoides*	Vu	4	0.998	0.485	Bio15	48.30
Magnoliopsida	Gentianales	*Gentianella hyssopifolia*	Vu	12	0.995	0.146	Altitud	33.80
Magnoliopsida	Gentianales	*Gentianella jamesonii*	En	4	0.979	0.402	Altitud	91.60
Magnoliopsida	Gentianales	*Gentianella longibarbata*	En	6	1	0.539	Altitud	49.70
Magnoliopsida	Gentianales	*Halenia serpyllifolia*	En	4	0.99	0.676	Altitud	51.30
Magnoliopsida	Geraniales	*Geranium chimborazense*	Vu	4	1	0.593	Altitud	56.40
Magnoliopsida	Geraniales	*Geranium ecuadoriense*	Vu	6	0.999	0.771	Altitud	84.00
Magnoliopsida	Geraniales	*Geranium loxense*	Vu	8	0.99	0.454	Altitud	31.90
Magnoliopsida	Geraniales	*Geranium sericeum*	Vu	6	0.998	0.365	Altitud	39.30
Magnoliopsida	Juglandales	*Juglans neotropica*	En	30	0.981	0.031	Altitud	39.50
Magnoliopsida	Juglandales	*Juglans olanchana*	En	28	0.969	0.023	Bio4	23.70
Magnoliopsida	Lamiales	*Lepechinia mutica*	Vu	4	0.997	0.569	Bio4	43.00
Magnoliopsida	Lamiales	*Aegiphila monticola*	En	15	0.981	0.3	Bio1	31.70
Magnoliopsida	Lamiales	*Aegiphila*	Vu	146	0.893	0.035	Bio4	25.40

		panamensis						
Magnoliopsida	Lamiales	*Aegiphila schimpffii*	En	4	0.997	0.645	Bio3	26.90
Magnoliopsida	Lamiales	*Vitex cooperi*	En	33	0.963	0.052	Bio19	33.40
Magnoliopsida	Laurales	*Nectandra olida*	Vu	4	1	0.809	Bio15	72.30
Magnoliopsida	Laurales	*Nectandra ramonensis*	Vu	13	0.909	0.302	Bio4	26.10
Magnoliopsida	Laurales	*Ocotea benthamiana*	Vu	11	0.96	0.443	Bio4	26.60
Magnoliopsida	Laurales	*Ocotea rivularis*	Vu	12	1	0.459	Bio13	41.40
Magnoliopsida	Laurales	*Ocotea rotundata*	Vu	6	0.988	0.612	Altitud	40.60
Magnoliopsida	Laurales	*Persea obtusifolia*	Vu	13	0.944	0.167	Bio1	27.70
Magnoliopsida	Laurales	*Persea schiedeana*	Vu	35	0.854	0.022	Altitud	24.90
Magnoliopsida	Laurales	*Mollinedia ruae*	CR	15	0.994	0.079	Bio4	34.50
Magnoliopsida	Laurales	*Siparuna cascada*	Vu	14	0.989	0.299	Bio15	67.50
Magnoliopsida	Laurales	*Siparuna multiflora*	Vu	12	0.983	0.341	Bio18	44.70
Magnoliopsida	Laurales	*Siparuna palenquensis*	En	8	0.983	0.36	Bio18	64.80
Magnoliopsida	Lecythidales	*Couratari guianensis*	Vu	16	0.968	0.227	Bio19	41.20
Magnoliopsida	Lecythidales	*Couratari scottmorii*	Vu	9	0.997	0.557	Bio13	53.00
Magnoliopsida	Lecythidales	*Eschweilera rimbachii*	Vu	16	0.965	0.338	Bio18	55.20
Magnoliopsida	Lecythidales	*Eschweilera sclerophylla*	Vu	9	0.999	0.045	Bio19	66.90
Magnoliopsida	Lecythidales	*Grias multinervia*	Vu	9	0.993	0.466	Bio3	44.30
Magnoliopsida	Lecythidales	*Gustavia dodsonii*	En	12	0.921	0.241	Bio18	59.90
Magnoliopsida	Lecythidales	*Gustavia monocaulis*	En	5	0.979	0.585	Bio2	50.30
Magnoliopsida	Magnoliales	*Anaxagorea phaeocarpa*	En	16	0.989	0.104	Bio14	33.30
Magnoliopsida	Magnoliales	*Cymbopetalum torulosum*	Vu	15	0.884	0.106	Bio19	29.60
Magnoliopsida	Magnoliales	*Rollinia pachyantha*	En	6	0.756	0.386	Bio19	75.00
Magnoliopsida	Magnoliales	*Stenanona panamensis*	En	4	1	0.889	Bio15	47.80
Magnoliopsida	Malvales	*Bombacopsis quinata*	Vu	23	0.938	0.041	Bio19	23.30
Magnoliopsida	Malvales	*Huberodendron patinoi*	Vu	12	0.937	0.26	Bio4	34.80
Magnoliopsida	Malvales	*Matisia grandifolia*	En	7	0.99	0.497	Bio18	47.90
Magnoliopsida	Malvales	*Quararibea gomeziana*	En	5	1	0.912	Bio15	45.80
Magnoliopsida	Malvales	*Quararibea pterocalyx*	Vu	16	0.971	0.204	Bio2	42.50
Magnoliopsida	Myrtales	*Terminalia*	En	21	0.991	0.188	Bio19	35.90

		bucidoides						
Magnoliopsida	Myrtales	*Axinaea pauciflora*	Vu	7	0.986	0.516	Bio15	32.50
Magnoliopsida	Myrtales	*Axinaea sclerophylla*	Vu	15	0.995	0.363	Altitud	40.50
Magnoliopsida	Myrtales	*Axinaea sessilifolia*	En	5	0.976	0.511	Bio19	47.40
Magnoliopsida	Myrtales	*Blakea eriocalyx*	En	15	0.948	0.261	Bio4	44.50
Magnoliopsida	Myrtales	*Blakea hispida*	Vu	12	0.952	0.293	Bio15	85.70
Magnoliopsida	Myrtales	*Blakea oldemanii*	Vu	9	0.997	0.221	Bio4	62.90
Magnoliopsida	Myrtales	*Blakea rotundifolia*	Vu	15	0.945	0.055	Bio4	67.10
Magnoliopsida	Myrtales	*Brachyotum benthamianum*	Vu	15	0.999	0.454	Altitud	38.50
Magnoliopsida	Myrtales	*Brachyotum gleasonii*	Vu	6	0.983	0.552	Altitud	40.90
Magnoliopsida	Myrtales	*Brachyotum gracilescens*	Vu	17	0.991	0.102	Altitud	59.80
Magnoliopsida	Myrtales	*Brachyotum harlingii*	Vu	7	0.976	0.297	Bio15	32.80
Magnoliopsida	Myrtales	*Brachyotum jamesonii*	Vu	12	0.992	0.118	Altitud	37.80
Magnoliopsida	Myrtales	*Brachyotum johannes-julii*	Vu	6	0.996	0.617	Altitud	31.30
Magnoliopsida	Myrtales	*Brachyotum rugosum*	Vu	10	0.991	0.68	Bio19	19.80
Magnoliopsida	Myrtales	*Bucquetia nigritella*	Vu	4	0.989	0.548	Bio18	52.40
Magnoliopsida	Myrtales	*Graffenrieda harlingii*	Vu	16	0.991	0.401	Altitud	34.10
Magnoliopsida	Myrtales	*Meriania furvanthera*	Vu	5	0.995	0.558	Bio3	27.40
Magnoliopsida	Myrtales	*Miconia bolivarensis*	Vu	5	1	0.45	Bio4	48.10
Magnoliopsida	Myrtales	*Miconia caelata*	Vu	9	0.991	0.606	Bio13	21.40
Magnoliopsida	Myrtales	*Miconia cajanumana*	Vu	5	0.989	0.604	Altitud	31.80
Magnoliopsida	Myrtales	*Miconia dapsiliflora*	Vu	6	0.999	0.584	Bio4	75.00
Magnoliopsida	Myrtales	*Miconia dodsonii*	En	7	0.994	0.525	Altitud	31.30
Magnoliopsida	Myrtales	*Miconia explicita*	Vu	7	0.993	0.273	Bio18	57.40
Magnoliopsida	Myrtales	*Miconia ledifolia*	En	9	0.995	0.357	Altitud	48.00
Magnoliopsida	Myrtales	*Miconia pernettifolia*	Vu	9	0.996	0.408	Altitud	35.90
Magnoliopsida	Myrtales	*Miconia stenophylla*	Vu	7	0.992	0.437	Bio18	34.10
Magnoliopsida	Myrtales	*Miconia suborbicularis*	Vu	6	0.993	0.534	Altitud	40.20
Magnoliopsida	Myrtales	*Tibouchina gleasoniana*	Vu	9	0.991	0.444	Bio4	48.90
Magnoliopsida	Myrtales	*Triolena pedemontana*	Vu	12	0.994	0.065	Bio18	33.00
Magnoliopsida	Myrtales	*Eugenia*	En	16	0.868	0.402	Bio4	43.10

		salamensis						
Magnoliopsida	Myrtales	*Fuchsia harlingii*	Vu	7	0.994	0.541	Altitud	38.00
Magnoliopsida	Piperales	*Hedyosmum purpurascens*	Vu	8	0.994	0.493	Altitud	30.00
Magnoliopsida	Piperales	*Peperomia persulcata*	Vu	8	0.96	0.11	Altitud	31.90
Magnoliopsida	Piperales	*Piper sodiroi*	Vu	7	0.999	0.421	Bio4	58.30
Magnoliopsida	Polygalales	*Spachea correae*	Vu	12	0.977	0.276	Altitud	21.70
Magnoliopsida	Polygalales	*Monnina equatoriensis*	Vu	11	0.994	0.55	Altitud	45.80
Magnoliopsida	Polygalales	*Monnina loxensis*	Vu	12	0.999	0.399	Bio4	25.20
Magnoliopsida	Proteales	*Euplassa occidentalis*	Vu	7	0.98	0.455	Bio15	77.60
Magnoliopsida	Rosales	*Licania longicuspidata*	En	5	0.996	0.395	Bio15	26.00
Magnoliopsida	Rosales	*Weinmannia loxensis*	Vu	6	0.984	0.517	Bio3	24.40
Magnoliopsida	Rosales	*Ribes lehmannii*	Vu	16	0.995	0.131	Altitud	53.40
Magnoliopsida	Rosales	*Lachemilla angustata*	Vu	7	0.991	0.372	Altitud	49.70
Magnoliopsida	Rosales	*Lachemilla rupestris*	Vu	8	0.997	0.44	Altitud	50.00
Magnoliopsida	Rosales	*Lachemilla sprucei*	Vu	6	0.999	0.355	Altitud	87.70
Magnoliopsida	Rosales	*Polylepis incana*	Vu	39	0.993	0.016	Altitud	61.80
Magnoliopsida	Rosales	*Polylepis lanuginosa*	Vu	12	0.878	0.26	Altitud	35.40
Magnoliopsida	Rosales	*Polylepis pauta*	Vu	18	0.999	0.304	Altitud	53.80
Magnoliopsida	Rosales	*Polylepis reticulata*	Vu	27	0.876	0.115	Altitud	69.10
Magnoliopsida	Rosales	*Polylepis weberbaueri*	Vu	9	0.991	0.046	Altitud	40.00
Magnoliopsida	Rosales	*Rubus laegaardii*	Vu	9	0.987	0.49	Altitud	55.00
Magnoliopsida	Rubiales	*Elaeagia pastoensis*	Vu	19	0.854	0.156	Bio19	53.40
Magnoliopsida	Rubiales	*Palicourea calothyrsus*	Vu	16	0.995	0.027	Bio4	44.40
Magnoliopsida	Rubiales	*Palicourea canarina*	Vu	9	0.954	0.234	Bio15	68.60
Magnoliopsida	Rubiales	*Palicourea corniculata*	Vu	7	0.995	0.217	Bio15	40.70
Magnoliopsida	Rubiales	*Palicourea jaramilloi*	Vu	9	0.945	0.214	Altitud	49.80
Magnoliopsida	Rubiales	*Palicourea lobbii*	Vu	9	0.991	0.544	Altitud	53.10
Magnoliopsida	Rubiales	*Palicourea prodiga*	Vu	6	0.997	0.612	Bio15	32.00
Magnoliopsida	Rubiales	*Palicourea sodiroi*	Vu	9	0.998	0.458	Bio4	70.10
Magnoliopsida	Rubiales	*Psychotria fusiformis*	Vu	12	0.977	0.286	Bio15	91.30
Magnoliopsida	Rubiales	*Stilpnophyllum grandifolium*	En	12	0.998	0.022	Bio15	74.10
Magnoliopsida	Rubiales	*Tocoyena pittieri*	Vu	27	0.909	0.127	Bio19	34.80

Magnoliopsida	Santalales	*Dendrophthora dalstroemii*	En	4	0.995	0.518	Altitud	24.10
Magnoliopsida	Sapindales	*Protium pittieri*	Vu	30	0.974	0.064	Bio19	28.10
Magnoliopsida	Sapindales	*Cedrela fissilis*	En	12	0.986	0.251	Bio6	26.20
Magnoliopsida	Sapindales	*Cedrela odorata*	Vu	140	0.803	0.067	Bio4	27.20
Magnoliopsida	Sapindales	*Guarea cartaguenya*	Vu	13	0.971	0.304	Bio18	33.30
Magnoliopsida	Sapindales	*Ruagea microphylla*	En	6	0.999	0.633	Bio4	26.40
Magnoliopsida	Sapindales	*Swietenia humilis*	Vu	54	0.952	0.036	Bio14	33.60
Magnoliopsida	Sapindales	*Swietenia macrophylla*	Vu	43	0.806	0.077	Bio4	25.80
Magnoliopsida	Sapindales	*Zanthoxylum panamense*	En	24	0.945	0.086	Bio12	27.50
Magnoliopsida	Sapindales	*Guaiacum sanctum*	En	69	0.916	0.001	Bio14	29.30
Magnoliopsida	Scrophulariales	*Aphelandra attenuata*	Vu	4	0.985	0.635	Bio18	64.80
Magnoliopsida	Scrophulariales	*Amphitecna isthmica*	Vu	20	0.957	0.243	Bio19	32.60
Magnoliopsida	Scrophulariales	*Alloplectus martinianus*	Vu	6	0.962	0.36	Bio15	37.10
Magnoliopsida	Scrophulariales	*Columnea albiflora*	Vu	12	0.974	0.396	Bio15	46.00
Magnoliopsida	Scrophulariales	*Columnea capillosa*	Vu	4	0.992	0.405	Bio15	63.20
Magnoliopsida	Scrophulariales	*Columnea eubracteata*	Vu	12	0.998	0.284	Bio4	47.10
Magnoliopsida	Scrophulariales	*Columnea mastersonii*	Vu	16	0.996	0.302	Bio4	47.80
Magnoliopsida	Scrophulariales	*Columnea ovatifolia*	Vu	6	1	0.543	Bio4	61.60
Magnoliopsida	Scrophulariales	*Corytoplectus cutucuensis*	En	6	0.985	0.173	Bio15	77.40
Magnoliopsida	Scrophulariales	*Gasteranthus lateralis*	Vu	14	0.946	0.378	Bio4	48.30
Magnoliopsida	Scrophulariales	*Gasteranthus ternatus*	En	4	0.999	0.733	Bio4	48.10
Magnoliopsida	Scrophulariales	*Monopyle sodiroana*	En	16	0.988	0.3	Bio18	59.70
Magnoliopsida	Scrophulariales	*Bartsia alba*	Vu	6	0.999	0.438	Altitud	77.30
Magnoliopsida	Scrophulariales	*Bartsia pumila*	Vu	4	1	0.528	Altitud	41.20
Magnoliopsida	Scrophulariales	*Calceolaria adenanthera*	Vu	6	0.992	0.293	Altitud	57.40
Magnoliopsida	Scrophulariales	*Calceolaria dilatata*	Vu	12	0.89	0.075	Bio4	38.60
Magnoliopsida	Scrophulariales	*Calceolaria gossypina*	En	6	0.997	0.555	Altitud	95.50
Magnoliopsida	Scrophulariales	*Calceolaria oxyphylla*	Vu	7	0.992	0.379	Altitud	33.60
Magnoliopsida	Scrophulariales	*Calceolaria pedunculata*	Vu	12	0.996	0.059	Altitud	29.40
Magnoliopsida	Scrophulariales	*Calceolaria*	Vu	5	0.999	0.502	Bio19	44.90

		serrata						
Magnoliopsida	Scrophulariales	*Calceolaria spruceana*	Vu	7	0.977	0.311	Altitud	82.10
Magnoliopsida	Scrophulariales	*Calceolaria stricta*	Vu	6	0.995	0.449	Bio4	33.80
Magnoliopsida	Solanales	*Iochroma lehmannii*	Vu	4	0.999	0.497	Bio19	27.40
Magnoliopsida	Solanales	*Solanum dolichorhachis*	Cr	4	0.987	0.312	Bio18	60.70
Magnoliopsida	Solanales	*Solanum interandinum*	Vu	20	0.99	0.205	Altitud	50.80
Magnoliopsida	Solanales	*Solanum leiophyllum*	Vu	4	0.971	0.523	Altitud	67.30
Magnoliopsida	Theales	*Caryocar costaricense*	Vu	14	0.76	0.218	Bio13	27.70
Magnoliopsida	Theales	*Clusia croatii*	Vu	81	0.972	0.067	Bio4	25.70
Magnoliopsida	Theales	*Clusia osseocarpa*	Vu	10	0.876	0.095	Bio2	40.80
Magnoliopsida	Theales	*Lacunaria panamensis*	En	18	0.997	0.308	Bio3	25.00
Magnoliopsida	Theales	*Freziera minima*	Vu	6	0.963	0.593	Altitud	33.80
Magnoliopsida	Urticales	*Cecropia longipes*	En	10	0.993	0.295	Bio6	43.20
Magnoliopsida	Urticales	*Cecropia maxima*	Vu	5	0.998	0.623	Bio4	67.00
Magnoliopsida	Violales	*Casearia mexiae*	En	5	0.985	0.513	Altitud	50.00
Magnoliopsida	Violales	*Passiflora jamesonii*	Vu	8	0.984	0.143	Altitud	77.20
Magnoliopsida	Violales	*Passiflora loxensis*	En	7	0.99	0.464	Altitud	41.90
Magnoliopsida	Violales	*Passiflora roseorum*	Vu	8	0.996	0.443	Altitud	44.40

Cuadro suplementario CS2: Clasificación de uso del suelo utilizado en el análisis según clasificaciones de tipo de vegetación en Global Land Cover 2000.

Clasificación Global Land Cover 2000	Clasificación usada en el análisis
Agriculture – intensive	áreas trasformadas
Bamboo dominated forest	vegetación natural
Barren / bare soil	vegetación natural
Burnt area (resent burnt area)	áreas transformadas
Closed deciduous forest	vegetación natural
Closed evergreen tropical forest	vegetación natural
Closed montane grasslands	áreas transformadas
Closed semi deciduous forest	vegetación natural
Closed semi-humid forest	vegetación natural
Closed shrublands	áreas transformadas
Closed steppe grasslands	áreas transformadas
Consolidated Rock Sparse Vegetation	vegetación natural
Cropland	áreas transformadas
Cropland and Shrubland/woodland	áreas transformadas
Desert	vegetación natural
Forest plantations (Llanos of Venezuela)	áreas transformadas

Fresh water flooded forests	vegetación natural
Grass Savannah	vegetación natural
Herbaceous Wetlands	vegetación natural
Mangroves	vegetación natural
Montane forests > 1000m - open semi humid	vegetación natural
Montane forests >1000m - closed semi -deciduous	vegetación natural
Montane forests >1000m - open deciduous	vegetación natural
Montane forests >1000m - open evergreen	vegetación natural
Montane forests >1000m - open semi- deciduous	vegetación natural
Montane forests >1000m - transition forest	vegetación natural
Montane forests > 1000m - closed semi humid	vegetación natural
Montane forests > 1000m flooded forest	vegetación natural
Montane forests > 1000m flooded forest	vegetación natural
Montane forests > 1000m flooded forest	vegetación natural
Montane forests >1000m - bamboo dominated	vegetación natural
Montane forests >1000m - closed deciduous	vegetación natural
Montane forests >1000m - closed temperate deciduous	vegetación natural
Montane forests >1000m - dense evergreen	vegetación natural
Montane forests >1000m - open temperate deciduous	vegetación natural
Montane forests >1000m - temperate mixed	vegetación natural
Montane forests >1000m -temperate closed broadleaf	vegetación natural
Montane forests 500-1000 - closed semi humid	vegetación natural
Montane forests 500-1000 - open semi humid	vegetación natural
Montane forests 500-1000 - bamboo	vegetación natural
Montane forests 500-1000 - dense evergreen	vegetación natural
Montane forests 500-1000 - open evergreen	vegetación natural
Montane forests 500-1000m - flooded forest	vegetación natural
Montane forests 500-1000m - closed deciduous	vegetación natural
Montane forests 500-1000m - closed semi -deciduous	vegetación natural
Montane forests 500-1000m - closed temperate deciduous	vegetación natural
Montane forests 500-1000m - flooded forest	vegetación natural
Montane forests 500-1000m - flooded forest	vegetación natural
Montane forests 500-1000m - open deciduous	vegetación natural
Montane forests 500-1000m - open semi- deciduous	vegetación natural
Montane forests 500-1000m - open temperate deciduous	vegetación natural
Montane forests 500-1000m - temperate mixed	vegetación natural
Montane forests 500-1000m - transition forest	vegetación natural
Montane forests 500-1000m -temperate closed broadleaf	vegetación natural
Mosaic agriculture / degraded forests	áreas transformadas
Mosaic agriculture / degraded vegetation	áreas transformadas
Open deciduous forest	vegetación natural
Open evergreen tropical forest	vegetación natural
Open montane grasslands	vegetación natural
Open semi deciduous forest	vegetación natural
Open semi-humid forest	vegetación natural
Open shrublands	áreas transformadas
Open steppe grasslands	vegetación natural
Periodically flooded savannah	áreas transformadas
Permanent swamp forests	vegetación natural
Polar Grassland with a Dwarf-Sparse Shrub Layer	vegetación natural
Polar Grassland with a Sparse Shrub Layer	vegetación natural
Semi deciduous transition forest	vegetación natural
Shrub Savannah	vegetación natural

Snow and Ice	vegetación natural
Sparse desertic steppe shrub /grasslands	vegetación natural
Subpolar Needleleaved Evergreen Forest Open Canopy - lichen understory	vegetación natural
Temperate closed deciduous broadleaf forests	vegetación natural
Temperate closed evergreen broadleaf forest	vegetación natural
Temperate mixed evergreen broadleaf forests	vegetación natural
Temperate open deciduous broadleaf forests	vegetación natural
Temperate or Sub-polar Broadleaved Deciduous Forest - Closed Canopy	vegetación natural
Temperate or Subpolar Broadleaved Deciduous Shrubland - Open Canopy	vegetación natural
Temperate or Subpolar Broadleaved Evergreen Shrubland - Closed Canopy	vegetación natural
Temperate or Subpolar Grassland	vegetación natural
Temperate or Subpolar Grassland with a Sparse Shrub Layer	vegetación natural
Temperate or Subpolar Grassland with a Sparse Tree Layer	vegetación natural
Temperate or Sub-polar Mixed Broaddleleaved or Needleleaved Forest - Open Canopy	vegetación natural
Temperate or Sub-polar Mixed Broadleaved and Needleleaved Dwarf-Shrubland - Open Canopy	vegetación natural
Temperate or Sub-polar Mixed Broadleaved or Needleleaved Forest - Closed Canopy	vegetación natural
Temperate or Sub-polar Needleleaved Evergreen Forest - Closed Canopy	vegetación natural
Temperate or Sub-polar Needleleaved Evergreen Forest - Open Canopy	vegetación natural
Temperate or Subpolar Needleleaved Evergreen Shrubland - Open Canopy	vegetación natural
Temperate or Sub-polar Needleleaved Mixed Forest - Closed Canopy	vegetación natural
Tropical or Sub-tropical Broadleaved Deciduous Forest - Closed Canopy	vegetación natural
Tropical or Sub-tropical Broadleaved Evergreen Forest - Closed Canopy	vegetación natural
Tropical or Sub-tropical Broadleaved Evergreen Forest - Open Canopy	vegetación natural
Unconsolidated Material Sparse Vegetation (old burnt or other disturbance)	áreas transformadas
Urban	áreas transformadas
Urban and Built-up	áreas transformadas
Water bodies	vegetación natural
Wetlands	vegetación natural

CAPITULO IV

DIVERSIDAD AMBIENTAL DE LAS ECOREGIONES DE MESOAMÉRICA, CHOCÓ Y LOS ANDES TROPICALES. IMPLICACIONES PARA SU CONSERVACIÓN.

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Environmental Heterogeneity of WWF Ecoregions and Implications for Conservation in Neotropical Biodiversity Hotspots

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Environmental heterogeneity of WWF ecoregions and implications for conservation in Neotropical biodiversity hotspots

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SUMMARY

Mesoamerica, Chocó and the tropical Andes are recognised as biodiversity hotspots where conservation action is urgently needed. Because World Wildlife Fund for Nature (WWF) ecoregions are commonly used as the basis for conservation decisions, an understanding of WWF ecoregions' environmental heterogeneity and their representation in current protected areas (PAs) is important for identifying priority areas for conservation. We developed 13 environmental domain classifications based on 22 climatic and topographical variables and used the Shannon diversity index to quantify environmental diversity for each ecoregion. To estimate representation in the PAs, we overlapped the area of each environmental domain and ecoregion with the World Database on Protected Areas 2007. The most environmentally-diverse ecoregions were poorly represented in the PAs and several ecoregions showed low environmental heterogeneity representation inside the PAs, such as the Balsas depression, Sierra Madre del Sur and the Chiapas Sierras in Mexico, some sierras in Central America, the Middle Magdalena, inter-Andean valleys, the Eastern Cordillera of Colombia, and the Western Moist Forest of Ecuador. Using WWF ecoregions as equivalent units for conservation and management can be misleading, given their environmental heterogeneity; therefore, their use for assessing environmental representation in PAs is limited. An underestimation of environmental heterogeneity representation in PAs can have misleading implications for conservation actions in regions where detailed biological information is lacking. Our study suggests that conservation efforts should focus on the environmental domains and ecoregions showing high environmental heterogeneity that is poorly represented in PAs.

Keywords: Conservation, environmental classifications, environmental domains, Neotropical biodiversity hotspots, protected areas, WWF ecoregions.

INTRODUCTION

Mesoamerica, the Chocó region and the tropical Andes are recognised as Neotropical biodiversity hotspots with exceptional species richness and endemism (N. Myers *et al.* 2000). However, these areas have been deforested at the rate of 1%, in which less than 25% of natural habitat currently remains. This loss of habitat poses a threat to biodiversity, and conservation actions are urgently needed (R. Dirzo & P.H. Raven 2003; World Bank 2007).

Current protected areas (PA) are a cornerstone of conservation strategies ideally representing and efficiently conserving regional biodiversity (C.R. Margules & R.L. Pressey 2000). However, PAs appear to poorly represent the biodiversity of the Mesoamerica, the tropical Andes and Chocó regions because many regional PAs were decreed using non-biological criteria, such as scenic value, political context and inaccessibility (S.J. Andelman & M.R. Willig 2003; D. Armenteras *et al.* 2003; E.M. Cue-Bar *et al.* 2006; T. Fuller *et al.* 2006). These *ad hoc* PAs usually fail to adequately represent regional biodiversity, which is a necessary step for sound systematic conservation planning (C.R. Margules & R.L. Pressey 2000).

It is well known that habitat heterogeneity and environmental heterogeneity correlate with species richness (F. Klijn & H.A.U. Haes 1994; M.L. Rosenzweig 1995). Hierarchical modelling relating different ecosystem components shows how species diversity depends on higher components, such as climate (R.G. Bailey 1985; R.G. Bailey

1987; F. Klijn & H.A.U. Haes 1994; M.M. Yarrow & S.N. Salthe 2008). Climatic and topographic variables are suitable for developing environmental classifications for a particular region. Environmental classifications are based on the identification of environmental domains; each domain is an n-dimensional space defined by specific values of environmental variables, such as climate and topography, where biological processes presumably occur. If distinct environmental conditions produce different environmental domains representing different regional species diversity, then, environmental factors can be used as surrogates for biodiversity (B.G. Mackey *et al.* 1988; L. Belbin 1993; A. Trakhtenbrot & R. Kadmon 2005). This can be particularly useful given the lack of complete and detailed information on the distribution of most taxonomic groups and the increasing availability of detailed quantitative data for environmental conditions (D.P. Faith & P.A. Walker 1996; R.L. Pressey *et al.* 2000; I. Oliver *et al.* 2004; S. Sarkar *et al.* 2005). Moreover, environmental domain classifications could be efficiently used in conservation planning (A. Trakhtenbrot & R. Kadmon 2005). The representation of different environmental domains in PAs will provide suggestions for successful conservation programs urgently needed in this region. Increasing the representation of the various environmental domains in conservation areas leads to an increase in habitat heterogeneity and presumably to species diversity representation.

Environmental heterogeneity within a region can be measured by classifications based on a cluster analysis of environmental variables. This quantitative approach provides some advantages over qualitative approaches for ecosystem-based classifications; subjective elements in cluster classifications such as variables used, weight given to the different variables and number of clusters, are maintain equal along the study area. These factors result in greater consistency and spatial accuracy in the classification progress (A.E. Lugo *et al.* 1999; J.R. Leathwick *et al.* 2003). The high resolutions and statistical approaches used in the production of environmental domain classifications ensure that the classified domains are unambiguously determined by specific variables, which enables the classification to be reproducible and objective (M.J. Metzger *et al.* 2005). Inversely, qualitative approaches of ecosystem-based classifications, such as the WWF ecoregions, are determined using a variety of ecoregion classification systems. As a result, the importance of each factor in determining the character of the ecosystem varies from place to place since the factor has not been statistically defined and depends on the experience and judgement of the originators (Metzger *et al*. 2005). In conservation planning, the use of the same variables and parameters across the study area allows for objective comparison between sites, which results in more transparent conservation decisions (C.R. Margules & S. Sarkar 2007). Quantitative classifications allow this kind of specific comparison.

WWF ecoregion classification (D.M. Olson *et al.* 2001) has been used for global and regional conservation prioritization and assessment (E. Dinerstein *et al.* 1995; R.D. Loyola *et al.* 2007). WWF ecoregions should represent distinct environmental domains, and conservation areas within these regions should represent environmental diversity. In this study, we provide an evaluation of the ecoregion's environmental heterogeneity based on a survey of the representation of different environmental domains and an evaluation of their representation in PAs. A set of environmental classifications are presented for Mesoamerica, Chocó and the tropical Andes to (1) represent their environmental heterogeneity, (2) assess the diversity of environmental domains within the WWF ecoregions and the representation of environmental domains and the WWF ecoregions in

PAs, and (3) select critical areas for conservation by identifying regions with environmental characteristics poorly represented in PAs.

METHODS

The study region was defined using the ecoregional classification of Olson *et al*. (2001), which included 53 WWF ecoregions in Mexico, Belize, Guatemala, Honduras, El Salvador, Nicaragua, Costa Rica, Panama, Colombia and Ecuador (Figure 1). Marine habitats were excluded from the analyses, and the northern boundary was delimited by the Transvolcanic Belt of central Mexico. A transitional Nearctic-Neotropical biogeography region, including up to the Balsas depression, that has a greater influence of tropical Mesoamerican elements was also included (J.J. Morrone 2005). The southern boundary was delimited by the topographical transition where the Andes split into the three separate Cordillera ranges. Boundary ecoregions are listed from west to east in Table 1.

The environmental variables used included elevation, slope, aspect with 19 bioclimatic variables (Table 2) from WorldClim (R.J. Hijmans *et al.* 2005). The WorldClim bioclimatic variables are developed using monthly temperature and rainfall values, which represent annual trends, seasonality and extreme or limiting environmental factors. The elevation was obtained from the U.S. Geological Survey's Hydro-1K DEM data set (USGS 1998) and the slope and aspect were derived by using the Spatial Analyst extension of ArcMap 9.0. All environmental layers were resampled at a 0.02° × 0.02° resolution in ArcGIS for a total of 437906 cells with an average area of 4.825 km^2 (SD = 0.105; max = 4.946; min = 4.597).

We used the PATN v.3.11 software (L. Belbin 1989) to develop the environmental domain classifications. The input data was a matrix of 437,906 rows and 22 columns, representing the values of each environmental variable (22 variables) within the study region (437906 cells). The ALOC algorithm (L. Belbin 1987), designed for large data sets, was used to construct a non-hierarchical classification that organised cells, based on an environmental distance measurement, into a number of predetermined groups or domains. All variables were equally weighted and the Gower metric distance measure was used to standardise the variables, which allowed for the combination of variables with different measurement units (P.H.A. Sneath & R.R. Sokal 1973). This method of classification has been widely used (J.R. Leathwick *et al.* 2003; A. Trakhtenbrot & R. Kadmon 2005; B.G. Mackey *et al.* 2008) to produce classifications unambiguously determined by specific variables, a process which makes the classification replicable and objective across the study area (M.J. Metzger *et al.* 2005).

Thirteen classifications were constructed starting with 53 groups, which equals the number of WWF ecoregions. The classification was increased by intervals of 50 groups until classification reached 653 groups; this is the maximum number of groups given the number of objects. To compare the correspondence between each of the 13 classifications and the WWF ecoregions, we used a tree diagram of similarity clusters for each classification based on the relationship between the groups within each cluster. This was completed by calculating the average value for each environmental variable across the cells composing each group and thereafter organizing the groups within each classification using the Euclidian distance and UPGA classification methods ("unweighted pair-group average") with the Statistica 8 software (I. StatSoft 2007). A comparison between the

UPGA clusters of each classification and the WWF ecoregions was visualised by importing the classifications into ArcMap (Esri 2004). In the tree diagram, the two groups most closely related were highlighted in ArcMap and then visually inspected to ensure that these groups were inside the geographical area of a specific WWF ecoregion. If the group was within the area, the next more similar group that conformed the cluster was selected and visually inspected again. This process was repeated to determine if the UPGA clusters were nested inside the WWF ecoregions.

The area occupied by each environmental domain in the WWF ecoregion was calculated for each classification using the projection “Equal Area Cylindrical” in ArcView extension Projector! This area was used as a multidimensional scaling in the Statistica 6 software to graphically observe if there was discrimination between ecoregions and if the degree of discrimination varied between classifications.

To evaluate the environmental diversity of the ecoregions and the environmental heterogeneity representation in PAs, we selected those classifications that were nested within the WWF ecoregions, and that could best discriminate between them. We calculated the Shannon diversity index as a measure of environmental or habitat diversity within each WWF ecoregion (sensu E.C. Pielou 1977) based on the area occupied by each environmental domain using the Species Diversity and Richness 3.02 software (P.A. Henderson & R.M. Seaby 2002). For statistical comparisons of diversity, we used the upper and lower 95% confidence intervals for each Shannon diversity value. The area represented in the PA was calculated for each selected environmental domain classification (EDC), and for the WWF ecoregions, by overlapping the datasets for the study region and the World Database on Protected Areas 2007.

RESULTS

Classifications with 300 environmental domains or more were nested into WWF ecoregions following visual comparisons. Conversely, classifications with less than 300 domains had the same domain extend across the study region but were not related to a particular WWF ecoregion. Clusters produced by classifications with more than 300 domains showed greater spatial relation with WWF ecoregions because the environmental domains were more aggregated and related to a specific geographical area. The results of visual comparisons were supported by multidimensional scaling, and more ecoregions were discriminated by classifications containing 300 domains or more (see Appendix S1 and S2 at http://www.ncl.ac.uk/icef/EC.Supplement.htm). All multidimensional scaling analyses adequately fit to the Sheppard curve, indicating that two dimensions were sufficient to describe the data. From the 53 WWF ecoregions, 17 were discriminated by the multidimensional analysis.

Given that there was no unique classification to improve discrimination between the WWF ecoregions, classifications of 349 and 541 environmental domains were selected for further analysis. The classification of 53 environmental domains was selected for a direct comparison with the 53 WWF ecoregions (Figure 2A-C). These criteria provided a range of EDC for evaluating our results. We found no significant differences in the environmental diversity between WWF ecoregions based on the three selected EDCs. The most environmentally-diverse ecoregions were the northwestern Andean montane forest, the Magdalena Valley montane forest, the Eastern Cordillera real montane forest, the Cordillera Oriental montane forest and the Central American pine-oak forest. Conversely, ecoregions showing the least environmental diversity were Cordillera Central páramo, Lake, the Miskito pine forest, the northern Mesoamerica Pacific mangroves, the Santa Marta páramo and the Veracruz dry forest (Table 3). Overall, Shannon diversity values increased as the number of environmental domains increased with the exception of six ecoregions (Figure 3, Table 3). Sierra de los Tuxtlas and Lake showed significantly greater Shannon diversity values (95% confidence intervals) for the 349 classification than for the 541 classification. Chimalapas montane forest, Cordillera Central páramo, Catatumbo moist forests and Veracruz dry forest showed no statistical difference in the Shannon diversity values (95% confidence intervals) between the 349 and 541 classifications (Table 3).

Few WWF ecoregions or environmental domains from the three EDCs were well represented in the PA. This resulted in a logarithmic distribution when evaluating the representation of each ecoregion or environmental domain in the PAs (Figure 4). The maximum percentage of representation in the 53 EDC was 65%. Conversely, the maximum percentage of representation in the WWF ecoregions and in the 349 and 541 EDC ranged between 98% and 100% (Figure 4). Several ecoregions were consistently identified as having a low PA representation, such as the Balsas depression, Sierra Madre del Sur and Chiapas mountains in Mexico, Central American mountain chains, Middle Magdalena, inter-Andean valleys, the Eastern Cordillera in Colombia, and Western Moist Forest in Ecuador (Figure 4, see Appendix S3 at http://www.ncl.ac.uk/icef/EC.Supplement.htm). The most environmentally-diverse WWF ecoregions were poorly represented in the PA (Table 3).

WWF ecoregions and environmental classification varied in PA representation. A high representation of an ecoregion in PA did not ensure appropriate representation of its environmental diversity. For example, 27.05% of Peten Veracruz moist forest was represented in the PA, which primarily covered the eastern sector of the ecoregion, leaving the western sector underrepresented. This selection highlighted the different environmental conditions in these areas according to the environmental classifications. Similarly, the Guajira Barranquilla Xeric-Scrub region showed 35% inside the PA that was concentrated on the Guajira peninsula, which disregarded the area around Barranquilla with different environmental domains (Figure 5). Environmental domains of the 53, 349 and 541 EDC showing less than 1% in the PA were identified as environmental domains with low representation in the PA and high priorities for conservation. These regions represent unique environmental habitats but were not represented in the PA (Figure 5).

We identified three groups for improved representation of environmental diversity in the WWF ecoregions could be achieved: (1) low representation of environmental domains in high conservation priority WWF ecoregions, as recognised by Soutullo et al. (2008); (2) low representation of environmental domains in WWF ecoregions with <10% representation in the PA, and (3) low representation of environmental domains in WWF ecoregions with >10% representation in the PA. A value of 10% was set because this percentage is the minimum target representation for each WWF ecoregion established by the CBD (Convention on Biological Diversity 2002).

DISCUSSION

WWF ecoregions have been used as a global template for assessments of species diversity, endemism or vulnerability to identify conservation priority areas (T.M. Brooks *et al.* 2006). Additionally, local governments and international NGOs guide economic resources and conservation efforts to selected WWF ecoregions (N. Myers *et al.* 2000; D.M. Olson & E. Dinerstein 2002; S. Ferrier *et al.* 2004). Moreover, local governments signing the CBD usually consider WWF ecoregions as criteria that cause 10% of the world's ecological regions to occur in their countries.

At a global scale, WWF ecoregion assessments are useful as a broad context for evaluating regional scale conservation priorities (S. Ferrier *et al.* 2004). We assessed climatic and topographic diversity at a regional scale across the WWF ecoregions from the Neotropical hotspots Mesoamerica, tropical Andes and Choco regions by selecting environmental domain classifications that maximise the overlap with WWF ecoregions. We observed that classifications of more than 300 environmental domains reflected the general pattern of the 53 WWF ecoregions, which highlighted the complexity of environmental conditions in the region. Our results showed that 17 ecoregions (at the most) could be clearly distinguished by a set of environmental domains, while 36 ecoregions could not be statistically discriminated. Other studies have shown similar results, indicating that WWF ecoregion classifications can only be partially described using environmental and biological data (R.S. Thompson *et al.* 2004; R. McDonald *et al.* 2005). WWF ecoregions are defined as "relatively large units of land containing a distinct assemblage of natural communities and species with boundaries that approximate the original extent of natural communities prior to major land-use change" (D.M. Olson *et al.* 2001). WWF ecoregions have been determined using a variety of ecoregion classification systems. As a result, the importance of each factor in determining characteristics of ecosystems can vary from region to region and expert opinion because these characteristics have not been statistically defined (M.J. Metzger *et al.* 2005). Given the complexity of defining WWF ecoregions, it is difficult to predict correlations with environmental data (R.S. Thompson *et al.* 2004). Environmental diversity is not a factor explicitly represented in the WWF ecoregion classification, which results in some ecoregions having more environmental diversity than others. Our study showed that similar patterns of diversity were maintained in the WWF ecoregions under different classifications; this result reflects that regardless of the number of domains included, some WWF ecoregions were consistently more diverse than others. Thus, using WWF ecoregions as the equivalent for conservation and management units can be misleading.

WWF ecoregion classification can also be inappropriate for assessing environmental representation. We do not intend to underestimate the relevance of WWF ecoregions worldwide but rather caution about the possible shortcomings in application to conservation. WWF ecoregions provide robust biogeographic units reflecting the distribution of communities largely determined by historical processes, habitat type and large-scale ecological processes, which provides the critical spatial link between global priority setting efforts and site-based assessment (E. Wikramanayake *et al.* 2002).

We found that the environmental diversity of a WWF ecoregion was not always appropriately represented in a PA despite an overlap greater than 10% with some WWF ecoregions fulfilling the CBD conservation target. When large WWF ecoregions showed middle or high environmental diversity values, ecoregions representation in the PAs was not representative of the areas' environmental heterogeneity, such as in the Peten-Veracruz moist forest and Guajira-Barranquilla xeric scrub ecoregions. This study demonstrates that underestimation of environmental heterogeneity in the WWF ecoregions misleads representation of the PAs. Ferrier *et al*. (2004) noted that outcome is the result of the mismatch between the scale of global assessments and the scale at which conservation decisions occur. WWF ecoregions include extensive regions containing land of varying conservation value. There is a need to transition prioritization exercises from a global scale into actual conservation plans at the local level (A. Soutullo *et al.* 2008). Additionally, conservation targets should be set to meet adequate representation of regional and local biodiversity.

Setting targets for adequate representation of biodiversity does not imply that total target values must increase with increasing diversity values; rather, this suggests that targets are established in an efficient way so that they achieve a maximum representation of biological or environmental characteristics for the same target value (C.R. Margules & S. Sarkar 2007). By effectively allocating the 10% target within WWF ecoregions, better representation of biological or environmental diversity is achieved. In this sense, the selection of conservation sites by a systematic conservation planning process is appropriate to maximise representation of biodiversity in PAs (Margules & Pressey 2000).

Low environmental heterogeneity representation can have adverse implications for conservation, particularly in Neotropical countries where detailed biological information is still lacking (S. Ferrier 2002; K. Kim & L. Byrne 2006). Habitat heterogeneity is directly related with landscape diversity, suggesting that habitat selection and adaptation to local environmental conditions may be the primary processes structuring diversity among landscapes within the WWF ecoregions (J.A. Veech & T.O. Crist 2007). If an adequate representation of environmental domains in PAs is observed, we can have a greater confidence in an adequate representation of habitat heterogeneity and regional species diversity. Given the limited knowledge of detailed species distributions, the full range of environmental conditions should be represented in PAs (J.B. Kirkpatrick & M.J. Brown 1994).

To effectively include different environmental conditions in the PAs, selection include a classification of areas with similar ecosystem characteristics. Environmental domain classifications have been built with a standardised method for simplifying environmental data into strata, which allows for an objective classification across a study region based on environmental variables. The development of a quantitative classification of environmental conditions is the first step for producing tools to derive stratified samples (R.G.H. Bunce *et al.* 1996; M.J. Metzger *et al.* 2005). This process is crucial when large-scale continuous gradients are involved and statistical analysis provides robust divisions based on the balance between the variables building the database (M.J. Metzger *et al.* 2005).

This is the first study to use the environmental domain classification at this scale to analyse the complex environmental diversity in the Mesoamerican, Chocó and tropical Andean regions (N. Myers *et al.* 2000). Other successful environmental classifications have been conducted for Papua New Guinea (H.A. Nix *et al.* 2000), Australia (Mackey *et al.* 2008) New Zealand (J.R. Leathwick *et al.* 2003), South Africa and Lesotho (A. Bonn & K.J. Gaston 2005) for conducting surveys, monitoring and management of biodiversity. Assessments of environmental classifications have shown that some environmental domains have received little conservation attention. This poor representation has been attributed to some environmental domains being highly correlated with areas of intensive human activity, which implies that there is a high economic cost when acquiring land for conservation. Moreover, governmental policies for actual reserve designation have focused on wild landscapes made using historical *ad hoc* decisions (C.R. Margules & S. Sarkar 2007). The underrepresentation of some environmental domains will inevitably result in differential loss of biodiversity features restricted to these environmental domains (J.R. Leathwick *et al.* 2003).

By itself, environmental domain analysis is not an appropriate tool for the planning of reserves to encompass the biodiversity. Species distributions are not always symmetrical in response to environmental gradients and do not have equal probabilities of distributions along the environmental space. Consequently, rather than being spread evenly across the environmental space, complementary areas may cluster around particular regions of high species density (M.B. Araújo *et al.* 2003). Reserves selected on the basis of environmental domains are likely to represent more widespread biotic attributes in a satisfactory manner but will likely fail to include many rare and endemic species and communities (A. Bonn & K.J. Gaston 2005; A. Trakhtenbrot & R. Kadmon 2006).The environmental domains poorly represent location of species abundances, which will prevent the detection of potentially viable populations (A. Bonn & K.J. Gaston 2005). Lastly, environmental domain classifications indicate only the range of dynamic states expected at a site but not the particular phase occurring at a particular site at a given time (J.R. Leathwick *et al.* 2003).

The effectiveness of environmental surrogates may be enhanced by using community-level modelling techniques in conjunction with the best available biological distribution data to correlate the relationship between environmental gradients and community richness and composition (A. Arponen *et al.* 2008). Several methods propose combined analyses, such as the environmental diversity approach (ED) (D.P. Faith & P.A. Walker 1996; D.P. Faith 2003; D.P. Faith *et al.* 2004) and the generalised dissimilarity modelling (GDM) (S. Ferrier *et al.* 2007). However, it is unclear how quality and quantity of biological data influence the environmental community modelling approach (Arponen *et al*. 2008).

Given that species distributions are influenced by climate variable correlations and their relationships with biotic attributes (X. Morin & M.J. Lechowicz 2008), research is needed to enable a better understanding of variable correlations with biological data. This will allow selection of subjective characteristics of environmental classifications, such as variables, weights given to the variables and number of groups, to result in classifications that better represent biological composition, ecosystem drivers and ecosystem responses reflective of ecological and evolutionary processes (Mackey *et al.* 2008).

In our view, the decisions for selecting modelling approaches depend on environmental, biological, or community-level data availability, which will ensure that planning remains a dynamic process as better quality data become available. Incorporating other important environmental variables as they become available in better resolutions (such as soil type and topographical data), can better represent primary environmental regimens that drive biotic responses, such as heat, light, water and mineral nutrients. These advances will result in a classification that more accurately reflects the environmental domains that will better representing species diversity (Mackey *et al*. 2008). If we consider the limitations of using exclusively environmental information but note the novelty of an objective comparison using environmental data across biological complex regions, it is feasible to identify regions where unique environmental conditions are underrepresented in the PAs. Our study suggests that conservation efforts should focus on poorly represented environmental domains in the PAs of relevant WWF ecoregions, such as the Chocó-Darién moist forest, the Cordillera Oriental montane forest, the Eastern Cordillera real montane forest, the Magdalena Valley montane forest, the Central American pine-oak forest and the Sierra Madre del Sur pine-oak forest (A. Soutullo *et al.* 2008). Inclusion of poorly represented environmental domains must be additionally incorporated in conservation programs in Mesoamerica, Chocó and tropical Andes countries where ecoregions have environmental domains consistently underrepresented in the PAs. This includes the Balsas depression in Mexico, the Sierra Madre del Sur, the Chiapas Mountains ecoregions, the ecoregions across the mountain chains in Central America, the Middle Magdalena and the inter-Andean valleys ecoregions in Colombia and the Western Moist Forest in Ecuador.

A preliminary assessment of conservation priorities using threatened species distributions in the same study region showed similar results of underrepresented biological data in PAs and identified similar regions for biological conservation with exception of the Middle Magdalena in Colombia (S. Sarkar *et al.* 2009). Geographic areas such as the Mesoamerican and Andean mountain chains and the inter-Andean valley, identify by biodiversity and environmental modelling approaches as being areas with conservation value, are often associated with high human population densities in urban settlements. This situation impose many challenges to accurately determine conservation areas and requires integrative landscape management and innovative restoration perspectives to ensure biodiversity conservation (J. Ervin 2003; A.S.L. Rodrigues *et al.* 2004; T. Fuller *et al.* 2006; C.R. Margules & S. Sarkar 2007; K. Chan & G.C. Daily 2008; S. Sarkar *et al.* 2009).

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TABLES and FIGURES

Table 1: Delimiting study area boundaries of ecoregions.

Boundary	Ecoregions (from west to east)
Northern Boundary: Mexico Transvolcanic Belt	Balsas dry forests, Sierra Madre del Sur pine-oak forests, Oaxaca montane forests, Sierra Madre de Oaxaca, Veracruz dry forests, Tehuacán Valley scrubland, Mesoamerican Gulf-Caribbean mangroves, Sierra de los Tuxtlas, Petén-Veracruz moist forests, Pantanos de Centla, and Yucatan

	dry forests
Southern boundary: Andes split into three separate ranges: Cordillera Occidental, Cordillera Central and Cordillera Oriental	Western Ecuador moist forests, Northwestern Andean montane forests and Eastern Cordillera Real montane forests

Table 2. The 19 bioclimate variables used as attributes in the environmental domain classification.

Variable
mean annual temperature
mean diurnal range
isothermality
temperature seasonality
max temperature of warmest month
min temperature of coldest month
annual temperature range
mean temperature of wettest quarter
mean temperature of driest quarter
mean temperature of warmest quarter
mean temperature of coldest quarter

annual precipitation
precipitation of wettest month
precipitation of driest month
precipitation seasonality
precipitation of wettest quarter
precipitation of driest quarter
precipitation of warmest quarter
precipitation of coldest quarter

Table 3. Representation of the WWF ecoregions in protected areas (PAs) and the Shannon environmental diversity index (H) for the three selected environmental domain classifications (H_53; H_349; H_541). Values in brackets indicate lower and upper 95% confidence intervals of Shannon values.

	% in		H	
WWF ecoregion	PA	H_53	H_349	H_541
Amazon-Orinoco-Southern Caribbean mangroves	22.05	1.16 (1.11-1.20)	2.07 (2.03-2.1)	2.28 (2.24-2.32)
Balsas dry forests	4.15	1.41 (1.39-1.41)	2.65 (2.64-2.66)	3.00 (2.99-3.0)
Belizean pine forests	34.66	1.21 (1.17-1.25)	1.86 (1.82-1.89)	2.17 (2.13-2.2)
Catatumbo moist forests	12.14	1.70 (1.68-1.72)	2.08 (2.04-2.11)	2.08 (2.05-2.11)
Cauca Valley dry forests	0.00	1.04 (1.01-1.07)	2.14 (2.11-2.17)	2.44 (2.41-2.47)
Cauca Valley montane forests [1]	4.09	2.31 (2.3-2.32)	3.67 (3.66-3.68)	4.10 (4.09-4.10)

Central American Atlantic moist forests	28.09	1.95 (1.94-1.95)	3.26 (3.25-3.26)	3.52 (3.51-3.53)
Central American dry forests	5.27	1.75 (1.74-1.76)	2.84 (2.82-2.84)	3.24 (3.23-3.24)
Central American montane forests	39.42	2.36 (2.34-2.37)	3.37 (3.35-3.38)	3.59 (3.57-3.61)
Central American pine-oak forests [1, 2]	7.33	2.41 (2.39-2.41)	3.75 (3.74-3.75)	4.11 (4.10-4.11)
Chiapas Depression dry forests	1.73	1.56 (1.54-1.57)	2.12 (2.09-2.13)	2.41 (2.39-2.43)
Chiapas montane forests	4.40	1.87 (1.85-1.89)	2.44 (2.41-2.47)	2.81 (2.78-2.83)
Chimalapas montane forests	6.56	1.67 (1.63-1.69)	2.66 (2.62-2.69)	2.66 (2.61-2.69)
Choco-Darien moist forests [2]	19.52	2.20 (2.19-2.21)	3.38 (3.37-3.39)	3.70 (3.69-3.7)
Cordillera Central paramo	23.77	0.93 (0.87-0.98)	1.37 (1.25-1.46)	1.51 (1.38-1.60)
Cordillera Oriental montane forests [2]	18.95	2.92 (2.92-2.93)	4.17 (4.16-4.17)	4.47 (4.46-4.47)
Costa Rican seasonal moist forests	10.08	2.40 (2.38-2.42)	2.96 (2.94-2.98)	3.25 (3.22-3.26)
Eastern Cordillera real montane forests [2]	15.37	2.60 (2.59-2.61)	3.84 (3.83-3.84)	4.15(4.14-4.16)
Eastern Panamanian montane forests	77.09	2.00 (1.97-2.03)	2.51 (2.47-2.55)	2.65 (2.59-2.69)
Guajira-Barranquilla xeric scrub	33.65	0.40 (0.39-0.42)	1.81 (1.79-1.83)	2.19 (2.18-2.20)
Isthmian-Atlantic moist forests	27.67	2.26 (2.25-2.27)	3.24 (3.23-3.24)	3.56 (3.55-3.56)
Isthmian-Pacific moist forests	12.48	1.42 (1.40-1.43)	2.64 (2.62-2.65)	3.02 (3.00-3.03)
Lake	3.49	0.54 (0.52-0.57)	1.15 (1.12-1.16)	0.58 (0.56-0.60)
Magdalena Valley dry forests	0.04	1.77 (1.76-1.78)	2.58 (2.56-2.59)	2.70 (2.68-2.71)
Magdalena Valley montane forests [1, 2]	2.39	2.77 (2.76-2.77)	4.17 (4.16-4.17)	4.47 (4.46-4.47)
Magdalena-Urabo moist forests	1.40	1.55 (1.54-1.55)	2.50 (2.49-2.50)	2.83 (2.82-2.83)
Mesoamerican Gulf-Caribbean mangroves	52.94	2.06 (2.05-2.07)	3.12 (3.11-3.13)	3.35 (3.34-3.36)
Miskito pine forests	12.52	0.82 (0.81-0.83)	1.74 (1.73-1.75)	2.00 (1.99-2.01)
Motagua Valley thornscrub	18.80	2.13 (2.09-2.16)	2.34 (2.28-2.38)	2.75 (2.69-2.78)
Northern Andean paramo	31.66	1.33 (1.31-1.34)	3.11 (3.09-3.11)	3.52 (3.50-3.52)
Northern Mesoamerican Pacific mangroves	3.95	0.53 (0.46-0.59)	1.01 (0.94-1.06)	1.27 (1.18-1.35)
Northwestern Andean montane forests	14.53	2.95 (2.95-2.96)	4.43 (4.42-4.43)	4.79 (4.78-4.79)
Oaxacan montane forests	0.00	1.64 (1.62-1.66)	2.78 (2.75-2.80)	2.99 (2.96-3.01)
Panamanian dry forests	3.70	1.18 (1.15-1.19)	2.07 (2.03-2.09)	2.14 (2.10-2.16)
Pantanos de Centla	30.69	1.50 (1.49-1.50)	1.58 (1.56-1.59)	1.63 (1.61-1.64)
Patía Valley dry forests	0.00	1.16 (1.10-1.19)	2.38 (2.33-2.41)	2.70 (2.64-2.74)
Peten-Veracruz moist forests	27.50	2.04 (2.03-2.04)	3.49 (3.48-3.49)	3.80 (3.79-3.81)
Santa Marta montane forests	71.72	1.87 (1.83-1.89)	2.67 (2.64-2.69)	2.75 (2.73-2.77)
Santa Marta paramo	98.92	0.70 (0.65-0.75)	1.31 (1.26-1.34)	1.58 (1.53-1.61)
Sierra de los Tuxtlas	43.22	1.68 (1.64-1.71)	2.28 (2.24-2.31)	2.07 (2.03-2.09)
Sierra Madre de Chiapas moist forests	15.88	2.07 (2.05-2.08)	2.87 (2.84-2.89)	3.06 (3.03-3.08)
Sierra Madre de Oaxaca pine-oak forests [1]	5.23	2.35 (2.34-2.36)	3.64 (3.62-3.65)	3.91 (3.88-3.92)
Sierra Madre del Sur pine-oak forests [1, 2]	1.62	2.23 (2.22-2.23)	3.49 (3.48-3.49)	3.91 (3.9-3.91)
Sinú Valley dry forests	3.61	0.97 (0.95-0.99)	1.75 (1.73-1.77)	2.25 (2.23-2.27)

South American Pacific mangroves	15.50	2.38 (2.36-2.39)	2.99 (2.96-3.00)	3.09 (3.06-3.11)
Southern Mesoamerican Pacific mangroves	37.55	1.25 (1.21-1.28)	2.31 (2.28-2.32)	2.63 (2.61-2.65)
Southern Pacific dry forests	2.08	1.71 (1.69-1.73)	2.76 (2.74-2.77)	3.08 (3.07-3.09)
Talamancan montane forests	65.09	2.16 (2.14-2.17)	3.30 (3.29-3.31)	3.69 (3.67-3.70)
Tehuacan Valley matorral	16.13	0.91 (0.89-0.92)	2.07 (2.05-2.09)	2.31 (2.29-2.32)
Veracruz dry forests	0.01	0.62 (0.59-0.64)	1.30 (1.27-1.33)	1.31 (1.29-1.34)
Western Ecuador moist forests	4.53	1.44 (1.43-1.46)	2.43 (2.42-2.44)	2.77 (2.75-2.77)
Yucatan dry forests	1.67	0.99 (0.98-0.99)	1.83 (1.82-1.84)	2.07 (2.06-2.08)
Yucatan moist forests	16.42	1.12 (1.12-1.13)	2.15 (2.15-2.16)	2.30 (2.29-2.30)

[1] ecoregions with high environmental diversity and low representation in PAs

[2] high conservation priority ecoregions

Figure Legends:

Figure 1: Study region (light gray) delimited by the WWF ecoregions, including 10 countries from the Mesoamerica, Choco and tropical Andes regions. The current protected areas (PAs) according to the World Database on Protected Areas 2007 are depicted in dark gray.

Figure 2: Environmental domain classifications (EDCs) selected for further analyses. Each colour corresponds to a different domain in each classification. A) 53 EDC, B) 349 EDC, and C) 541 EDC.

Figure 3: Environmental diversity (Shannon index) for the WWF ecoregions based on the different classifications. Light colours represent low environmental diversity and dark colours represent high environmental diversity. (A) 53 EDC;, (B) 349 EDC, and (C) 541 EDC.

Figure 4: Percentage of representation in PAs for (A) dash line WWF ecoregions, continuous line 53 EDC, (B) dash line 349 EDC, continuous line 541 EDC. The curve shows the percentage of representation on the *y* axis and ecoregions or domains on the *x* axis. Logarithmic shapes of the curves indicate that few WWF ecoregions or environmental domains showed high percentages of representation in the PAs.

Figure 5: Environmental domains with less than 1% representation in the PAs (shaded areas), including the 53, 349 and 541 EDC, respectively. Environmental domains within high conservation priority ecoregions (compiled by Soutullo et al. 2008) are depicted in black. Environmental domains within ecoregions with less than 10% representation in the PAs are depicted in dark gray. Environmental domains within ecoregions with more than 10% representation in the PAs are depicted in light gray. Figures inside boxes illustrate examples of poorly representated environmental domains within two ecoregions of more than 10% representation in the PAs.

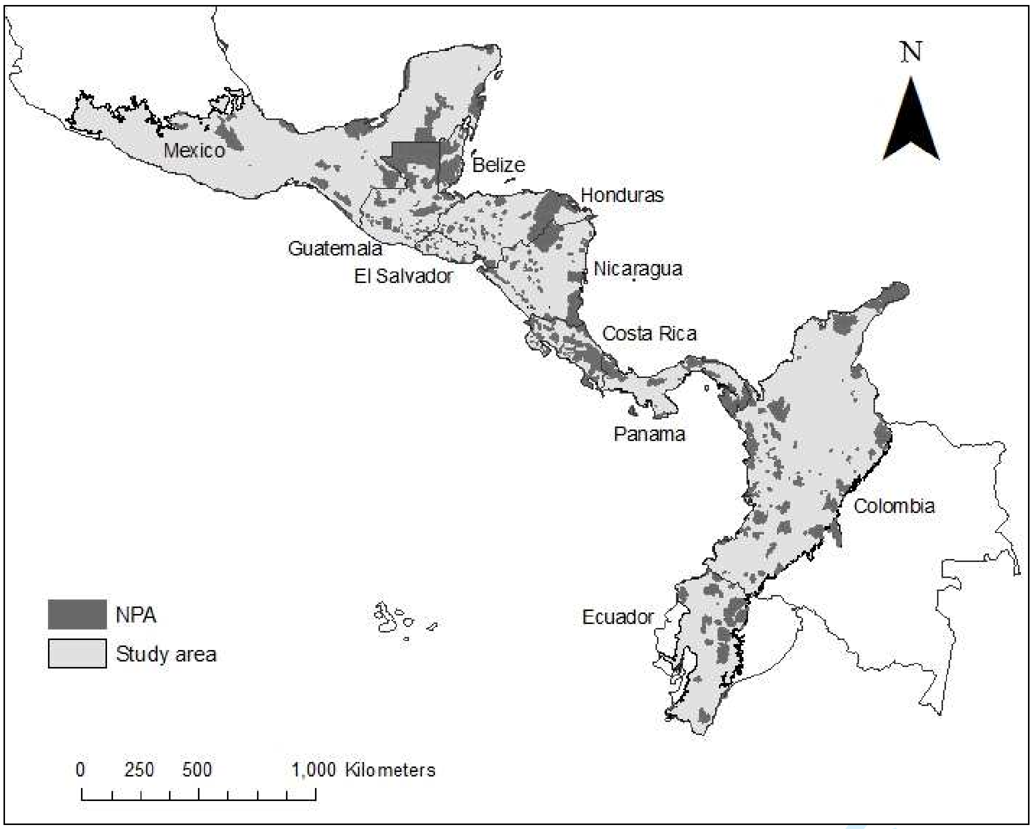

N
Mexico
Belize
Honduras
Guatemala
El Salvador
Nicaragua
Costa Rica
Panama
Colombia
Ecuador
NPA
Study area
0 250 500 1,000 Kilometers

Figure 1.

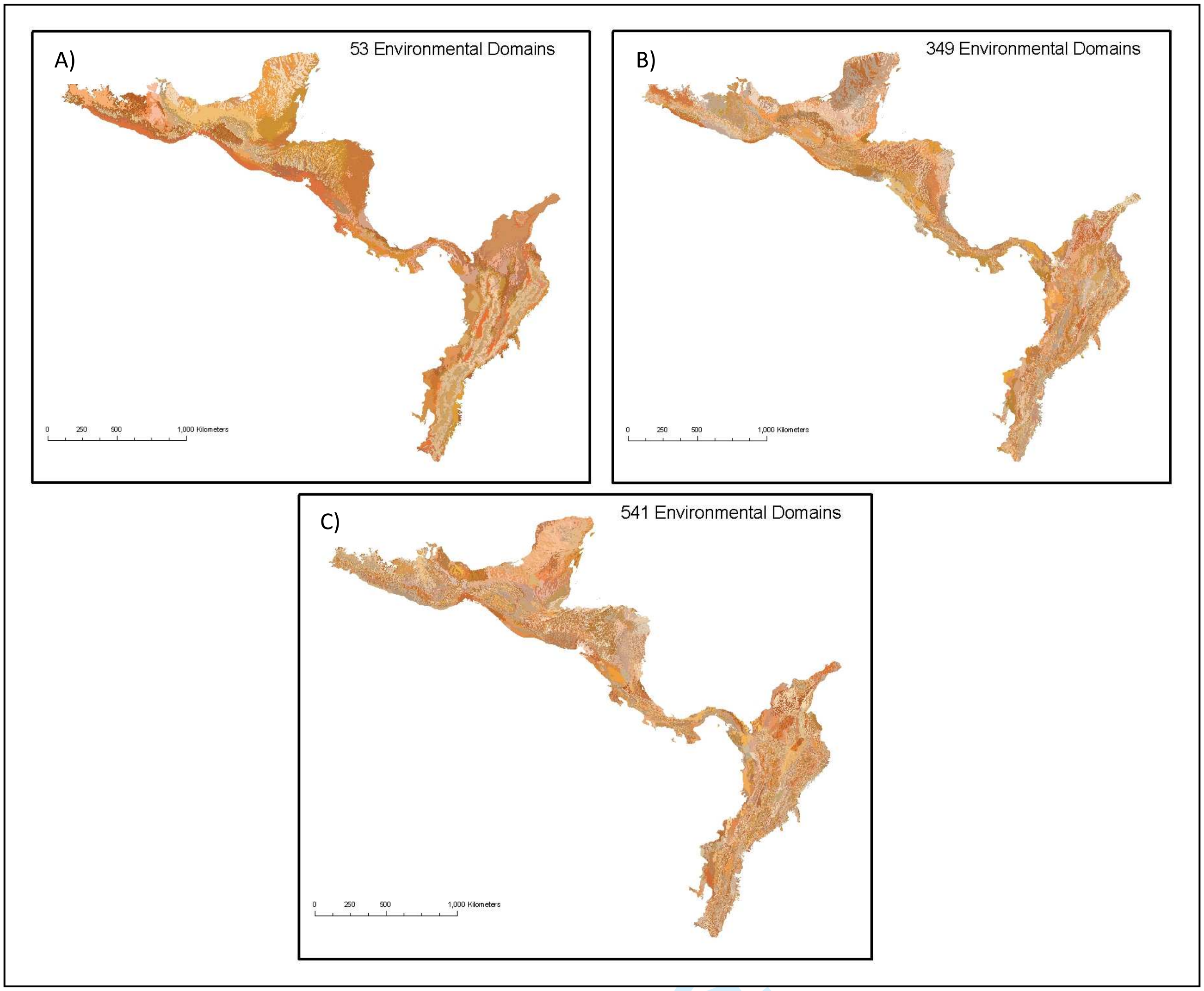

A)
53 Environmental Domains
0 250 500 1,000 Kilometers
B)
349 Environmental Domains
0 250 500 1,000 Kilometers
C)
541 Environmental Domains
0 250 500 1,000 Kilometers

Figure 2.

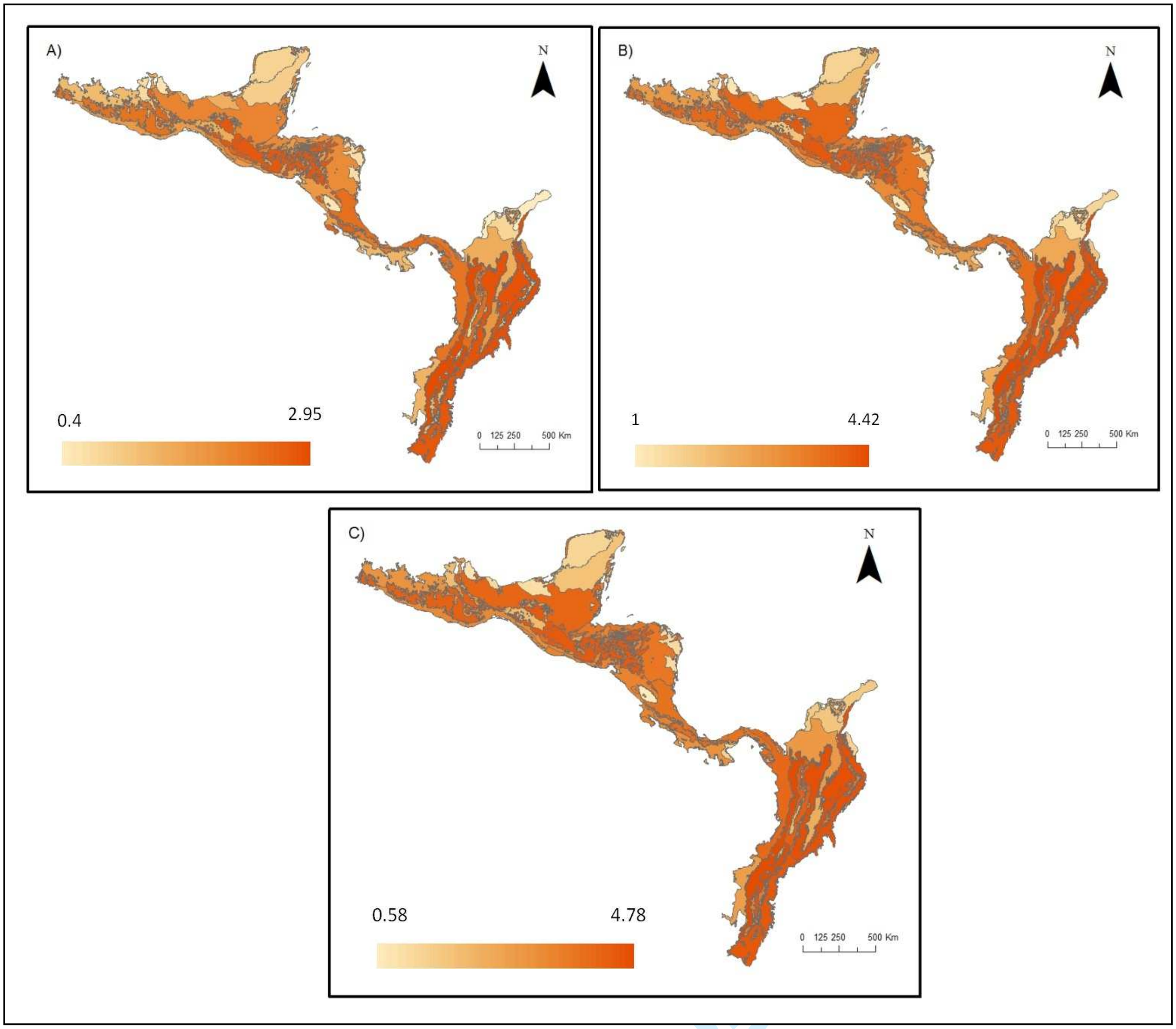

A)
N
0.4
2.95
0 125 250 500 Km
B)
N
1
4.42
0 125 250 500 Km
C)
N
0.58
4.78
0 125 250 500 Km

Figure 3.

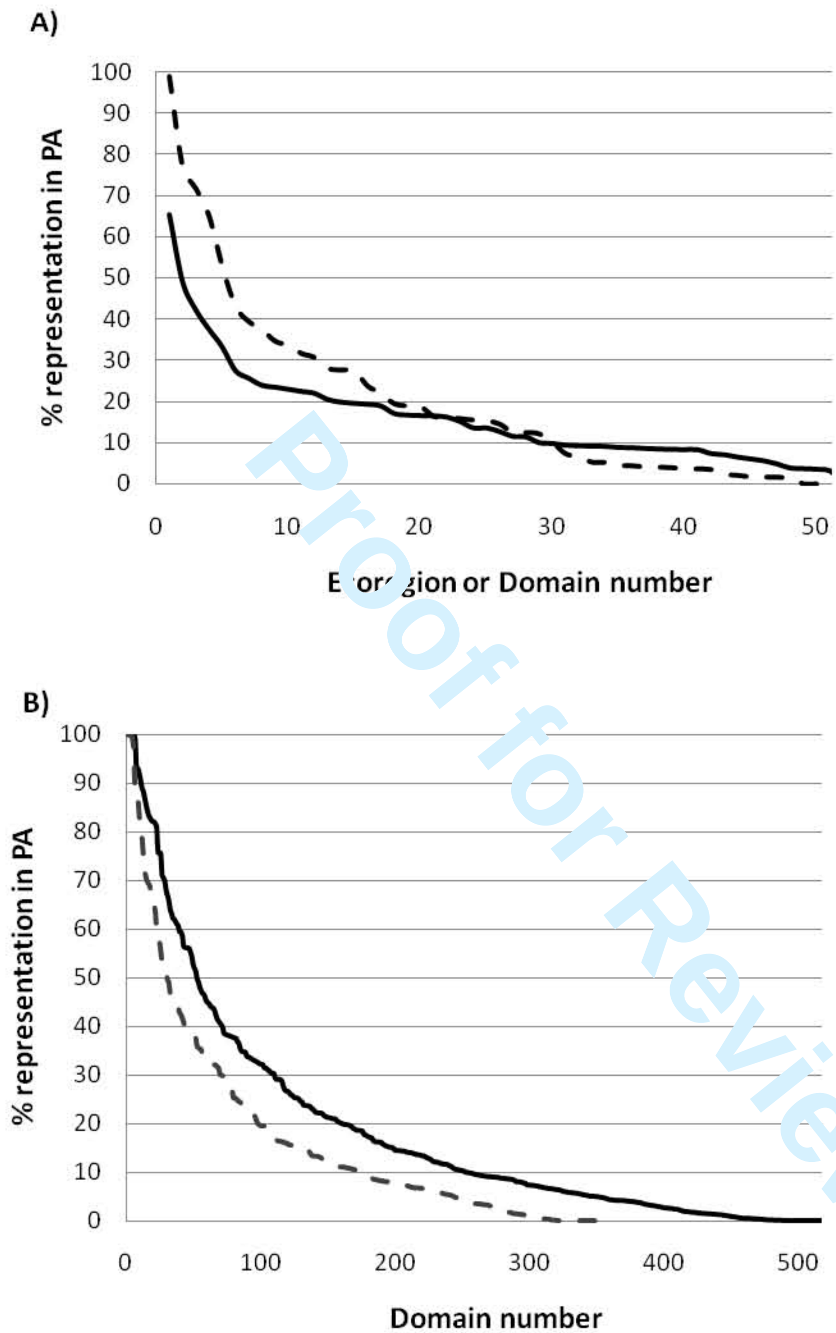

A)
% representation in PA
100
90
80
70
60
50
40
30
20
10
0
0
10
20
30
40
50
or Domain number
B)
% representation in PA
100
90
80
70
60
50
40
30
20
10
0
0
100
200
300
400
500
Domain number

Figure 4.

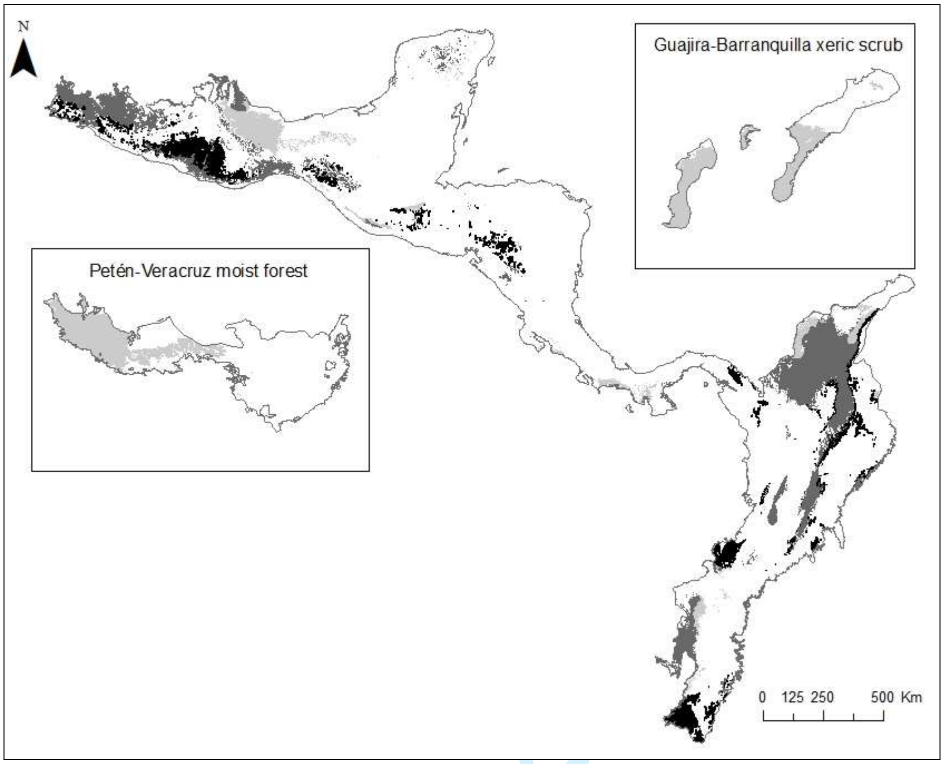

N
Guajira-Barranquilla xeric scrub
Petén-Veracruz moist forest
0 125 250 500 Km

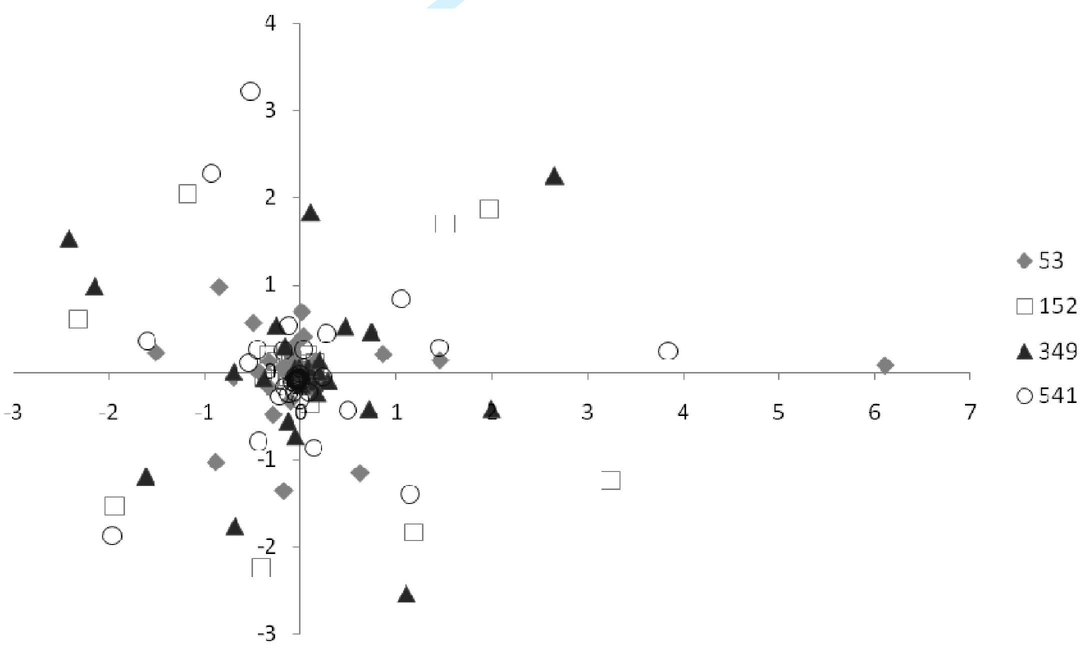
Figure 5

Supporting Information

Appendix S1. WWF ecoregions were graphically discriminated by the multidimensional scaling performed on each of the 13 environmental domain classifications (EDC). EDC selected for further analysis are highlighted in grey.

Number of EDC	Stress	Number of discriminated ecoregions
53	0.102	13
104	0.121	9
152	0.111	8
200	0.125	10
249	0.124	9
300	0.139	17
349	0.141	16
397	0.140	15
442	0.141	17
493	0.146	15
541	0.121	15
589	0.139	14
638	0.138	16

Appendix S2. Graph showing the distance between WWF ecoregions obtained by the multidimensional scaling performed on each environmental domain classification (EDC). As an example, results are shown for the 53 (diamonds), 152 (squares), 349 (triangles), and 541 (circles) EDC. WWF ecoregions can be partially discriminated by the different EDC given that many ecoregions were separated by small distances at the centre of the graph. For the 53, 349 and 541 EDCs, 13 to 16 different ecoregions showed distances that allowed discrimination. Eight ecoregions could be identified for the 152 EDC. Each geometric point represents a different WWF ecoregion.



:

Appendix S3 The percentage of representation in the PAs for (A) the WWF ecoregions, (B) 53 EDC, (C) 349 EDC, and (D) 541 EDC.

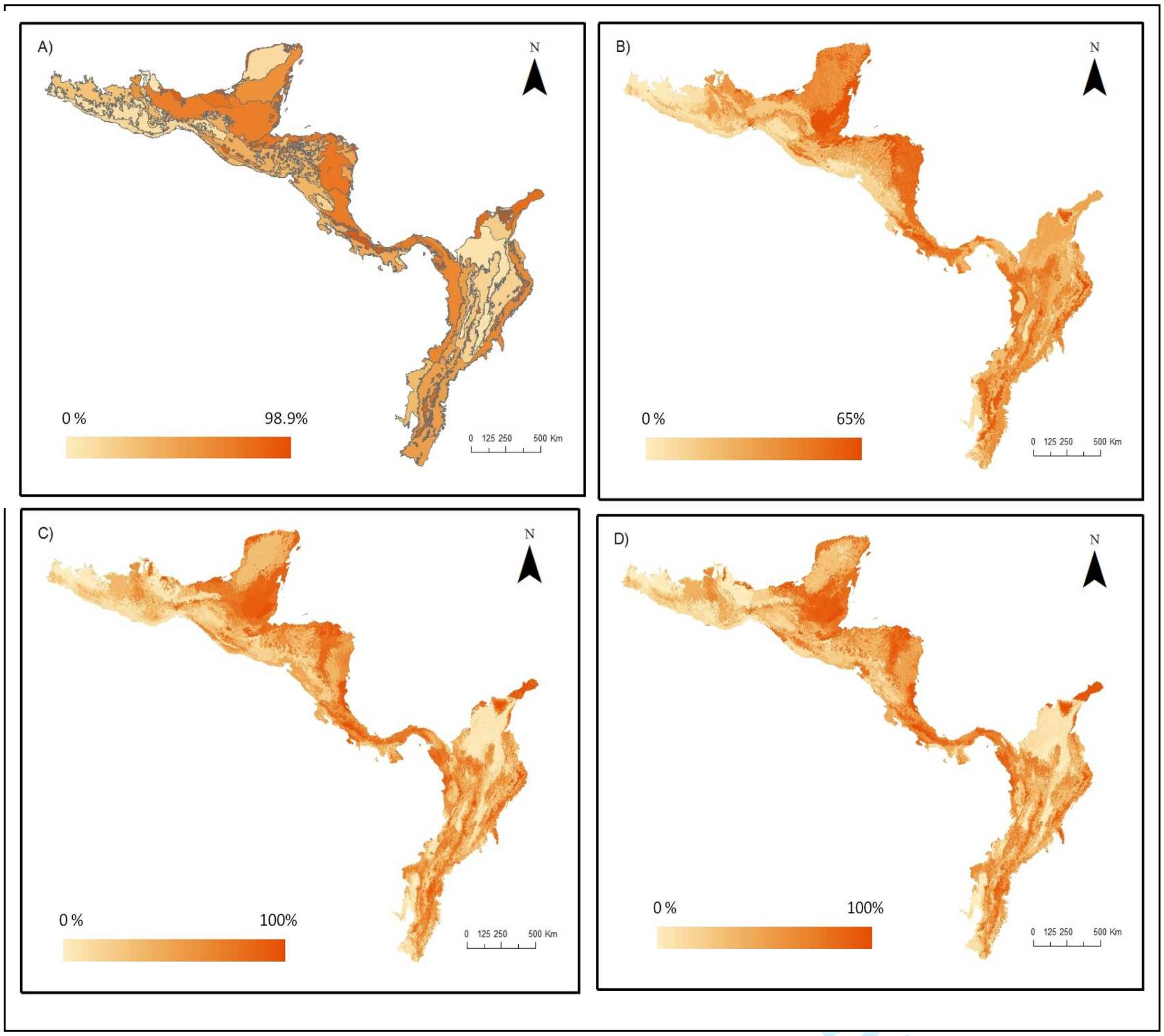
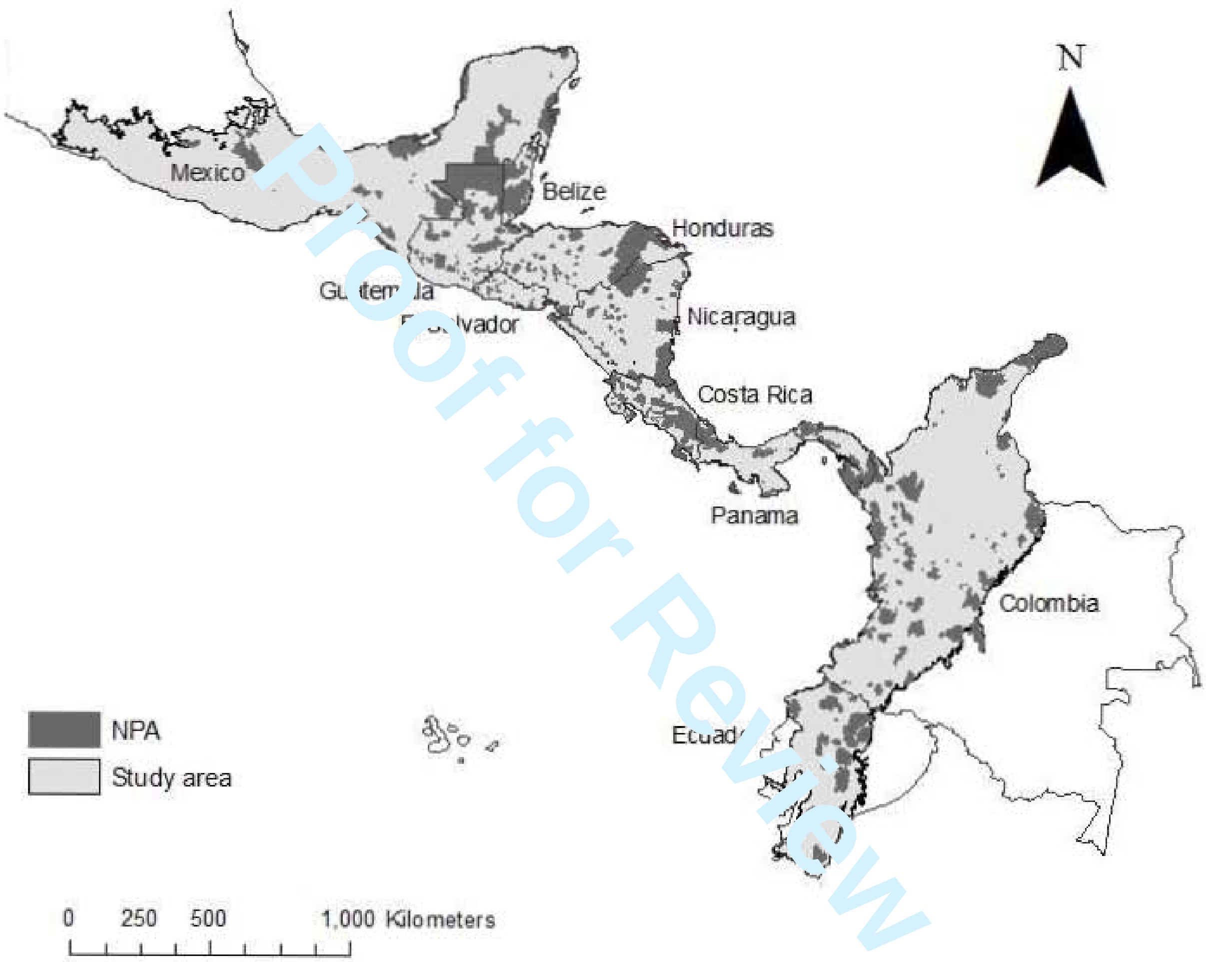


Figure 1: Study region (light gray) delimited by the WWF ecoregions, including 10 countries from the Mesoamerica, Choco and tropical Andes regions. The current protected areas (PAs) according to the World Database on Protected Areas 2007 are depicted in dark gray.



CAPITULO V

COMPARACIÓN DE ÁREAS SELECCIONADAS PARA LA CONSERVACIÓN USANDO MODELOS DE NICHO ECOLÓGICO DE ESPECIES AMENAZADAS Y DOMINIOS AMBIENTALES.

Environmental Domains and Species Distributions as Surrogates for Selecting Conservation Areas.

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Introduction

Habitat transformation is the primary cause of the biodiversity and extinction crisis that the word is facing today; the establishment of new reserves that adequately conserves biodiversity is urgently needed (Dirzo and Raven 2003; Margules and Pressey 2000). The practice of conservation planning in a systematic way is resent in the history of conservation reserves; as a result reserves have often been located in places that do not contribute to the representation of biodiversity, tending to be concentrated on remote and unproductive land, meaning that species occurring in productive landscapes are not protected (Margules and Pressey 2000). In order to fulfill conservation, reserves need to meet two objectives: biodiversity representativeness and assure its persistence (Margules and Pressey 2000), these objectives are generally planned based on biological and physical data (Rondinini et al. 2006).

Because of the complexity of biodiversity, surrogates such as sub-sets of species, species assemblages and habitat types have to be used as measures of biodiversity (Margules and Pressey 2000). The choice for a surrogate typically depends on factors including the goals to be achieved by the areas selected for conservation (Margules and Pressey 2000), the scale of analysis (Ferrier 2002), and constraints associated with data availability (Rondinini et al. 2006). This last point makes surrogate selection a subjective choice (Rondinini et al. 2006)

Because it is necessary to accept that we have an incomplete knowledge of biodiversity distribution and adopt methods for making most from the data available (Margules and Pressey 2000). Surrogacy analysis are urgently needed to better understand the advantages or consequences of using certain sets of species as biodiversity surrogates (Rodrigues and Brooks 2007).

Among the available types of biodiversity data, species occurrence data are widely used in conservation planning (Rondinini et al. 2006). Different types of species occurrence data exist: point localities, geographic ranges, and predicted distribution data. Predictive distribution models have the ability to use incomplete information, to generate spatially comprehensive predictions of species distributions, resulting more comprehensive and representative that point locality data (Rondinini et al. 2006).

The improvement of remote sensing techniques has facilitated the use of physical variables as surrogates, providing data to assess environmental classes with adequate spatial consistency across wide areas, which is an advantage for systematic conservation planning, as areas with the same kind of information and detail can be compare (Margules and Pressey 2000).

There are theoretical reasons why environmental variables should be good estimators of the spatial distribution patterns of species (Bailey 1985; Bailey 1987; Klijn and Haes 1994; Rosenzweig 1995; Yarrow and Salthe 2008) but while there are some empirical studies that add support (Ferrier et al. 1997; Trakhtenbrot and Kadmon 2005), some others do not (Araújo et al. 2001; Bonn and Gaston 2005). New statistical techniques are also being developed to compare how well different environmental surrogates reflect the distribution patterns of species (Ferrier et al. 2000; Rodrigues and Brooks 2007; Williams et al. 2006).

The following work compares and analyzes the areas for conservation selected using two types of surrogates, endangered species distribution models and environmental domains classifications; and proposes a set of priority conservation areas that complements the establish Natural Protected Areas (NPA), providing an efficient representation of endanger species and environmental heterogeneity in Mesoamerica, Tropical Andes and Chocó regions, under an scenario of land use coverage.

Methods:

The regions of Mesoamerica, Tropical Andes and Chocó are recognized as megadiverse regions, were high deforestation rates represent a serious threat to biodiversity (Myers et al. 2000). The Study area was define by 53 WWF ecoregions (Olson et al. 2001) in the 10 Countries (México, Belize, Guatemala, Honduras El Salvador, Nicaragua, Costa Rica, Panamá, Colombia and Ecuador) that encompass the study area in Mesoamerica, Tropical Andes and Chocó (Figure 1). The total study area was divided into 0.02 x 0.02 degrees cells, for a total of 383483 cells.

IUCN enlists 1074 species of terrestrial vertebrates and 2559 plants in the three categories of threaten species (CR, EN, VU) for Mesoamerica, Tropical Andes and Chocó, (http://www.iucnredlist.org/search/search-basic Last access May 2007). Threatened species deserve special attention when establishing new protected areas due to their high vulnerability and urgent conservation need (Mace et al. 2008). Ecological niche models were develop in the software MaxEnt (Phillips et al. 2006) for 313 EN, VU and CR red list species of vertebrates and plants (Table 1). Models were done in a logarithmic form, using the 19 bio variables from WorldClim (Hijmans et al. 2005), data of altitude obtained from the U.S. Geological Survey's Hydro-1K DEM data set (USGS 1998) and slope and aspect derived using the Spatial Analyst extension of ArcMap 9.2.(Esri 2004).

Environmental domain classifications were done using the same variables as in the niche models and performing a nonhierarchical cluster that organized cells, based on an environmental distance measurement, into 348 groups or domains. This was done in the software PATN v.3.11 (Belbin 1989) using the ALOC algorithm (Belbin 1987). All variables were equally weighted and the distance measure used was the Gower Metric, which standardized the variables allowing the combination of variables with different measurement units (Sneath and Sokal 1973).

Site selection was done using the software ConsNet (Ciarleglio et al. 2009). Sixteen different scenarios were created for assuring representation targets of 10, 20, 30 and 40% of each of the 313 species niche models, and each of the 348 environmental domains. Each of the representation target was run independently for species and ED representation, and for each surrogate type and representation target a selection was performed considering NPA a priori, that is starting the selection including the NPA´s, and another selection excluding NPA.

All the results were imported into ArcMap 9.2 were analysis of overlapping areas was done by intercepting the different solutions. Intercept results are presented as percentage of a solution, which corresponds to the area overlap between two solution. Overlap were done between solution from different surrogates but the same target representation, and between solutions where NPA were not included a priori with the NPA´s 2007 World Database (WDPA 2007). Comparisons using random samples were also done; random samples were created using the random sampling tool from Hawth´s Analysis Tools for ESRI´s ArcGIS (V3.x). Area calculations were done using the Projector! extension in Arcview 3.3 reprojecting the data into Equal Area Cylindrical.

For the statistical comparison between samples we used the Kolmogorov-Smirnov test as our data were not normally distributed, Shapiro-Wilk test $p<0.00009$. Statistical analyses were done in STATISTICA V. 8.0. (StatSoft 2007).

Results:

The area needed for representing the 348 environmental domains was almost as double as the area needed for representing the 313 species (Figure 2). For achieving a 40% target of the species, 8% of the study region will be needed to complement the NPA´s, and for representing the same target of ED, 25% of total study area. Meaning that when we include the NPA´s the total area set for conservation purposes will be around 25% for representing the species and 40% for the ED. As target increase, the difference in area between the selection done without NPA and including the NPA becomes smaller (Figure 2).

When analyzing the area selected in each country including the NPA´s, percentages vary between 14% and 53% for the species and 35% and 54% for the ED. The countries were more percentage of total area was selected in order to represent the species were Belize, Costa Rica, Ecuador, Guatemala and Panama. Similarly Belize, Costa Rica and Guatemala had more than 50% of their country selected in order to represent a 40% of the ED. Mexico was the country with less area selected, less than 20% for representing the 40% of species and less than 40% for represent the 40% of ED. Differences in the area selected in each country were greater when representing species than when representing ED, (Figure 3).

Total area in NPA´s from each country was positively correlate with the total area selected, $r>0.9$ and $p<0.05$ for both species and ED representation. The percentage of the country represented by NPA´s inversely correlate with the area needed to complement the NPA for ED representation $r=-0.94$, $p<0.05$ but was not correlated with the complementary area needed to represent the species $r=-0.48$, $p<0.05$. Ecuador and El Salvador are the countries that need to designate more new area into conservation in order to represent the 40% of species and ED. For the ED they will need more than 30% of their country, while for species Ecuador will need more than 20% and El Salvador more than 15%. Mexico, Honduras and Guatemala will need less than 5% for representing the species and Panama, Guatemala and Costa Rica, will need less than 15% for representing the ED. Belize is the country that will need less complementary area, with less than a 0.6% for species or ED representation (Figure 4).

When the selection was done without the NPA a priori, Ecuador and Costa Rica were the countries were more area was selected to represent species with more than 50% selected, while in Colombia, México and Honduras, less than 20% was selected, (Figure 5a). For ED representation all the countries except from Belize, showed a selection of more than 30% of the country for achieving a 40% target. Ecuador, Nicaragua, Honduras and Guatemala were the countries with the biggest percentages selected, (Figure 5b).

When we did a selection without including the NPA´s a priori and overlap the established NPA´s for each country, Belize and Guatemala showed the higher percentage of overlap for the species, more than 60%, and more than 40% for the ED selection. El Salvador showed less than 10% of overlap with species or ED selection, (Figure 6).

Of the total area selected for representing species that complemented the NPA’s, between 14% and 47% was also selected for complementing ED, percentage of overlap varied in each country, (Figure 7a). In Ecuador and El Salvador, which are the countries that needed more extra area to represent the species, more than 40% and an 80% of overlap was found for the 40% target respectively. Although Guatemala and Mexico are two of the countries that needed less area to complement the NPA’s for the species representation, the overlap in its selection is lower compare to the other two countries were little area is needed, Belize and Honduras, (Figure 7a).

Percentage of overlap was grater for the species than for the ED selections, just a 16% of the total area selected for complementing NPA´s in order to represent ED was overlap by the species selected area, (Figure 7b). No country except Belize had more than 56% of the area selected for representing ED overlapping with the area selected for representing species. Again Ecuador and El Salvador have one of the highest percentages of overlap although they are the countries that needed more area to represent the ED. Mexico, Honduras, Guatemala and Colombia had the lowest percentages of ED selection overlap with area selected for species representation, (Figure 7b).

The percentages of overlap change when the selection was done without the NPA a priori. Statistical difference showed that percentage of species overlap by ED selection were higher when NPA were included in the selection, Kolmogorov-Smirnov Test $p<0.025$, mean percentage of overlap with NPA included was 54.6% and without NPA 38%. While percentage of ED overlap by species selection were higher when the NPA were not included a priori, Kolmogorov-Smirnov Test $p<0.001$, mean percentage of overlap with NPA included 12.8% and without NPA 29%.

Between 21% and 55% of the total area selected for species representation was also selected for representing the ED when selection was done without NPA´s (Figure 8a). Consistently for all the countries the percentage of overlap increases with increasing representation target. In all Central American countries except Belize and Costa Rica percentage of species selection overlapping ED selected area were greater than 50% for some of the representation targets, having the greatest overlapping percentages of species selections, (Figure 8a). The percentage of total ED selection that overlaps the species selection ranged between 9% and 30%, (Figure 8b). Colombia, Guatemala, Honduras and Mexico showed the lower percentages of ED selection overlapping by the species selection. Costa Rica, Ecuador, El Salvador, Nicaragua and Panama showed the higher percentages of ED selection overlapping the species selection, (Figure 8b).

Percentages of area selected for the species or the ED representation that overlap area selected for ED or species at the same target, were statistically higher than the percentage of overlap if the same amount of area were selected at random. For species percentages of overlap Kolmogorov-Smirnov Test had a $p<0.001$, mean with random selection 19.8% and mean with complementarity selection 38.3%. For ED percentages of overlap Kolmogorov-Smirnov Test had a $p<0.001$, mean with random selection 13% and mean with complementarity selection 29%.

Figures 9 shows the area selected for species and the ED and the area were they overlap in order to complement the NPA´s for achieving a 40% representation target. Area selected for both species and ED representation concentrates in some regions, in Mexico northern from the Isthmus of Tehuantepec, in Centro America pine-Oak forest and dry forest, and in Southern Colombian and Ecuador Andes ranges.

Discussion:

Recently, the issue of surrogacy has been the subject of intense debate, with candidates including species occurrences (e.g.,Brooks et al. 2004), spatial patterns in species richness and turnover (e.g.,Ferrier et al. 2004), environmental diversity (e.g.,Faith 2003), and vegetation types or land classes (e.g.,Lombard et al. 2003). To our knowledge there has been no consistency in determining which the best possible surrogate set is, and studies have fund different results in surrogacy effectiveness (Rodrigues and Brooks 2007).

Methods for the comparison of surrogacy effectiveness include comparing the spatial overlap between complementary sets of sites selected (e.g.,Grenyer et al. 2006), measuring the correspondence between the sequence of complementary site selection (e.g.,Polasky et al. 2001), investigating the effectiveness of sets of sites based on surrogate data in representing the target taxa (e.g.,Rodrigues and Brooks 2007), spatial patterns of site irreplaceability (e.g.,Ferrier 2002), and complementarity between targets and surrogates (e.g.,Williams et al. 2006). The approaches based on comparisons between the spatial location of the sites selected using surrogates and target features are not enough proof of the surrogacy effectiveness, because usually complementary representation problems typically have multiple solutions varying on their spatial arrangement (Rodrigues and Brooks 2007), our analysis uses percentages of overlap between different solutions, but as different representation targets were used and some solution included NPA and other not, there are 8 different spatial solution for each surrogate set, so variability in solutions is a issue incorporated in our results, as so we believe they are a good proof of surrogacy effectiveness.

Theoretically higher levels in the biological hierarchy, such as habitat types and ecosystems can integrate more of the ecological processes that contribute to the maintenance of ecosystem function (McKenzie et al. 1989), additionally different kinds of environments are assumed to support different sets of species (Yarrow and Salthe 2008), promoting the use of environmental classifications as surrogates sets (Belbin 1993; Faith et al. 2001). As in our work, others have shown that the use of environmental surrogates in conservation area selection better represent other groups of species that if areas were chosen at random (Ferrier et al. 1997; Sarkar et al. 2005; Trakhtenbrot and Kadmon 2005).

But as our results shows, environmental surrogates do not represent all the area selected for species surrogates, that might be because the relationship between environmental classes and the distribution and abundance patterns of species can be unclear and difficult to quantify and some species may require a combination of environmental variables not recognized by the environmental classification used (Margules & Sarkar 2007). Additionally environmental classification can´t represent those species were ecological and evolutionary history play a lead role in determining their distribution patters, needing those species to be identify and map as independent surrogates (Margules and Sarkar 2007).

Surrogates are chosen with the expectation that they will sample the biological or environmental space uniformly, for example representing all members of a taxon or the entire range of environmental parameters (Margules and Pressey 2000). In that sense, data needed to delimitate environmental classifications are more generally available at a consistent level of detail across wide geographical areas, than field records of species occurrence (Margules and Sarkar 2007). Environmental classifications results in environmental domain distributions across the entire study region, but contrary to species distribution, environmental domains distributions do not overlap each other, ending up with more area selected giver its lower complementarity. Due to their continuum distribution across all study region percentages of area needed to represent environmental surrogates are more similar across the different countries than percentages needed to represent species distributions.

The spatial arrangement of the different surrogates can be the explanation to others of our results, for example, Mexico NPA´s perform poorly representing the species and environmental sets use in this work as surrogates, area needed to complement the

ED representation did inversely correlate with its poor NPA´s representation, but for complementing the species representation Mexico was one of the countries that least area needed; the question that arises is then if Mexico has poor endangered species diversity, or if its diversity is represented in others regions not included in this analysis, or most probably if there are limitations in biological data to accurately represents endangered species diversity.

Although in many localities data on the distribution of species are available, researchers have been well aware of their taxonomic and geographical bias (Reddy and Davalos 2003). Species distribution models have been applied to conservation planning as they can cope with species data limitations (Elith et al. 2006), and have proven to performed better as surrogates than using the locality points (Ferrier et al. 1997). Results of the performance of species as surrogates vary, some studies report pessimistic results about the adequacy of some taxonomic set of species as surrogates (Dobson et al. 1997; Flather et al. 1997; Lund and Rahbek 2002). Given our result of species poor performance in representing different environmental conditions, our hypothesis is that they will not adequately represent other species sets, but other works have found that threatened species seems to be effective surrogates for a broader set of species, performing better as surrogates than using environmental data (Rodrigues and Brooks 2007).

Given our low percentages of overlap between areas identify to represent environmental domains or species distributions, and the weak surrogacy value found in surrogacy effectiveness literature (Rodrigues and Brooks 2007), as suggested by other authors a sensible strategy for prioritizing areas will be to sample all environmental classes and to include as many species as possible, resulting that in practice, combinations of surrogates will be the most successful option (Margules and Sarkar 2007). Surrogate sets has to go beyond distributional representation and look toward for representing ecological processes, as they are key for the persisting of biodiversity. Surrogates of persistence are particularly needed for threatened species, which have lower probabilities of long-term persistence in the absence of conservation effort (Rodrigues and Brooks 2007).

A promising research line is therefore to understand if conservation planning for the persistence of some species in a particular taxonomic group is effective surrogate for the persistence of others species, what will be expected if there is a coincidence in the spatial patterns of threat for both groups (Rodrigues and Brooks 2007). Persistence in surrogates will also play an important role in ecosystem services conservation planning, which in the actual international conservation arena is gaining urgency and importance (Egoh et al. 2007).

Performance of surrogates depends on the spatial scale; as noted by Margules and Sarkar (2007) an interesting question will be if the relation with scale will be equal across different geographical regions. As our results shows differences across the percentage of overlap in the different countries, we hypothesis that not only will the scale relation be different but the overall performance of surrogates set will vary depending on the geographical region, as also found Ferrier and Watson (1997), and as mention earlier in the discussion it will be related to the processes that delimit the species distributions.

Another interesting question will be if the established NPA will make surrogates work different, our results suggested that they play an important role, as their inclusion of exclusion from site selection was reflected in difference in overlapping area between different surrogates. Environmental surrogates seems to better represent the species when NPA were included in the selection, further research has to be done on this, but based on our results we will suggest that for real implementation scenarios it is important to include environmental surrogates.

For lower representation target the different in area needed for assuring surrogate representation was bigger between selected area with a priori inclusion of NPA and selected area without including NPA, no matter which surrogate we used. This issue is related with the cost of postponing conservation planning (Fuller 2007), because conservation in the real world competes with other land uses needs, and as establish by the CBD (Convention on Biological Diversity 2002) a representation target of 10% is what countries have committed to represent in conservation areas, biological representation in NPA has to be properly planed for small target representation where the cost of inadequate planning is more.

Our recommendation for Mesoamerica, Tropical Andes and Chocó regions is to use sets of different surrogates types, the regions great biological diversity implies that several processes are delimitating species distributions, as a result only one surrogate type can´t effectively represent all of their biodiversity. Until we can find surrogates for the process that generate biodiversity the use of species and environmental characteristics is the most promising option in order to represent different biological characteristics. Finally those areas where there was an overlap in areas selected both for species and environmental surrogates should address special attention for local conservation assessments (Grenyer et al 2006), due to their high complementarity values in representing different biological characteristics.

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Figure legends:

Figure 1: Natural Protected Areas and Countries encompassing the study area in Mesoamerica, Chocó and Tropical Andes regions.

Figure 2: Percentage of total study area needed to represent 10%, 20%, 30% and 40% targets of species and environmental domains. The different curves represent selections including NPA´s (squares), selection without NPA´s (triangles) and the area needed to complement the NPA´s, which equals total area selected including NPA´s minus the countries NPA´s area.

Figure 3 Percentage of each country that is selected to represent 10%, 20%, 30% and 40% target of a) species and b) ED.

Figure 4 Percentage of each country that is needed to complement the NPA´s in order to represent 10%, 20%, 30% and 40% target of a) species and b) ED.

Figure 5 Percentage of each country that is selected without the NPA´s a priori in order to represent 10%, 20%, 30% and 40% target of a) species and b) ED.

Figure 6 Percentage of country´s NPA that overlap the selection done without including the NPA´s a priori for 10%, 20%, 30% and 40% target of a) species and b) ED.

Figure 7 Percentage overlapping area selected for complementing NPA for a) species representation that overlap the selection for ED representation and b) ED representation that overlap the selection for species representation.

Figure 8 Percentage of area selected without NPA for representing a) species and b) ED that overlap area selected for both species and ED.

Figure 9 Selected area for complementing NPA´s in order to represent 40% target of species (blue area) and ED (yellow area). Overlapping area between species and ED selection is shown in red. Green areas represent the established NPA´s

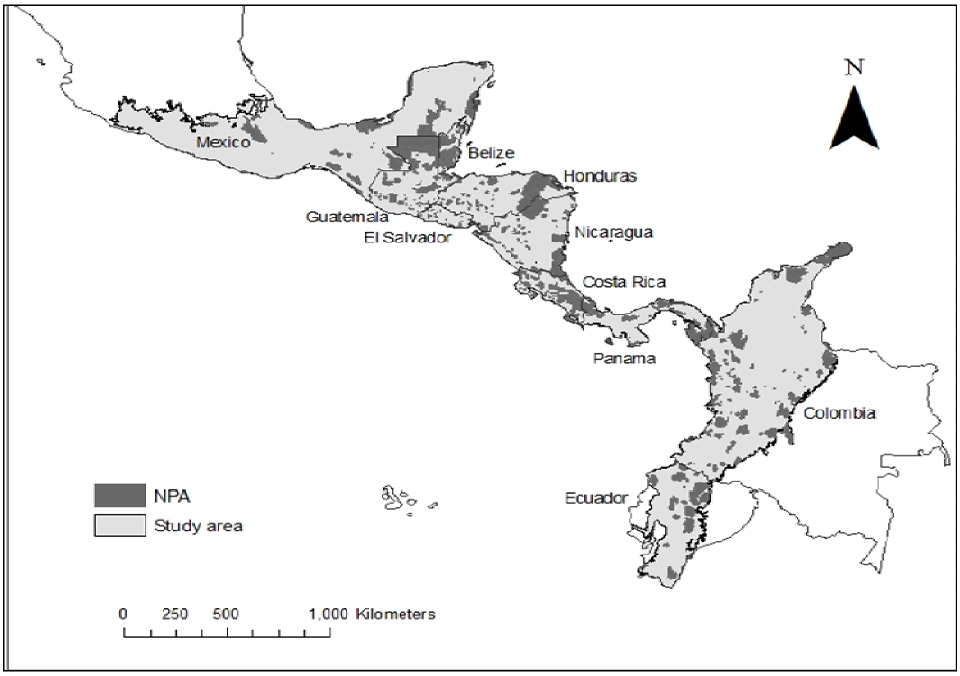
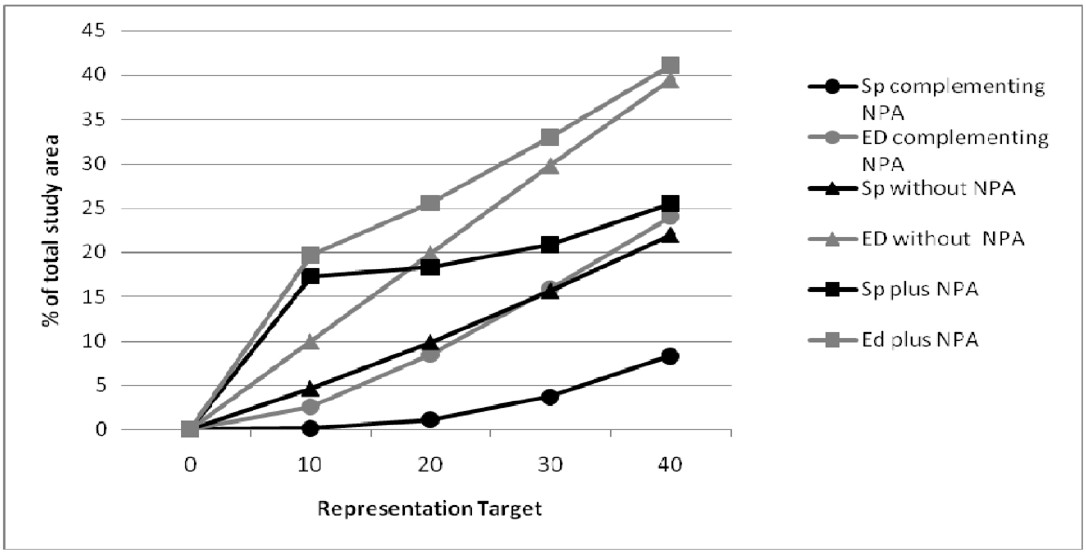


Figure 1: Natural Protected Areas and Countries encompassing the study area in Mesoamerica, Chocó and Tropical Andes regions.



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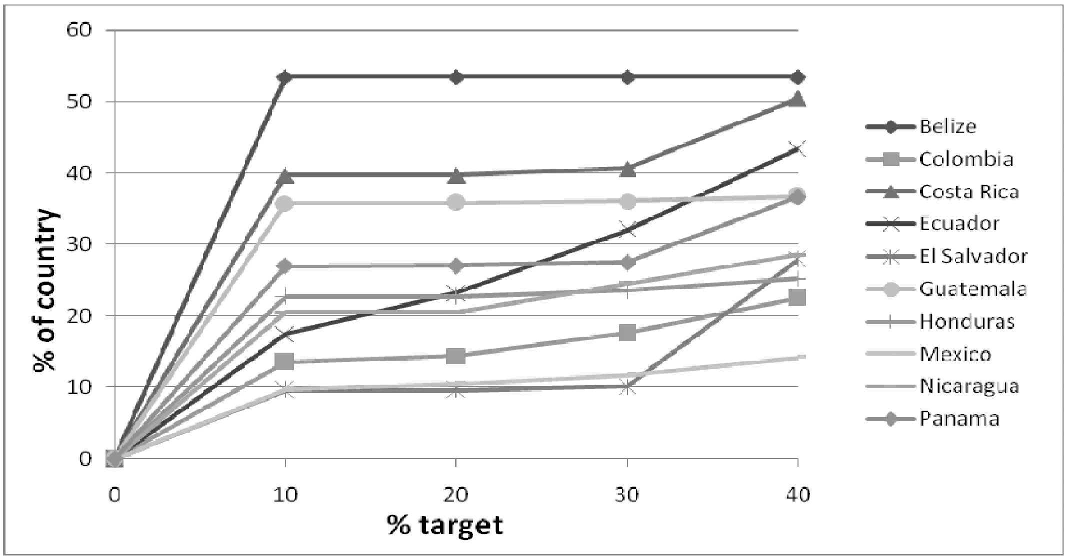
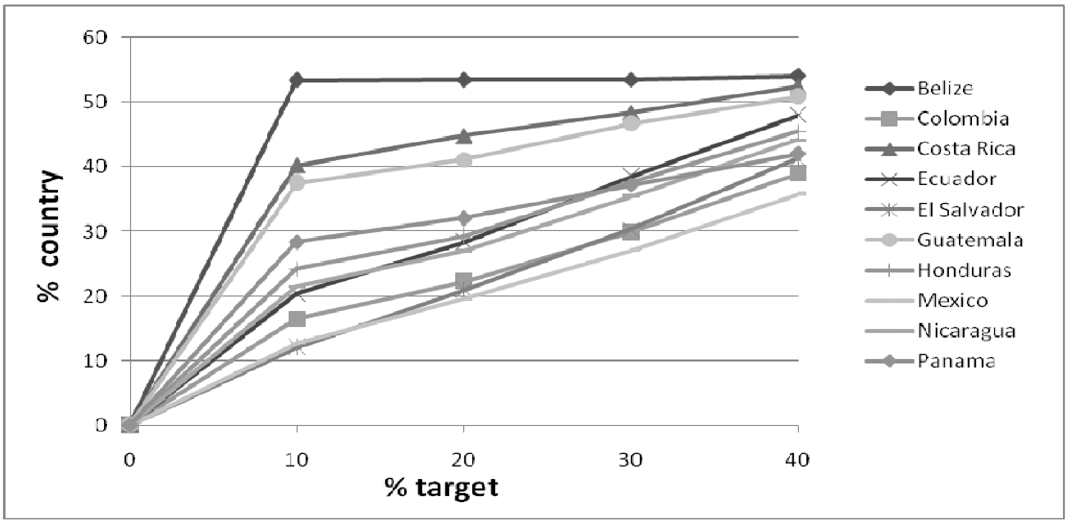
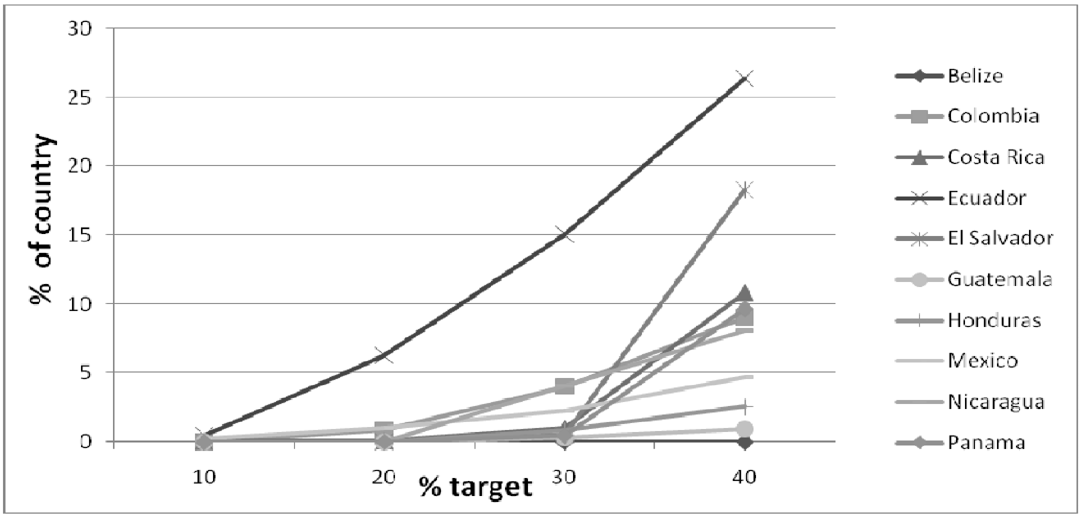


a) SP



b) ED

Figure 3 Percentage of each country that is selected to represent 10%, 20%, 30% and 40% target of a) species and b) ED.



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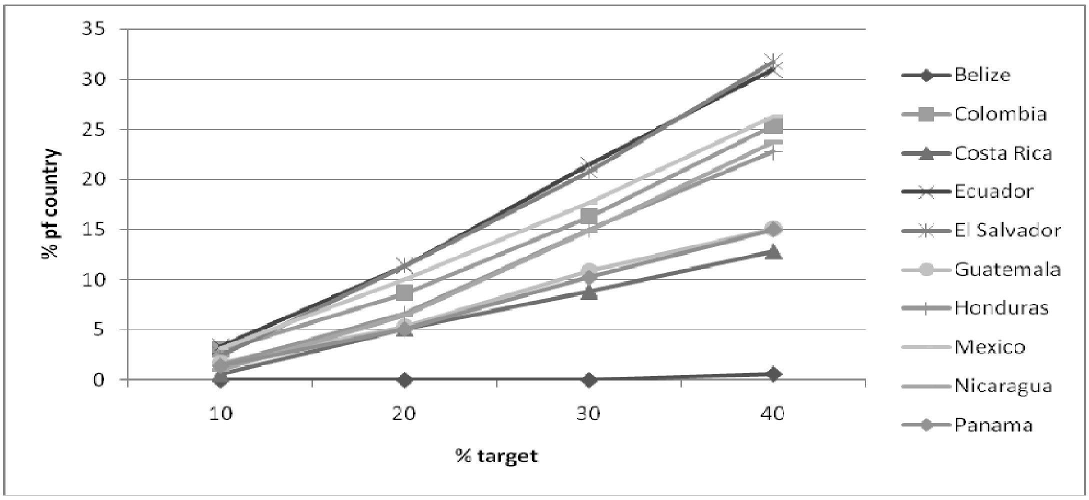
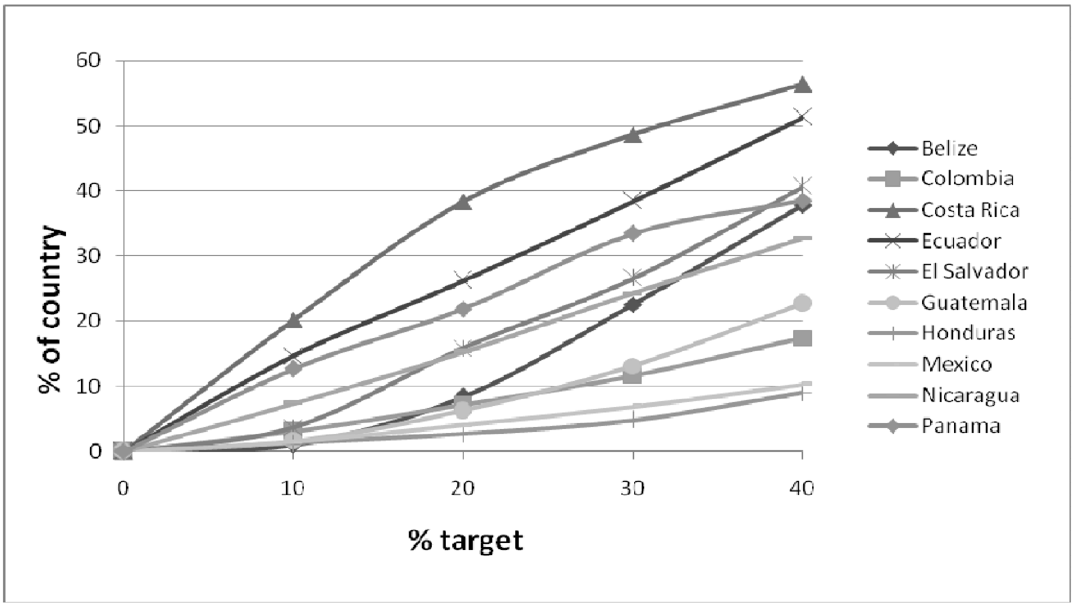
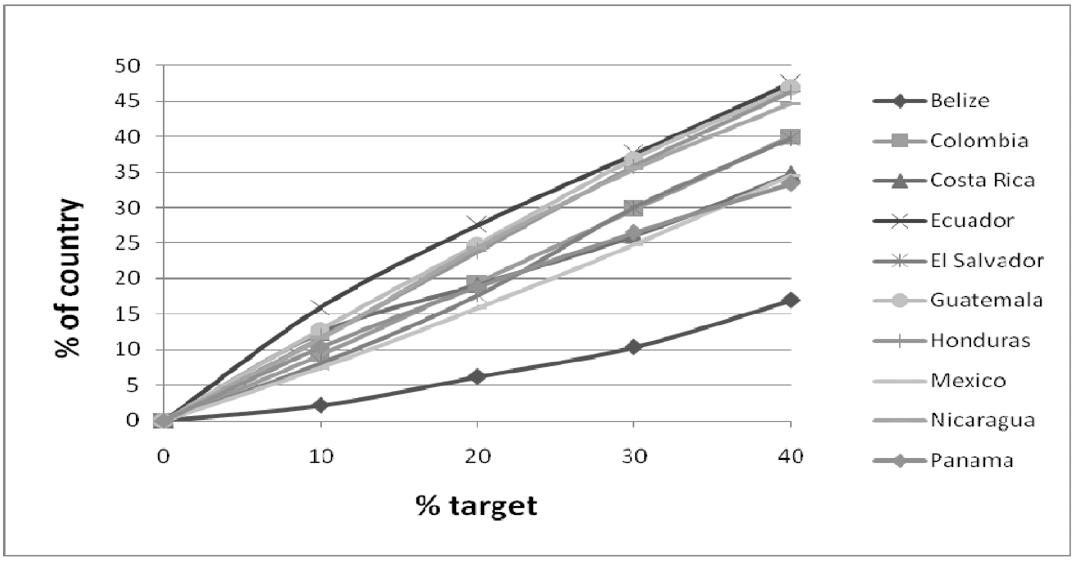


b) ED

Figure 4 Percentage of each country that is needed to complement the NPA´s in order to represent 10%, 20%, 30% and 40% target of a) species and b) ED.

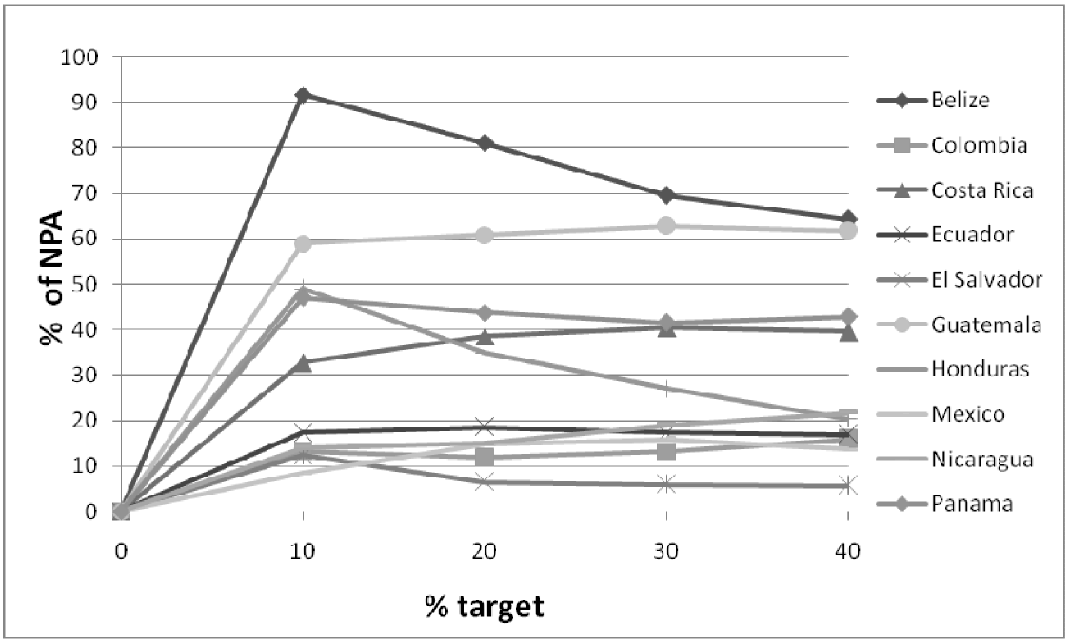


a) sp

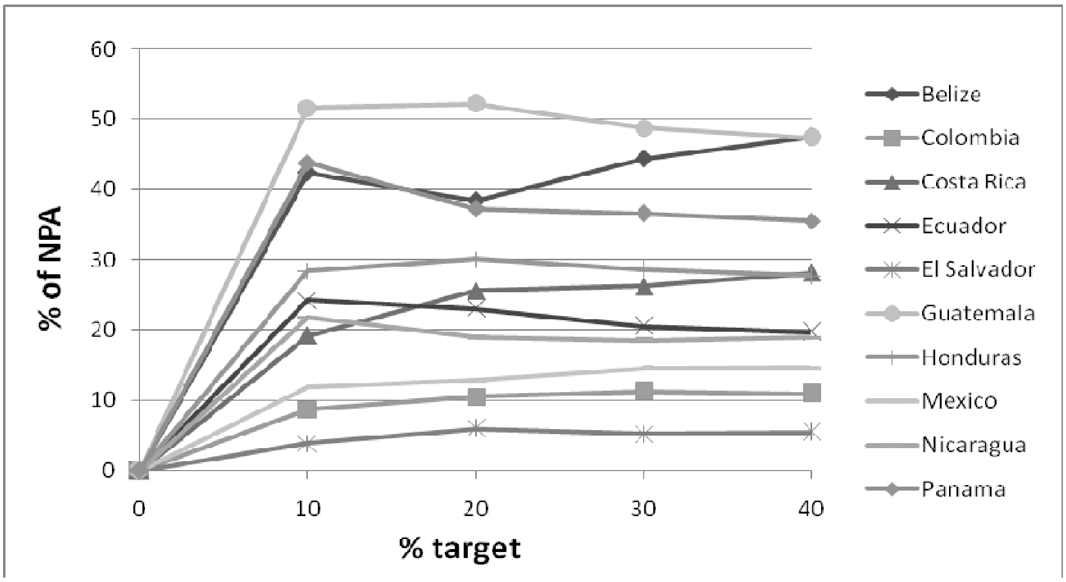


b) ED

Figure 5 Percentage of each country that is selected without the NPA´s a priori in order to represent 10%, 20%, 30% and 40% target of a) species and b) ED.

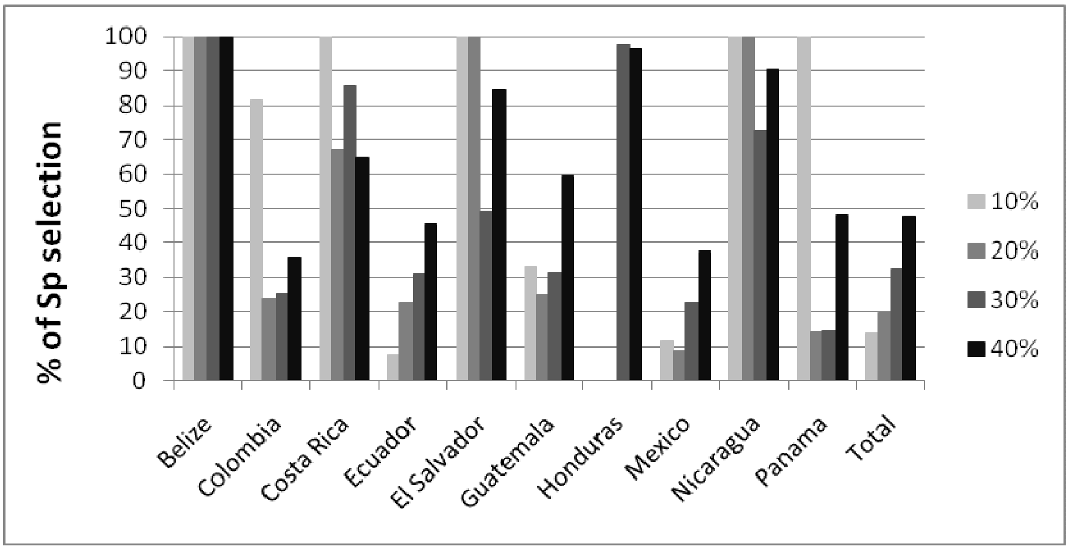


a) sp

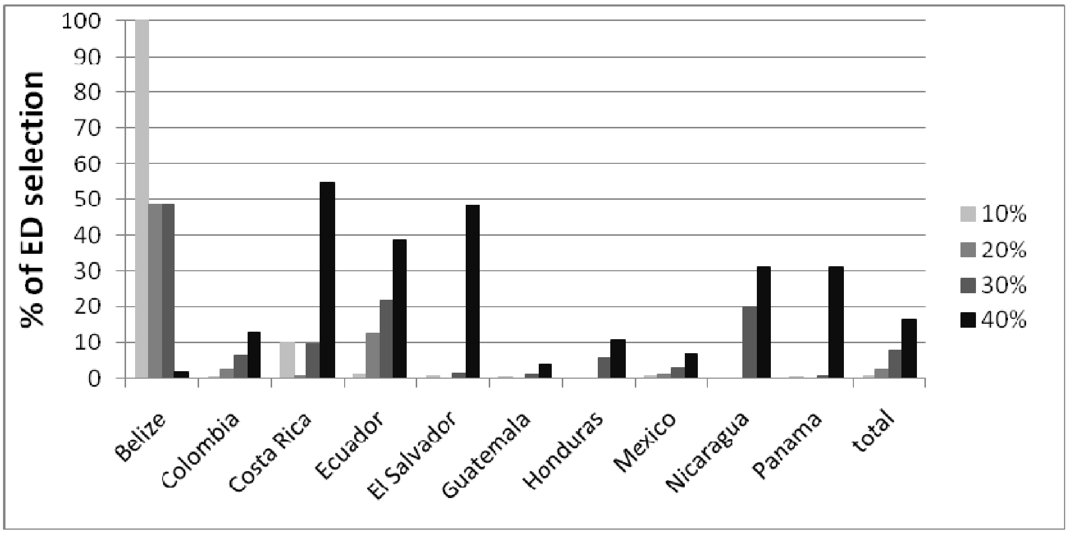


b) ED

Figure 6 Percentage of country´s NPA that overlap the selection done without including the NPA´s a priori for 10%, 20%, 30% and 40% target of a) species and b) ED.

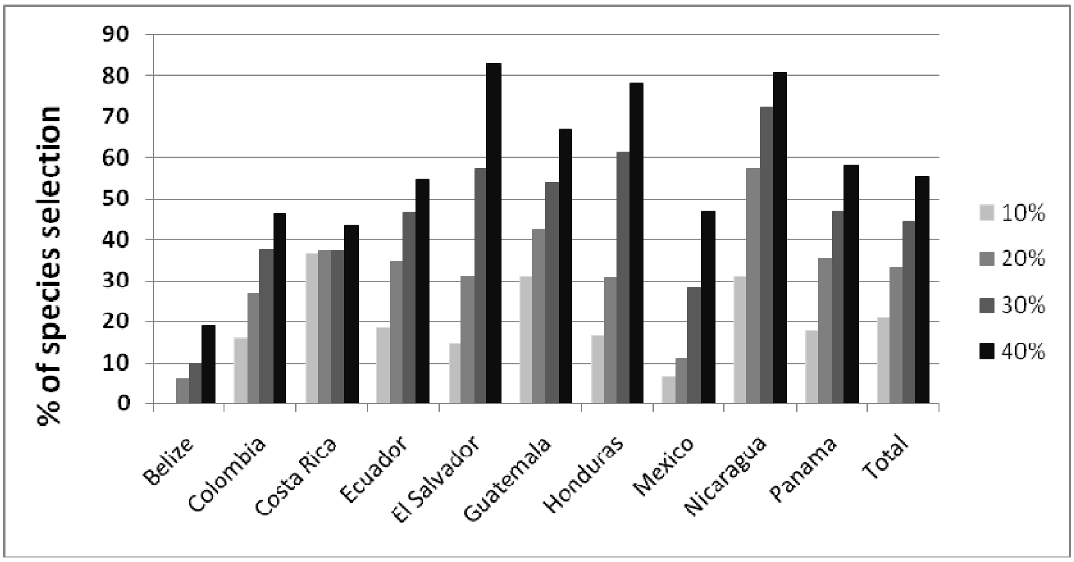


a) sp

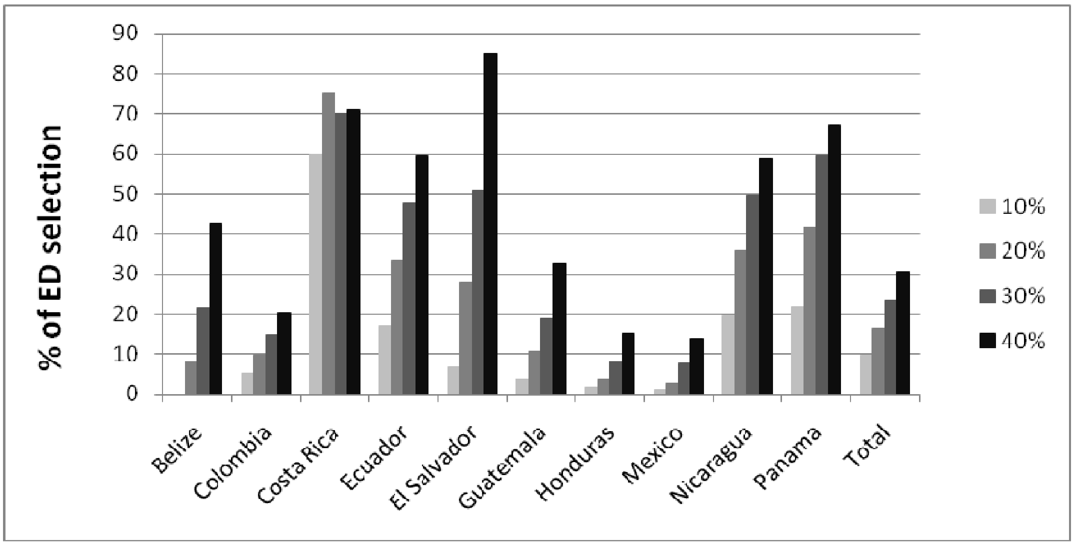


b) ED

Figure 7 Percentage overlapping area selected for complementing NPA for a) species representation that overlap the selection for ED representation and b) ED representation that overlap the selection for species representation.



a) sp



b) Ed

Figure 8 Percentage of area selected without NPA for representing a) species and b) ED that overlap area selected for both species and ED.

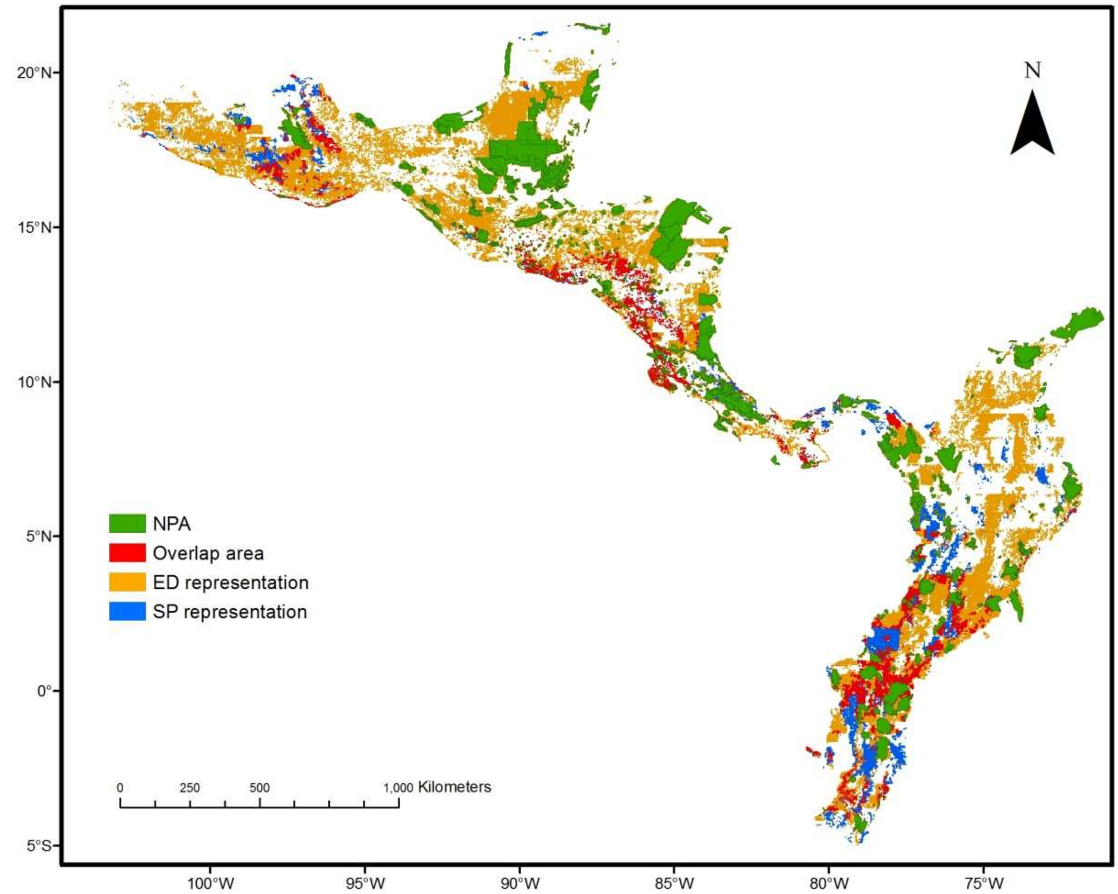


Figure 9 Selected area for complementing NPA´s in order to represent 40% target of species (blue area) and ED (yellow area). Overlapping area between species and ED selection is shown in red. Green areas represent the established NPA´s

Table 1: Number of species used for each taxon and for each IUCN category.

	CR	En	Vu	Total
	Used	**Used**	**Used**	**Used**
Animalia	**4**	**18**	**6**	**28**
Amphibia	4	18	2	24
Aves			1	1
Mammalia			3	3
Plantae	**7**	**73**	**205**	**285**
LYCOPODIOPHYTA				
Lycopodiopsida		1	3	4
POLYPODIOPHYTA				
Polypodiopsida		3	11	14
TRACHEOPHYTA				
Coniferopsida			1	1
Liliopsida	1	11	24	36
Magnoliopsida	6	58	166	230
Total	**11**	**91**	**211**	**313**

CAPITULO VI

ANÁLISIS MULTICRITERIO PARA LA SELECCIÓN DE ÁREAS DE CONSERVACIÓN, MAS ALLÁ DE LA REPRESENTATIVIDAD BIOLÓGICA

Planning for conservation beyond biological representativeness: towards integrative landscape management

Running head: Multicriteria Conservation Planning

Keywords: Conservation Planning, Biodiversity Representation, Threat, Opportunities, Landscape Management, Neotropics.

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Abstract

Prioritization of conservation areas is commonly based on biodiversity surrogate representation. This work proposes a prioritization of biodiversity conservation areas in the Mesoamerica, Tropical Andes and Chocó regions, going beyond biological and environmental representation by incorporating land use patterns in transformed landscapes, opportunities for the involvement of poor communities in conservation actions, and the reduction of the impact of natural disasters. Area prioritization was done using the ConsNet software package under three scenarios with different fundamental objectives: representation of biodiversity, anthropogenic threat reduction, and improving implementation. Initial solutions were built by solving the minimum area problem for a 10% target of representation of 313 threatened species and 348 environmental domains. Refinement of the initial solution was obtained using a General Multi-Criteria Analysis (GMCA) protocol. Two best solutions were independently identified for each scenario, one including established Natural Protected Areas (NPA), and other excluding them. In order to complement the NPA across the three different scenarios, from the 53 ecorregions that encompass the study area, eight needed more than 50% of the selected area in cultivated areas and five had more than 60% of the selection in low human influence area. NPAs in Belize, Ecuador and Nicaragua coincided to a greater degree with areas selected for the implementation scenario; NPAs in Honduras, El Salvador and Mexico showed more coincidence with areas selected for the threat scenario. Except for Belize, areas selected in all countries' NPAs showed low coincidence with the biodiversity-based scenario. The scenarios build showed different performance across the variables, reflecting the complex interaction and trade-offs between socio-economic, land use and biological variables. This analysis identifies several areas at which conservation actions should be accorded high priority. Given the high percentage of selected areas in cultivated areas, conservation initiatives have to be developed towards the integration of rural landscapes and natural vegetation remnants.

1. Introduction

Strategies for the conservation of natural habitats that can curtail the increasing loss of biodiversity require an efficient allocation of scarce resources. The design and establishment of conservation area networks to decide where to allocate such scarce resources for biodiversity management is central to the conservation of biological diversity (Margules & Pressey 2000). The prioritization of areas for planning conservation networks has commonly been done using data on biological or environmental representation were targets and goals are established for surrogate coverage and total network area, and sometimes used in conjunction with spatial configuration criteria such as size, shape, dispersion, connectivity, alignment, and replication (see eg.: Cerdeira et al. 2005; Franco et al. 2009; Fuller et al. 2006; Peralvo et al. 2007; Sarkar et al. 2009).

In the current global context, the threats to and loss of biodiversity are increasingly linked to human activities arising from a variety socioeconomic and cultural factors (Soulé 1991). Therefore, protection of biodiversity needs to be integrated with natural resource management in such a way that sustainable livelihoods constitute a major goal (Margules & Sarkar 2007).

Protected areas have recently been included in international agendas as an important consideration for sustainable development (United Nations 2008); in developing countries with high biodiversity levels, widespread poverty conditions and rapid population growth, conservation areas will become increasingly important for the continued persistence of biota (Naughton-Treves et al. 2005). Biodiversity conservation areas could also play a role in reducing the likelihood and impacts of natural disasters (Miller et al. 2001; Mitchell & Ricardo-Grau 2004), given that natural disasters are often related to natural habitat loss and degradation (de Sherbinin 2002).

The inclusion of socio-political criteria to set targets and goals in the selection and prioritization of areas for conservation action has been explicitly recognized in the literature (Faith & Walker 1996; Margules & Pressey 2000; Margules & Sarkar 2007), and a few exercises have so far included some of them (Cowling et al. 2003; Faith & Walker 1996; Faith et al. 1996; Moffett et al. 2005; Williams et al. 2003). A diversity of methods exist for approaching multicriteria problems; the choice of an appropriate method depends on the context of analysis (Sarkar 2005; Moffett & Sarkar 2006; Margules & Sarkar 2007).

Although humans have extensively transformed the natural ecosystems globally through different land use forms (Ellis & Ramankutty 2008; Haberl et al. 2007; Sanderson 2002), many rural landscapes still provide important habitat and suitable environmental conditions for many species to exist (Arroyo-Mora et al. 2005). Rural landscapes, therefore, should be included in the design of conservation area networks, especially in tropical areas where natural remnants keep being systematically eliminated and increasingly isolated (DeFries et al. 2005; Sánchez-Azofeifa et al. 2003). However, the ability of settled and transformed landscapes to harbor biota depends on the proportion, the spatial configuration of the relict areas, and the types of uses surrounding them (Geneletti 2007).

To the best of our knowledge there has been no conservation prioritization exercise in Latin America using data other than biological information, nor has there been a multi-criteria analysis performed at a regional scale in high priority hotspots. However, there is a pressing need to explore conservation planning scenarios for plausible opportunities to minimize socioeconomic costs while preventing the extinction of globally threatened species (Cameron & Williams 2008).

This paper addresses a regional scale prioritization of biodiversity conservation areas in the Mesoamerica, Tropical Andes and Chocó regions, looking towards complementing the biological representation objective under three different scenarios based on the assessment of social and land use data. The allocation of conservation areas takes into account land use patterns in transformed landscapes, opportunities for involvement of poor communities in conservation actions, and the reduction of natural disaster impacts.

2. Study Area Background

Mesoamerica, the Tropical Andes, and the Chocó regions are recognized as biodiversity hotspots given their species richness, exceptional concentration of endemic species, and the high ongoing rates of natural vegetation loss (Lamoreux et al. 2006; Myers et al. 2000). Moreover, these regions share a recent evolutionary history giving rise to an overall high regional biodiversity (Stehli & Webb 1985).

These regions show at the same time high levels of ongoing landscape transformation and threats to biodiversity, with currently only one-third of their pristine natural remaining (Brooks et al. 2002); for example the remnants of natural ecosystems in the Colombian Andes is 36.9% and in the Colombian Chocó 66.5% (Etter et al. 2006). Some ecoregions of the area such as Central American pine-oak forests, the Sierra Madre del Sur pine-oak forests and Eastern Cordillera Real montane forests (Mexico), and the Cordillera Oriental montane forests, Chocó-Darien moist forests and Magdalena Valley montane forests in Colombia, show very low representation in NPAs (Soutullo et al. 2008). Several assessments of NPAs in these regions coincide in their poor performance in representing biological or environmental diversity (Fandiño & van Wyngaarden 2005; Sarkar et al. 2009; Sierra et al. 2002). Although many studies have addressed conservation prioritization for these hotspots, most of them are local or national scale studies based on single taxonomic groups as surrogates (see eg: Alvarez Mondragón & Morrone 2004; Greenbaum & Komar 2005; Jaramillo 2006; Powell & Bjork 1995), and few incorporate non biological parameters, such as spatial design and human impact measurements (Fandiño & van Wyngaarden 2005; Fuller et al. 2006; Sierra et al. 2002).

Between 1990 and 2005, the study region experienced mean annual deforestation rates that exceed 1%, while population increased at an annual rate of 2% (World Bank 2007). If current deforestation rates continue, these regions are predicted to lose most of their species in the near future (Brooks et al. 2002). Moreover, these regions face underdevelopment indicators including high levels of social inequality, an average >7% unemployment rate, 32% of the children working, as well as increasing social conflicts and high emigration rates (nearly three million people emigrated from their country of origin between 2000 and 2005), all leading to 42% of their population living below the national poverty line (World Bank 2007). At the same time a diverse range of distinct ethnic groups, including indigenous, afro-American and mestizo communities are found, many living in natural areas or in the agricultural frontiers. Such social and economic complexities challenge the traditional biodiversity conservation strategies. There is a pressing need for conservation actions that include priority conservation areas in rural landscapes (Harvey et al. 2008), and conservation areas that contribute to poverty reduction and can help respond to climate change while simultaneously supporting biodiversity conservation (Roe 2008).

3. Materials and Methods

3.1 Study Area:

The study area was delineated using the ecoregional classification of terrestrial ecosystems (Olson et al. 2001) in 10 countries: Mexico, Belize, Guatemala, Honduras El Salvador, Nicaragua, Costa Rica, Panama, Colombia and Ecuador, to encompass a total of 53 ecoregions. Marine areas were excluded from consideration (see Figure 1).

The northern boundary was defined by the transition between the Neartic and Neotropical biogeographic region in Mexico, including up to the Balsas Depression which is the most northern region with an important influence of tropical Mesoamerican elements (Morrone 2005). The southern limit of the study area was defined to include the ecoregions Eastern Cordillera Real montane forests, Northwestern Andean montane forests, and Western Ecuador moist forests that coincide with the geographical area where the northern Andes split into three separate ranges. This is the study area delineated by Sarkar et al. (2009) as the basis of several ongoing analyses of the region.

The extent of the area is approximately 1,659,847 km^2, between 21˚36’ north to 5˚1’ south, and from 71˚5’ to 103˚30’ west. Altitudinal range spans from sea level to snow peaks of more than

5,000 m in Mexico, Colombia and Ecuador. Total population of the region is approximately 121 million, 27 million (22%) living in rural areas.

3.2 Methods.

3.2.1 Fundamental Objectives and Scenario building

In order to represent three fundamental objectives in conservation planning: i) representation of biodiversity, ii) reducing threat to biodiversity, and iii) increasing implementation (Margules & Sarkar 2007), we constructed three different scenarios that were related to the different fundamental conservation objectives. Because we are working with scenarios we won´t formally integrate the fundamental objectives but only explain how they can be use together using a qualitative integration of what happens in each scenario.

For each fundamental objective we construct the objectives hierarchy, relating means objectives with a given ser of alternative biological, environmental and non biological attributes which can be easily measure (Figures 2-4).

3.2.3. Data

Variables used in the analysis included data from three sources: species distribution models models; environmental domains; and non biological variables.

Distribution models and Environmental Domains:

For the construction of the species distribution models and the environmental domain classifications, the following environmental variables were used: elevation, slope, aspect and 19 bioclimatic variables from WorldClim (Hijmans et al. 2005). Bioclimatic variables from WorldClim are calculated from monthly temperature and rainfall values, representing annual trends, seasonality and extreme or limiting environmental factors (Hijmans et al. 2005). Elevation was obtained from the U.S. Geological Survey’s Hydro-1K DEM data set (USGS 1998), and slope and aspect were derived from the same dataset using the Spatial Analyst extension of ArcMap 9.0 (Esri 2004). All physical and environmental layers were resampled to a 0.02° resolution.

Point occurrence data for 313 species, 28 animals and 285 plants listed in the categories CR, EN and VU of the IUCN Red List were obtained from several scientific collections: Animal species data were obtained from MaNIS (http://manisnet.org), HerpNET (http://www.herpnet.org/), ORNIS, (http://olla.berkeley.edu/ornisnet/), and REMIB (Red Mundial de Información sobre Biodiversidad; http://www.conabio.gob.mx/remib/doctos/remib_esp.html) and Smithsonian National Museum of Natural History (http://www.mnh.si.edu/rc/). Plant species data were obtained from the Missouri Botanical Garden (http://mobot.mobot.org/W3T/Search/vast.html).

To build the species distribution models, the Maxent software package Version 2.2 (Phillips et al. 2006) was used. Maxent has shown to be robust for modeling distributions from presence-only data (Elith et al. 2006). Species distribution model accuracy was evaluated by constructing the models randomly using 75% of the available records, with the other 25% used for testing. Following Pawar et al.(2007) all selected distribution models for further analysis showed a AUC greater than 0.75 and a P-value lower than 0.05 (for the sensitivity and specificity tests). Expert knowledge was used to drop species distribution models which showed systematic over-prediction. All species distribution models were reclassified into 0 and 1values, using the training presence threshold (Phillips et al. 2006).

To develop a Environmental Domain Classification we used the software PATN v.3.11 (Belbin 1993). All physical and environmental variables were equally weighted, and the association measure Gower Metric standardized the variables allowing the combination of variables with different measurement units. A non-hierarchical clustering was performed using the ALOC algorithm (Trakhtenbrot & Kadmon 2006). A classification of 348 environmental domains was chosen given the ability of the different domains to represent WWF ecoregions (Londoño-Murcia et. al. in press).

Socioeconomic, land use and risk variables:

Land use-cover data were obtained from the Global Land Cover 2000 database (Eva et al. 2003; Latifovic et al. 2003). Data were reclassified into three categories: urban land, cultivated land and natural vegetation (Supplementary Material Table 1).

Natural disaster probabilities for Cyclones, Draught events, Floods and Landslides were obtained from Columbia University (CHRR 2005a, b, c, 2005), for the period 1 January 1980 through 31 December 2000, with a resolution of 2.5 by 2.5 minute grid. Cells were classified into 10 classes, with higher values meaning a greater frequency of the hazard occurring.

Population Growth rate was calculated as:

$$\left(\frac{1}{t2-t1}\right)\ln\left(\frac{A2}{A1}\right)$$

where, t2 equals human population in the year A1 = 2015, and t1 equals human population in the year A2= 2005. Projection data for 2005 and 2015 was obtained from Gridded Population of the World (CIESIN 2005) which consists of estimates of human population for the years 1990, 1995 and 2000 with a resolution of 2.5 by 2.5 minute grid.

To address poverty and welfare aspects we used Infant Mortality rates obtained from the Center for International Earth Science Information Network from Columbia University as a proxy. These data were used as proxies for poverty and welfare rather than direct measures, such as gross domestic product or population living on less than one U.S. dollar per day, because these indicators are not available at a subnational level for most countries. Data are presented in 2.5 minute resolution and represent percent of children, under the age of 5, for the year period: 1990-2002 (Storeygard et al. 2008) .

Global Patterns in Net Primary Productivity (NPP) and Global Patterns in Human Appropriation of Net Primary Productivity (HaNPP) data were obtained from the Columbia University Center for International Earth Science Information Network (CIESIN), grids have a resolution of 0.25 dd. NPP and HaNPP are measured in grams of carbon, HaNPP have data for 1995 and NPP use data obtained between 1982 and 1998 (Imhoff et al. 2004) . NPP in agricultural lands was used as a proxy for the acquisition cost of the land.

Distance to roads was calculated for each grid cell as the distance from the center of each grid cell to the nearest road from the road map of the Digital Chart of the World (scale 1:1,000,000) (DCW 1992); and Distance to settlements was calculated from the nearest settlement point presented in the Global Rural-Urban Mapping Project with data from the years 1990, 1995 and 2000 (CIESIN et al. 2004). Calculations were done using Near_Dist function of ArcMap 9.

Human Population was obtained from The LandScan 2007 Global Population Database, developed by Oak Ridge National Laboratory (ORNL) for the United States Department of Defense and has a 1 km resolution (ORNL 2007).

Natural Protected Areas were obtained from the World Database on Protected Areas 2007 developed by United Nations Environmental program and the International Union for Conservation of Nature (WDPA 2007).

3.4 Preparation of variables for modeling:

All variables were resampled to 2.5 minute grid resolution, for a total of 82,666 cells. Basic descriptive statistics for each variable were calculated (Supplementary Material Table 2), and Pearson Correlation was performed between all pairs of variables to evaluate independence between the variables used (Supplementary Material Table 3).

Threshold values used in the prioritization were obtained for the variables distance to roads and distance to settlements. Site selection was performed maximizing the distance to roads and the distance to settlements adding cells that had values greater than 30km and 80km respectively, because lower values were more commonly found in high human influence areas (Supplementary Material Figure 1).

Net Primary Productivity was minimized for cells having > 50% of cultivated area, and Infant Mortality was maximized over the upper quartile value in cells that had > 50% natural vegetation. For all other variables all cells were used in the selection process.

3.5 Site selection modeling:

The site selection for each scenario was done using the software ConsNet (Ciarleglio et al. 2009), which is a comprehensive software package to design and analyze conservation area networks. It is built on a Modular Abstract Self-Learning Tabu Search framework. Tabu search is a metaheuristic framework that relies on memory structures to organize and navigate the search space (Ciarleglio 2008).

Initial solutions were built to solve the minimum area problem for a 10% representation target for each of the 313 species and 348 Environmental Domains, and giving preference for adjacency between cells. ConsNet constructs 3 initial solutions based on three different algorithms, MDS2 adjacency, RF4 adjacency and ILV4 adjacency (Ciarleglio et al. 2008). Although these initial solutions are not optimal they allow a fast assessment of the problem (Ciarleglio et al. 2008) and provide the maximum and minimum values for each variable that are needed in the multicriteria analysis. As these initial solutions are the starting point for the search of the solution of the different scenarios, the threat and implementation scenarios do assure the appropriate representation of species and environmental domain distributions.

The best solution for each scenario is obtained using a General Multi-Criteria Analysis (GMCA). ConsNet uses a modified Analytic Hierarchy Process (AHP) (Moffett et al. 2006), to create a general multi-criteria objective function. The modification makes the process consistent with multi-attribute value theory (MAVT) (Moffett et al. 2006) and, thus, with classical economic analysis. (The results are different from what would be obtained using the AHP but the transparent elicitation process of the AHP can still be used.) The GMCA objective is a weighted linear combination of several variables, were maximization or minimization of each variable is indicated by the user, and a fixed minimum and maximum values are used to scale the variable values, as they represent the range of acceptable values obtained from the initial solution. Maximum and minimum values of variables correspond to the sum of the values of the selected cells in the solution. To search for the maximization of a variable, the input minimum value in the GMCA was the maximum value obtained in the initial solution, and the maximum value in the GMCA was the total variable sum across all cells in the analysis. To search for the minimization of a variable, the maximum values in the GMCA was the minimum value obtained in the initial solutions, and the minimum values in the GMCA was either 0 if NPA were not included, or the sum per variable in the cells that correspond to the NPA.

The importance of each variable was assessed through a series of pair wise comparisons, assigning equal weights to each variable. Using the weights and the range defined for each variable (maximum and minimum values), a score was assigned to each solution. Higher scores represented better solutions. The search for the best solution started from the initial solution with the highest score and lasted 86,000 interactions. In a typical iteration one cell in the analysis area changes status (from off to on, or on to off), therefore a thorough search will have at least as many iterations as there are cells in the analysis area (Ciarleglio et al. 2008).

For each scenario as well as for each variable, the two best solutions were searched independently, one including NPAs and one excluding NPAs. All best solutions should meet a 10% representation target for each of the 313 species and 348 Environmental Domains, while minimizing the perimeter/area relationship. For all best solutions except for the Scenario 1 (biodiversity objective), area was fixed to the minimum value obtained in the initial solutions, while for Scenario 1 a search was done in order to obtain the minimum possible area (Figure 5).

4. Results:

4.1. Performance of NPAs

When analyzing the overlap of NPAs with area selected in scenarios constructed without NPAs (Figure 6), for all the countries the Scenario 1 (biodiversity) was the least coincident with the existing NPAs. Across all three scenarios Belize was the country showing the highest overlap, in particular for the implementation scenario where there was a 100% agreement. In general México and Colombia were the countries with the lowest overlap, with less than 25% of coincidence for all scenarios. NPAs in Belize, Ecuador and Nicaragua coincided to a greater degree with areas selected for Scenario 3 (implementation), while Honduras, El Salvador and Mexico NPAs showed a greater coincidence with areas selected for Scenario 2 (threat).

NPAs in general show a low efficiency in representing the 313 endangered species and the environmental domains (climatic and topographic diversity), and do not meet the 10% target representation. Solutions meeting the 10% target representation generated without including NPAs are less costly in area (176,736 km^2 SD 6,520.2km^2) than the NPA network (356,027 km^2) itself. NPAs however, performed well for variables to be maximized such as Cyclone and Landslide risk, and variables to be minimized such as NPP and Human population (Table 1).

4.3. Complementation of the NPA network

To fulfill the 10% target of biological representation not achieved by the NPAs, additional areas had to be chosen. Figures 7 through 9 show the additional areas selected for each of the three scenarios where NPAs were included. The complementary areas selected in addition to the NPAs did also permit to maximize Infant Mortality, Cyclones, Flood hazards and Drought risks, but failed to do so for the variables of Landslides, low human influence vegetation, and distance to roads and settlements. For variables to be minimized, the additional areas did so only for Human population growth and HaNNP, but failed to do so for NPP, Human impact and Total human population (Table 1).

When analyzing all selected additional areas to the NPAs, Belize, Costa Rica, Honduras, Nicaragua and Panama showed a greater percentage of selected sites occurring in natural vegetation. For Colombia the opposite was true, with the greatest percentage of the selected areas is located in cultivated areas (Table 2).

From the 53 ecorregions that encompass the study area, 13 needed more than 10% of their area represented in additionaly selected areas in order to complement the existing NPAs (Table 3). Five of these13 ecoregions, the Magdalena Valley Dry Forests, Magdalena Uraba Moist Forests and Patía Valley Dry Forests exclusively found in Colombia, and the Southern Pacific Dry Forests in Mexico, and the Chiapas Depression Dry Forests in Mexico and Guatemala had more than 50% of the area selected in areas with human transformed vegetation (Table 3). The highest proportion of selected areas in human transformed vegetation were the Sierra de los Tuxtlas in Mexico and the Motagua Valley Thornscrub in Guatemala ecoregions with 85%. On the other hand, the Miskito Pine Forest in Nicaragua and Honduras, the Balsas Dry Forest, the Northern Mesoamerican Pacific Mangroves and the Sierra Madre del Sur Pine-Oak forest in Mexico had more than 60% selected in natural vegetation areas (Table 3).

5. Discussion

The results of this study are an example of how conservation planning can integrate socio-economic variable without compromising efficiency and the adequate representation of biodiversity. Regional biodiversity planning can take advantage of complementarity in a way that promotes “regional sustainability” as a balance between competing needs of society, allocating different areas to different land uses as to maximize net benefits to society, where two or more “services” con be met in a single area (Faith & Walker 2002).

We did not find an optimal solution. Scenarios built showed different performance across variables, which should be reflecting the complex interaction and trade offs between socio-economic, land use and biological diversity (Geist & Lambin 2002; Lambin et al. 2001; Luck 2007). Increasing our understanding of the dynamic interaction between socio-economic data, land use and biological persistence, should help assigning better threshold values and weights to the different variables, which would improve the outcomes. However it remains to be seen if achieving a proper quantitative knowledge about relationships between the variables is possible given the complex of landscape processes. In the lack of such knowledge we used threshold values obtained from the initial solutions, in order to obtain better solutions for different criteria starting from the most efficient solution based on pure biological and environmental data. To our knowledge this is the most practical way to bypass the lack of information while assuring that threshold values of variables do not affect the representation of biodiversity.

Better data and better understanding of landscape process are possible at a local scale, ultimately conservation implementation has to be done at a local scale. Regional scale planning highlights the potential of sites at a regional level for trade-offs, making it a higher global priority for conservation efforts. Determining which areas in a region are priorities for land use management and landuse allocation (Faith & Walker 2002), can help identify which local areas within the region are priority areas for protection of biodiversity, poverty reduction or climate change responses. Our study provides a framework for sustainability, both across sectors and across spatial scales identifying areas that are globally and regionally important, providing a link between planning levels (Faith & Walker 2002) .

We identify several areas with a high priority for allocating resources to develop conservation projects at local scales, providing a link between network design and management options, by including a regional perspective for site selection where management options range from pure natural protection strategies, to climate change mitigation projects and conservation initiatives with high emphasis on sustainable livelihoods.

The selected areas in the Andes region and northern Mesoamerica region, suggest the importance of developing conservation projects that strategically include the management of rural landscapes. The representation of environmental diversity in these countries needs to include human modified landscapes to a substantial degree to meet the conservation targets (Sarkar et al. 2009).

Natural protected area networks are unlikely to protect more than a tiny fraction of Earth's biodiversity over the long run (Chan & Daily 2008; Ervin 2003; Rodrigues et al. 2004). Conservation planning needs to get involved in local and regional planning, to overcome the perspective of the NPA network being the only conservation alternative, and include all possible conservation management options in rural landscapes, which will provide ecological services (Geneletti 2007) and biodiversity protection (Cunningham et al. 2008; Chan & Daily 2008).

Transformed landscapes in Northern Mesoamerica and Tropical Andes show good opportunities for the involvement of people in conservation initiatives, where locals can get economic benefits and at the same time help improving habitat quality. Although society as a whole becomes more urban, poverty remains concentrated in rural areas, spatially, rural poverty is often highest in the places where biodiversity is greatest (Naughton-Treves et al. 2005). As a result of their geographic location, their vulnerability to environmental hazards and their direct reliance on ecosystem services, poorer communities in the poorest countries are going to be the most affected by global warming and related climate impacts such as wildfires and floods (IPCC 2007). Conservation management options for the site selected in the implementation scenario can help poor people to adapt to and mitigate the effects of climate change, at the same time programs such as Payment for Environmental Services (PES) or Reducing Emissions from Deforestation and Degradation (REDD) can be used as principal tools for countryside conservation (Pagiola et al. 2002) and implementing sustainable livelihoods strategies (Mehta &

Kill 2007). In that sense the climate change agenda can be seen as an opportunity to bridge biodiversity conservation and poverty alleviation, by searching for sustainable solutions (Roe 2008).

In the rest of Mesoamerica and Chocó regions, conservation actions can be developed to a larger extent towards protection of natural landscapes remnants, given the higher proportion of selected areas in low human influence vegetation. Again PES and REDD strategies can be implemented, as well as new NPA established.

The reason that in our results the established NPAs had a better performance for the implementation and threat scenarios in some countries can be explained because they were selected in poor productivity areas, far away from development centers (Naughton-Treves et al. 2005), and for having a higher proportion of low human influence vegetation. In that respect NPAs seem to do their job well with the majority of parks successfully stopping land clearing (Aaron et al. 2001). However, although deforestation rates are lower inside NPAs, habitat loss in buffer zones outside NPA is increasing the isolation of protected areas and may prevent them from functioning as an effective network (Sánchez-Azofeifa et al. 2003). Viewing NPA as an "intact" area that can provide vital ecosystem services and help contribute to the quality and restoration of surrounding areas, but more attention is needed toward effective management outside NPA (Naughton-Treves et al. 2005).

Although there has been a significant increase in NPAs during the last 20 years (Naughton-Treves et al. 2005), the low efficiency of NPAs that our results show for countries other than Belize in the region is a matter of worry, since these countries are priority areas for global conservation strategies, and therefore to have adequate representation in NPAs is much desired. Other works have identify areas in order to improve NPA representation of biodiversity that are similar to the areas obtain in this work for the biodiversity scenario: Chiapas Depression dry forest, Southern Pacific dry forest north of the NPA Laguna de Chacahua, Balsas dry forest, Sierra Madre del Sur pine-oak forest and northern Peten Veracruz moist forest in Mexico (Sarkar et al. 2009); Cuenca de Polochic-Sierra de las Minas-Motagua, Rio Platano-Caratasca-Savannas, Cordillera Volcánica Occidental and Talamanca-Caribe in Central America (Calderón et al. 2004); Central American pine-oak forest and Central American montane forest in

Guatemala (Sarkar et al. 2009). Magdalena Valley dry forest, Magdalena Urabá moist forest and Chocó-Darién moist forest in Colombia (Sarkar et al. 2009); Center and Southern Western Ecuador moist forest , south western Eastern Cordillera real montane forest, northern Northwestern Andean montane forest and a sector in Nosrwestern Andean montane forest between the NPA Los Illinizas and NPA Cotacahi-Cayapas in Ecuador (Sierra et al. 2002); comprising priority areas for conservation in the Mesoamerica, Tropical Andes and Chocó hotspots.

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Figure legends:

Figure 1. Location of the study area showing country borders and Natural Protected Areas (IUCN categories I-V).

Figure 2: Objectives hierarchy developed for the fudamental objective: Representation of biodiversity.

Figure 3: Objectives hierarchy developed for the fudamental objective: Reducing threat to biodiversity.

Figure 4: Objectives hierarchy developed for the fudamental objective: Increasing Implementation.

Figure 5. Variables used in the construction of the different scenarios and their threshold values, and maximization or minimization goal of variables.

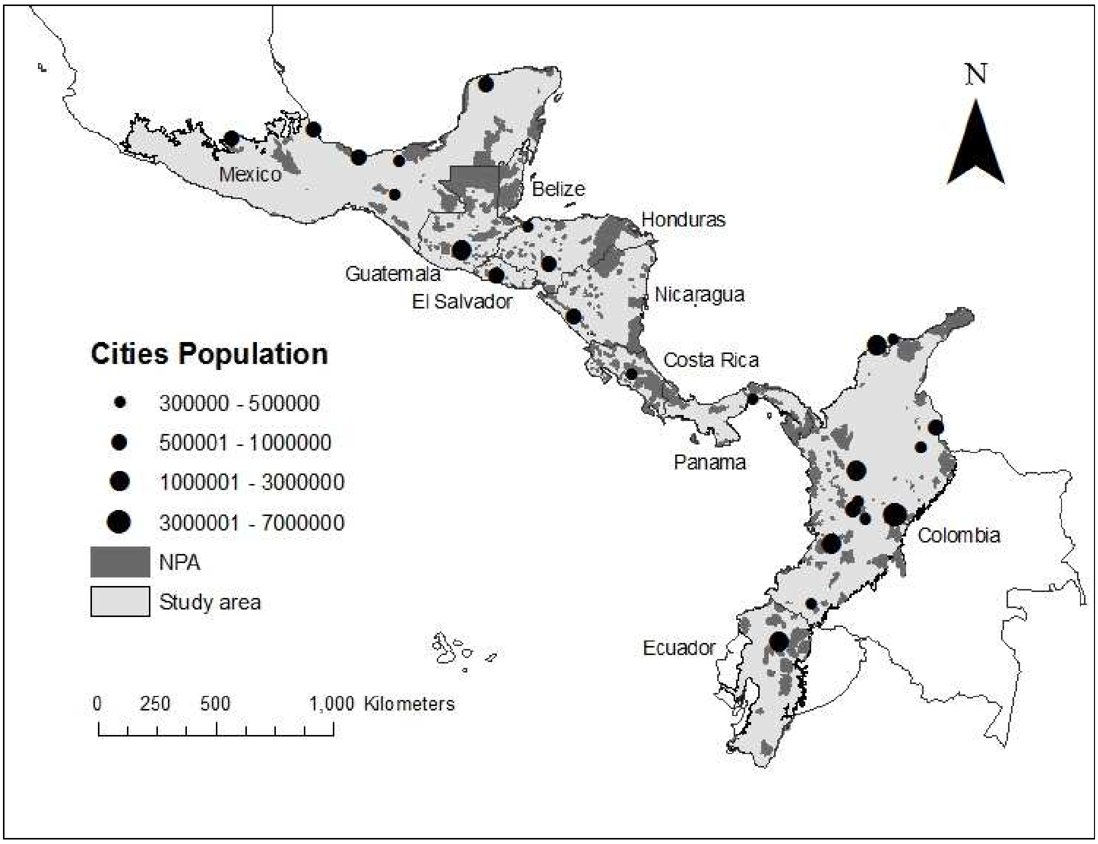
Figure 6. Percentage overlap between the existing NPAs and the selected areas for the three scenarios built by excluding NPAs

Figure 7. Site selected in the solution of the threat scenario including NPA. Gray shows the decreed Natural Protected Areas and black areas include site selected for the threat scenario solution that complemented the NPA. Site selection for Low human influence vegetation, sites with mayor distance to roads and settlements and site selected for minimizing Human Impact are shown.

Figure 8. Sites selected in the solution of the Implementation Scenario including NPA. Gray shows the decreed Natural Protected Areas, and black areas include sites selected in the scenario

solution that complement the NPAs. Site selected to maximize Infant Mortality, minimize NPP in cultivated areas, and maximize Drought, Flood and Landslide hazards are shown in colors.

Figure 9. Site selected in the solution of the Biological scenario including NPA. Gray shows the decreed Natural Protected Areas; green areas are sites selected in the Biodiversity Scenario solution that complement the NPAs. Sites selected to maximize Biological representation are shown in red.



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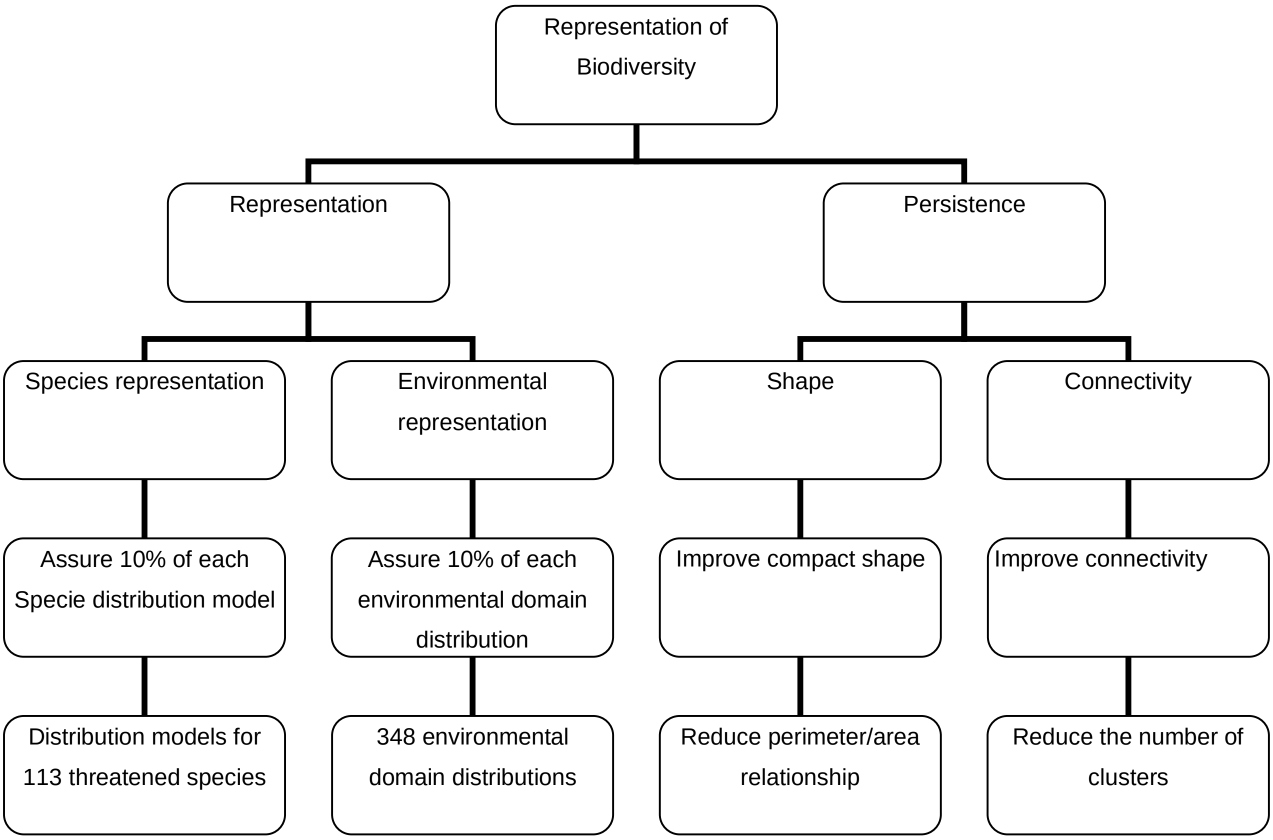


Figure 2: Objectives hierarchy developed for the fudamental objective: Representation of biodiversity.

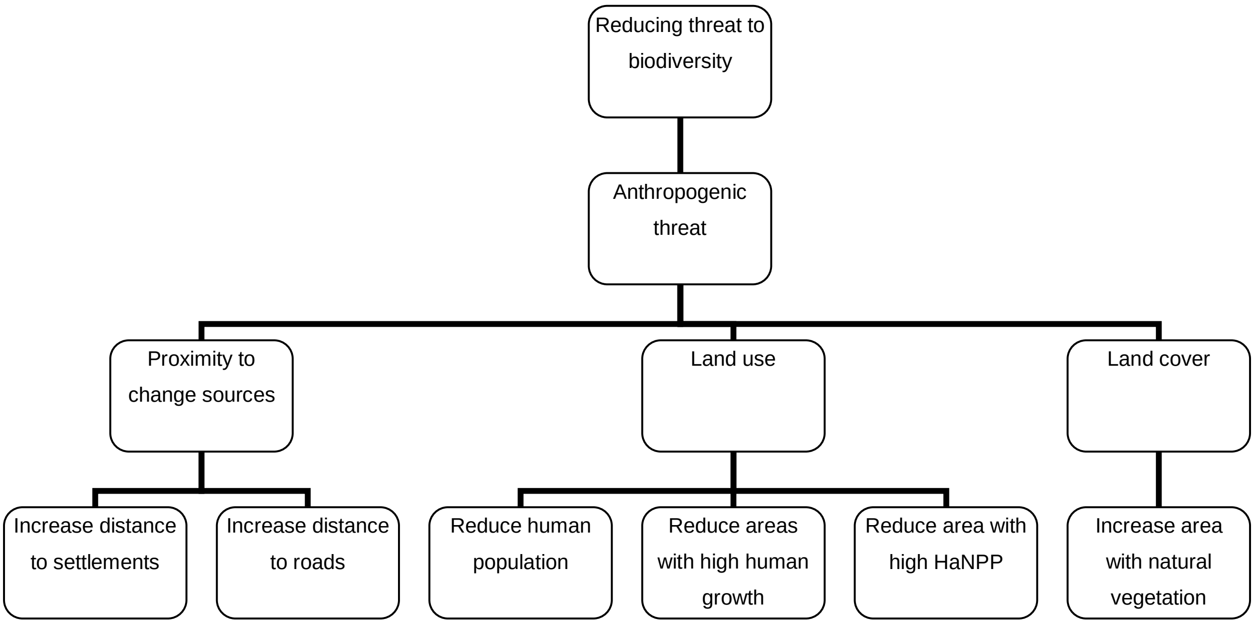
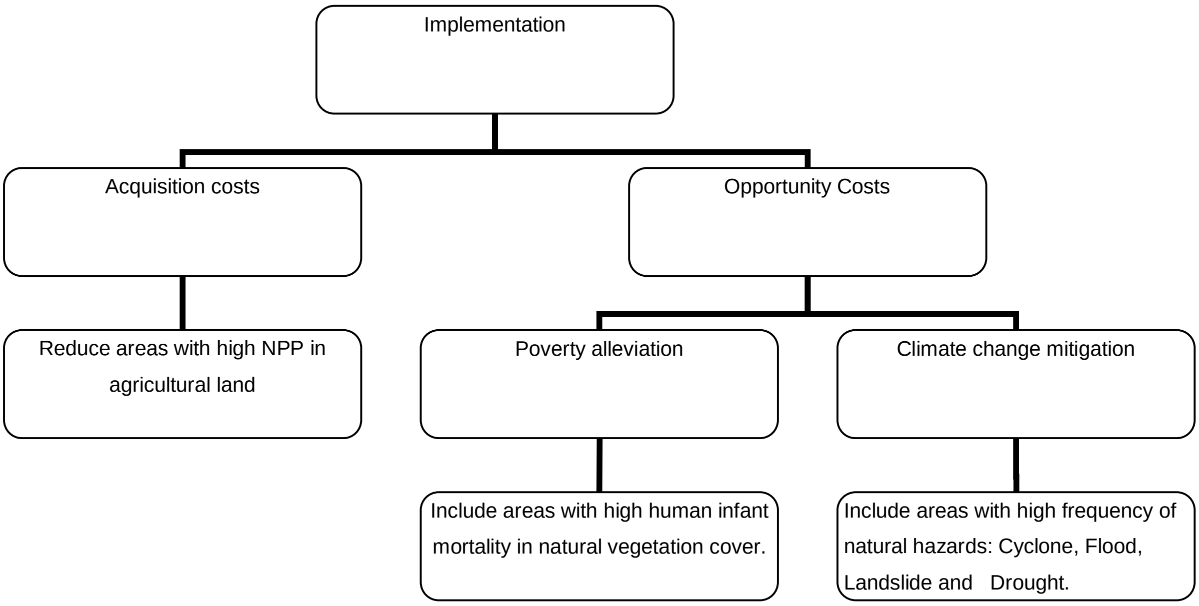


Figure 3: Objectives hierarchy developed for the fudamental objective: Reducing threat to biodiversity.



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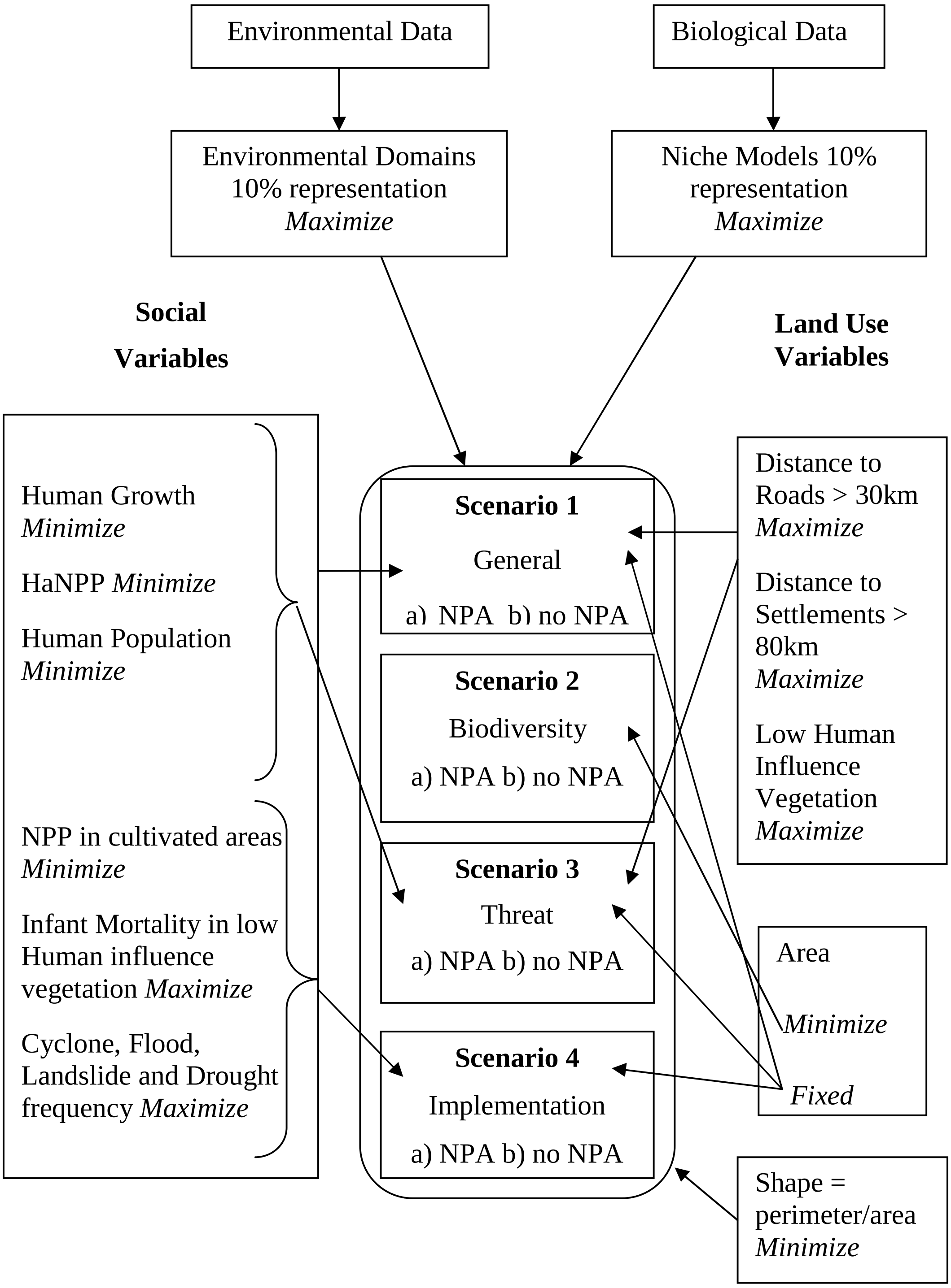

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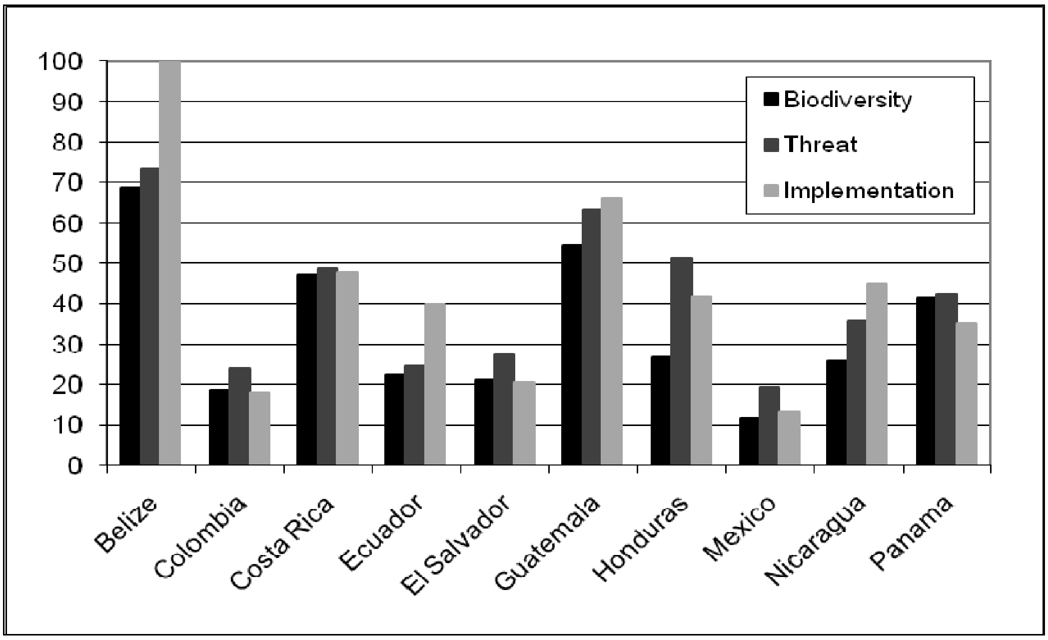

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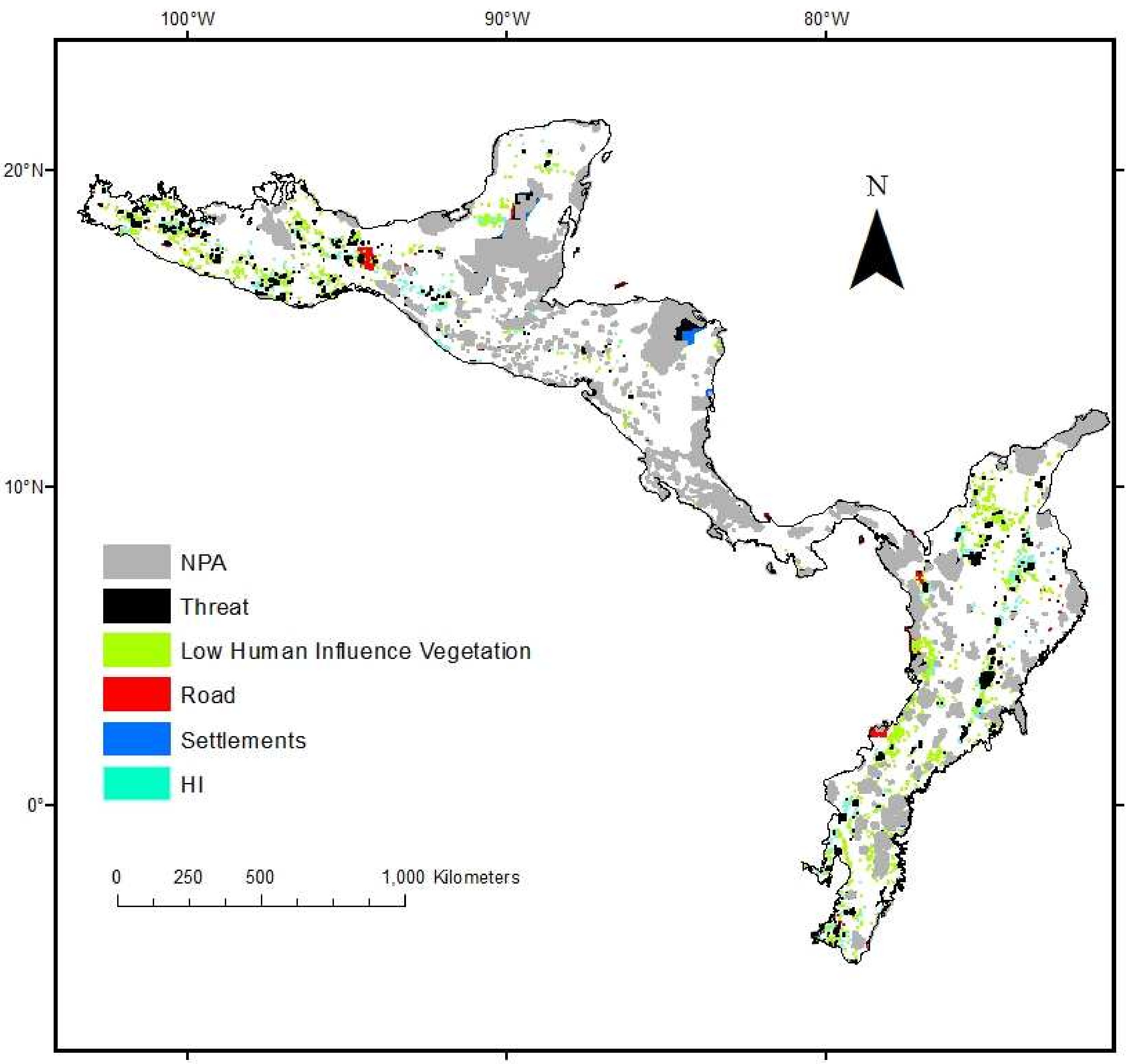

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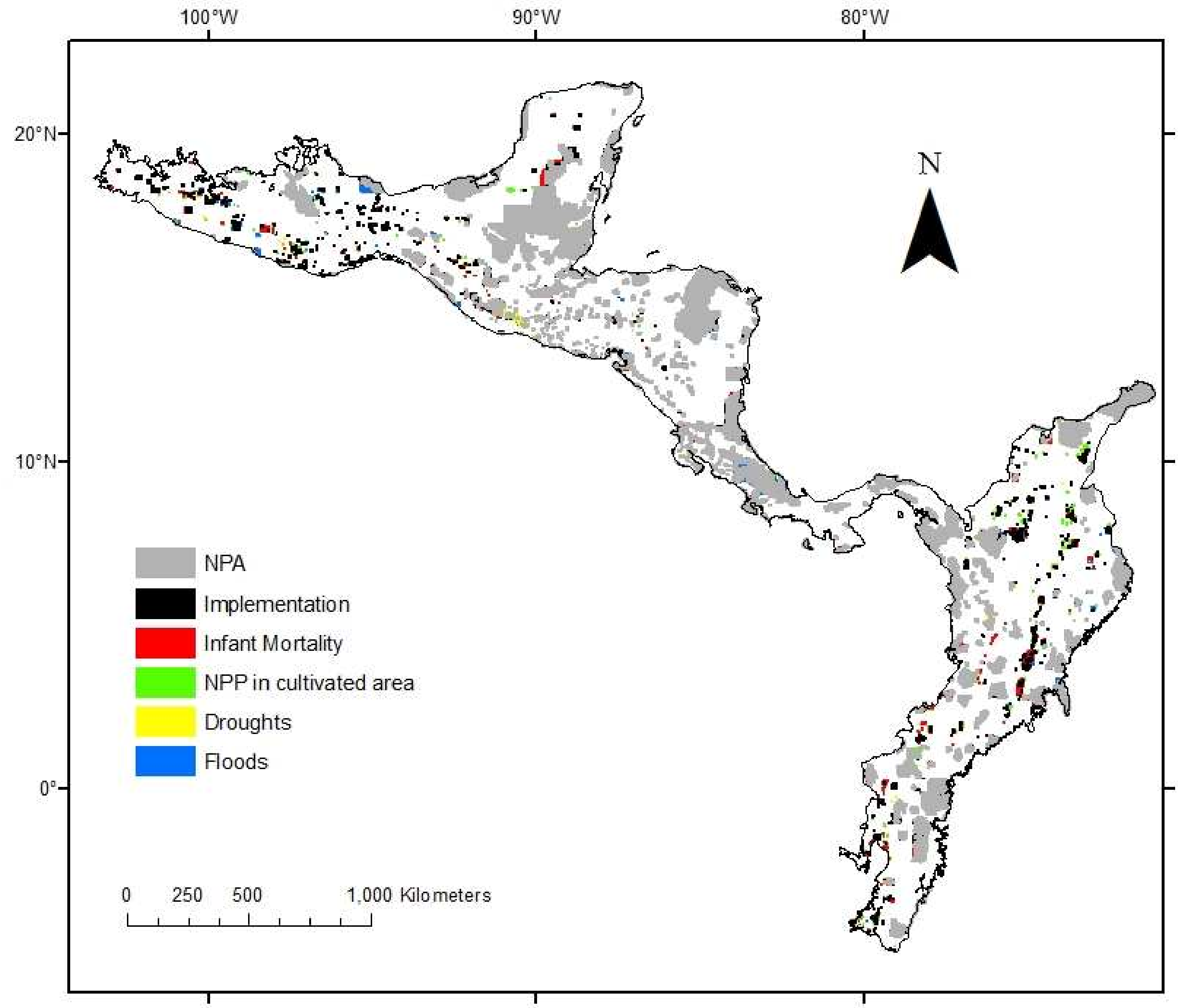

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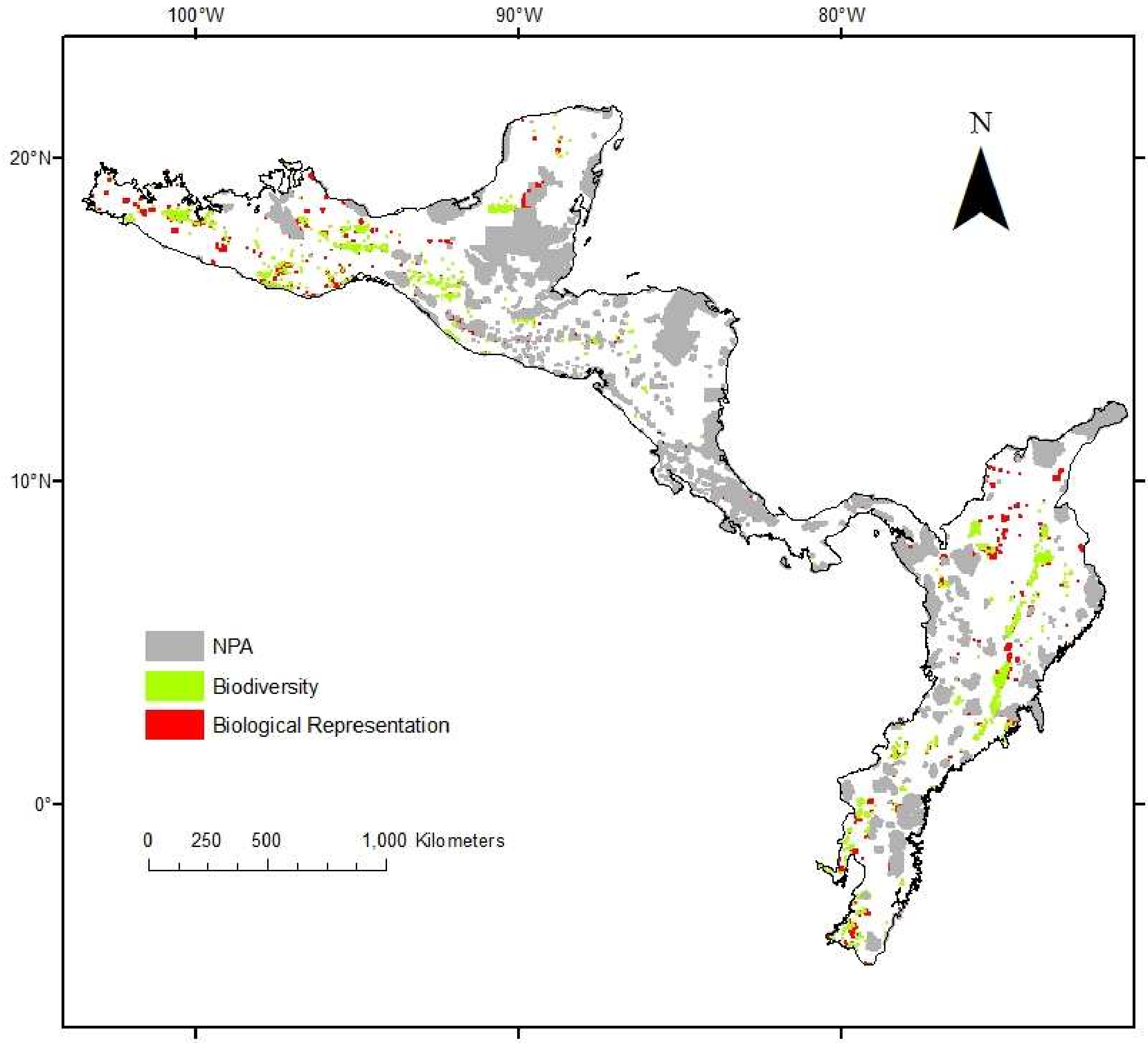

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Table 1. Mean values obtained for each variable in the best solution for each of the three scenarios, with and without NPAs.

		Solution excluding NPAs			Solution including NPAs		
	NPA	Biodiversity	Threat	Implementation	Biodiversity	Threat	Implementation
Distance to roads	11	8	15	11	10	11	10
Distance to settlements	43	32	44	38	41	43	42
Human Growth	0.02	0.01	0.01	0.01	0.02	0.02	0.02
HaNPP	8.48E+10	8.48E+10	6.03E+10	7.33E+10	8.44E+10	8.12E+10	8.35E+10
NPP	6.10E+11	6.42E+11	6.37E+11	6.34E+11	6.11E+11	6.13E+11	6.10E+11
Human Population	677.47	1188.26	847.82	1235.34	784.97	721.22	709.56
Low Human Influence vegetation	15.48	12.96	13.57	15.47	14.87	15.04	15.11
Infant Mortality (>50% low human influence vegetation)	404.82	408.13	413.08	430.66	405.06	405.06	406.08
Cyclone Frequency	3.12	2.60	2.80	2.59	3.10	3.14	3.18
Draught Frequency	4.59	4.14	4.34	4.26	4.56	4.59	4.59
Floods Frequency	7.06	7.73	7.36	7.80	7.19	7.14	7.18
Landslides Frequency	7.27	7.14	7.19	7.16	7.24	7.22	7.25
Shape= perimeter/area	0.14	0.36	0.35	0.20	0.16	0.14	0.14
Area	356027.59	166,955.86	179,999.89	179,992.31	389,358.66	399,997.23	399,999.78

Table 2: Percentage of area selected for complementing the NPA in all scenarios that corresponded to low human influence vegetation and to cultivated areas

	Total (Km^2)	Nat_Veg (%)	Int_Veg (%)
Belize	**603**	84,77	15,22
Belizian pine forests	116	83,42	16,57
Central American Atlantic moist forests	0	0	0
Mesoamerican Gulf-Caribbean mangroves	41	99,44	0,55
Petén-Veracruz moist forests	445	83,75	16,24
Yucatán moist forests	0	0	0
Colombia	**35938**	38,70	61,29
Amazon-Orinoco-Southern Caribbean mangroves	120	32,97	67,02
Catatumbo moist forests	447	31,74	68,25
Cauca Valley dry forests	297	25,74	74,25
Cauca Valley montane forests	422	41,07	58,92
Central American dry forests	0	0	0
Chocó-Darién moist forests	3046	77,32	22,67
Cordillera Oriental montane forests	2078	52,96	47,03
Eastern Cordillera real montane forests	239	62,85	37,14
Eastern Panamanian montane forests	107	82,23	17,76
Guajira-Barranquilla xeric scrub	1757	30,48	69,51
Magdalena Valley dry forests	*7109*	*29,24*	*70,75*
Magdalena Valley montane forests	2697	45,10	54,89
Magdalena-Urabá moist forests	*13647*	*29,87*	*70,12*
Northern Andean páramo	319	87,17	12,82
Northwestern Andean montane forests	781	64,74	35,25
Patía Valley dry forests	*495*	*34,94*	*65,05*
Santa Marta montane forests	91	60,30	39,69
Santa Marta páramo	0	0	0
Sinú Valley dry forests	1583	16,79	83,20
South American Pacific mangroves	411	95,43	4,56
Western Ecuador moist forests	285	71,56	28,43
Costa Rica	**732**	75,48	24,51
Central American dry forests	64	55,26	44,73
Costa Rican seasonal moist forests	246	72,03	27,96
Isthmian-Atlantic moist forests	213	86,86	13,13
Isthmian-Pacific moist forests	152	70,40	29,59
Mesoamerican Gulf-Caribbean mangroves	0	0	0
Southern Mesoamerican Pacific mangroves	0	0	0
Talamancan montane forests	56	84,42	15,57
Ecuador	**11968**	50,51	49,48
Cordillera Central páramo	**0**	0	0
Eastern Cordillera real montane forests	5453	67,02	32,97

Northern Andean páramo	410	81,62	18,37
Northwestern Andean montane forests	2021	51,29	48,70
South American Pacific mangroves	0	0	0
Western Ecuador moist forests	*4083*	*24,94*	*75,05*
El Salvador	**175**	58,76	41,23
Central American dry forests	90	53,24	46,75
Central American montane forests	20	78,27	21,72
Central American pine-oak forests	62	61,51	38,48
Sierra Madre de Chiapas moist forests	0	0	0
Southern Mesoamerican Pacific mangroves	2	22,92	77,07
Guatemala	**3965**	54,33	45,66
Central American Atlantic moist forests	229	62,48	37,51
Central American dry forests	378	42,23	57,76
Central American montane forests	634	89,59	10,40
Central American pine-oak forests	1328	63,62	36,37
Chiapas Depression dry forests	*121*	*48,41*	*51,58*
Chiapas montane forests	17	0	100
Mesoamerican Gulf-Caribbean mangroves	0	0	0
Motagua Valley thornscrub	*600*	*12,26*	*87,73*
Petén-Veracruz moist forests	174	65,20	34,79
Sierra Madre de Chiapas moist forests	335	45,92	54,07
Southern Mesoamerican Pacific mangroves	147	25,95	74,04
Yucatán moist forests	0	0	0
Honduras	**4990**	83,29	16,70
Central American Atlantic moist forests	2073	90,84	9,15
Central American dry forests	451	16,82	83,17
Central American montane forests	20	40,16	59,83
Central American pine-oak forests	537	75,20	24,79
Mesoamerican Gulf-Caribbean mangroves	181	91,61	8,38
Miskito pine forests	*1713*	*94,22*	*5,77*
Southern Mesoamerican Pacific mangroves	12	29,59	70,40
Mexico	**42195**	62,98	37,01
Balsas dry forests	*8103*	*60,68*	*39,31*
Belizian pine forests	0	0	0
Central American dry forests	64	45,29	54,70
Central American pine-oak forests	928	31,58	68,41
Chiapas Depression dry forests	*1662*	*24,25*	*75,74*
Chiapas montane forests	146	46,91	53,08
Chimalapas montane forests	0	0	0
Mesoamerican Gulf-Caribbean mangroves	137	82,95	17,04
Northern Mesoamerican Pacific mangroves	*84*	*78,17*	*21,82*
Oaxacan montane forests	*775*	*60,18*	*39,81*
Pantanos de Centla	169	37,25	62,74
Petén-Veracruz moist forests	7649	46,44	53,55

Sierra de los Tuxtlas	*391*	*12,01*	*87,98*
Sierra Madre de Chiapas moist forests	377	54,49	45,50
Sierra Madre de Oaxaca pine-oak forests	549	71,80	28,19
Sierra Madre del Sur pine-oak forests	*6373*	*85,76*	*14,23*
Southern Mesoamerican Pacific mangroves	10	1,49	98,50
Southern Pacific dry forests	*6298*	*44,26*	*55,73*
Tehuacán Valley matorral	75	100	0
Veracruz dry forests	445	48,49	51,50
Yucatán dry forests	1751	95,19	4,80
Yucatán moist forests	6202	92,56	7,43
Nicaragua	**2304**	79,52	20,47
Central American Atlantic moist forests	818	86,96	13,03
Central American dry forests	312	39,13	60,86
Central American montane forests	2	94,15	5,84
Central American pine-oak forests	218	47,35	52,64
Costa Rican seasonal moist forests	33	98,76	1,23
Isthmian-Atlantic moist forests	491	99,17	0,82
Lake	97	100	0
Mesoamerican Gulf-Caribbean mangroves	54	100	0
Miskito pine forests	*276*	*79,88*	*20,11*
Southern Mesoamerican Pacific mangroves	0	0	0
Panama	**942**	70,91	29,08
Chocó-Darién moist forests	218	92,47	7,52
Eastern Panamanian montane forests	3	100	0
Isthmian-Atlantic moist forests	365	82,58	17,41
Isthmian-Pacific moist forests	307	39,44	60,55
Mesoamerican Gulf-Caribbean mangroves	1	100	0
Panamanian dry forests	0	0	0
South American Pacific mangroves	22	91,91	8,08
Southern Mesoamerican Pacific mangroves	0	0	0
Talamancan montane forests	21	73,46	26,53

Table 3. Ecoregions area and percentage of area selected for complementing the NPA.

Ecoregion	Total Area (Km^2)	% selected
Magdalena Valley dry forests	19635	36,2
Motagua Valley thornscrub	2336	25,6
Patía Valley dry forests	2270	21,8
Magdalena-Urabá moist forests	76741	17,7
Southern Pacific dry forests	41790	15,0
Balsas dry forests	62441	12,9
Western Ecuador moist forests	33886	12,8

Chiapas Depression dry forests	14021	12,7
Northern Mesoamerican Pacific mangroves	720	11,7
Miskito pine forests	18053	11,0
Sierra Madre del Sur pine-oak forests	61173	10,4
Sierra de los Tuxtlas	3825	10,2
Oaxacan montane forests	7600	10,2
Yucatán moist forests	69533	8,9
Eastern Cordillera real montane forests	76445	7,4
Veracruz dry forests	6610	6,7
Catatumbo moist forests	6764	6,6
South American Pacific mangroves	6628	6,5
Guajira-Barranquilla xeric scrub	27462	6,3
Sinú Valley dry forests	24980	6,3
Sierra Madre de Chiapas moist forests	11258	6,3
Petén-Veracruz moist forests	148770	5,5
Central American montane forests	13299	5,0
Amazon-Orinoco-Southern Caribbean mangroves	2509	4,8
Chocó-Darién moist forests	72846	4,4
Belizian pine forests	2830	4,1
Cauca Valley dry forests	7344	4,0
Sierra Madre de Oaxaca pine-oak forests	14345	3,8
Eastern Panamanian montane forests	3044	3,6
Cordillera Oriental montane forests	58681	3,5
Yucatán dry forests	49723	3,5
Central American Atlantic moist forests	89473	3,4
Northwestern Andean montane forests	81164	3,4
Chiapas montane forests	5778	2,8
Central American pine-oak forests	111342	2,7
Southern Mesoamerican Pacific mangroves	6281	2,7
Costa Rican seasonal moist forests	10628	2,6
Magdalena Valley montane forests	105053	2,5
Northern Andean páramo	29635	2,4
Central American dry forests	67519	2,0
Mesoamerican Gulf-Caribbean mangroves	21545	1,9
Santa Marta montane forests	4784	1,9
Isthmian-Atlantic moist forests	57820	1,8
Isthmian-Pacific moist forests	28955	1,5
Cauca Valley montane forests	32055	1,3
Lake	8014	1,2
Pantanos de Centla	17082	0,9
Tehuacán Valley matorral	9892	0,7
Talamancan montane forests	16341	0,4
Chimalapas montane forests	2083	0
Cordillera Central páramo	500	0

Panamanian dry forests	5070	0
Santa Marta páramo	1243	0

Supplementary Material:

SM Table 1: Clasificación de uso del suelo utilizado en el análisis según clasificaciones de tipo de vegetación en Global Land Cover 2000.

Global Land Cover 2000 Classification	Classification unit in the analysis
Agriculture – intensive	cultivated land
Bamboo dominated forest	natural vegetation
Barren / bare soil	natural vegetation
Burnt area (resent burnt area)	áreas transformadas
Closed deciduous forest	natural vegetation
Closed evergreen tropical forest	natural vegetation
Closed montane grasslands	cultivated land
Closed semi deciduous forest	natural vegetation
Closed semi-humid forest	natural vegetation
Closed shrublands	cultivated land
Closed steppe grasslands	cultivated land
Consolidated Rock Sparse Vegetation	natural vegetation
Cropland	cultivated land
Cropland and Shrubland/woodland	cultivated land
Desert	natural vegetation
Forest plantations (Llanos of Venezuela)	cultivated land
Fresh water flooded forests	natural vegetation
Grass Savannah	natural vegetation
Herbaceous Wetlands	natural vegetation
Mangroves	natural vegetation
Montane forests > 1000m - open semi humid	natural vegetation
Montane forests >1000m - closed semi -deciduous	natural vegetation
Montane forests >1000m - open deciduous	natural vegetation
Montane forests >1000m - open evergreen	natural vegetation
Montane forests >1000m - open semi- deciduous	natural vegetation
Montane forests >1000m - transition forest	natural vegetation
Montane forests > 1000m - closed semi humid	natural vegetation
Montane forests > 1000m flooded forest	natural vegetation
Montane forests > 1000m flooded forest	natural vegetation
Montane forests > 1000m flooded forest	natural vegetation
Montane forests >1000m - bamboo dominated	natural vegetation
Montane forests >1000m - closed deciduous	natural vegetation
Montane forests >1000m - closed temperate deciduous	natural vegetation
Montane forests >1000m - dense evergreen	natural vegetation
Montane forests >1000m - open temperate deciduous	natural vegetation
Montane forests >1000m - temperate mixed	natural vegetation
Montane forests >1000m -temperate closed broadleaf	natural vegetation
Montane forests 500-1000 - closed semi humid	natural vegetation
Montane forests 500-1000 - open semi humid	natural vegetation
Montane forests 500-1000 - bamboo	natural vegetation

Montane forests 500-1000 - dense evergreen	natural vegetation
Montane forests 500-1000 - open evergreen	natural vegetation
Montane forests 500-1000m - flooded forest	natural vegetation
Montane forests 500-1000m - closed deciduous	natural vegetation
Montane forests 500-1000m - closed semi -deciduous	natural vegetation
Montane forests 500-1000m - closed temperate deciduous	natural vegetation
Montane forests 500-1000m - flooded forest	natural vegetation
Montane forests 500-1000m - flooded forest	natural vegetation
Montane forests 500-1000m - open deciduous	natural vegetation
Montane forests 500-1000m - open semi- deciduous	natural vegetation
Montane forests 500-1000m - open temperate deciduous	natural vegetation
Montane forests 500-1000m - temperate mixed	natural vegetation
Montane forests 500-1000m - transition forest	natural vegetation
Montane forests 500-1000m -temperate closed broadleaf	natural vegetation
Mosaic agriculture / degraded forests	cultivated land
Mosaic agriculture / degraded vegetation	cultivated land
Open deciduous forest	natural vegetation
Open evergreen tropical forest	natural vegetation
Open montane grasslands	natural vegetation
Open semi deciduous forest	natural vegetation
Open semi-humid forest	natural vegetation
Open shrublands	cultivated land
Open steppe grasslands	natural vegetation
Periodically flooded savannah	cultivated land
Permanent swamp forests	natural vegetation
Polar Grassland with a Dwarf-Sparse Shrub Layer	natural vegetation
Polar Grassland with a Sparse Shrub Layer	natural vegetation
Semi deciduous transition forest	natural vegetation
Shrub Savannah	natural vegetation
Snow and Ice	natural vegetation
Sparse desertic steppe shrub /grasslands	natural vegetation
Subpolar Needleleaved Evergreen Forest Open Canopy - lichen understory	natural vegetation
Temperate closed deciduous broadleaf forests	natural vegetation
Temperate closed evergreen broadleaf forest	natural vegetation
Temperate mixed evergreen broadleaf forests	natural vegetation
Temperate open deciduous broadleaf forests	natural vegetation
Temperate or Sub-polar Broadleaved Deciduous Forest - Closed Canopy	natural vegetation
Temperate or Subpolar Broadleaved Deciduous Shrubland - Open Canopy	natural vegetation
Temperate or Subpolar Broadleaved Evergreen Shrubland - Closed Canopy	natural vegetation
Temperate or Subpolar Grassland	natural vegetation
Temperate or Subpolar Grassland with a Sparse Shrub Layer	natural vegetation
Temperate or Subpolar Grassland with a Sparse Tree Layer	natural vegetation
Temperate or Sub-polar Mixed Broaddleleaved or Needleleaved Forest - Open Canopy	natural vegetation
Temperate or Sub-polar Mixed Broadleaved and Needleleaved Dwarf-Shrubland - Open Canopy	natural vegetation
Temperate or Sub-polar Mixed Broadleaved or Needleleaved Forest - Closed Canopy	natural vegetation

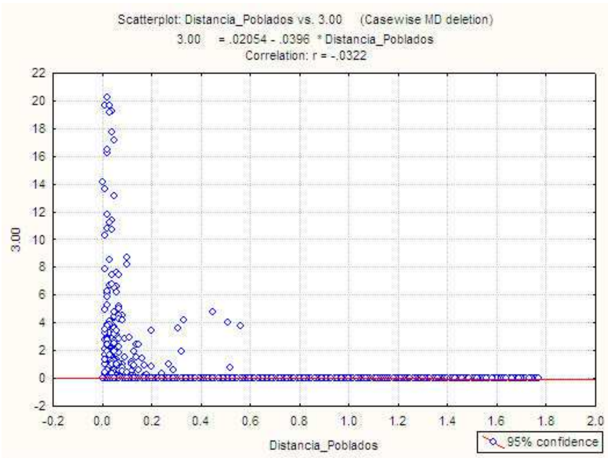
Temperate or Sub-polar Needleleaved Evergreen Forest - Closed Canopy	natural vegetation
Temperate or Sub-polar Needleleaved Evergreen Forest - Open Canopy	natural vegetation
Temperate or Subpolar Needleleaved Evergreen Shrubland - Open Canopy	natural vegetation
Temperate or Sub-polar Needleleaved Mixed Forest - Closed Canopy	natural vegetation
Tropical or Sub-tropical Broadleaved Deciduous Forest - Closed Canopy	natural vegetation
Tropical or Sub-tropical Broadleaved Evergreen Forest - Closed Canopy	natural vegetation
Tropical or Sub-tropical Broadleaved Evergreen Forest - Open Canopy	natural vegetation
Unconsolidated Material Sparse Vegetation (old burnt or other disturbance)	cultivated land
Urban	urban land
Urban and Built-up	urban land
Water bodies	natural vegetation
Wetlands	natural vegetation

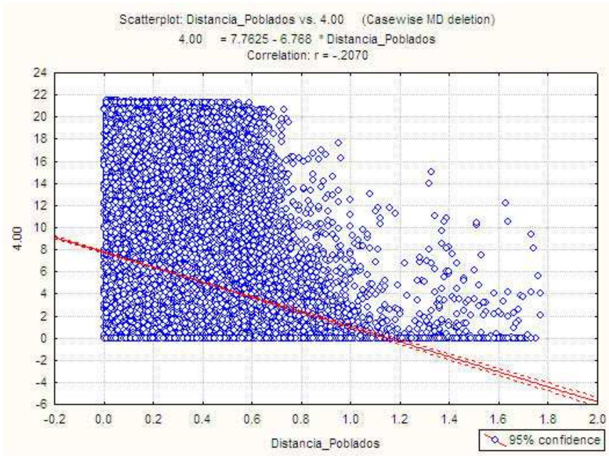
SM Table 2: Basic descriptive statistics for al variables

	N	Mean	Median	Mode	Min.	Max.	Lower	Upper	Std.Dev.
Distance to settlements (km)		30	24	13	0	181	13	41	24
Distancie to roads (km)	82666	7	4	na	0	89	2	9	9
Cyclone	31813	3.07	1.00	1.00	1.00	9.00	1.00	5.00	2.88
Flood	77630	7.70	8.00	8.00	1.00	10.00	7.00	9.00	2.25
Landslide	24589	6.98	7.00	6.00	6.00	10.00	6.00	8.00	1.08
Drought	62613	4.32	4.00	1.00	1.00	10.00	1.00	7.00	3.03
HaNPP	82270	9.47E+10	3.73E+10	1.38E+10	0.00	4.40E+12	1.57E+10	8.63E+10	2.53E+11
NPP	82666	6.28E+11	6.63E+11	0.00	0.00	1.05E+12	4.67E+11	8.45E+11	2.57E+11
Infant Mortality rate in non	77902	291.43	279.00	245.00	81.00	672.00	215.00	342.00	95.46
Upper quartile Infant	14739	409.54	393	342	342	672	364	459	56.17
LandScan	82666	1467.77	185.00	0.00	0.00	936883.00	43.00	668.00	12251.2
Human Growth	82666	0.01	0.01	0.00	-0.06	0.07	0.01	0.02	0.01
Natural vegetation (km^2)	82666	11.90	12.95	0.00	0.00	21.47	4.45	20.08	7.71

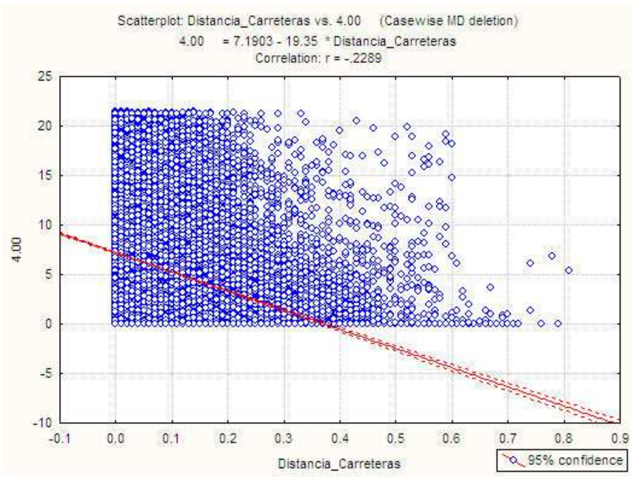
SM Table 3: Pearson Correlation performed between all pairs of variables.

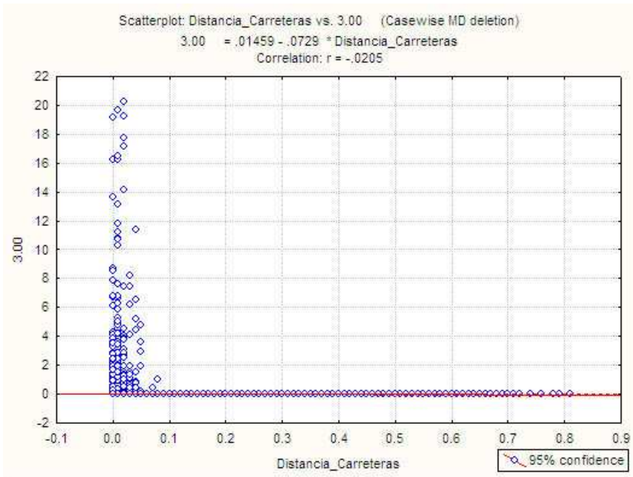
	Distance to Roads	HaNPP	NPP	HI	Infant Mortality	LandScan	Human Growth Index	Natural Vegetation	Human Influence Vegetation
Distance to settlements	.2990 p=0.00	-.3114 p=0.00	.1541 p=0.00	-.3962 p=0.00	.1235 p=0.00	-.1992 p=0.00	-.1329 p=0.00	.1808 p=0.00	-.1876 p=0.00
Distance to Roads	1.0000 p= ---	-.1593 p=0.00	.1083 p=0.00	-.4405 p=0.00	.0202 p=0.08	-.1150 p=0.00	-.0757 p=.000	.1458 p=0.00	-.1495 P=0.00
HaNPP		1.0000 p= ---	-.1130 p=0.00	.4099 p=0.00	-.0851 p=0.00	.4769 p=0.00	.2264 p=0.00	-.0508 p=0.00	.0602 p=0.00
NPP			1.0000 p= ---	-.0383 p=0.00	.3167 p=0.00	-.0496 p=0.00	.1625 p=0.00	-.2034 P=0.00	.1977 p=0.00
Human_Impact				1.0000 p= ---	-.1945 p=0.00	.3957 P=0.00	.2454 p=0.00	-.4954 p=0.00	.5076 p=0.00
Infant Mortality					1.0000 p= ---	-.0080 p=0.49	-.0315 p=0.00	-.0249 p=0.03	.0031 p=0.79
LandScan Human						1.0000 p= ---	.0956 P=0.00	-.0530 P=0.00	.0564 p=0.00
Human Growth Index							1.0000 p= ---	-.1029 p=0.00	.1200 p=0.00
Natural Vegetation								1.0000 p= ---	-.9992 p=0.00





b)





c) d)

Figure 1. Scatter plots showing the relationship between distance to roads and area of urban (a) and cultivated areas (b), and distance to settlements and area of urban (c) and cultivated areas (d).

Discusión General

Vacios de investigación para la selección de áreas prioritarias de conservación en Mesoamérica, Chocó y Andes Tropicales.

El trabajo para identificar áreas de conservación en las regiones de Mesoamérica, Los Andes Tropicales y el Chocó ha estado activo desde 1990, con gran interés no sólo por parte de instituciones nacionales sino también de organizaciones internacionales. La participación de ONG´s internacionales es muy relevante pues políticamente juegan un papel importante en la implementación de áreas de conservación (Jepson 2005).

El aumento en el número de publicaciones sobre selección de áreas de conservación a partir del año 2002, demuestra el desarrollo académico sobre la materia. Cabe resaltar que México es el país líder en el tema dentro de la región de estudio de este trabajo. Sin embargo, pese al desarrollo en metodologías y conceptos para la selección de áreas de conservación, es necesario mejorar los ejercicios realizados teniendo en cuenta los puntos que mencionaré a continuación.

1. La escala y resolución espacial de los trabajos no presentan una relación lineal, esto quiere decir que hay muchos estudios a escalas locales que tienen una resolución menos fina que estudios a escalas regionales. Esto tiene implicaciones importantes no sólo en el proceso de selección, sino también en la implementación de los resultados. Resoluciones finas en trabajos a escalas locales son importantes porque ello garantiza una mayor eficiencia en los resultados, lo cual se refleja en una mejor evaluación de costos y oportunidades de implementación (Pressey & Logan 1998).

2. Los sustitutos que se han utilizado en los trabajos de priorización de áreas se restringen a algunos grupos de vertebrados y plantas, sobre todo aves y mamíferos. Debido a la poca eficiencia que se ha demostrado para la mayoría de los sustitutos (Rodrigues & Brooks 2007) es necesario que se desarrollen más trabajos en la región que involucren otros grupos taxonómicos, o usando combinaciones de sustitutos para que exista una mejor representatividad de la biodiversidad en la selección de áreas de conservación (Margules & Sarkar 2007).

3. El uso de sustitutos abióticos es escaso en los trabajos realizados en la región de estudio. Dado que los datos biológicos presentan altos sesgos de muestreo (Reddy & Davalos 2003) y que son limitados para las regiones del mundo con mayor biodiversidad y con más necesidad de conservación (Pimm 2000), como el caso de Mesoamérica, los Andes Tropicales y el Chocó, las ventajas de usar datos ambientales deben ser reconocidas por las personas que trabajan en la planeación de áreas de conservación. Entre sus ventajas están que tienen resoluciones suficientemente finas, se distribuyen de manera homogénea para toda el área de estudio y son, en muchos sitios los únicos datos disponibles (Noss 1987, Margules & Pressey 2000, Oliver et al 2004). Los sustitutos basados en datos ambientales han sido usados en trabajos en otras regiones, algunas veces en combinación con información de grupos taxonómicos (ej. Pressey et al. 2000, Ferrier 2002, Lombard et al. 2003), dando prueba de su aplicabilidad en la materia.

4. Como el papel de la conservación es también la persistencia de las especies, los análisis de viabilidad de poblaciones juegan un papel importante en la conservación (Margules y Sarkar 2007). Desafortunadamente los criterios de parámetros poblacionales o requerimientos ecosistémicos para la persistencia de la biodiversidad son muy limitados. Una línea de acción de la conservación debe ser el integrar de mejor manera los estudios ecológicos y poblacionales con los estudios de planes de conservación, tratando de encontrar datos que ayuden a representar mejor la persistencia de las especies (Pressey et al., 2003, Fleishman et al 2006).

5. Por último, la conservación biológica necesita profundizar el conocimiento de las incertidumbres y complejidades de la implementación (Pressey & Bottrill 2008), tomando en cuenta criterios socio-económicos para la toma de decisiones en el uso de los recursos naturales (Cowling & Pressey 2003, Margules & Sarkar 2007). La necesidad de incluir estos criterios en la conservación se ha reconocido con antelación (Vane-Wright 1996, Pressey & Bottrill 2008) y en la última década ha habido un incremento en los trabajos que los incluyen, así como en los aspectos metodológicos para analizarlos (Pressey & Bottrill 2008). Sin embargo para Mesoamérica, Los Andes Tropicales y el Chocó, no existen trabajos que incorporen estos criterios de forma explícita en la selección de áreas de conservación. Quizás esta pude ser una de las razones de porque después de dos décadas de trabajos en priorización de áreas de conservación, la implementación sea baja y las áreas naturales protegidas (ANP) sigan representando la biodiversidad de

manera incompleta (ej. Powell et al. 2000, Fandiño-Lozano y van Wyngaarden 2005, Garcia 2006)

Frente a estos vacios el presente trabajo hace un valioso aporte para la selección de áreas de conservación en la región, abordando algunos de los aspectos metodológicos y conceptuales que se mención en los puntos 2, 3 y 5.

Mayor diversidad de sustitutos

Un sustituto de biodiversidad debe cumplir varios requisitos para ser usado en la conservación biológica: ser cuantificable, que se tenga conocimiento sobre su biología y ecología, si es un sustituto biológico, y ser evaluado de manera apropiada, para que identifique claramente áreas importantes para la conservación (Sarkar y Margules 2002, Cabeza et al. 2008). Los sustitutos que se usaron en este ejercicio cumplen con estos requisitos, utilizando como sustitutos especies amenazadas y dominios ambientales.

Especies como sustitutos

Una de las principales limitaciones de los datos para especies amenazadas es la carencia de registros, además de las limitantes que poseen los registros de museo para su aplicación a modelos predictivos de sus distribuciones espaciales, ya que aunque se cuente con registros suficientes los sesgos de colecta, las identificaciones taxonómicas y espaciales erróneas resultan en modelos inapropiados (Graham et al. 2004). Este trabajo es el producto de una búsqueda exhaustiva de registros en las colecciones de consulta libre en internet, y demuestra el gran potencial de los registros pero también el limitado número de especies que pueden ser utilizadas. Se reconoce la necesidad de trabajar más en la integración de datos por parte de las colecciones biológicas de los diferentes países incluidos en este trabajo, pues son pocas las que están asociadas a bases de datos globales.

Se pudo trabajar con 313 especies en las categorías de riesgo de la UICN, lo que representan el 2.6% de las vertebrados y el 11% de las plantas reportadas en estas categorías. Con ello se hace evidente que porcentajes bajos como los encontrados son una limitante importantes, además de que no se incluyen especies amenazadas a nivel local, o las especies endémicas, que deben ser consideradas para la toma de decisiones de manera urgente. Sin embargo, hay que resaltar que es el primer trabajo en representar tal diversidad de especies amenazadas a una escala regional. La escala regional permite una evaluación más homogénea en toda el área de estudio, identificando especies o sitios donde se requiera más urgencia de conservación o donde existan mayores probabilidades de persistencia para las especies (Rodrigues y Gaston 2002).

Conservación de especies utilizadas como sustitutos

El nicho potencial de muchas de las especies amenazadas utilizadas en este trabajo se encuentra en áreas transformadas, dado que teóricamente la adaptación a condiciones de hábitat transformado es improbable, y que las poblaciones remanentes en hábitats naturales tienen pocas probabilidades de persistir sin una tasa de inmigración significativa desde hábitats naturales adyacentes (Peterson y Holt 2003). Los grupos taxonómicos que tienen una proporción alta de su nicho potencial en áreas transformadas requieren mayor prioridad de conservación en las áreas naturales remanentes.

De acuerdo a nuestros resultados, los programas de conservación para plantas amenazadas en México son urgentes dada su alto porcentaje de distribución potencial en áreas transformadas y su baja representatividad en las ANP. Otros trabajos concuerdan con esta idea (ej. Luna et al 2007 y Sanchez-Gonzalez 2008). Los ecosistemas de pino-encino en la región Mesoamericana fueron seleccionados reiteradamente en los diferentes análisis, esto se debe a que poseen composiciones de especies amenazadas que los hacen prioritarios en la selección de áreas de conservación. Otros trabajos han identificado la necesidad de unir esfuerzo entre países de Mesoamérica que tiene este tipo de ecosistema, para poder asegurar la existencia de especies amenazadas, como por ejemplo el Quetzal (Solórzano et al. 2003)

Para unir esfuerzos regionales en la protección de especies globalmente amenazadas se recomienda que para las clases Anfibia, Liliopsida, Polipodiopsida y los órdenes Asterales, Fabales, Laurales, Myrtales, Scrophulariales y Rubiales se ponga especial atención en Guatemala, El Salvador, Honduras, Panamá, Nicaragua y México, donde existen los mayores porcentajes de distribuciones potenciales en áreas transformadas, sobre todo los ordenes Myrtales, Scrophulariales y Rubiales que tienen porcentajes inferiores al 31% en ANP. A su vez se debe tener en cuenta a Belize y Ecuador ya que son los países que mostraron mayores porcentajes de las distribuciones de especies amenazadas en áreas naturales remanentes. Belize, Panamá, Nicaragua y Costa Rica tienen además los mayores porcentajes de distribuciones de especies amenazadas representadas en las ANP, pues estos países pueden contar con suficiente área en sitios conservados y en ANP que ofrezcan mejores oportunidades para el éxito de los planes de conservación de las especies amenazadas a nivel regional.

Clasificaciones ambientales como sustitutos

Aunque los dominios ambientales mostraron ser una herramienta adecuada para evaluar la diversidad ambiental de Mesoamérica, Los Andes Tropicales y el Chocó, la carencia de criterios biológicos dentro de su delimitación debe considerarse cuando se hacen inferencias sobre su eficiencia en representar comunidades naturales.

Nuestros análisis sobre la comparación entre sustitutos de especies y clasificaciones ambientales se basaron en la sobreposición de las áreas seleccionadas para la conservación usando uno u otro sustituto. Se debe reconocer que existen otros métodos para compara la eficiencia de sustituto (Rodrigues & Brooks 2007), que pueden arrojar resultados diferentes. Sin embargo, dado el numero de soluciones que fueron evaluadas concideramos que los resultados evaluan algun grado de variablidad en la configuración espacial de las soluciones. Aunque los resultados confieren mas eficiencia a los dominios ambientales como susutitutos que a las espcecies, como tambien lo han encontrado otros trabajos (Ferrier et al. 1997; Sarkar et al. 2005; Trakhtenbrot & Kadmon 2005), su efectividad fue relativamente baja, resultado similar al encontrado por Rodrigues y Brooks (2007).

Lo anterior se debe a que las condiciones que delimitan las distribuciones de las especies no están restringidas a las características abióticas, y los factores ecológicos y biogeográficos juegan un papel muy importante y no son representados en las clasificaciones ambientales. Posiblemente también las características abióticas de la distribución de ciertas especies no quedan plasmadas en la clasificación realizada (Margules and Sarkar 2007). La gran diversidad de la región de estudio implica que hay procesos diferentes que delimitan las distribuciones de los componentes biológicos, por lo cual un único tipo de sustituto no puede representarlos a todos adecuadamente. Podemos concluir entonces que hasta que no encontremos sustitutos para los procesos que delimitan las distribuciones de las especies, lo más prometedor para la identificación de áreas de conservación es el uso combinado de especies de diferentes taxones y características abióticas (Margules and Sarkar 2007).

Como una de las metas de la conservación es la persistencia de las especies, un campo que será crucial en el tema de sustitutos es pasar de evaluar si un sustituto representa adecuadamente a otro, a evaluar si representa adecuadamente su persistencia. Es decir, evaluar si sitios seleccionados que aseguren la persistencia de ciertas especies también aseguran la persistencia de otras (Rodrigues & Brooks 2007). También habrá que identificar sustitutos que representen procesos ecosistémicos que aseguren la persistencia de la biodiversidad y la provisión de servicios ambientales (Egoh et al. 2007).

Heterogeneidad ambiental, implicaciones para la conservación

Como lo demuestran nuestros resultados, las clasificaciones ambientales no son suficientes para delimitar muchas de las ecoregiones delimitadas por la WWF (Olson et al. 2001), resultado que concuerda con el de otros autores (Thompson et al. 2004, McDonald et al. 2005). Las ecoregiones de la WWF han sido utilizadas es numerosas iniciativas de conservación (Brooks et al. 2006), guiando la inversión de recursos para la investigación y conservación a ciertas ecoregiones (Myers et al. 2000, Olson & Dinerstein 2002, Ferrier et al. 2004). A una escala global las ecoregiones son útiles para enfocar esfuerzos de conservación (Ferrier et al. 2004). Sin embargo los criterios para delimitar las ecoregiones no han sido los mismos en todo el mundo, y como resultado algunas ecoregiones están delimitadas por características biológicas y otras por características ambientales (Metzger et al. 2005). Por ello nuestras clasificaciones ambientales pudieron representar adecuadamente algunas ecoregiones, mientras que la carencia de datos biológicos impidió hacerlo para otras.

Esto tiene implicaciones en conservación porque como lo muestran nuestros resultados, algunas ecoregiones resultan más diversas ambientalmente que otras, y al tratarlas como unidades equivalentes para asignar recursos de conservación se pueden estar dejando de lado ecoregiones que tengan mayores valores de diversidad ambiental. Subestimar los valores de diversidad dentro de las unidades de conservación crea falsos porcentajes de representación en las ANP. Esto es muy relevante al relacionarlo con valores propuestos de manera internacional con implicaciones políticas, como los de la Convención de la Diversidad Biológica (2002), que plantea que el 10% de cada ecoregion debe ser protegido. Si una ecoregion es muy diversa ambientalmente no se puede decir que está bien representada si ya se tiene cubierto el 10%, a no ser que ese 10% esté adecuadamente distribuido dentro de la ecoregión para representar su diversidad ambiental. Como lo destacan Ferrier et al. (2004), este es el resultado de asignar valores de conservación con base en escalas globales, cuando las decisiones de conservación que se deben tomar a escalas locales.

Evaluar la diversidad ambiental dentro de cada una de las ecoregiones y compararlas a niveles regionales resulta muy importante para reconocer vacios de conservación en estas unidades tan utilizadas en conservación. Así, este trabajo identifica, las ecoregiones presentes en la depresión del Balsas en México, Sierra Madre del Sur en México y Chiapas, las cadenas montañosas de Centro América, el magdalena medio, los valles interandinos y la cordillera oriental en Colombia y la selva húmeda del oeste y el bosque montano del este de la Cordillera Real en el Ecuador como prioridades donde se debe mejorar la representación de su diversidad ambiental en las áreas de conservación. Otros trabajos también han identificado la necesidad de mejorar la conservación en algunas de estas ecoregiones (Soutullo et al. 2008).

Selección de áreas, importancia del concepto de complementariedad

Durante las últimas dos décadas, las metodologías para la selección de áreas se han basado en el concepto de complementariedad, entendido como la contribución cuantitativa de un sitio para representar las características o especies que aun no han sido representadas en los sitios seleccionados (Kirkpatrick 1983, Pressey et al. 1993, Possingham et al. 2000, Margules & Pressey 2000). La complementariedad resulta ser más eficiente, es decir encuentra el menor número de sitios donde se incluya un número definido de especies o características que se quieran representar, en comparación a cuando se seleccionan áreas con base en la riqueza de especies (ej. Margules et al. 1988,Williams et al. 1996).

Este trabajo utilizó dos programas para seleccionar áreas de conservación con base en la complementariedad, uno que usa algoritmos heurísticos, ResNet (Garson et al. 2002, Sarkar et al. 2002) y otro metaheurísticos, ConsNet (Ciarleglio 2009). Este tipo de algoritmos generan resultados casi óptimos al momento de resolver el problema de complementariedad en una región determinada, por eso los análisis de este trabajo garantizan la efectividad en términos de la cantidad de área seleccionada para representar los diferentes porcentajes de los distintos sustitutos.

Cantidad de área seleccionada

El área para representar adecuadamente la heterogeneidad ambiental de la región es mayor que el área necesaria para representar a las especies. Ello se debe a que las distribuciones de especies se sobreponen entre sí, lo cual no ocurre con los dominios ambientales. Esta condición de sobreposición en las distribuciones le confiere mayor complementariedad a las distribuciones de especies con respecto a los dominios ambientales, resultando en una menor área seleccionada.

Sin embargo, para lograr porcentajes altos de representatividad de las especies amenazadas se necesitan grandes áreas naturales destinadas a la conservación, hasta el 98% del territorio natural en algunos países. La gran cantidad de área necesaria para representar las especies es el resultado de que su distribución está dispersa por toda el área de estudio, lo cual rectifica la importancia del área en términos de biodiversidad. Solo 0.5% del total de la región de estudio, correspondiente a 8218 km^2 no mostró presencia de ninguna distribución potencial utilizando solo 313 especies amenazadas. Es improbable que un país disponga de una gran cantidad de área para destinarla a la conservación. Por tal razón, los análisis que se realizaron en los capítulos 5 y 6 utilizaron porcentajes de representación hasta el 40%, donde para algunos países se requiere el 60% del territorio.

Las áreas que son seleccionadas por los dos sustitutos en este trabajo se concentran en tres áreas geográfica: en México al norte del istmo de Tehuantepec, en Centro América en los bosques de pino-encino y en las selvas secas, y en las cordilleras de los Andes al sur de Colombia y en todo Ecuador. Estas regiones deben ser reconocidas especialmente por su alto valor complementario para representar diferentes características de la región (Grenyer et al 2006).

La adecuada representación de las características biológicas y ambientales de una región no es suficiente para garantizar el éxito de la conservación, ya que la conservación en el mundo real requiere aproximaciones complejas, incluyendo consideraciones de tiempos de implementación y persistencia (Rodrigues & Brooks 2007). Las consideraciones en los tiempos de implementación son importantes porque es imposible establecer todos los sitios prioritarios al mismo tiempo (Pressey & Taffs 2001). Además, para minimizar la pérdida de biodiversidad es necesario proteger algunos sitios primero que otros (Margules & Pressey 2000, Pressey & Taffs 2001).Deben establecerse asimismo prioridades de implementación a través de un balance entre prioridades basadas en la persistencia de las especies, y el considerar las oportunidades reales y limitaciones que afectaran las acciones de conservación (Pressey & Bottrill 2008).

Aproximación multicriterio a la selección de prioridades de conservación

Con respecto a otros escenarios donde se exploraron mejores oportunidades de persistencia y oportunidad de implementación, asegurar exclusivamente la representatividad de especies y dominios ambientales mostró una mayor eficiencia en términos de área seleccionada, pero se obtuvieron valores menores para la representatividad total de sustitutos y valores menos óptimos para otras variables que representan mayor amenaza y menores oportunidades de mejores estrategias de implementación. Esto se ha observado en otros trabajos que incorporan parámetros no biológicos en la selección de áreas de conservación (Ando et al. 1998; Faith et al. 1996). Al ignorar el costo de otros criterios no biológicos incorporados a la selección de áreas de conservación, se están perdiendo oportunidades importantes para lograr mejores resultados en conservación (Naidoo et al. 2006).

Los resultados de este trabajo demuestran cómo se pueden integrar otros parámetros sociales y económicos sin comprometer la eficiencia de la representación en las áreas seleccionadas para la conservación, demostrando que la planeación de la conservación puede tomar ventaja del concepto de complementariedad, al encontrar un balance en las necesidades de la sociedad y al asignar diferentes estrategias de manejo de los recursos a distintas áreas para maximizar los beneficios sociales (Faith & Walker 2002).

En el análisis multicriterio no encontramos una única solución óptima, lo cual sugiere que se debe mejorar el conocimiento de las relaciones entre variables socio-económicas, uso del suelo y persistencia de la biodiversidad. De esa manera se podrán asignar mejor los valores umbrales y los pesos de importancia a las variables dentro de los análisis, lo que mejorará los resultados de la selección de áreas. Sin embargo, dadas las complejas dinámicas de los procesos de uso del suelo, dependientes no sólo de variables cuantificables sino de procesos históricos, políticas y oportunidades (Soulé 1991), sólo análisis a escalas locales permitirán hacer esto posible. Este trabajo analiza áreas que son regionalmente importantes y donde se debe hacer un zoom local para planear adecuadamente su conservación, ofreciendo un ejercicio esencial entre diferentes niveles de planeación, de lo regional a lo local (Faith & Walker 2002).

Evaluación de las áreas naturales protegidas establecidas

El análisis multicriterio cambió un poco la perspectiva de la efectividad de las ANP, mientras que todos los análisis anteriores demostraron que la representatividad de la biodiversidad en las ANP es baja, especialmente en Colombia, Mexico y Ecuador como también lo señalan otros trabajos (ej. Sierra et al. 2002, Fandiño-Lozano y van Wyngaarden 2005, Garcial 2006, Fuller et al. 2006), se encontró que las áreas seleccionadas para los escenarios de amenaza e implementación se sobreponían en mayor medida con las ANP, evidenciando que las ANP son eficientes en disminuir las tasas de deforestación dentro de ellas (Aaron et al. 2001). Ello puede ser producto de que han sido establecidas en sitios con mayor porcentaje de vegetación natural y más alejadas de las fuentes de disturbio, carreteras o ciudades (Naughton-Treves et al. 2005).

Las ANP son una estrategia eficiente para la conservación si se toman como áreas núcleo, que proveen servicios ambientales vitales y que contribuyen al mejoramiento y restauración del hábitat que las rodea, pero se requiere una mayor atención hacia el manejo eficiente de las áreas que rodean a las ANP (Naughton-Treves et al. 2005). Para que la conservación funcione en estos países se debe usar una perspectiva de organización del territorio en todo el paisaje. Políticas excluyentes como restringir áreas para reservas de conservación, juegan un papel menos importante que políticas de conservación que integren un uso sustentable de los recursos, como las reservas de la biosfera (Figueroa & Sánchez-Cordero 2008).

Las ANP por sí solas protegen una pequeñísima fracción de la biodiversidad a largo plazo (Ervin 2003, Rodrigues et al. 2004, Chan y Daily 2008). Por tal razón las estrategias de conservación deben integrar el diseño de áreas de conservación dentro de un manejo adecuado del paisaje, utilizando la matriz de hábitat transformado en que se encuentran.

La diferencia en la cantidad de área selecionada entre los ejercicios que incorporaron las ANP a priori y los que no, se hace más pequeña a medida que los porcentajes de representatividad aumetan, esto tiene implicaciones respecto al costo de postponer la planeación de la conservación (Fuller 2007). Ya que no se puede pretender tener altos porcentajes de representatividad, la planeación en escenarios reales requiere de una cuidadosa selección para que sea mejor la relación entre costo y efectividad.

Diferentes opciones para la conservación

Este trabajo permitió hacer un enlace entre selección de áreas de conservación y opciones de manejo, mediante la creación de diferentes escenarios en el análisis multicriterio. Las estrategias de manejo consideran estrategias de conservación en áreas naturales, estrategias para mitigar el cambio climático y estrategias de desarrollo sostenible, estar últimas deben ser implementadas en áreas naturales con altos niveles de pobreza. En este sentido se puede ver a la conservación como un puente de oportunidades para encontrar soluciones a múltiples problemas (Roe 2008).

Aunque en la selección de áreas de conservación generalmente no es considerada el área transformada en cultivos, algunos estudios demuestran que son sistemas importantes para proveer ciertos servicios ecológicos y que permiten proteger niveles significativos de biodiversidad (Daily et al. 2003, Geneletti 2007). Pequeños cambios del manejo de los sistemas agrarios pueden contribuir positivamente a la biodiversidad (Chan & Daily 2008). El grado de transformación de los ecosistemas en el norte de Mesoamérica y en Los Andes Tropicales se puede ver como una oportunidad para que la gente se involucre en iniciativas de conservación, donde se puedan beneficiar económicamente y ayudar a mejorar la calidad del hábitat.

En las regiones del sur de Mesoamérica y el Chocó, las acciones de conservación pueden ir enfocadas más hacia la protección de remanentes de vegetación natural, con estrategias como pago por deforestación evitada (REDD) o pago por servicios ambientales (Pagiola et al. 2002). Acciones de ese tipo pueden ser efectivas para mejorar la calidad de vida de las personas que habitan estas áreas naturales (Mehta & Kill 2007).

Aun queda mucho por hacer respecto a la selección de áreas, especialmente cuando involucre análisis multicriterio. La propuesta del presente estudio es hacer planes sistemáticos de conservación a escalas locales en las regiones que a continuación se ilustran para los diferentes países (figura 1). Dichas áreas tienen alta prioridad y merecen nuestra atención porque son identificadas como prioritarias para la conservación en nuestro trabajo y también por otras iniciativas, que usan metodologías diferentes, por ejemplo, la de identificación de regiones terrestres prioritarias de México realizada por la Comisión Nacional para el Conocimiento y uso de la Biodiversidad, y los trabajos de Soriano y colaboradores (2001) en Ecuador, Fandiño-Lozano y van Wyngaarden (2005) en Colombia y Calderón y colaboradores (2004) en Centro América.

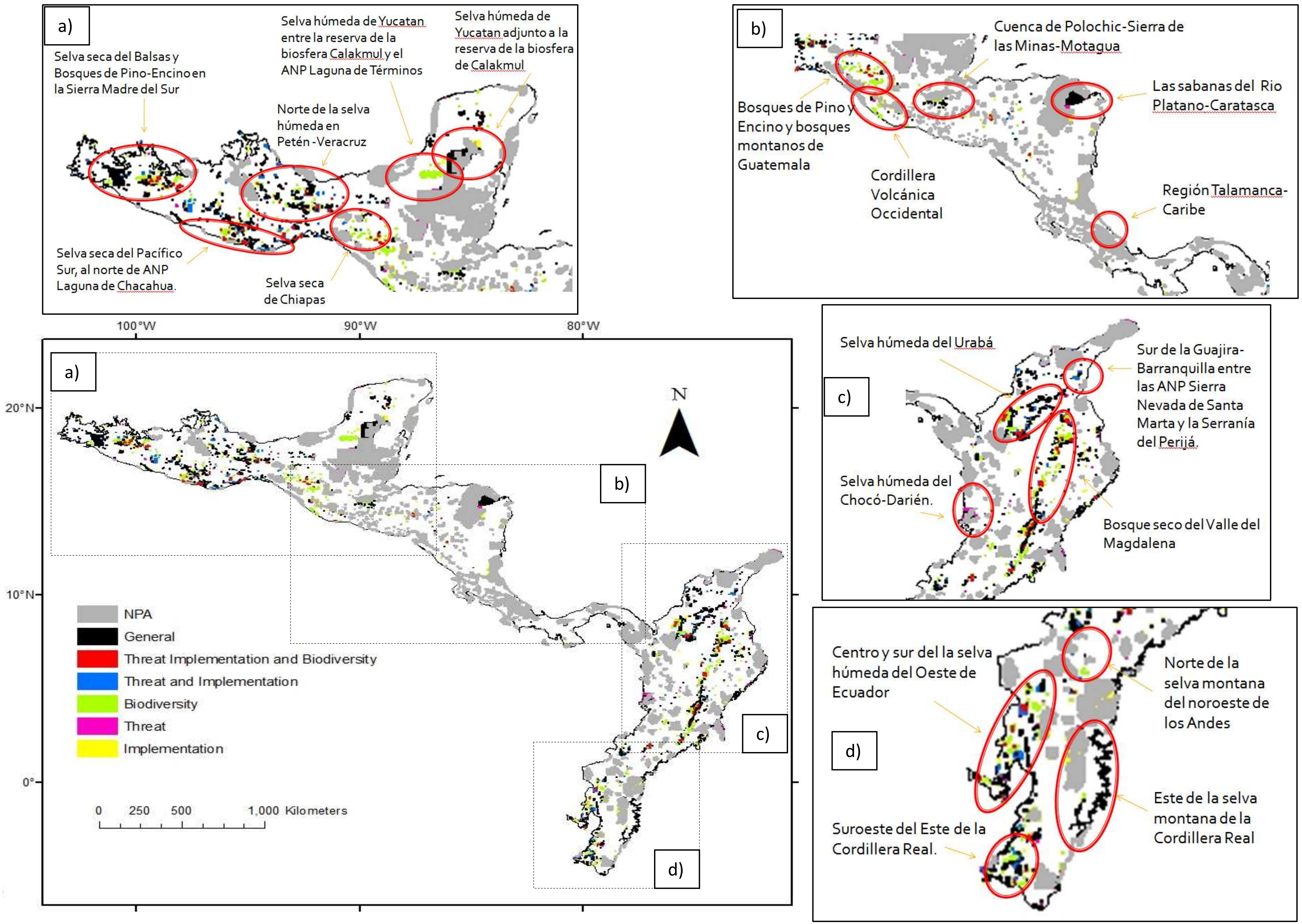


Figura 1. Selección de áreas prioritarias para la conservación en Mesoamérica, Chocó y Los Andes Tropicales. En gris se representan las ANP establecidas, las selecciones se muestran en colores dependiendo del escenario en que fueron seleccionadas. Para más detalles ver capitulo 6.

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