UNIVERSIDAD DE CÓRDOBA

# Teledetección y sistemas de información geográficos aplicados al seguimiento de procesos de deforestación en bosques secos de Ecuador

Tesis Doctoral

Carlos Alfredo Rivas Cobo

TITULO: *TELEDETECCIÓN Y SISTEMAS DE INFORMACIÓN GEOGRÁFICOS APLICADOS AL SEGUIMIENTO DE PROCESOS DE DEFORESTACIÓN EN BOSQUES SECOS DE ECUADOR*

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UNIVERSIDAD DE CÓRDOBA

**Programa de doctorado en Biociencias y ciencias agroalimentarias**

**Tesis doctoral**

# Teledetección y sistemas de información geográficos aplicados al seguimiento de procesos de deforestación en bosques secos de Ecuador

Remote sensing and geographic information systems applied to the monitoring of deforestation processes in dry forests of Ecuador

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Fecha depósito:

16/11/2021

**TÍTULO DE LA TESIS:** TELEDETECCIÓN Y SISTEMAS DE INFORMACIÓN GEOGRÁFICOS APLICADOS AL SEGUIMIENTO DE PROCESOS DE DEFORESTACIÓN EN BOSQUES SECOS DE ECUADOR.

**DOCTORANDO/A:** Carlos Alfredo Rivas Cobo

## INFORME RAZONADO DEL/DE LOS DIRECTOR/ES DE LA TESIS

(se hará mención a la evolución y desarrollo de la tesis, así como a trabajos y publicaciones derivados de la misma).

La tesis doctoral presentada por el doctorando Carlos Alfredo Rivas Cobo con el título "Teledetección aplicada al seguimiento de procesos de deforestación en bosques secos de Ecuador" corresponde a su trabajo de tesis doctoral realizado bajo nuestra dirección. Consideramos que el doctorando ha cumplido correctamente las etapas implícitas en el desarrollo de la tesis doctoral, lo que ha permitido su correcta formación como investigador. Durante el desarrollo de la tesis doctoral, el doctorando ha desarrollado las destrezas propias del proceso de investigación (planteamiento y contraste de hipótesis, búsqueda bibliográfica, análisis estadístico, redacción de artículos…) y ha sabido sobrepasar las dificultades inherentes al iniciarse en la investigación. Creemos que el doctorando tiene la formación necesaria para continuar trabajando en la misma línea de investigación, y desarrollar su carrera investigadora. El documento final de su tesis doctoral cumple con los requisitos establecidos en la normativa actual para poder finalizar su tesis doctoral. Concretamente, la tesis incluye dos capítulos ya publicados en revistas incluidas en el Journal Citation Report, una del cuartil 2 (Q2) y otra del cuartil 1 (Q1), en los que el doctorando figura como primer autor y los directores como co-autores. Además, consideramos que el documento está redactado correctamente, siendo su contenido consecuencia del trabajo realizado por el doctorando bajo la coordinación de los directores, siendo información inédita no publicada en otras partes. Las publicaciones derivadas de la tesis son:

-Rivas CA, Navarro-Cerillo RM, Johnston JC, Guerrero-Casado J (2020). Dry forest is more threatened but less protected than evergreen forest in Ecuador's coastal region. Environmental Conservation 47: 79–83. https://doi.org/10.1017/S0376892920000077

-Rivas CA, Guerrero-Casado J, Navarro-Cerillo RM (2021) Deforestation and fragmentation trends of seasonal dry tropical forest in Ecuador: impact on conservation. Forest Ecosystems 8: 1–13. https://doi.org/10.1186/S40663-021-00329-5

Por todo ello, se autoriza la presentación de la tesis doctoral.

Córdoba, 15 de Noviembre de 2021

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## Resumen

Ecuador es un país mega diverso, con cuatro regiones claramente diferenciadas: Costa, Sierra, Las islas Galápagos y la Amazonía, presentando la región de la Costa un menor grado de protección. Esta región está dominada fundamentalmente por dos ecosistemas: los bosques secos y húmedos. El bosque seco es el ecosistema tropical con menos superficie remante a nivel mundial, esto lo ha llevado a un estado crítico en gran parte de Sudamérica, estando menos estudiado que los bosques verdes. En Ecuador la situación es similar, el bosque seco ecuatoriano se encuentra en estado crítico a pesar de formar parte del punto caliente de biodiversidad mundial "Chocó/Darien/Western Ecuador". Los bosques secos ecuatorianos se encuentran principalmente en la región de la Costa y se dividen, a su vez, en dos: deciduo y semideciduo.

Al presentar la región costa menor protección y dos tipos de ecosistemas mayoritarios, el primer capítulo de la tesis tiene como objetivo comparar el grado de protección y conservación de los remanentes de bosque seco y húmedo de la costa ecuatoriana. Para esta comparación se usó una categorización que clasifica los ecosistemas de Ecuador a partir de cinco parámetros: amenaza, vulnerabilidad, conectividad, fragmentación y fragilidad, y se estudiaron las figuras de protección "Patrimonio de Áreas Naturales del Estado" (PANE) y "Bosque protector". Los resultados pusieron en evidencia el mal estado de conservación del bosque seco, presentando peores parámetros en conectividad, fragilidad y amenaza que los bosques húmedos. Además, presentaba menos superficie protegida, tanto en bosque remanente como en superficie dentro de las áreas protegidas. Estos datos demuestran que existe una discriminación del bosque seco respecto al húmedo en su protección, siendo necesario aumentar la protección de los bosques secos como parte de una estrategia nacional integral que garantice la conservación de los altos niveles de biodiversidad y endemicidad de este tipo de bosques.

La deforestación de Ecuador ha sido la más alta de Sudamérica entre 1990 y 2010, lo que ha llevado a muchos ecosistemas a presentar una alta tasa de fragmentación, siendo una de las mayores amenazas para los bosques del país. El segundo capítulo, tiene como objetivo conocer la evolución de la deforestación y de la fragmentación del bosque seco durante casi 30 años (entre 1990 y 2018), y evaluar si las figuras de protección son efectivas para evitar estos fenómenos. El bosque seco se dividió en tres tipos (bosques deciduos, semideciduos y una transición entre ambos), y se teselaron en hexágonos de 10 $km^2$. Los resultaros mostraron un alto grado de deforestación en los bosques secos, aumentando sus parámetros de

fragmentación y el índice de fragmentación reticular en los tres tipos de bosques, siendo el bosque semideciduo el más afectado, caracterizado por el valor más alto del índice de fragmentación reticular y presentando un tamaño medio de tesela o parche muy preocupante. No todas las figuras de protección fueron efectivas para reducir la fragmentación, ya que las áreas PANE son las que menor aumento de fragmentación tuvieron, luego el bosque sin protección, y por último el Bosque protector, demostrando que esta última figura de protección no es efectiva para detener la fragmentación y la deforestación.

Debido a la complejidad del estudio de la fragmentación, dado que existen múltiples parámetros para medirla, aunque la mayoría se pueden agrupar en 4 parámetros, y que ningún parámetro de tesela por sí mismo explica el estado de fragmentación de una tesela, en el capítulo tres se propone un índice que integra los principales parámetros de fragmentación a escala de tesela. Esta fórmula se aplicó al caso concreto del bosque seco ecuatoriano, arrojando un índice de fragmentación muy alto, y un aumento de la fragmentación de un 7% entre 1990 y 2018. La métrica de fragmentación también se mostró efectiva cuando se extrapolaba y se calculaba a escala de zonas. Para estos cálculos zonales se usaron hexágonos de 10 $km^2$ y el área de influencia de las teselas. Este parámetro es uno de los aspectos más novedosos del índice propuesto, ya que calcula la fragmentación en función del área de influencia de la tesela, que se corresponde con el área que puede ocupar dicha tesela. Posteriormente se calcularon patrones espaciales, más concretamente un análisis de puntos calientes, donde el índice se mostró muy eficiente para identificar teselas en riesgo de desaparición.

La conectividad es un factor clave en el mantenimiento de la biodiversidad, y se ve afectada por la fragmentación y la deforestación, jugando un papel fundamental en los ecosistemas altamente fragmentados como el bosque seco ecuatoriano. El capítulo cuatro tiene como objetivo calcular la evolución de la conectividad en los bosques secos entre 1990-2018. Para el cálculo de la conectividad se usó el software Graphab, que se basa en la teoría de grafos, demostrando ser una potente herramienta para calcular la conectividad. Los resultados mostraron que el bosque seco ha perdido más del 50% de conectividad global en algunos parámetros, y que existen zonas con una conectividad muy baja y en riesgo de desaparición. Además, se observó que la conectividad por tesela no es útil al extrapolarla a escala de paisaje, pero sí es útil en la toma de decisiones relativas a la escala de tesela, como por ejemplo en políticas de conservación. El análisis de usos de suelo permitió, además, identificar que la matriz que rodea el bosque seco se ha vuelto menos permeable, cambiando el bosque por terrenos agrícolas mayoritariamente, aunque también aumentaron los terrenos antrópicos y las zonas de matorral. Esto ha hecho que las conexiones entre las teselas cada vez sean menores en número y en distancia. En este aspecto destaca la importancia de las teselas pequeñas y de los otros tipos de bosques naturales para mantener la conectividad del bosque seco. Estas teselas pequeñas son de gran importancia también a la hora de las actuaciones de restauración ecológica, donde se demostró la importancia de la conectividad en la toma de decisiones, siendo más efectivas las acciones de restauración con pequeñas teselas que la restauración de grandes superficies concentradas en teselas extensas, ya que la reforestación con teselas pequeñas requiere de menos superficie y puede tener mejor resultados en la conectividad global.

En conclusión, los bosques secos de Ecuador están poco protegidos, sufriendo una alta tasa de deforestación, aumento de la fragmentación y perdida de conectividad, que los ha llevado a un pobre estado de conservación y riesgo de desaparición.

## Abstract

Ecuador is a mega diverse country with four clearly differentiated regions: The Coast, the Andean, the Galapagos Islands and the Amazon. The Coast region presenting a lower degree of protection. This region is fundamentally dominated by two ecosystems: dry and humid forests. The dry forest is the tropical ecosystem with the least remaining surface in the world and as a result, its situation is critical in South America and therefore less investigated than humid forests. In Ecuador has a similar situation, the Ecuadorian dry forest is in critical condition despite being part of the global biodiversity hotspot "Chocó / Darien / Western Ecuador". Ecuadorian dry forest is mainly found in the Costa region and subdivided into two: deciduous and semi-deciduous.

Costa region suffers lack of protection and two types of major ecosystems, the first chapter of the thesis aims to compare the degree of protection and conservation of the remnants of dry and humid forest on the Ecuadorian coast. In the study we used a categorization that classifies the ecosystems of Ecuador based on five parameters: threat, vulnerability, connectivity, fragmentation and fragility, and the protection figures “Patrimony of Natural Areas of the State” (PANE) and “Forest Protective". The results showed the poor state of conservation of the dry forest, presenting worse parameters in connectivity, fragility and threat than the humid forests. In addition, it also has less protected area, both in the remaining forest and on the surface within the protected areas. This led to a discrimination between the dry forest versus the humid forest in its protection, making necessary to increase the protection of dry forests as part of a comprehensive national strategy that guarantees the conservation of the high levels of biodiversity and endemicity of this type of forest.

Ecuador's deforestation has been the highest in South America between 1990 and 2010, which led to high number of ecosystems show an increase fragementation rate, and therefore being one of the greatest threats to the country's forests. The second chapter aims to know the evolution of deforestation and fragmentation of the dry forest during almost 30 years (between 1990 and 2018) and to evaluate if the protection figures are effective to avoid these phenomena. The dry forest was divided into three types (deciduous, semi-deciduous and a transition between the two) and they were tessellated into 10 km$^2$ hexagons. The results showed a high degree of deforestation in dry forests, increasing its fragmentation parameters and the reticular fragmentation index in the three types of forests. With semi-deciduous forest being the most affected, characterized by the highest value

of the reticular fragmentation index, and presenting a very worrying average tile size. Not all the protection measures were effective to reduce fragmentation since the PANE areas are the ones with the lowest increase in fragmentation, secondly the unprotected forest and finally the Protective Forest, showing that this last protection figure is not effective to stop fragmentation and deforestation.

Due to the complexity of the study of fragmentation as there are many parameters to measure it (although most can be explained by 4 of the parameters) and that no patch parameter by itself explains the fragmentation state of a patch, in the chapter three proposes an index that integrates the main fragmentation parameters at the patch scale. This formula was applied to the specific case of the Ecuadorian dry forest, showing a very high fragmentation index, and an increase in fragmentation of 7% between 1990 and 2018. The fragmentation metric was also effective when extrapolated and calculated from zone scale. For these zonal calculations, 10 $km^2$ hexagons and the area of influence of the patches were used. Subsequently, spatial patterns were calculated, more specifically an analysis of hot spots, where the index was very efficient to identify tiles at risk of disappearance.

Connectivity is a key factor to maintaining biodiversity and is affected by fragmentation and deforestation, playing a fundamental key in highly fragmented ecosystems such as the Ecuadorian dry forest. Chapter four aims to calculate the evolution of connectivity in dry forests between 1990-2018. For the calculation of connectivity, Graphab software was used which is based on graph theory, proving to be a powerful tool for calculating connectivity. The results showed that the dry forest has lost up to 50% of global connectivity in some parameters, as well as that there are areas with very low connectivity and at risk of disappearance. In addition, it was observed that connectivity by patch is not useful when extrapolated to the landscape scale, but it is useful in making decisions related to the patch scale such as conservation policies.

Land use analysis allowed to identify that the matrix that surrounds the dry forest has become less permeable, changing the forest for mostly agricultural lands, although anthropic lands and scrub areas also increased. This has made that the connections between the patches are decreasing in number and in distance. In this regard, the importance of small patches and other types of natural forests to maintain the connectivity of the dry forest is crucial. These small patch are also of great importance when it comes to ecological restoration actions where the importance of connectivity in decision-making was demonstrated, restoration actions with small patches being more effective than large surfaces concentrated in patches large, since reforestation with small patches requires less surface area and may have better results in global connectivity.

In conclusion, the dry forests of Ecuador are poorly protected suffering a high rate of deforestation, increased fragmentation and loss of connectivity, which has led to a poor state of conservation and risk of disappearance.

# Introducción

Aproximadamente el 73% de la superficie de la tierra ya ha sido alterada por los humanos (Locke et al., 2019), los bosques perdieron 178 millones de hectáreas entre 1990 y 2020, principalmente por la expansión de la frontera agrícola, pasando del 32,5% al 30,8% de superficie forestal mundial. Liderando la deforestación entre 2010 y 2020 se encuentra el continente africano, y entre 1990 y 2010 Sudamérica (FAO y PNUMA, 2020). La deforestación en Sudamérica ha sido principalmente por motivos agrícolas, y se espera que en 2050 haya crecido un 30%, aumentando la fragmentación de hábitats (Donald y Evans, 2006; Tilman et al., 2001). Durante el último siglo, la fragmentación de los bosques y la división de hábitats continuos en fragmentos más pequeños y más aislados ha modificado profundamente las características y la conectividad de los bosques, y ha provocado graves pérdidas de biodiversidad (Haddad et al., 2015).

Esta tesis doctoral aborda el estado de conservación, fragmentación y conectividad del bosque seco ecuatoriano, un ecosistema muy alterado, pero a su vez muy poco estudiado. En esta introducción se analiza los efectos de fragmentación y conectividad en los ecosistemas y posteriormente se describen los bosques secos mundiales, el bosque seco ecuatoriano y la deforestación que ha sufrido.

### Efectos de la fragmentación en la conservación de los bosques

El concepto de fragmentación fue introducido por Curtis (1956) y Moore (1962), definiendo la fragmentación como una ruptura del hábitat donde se incrementa el número de teselas. Posteriormente, McArthur y Wilson (1967) definen la fragmentación como un efecto de "isla biogeográfica" donde los organismos quedan aislados (Fahrig, 2019). Más concretamente, la fragmentación puede definirse como el proceso que subdivide el hábitat en fragmentos más pequeños, más aislados y geométricamente más complejos, resultado del cambio del uso de suelo (McGarigal y Marks, 1995; Chakraborty et al., 2017).

Durante décadas los científicos han estudiado la fragmentación y sus consecuencias para los ecosistemas y especies animales. Actualmente existe una controversia (Fig. 1) en cuanto a los efectos de la fragmentación, ya que un gran número de los autores la consideran perjudicial (Arasa-Gisbert et al, 2021), mientras que otros lo debaten, alegando que la mayoría de los efectos en la biodiversidad son positivos o sin efectos significativos (Fahrig, 2003, 2017, 2019; Fletcher et al., 2018).

**Causas y consecuencias de la fragmentación de hábitats**

| | | |
|---|---|---|
| Definición | La fragmentación es un proceso y los efectos de la perdida de hábitat y fragmentación son indisociables | La fragmentación es un patrón y los efectos de la perdida de hábitat y la fragmentación se deben medir de manera aislada. |
| Unidad de análisis | Sitio, tesela o paisaje | Paisaje |
| Variables explicativas | Características del paisaje, de la tesela o del sitio | Características del paisaje |
| Extrapolación de datos | Extrapolación de los datos de teselas y sitio a escala del paisaje. | No hay extrapolación de los datos de teselas y sitio a escala del paisaje, evalúa datos a nivel de paisaje. |
| Influencia del hábitat | Las características del sitio, paisaje o tesela son considerados efectos de la fragmentación, aunque este relacionado con la perdida de hábitat | La fragmentación esta relacionada con la perdida de hábitats, y esta variable debe estar controlada mediante el uso de estadística o de forma experimental |
| Consecuencias | La fragmentación tiene efectos fuertes y perjudiciales sobre la biodiversidad | La fragmentación tiene efectos débiles y normalmente positivos para la biodiversidad |
| Protección | Conservar las teselas grandes, al estar las pequeñas más afectadas por la fragmentación y por tanto, con menor valor de conservación | Conservar la mayor cantidad de hábitats posible, ya que las teselas pequeñas pueden tener el mismo valor que las teselas grandes. |

**Figura 1:** Causas y consecuencias de la fragmentación de los hábitats y sus posibles efectos sobre la biodiversidad, donde se puede observar dos grandes tendencias dependiendo de la definición y conceptualización de la fragmentación así como de las metodologías empleadas (Arasa-Gisbert et al, 2021). En azul el estudio de la fragmentación como proceso y en verde como patrón.

La mayoría de los estudios de fragmentación se han desarrollado en Norte América, Europa, Australia y Nueva Zelanda (aproximadamente un 70%) y sólo un 13,2% en Sudamérica (Fardila et al., 2017). La fragmentación se relaciona con efectos negativos en la biodiversidad, de hecho, se considera una de sus principales amenazas (Rogan y Lacher, 2018; Betts et al., 2019; Kuipers et al., 2021). Haddad et al., (2015) demuestra que la fragmentación del hábitat reduce la biodiversidad entre un 13% y un 75% y deteriora las funciones clave del ecosistema al disminuir la biomasa y alterar los ciclos de nutrientes. Por estos motivos, la fragmentación es considerada una de las principales amenazas de los bosques en los trópicos (Trejo y Dirzo, 2000; Fuchs et al., 2003).

Además, existen numerosas controversias respecto a cómo medir la fragmentación (Fig. 1), diversos autores consideran que es un proceso en el cual las especies reaccionan a la modificación del hábitat, tales como la perdida de hábitat, el tamaño, el aislamiento o el borde de los parches. Sin embargo, autores como Fahrig (2003, 2017) defienden que la fragmentación debe ser medida como un patrón, y no debe ser vinculada exclusivamente a la perdida de hábitat (Arasa-Gisbert et al, 2021).

En ese sentido, existen muchas métricas para medir la fragmentación, pero estas se pueden agrupar en categorías de 4 a 6 métricas (Cushman et al., 2008; Chen et al., 2014) que miden parámetros similares. Las métricas de área y de aislamiento o proximidad son las métricas más usadas en el estudio de la fragmentación (48% y 42,4%, respectivamente; Fardila et al., 2017). Las métricas se pueden clasificar en tres grupos: las de configuración, las de composición, y las de variación medioambiental y posición geográficas (Fig. 2).

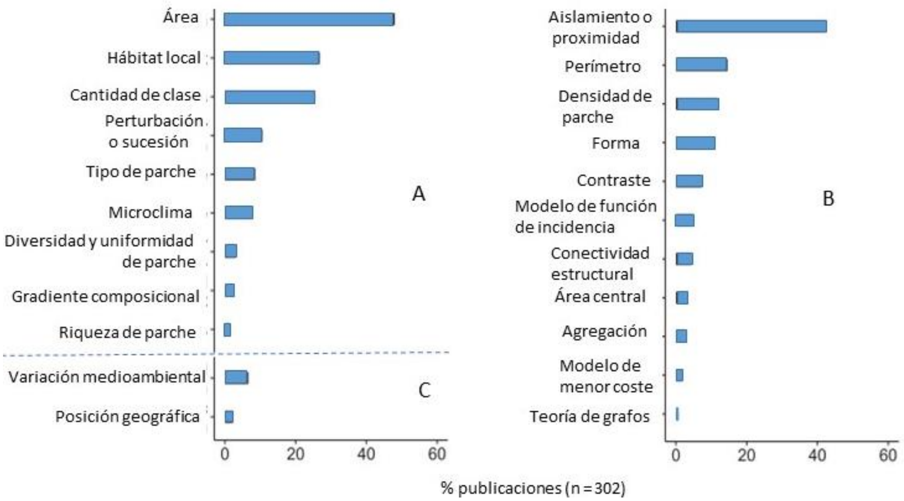


**Figure 2:** Métricas de fragmentación y porcentaje de aparición en publicaciones científicas entre 1994 y 2016. Adaptado de Fardila et al, (2017). A) métricas de composición. B) Métricas de configuración. C) Métricas de variación medioambiental y posición geográficas.

Estos efectos de la fragmentación en los parches o en el paisaje se han estudiado a través de su afectación en la distribución, la dinámica, el movimiento, la genética, la diversidad, la morfología, la fisiología, la interacción de poblaciones o individuos que albergan, la retención de nutrientes, la productividad e incluso los microclimas (Haddad et al. 2015; Fardila et al., 2017), siendo las principales aquellas relacionadas con la distribución y la diversidad de especies, encontrándose en más del 50% de los estudios (Fardila et al., 2017).

**Efectos de la conectividad en la conservación de los bosques**

La deforestación conduce a la fragmentación y el aislamiento de las manchas forestales dentro de una matriz más "hostil" de otros usos del suelo (Donald y Evans, 2006), y esto lleva a una disminución de la conectividad. La conectividad es el grado en el cual el paisaje facilita o impide movimientos de especies o flujos de procesos naturales que sustentan la vida en la Tierra (Hilty et al., 2020). La pérdida de conectividad se considera una de las mayores amenazas para la biodiversidad (Pascual-Hortal y Saura, 2006), y por esta razón se considera una parte clave para su mantenimiento y una cualidad del paisaje fundamental para contrarrestar los efectos negativos de la fragmentación de hábitats (Saura y Lidón, 2010), convirtiéndose en una prioridad para preservar los ecosistemas y sus funciones (UNEP, 2015; Dickson et al., 2019).

La conectividad suele disminuir, o incluso perderse completamente, por la fragmentación y el cambio de uso de suelo producida principalmente por actividades antropogénicas (Hilty et al., 2020). La conectividad vincula una gran variedad de temas ecológicos como son la evolución, la dispersión, y las migraciones de especies, el desarrollo de las estructuras genéticas, las dinámicas fuente-sumidero o las adaptaciones al cambio climático; pero la conectividad también afecta a la conservación, ya que es un factor fundamental al considerar las áreas naturales, áreas de restauración o incluso las especies invasoras y la administración de recursos transfronterizos (Kool et al., 2013).

La conectividad es una cualidad del paisaje y se puede dividir en dos grandes tipos (Fig. 3). La primera es la conectividad estructural, que se basa en modelo binario (hábitat, no hábitat), y en la disposición espacial del hábitat dentro un paisaje. La segunda es la conectividad funcional, que se basa en una matriz donde cada atributo (uso de suelo) facilita o impide el movimiento de los organismos y, por lo tanto, se puede medir la conectividad entre los parches de hábitats en función de las características que queramos estudiar (Keeley et al., 2021). En este caso sirve para medir, por ejemplo, la facilidad de movimiento de genes, gametos, propágulos o individuos a través de diferentes ecosistemas (Hilty et al., 2020). Los estudios más complejos y realistas de conectividad también incluyen modelos de dispersión e información demográfica de las poblaciones en el ámbito de estudio (Drake et al., 2021), esta última denominada conectividad ecológica para especies (Hilty et al., 2020).

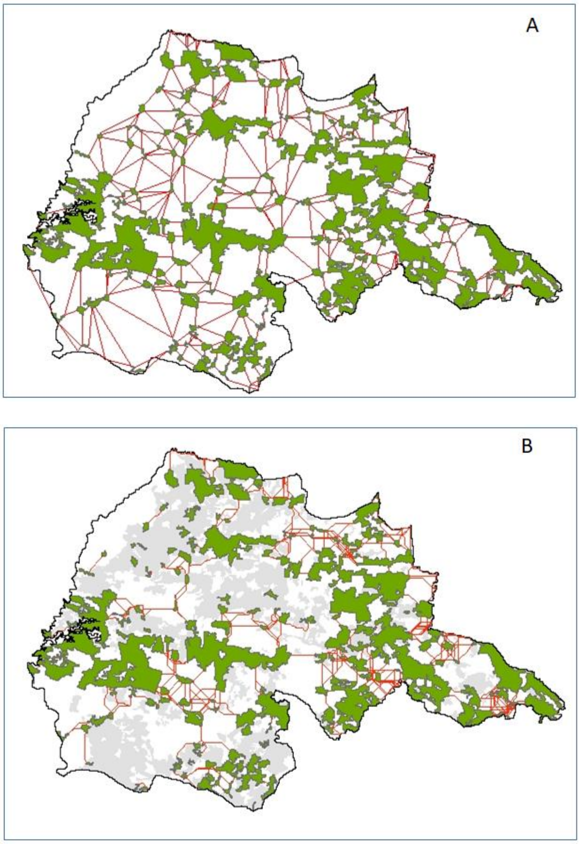


**Figura 3:** Ejemplo que ilustra las diferencias entre los dos grandes tipos de conectividad: en verde las teselas de hábitats, en rojo los links o conectores entre los hábitats. A) Conectividad estructural. B) Conectividad funcional donde las áreas blancas son zonas de fácil desplazamiento por donde se establecen los links entre parches de hábitats, y las zonas grises áreas de difícil desplazamiento.

Las métricas de conectividad pueden ser a nivel global, local (por parche o tesela), de vecindad o proximidad y por componente (Fig. 4) (Rayfield et al., 2011). Los componentes son grupo de nodos (hábitat) conectados, esto significa que los organismos pueden moverse (enlazar) entre teselas de hábitats (nodos) dentro del mismo componente, pero no con un componente distinto, y por lo tanto, las teselas de diferentes componentes no pueden comunicarse, estando aislados entre ellos (Herrera et al., 2017).

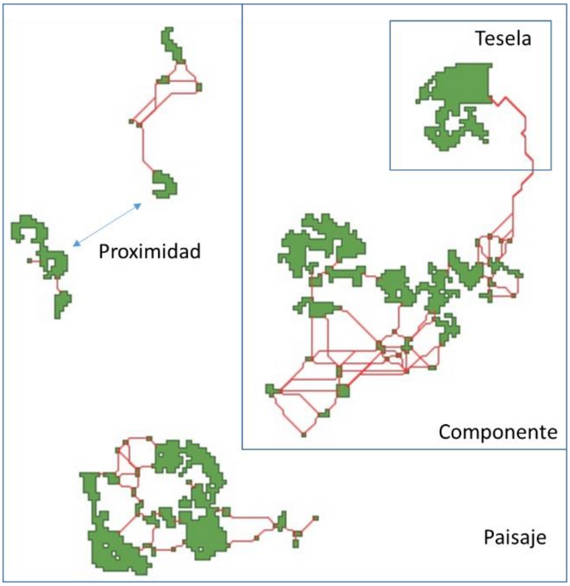


**Figura 4:** Ejemplo de redes formadas por teoría de grafos y menor coste, las teselas de hábitat (en verde) son conectados por links (en rojo) formando diferentes niveles estructurales.

Ante este contexto, la teoría de grafos se ha mostrado como una herramienta potente y útil para evaluar todo tipo de conectividad basándose en un conjunto de nodos y bordes (o link), donde los nodos son los elementos individuales dentro de la red y los bordes representan la conectividad entre nodos. Los bordes pueden ser binarios (conectados o no) o contener información adicional sobre el nivel de conectividad (Minor y Urban, 2008). En el campo de la conectividad de ecosistema; los nodos son el hábitat y los links son las conexiones entre los hábitats (Pascual-Hortal y Saura, 2006). La teoría de grafos se ha usado ampliamente en conectividad (e.g. Minor y Urban, 2008; Galpern et al., 2011; Luque et al., 2012), ya que permite analizar las redes de forma eficiente y sus características, pudiendo usarse para identificar la conectividad tanto funcional como estructural. Sin embargo, para el análisis de la funcionalidad habría que añadir otros parámetros que permitan cuantificar la permeabilidad de la matriz como, por ejemplo, el uso de tierra donde se encuentra el grafo, ya que si no se tiene en cuenta la matriz, las métricas de conectividad no representan la realidad, o sólo podrían hacerlo con especies muy móviles como las aves (Keeley et al., 2021).

En la teoría de grafos, las métricas de tesela permiten identificar teselas que tienen una importancia determinada o un alto grado de influencia en todo el sistema (Kool et al., 2013). Esto hace que las métricas de tesela tengan un uso muy eficiente en planificación y conservación (Keeley et al., 2021), aunque cada métrica tiene unas características distintas. Esto hace que para la conservación lo idóneo es que se tengan en cuenta varias métricas, ya que cada una aporta información distinta. Todo esto hace que se sigan desarrollando nuevas métricas para conseguir así obtener la mayor cantidad (y variedad) de información posible que permitan tomar las mejores decisiones (Petsas et al., 2021).

**Bosques secos tropicales a nivel mundial**

El bosque seco tropical (BST) es posiblemente uno de los ecosistemas más amenazados del mundo (Hoekstra et al., 2005; Portillo-Quintero y Sánchez-Azofeifa, 2010), y con menos superficie remanente (menos del 25%) (Ferrer-Paris et al., 2018), esto podría indicar que ya ha llegado o está llegando a su punto de colapso (Arroyo-Rodríguez et al., 2020). Los bosques secos tropicales están desapareciendo a un ritmo alarmante, presentando unas tasas excepcionalmente altas de cambios en el uso de la tierra y por cambio de clima (Siyum, 2020). Las principales amenazas del bosque seco son el cambio climático, el fuego, la fragmentación y la conversión a agricultura (Miles et al., 2006).

Los bosques secos tropicales y subtropicales se encuentran en México, el este de Bolivia y el centro de Brasil, los valles de los Andes del norte, a lo largo de las costas de Ecuador y Perú, el Caribe, África sudoriental, las islas menores de la Sonda , India central, Indochina, Madagascar, y Nueva Caledonia (World Wildlife Fund, 2021) (Fig. 5). Por continentes, principalmente se encuentran en Sudamérica (54,2%) seguido de norte y centro américa (12,5%), África (13,1%), Eurasia (16,4%), y por último Australia y la región insular del sudeste asiático (3,8%). Con un estimado total de 1.048.700 $km^2$ de bosque seco a nivel mundial (Miles et al., 2006).

Los BST tienen múltiples definiciones (Tabla 1), aunque genéricamente se pueden considerar que crecen en zonas que presentan una temperatura media anual mínima de 17 °C, con precipitación media anual comprendida entre 250 mm y 2000 mm, y al menos tres meses de sequía, y donde la mayoría de los árboles son caducifolios (Guerra-Martínez et al., 2020).

Tradicionalmente, la investigación sobre bosques secos es mucho menor que la investigación en bosques tropicales húmedos, donde la financiación internacional ha sido tradicionalmente mayor (Sanchez-Azofeifa, et al 2005). Esto ha llevado a que los bosques secos se hayan estudiado menos que las selvas tropicales húmedas vecinas, con una proporción de aproximadamente un estudio de bosque seco por cada seis de selvas tropicales (Sánchez-Azofeifa et al., 2005), aunque esta proporción ha disminuido en los últimos años hasta una ratio de 4,5. Sin embargo, los estudios tropicales que se centran en ecosistemas secos siguen siendo solo el 10% del total (Escribano-Ávila et al., 2017).

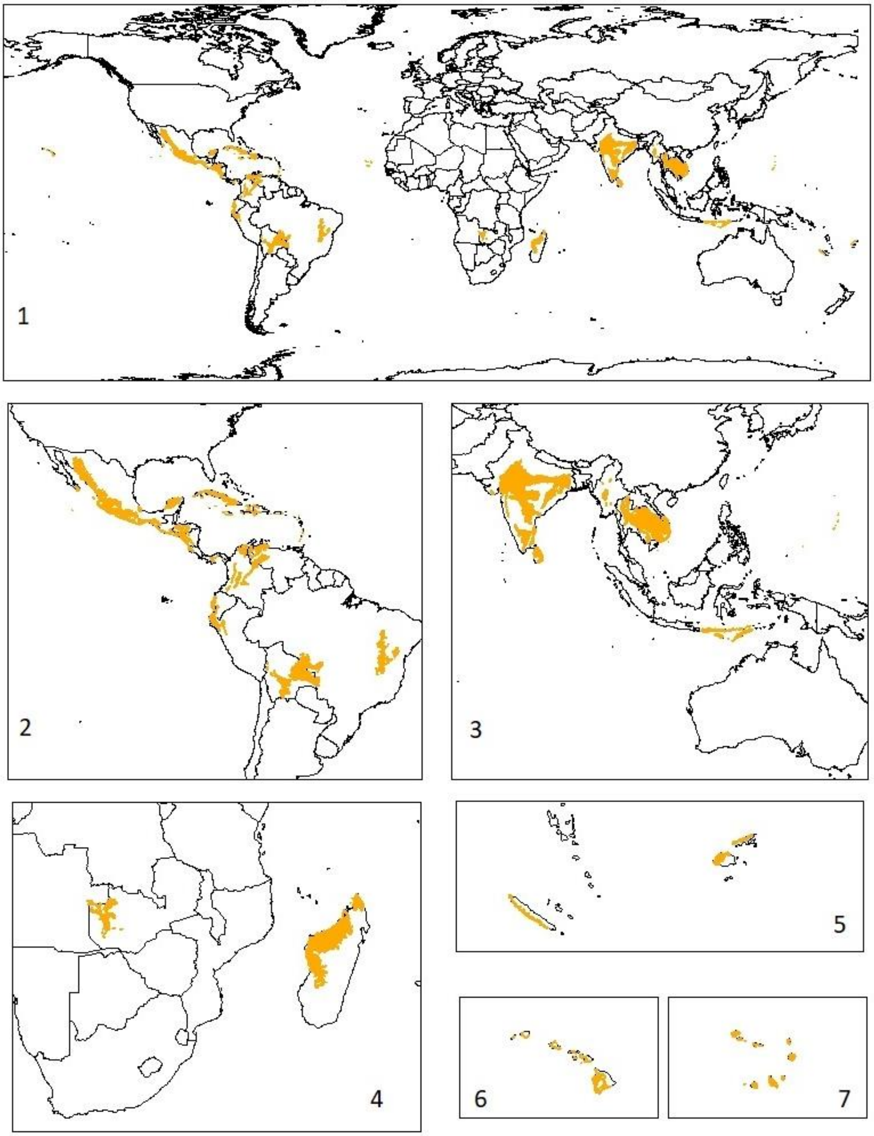


**Figure 5:** 1) Distribución mundial del bosque tropical seco según WWF. 2) Distribución del bosque seco en América. 3) Distribución del bosque seco en Asia. 4) Distribución del bosque seco África. 5) Distribución del bosque seco en Nueva caledonia y Fiyi. 6) Distribución del bosque seco en Hawái. 6) Distribución del bosque seco en Cabo verde.

**Tabla 1:** Algunas definiciones del bosque seco tropical, adaptado de Siyum (2020).

| Definición o concepto. | Fuente. |
|---|---|
| Los BST se encuentran en regiones tropicales de una pronunciada estacionalidad con periodos de sequía de poca o nula precipitación. | Mooney et al. (1995). |
| Los BST se encuentran en regiones libres de heladas con 500 a 2000 mm de precipitación anual con una estación seca de 4 a 7 meses. | Walter (1971), Murphy y Lugo (1986) y Miles et al. (2006). |
| Los BST se encuentran en regiones con clima tropical con un periodo seco de 5 a 8 meses y con precipitaciones anuales entre 500 y 1500 mm | FAO (2001, 2012). |
| Los BST se encuentran en regiones con un clima estacional con periodo seco de como mínimo tres meses con anuales entre 250 y2000 mm | Menaut et al. (1995), Mayaux et al. (2005), Meir y Pennington (2011). |
| Los BST se encuentran en regiones con un clima estacional con periodo seco de 3 meses como mínimo al año con precipitación <100 mm por mes, la precipitación anual total oscila entre 700 y 2000 mm y temperatura media anual es ≥ 25 ° C, presentando con al menos el 50% de los árboles caducifolios de sequía. | Sanchez-Azofeifa et al. (2005). |
| Los BST se encuentran en regiones con clima similar al bioma de sabana, pero presentan diferencias de vegetación, al presentar los BST una vegetación arbórea alta (> 10 m) y con una capa de árboles intermedias tolerantes a la sombra, con un piso de hojarasca y zonas de herbáceas o gramíneas. | Charles-Dominique et at. (2015) y Ratnam et al.(2011). |

**Ecosistemas de Ecuador: bosques secos y deforestación.**

Ecuador es uno de los países más mega diversos del mundo (Lessmann et al., 2014), ubicado en Sudamérica, con frontera con Perú y Colombia (Fig. 6.A), con una extensión de 257.148 $km^2$ (Ministerio del Ambiente de Ecuador, 2015a), y una población de más 17,5 millones de personas (Instituto Nacional de Estadística y Censo, 2021). Presenta 4 regiones de una alta biodiversidad, que son las islas Galápagos, la Amazonía, la Sierra o región Andina, y la Costa (Varela y Ron, 2019) (Fig. 6. B). Esta última región presenta dos grandes tipos de bosques, el bosque húmedo, mayoritariamente al norte, y el bosque seco principalmente en el sur (Fig. 6.C) (Ministerio del Ambiente del Ecuador 2013). La región de la Costa ecuatoriana se encuentra localizada entre la región montañosa de los Andes, al este, y el océano Pacífico, al oeste, al norte del río Jubones a 300 m sobre el nivel del mar y al sur del mismo a 400 m sobre el nivel del mar (Ministerio del Ambiente del Ecuador, 2013).

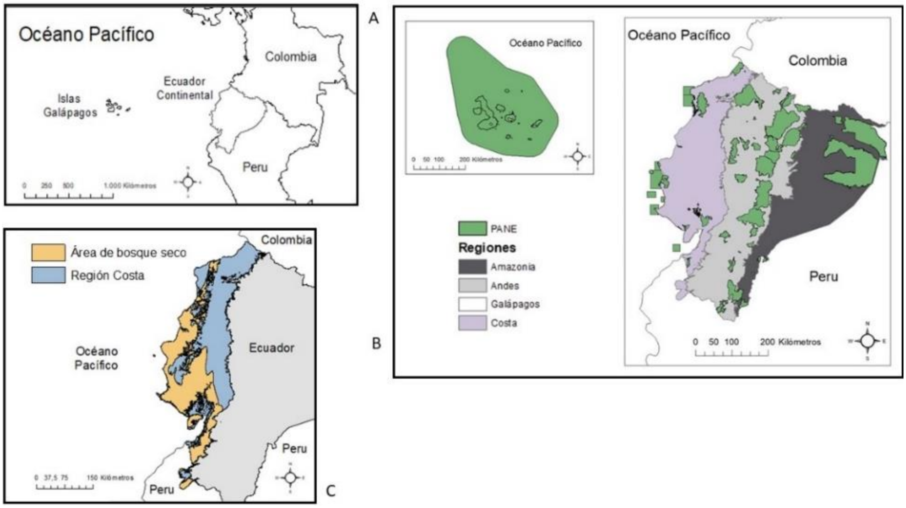


**Figura 6**: A) Ubicación de Ecuador en Sudamérica. B) Regiones de Ecuador y Patrimonio de Áreas Protegidas del Estado (PANEs). C) Región costa mostrando la zona de bosque seco tropical.

A pesar de que la Costa y la vertiente oriental de los Andes son *hotspots* mundiales de biodiversidad (Myers et al., 2000), y la Amazonía y Galápagos son regiones también altamente diversas, Ecuador continental presentó una deforestación bruta anual promedio 94.353 ha $año^{-1}$ entre 2014 y 2016, y el país completo ha liderado las tasas de deforestación en Sudamérica entre 1990-2000 y 2000-2010, con valores anuales de -1,5 y -1,8% respectivamente (FAO, 2011). Esto supone que ha perdido entre 1990 y 2020 una superficie de

21.340 km$^2$ de bosque, lo que supone aproximadamente un 15% de sus bosques (FAO, 2020b).

La región de la Costa ha sido la que más ha sufrido la deforestación (Sierra, 2013), siendo la región que lidera la industria ganadera y agrícola del país (Instituto Nacional de Estadística y Censo, 2019a). El bosque seco de esta región ha liderado las tasas de deforestación en Ecuador (Fig.7), con una reducción de sus bosques del 1,4% anual entre 2008 y 2014 (Manchego et al., 2018), y se estima que más de 60% de su superficie ha sido destruida por actividades humanas (Ron, 2020). Esta deforestación ha llevado al bosque seco del Tumbes-Piura a un estado de peligro crítico de extinción según los criterios de la UICN (Ferrer-Paris et al., 2019).


**Figura 7:** A) Deforestación del bosque seco y posterior quema del terreno para establecer algún tipo de cultivo. Autor: José Guerrero. B) Deforestación del bosque para ser sustituido por terreno agrícola Autor: José Guerrero.

Los bosques secos estacionales en Ecuador suelen crecer en zonas con una precipitación anual de 400 a 600 mm en un período de 3 a 4 meses, generalmente en febrero, marzo y abril; la temperatura media anual es de 24,9 °C y la evapotranspiración potencial es de 1783 mm año$^{-1}$ (Ministerio del Ambiente del Ecuador, 2012). Estas condiciones climáticas extremas hacen que los árboles pierdan sus hojas durante la época seca, de ahí su nombre bosque seco, con una fenología decidua o semidecidua, por eso también se les llama bosques secos deciduos (o semideciduos) o estacionales (Fig. 8). En los bosques deciduos propiamente dichos, el 75% de los individuos arbóreos o matorrales pierden sus hojas durante la época seca, que dura entre 6 y 8 meses. En la fenología semidecidua, son entre el 75% y el 25% los que pierden las hojas y los periodos secos duran menos, entre 1 y 6 meses (Prentice, 1990; Ministerio del Ambiente del Ecuador, 2013; Rivas et al., 2020).


**Figura 8:** Fisionomía del Bosque seco ecuatoriano. A y B) Imágenes en época seca donde se observa que los árboles han perdido sus hojas Autor: José Guerrero y Lilian Sosa. C y D) Imágenes en época lluviosa donde se observa la cubierta vegetal completa típica del bosque seco. Autor: José Guerrero y Emilio Jarre.

La región Tumbesina presenta un alto grado de endemismo, con 16 y 39 especies endémicas de mamíferos y aves respectivamente (Loaiza, 2013; BirdLife International, 2019), así como un alto valor de diversidad florística, con aproximadamente el 80% de sus componentes endémicos (Ministerio del Ambiente del Ecuador, 2012), siendo uno de los lugares con mayor probabilidad de encontrar nuevas especies (Moura and Jetz, 2021).

Con el objetivo de conservar esta diversidad, Ecuador tiene una red de áreas protegidas llamado Sistema Nacional de Áreas Protegidas (SNAP), dentro del cual se encuentran las denominadas "Patrimonio de Áreas Naturales del Estado" (PANE). Las PANE se encuentran distribuidas muy desigualmente, ya que en la Costa son áreas relativamente pequeñas, que cubren sólo un 5% de la región, y por contraste, los Andes y la Amazonía presentan una protección del 20,6% y del 23,2% respectivamente (Lessmann et al., 2014), mientras que la región insular o islas Galápagos tienen protegidos un 97% de su superficie terrestre y una extensión total (marina y terrestre) de 133 mil kilómetros cuadrados, lo que la convierte en uno de las 10 unidades de protección más grandes del mundo (Parque Nacional Galápagos, 2021) (Fig. 6.B).

**Objetivos y estructura de la tesis.**

Basado en la información mostrada en la introducción, la hipótesis general de esta tesis fue que el bosque seco ecuatoriano se está deforestando, aumentando la fragmentación y perdiendo conectividad, presentando un grado de protección muy bajo. A partir de esta hipótesis, el objetivo principal de la tesis doctoral fue conocer el grado de protección del bosque seco ecuatoriano y estudiar la evolución de su deforestación, fragmentación y conectividad.

**Objetivos específicos:**

Debido a la baja protección de la región de la Costa ecuatoriana con respecto al resto del país, así como a la marginación en los estudios del bosque seco en comparación con el bosque siempre verde, en el **capítulo uno**, utilizando datos del Ministerio del Ambiente, se realizó una comparación del estado actual de conservación y de protección de ambos tipos de bosque en la costa ecuatoriana, donde son los ecosistemas predominantes, con el objetivo de analizar si existe discriminación en el grado de protección no justificada por su estado actual de conservación.

Aunque el bosque seco tropical ecuatoriano ha sido fuertemente deforestado en las últimas décadas, no hay estudios detallados sobre la fragmentación de este ecosistema. Por este motivo, el **capítulo dos** tuvo como objetivo evaluar la fragmentación del bosque seco estacional ecuatoriano entre 1990 y 2018, con el objetivo final de detectar las zonas más fragmentadas y más vulnerables a desaparecer, para de esa manera contribuir a desarrollar planes efectivos de conservación.

Debido a la gran variedad de métricas de fragmentación disponibles, a la disparidad de opiniones y de interpretaciones, y a los parámetros y consecuencias que estudian, en el **capítulo tres** se agruparon diversas métricas con el fin de generar una única fórmula de cálculo del estado de fragmentación a nivel de tesela que pudiera extrapolarse al paisaje, con el objetivo de dar una nueva visión de la fragmentación más holística, y conseguir así una mejor interpretación de la fragmentación del bosque seco y de su evolución.

Debido a la alta deforestación, y fragmentación del bosque seco ecuatorial, los efectos que ambas perturbaciones pueden tener sobre la conectividad funcional del ecosistema, y a la falta de estudios, **en el capítulo cuatro** se analizó la evolución de la conectividad funcional entre 1990 y 2018 en los bosques secos costeros de Ecuador, con el objetivo de conocer su estado por zonas, teselas y de forma global, lo cual permite una mejor toma de decisiones para su conservación, posibles repercusiones en el ecosistema y así poder avanzar en el estudio de este parámetro tan importante para la conservación y planificación del terreno.

# Capítulo 1

## Dry forest is more threatened but less protected than evergreen forest in Ecuador's coastal region.

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Capítulo publicado en Environmental Conservation (2020) 47: 79–83. https://doi.org/10.1017/S0376892920000077

**Summary**

The Ecuadorian coast has two main types of forests, which are differentiated by their phenology: dry forests are deciduous and more humid forests are evergreen. Less of the dry forest on the Ecuadorian coast is protected (13% of its area) than the evergreen forests (28%), and the area devoted to the protection of dry forest (1069 $km^2$) is substantially less than to the protection of evergreen forests (2800 $km^2$). Yet, the conservation status of dry forests is more critical, with 83% of their area classified as very low connectivity, 70% as highly fragile and 86% as highly threatened. In addition the dry forest has fewer protected areas than the evergreen forests. It is therefore necessary to increase the protection of deciduous ecosystems as part of a comprehensive national strategy, because they support high levels of biodiversity and many endemic species.

**Keywords:** deciduous forests; dry forests; evergreen forests; forest conservation; protected areas.

## 1. Introduction

The human population of Ecuador, which is one of the most diverse countries on the planet (Lessmann et al., 2014), has grown by 452% in 60 years, reaching 17 million inhabitants in 2019 (Instituto Nacional de Estadística y Censo, 2019b). This growth has resulted in the fragmentation of many natural ecosystems. The Ecuadorian coastal ecosystems are the most susceptible, due to their low resilience to human pressures (Ministerio del Ambiente de Ecuador, 2015a), and Ecuador has experienced a high rate of deforestation; almost 19,000 $km^2$ of natural forest were lost between 1990 and 2008, *c*. 70% of the loss being in the 1990s, with an average annual net deforestation of 1292 $km^2$ (Sierra, 2013). The most affected areas are located in the Coast Region, especially in dry forests, which are the most threatened forest type in the region (Manchego et al., 2018).

The dry forest is important for provides the inhabitants of the area with wood products for structures and occasionally for market, resulting in the degradation of the forest structure, functionality and dynamics (Ministerio del Ambiente del Ecuador, 2012). The tropical forests of the Tumbesian region are characterized by a high degree of endemism (Espinosa et al., 2012); they harbour 16 endemic mammals (Loaiza, 2012) and 39 endemic bird species (Bird Life International, 2019). Endangered mammals such as the Sechuran fox (*Pseudalopex sechurae* Thomas) and the white-tailed deer (*Odocoileus peruvianus* Gray) are confined to the dry ecosystems, while the critically endangered white-fronted Ecuadorian capuchin monkey (*Cebus aequatorialis* Allen) frequently occurs in the dry forests (Guerrero-Casado et al., 2020).

Dry forests in the Neotropics have traditionally been studied less than the neighbouring rainforests, with a ratio of approximately one study of dry forests per six of rainforests (Sanchez-Azofeifa et al., 2005). Although this ratio increased slightly to one research paper on tropical dry forests per 4.5 on rainforests during 1997-2014, tropical studies that focus on dry ecosystems are still only 10% of the whole (Escribano-Avila et al., 2017). The Ecuadorian tropical dry forest in particular has been studied significantly less than the other Ecuadorian ecosystems (Escribano-Ávila, 2016). Moreover, scientific data from these forests is very skewed towards certain areas, while most areas remain completely unsurveyed (Escribano-Ávila, 2016). This paucity of data regarding the status, biodiversity and ecological processes of theses dry forests systems, as well as the ecosystem services they may provide to surrounding human communities, leaves researchers, conservationists, and policy makers ill-equipped to address the many threats facing this forest type.

Previous works have suggested that priority conservation areas should be established in the Coastal Region of Ecuador, because of its low representatives in the protected areas (PAs), its high degree of threat, the huge land-use changes, its great diversity and high vulnerability (Sierra et al., 2002; Lessmann et al., 2014; Cuesta et al., 2017). Particularly, the dry forests of the Coastal Region are the ecosystems with the poorest representation in the PAs at a national level (Sierra et al., 2002).

Given the accelerated rate of deforestation has resulted in protected areas often becoming the last refuge for threatened species and natural ecosystem processes (Laurance et al., 2012), networks of protected areas should be representative, encompassing all relevant biodiversity targets (D'Aloia et al., 2019; Margules and Pressey, 2000). Nevertheless, the declaration of a certain area as protected is affected by multiple factors including its representativeness, degree to which it is endangered, the species harboured in that area, community support (both local and international), aesthetic issues and the ecosystem services. Phenology of the ecosystems could also be considered a determining factor; ecosystems differ in their phenology, features that may be influential for establishing conservation priorities. In this study, we tested the hypothesis that dry forests may be significantly underrepresented in Ecuador's network of protected areas when compared to evergreen forests. We particularly hypothesise that dry forests in Ecuador (deciduous and semi-deciduous forests) may be less protected owing to the fact that they are apparently of less conservation value (particularly during the dry season) than evergreen forests. Additionally, we hypothesized that dry forests may be facing greater anthropogenic threats and degradation than evergreen forests. To address these questions, we compared the area devoted to the protection of dry (deciduous and semi-deciduous) forests and evergreen forests in the Coastal Region of Ecuador. We further compared their degree of fragmentation, vulnerability, connectivity, threat and fragility, with the aim of understanding the conservation status of these forest types and relationship between forest type and degree of protection.

## 2. Methods

### 2.1 *Study area*

Our study area included the entirety of the Coastal Region of Ecuador, defined as the region located along the Pacific Ocean and west of the Andes mountain range, north of the Jubones River at 300 m above sea level and south of it at 400 m above sea level (Ministerio del Ambiente del Ecuador, 2013) (Fig. 9). The ecosystems in the Coastal Region of Ecuador can be classified into dry and evergreen forest phenologies (Ministerio del

Ambiente del Ecuador, 2013). Dry or deciduous forests (*sensu lato*) have deciduous (*sensu stricto*) and semi-deciduous phenology, in which the dry periods last up to eight months and 25%-75% of trees and shrub species lose their leaves during the dry season (Prentice, 1990). Evergreen forests (*sensu lato*), which include seasonal evergreen (Josse et al., 2003) and evergreen forests (*sensu stricto*), refer to the types of vegetation with dry seasons (periods of low or no precipitation) that last less than a month per year and where more than 75% of trees and shrub species maintain foliage throughout the year (Ministerio del Ambiente del Ecuador, 2013).

The Ecuadorian state divides the natural spaces into protective forests (PFs) and PAs, the latter being further subdivided. PFs are forests that are located in areas of rugged topography, in headwaters of watersheds or in areas that, owing to their climatic conditions (edaphic and hydric) are not suitable for agriculture or livestock, and whose functions are to conserve water, soil, flora and fauna. There are 202 PFs in Ecuador, which cover an area of 2,425,002 ha, representing 9.72% of the national territory (Ministerio del Ambiente, 2019a). Thirty-nine of the 202 PFs are located in the Coastal Region, representing 19% of all PFs (Fig. 9).

Furthermore, the national system of PAs (Patrimonio de Areas Naturales del Estado [PANE]) is the group of natural areas that ensure coverage and connectivity of important terrestrial, marine and coastal marine ecosystems, as well as cultural resources and main water sources (Ministerio del Ambiente del Ecuador 2016). The PANE covers the four regions of the country and consists of 56 PAs that extend over *c*. 20% of the area of Ecuador. However, distribution of the PANE areas is spatially uneven, and although the Coastal Region harbours 22 of them, these are relatively small, and cover only 5% of the Coastal Region, which is significantly less than in the Andes (20.6 %) and Amazon (23.2 %) (Lessmann et al., 2014). These PAs within the Coastal Region are focused on the largest patches of remaining vegetation (Fig. 9).

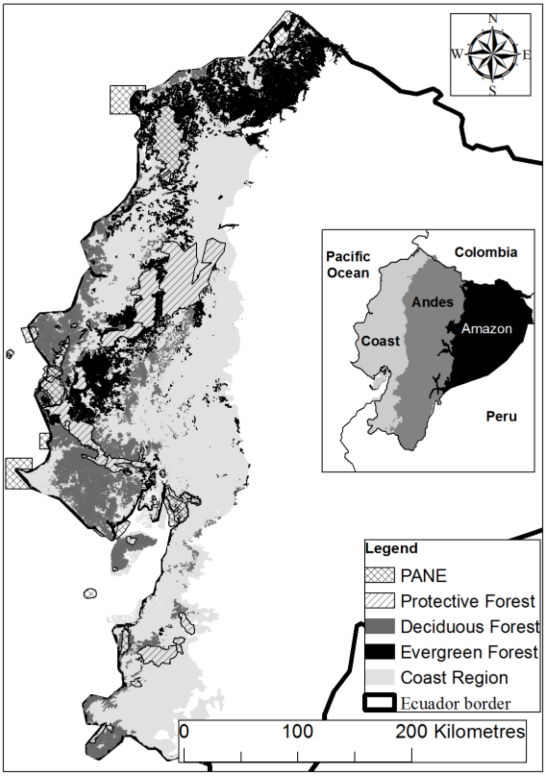


**Figure 9:** Map of the Ecuadorian coast showing dry forests, evergreen forests, Patrimonio de Areas Naturales del Estado (PANE) and protective forests.

2.2 *Analysis of ecosystem features*

Various layers of geographic information obtained from the Ecuadorian Ministry of Environment were used to address our research question: biogeographic sectors; phenology; ecosystems, protected areas (PANE) and protective forests (available at http://mapainteractivo.ambiente.gob.ec/portal/). The first step was to overlay the layers of the ecosystems with the PAs and PFs in the littoral zone in order to assess correlations between phenology and protected status in Ecuador. This was done by grouping the ecosystems into dry or evergreen forests. We did not consider either mangroves or grasslands when making this calculation. Moreover, we calculated the proportion of the area with different land uses (classified as deciduous, evergreen, mangrove, intervention and others) included in the PANE areas and PFs separately.

The shapefile of the ecosystems also contains information concerning the fragmentation, connectivity, vulnerability, threat and fragility of each ecosystem, which were classified into different categories (e.g. high, medium, low) for each of these five indicators. The degree of fragmentation of each ecosystem was calculated using the number of patches, their mean size and the coefficient of variation among them (Ministerio del Ambiente de Ecuador, 2015a) using the Patch Analysis Tool in ArcGIS and Fragstat software (Elkie et al., 1999; McGarigal et al.,2002). The fragmentation index of each ecosystem was ranked in four levels (very high, high, medium and low) according to the method of Jenks' natural breaks (Ministerio del Ambiente de Ecuador, 2015a). The connectivity was measured using Conefor 2.6 (Saura, 2006) software by means of the Equivalent Connected Area Index (Ministerio del Ambiente de Ecuador, 2017) which is defined as the area that should have a hypothetical and single continuous patch of forest (fully connected), corresponding to the same probability of connectivity as the set of patches of a habitat or ecosystem evaluated (Saura et al., 2011). The connectivity index was classified into five categories (high, medium, low and very low) also using Jenks' natural breaks (Ministerio del Ambiente de Ecuador, 2017). The vulnerability of each ecosystem was calculated by a combined weighted index using the number of species listed in CITES (CT), the number of endemic species (EN), the number of plants with a commercial value (CV), and the number of endangered plant species harboured according to the IUCN red list for Ecuador (IUCN), together with the representativeness in Ecuador in terms of surface (RE) and in terms of conservation (RC) by means of protected area, the fragmentation (FR) and the connectivity (CN) of the ecosystem. All these variables were combined to obtain an overall vulnerability index (Ministerio del Ambiente de Ecuador 2015a) as follows:

$$\text{Vulnerability species} = (\text{EN} \times 0.6) + (\text{IUCN} \times 0.2) + (\text{CT} \times 0.24) + (\text{CV} \times 0.4)$$

$$\text{Vulnerability Landscape} = (\text{RE} \times 0.23) + (\text{RC} \times 0.24) + (\text{FR} \times 0.29) + (\text{CN} \times 0.24)$$

$$\text{Vulnerability Inde} = (\text{Vulnerability species} \times 0.45) + (\text{Vulnerability Landscape} \times 0.55)$$

The Vulnerability Index obtained was divided into three quantiles (high, medium and low).

The threat to the ecosystems was evaluated by another combined weighted index employing five variables: water resource use (WR), climate change impact (CC), forest exploitation (FE), extraction of natural resources (ENR), and the probability of land conversion (PC). Thus:

$$\text{Threat} = (\text{CC} \times 0.11) + (\text{FE} \times 0.12) + (\text{WE} \times 0.11) + (\text{ENR} \times 0.34)$$

Finally, Vulnerability and Threat were combined to obtain five levels of fragility (very high, high, medium, low and very low; Table 2) (Ministerio del Ambiente de Ecuador, 2015b). In these five indicators, no values were assigned to areas identified as non-natural ecosystems, such as crops, urban areas or intervening forests (for more details about the methodology, see Ministerio del Ambiente de Ecuador, 2015a, 2015b, 2017).

**Table 2:** The five categories of fragility according to the combination of the threat and vulnerability indexes (Ministerio del Ambiente de Ecuador, 2015b).

| Vulnerability | | **Threat** | |
|---|---|---|---|
| | **High** | **Medium** | **Low** |
| **High** | Very High | High | Medium |
| **Medium** | High | Medium | Low |
| **Low** | Medium | Low | Very Low |

For each ecosystem, we calculated the percentage of fragmentation, connectivity, fragility, vulnerability and threat according to the area under each category. Finally, we compared the values for each category between deciduous and evergreen forests.

## 3. Results

### 3.1 *Degree of protection*

The remaining deciduous forests have less protection (12.9%) than evergreen forests (27.9%) (Table 3), and deciduous phenology is less represented in (PFs; 11.3 %) and PANE areas (14.72 %) than evergreen phenology and mangroves (Table 4). It can also be observed that the percentage of intervention is greater in PFs (54.31 %) than in PANE areas (22.8 %).

**Table 3:** Area and the percentage protected by the PANE and protective forest (PF) according to phenology in the whole region (overall), and in the remaining forests of the Ecuadorian Coast Region.

| | Total ($km^2$) | PANE ($km^2$) | PANE (%) | PANE+PF ($km^2$) | PANE+PF (%) |
|---|---|---|---|---|---|
| Overall | | | | | |
| Deciduous | 30426.19 | 1443.01 | 4.74 | 3078.76 | 10.12 |
| Evergreen | 33535.99 | 2584.61 | 7.71 | 6448.15 | 19.23 |
| Remaining Forests | | | | | |
| Deciduous | 8288.63 | 446.51 | 5.39 | 1069.07 | 12.90 |
| Evergreen | 10018.24 | 1097.80 | 10.96 | 2801.42 | 27.96 |

**Table 4**: Percentage of forests by phenology within national system of protected areas (PANE) and protective forests.

| Phenology | PANE | | Protective Forests | |
|---|---|---|---|---|
| | Area (km²) | % | Area (km²) | % |
| Deciduous | 446.51 | 14.72 | 622.55 | 11.30 |
| Evergreen | 1097.80 | 36.18 | 1703.62 | 30.91 |
| Intervention | 691.81 | 22.80 | 2992.84 | 54.31 |
| Mangrove | 651.40 | 21.47 | 0 | 0 |
| Others | 146.39 | 4.83 | 192.02 | 3.48 |
| Total | 3033.91 | 100.00 | 5511.04 | 100.00 |

3.2 *Analysis of ecosystem features*

Among fragmentation, connectivity, threat, vulnerability and fragility, fragmentation is the most severe and widespread of all of the factors; both evergreen and deciduous phenologies show high to very high degrees of fragmentation covering the majority of the area. Additionally, more than 80% of the area under deciduous ecosystems is classified as very low connectivity; 70% of the deciduous phenology is classified as very fragile, compared to only 0.2% of the evergreen; and 86% of the area under deciduous ecosystems is classified as highly threatened as opposed to only 0.6% of the evergreen. However, the evergreen ecosystems showed higher values of vulnerability (Fig. 10).

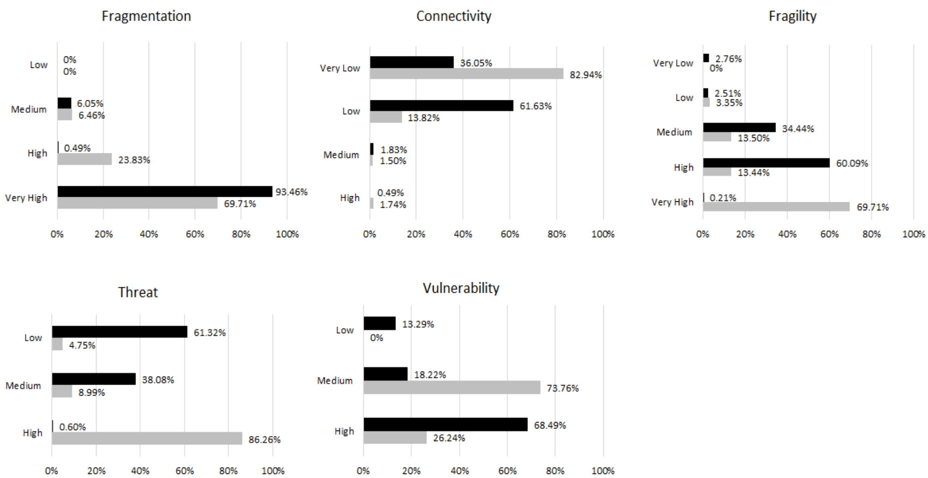


**Figure 10:** Percentages of the area with the different degrees of fragmentation, connectivity, fragility, vulnerability and threat of the ecosystems in the coastal region of Ecuador according to their phenology: dry forests (grey) and evergreen forests (black).

## 4. Discussion

According to our data, the proportion of dry forests covered by PAs is much lower than that of the evergreen forests in this region. This bias could be attributed to the lower number of species which inhabit the dry forests (Sierra et al., 2002) as well as its lower landscape value, particularly during the dry season.

The dry forest is one of the most threatened ecosystems in the Americas; specifically, the Tumbes-Piura dry forest ecosystem is critically endangered (Ferrer-Paris et al., 2018). Despite the threat, it seems to receive less attention than the other Ecuadorian ecosystems. For instance, according to our calculations, despite the fact that the mangroves cover much less area than the deciduous forests, the PANE devotes more area to protecting mangroves than deciduous forests (Table 4). Additionally, our results show that dry forests have a lower coverage of PAs than evergreen forests, even though these dry forests have more critical conservation status, because more of their areas is classified as having a high threat and very high fragility, and being highly fragmented with very little connectivity. Fragility is the most worrying indicator, since it encompasses the other parameters, and 70% of the area of dry ecosystems has very high levels (Fig. 10). This suggests that deciduous ecosystems are much more subject to anthropogenic pressures, and that the designation of remaining dry forests as PAs is necessary for their conservation. Nevertheless, the protection of dry forests could be insufficient to ensure the mitigation of deforestation, since in Ecuador this is still occurring within PAs (van Der Hoek, 2017). More comprehensive conservation strategies are needed to effectively reduce the deforestation rate. In particular, the PF is an ineffective conservation tool, since in the Coastal Region more than half of its area is classified as being subject to land use (Table 3).

It is important to recognize that the Vulnerability Index of the evergreen forests status is also worrisome, but the calculation of this parameter takes into account the number of endemic species, threatened species, species with a commercial value, and species listed in CITES (Ministerio del Ambiente 2015a), values that are expected to be greater overall in evergreen forests. When considering conservation priorities, the deciduous forests are practically limited to the coastal region and, therefore, the loss of these remnant dry forests would lead to the extinction of this ecosystem in Ecuador.

In summary, deciduous forest systems face higher degrees of anthropogenic threat yet receive much less official protection and conservation interventions than evergreen forests in coastal Ecuador. While more than 80% of their original area has already been deforested (Sierra, 2013), the remaining dry ecosystems in the Coastal Region of Ecuador are included in the Choco/Darien/Western Ecuador biodiversity hotspot (Myers et al., 2000) and harbour high levels of endemic species and others with constrained distributions (Sierra et al., 2002; León-Yánez et al.,2011). Despite their unique value in terms of biodiversity, anthropogenic factors continue to degrade these already highly threatened and fragile dry forests. Since the evergreen forests have received more attention by researchers and natural resources managers, conservation efforts should now be more focused on the preservation of the remaining dry ecosystems, protecting them as part of a more complex regional conservation plan in which sustainable exploitation by the local communities improves their income and preserves biodiversity (Escribano-Avila et al., 2017). This plan is vital to preventing continued anthropogenic habitat loss, partly due to lack of protection, and the establishment of new protected area networks should also ensure the connectivity of these ecosystems due to the high levels of fragmentation.

**Acknowledgements.** This work was carried out in the frame of the project Conservation Status and Threats of Wild Mammals in the Dry and Humid Tropical Forest in Manabí – Ecuador (AP-C1-2018-FCV0012), funded by the 'Universidad Técnica de Manabí'.

# Capítulo 2

## Deforestation and fragmentation trends of seasonal dry tropical forest in Ecuador: impact on conservation

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Capítulo publicado en Forest Ecosystems (2021) 8: 1–13. https://doi.org/10.1186/S40663-021-00329-5

**Abstract**

**Background:** Fragmentation and deforestation are one of the greatest threats to forests, and these processes are of even more concern in the tropics, where the seasonal dry forest is possibly one of the most threatened ecosystems with the least remaining surface area.

**Methods:** The deforestation and fragmentation patterns that had occurred in Ecuadorian seasonal dry forests between 1990 and 2018 were verified, while geographic information systems and land cover shapes provided by the Ecuadorian Ministry of the Environment were employed to classify and evaluate three types of seasonal dry forests: deciduous, semi-deciduous, and transition. The study area was tessellated into 10 $km^2$ hexagons, in which six fragmentation parameters were measured: number of patches, mean patch size, median patch size, total edge, edge density and reticular fragmentation index (RFI). The RFI was also measured both outside and inside protected natural areas (unprotected, national protected areas and protected forest). Moreover, the areas with the best and worst conservation status, connectivity and risk of disappearance values were identified by means of a Getis-Ord Gi* statistical analysis.

**Results:** The deforestation of seasonal dry forests affected 27.04% of the original surface area still remaining in 1990, with an annual deforestation rate of -1.12% between 1990 and 2018. The RFI has increased by 11.61% as a result of the fact that small fragments of forest have tended to disappear, while the large fragments have been fragmented into smaller ones. The semi-deciduous forest had the highest levels of fragmentation in 2018. The three categories of protection had significantly different levels of fragmentation, with lower RFI values in national protected areas and greater values in protected forests.

**Conclusions:** The seasonal dry forest is fragmenting, deforesting and disappearing in some areas. An increased protection and conservation of the Ecuadorian seasonal dry forest is, therefore, necessary owing to the fact that not all protection measures have been effective.



**Keywords:** Deciduous Forest, Semi-deciduous Forest, Remnant forest, Patch isolation, Habitat loss, Protected areas.

## 1. Introduction

The term 'forest fragmentation' refers to the spatial configuration and amount of treed-vegetation (Hermosilla et al., 2018), a landscape-level process during which anthropogenic factors progressively subdivide forest tracts into (initially, but not necessarily ultimately) smaller, geometrically more complex and more isolated patches as a result of natural processes and land use activities (McGarigal and Marks, 1995; Chakraborty et al., 2017). This concept can refer to the entire process of forest loss and isolation or, more specifically, to changes in the spatial configuration of remnants of forest that are the result of deforestation (Fahrig, 2003; Kupfer, 2006). The fragmentation process involves changes in the composition, structure and function of the landscape, and occurs on a mosaic background of natural patches created by changing landforms and natural disturbances (McGarigal and Marks, 1995; Asbjornsen et al., 2004).

At the landscape level, the most common effect of fragmentation is the formation of new edges or the modification of existing ones, which play a fundamental role in the structure and functioning of ecosystems (Forman and Godron, 1989; Asbjornsen et al., 2004). These changes can alter ecological functions related to biodiversity, the nutrient cycle and the hydrological cycle, and may even affect the microclimate of the area (Asbjornsen et al., 2004; Taubert et al., 2018).

The increase in forest fragmentation is one of the main threats to natural tree populations in the tropics around the world (Trejo and Dirzo, 2000; Fuchs et al., 2003), where large areas of forests have been transformed into pastures and crops, thus creating a mosaic of agricultural areas and forests in which forests remain as small scattered patches (Asbjornsen et al., 2004; Taubert et al., 2018). The tropical forests in South America underwent a net loss of 2.6 million hectares in the 2010-2020 period, although the deforestation rate has decreased significantly when compared to 2000-2010 (FAO, 2020a). More specifically, Ecuador maintained the highest deforestation rates in South America during the periods 1990-2010, with annual rates of between -1.5% to-1.8% (FAO, 2011) and with an overall deforestation of 21,340 km$^2$ between 1990 and 2020 (FAO, 2020b). One consequence of this intensive fragmentation is that 47 ecosystems of mainland Ecuador have been classified as very-highly or highly fragmented, i.e. 30% of the natural areas (Ministerio del Ambiente de Ecuador, 2015). Those most affected are located in the coastal region, in which there was an area of annual deforestation of 678.13 $km^2$ between 1990 and 2008 (Sierra, 2013). The deforestation and degradation of the seasonal dry forests in this region have been particularly intense, thus making them the most threatened type of forest in the country, in addition to being less protected than the evergreen forests (Manchego et al., 2018; Rivas et al., 2020). Deforestation has, in fact, become the greatest threat to seasonal dry forest ecosystems in Ecuador, with an average change in area reduction of 1.4% per year between 2008 and 2014 (Tapia-Armijos et al., 2015; Manchego et al., 2018). Indeed, tropical dry forests are among the most threatened ecosystems in the world (Hoekstra et al., 2005; Portillo-Quintero and Sánchez-Azofeifa, 2010), and are the ecosystems of which the least amount of original surface remains (less than 25%) (Ferrer-Paris et al., 2018). This deforestation has, according to the IUCN criteria, led the equatorial dry forest to be classified as in critical danger of extinction (Ferrer-Paris et al., 2018), and approximately 70% of the remaining surface has very high levels of fragmentation (Rivas et al., 2020). Intense deforestation is consequently considered to be the main threat to the biodiversity of the tropical seasonal dry forests of the Tumbension region, which are characterised by a high degree of endemism since they harbour 16 endemic mammals (Loaiza, 2013) and 39 endemic bird species (Bird Life International, 2019). The Ecuadorian seasonal dry forests contain high levels of floristic diversity, and approximately 80% of their components are regionally endemic as part of the Tumbesian Endemism Centre (Ministerio del Ambiente del Ecuador, 2012). Seasonal dry forest areas also provide local communities with wood and food products, which results in the degradation of the structure, functionality and dynamics of the forest (Ministerio del Ambiente del Ecuador, 2012).

Despite the worrying state of conservation, tropical seasonal dry forests have traditionally been studied to a lesser degree than their neighbours, humid forests, with a ratio of approximately one study in dry forests to six in humid forests (Lessmann et al., 2014). One issue that has not been addressed in any great depth is the fragmentation of the Ecuadorian tropical dry forest in the last few decades, and how this fragmentation has transformed the landscape. According to the framework of the Convention on Biological Diversity (Biodiversity Indicators Partnership, 2011), international organisations worldwide, such as the EBONE "Europe Biodiversity Observation Nature" (Parr et al., 2010) or the BIP "Biodiversity Indicators Partnership", have recommend analysing ecosystems through the use of fragmentation indices. Class indices separately quantify the quantity and distribution of each type of patch in the landscape, and fragmentation indices can, therefore, be considered for each type of patch (McGarigal and Marks, 1995). The objective of this study was consequently to assess the fragmentation of the Ecuadorian seasonal dry forest between 1990 and 2018. The specific objectives of this work were the following: i) to study the deforestation and fragmentation of Ecuadorian seasonal dry forests during five different periods (1990, 2000, 2008, 2014, 2016 and 2018); ii) to describe the spatial patterns of fragmentation during these study periods; iii) to analyse different parameters of fragmentation (e.g. edge density, number of patches, mean patch size) in the three types of dry forests (deciduous, semi-deciduous and transition forest) in the region between the years 1990-2018; iv) to analyse fragmentation in order to find patterns that indicate the most vulnerable areas; and v) to compare the fragmentation index in protected and unprotected areas. The intention of this was to provide useful information on the state of the Ecuadorian dry forest and the areas with the worst conditions and conservation, which would be useful as regards developing effective protection measures according to the present conservation status and future trends.

## 2. Materials and methods

2.1 *Study area*

Our study area included the seasonal dry forest in the coastal region of Ecuador (Fig. 11a), also known as Western Ecuador, located along the Pacific Ocean and the west slope of the Andes mountain range. The coastal region is characterised by three large structural elements that influence the distribution patterns of the biota: The Guayas River, the Esmeraldas River and the Coastal mountain range. This region has a total of 24 ecosystems, 22 of which are divided into two biogeographic regions that are clearly distinguishable as regards their composition and floristic structure, in addition to their bioclimate: the predominantly humid region of Chocó and the region of the Equatorial Pacific, which are mostly dry (Ministerio del Ambiente del Ecuador, 2013). Seasonal dry forests in Ecuador thrive in extreme climatic conditions, with an annual rainfall of 400-600 mm in a period of 3-4 months, generally in February, March and April; the average annual temperature is 24.9 ºC, and the potential evapotranspiration is 1,783 mm / year (Ministerio del Ambiente del Ecuador, 2012). In the present study, we considered the seasonal dry forest of the Ecuadorian Pacific, which is divided into deciduous and semi-deciduous areas. In deciduous forests, 75% of individuals of the arboreal or shrub species lose their leaves during the dry period, which lasts between six and eight months, whereas in the semi-deciduous forest, between 75 and 25% of individuals of the arboreal or shrub species lose their leaves and are located in areas in which the dry periods last between one to six months a year (Prentice, 1990; Ministerio del Ambiente del Ecuador, 2013; Rivas et al., 2020).

2.2 *GIS sources*

In order to limit the potential extent of the seasonal dry forests, the layers of phenology and land use were obtained from the Ecuadorian Ministry of the Environment (available at http://ide.ambiente.gob.ec/mapainteractivo). Land uses have been obtained by the Ecuadorian Ministry of the Environment, using Landsat satellite images and Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER), orthorectification and these have later been certified by experts and by means of fieldwork, with a pixel size of 30 meters (Peralvo and Delgado, 2010; Ministerio del Ambiente, 2012; MAE and MAGAP, 2015; Ministerio del Ambiente, 2017). The Kappa index is approximately 0.7 (Ministerio del Ambiente, 2012). We selected those land-uses classified as native forests in zones with a deciduous and semi-deciduous phenology, since seasonal dry forest predominates in these areas. Flooded areas (mangrove areas) and areas without vegetation cover or without woody vegetation were eliminated. Once the area of seasonal dry forest had been obtained, a transition zone was created between the deciduous and semi-deciduous forest by applying a 10 km buffer in the area that divided both ecosystems. We eventually obtained three analysis zones: deciduous, semi-deciduous and transition forest (Fig. 11b). Land-uses are available for the years 1990, 2000, 2008, 2014, 2016 and 2018, and they were reclassified into two main land-uses: native forest and non-forest zones. Shrub and Herbaceous Vegetation, Agricultural Land, Body of Water, Anthropic Zone, Other Land, No Information and Forest Plantations were classified as non-forest areas, while the 'native forest' land use was classified as forest (Ministerio del Ambiente, 2017).

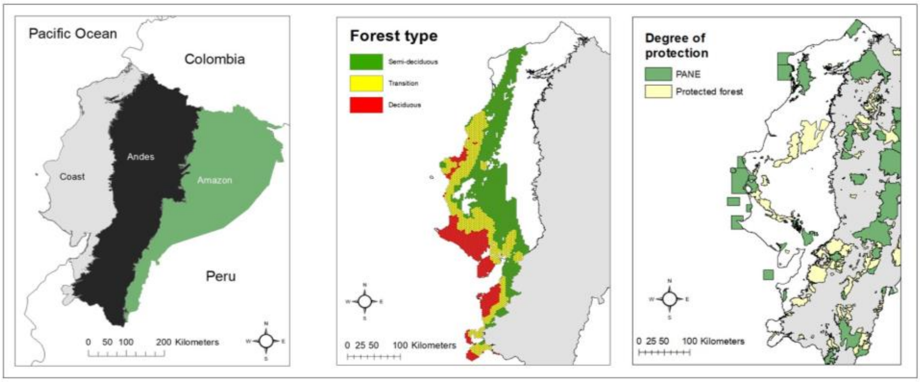


**Figure 11:** a) Map of continental Ecuador showing its three main geographical regions; b) Division of the dry forest by phenologies in hexagons of 10 km$^2$; c) Protected areas.

2.3 *Deforestation and Fragmentation analysis*

We calculated the changes in these two main land-uses throughout the temporal periods. The deforestation rate was calculated by employing the formula proposed by Puyravaud (2003) (Eq. 1).

Deforestation rate (DR) = 1/(t2-t1) * Ln (A2/A1) *100, [1]

where A1 and A2 are the forest cover at times t1 and t2, respectively.

The study area was divided into 10 $km^2$ tiles (Fig. 11b) made of hexagonal polygons, since this is considered the most suitable geometry when studying interaction and connectivity (Birch et al., 2007). The use of polygons improves the ability to assess landscape metrics in a more homogeneous manner. We selected 10 $km^2$ because 99.8% of the world's forest fragments cover less than 10 $km^2$ (FAO and PNUMA, 2020). One of the three types of forests (deciduous, semi-deciduous, and transition) was assigned to each tile on the basis of the predominant type within each segment. The Patch Analysis Tool (Rempel et al., 2012) in ArcGIS was employed in order to calculate different landscape metrics for each tile (Table 5). These were: Number of patches (NumP), average patch size (MPS), median patch size (MedPS), total edge (TE), and edge density (ED) These parameters were then used to calculate the reticular fragmentation index (RFI) on the basis of the percentage without forest (PSB%) and the percentage of edge density (ED%) (Table 5), using the formula proposed by Leautaud Valenzuela (2014). A 1990 forest fragment of 0.2 hectares was used as a reference value in order to determine 100% of the PSB% and ED% metrics. This size was used because smaller sizes distorted the calculation. The RFI was divided into five categories: very high (>80%), high (60%- 80%), medium (40%- 60%), low (40%-20%) and very low (< 20%). An RFI of 100% was attributed to those tiles from which the native forest had disappeared.

**Table 5:** Description of the fragmentation metric parameters analysed according to McGarigal and Marks, 1994; McGarigal and Marks, 1995 and Leautaud Valenzuela, 2014.

| Parameter | Abbreviation | Definition | Unit |
|---|---|---|---|
| Number of Patches | NumP | Total number of patches inside the tiles. The more patches there are, the more fragmented the forest is considered to be | Number |
| Mean Patch Size | MPS | The average patch size of the forest within the tile.<br>A smaller average forest patch size is considered indicative of a more fragmented forest | $kilometers^2$ |
| Median Patch Size | MedPS | The middle patch size, or 50th percentile of the forest patches inside the tile.<br>Median patch size can hide the presence of very large or very small patches. | $kilometers^2$ |
| Total Edge | TE | Perimeter of patches within each tile. The greater the perimeter, the more exposed to disturbances. Greater TE (if the fragmentation is related to an anthropogenic disturbance) | kilometers |
| Edge Density | ED | Amount of edge (Km) relative to the forest area ($km^2$) within the tile<br>ED= TE/Forest area within the tile<br>A high ratio of perimeter to forest patch area may be associated with more fragmented forests (if fragmentation is related to anthropogenic disturbance) | kilometers/ $kilometers^2$ |
| Edge Density Percentage | ED | Edge percentage relative to landscape area.<br>A high ratio of perimeter to forest patch area may be associated with more fragmented forests (if fragmentation is related to anthropogenic disturbance) | Percentage |
| Percentage without forest | PSB | Non-forest area (%) without forest within the tile.<br>Higher percentage of area without forest within the tile would indicate greater fragmentation | Percentage |
| Reticular fragmentat ion index. | RFI | Reticular fragmentation index of each tile.<br>RFI= (PSB%+ED%)/2<br>A higher RFI signifies a greater percentage of fragmentation within the tile. | Percentage |

2.4 *Fragmentation patterns*

The fragmentation patterns were described by employing the Getis-Ord Gi*analysis (Ord and Getis, 1995) for the years 1990 and 2018 and by considering the RFI values. The resulting z-scores and P-values indicate a spatial cluster of high or low RFI values. At 5% significance ($p \leq 0.05$), a z-score greater than 1.96 indicates a hot spot, while a z-score smaller than –1.96 indicates a cold spot and the remaining values are classified as not significant ($-1.96<z<1.96$; $p >0.05$), thus suggesting a random spatial process (Feng et al., 2018). A transition matrix was created using the categories of the Getis-Ord Gi analysis for the years 1990 and 2018 to identify the probability of a hexagon disappearing or of its state changing, based on its initial state (1990) upon its categorisation.

2.5 *Fragmentation in protected and unprotected areas*

In order to test the trend of the RFI in protected and unprotected areas (Fig. 11c), an RFI trend index was calculated as follows (Eq. 2):

$$\text{RFI trend index} = (\text{RFI2018}-\text{RFI1990})/\text{RFI1990} \qquad [2]$$

We then assigned one of the following three protection categories to each tile: unprotected, protected by the Heritage of Natural Areas of the Ecuadorian State (PANE in Spanish), and Protected forests. The RFI trend index of these three categories was then compared (see statistical analysis in section 2.6) in order to verify whether the degree of protection prevents fragmentation more effectively.

2.6 *Statistical analysis*

Wilcoxon paired tests were then used to compare the value of the fragmentation indicators (RFI, NumP, MPS, MedPS, TE, and DE), which were considered as dependent variables, between 1990 and 2018 in each of the three types of forest (deciduous, semi-deciduous and transition). The tiles were employed to pair these tests in order to consider the variations among the same grids between the two periods. A Kruskal-Wallis test was used to compare the RFI values (dependent variable) obtained for the three types of forest in the year 2018. The same type of test was similarly used to compare the RFI trend indices (dependent variable) obtained for the three categories of protection (unprotected, PANE, and Protected forests). In both the Kruskal-Wallis tests, pairwise comparisons (post hoc) were developed in order to verify the differences among the levels of the independent variables. The tiles were the experimental units in all the statistical analyses. InfoStat software was used in all the statistical analyses.

## 3. Results

3.1 *Evolution of loss and fragmentation of forest.*

Since 1990, 2,631.91 km$^2$ has been lost (Table 6), which signifies a loss of 27.04% of the original surface still remaining in 1990, and an annual deforestation rate of 94 km$^2$ (-1.12%).

**Table 6:** Surface of equatorial dry native forest and other land uses in each of the periods included in this study.

| | 1990 | 2000 | 2008 | 2014 | 2016 | 2018 |
|---|---|---|---|---|---|---|
| Native Forest (Km$^2$) | 9730.91 | 8848.42 | 7931.61 | 7497.77 | 7276.01 | 7099.26 |
| Other land uses (Km$^2$) | 17973.63 | 18856.12 | 19772.94 | 20206.78 | 20428.55 | 20605.30 |
| Deforestation rate | - | -0.95 | -1.38 | -0.94 | -1.50 | -1.23 |

This deforestation has changed the degree of fragmentation of the dry forest (Table 7 and Fig. 12), since the mean and median RFI values have increased, particularly from 1990 to 2008 (Table 7). In 1990, 42.28% of the tiles of seasonal dry forest were classified with a low or very low RFI, while this figure dropped to 29.15% in 2018. Moreover, 432 tiles that had some forest patches in 1990 had no patches in 2018 (S1).

**Table 7:** Variation of the mean and median RFI values (%) per tile in each of the 6 periods. SD = standard deviation.

| | | 1990 | 2000 | 2008 | 2014 | 2016 | 2018 |
|---|---|---|---|---|---|---|---|
| | Mean | 40.65 | 46.82 | 51.62 | 50.98 | 51.62 | 52.26 |
| RFI | S.D. | 21.42 | 26.75 | 29.33 | 27.38 | 27.38 | 27.47 |
| | Median | 44.9 | 48.57 | 50.17 | 50.55 | 50.76 | 51.11 |

3.2 *Spatial evolution of fragmentation*

Deforestation has occurred principally in the north of the study area, which formally contained small forest fragments that have disappeared or been considerably reduced since 1990 (Fig. 12 and 13). Other affected areas were located in the Guayas areas and in the central-south, where many of the segments had disappeared, leading to a significant increase in RFI. The edge areas of the large forest fragments have been deforested, as has also occurred with the small fragments, which has resulted in the disappearance of those forests throughout the territory analysed (Fig. 13).

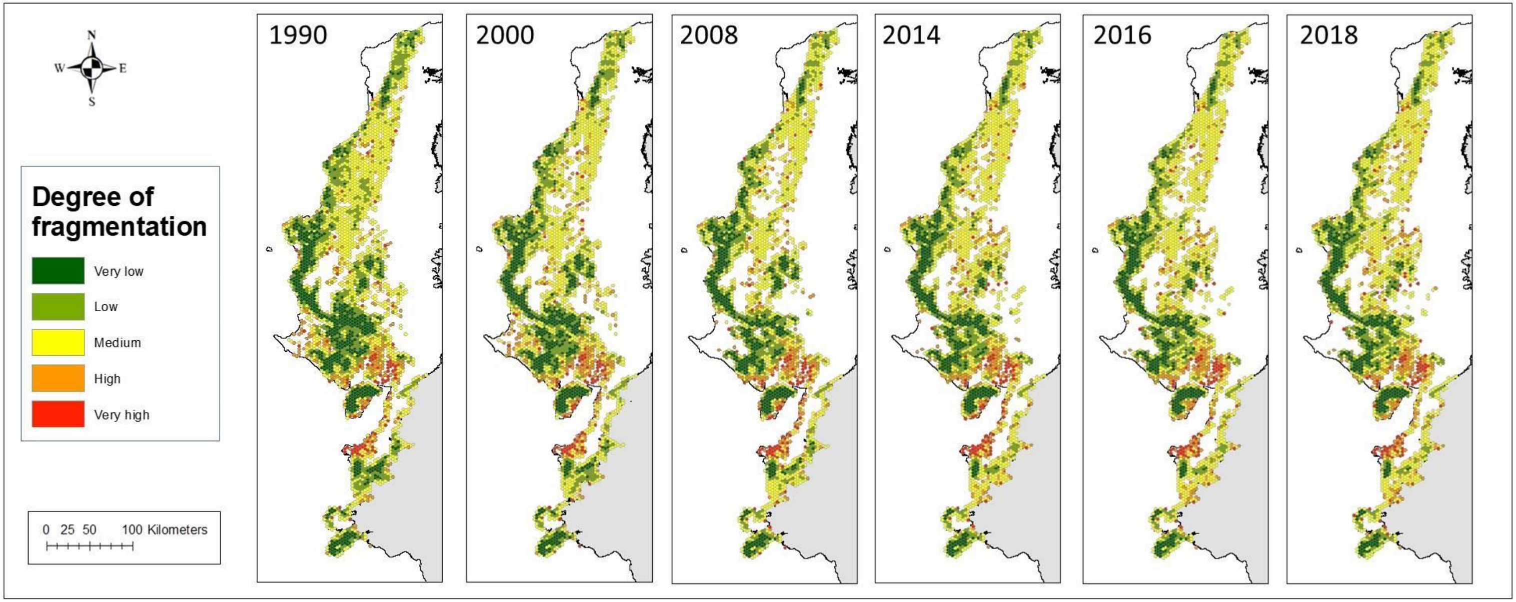


**Figure 12:** Spatial evolution of the RFI of the equatorial dry forest per tile in each of the six time periods


**Figure 13:** Comparison of the central north and central south of the Ecuadorian coast, showing details of forest fragments (green areas) in the years 1990 and 2018.

3.3 *Fragmentation indexes for the three types of forests*

According with the Wilcoxon paired tests, the RFI value was significantly higher in 2018 than in 1990 for the three types of forests (Table 8), and the NumP, MPS, MedPS and TE values were significantly higher in 1990 than in 2018. The ED was significantly higher in 2018 than in 1990 for the semi-deciduous and transition forests, whereas it was lower in 2018 for deciduous forests (Table 8). The semi-deciduous forest attained the highest increase in RFI from 1990 to 2018 (highest mean difference), with the highest levels of fragmentation occurring in 2018 (Table 8). With regard to the number of patches (NumP) within the tile, this has not undergone a great variation as regards either the total or forest types.

**Table 8:** Variation between 1990 and 2018 as regards the values obtained for the different fragmentation indicators by forest type, showing the Z and p-value from the Wilcoxon paired test. Mean-dif = value 2018 - value 1990; SD = standard deviation.

| Forest type | | Deciduous | | Semi-deciduous | | Transition | |
|---|---|---|---|---|---|---|---|
| Year | | 1990 | 2018 | 1990 | 2018 | 1990 | 2018 |
| RFI (%) | Mean | 39.14 | 49.75 | 45.92 | 59.47 | 33.94 | 42.99 |
| | SD | 27.34 | 32.96 | 16.17 | 23.34 | 22.22 | 26.79 |
| | Median | 42.18 | 49.12 | 49.1 | 54.07 | 33.73 | 42.56 |
| | Mean-dif (±SD) | 11.20 (±19.18) | | 16.53 (±24.17) | | 9.58 (±17.45) | |
| | Z | 15.08 | | 28.32 | | 20.34 | |
| | p-value | p<0.0001 | | p<0.0001 | | p<0.0001 | |
| Num P | n | 595 | 511 | 1528 | 1334 | 1089 | 1021 |
| | Mean | 7.48 | 7.39 | 5.72 | 5.14 | 5.58 | 4.73 |
| | SD | 10.79 | 11.62 | 7.15 | 7.25 | 6.93 | 6.51 |
| | Median | 3 | 3 | 4 | 3 | 3 | 3 |
| | Mean-dif (±SD) | -1.11 (±5.26) | | -1.15 (±4.30) | | -1.13 (±4.86) | |
| | Z | -7.02 | | -11.26 | | -8.51 | |
| | p-value | p<0.0001 | | p<0.0001 | | p<0.0001 | |
| MPS ($km^2$) | Mean | 2.34 | 2.35 | 0.92 | 0.58 | 2.51 | 2.15 |
| | SD | 3.55 | 3.40 | 1.90 | 1.46 | 3.36 | 3.06 |
| | Median | 0.35 | 0.47 | 0.17 | 0.12 | 0.72 | 0.64 |
| | Mean-dif (±SD) | -0.31 (±1.88) | | -0.38 (±1.31) | | -0.49 (±2.19) | |
| | Z | -8.03 | | -18.00 | | -9.21 | |
| | p-value | p<0.0001 | | p<0.0001 | | p<0.0001 | |
| MedPS ($km^2$) | Mean | 2.11 | 2.16 | 0.76 | 0.47 | 2.24 | 1.91 |
| | SD | 3.64 | 3.48 | 1.92 | 1.46 | 3.49 | 3.16 |
| | Median | 0.02 | 0.07 | 0.05 | 0.05 | 0.08 | 0.14 |
| | Mean-dif (±SD) | -0.25 (±2.19) | | -0.32 (±1.44) | | -0.43(±2.50) | |
| | Z | -4.59 | | -11.30 | | -5.42 | |
| | p-value | p<0.0001 | | p<0.0001 | | p<0.0001 | |
| TE (km) | Mean | 15.42 | 15.01 | 15.59 | 11.91 | 20.90 | 18.02 |
| | SD | 12.14 | 10.91 | 12.59 | 10.67 | 12.85 | 11.76 |
| | Median | 13.39 | 13.73 | 12.83 | 8.73 | 20.21 | 17.15 |
| | Mean-dif (±SD) | -2.48 (±7.98) | | -4.86 (±10.13) | | -3.94 (±10.89) | |
| | Z | -6.6 | | -18.57 | | -10.22 | |
| | p-value | p<0.0001 | | p<0.0001 | | p<0.0001 | |
| ED ($km/km^2$) | Mean | 42.92 | 35.57 | 20.04 | 29.52 | 23.26 | 31.51 |
| | SD | 259.28 | 106.75 | 45.19 | 74.97 | 97 | 214 |
| | Median | 7.05 | 9.02 | 13.02 | 17.43 | 7.13 | 8.72 |
| | Mean-dif (±SD) | -7.35 (± 269) | | 9.49 (± 71.8) | | 8.26 (± 218) | |
| | Z | 6.04 | | 16.71 | | 10.03 | |
| | p-value | p<0.0001 | | p<0.0001 | | p<0.0001 | |

The Kruskal-Wallis test showed differences among the RFI values obtained for the three types of seasonal dry forests in 2018 ($H=295.65$; $p<0.0001$), with the highest value being attained for semi-deciduous forest and the lowest for the transition forest (Table 9).

**Table 9:** RFI values in 2018 for the three types of forests. N = number of tiles; SE = standard error of means. Lower case letters indicate significant differences according to the post hoc test.

| Kind of forest | N | Mean | S.E. | Median |
|---|---|---|---|---|
| Deciduous | 608 | $49.75^{a}$ | 1.19 | 49.12 |
| Semi-deciduous | 1634 | $59.47^{b}$ | 0.40 | 54.07 |
| Transition | 1106 | $42.99^{c}$ | 0.69 | 42.65 |

3.4 *Fragmentation patterns*

The Getis-Ord Gi * analysis shows the hot and cold fragmentation spots (Fig.14). These results indicate that the hot areas, which had a worse structural connectivity, were more vulnerable to disappearance and had a worse state of conservation. The comparison of 1990 with 2018 highlights this evolution (Fig.14 and Table 10). In 2018, there were 981 (29.70%) tiles catalogued as cold spots, 1044 (31.60%) with no significant differences, and 843 (25.52%) as hotspots; in 1990, meanwhile, there were 1063 (32.18%), 1286 (38.93%) and 863 (26.12%) respectively. The transition matrix (Table 10) shows that of the 863 tiles classified as hotspots in 1990, 213 (24.68%) disappeared during the studied period, and 572 (66.28 %) remained in the hotspot category.

**Table 10:** Transition matrix according to the categorisation provided by the Getis-Ord Gi analysis, showing the number of tiles whose state changed from 1990 (columns) to 2018 (rows). NS = not significant changes.

| | | 2018 | | | | |
|---|---|---|---|---|---|---|
| | Category | Cold | NS | Hot | No forest | Total |
| 1990 | Cold | 823 | 202 | 14 | 24 | 1063 |
| | NS | 154 | 736 | 198 | 198 | 1286 |
| | Hot | 0 | 78 | 572 | 213 | 863 |
| | Forested | 4 | 28 | 59 | 0 | 91 |
| | Total | 981 | 1044 | 843 | 435 | 3303 |

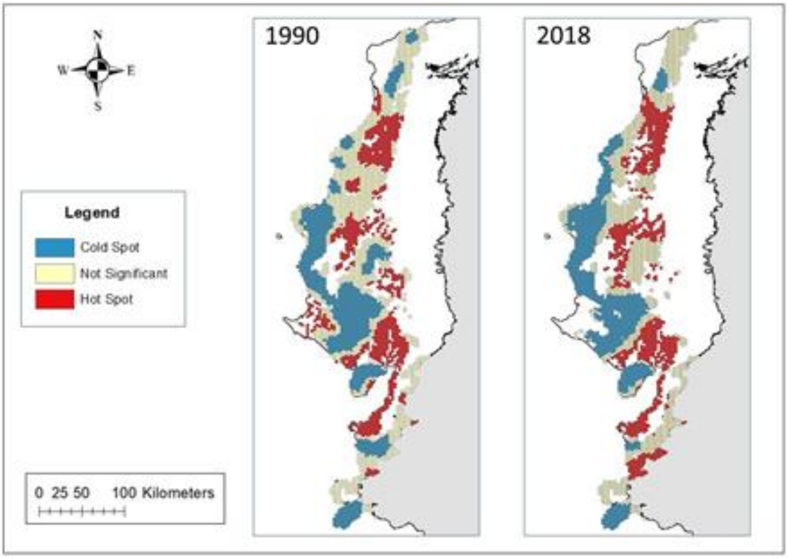


**Figure 14:** Comparison of the hot and cold spots by means of Getis-Ord Gi* analysis in the years 1990 and 2018.

3.5 *Fragmentation in protected and unprotected areas*

Of the 2,707 tiles into which the seasonal dry forest was divided, only 7.24% was covered by PANE and 8.32% by Protected forests, while 84.45% were unprotected. The RFI trend was significantly different for the three protection categories ($H = 19.60$; $P<0.001$), with the lowest value for PANE ($0.3 \pm 0.07$), the highest values for Protected forests ($0.97 \pm 0.17$), and the intermediate values for unprotected areas ($0.42 \pm 0.03$).

## 4. Discussion

Ecuador is undergoing a high rate of deforestation, and the seasonal dry forest is no exception (Sierra, 2013). Our results show that 2,631.91 km$^2$ of seasonal dry forest have been converted to other land uses in the last three decades (87% of the forest that was deforested between 1990 and 2018 had been transformed into agricultural land by 2018, while 7% had been transformed into scrubland), with the extinction of many patches, thus causing a constant increase in fragmentation. This fragmentation has occurred throughout the study area, although we have identified areas with higher fragmentation values and that are spatially aggregated (hotspots), thus suggesting that an important area of the remaining forests runs a high risk of disappearing in the next few years. All these data suggest the urgency of implementing effective conservation measures to preserve the remaining Ecuadorian seasonal dry forest patches and promoting connectivity, with the eventual goal of preventing the disappearance of new areas and ensuring the functional ecology of the remaining forests.

4.1 *Deforestation of Ecuadorian seasonal dry forests*

We observed a dramatic level of deforestation of native forests, and consequently assume that this is a threat to the flora and fauna that inhabit these forests. According to our results, the Ecuadorian seasonal dry forests underwent a net loss of 27% from 1990 to 2018, signifying an annual deforestation rate of -1.12%. This annual deforestation rate was higher than the rates found in other Latin American countries (Brazil -0.56%; Colombia, -0.31%; Peru -0.18%), but lower than that of Paraguay (-1.53%) (FAO, 2020b). When compared to other dry forests in the region, the deforestation rate was in the same range as that of Paraguay and Chile (between -1% and -2%), with lower rates than those found for Argentina and Mexico (>-2%), but greater than those found for Brazil, Costa Rica and Venezuela (< -1%) (Armenteras et al., 2017).

4.2 *Fragmentation*

According to our calculations, all the landscape metrics attained worse fragmentation values in 2018 than in 1990 for all three types of forest. The number of patches decreased and the forests had a smaller mean patch size, which led to an overall increase in the fragmentation index (RFI). But if this information is analysed together with the other fragmentation metrics, it will be noted that this is associated with the disappearance of the smaller patches and the fragmentation of large patches, which has kept the number of patches constant, but has increased the fragmentation. The edge density (ED) in the semi-deciduous and transitional forest has probably increased as a consequence of the forest fragments getting smaller (lower mean MPS) and the increase in the number of small fragments (median MPS). However, the ED in the deciduous forest is probably decreasing because the small fragments are disappearing and the largest ones are becoming smaller (thus keeping the MPS constant), and since smaller fragments had higher edge density values, their disappearance may have led to a decrease in the ED value (Hargis et al., 1998). This process makes this measure less sensitive because, although these small patches disappear, the landscape fragmentation increases (Whelan and Maina, 2005; Tulloch et al., 2016). Small patches have been shown to be fundamental to ecosystems, particularly in those that are highly fragmented, and their disappearance may, therefore, have negative consequences for them (Tulloch et al., 2016).

Of the three types of forest considered, the semi-deciduous forest was the most fragmented. For instance, the average patch size (MPS) of the semi-deciduous forest attained very worrying values (0.58 $km^2$), considering that the tile area is 10 $km^2$. This may be owing to the fact that the more humid forests are more fertile and are, therefore, more prone to the establishment of crops and pastures (Ministerio del Ambiente del Ecuador, 2012; Lessmann, et al., 2014). Moreover, many areas of seasonal dry forest have degenerated into savannah, scrub or grasslands owing to the high pressure of livestock and overgrazing, which could be the cause of the disappearance of the small fragments of deciduous forest, thus limiting forest growth and extension (Trejo and Dirzo, 2000; Sales et al., 2020). Conversely, drier areas are often perceived as areas that are poorer in resources (Siyum, 2020), which could explain the lower conversion of the deciduous forest when compared to the semi-deciduous forest.

4.3 *Connectivity*

Upon comparing the images from 1990 and 2018 (Fig.12, 13 and 14), it will be noted that a quarter of the forest fragments classified as hotspots in 1990 had completely disappeared by 2018, which indicates that these areas are more prone to disappearance. Many factors may lead to differences in deforestation among areas, such as the growth rates of the local human population, the presence of particular hardwood species, the development of specific types of agriculture, the distance to roads and trails, the distance to rivers or the suitability of the land (e.g. soil features or being steep) for agricultural purposes in general (Andam et al., 2008; Barber et al., 2014; van Der Hoek, 2017). Future studies should, therefore, be carried out to evaluate which factors explain a greater or lesser fragmentation of the landscape in order to identify those forests that are still well conserved and run the greatest risk of becoming fragmented.

Deforestation for agricultural and livestock purposes has been identified as one of the main reasons for the loss of seasonal dry forests in Ecuador (Tapia-Armijos et al., 2015; Prieto-Torres et al., 2018), and this also occurs in other countries, such as in the Brazilian Cerrado (Trigueiro et al., 2020) or in the Mexican Yucatan (Smith et al., 2019), and in other dry forests in Latin America (Armenteras et al., 2017). The spatial analysis of fragmentation indicates two large areas of high concentrations of fragmentation (hotspots) in Ecuadorian seasonal dry forests (Fig. 14). The first area is located in the province of Manabí, which is the province with the highest agricultural production, and in which 777,088 hectares correspond to cultivated and natural pastures, contributing more than 20% of the country's agricultural area (Instituto Nacional de Estadística y Censo, 2019a). The second fragmented area corresponds to the urban areas of Guayaquil and Machala, the first and third largest cities as regards human population, respectively (Instituto Nacional de Estadisticas y Censos, 2010). The population of Ecuador has increased dramatically in the last few decades, since it has grown by 452% in 60 years (Villacís Byron, 2012), reaching 17 million inhabitants in 2019 (Instituto Nacional de Estadística y Censo, 2019b). This has led to an increase in demands for food and an increase in the areas devoted to agricultural and livestock production, which are the greatest threats to tropical forests in South America and Africa (Laurance et al., 2014). The dry forests are used by the local population, since they have environmental, social and economic importance for various segments of the rural communities (Briceño et al., 2016). The quality of wood products has historically led to interventions in these forests in order to extract wood and food products as a means of subsistence. The seasonal dry forest provides essential ecosystem goods and services, livelihoods and is vital to the well-being of its residents, since it provides supplies (water, wood, food, biofuels) (Nelson et al., 2020; Siyum, 2020). Population growth consequently also increases the pressure on remnants of forest. In summary, the great importance of agriculture for the economy of the region, together with the growth of the human population, have increased the conversion of natural forests into agricultural land, thus leading to a rise in fragmentation.

The analyses of hotspots showed the areas with a worst conservation status, low connectivity and high fragmentation, and these may be priority areas for forest restoration and an increase in connectivity. Furthermore, in areas identified as cold spots, the actions should be focused principally on preventing deforestation. Fragmentation can have negative consequences for populations of wild species that inhabit the dry forest (Solórzano et al., 2021), since many remaining patches are becoming isolated and exposed to disappearance (Margules and Pressey, 2000). The synergistic effects of fragmentation lead to changes in climate, which can, in turn, change the structure of the vegetation, soil cover and nutrient status, thus affecting the species that inhabit these forest fragments (Margules and Pressey, 2000). Changes take place in these isolated fragments, which can lead to the collapse of populations (Laurance et al., 2012).

### 4.4 *Conservation implications*

Tropical seasonal dry forests are the ecosystems with the least remaining surface in Ecuador (Ferrer-Paris et al., 2018) and are possibly the most threatened tropical ecosystems in the world (Escribano-Avila et al., 2017). They are considered an endangered ecosystem owing to the high degree of endemism and species richness; however, less than 10% of their area is protected (Prieto-Torres et al., 2018). Protected areas are important for conservation (Barber et al., 2014; van Der Hoek, 2017), and should be expanded in the case of the Ecuadorian dry forest (see below), which is less protected than other ecosystems (Rivas et al., 2020).

Previous works have shown that the dry ecosystems in the Coastal Region of Ecuador are underrepresented in the PANE (Sierra el al., 2002; Lessmann et al., 2014; Escribano-Avila et al., 2017), thus suggesting that it is necessary to create new protected areas in order to preserve these ecosystems (Lessmann et al., 2014; Cuesta et al., 2017). As our results show, there has been less fragmentation in the PANE, while it has increased more in unprotected areas and has been particularly dramatic in Protected forests. The protected areas included in the PANE have, therefore, been partly effective as regards preventing deforestation, with a smaller increase in the RFI value than in unprotected areas from 1990 to 2018. These results coincide with those of two previous works, which demonstrated that the deforestation rates were lower inside protected areas, although deforestation still took place in those areas (van Der Hoek, 2017; Ford et al., 2020). Protected forests are not, however, an effective conservation tool for the conservation of seasonal dry forests since, according our results, the RFI increased even more than in unprotected areas. Indeed, more than half of the areas in Protected forests were classified as non-forest land use (Rivas et al., 2020). Although intensive agriculture and deforestation is prohibited in those forests (Sandoval et al., 2017), our results showed that the RFI dramatically increased inside the Protected forests, signifying that the current management system of these forests needs to be reviewed with the aim of ensuring their intended conservation goals. Protection measures should, therefore, be implemented, and they should be established in areas of high priority, which would reduce fragmentation and increase structural connectivity. Several scientists have recently highlighted the importance of small forest patches in fragmented landscapes as regards biodiversity conservation (Tulloch et al., 2016; Fahrig et al., 2019; Volenec and Dobson, 2020), thus suggesting that it is necessary to maximize the total amount of habitat conserved, irrespective of its rate of fragmentation (Fahrig et al., 2019; Ríos et al., 2021). Specific conservation measures, such as the creation of small reserves (including private protected areas: Guerrero-Casado et al., 2021), should, therefore, be implemented in order to protect the few remnants of seasonal dry forest.

## 5. Conclusion

According to our results, the Equatorial seasonal dry forest has undergone a continuous process of deforestation that has led to the loss of more than 2,600 $km^2$ of native dry forest in the last three decades, which is causing an increase in fragmentation, with semi-deciduous forest being the most affected. Fragmentation has increased since 1990, and the number of patches has decreased as a result of the reduction in the forest area, thus increasing the border and patching the forest in isolated fragments, and consequently making it more exposed to anthropic and natural changes. Fragmentation occurs throughout the entire distribution area of seasonal dry forest, which degrades the ecosystem, increases its vulnerability, reduces the area and decreases its connectivity, thus leading to high values of biodiversity loss. Our results show that many areas of seasonal dry forests run a great risk of disappearing if effective protection is not provided or conservation measures are not taken, and it is, therefore, urgent to establish conservation measures that will avoid the continued fragmentation of these forests.

**Acknowledgements.** José Guerrero-Casado is currently supported by the European Regional Development Fund (ERDF) and the Consejería de Transformación Económica, Industria, Conocimiento y Universidades (project reference: 1264483-R). Rafael M Navarro Cerrillo is particularly grateful for the support of the ISOPINE (UCO-1265298) and ESPECTRAMED (CGL2017-86161-R) projects. We acknowledge the institutional support of the University of Cordoba-Campus de Excelencia CEIA3.

# Capítulo 3

## A new index to know the fragmentation status of a patch

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**Abstract**

There are many local fragmentation metrics, but most can be grouped into four types of metrics, and none by itself determines the degree of fragmentation of a patch. The main fragmentation metrics and the most damaging fragmentation effects for biodiversity were grouped together and a new metric/index, the patch fragmentation index (PFI), was proposed to determine the fragmentation at patch scale. Subsequently, the Ecuadorian coastal dry forest was used for its verification. Geographic information layers were used, temporal changed was measured in 1990, 2000, 2008 and 2018. Fragmentation per patch was calculated, zonal fragmentation (with the area of influence and hexagons of 10 km$^2$) and the spatial patterns using Getis-Ord Gi *. Ecuadorian dry forest presented 0.88 and 0.99 of mean and median respectively of PFI in 2018. The mediam has increased by 8.6% since 1990. 3,451 patches of forest have disappeared between 1990 and 2018. The tessellation in 10 km$^2$ present mean 0.66 in 2018, being lower than the medium per patch. The Getis-Ord Gi * analysis is shown to be effective, as 62% of the patches classified as hot patches in 1990 disappeared in 2018. The PFI has showed to be a useful tool to describe fragmentation at patch scale and can be extrapolated to landscape. PFI will provide a new vision and can help in decision-making in conservation and management of threatened ecosystems.

## 1. Introduction

International organizations like the Convention on Biological Diversity (Biodiversity Indicators Partnership, 2011), the "Europe Biodiversity Observation Nature" (Parr et al., 2010) and the "Biodiversity Indicators Partnership" have recommend to analyse the conservation status of the ecosystems through the use of fragmentation indices. Forest fragmentation refers to the amount and spatial configuration of wooded vegetation (Gustafson, 1998; Riitters et al., 2002; Hermosilla et al., 2018). Increased forest fragmentation results from the dissection of contiguous wooded areas into smaller, isolated patches of various sizes and shapes and with a larger edge surface exposed to the matrix (Arasa-Gisbert et al., 2021). Forest fragmentation is a continuous and progressive process that reduces the areas of intact forest cover, increases the edges of the forest and isolates the remaining patches in a forested landscape (Carranza et al., 2015). It has also been defined as process that must be separated from the loss of habitats, although they are closely linked, fragmentation is pattern and nor a process (Fahrig, 2019).

Forest loss and the subsequent fragmentation results in a variety of ecological, environmental, social and economic impacts (Kupfer, 2006) being one of the major components in global change (Rogan and Lacher, 2018). Fragmentation has been classified as one of the main threats to tropical trees (Whitmore and Sayer, 1992; Heywood et al., 1994; Trejo and Dirzo, 2000) and as one of the main threats to the loss of biodiversity (Arroyo-Rodríguez et al., 2020) with the ability to damage the quality of habitat for more than 80% of all species of mammals, reptiles, birds and amphibians (Soverel et al., 2010). Small patches tend to be considered the most affected by fragmentation due to the edge effect (Arasa-Gisbert et al., 2021), affecting biodiversity and other ecological processes (Fahrig, 2003a; Phillips et al., 2018).

For these reasons, estimate habitat fragmentation is a relevant issue in nowadays ecology (Teixido et al., 2020). There are many metrics to calculate fragmentation, being area metrics and isolation/proximity metrics those most commonly used in the fragmentation literature accounting for 48% and 42.4%, respectively (Fardila et al., 2017), but other parameters are also well studied such as: edge, shape, or patch density, fragmentation studies are mostly targeted to the distribution, diversity and dynamics of species populations (Fardila et al., 2017). In this regard, various authors have tried to identify parameters of fragmentation assessment at forest patches and landscape scales (Hermosilla et al., 2018; Dener et al., 2021), others even define fragmentation as the formation of new edges (Asbjornsen et al., 2004; Tapia-Armijos et al., 2015), smaller patches (Ries et al., 2004; Chakraborty et al., 2017) or more isolated (Fahrig, 2003; Kupfer, 2006; Tapia-Armijos et al., 2015) showing the complexity of the fragmentation assessment.

Fragmentation studies are shifting from patch scale to landscape scale (Fardila et al., 2017), which is related to many fragmentation effects do not depend on just one fragmentation parameter at patch level, but on several at the landscape level. In fact, Fahrig (2019) suggested that patch metrics cannot be extrapolated at the landscape-level and define fragmentation as a landscape-level process that must be studied through spatial patterns. Others authors have suggested that the loss of habitat and fragmentation (reduction in patch size, isolation and edge effect) occur at the same time, and therefore, their independent effects cannot be demonstrated (see for example, Didham et al., 2012). However, Haddad et al. (2015) correlated the reduction of habitat, increase of isolation and border with processes of residency, abundance, persistence and species richness (plants and animals), nutrient retention, succession rate, community composition, microclimate, trophic dynamics or other effects. Those, these authors considered that the effects of fragmentation continue to appear until decades later, probably leading to the extinction of species (perhaps being able to be colonized again by migrations), having strong and consistent effects on a wide range of ecosystems on the five continents. Other authors say that the effects of fragmentation are not always negative (Fahrig, 2017; Fahrig et al., 2019) highlighting the importance of small patches to maintain biodiversity and the connectivity of ecosystems (Herrera et al., 2017; Wintle et al., 2019), and that many smaller forest patches can be richer in species than the large ones (Tulloch et al., 2016; Fahrig, 2020). Indeed, the "habitat amount hypothesis" suggests that the habitat amount in the landscape can be more important than patch size and isolation (Fahrig, 2013; Rios et al., 2021). For all these reasons, calculation and interpretation of fragmentation are subjects of debate (Fahrig, 2017; Fletcher et al., 2018; Fahrig et al., 2019) opening a range of possibilities to researchers. There are many patch fragmentation metrics, but none of them alone explains or identifies the fragmentation status of a patch. The most common metrics identified and calculated in patch fragmentation are area (patch size), edge (perimeter of the patch), shape (mean patch fractal dimension (MPFD), shape index or area / perimeter), proximity to other patches (distance to the closest patch, distance to the furthest, or average distance to other patches) or core areas (McGarigal, 1994; Bestion et al., 2019). However, when these patch metrics are extrapolated to the landscape can be misled. This problem has already been addressed (Hargis et al., 1998; Jackson and Fahrig, 2013; Fahrig, 2019), describing fragmentation as a complex process where an interpretation of the results is required. Therefore, integrating the main fragmentation parameters (size, isolation, edge and shape) into a single formula would improve the assessment of patch fragmentation, being a suitable tool in decision-making. Once those patches have been identified, its location and its fragmentation state, their temporal and spatial evolution (e.g., increase, decrease or stationary state) could be monitored and analyse at landscape scale (Hermosilla et al., 2018). Therefore, our general objective was to propose and analyse a new fragmentation metric at patch scale, named as patch fragmentation index (PFI) and its validation on one highly fragmented ecosystem, the seasonal tropical dry forest in Ecuador. The specific objectives were: i) to analyse the efficiency of the PFI at patch scale as an indicator of habitat fragmentation; ii) to analyse if the PFI is useful to monitor the evolution of fragmentation at different scales (from patch to landscape); and iii) analyse the usefulness of the PFI as an indicator zonal fragmentation. The PFE can provide a new research tool for decision making in the field of forest fragmentation.

## 2. Methodology

### *2.1 Study area*

The selected study area was the seasonal dry forest in the coastal region of Ecuador (Fig. 15), which is part of the Chocó-Darien-Western Ecuador, one of the world biodiversity hotspot (Myers et al., 2000) and one of the area's most likely to find new species (Moura and Jetz, 2021). It is a highly fragmented ecosystem, with an important area classified as very high fragmented, very low connected and highly threatened (Rivas et al., 2020). Despite having these worrisome values, dry forest of Ecuador present a lower degree of protection than humid forests (Rivas et al., 2020). In order to limit the potential extent of the dry forests, the layers of phenology and land use were obtained from the Ecuadorian Ministry of the Environment (available at http://ide.ambiente.gob.ec/mapainteractivo), and the interception between the land use native forest and the deciduous and semi-deciduous phenology was used to delimit the extent of the tropical dry forests.

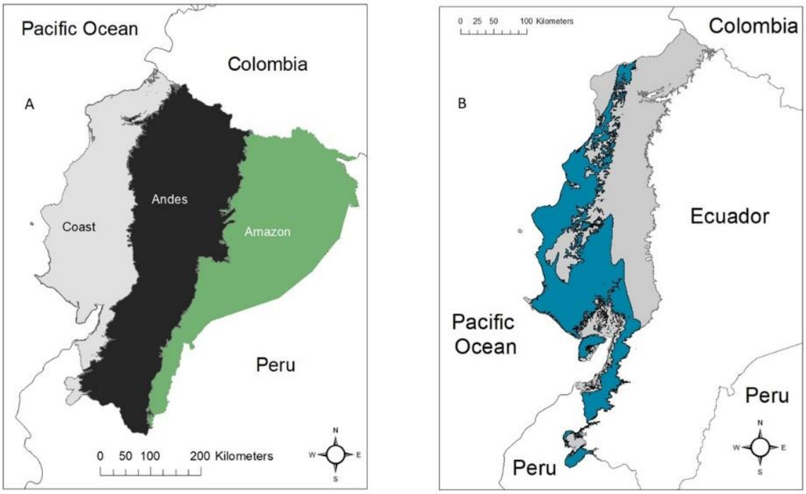



**Figure 15:** A) Map of mainland Ecuador and its three geographical regions. B) Map showing the seosonal dry forest (blue) on the equatorial coast (grey).

*2.2 Fragmentation metrics*

We identify the most common metrics used to calculate patch fragmentation (McGarigal, 1994; Bestion et al., 2019), which were those related to area (patch size), edge (perimeter of the patch), shape (MPFD, shape index or area / perimeter), proximity to other patches (distance to the closest patch, distance to the furthest, or average distance to other patches) or core areas. Here, we propose a new fragmentation metric, including some of the most important indexes (those that have the greatest consequences in fragmentation), thus eliminating a large part of the errors that were made when studying a single patch parameter.

The Patch Fragmentation Index (PFI) is based on contemplating the main effects of fragmentation, they are the loss of habitat (patch size), shape complexity and isolation of the patches (Arasa-Gisbert et al., 2021). The PFI formula includes these fundamental fragmentation parameters, such as the formation of new edges (the MFPD is a shape parameter), smaller patches (area of the patch) or more isolated patches (area of influence), making a parameter that encompasses several aspects making it complete and indicative of the state of the patch.

For the fragmentation calculation the following formula was used:

$$PFI = \frac{4}{5}\left(1 - \frac{Ap}{Ai}\right) + \frac{1}{5}\left(\frac{MFPD}{2}\right)$$

Where Ap= Patch area, Ai= Area of influence, MPFD= Complexity of shape. PFI= patch parameter between 0 and 1. The closer to zero indicates less fragmentations. The area of the patch is one of the most used metrics in fragmentation, since it indicates a small amount of habitat and greater proximity to the edge. The area of influence (Ai) is based on the maximum area that the patch could occupy if it had not suffered deforestation processes. The mean fractal patch dimension (MPFD) is a measure of the complexity of the shape. The mean fractal dimension approaches one for shapes with simple perimeters (such as circles or squares) and approaches two when the shapes are more complex (McGarigal and Marks, 1995). The complexity of the shape is related to fragmentation because in more complex shapes there is more edge in relation to the area of the patch, this increases the edge effect, which has been shown to have detrimental effects on fauna and flora. Patches with more complex shapes can also be divided more easily. The section (Ap/Ai) was weighted with more weight because this section accumulates the worst effects for biodiversity and encompasses more metrics than the section (MFPD), MFPD is divided by 2 because it is a value between 1 and 2 and PFI is values between 0 and 1. The maximum value of 1 is never reached as it signifies the disappearance of the patch and therefore it has neither area nor shape.

The PFI was used to calculate the state of the Ecuadorian dry forest in 1990. For a better representation, the forest patches were classified based on the PFI into five categories for the value of fragmentation: very high (>80%), high (60%- 80%), medium (40%- 60%), low (40%-20%) and very low (< 20%). For the calculation of areas of influence (Ai), Voronoi areas were created with the forest patches in 1990, which were intercepted with areas of dry forest habitat to delimit the zone of possible growth of the dry forest.

*2.3 Temporal change of fragmentation based on PFI.*

The areas of influence (Ai) created for the seasonal dry forest in 1990 were then used to calculate the PFI for the years 1990, 2000, 2008 and 2018, converting the seasonal dry forest in a raster of 100x100 meters to delimit native forest patches. In the areas of influences where disappeared the patches in the years 2000, 2008 and 2018, a value of PFI=1 was given (maximum fragmentation rate and patch is considered that disappears). In the areas of influence where increase the number of patches (in the years 2000, 2008 and 2018) the new patches were considered as part of the original in 1990, the increase in the number of patches can be produced by reforestation or because large patches are divided.

*2.4 Zoning of fragmentation*

Two zonal fragmentations were calculated: heterogeneous zones using the area of influence of the patches, and a homogeneous zoning using hexagons of 10 $km^2$. The area of influence was used to divide the study area into smaller areas; for this purpose, the PFI value of the patch was given to the area of influence and to calculate patch fragmentation in areas of homogeneous size the terrain was tested in hexagons of 10 $km^2$, in which each tile was given the average PFI of the patches that were inside it for the years 1990, 2000, 2008 and 2018. Only the hexagons with at least one forest patch inside were considered.

*2.5 Fragmentation patterns.*

An analysis of Gi* de Getis-Ord (Ord and Getis, 1995; Feng et al., 2018) or hot spots was used to analyse fragmentation spatial patterns of the PFI at patch level (similarly to Rivas et al., 2021), in which heterogeneous zoning was used (see previous section) and a transition matrix was created, showing the status of the patches in 1990 and 2018, and their status change. The resulting z-scores and P-values indicate where features with high or low values cluster spatially. This tool works by searching for each entity within the context of neighbouring entities.

## 3. Results

*3.1 Fragmentation of dry forest in Ecuador based on PFI.*

Figure 16 shows the patches of dry forest in the coastal region of Ecuador qualified according to their PFI value. Highly fragmented areas spatially coexisted with less fragmented areas, demonstrating how the PFI catalogues the patch based on its state of fragmentation.

Fragmentation per patch increased during the study period, showing a greater number of patches with an index close to 1 (the highest possible) in 2018, or many patches that have disappeared since 1990 (3451), with 6,908 patches in 1990, remaining only 3,457 patches in 2018, with nearly half of the patches deforested since 1990 (Fig. 17, Table 11). Change of dry forest showed a highly fragmented state with a greater mean value of PFI in 2018 (0.88) than in 1990 (0.81) (Table 11), increasing 8.6% in this period.

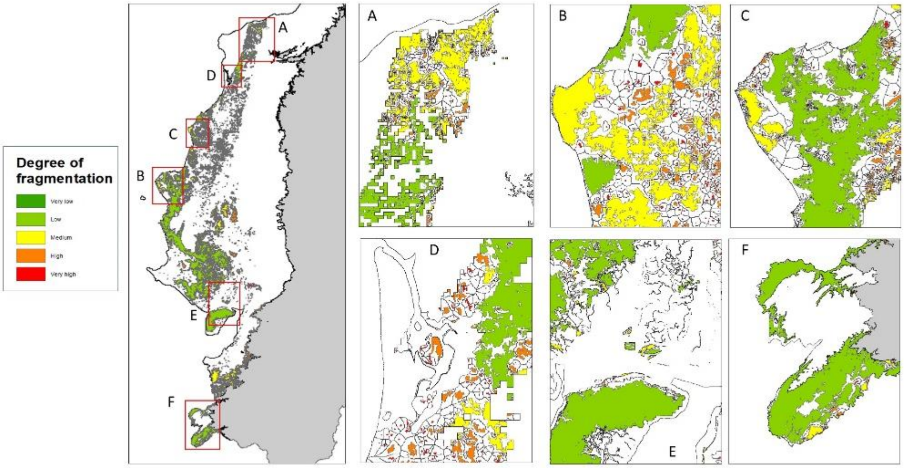



**Figure 16:** Fragmentation status of the seasonal dry forest in Ecuador in 1990 using the FPI. Figures in details A, B, C, D, E, F: State of fragmentation of the patches and their area of influence.

**Tabla 11:** Evolution of the mean and median value of PFI measured at the patch level. The number of patches deforested indicates deforested patches between 1990 and the indicated year.

| Year | 1990 | 2000 | 2008 | 2018 |
|---|---|---|---|---|
| Nº. patches deforested | 0 | 2481 | 3348 | 3451 |
| Mean | 0.81 | 0.85 | 0.87 | 0.88 |
| Median | 0.85 | 0.90 | 0.92 | 0.99 |
| E.D | 0.09 | 0.13 | 0.13 | 0.12 |

Through the PFI, temporal changes of some patches with different features can be monitored. For example, the evolution of the central patch, the largest of the landscape, showed higher fragmentation despite its large area in comparison with other surrounding patches (Fig. 17a). On the other hand, figure 17b showed an area where many patches disappear, and the main patch has worsened its status.

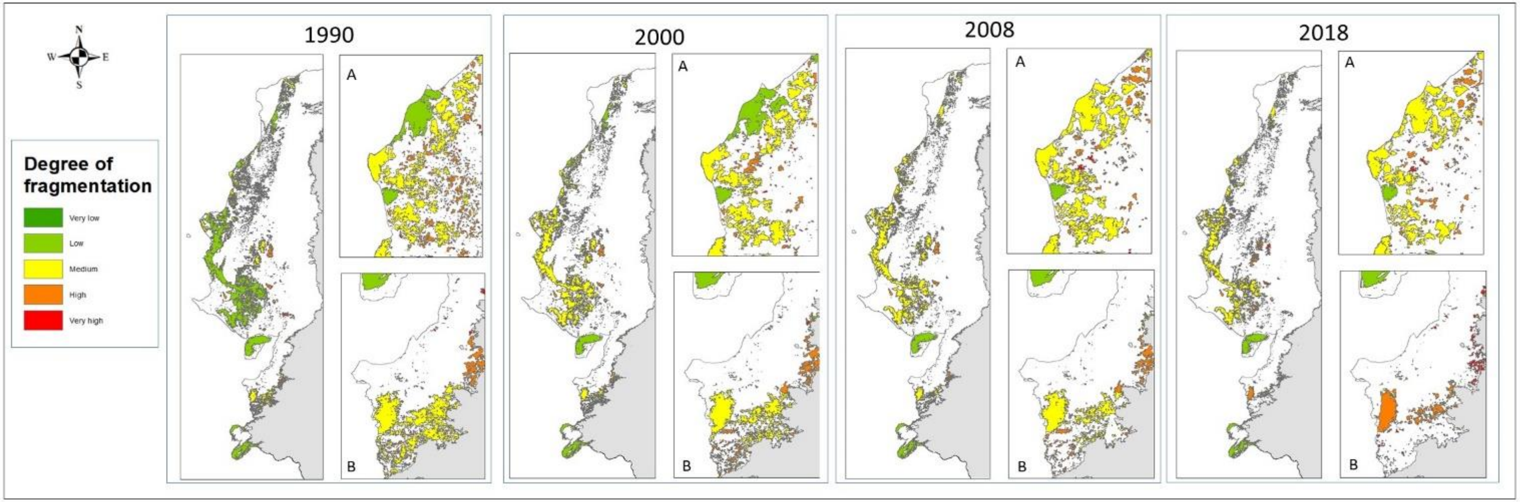



**Figure 17**: Evolution of fragmentation status of the seasonal dry forest in Ecuador using FPI. A and B figures in details

*3.2 Zoning of fragmentation of dry forest in Ecuador*

In the study area, with high deforestation and fragmentation rates, 3,457 area of influence have lost their patches of forest. Those were located mainly in the north-central and the eastern and western zones (Fig. 18). The homogeneous zoning remained similar over time, having a mean value lower than the zonal representation per patch (Fig. 19, Table 11, Table 12), In 2018, the homogeneous zoning was 25% lower than the zoning by the area of influence and between 1990 and 2018, and 350 hexagons have been lost.

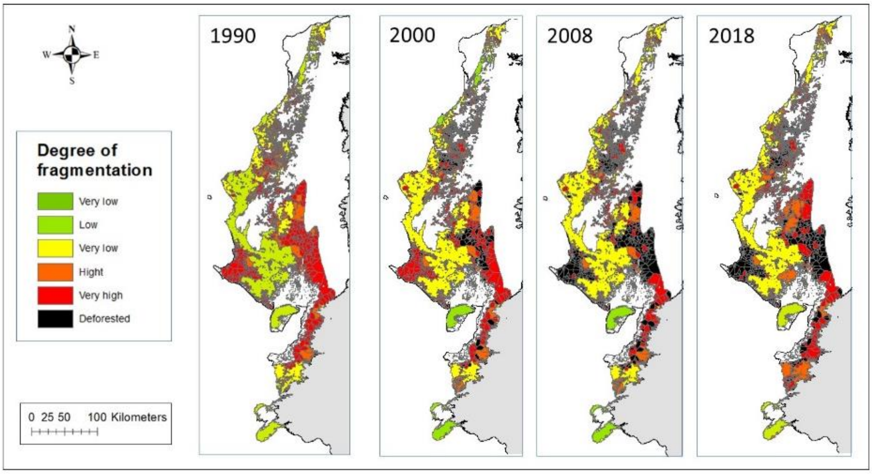



**Figure 18**: Evolution of the state of fragmentation using the area of influence as delimitation.

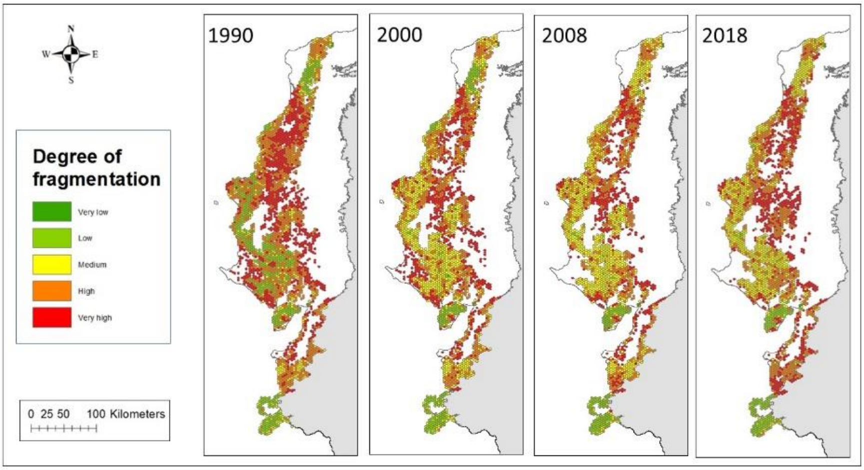



**Figure 19:** Evolution of Fragmentation Status using FPI and a homogeneous tessellation by hexagons of $10km^2$.

**Table 12:** Evolution of mean and median value of PFI in a homogeneous zoning

| Year | 1990 | 2000 | 2008 | 2018 |
|---|---|---|---|---|
| Nº. of hexagons | 3091 | 2919 | 2681 | 2741 |
| Mean | 0.67 | 0.63 | 0.62 | 0.66 |
| Median | 0.73 | 0.65 | 0.63 | 0.69 |
| S.D | 0.16 | 0.17 | 0.15 | 0.15 |

*3.4 Fragmentation patterns of dry forest of Ecuador*

Regarding the analysis of hot spots, the PFI was very efficient in identifying patches and areas at risk of becoming more fragmented or disappearing (Fig. 20), where 1808 (62.9 %) patches defined as hot spots of fragmentation in 1990 disappeared in in 2018 (Table 13).

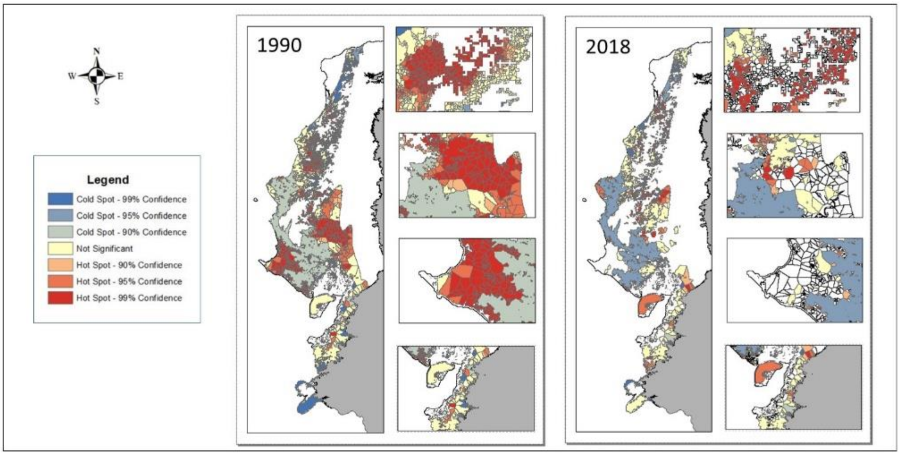



**Figure 20:** Hot spot analysis of the seasonal dry forest in 1990 and 2018, in which an important number of hotspots identified in 1990 have disappeared by 2018.

**Table 13:** Hot spot change matrix showing the number of patches that change from one to other status from 1990 to 2018.

| | | 2018 | | | | |
|---|---|---|---|---|---|---|
| | | Cold | NS | Hot | Deforested | Total |
| 1990 | Cold | 879 | 314 | 22 | 505 | 1,720 |
| | NS | 139 | 634 | 403 | 1,138 | 2,314 |
| | Hot | 19 | 228 | 819 | 1,808 | 2,874 |
| | Total | 1,037 | 1,176 | 1,244 | 3,451 | 6,908 |

## 4. Discussion

Fragmentation parameters have been studied for more than 20 years and although there are multiple parameters, these can be grouped into few groups (Riitters et al., 1995; Cushman et al., 2008; Chen et al., 2014) which reduce redundancies in the calculation and distortion of fragmentation (Cushman et al., 2008). Chen et al. (2014) showed that only 4 types of metrics explain 89% of the variation of fragmentation metrics. These four metrics, in order of importance, were: composition/area and contagion, edge (e.g., the third would be a combination of the first and second), and shape. Rogan and Lacher (2018) considered four similar parameters reflex the main habitat alterations (we exclude the parameters habitat matrix because refers to human-modified land that surrounds or intersperses throughout remnant native habitat patches in fragmented landscapes) and have shown to describe detrimental effects on species. Patch Fragmentation Index (PFI) proposed on this work is a useful and accurate tool to measure the patch fragmentation status because include the four key parameters.

Area effects is related to the size of the patch, which when decreasing, resources become limited, limiting the size of the population, and reduce reproductive success and colonization rates (Hanski and Gaggiotii, 2004). If these mechanisms work together, vortices can be produced, putting populations at greater risk of extinction (Rogan and Lacher, 2018). When considering the shape of a habitat patch, there is a higher level of edge in relation to the area (Rogan and Lacher, 2018). Complex shaped patches can be more easily divided into smaller patches, exposing central habitat and increasing fragmentation (Ewers et al., 2007). Also more complex shapes can increase the degree to which the edge of the patch infringes on the central habitat of the patch; thus reducing the amount of central habitat available to occupy species specialized in habitats (Collinge, 1996; Rogan and Lacher, 2018).

Edge effects can increase biodiversity at the edges of the patch, this occurs because species that live in the matrix or in surrounding areas are introduced into the patch, but nevertheless specialist species are affected (Rogan and Lacher, 2018). When measuring the richness of the species is measured within the patch, and an increase in biodiversity is considered, but the arrival of these species could be classified as "invasive species" that compete with the specialists and their populations may decline. Increased edge has also been correlated with higher seed and herbivore predation (Haddad et al., 2015; Rogan and Lacher, 2018). Isolation is the antithesis of connectivity, lack of connectivity has been linked to reduced movement among fragments, thus reducing fragment recolonization after local extinction (Haddad et al., 2015).

The loss of connectivity is considered one of the greatest threats to biodiversity (Pascual-Hortal and Saura, 2006). Connectivity tends to decrease or even be completely lost due to fragmentation and land use change produced mainly by anthropogenic activities (Hilty et al., 2020). Connectivity links a wide variety of ecological issues such as: evolution, dispersal, migrations, development of genetic structures, source-sink dynamics or adaptations to climate change (Kool et al., 2013).

*4.1 Fragmentation of dry forest in Ecuador.*

Fragmentation is a process that requires monitoring for effective forest management. Monitoring helps highlight areas of rapid change that need the attention of conservation professionals and forest managers on the ground and helps understand direct and indirect socio-economic drivers of loss (Smith et al., 2019). Changes in fragmentation at patch scale (increasing, declining or maintaining its state) cannot be assessed if measure at different times is not able. That is why temporal evolution of fragmentation is very useful in making decisions and identifying the fragmentation causes.

Our results showed a high fragmentation levels of the Ecuadorian dry forest, as has already been demonstrated previously using different methodologies (Ministerio del Ambiente de Ecuador, 2015; Rivas et al., 2021). PFI median value was close to 1 in 2018, increasing by 16% since 1990, indicating that the majority of patches presented a high state of fragmentation or have already disappeared on this period. Between 1990 and 2018, 3,457 patches were lost increasing overall fragmentation.

The PFI metric has also shown to be efficient when establishing fragmentation zones using the area of influence of the patches. Some authors zone the landscape using geometric figures such as squares (raster) or hexagons (Hermosilla et al., 2018; Wintle et al., 2019), but hexagon shape has shown to be more efficient (Birch et al., 2007), although even so they have drawbacks as shown in the figure 21.A (dry forest area within the study area). Different inconveniences can be presented such as islands smaller than the tessellation, mix with other ecosystems, other ecosystems within our study area, littoral zone marked by the coast and border area with another ecosystem or outside the study area such as another country (Fig. 21.A). The area of influence eliminates these errors because in its calculation only the study area is used or where the patch can inhabit. Furthermore, when the area is tessellated for study, the problems cited in table 14 plus the tessellation problems are presented (Fig. 21.A). These problems are solved used the area of influence, the problem of used the area of influence is that the zones do not have all the same dimensions or shapes (Fig 21.B), to solve this problem, a combination of tiles and PFI value can be used (Fig 21.C).

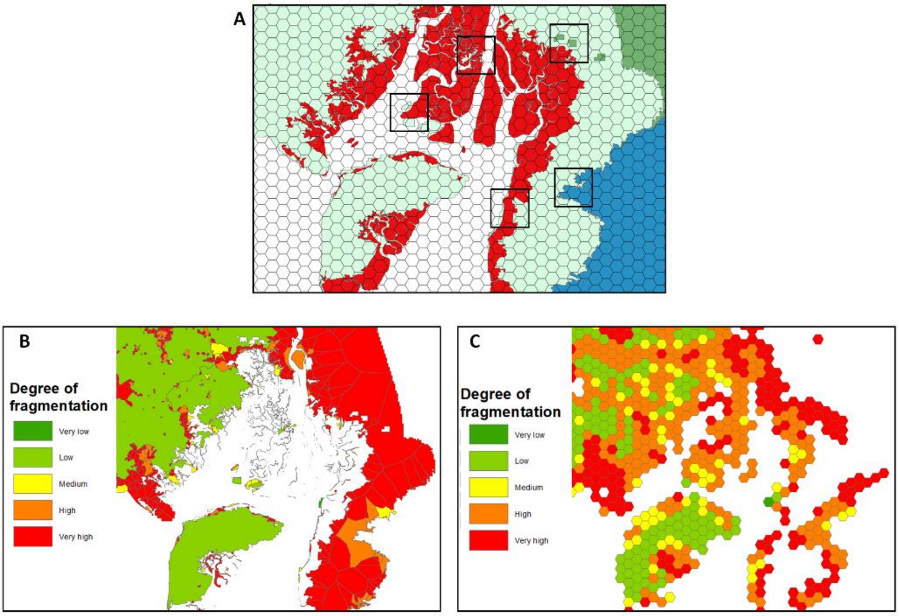
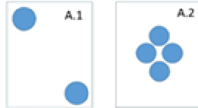
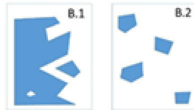
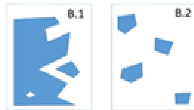
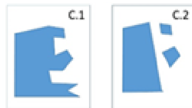
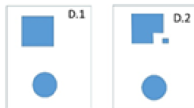
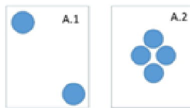



**Figure 21:** (A) Image of the Ecuadorian coast, in red floodplain areas, in light green areas of dry forest, in dark green areas of humid forest and in blue areas of mountains. (B) zones of the Ecuadorian coast classifying the patches by IFP (C) Zone of the Ecuadorian coast using IFP and homogeneous tessellation.

The PFI calculation based on tiles or hexagon tessellation allows a better assessment of fragmentation at the landscape level, in particular where there are a high number of small patches. For example, in 1990, only 62 patches had more than 10 $km^2$, out of a total of 6908, but these patches occupied 85% of the area (and the historical trend was similar), which can lead to a distortion if we extrapolate the patch data at the landscape level (Fahrig et al., 2019). The tessellation therefore helped to assess fragmentation at the landscape. Another advantage of the tessellation is that when using the area of influence in the PFI formula, the area of influence only measures the area where our ecosystems surface can increase, eliminating problems of mixing with other ecosystems or other land uses (e.g., coastal zones or borders of countries).

**Table 14**: Problems when measuring fragmentation with standard parameters.

| Metrics | Conclusion | | Description | Representation |
|---|---|---|---|---|
| Area | The larger the average area, the less fragmentation. | A | In figure A.1 there is a greater mean area, however figure A.2 shows less fragmentation. |  |
| Perimeter | The greater the perimeter area, the greater the fragmentation. | B | In figure B.1 there is a greater perimeter, however figure B.2 shows greater fragmentation. |  |
| Area/Perimeter | The smaller the area / perimeter, the greater the fragmentation. | B | In figure B.1 there is higher area / perimeter, however figure B.2 shows higher fragmentation. |  |
| MPFD and other shape parameters. | In more complex forms, greater fragmentation. | C | In figure C.1 the patches have more complex shapes, however figure C.2 shows greater fragmentation. |  |
| Distance | The shorter the mean distance, the less fragmentation. | D | In figure D.1 the patches have a greater mean distance, however figure D.2 shows greater fragmentation. |  |
| Number of patches | The lower the number of patches, the less fragmentation. | A | In figure A.1 there are fewer patches, however figure A.2 shows less fragmentation. |  |
| Core Area | Undisturbed areas within the patch, the greater the core area, the less fragmentation. | | The core area is based on the species, on the area that the species needs eliminating the edge effect, but it starts from the premise that the edges are detrimental to the species, but this is not always the case. | |

*4.2 Fragmentation patterns.*

From 1990 to 2018, 3,451 patches and 350 hexagons have been lost, which indicates that these areas may be more prone to be deforested. The analysis of hot spots based on the areas and not just on a patch is also extremely efficient in identifying areas are prone to be deforested, the, which would indicate that these areas are the most threatened. The study of spatial patterns can help in decision-making, indicating those areas with concentration of patches with high or low fragmentation (cold spots). It could also allow studying the structural connectivity of the landscape. In our case we observed how the most affected area was the central dry forest distribution which corresponds to the area of Manabí, the largest province in the country. This area has suffered a strong population increase (Instituto Nacional de Estadística y Censo, 2021) and is the main agricultural region of the country, contributing more than 20% of the country's agricultural area (Instituto Nacional de Estadística y Censo, 2019a).

In summary, PFI was able to improve the description of fragmentation status at patch scale, pointing out very large patches catalogued as highly fragmented or small patches classified as low fragmented. Also, PFI was effective in cataloguing the patch fragmentation and its area of influence, as well as showing its effectiveness in identifying patterns such as disappearance or conservation of patches. The PFI has been applied in a highly fragmented forest with complex patterns of deforestation and fragmentation. It would be interesting to apply this PFI to other ecosystems with simpler patterns.

## 5. Conclusion

Fragmentation is one of the main components of the extinction of species and there are multiple ways to measure it (S2), in this regard PFI provides a new tool to describe and interpreting fragmentation given patch of habitat. The majority of fragmentation metrics are designed for ecosystems with little complex fragmentation patterns, such as the Amazon rainforest or the taiga, but not all ecosystems have this structure, and therefore the traditional parameters are not efficient. In this regard, adding the area of influence in the PFI provides useful information about the status of a patch. The area of influence is essential to know the anthropic fragmentation of a patch, since we can find patches that due to the dynamics or situation of the ecosystems have a small size or a distribution that is distant from other patches, perhaps presenting a small patch fragmentation. Although the area of influence is delimited through the potential distribution of the patch with Voronoi triangles, this can be even more precise if we introduce site variables such as very steep slopes or rivers. PFI groups the main fragmentation parameters into one improving patch fragmentation analysis and extrapolating at the landscape level for a more complete view. It has also been very efficient in identifying landscape dynamics with the analysis of hot spots. This would allow taking conservation or reforestation analysis measures and having a realistic vision of the state and evolution of the patches and the landscape, also allowing the study of landscape dynamics.

# Capítulo 4

## Assessment of habitat connectivity in Equatorial tropical dry forests using Graphab resistance surfaces

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**Abstract**

The theory of graphs that has been developed is of the utmost importance for the calculation of connectivity, a parameter that plays a fundamental role in highly fragmented ecosystems, such as the seasonal tropical dry forest, one of the most threatened ecosystems worldwide. This research aims to discover the functional connectivity of the Ecuadorian seasonal dry forest between 1990-2018. The land-use changes, fragmentation and functional connectivity that occurred in Ecuadorian seasonal dry forests between 1990 and 2018 were verified using GIS, land cover shape and Graphab sofware. A reforestation plan was also developed using various connectivity metrics and was compared with that proposed by the Ecuadorian Ministry of the Environment. A surface of 9.5% of dry forest was lost between 1990 and 2018. Former forest areas were put mainly to agricultural uses, which increased by 12.96%. The number of total patches decreased from 6,908 to 5,357, signifying a loss of areas of 30% and leading to losses of up to 75% of connectivity. The distances between the links became shorter. Areas with low connectivity and a risk of disappearance were identified, and it was shown that the reforestation of small areas can increase functional connectivity. One of the parameters that should be taken into account for the conservation and reforestation of habitats is functional connectivity, since the change in the matrix affects connectivity, thus demonstrating the importance of small patches and other natural corridors as regards measuring landscape connectivity. Graphad software can be a very helpful tool when calculating connectivity and creating a reforestation plan.

**Keywords**: functional connectivity, land-use change, graph theory, fragmentation, dry forests, tropical forests.

## 1. Introduction.

Deforestation in South America has led to severe habitat fragmentation and the replacement of native forest with an agricultural matrix (Donald and Evans, 2006). It is expected that deforestation will have increased by 30% in 2050 (Donald and Evans, 2006; Tilman et al., 2001). Ecuador underwent the highest rate of deforestation in South America during the decades 1990-2000 and 2000-2010, at -1.5 and -1.8% per year$^{-1}$, respectively, (FAO, 2011) and lost 21,340 km$^2$ (1990-2020), which is approximately 15% of its remaining forests (FAO, 2020b). Seasonal dry forests in particular are undergoing one of highest rates of deforestation in South American, which has led the Ecuadorian dry forest to have a high rate of fragmentation (Rivas et al., 2021), and approximately 24% and 70% of the remaining forest have high and very high levels of fragmentation, which is one of the greatest threats to the Ecuadorian dry forest (Manchego et al. 2018; Tapia-Armijos et al. 2015). This makes these forests one of the most critically endangered ecosystems in South America (Ferrer-Paris et al. 2018), which may lead to the collapse of its biodiversity, even in protected areas (Laurance et al., 2012).

The negative impacts of fragmentation on ecosystems signifies that assessing the connectivity in highly fragmented landscapes such as the seasonal dry forest in Ecuador is essential if effective conservation plans are to be designed. Landscape connectivity can be defined as the degree to which the landscape facilitates or prevents movement between existing resources (Taylor et al., 1993; Dickson et al., 2019). Connectivity determines a host of ecological functions, including seed and animal dispersal, gene flow and the spread of disturbances. Understanding the flows of matter and energy (Kang et al., 2016) is, therefore, a key aspect in the preservation of biodiversity and ecological functions (UNEP, 2015; Hilty et al., 2020). Connectivity is evaluated at the landscape level, at which the reference scale is determined by the use of habitat or the scale of the movement of the targeted species (Tischendorf and Fahrig, 2000). Some authors have studied connectivity as structural connectivity, but advances provide a clearer distinction between structural and functional connectivity (Doerr et al., 2011). Functional connectivity can be ensured not only when existing habitat units are spatiality contiguous, but also when a permeable matrix or other connecting elements allow the movement of a particular organism between habitat areas that may be distant (Saura et al., 2010). This is a major concern for the maintenance of wildlife populations, ecological flows and many other landscape functions (Saura and Pascual-Hortal 2007), and can be evaluated through the concept of habitat species availability (Pascual-Hortal and Saura, 2006).

Several studies have shown the usefulness and relevance of the graph theory as regards generating ecological networks and model landscape connectivity (e.g. Sahraoui et al., 2017; Dickson et al., 2019). A graph is basically a set of nodes connected by a set of links (Savary et al., 2021). Although the graph theory is a common basis for all methods that handle network objects, each subject field has its own specific applications and software (Foltête et al., 2012). The graph theory is very useful for representing ecosystem connectivity, with the node being the habitat of the species and the links that connect the nodes representing the potential movement between patches (Rayfield et al., 2011). The use of graph theory as a means to estimate habitat connectivity is, therefore, rapidly increasing in popularity in both ecology and conservation biology (Rayfield et al., 2011; Savary et al., 2021; Tarabon et al., 2021), and this has led to the development of specific software such as Conefor or Graphab (Ray, 2005; Saura and Torné 2009; Shah and McRae, 2008). The latter provides an integrated set of methods, such as least cost distance calculation, landscape plot construction and connectivity measurement calculation, which can be visualised, used and analysed in a GIS (Foltête et al., 2012, Clauzel et al., 2019; Tarabon et al., 2021).

Here, we propose the application of Graphab software in order to estimate the connectivity of a highly fragmented ecosystem: the seasonal dry forest in the coastal region of Ecuador. This region has the lowest connectivity among natural vegetation in Ecuador owing to fragmentation caused by anthropic activities, and 83% and 14% of its ecosystems have very low and low connectivity, respectively (Rivas et al., 2020). It is, therefore, crucial to consider connectivity as the basis for conservation planning by which to maintain the viability of wild populations, reduce the risk of extinction and increase the stability and integrity of ecosystems (Pascual-Hortal and Saura 2006). The aim of this research is, therefore, to discover the functional connectivity of the equatorial dry forest between 1990-2018, with the following specific objectives: i) to study the fragmentation of the equatorial dry forest during three different periods (1990-2000, 2000-2008 and 2008-2018) and to study the connectivity between the patches and their evolution; ii) to identify areas without external connectivity, their evolution and their internal connectivity, and iii) to analyse the Ecuadorian state's reforestation plan and prepare a reforestation proposal by maximizing connectivity with the smallest possible area. The eventual aim of this work is to provide useful information on the functional connectivity of the Ecuadorian dry forest, which would be useful as regards developing effective protection measures and reforestation plans according to the present conservation status and future trends.

## 2. Materials and methods

### *2.1 Study area*

The study area focused on as a targeted habitat was the seasonal dry forest in the coastal region of Ecuador (Fig. 22). The coastal region of Ecuador is located along the Pacific Ocean and the west slope of the Andes Mountain range, and the Ecuadorian Ministry of the Environment defines it as a biographic sector (available at http://ide.ambiente.gob.ec/mapainteractivo). The entire Ecuadorian coast, plus 10 km of buffer area from the *Andean* region (mountain area), was selected. The coastal region has two clearly differentiable ecosystems: humid and dry (Ministerio del Ambiente del Ecuador 2013). In the present study, we considered the seasonal dry or deciduous forests (*sensulato*) of the Ecuadorian Pacific, which include deciduous (*sensustricto*) and semi-deciduous phenology, in which the dry periods last up to 8 months and 25–75% of tree and shrub species lose their leaves during the dry season (Prentice 1990; Rivas et al. 2020).

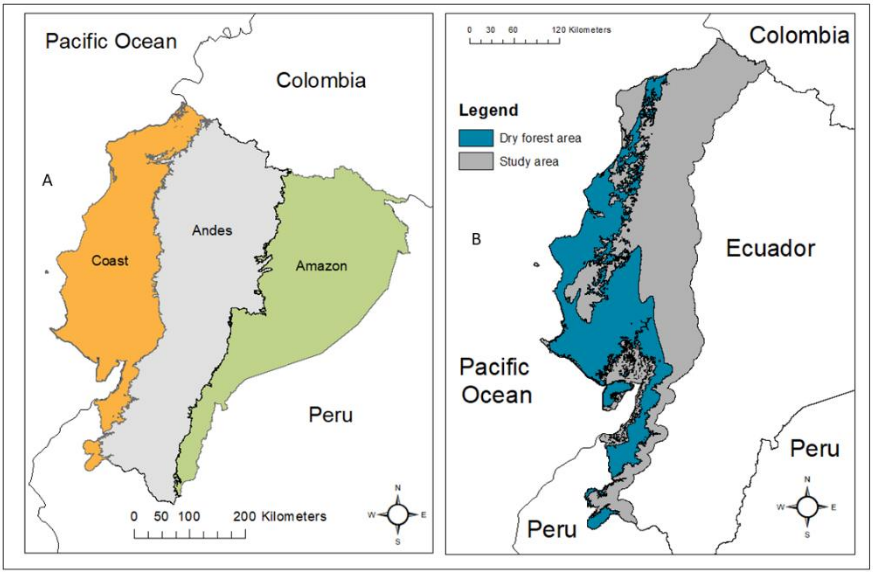


**Figure 22:** A) Map of continental Ecuador showing its three main geographical regions. B) The location of the seasonal dry forest in the study area (coast region of Ecuador).

The layers of phenology, flooded areas and land use were obtained from the Ecuadorian Ministry of the Environment (available at http://ide.ambiente.gob.ec/mapainteractivo) in order to limit the potential extent of the seasonal dry forests in the coastal region. The study was carried out by employing land uses maps created by the Ecuadorian Ministry of the

Environment using Landsat satellite images and Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER), orthorectification. These maps were later certified by experts and with field work, with a pixel size of 30 meters (MAE and MAGAP, 2015; Ministerio del Ambiente, 2017). Zones with deciduous and semi-deciduous phenology were selected, excluding flooded areas (mangrove areas) and areas without vegetation cover or without woody plants. The land use layers were then converted into raster files with a pixel size of 100x100 meters.

Changes in land use were calculated for the years 1990, 2000, 2008 and 2018. The original 12 land uses were then recategorised into six categories: 1-Anthropic zones (populated areas and infrastructures), 2-Flooded areas (artificial and natural water bodies), 3-Native forests (including humid and dry forests), 4- Other natural areas (area without vegetation cover, paramo, forest Plantation, shrub and herbaceous vegetation), 5-Agricultural land and 6-Areas for which no information was available (Table 15).

**Table 15:** Land-uses proposed by the Ecuadorian Ministry of the Environment and the assigned cost of moved through them.

| Name | Definition | Cost |
|---|---|---|
| Native forest | Arboreal ecosystem, primary or secondary, regenerated by natural succession; it is characterized by the presence of trees of different native no dry forest species, varied ages and sizes, with one or more strata. | 0 |
| Native dry forest | Arboreal ecosystem, primary or secondary, regenerated by natural succession; it is characterized by the presence of trees of different native dry forest species, varied ages and sizes, with one or more strata | 0 |
| Forest Plantation | Anthropically established tree mass with one or more forest species | 1 |
| Shrub vegetation | Areas with a substantial component of non-tree native woody species. Includes degraded areas in transition to dense canopy coverage and paramo. | 2 |
| Herbaceous Vegetation | Areas made up of native herbaceous species with spontaneous growth, which do not receive special care, used for sporadic grazing, wildlife or protection purposes. | 2 |
| Natural water | Surface and associated volume of static or moving water. | 5 |
| Artificial water | Surface and associated volume of static or moving water associated with anthropic activities and the management of water resources. | 5 |
| Populated Area | Areas mainly occupied by homes and buildings intended for communities or public services | 10 |

| | | |
|---|---|---|
| Infrastructure | Civil works of transport, communication, agro-industrial and social. | 10 |
| Area without vegetation cover | Areas generally devoid of vegetation, which due to their edaphic, climatic, topographic or anthropic limitations, are not used for agricultural or forestry use, however they may have other uses | 2 |
| Agricultural Land | Area under agricultural cultivation and planted pastures, or within a rotation between them, includes areas of annual crops, semi-permanent crops, permanent crops, grasslands and agricultural mosaic | 5 |
| No information | It corresponds to areas that have not been able to be mapped. | 5 |

### *2.2. Fragmentation analysis*

Fragmentation was analysed for 1990, 2000, 2008, and 2018, and forest patches were divided into three categories according to their size (<1 $km^2$, 1-10 $km^2$ and >10 $km^2$) in order to verify trends in the different periods. Five simple fragmentation metrics were calculated for each category: number of patches, average size, sum of areas per category, percentage of area per category and percentage of patches per category.

### *2.3. Connectivity analysis*

The connectivity metrics between the patches of native dry forest were calculated using the Graphab software (Foltête et al., 2012). The cost assigned to each land use was similar to that calculated by Pierik et al. (2016) (Table 15). Various Global metrics, component metrics and patch metrics concerning connectivity (Clauzel et al. 2019; Table S3) were calculated in order to create links between patches by means of cost analyses at three different distances (100, 50 and 5), in which a cost of 1 (each land cover has a different cost) and a of 100 meters corresponded to 10, 5 and 0.5 km. The links were calculated in 8 directions.

An analysis of the corridors or links between habitat patches of native dry forest was then carried out, during which their number, their distance in meters and their distance in costs in the years 1990, 2000, 2008, and 2018 at the same three distances were calculated. Once the connectivity analyses had been conducted, areas without connectivity were identified within the study area (e.g. component (Table S3) (Urban and Keitt, 2001)): a component is a group of connected nodes, signifying that organisms can move (link) between patches (nodes) into the same component, and patches of different components cannot, therefore, communicate because they are in isolation (Herrera et al., 2017). The number of components and their connectivity within the landscape were analysed for the years 1990, 2000, 2008, and 2018 by considering the same distances (10, 5 and 0,5 km).

*2.4. Spatial connectivity and fragmentation*

The study area was tessellated into 10 $km^2$ hexagons, and the Graphab programme was also used to measure the components generated in links of 0.5 km in the years 1990 and 2018. Only the hexagons containing patches of native dry forest were considered. The hexagons were classified by the number of components (zones without connectivity) in five categories: very high connectivity and very low fragmentation (1-2 components), high connectivity and low fragmentation (3-4 components), medium connectivity and medium fragmentation (4-7 components), low connectivity and high fragmentation (7-12 components), very low connectivity and very high fragmentation (more than 12 components).

The fragmentation patterns were described by employing the Getis-Ord Gi*analysis (Ord and Getis, 1995) for the years 1990 and 2018, considering the number of zones without connectivity or component values. The resulting z-scores and P-values indicate a spatial cluster of high or low values. The results were assigned to 7 values for each tile: Cold spot 99% confidence ($z < 2.58$; $p \leq 0.01$); Cold spot 95% confidence ($-2.58 < z < -1.96$; $p \leq 0.05$); Cold spot 90% confidence ($-1.96 < z < 1\text{-}.65$; $p \leq 0.1$); Not Significant ($-1.65 < z < 1.65$); Hot spot 99% confidence ($z > 2.58$; $p \leq 0.01$); Hot spot 95% confidence ($1.96 < z < 2.58$; $p \leq 0.05$), and Hot spot 90% confidence ($1.65 < z < 1.96$; $p \leq 0.1$). Clusters of high values were denominated as hot spots, while those with low values were cold spots.

*2.5. Reforestation plan.*

Connectivity metrics were calculated for two different reforestation scenarios: one performed according to our own connectivity data, and that proposed by the Ecuadorian Ministry of the Environment (Ministerio del Ambiente, 2019b). The metrics of that proposed by the Ministry were calculated using the land use map of 2018, and the areas with a very high priority of reforestation proposed by the ministry of the environment were added. Our own reforestation plan was created by using Graphad to identify areas or steppingstones that would increase the connectivity of the habitat with the lowest possible cost. The costs used for the calculation were 100 or 10 km (with cost 1).

### *2.8 Statistical analysis*

Three different generalised linear models were applied in order to test the effect of the year (1990 vs. 2018) and the distance (0.5, 5 and 10 km) on the connectivity correlation metric (Ccor; model-1), the distance in meters (model-2), and cost distance (model-3), which were considered as response variables. The interaction year*distance was also included in the three models, and a negative binomial distribution was used in order to consider data over dispersion. Finally, post hoc tests (Fisher LSD) within the linear models were developed to check for significant differences among the level of categorical independent variables. All the statistical analyses were carried out using Info-Stat software.

## 3. Results.

### *3.1 Land-use change*

Figure 23 shows the change in land use in the study area (76,550.7 $km^2$) and the dry forest area (27,704.55 $km^2$). The remaining dry forest area was reduced to 7,104.02 $km^2$ in 2018 (25.58%), with a loss of 2,647.21 $km^2$ and a percentage 37.26% of forest in 28 years (between 1990 and 2018). The native forest in the study area lost an area of 9,265.88 $km^2$ between 1990 and 2018.

The land on which forests had been lost was subsequently put mainly to agricultural use, which increased by 20.11% and 12.96% in the whole study area and in the dry forest area, respectively. However, the land use with the highest growth in relative terms was that of anthropic lands, although it still represented a small surface (less than 2% of the study area).

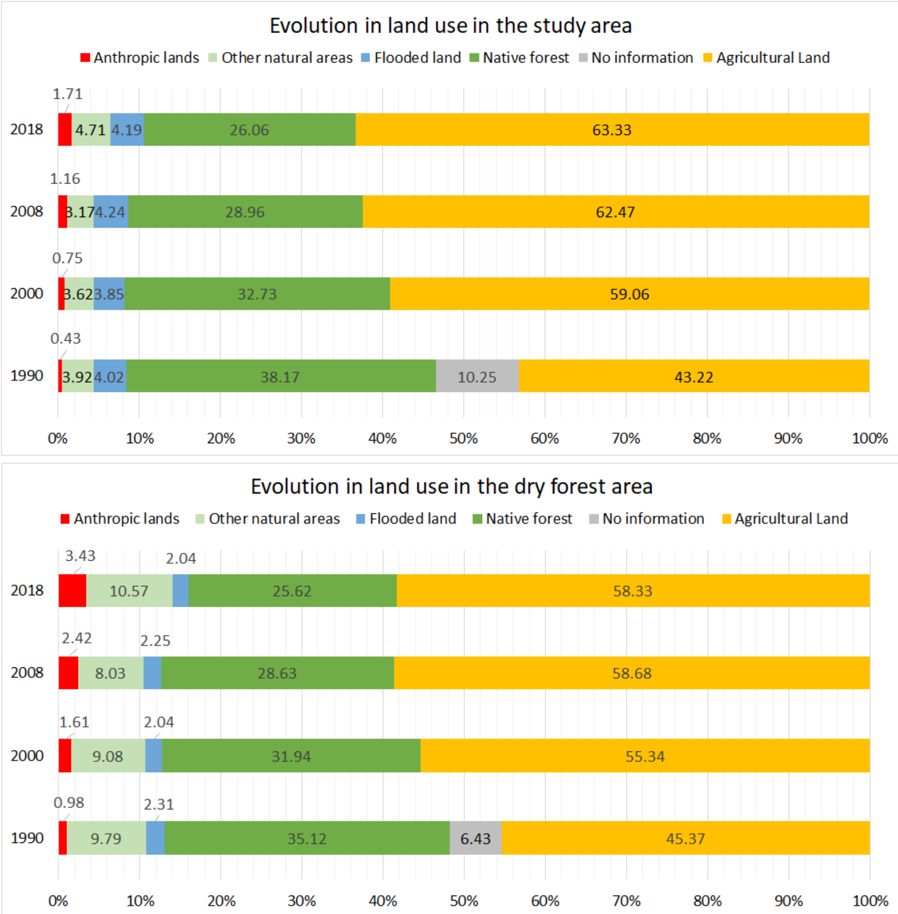


**Figure 23:** Evolution of land use in the study area and dry forest area in percentage in the years 1990, 2000, 2008 and 2018.

### *3.2 Fragmentation analysis*

The seasonal dry forests studied are becoming more and more fragmented (Table 16)**,** since the number of patches is decreasing, most of these patches are very small, and a significant number of these smaller patches are disappearing (<1 km²). Most of the dry forest area was included in a few patches larger than 10 km², which contain more than 80% of the total area. However, they lost 30% of their surface between 1990 and 2018, signifying that their average patch size was reduced from 122.71 km² to 93.3 km². With regard to the number of patches, the most frequent were those that were smaller than 1 km², which represents more than 93% of the total number of patches but which covered only 8% of the remaining area in 2018. The total number of patches decreased from 6,908 to 5,357 between 1990 and 2018.

**Table 16**: Evolution of the fragmentation parameters in the years 1990, 2000, 2008 and 2018 in three categories of patches according to their size

| Year | Area $km^2$ | Number of patches | Mean | Standard deviation | Area $Km^2$ | Percentage of area | Percentage of patches |
|---|---|---|---|---|---|---|---|
| 1990 | <$1km^2$ | 6554 | 0.1 | 0.17 | 680.33 | 6.98 | 94.88 |
| | 1-$10km^2$ | 286 | 2.54 | 1.84 | 726.5 | 7.45 | 4.14 |
| | >$10km^2$ | 68 | 122.71 | 494.28 | 8344.4 | 85.57 | 0.98 |
| | Total | 6908 | 1.41 | 50.17 | 9751.23 | 100 | 100 |
| 2000 | <$1km^2$ | 4696 | 0.11 | 0.18 | 526.9 | 5.95 | 93.84 |
| | 1-$10km^2$ | 250 | 2.72 | 1.95 | 679.9 | 7.67 | 5 |
| | >$10km^2$ | 58 | 132 | 401.65 | 7655.8 | 86.38 | 1.16 |
| | Total | 5004 | 1.77 | 45.14 | 8862.7 | 100 | 100 |
| 2008 | <$1km^2$ | 4026 | 0.12 | 0.18 | 478.2 | 6.02 | 92.79 |
| | 1-$10km^2$ | 256 | 2.05 | 2.85 | 730.2 | 9.19 | 5.9 |
| | >$10km^2$ | 59 | 114.2 | 359.32 | 6737.6 | 84.79 | 1.36 |
| | Total | 4339 | 1.83 | 43.59 | 7946.1 | 100 | 100 |
| 2018 | <$1km^2$ | 5018 | 0.11 | 0.17 | 569.1 | 8.01 | 93.67 |
| | 1-$10km^2$ | 277 | 2.71 | 1.95 | 750.2 | 10.56 | 5.17 |
| | >$10km^2$ | 62 | 93.3 | 266.8 | 5784.7 | 81.43 | 1.16 |
| | Total | 5357 | 1.33 | 30.17 | 7104.02 | 100 | 100 |

### 3.3 *Connectivity*

General connectivity decreased throughout the study period and increased with the distance of the links (Table 17). Some weighted connectivity metrics deteriorated dramatically during the study period: Flux in 0.5 km decreased by up to 75%, the IIC metric fell from 70% to 60%, and the EC metric increased from 57% to 69%, greatly increasing with the distance of the links. The values of the metrics derived from graph topology, such as NC, GD and H, fluctuated over time, but without being strongly influenced by the size of the links. For example, the NC (areas without connection between them) decreased from 4,094 with 0.5 km to 91 with 10 km in 1990. Area metrics (MSC, SLC, CCP and ECS) were not as affected throughout the study period as the other metrics were (Table 17).

The connectivity correlation metric (Ccor) was lower in 1990 than in 2018 (Model-1; F = 34.77; $P$<0.0001; Fig. 24) for the three distances considered, since the interaction was not significant (F = 2.08; $P$ = 0.1252), with lower Ccor values at the distance of 0.5 km (F = 478.81; $P$<0.0001) and there were no significant differences for 5 and 10 km (Fig. 24). Dry forests had very few fragments larger than 10 km$^2$ (62 in 2018), with high connectivity metric values (Fig. 25) and a majority of very small patches (5,018, representing more than 8.01% of the dry forest) with connectivity metrics close to zero.

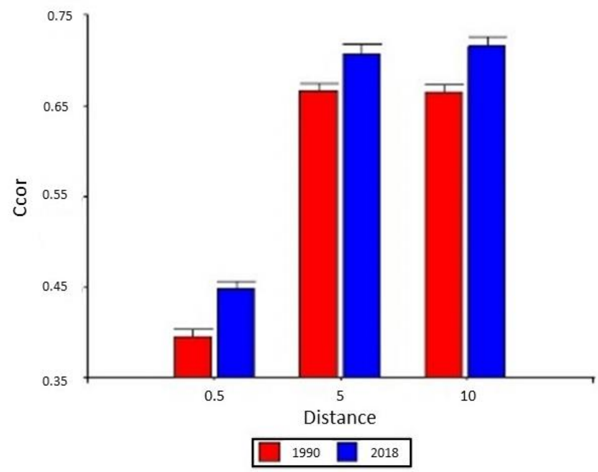


**Figure 24**: Connectivity correlation mean metric at the patch level in 1990-2018 at the three considered distances. Error bars represent the standard error of means.

**Table 17:** Evolution of connectivity metrics in the years 1990, 2000, 2008, 2018, 2018Min (reforestation plan proposed by the ministry) and 2018Ref (reforestation plan proposed in the current study) in the scenarios of links of 0.5, 5 and 10 km of Cost.

| Km | Metric | F | EC | PC | IIC | NC | GD | H | MSC | SLC | CCP | ECS |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 0.5 | 1990 | 2.19E+12 | 4.54E+09 | 4.92E-04 | 4.49E-04 | 4094 | 28.24 | 53848.43 | 2381834.39 | 4.36E+09 | 0.21 | 2.12E+09 |
| | 2000 | 1.57E+12 | 4.17E+09 | 4.14E-04 | 3.25E-04 | 2532 | 30.65 | 36751.45 | 3500312.01 | 4.01E+09 | 0.22 | 1.96E+09 |
| | 2008 | 5.57E+11 | 2.95E+09 | 2.08E-04 | 2.02E-04 | 2167 | 30 | 22124.21 | 3666866.64 | 2.71E+09 | 0.13 | 1.10E+09 |
| | 2018 | 5.61E+11 | 2.61E+09 | 1.63E-04 | 1.37E-04 | 3014 | 31.24 | 22667.94 | 2357007.3 | 2.33E+09 | 0.13 | 9.59E+08 |
| 5 | 1990 | 1.55E+13 | 5.51E+09 | 7.24E-04 | 4.92E-04 | 295 | 1339.14 | 728794.88 | 3.31E+07 | 4.94E+09 | 0.21 | 3.23E+09 |
| | 2000 | 8.59E+12 | 4.73E+09 | 5.33E-04 | 3.51E-04 | 311 | 666.45 | 330719.17 | 2.85E+07 | 4.39E+09 | 0.28 | 2.55E+09 |
| | 2008 | 5.29E+12 | 4.05E+09 | 3.91E-04 | 2.65E-04 | 243 | 541.4213 | 196575.16 | 3.27E+07 | 3.83E+09 | 0.26 | 2.09E+09 |
| | 2018 | 5.71E+12 | 3.78E+09 | 3.42E-04 | 1.85E-04 | 350 | 776.76 | 221094.61 | 2.03E+07 | 3.64E+09 | 0.28 | 2.04E+09 |
| 10 | 1990 | 3.51E+13 | 7.16E+09 | 0.00122108 | 5.50E-04 | 91 | 1623.24 | 1461629.46 | 1.07E+08 | 4.36E+09 | 0.21 | 6.09E+09 |
| | 2000 | 1.39E+13 | 5.45E+09 | 7.09E-04 | 3.77E-04 | 106 | 1312.02 | 550517.82 | 8.36E+07 | 5.30E+09 | 0.41 | 3.68E+09 |
| | 2008 | 9.52E+12 | 4.81E+09 | 5.51E-04 | 2.86E-04 | 69 | 1814.17 | 360617.75 | 1.15E+08 | 4.69E+09 | 0.4 | 3.24E+09 |
| | 2018 | 1.07E+13 | 4.30E+09 | 4.42E-04 | 1.99E-04 | 85 | 3823.02 | 453079.41 | 8.36E+07 | 5.16E+09 | 0.55 | 3.92E+09 |
| | 2018 M in | 1.82E+13 | 5.16E+09 | 6.36E-04 | 2.22E-04 | 86 | 1889.84 | 713465.73 | 9.02 E+7 | 5.43E+09 | 0.52 | 4.05E+09 |
| | 2018 Ref | 1.54E+13 | 4.80E+09 | 5.51E-04 | 2.08E-04 | 8 | 6421.89 | 625045.76 | 8.88E+08 | 6.57E+09 | 0.86 | 6.11E+09 |

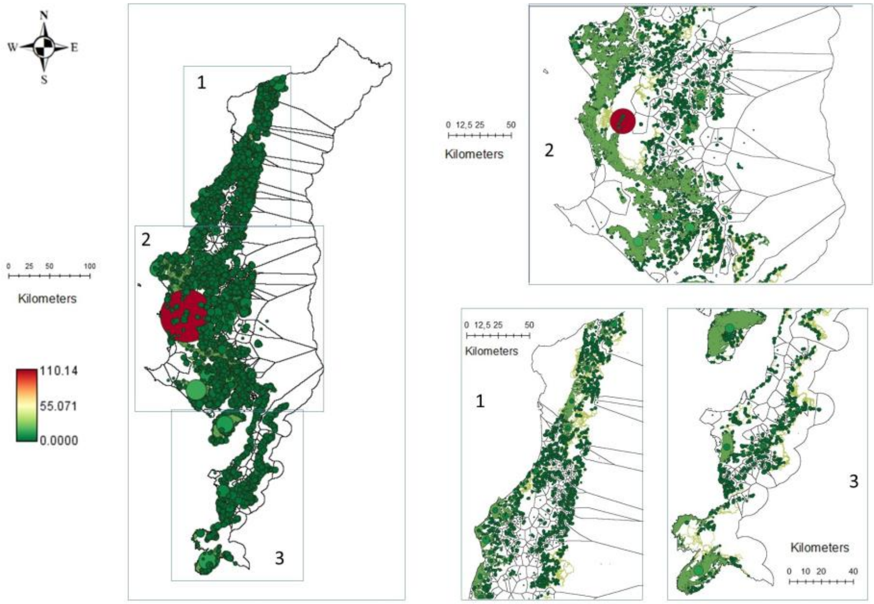


**Figure 25:** Connectivity correlation metric (Ccor) at the patch level, links and components in 2018 at the 5 km level. On the left the main image of the coast, the number 1,2 and 3 closest image to see more details, the size and color of the point indicate the value of the metric in the patch.

The number of connected patches increased as the distance increased (Table S4), but the number of connections decreased over time, since the number of connected patches decreased during the study period. There were 45,354 links in 1990 and 32,060 links in 2018. The median was zero as regards the links of 0.5-km distance in cost, while in the case of 5 and 10-km it was close to zero, which was well below average, and the distance in meters also decreased over time (Fig. 26).

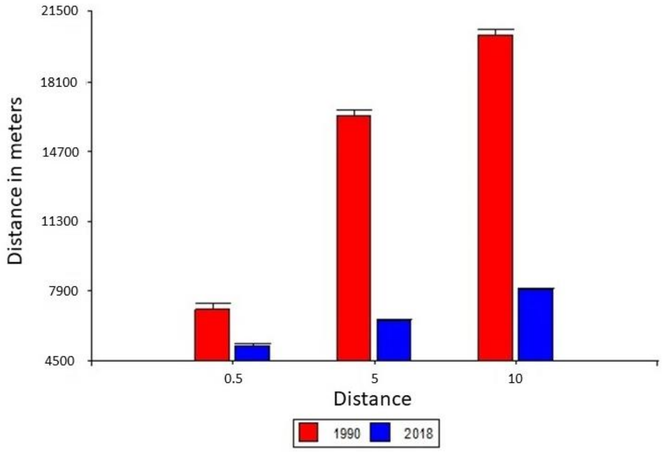


**Figure 26:** Distance in meters for the years 1990 and 2018 for the 0.5, 5 and 10 km links. Error bars represent the standard error of means.

With regard to the distance in meters, the results of Model-2 showed a significant effect of the year (F = 4840; $P<0.001$) and the distance (F = 1303; $P<0.001$), with higher distance values in 1990 than in 2018 for the three distances. The interaction year*distance was not significant (F = 223; p=0.215). Similar results were obtained for the cost distance (Model-3), with a significant interaction for the year (F = 149; $P<0.001$) and the distance (F = 1367; $P<0.001$), with lower distance cost values in 1990 than in 2018 for the three considered distances, but with no significant effect of the interaction year*distance (F=12.3; $P = 0.3465$).

Zonal connectivity decreased between 1990 and 2018 (Fig. 27). It should be noted that the zone of maximum connectivity (red) in the centre of the map decreased its connectivity at cost 5, and at cost 50. Moreover, the zone of maximum connectivity (red) located in the north was fragmented into multiple components with a low internal connectivity. At cost 100, the largest component with greater connectivity also underwent slight fragmentation, forming new areas without connectivity at its centre.

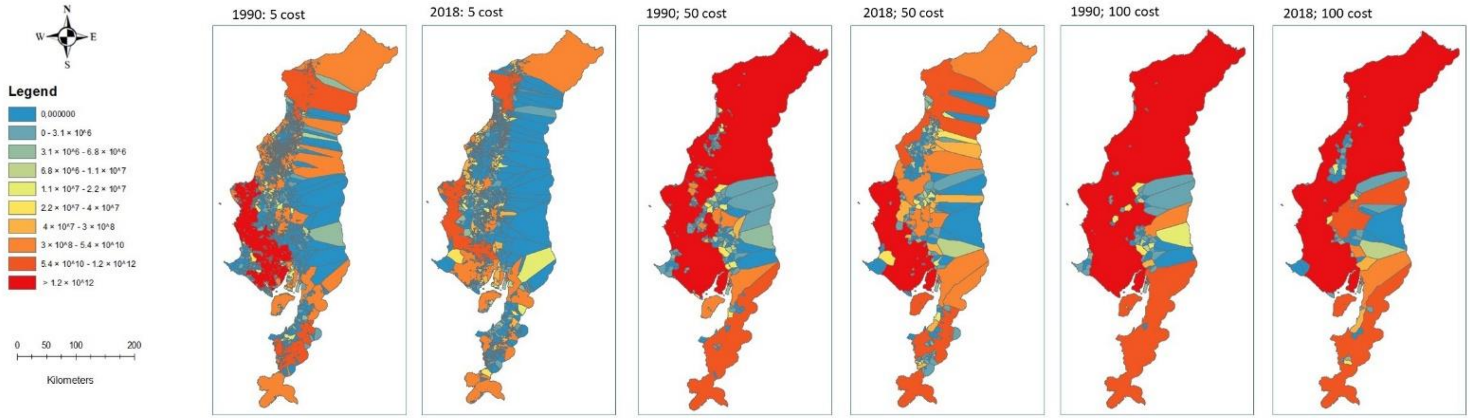


**Figure 27:** Metric Flux by components in the years 1990 and 2018 for costs of 5, 50 and 100 in the coast region of Ecuador, warmer colors indicate that component has more connectivity

*3.3 Spatial evolution of connectivity and fragmentation*

The high fragmentation and lack of connectivity was observed mainly in the northeast of the study area. This area was that most affected by fragmentation in 1990 and in 2018, and much of the dry forest habitats have now disappeared. The Getis-Ord Gi* analysis showed that the largest area classified as hot spots had partly disappeared in 2018 (Fig. 28). Moreover, new areas classified as hot spots of degradation appeared in 2018 in different places in the dry forest area.

*3.4. Reforestation plan.*

The Ecuadorian Ministry of the Environment has established an area of 653.24 $km^2$ with a very high preference for reforestation in the coastal region. Here, we propose the reforestation of only 1.62 $km^2$. The ministry's reforestation plan generally has better connectivity metrics than our proposal, and some connectivity metrics (e.g., F, EC, PC and IIC) are improved by between 15% and 10% (Table 17). However, our reforestation proposal has almost no areas without connectivity in the dry forest of continental Ecuador (Table 17 and Fig. 29), since the number of components (NC) is 8, while the ministry plan has 86, and the graphic size (GD) is 3.5 times higher in our proposal, connecting patches much further apart from each other, thus showing that all current forest fragments with less reforested area can be connected in order to decrease patch isolation.

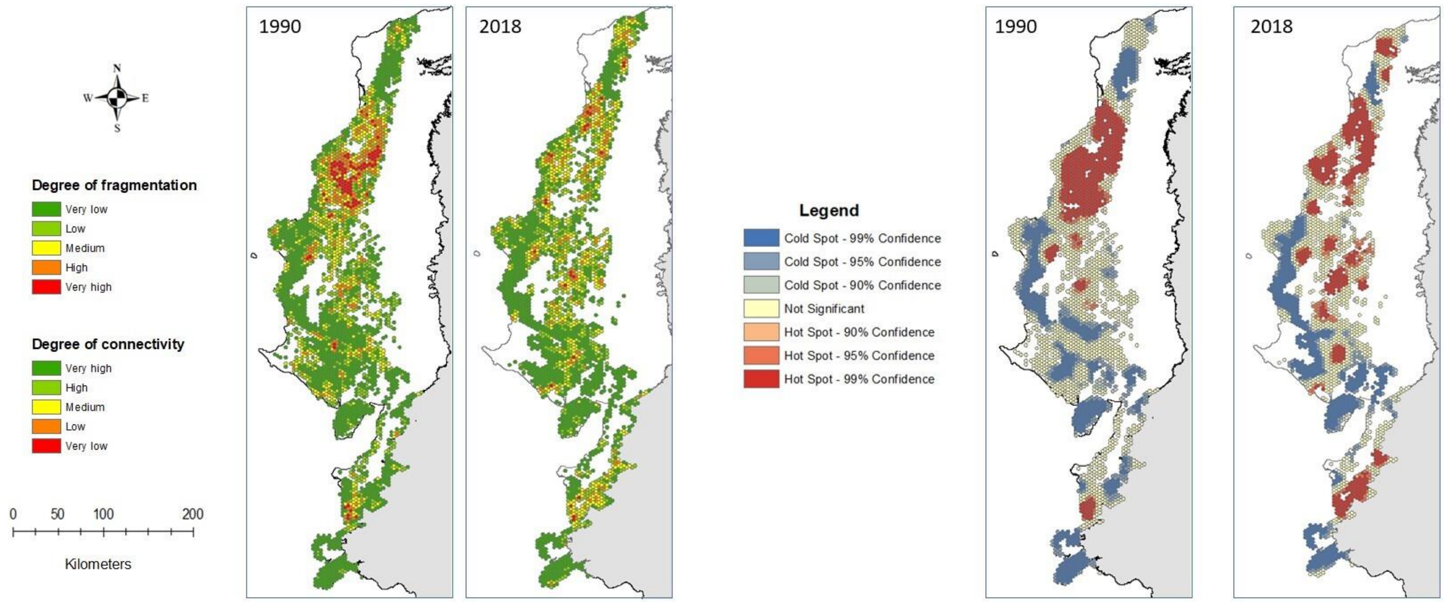


**Figure 28:** Spatial evolution of the connectivity and fragmentation of the equatorial dry forest per tile, and comparison of the hot and cold spots by means of Getis-Ord Gi* analysis in the years 1990 and 2018.

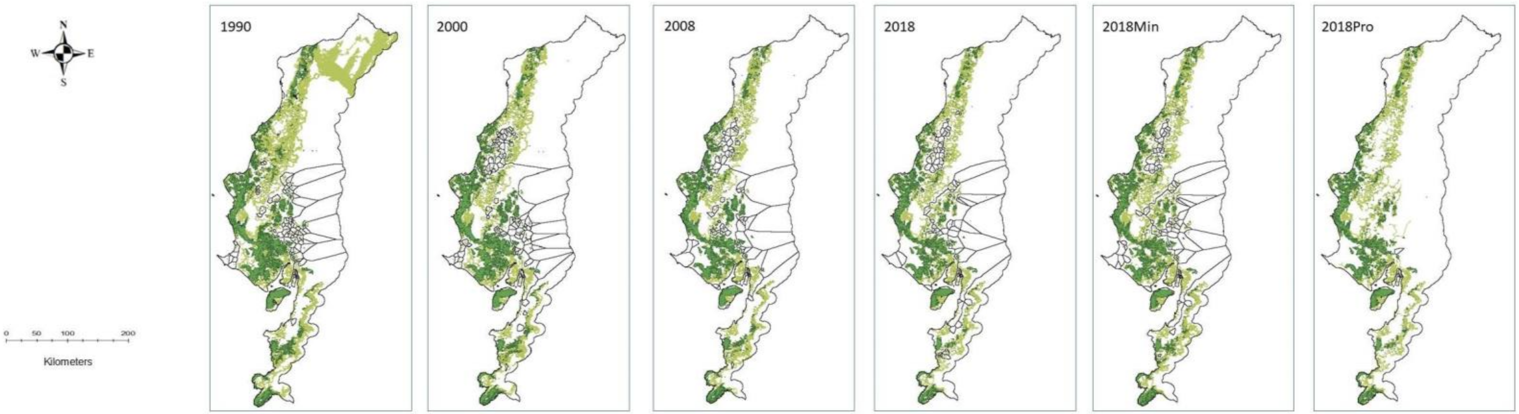


**Figure 29:** Evolution of connectivity with 10 km links for the four years and the two reforestation proposals (2018Min = proposed by the Ministry of the Environment; 2018pro = proposed by the authors) the habitat patches are observed in dark green, the links in light green and the components (NC) or areas without connection are represented by dark lines.

## 4. Discussion

Deforestation and fragmentation are some of the main threats to forests, which conserve much of the world's biodiversity (FAO and PNUMA, 2020). Dry forests in America are seriously threatened by increased deforestation (Ferrer-Paris et al., 2018), and the Ecuadorian dry forest is no exception (Rivas et al., 2021). Our results show that the coastal dry forest in Ecuador is undergoing sever deforestation, fragmentation and loss of connectivity as a result of the combined action of three factors: reduction of the total forest area, reduction of the number of patches and the changes in land use. The dry forests in Ecuador lost 2647.21 km$^2$ during the period 1990 to 2018, which have been transformed into other land uses, mostly agricultural, and which has led to the appearance of a large number of habitat fragments and the possibility of passing the threshold of extinction (Arroyo-Rodríguez et al., 2020).

### *4.1 Global Land use change and fragmentation analysis*

Forests throughout the world are being deforested, with their lands being converted into agricultural land (Laurance et al., 2014; Runyan and Stehm, 2018). This is also occurring with the Ecuadorian dry forest, in which agricultural land is, according to our results, replacing forests. This deforestation for conversion to agricultural land is caused by the increase in the world population (Laurance et al., 2014; Runyan and Stehm, 2018); in this respect, the Ecuadorian population has grown by 550% in only 71 years, reaching 17.5 million inhabitants in 2021 (Villacís Byron, 2012; Instituto Nacional de Estadística y Censo, 2021). Moreover, the Ecuadorian coast is the region that supports most of Ecuador's agricultural land, where the four main agricultural provinces of the country are located (Instituto Nacional de Estadística y Censo, 2019a). It is principally for this reason that this region has the highest deforestation rate in the country (Sierra, 2013). The increase in population is also reflected in the increase in anthropic lands, which are those that grow the most in relative terms, but which still occupy a lower proportion of the land.

Deforestation is leading to an increase in the fragmentation of the Ecuadorian dry forest (Rivas et al., 2021). In 2018, the number of patches was lower than in 1990, smaller patches disappeared, and large patches lost surface. In this scenario, small patches have been shown to be fundamental in highly fragmented ecosystems (Tulloch et al., 2016). The dry forest has a very small patch size (less than 1 km$^2$), both globally (FAO and PNUMA, 2020) and in Ecuador (Rivas et al., 2021). These very small patches may be affected by the edge effect (Hargis et al., 1998), which has serious consequences for biodiversity (Ruffell and Didham, 2016). Many areas of seasonal dry forest have degenerated into savannas, scrublands or grasslands owing to high pressure from livestock and overgrazing (Trejo and Dirzo, 2000; Sales et al., 2020). This would also explain our results concerning the increase in other natural areas (which include areas without vegetation cover, shrub, and herbaceous vegetation), as other natural areas in the dry forest area accounted for more than 10% in 2018, and increased between 2008 and 2018 by 703.7 $km^2$. These changes in land use and fragmentation have led 86% of the surface of the Ecuadorian dry forest to be classified as confronting a high degree of threat, and almost 70% as being of very high fragility (Rivas et al., 2020).

*4.2 Connectivity analysis*

Connectivity is, therefore, of vital importance in these highly fragmented ecosystems, and is a key factor for nature conservation (Foltête et al., 2012). However, dry forests are becoming more fragmented and more poorly connected. This loss of connectivity occurs throughout all of the coastal dry forests in Ecuador, although some areas have lost more connectivity (hotspots) and are, therefore, more vulnerable and more prone to habitat loss and should be defined as priority areas for protection or reforestation.

Connectivity parameters are classified into three families (Clauzel et al., 2019): Weighted metrics (W), which are based on distance and patch capacity criteria, area metrics (A), which are based primarily on the area criterion, and Topological metrics (T), which are derived from graph theory. This explains why, in general terms, our metrics decrease over time, since there are fewer areas and patches. The change of land use leads to an increasingly hostile matrix, thus making connectivity more difficult. Conversely, the increase in distance between the links favours connectivity, since more distant patches can connect or overcome the most hostile matrix, and connecting more distant patches eliminates their isolation, thus benefiting species that have a longer dispersal range (Ray, 2005; Herrera et al., 2017;)

The NC metric fluctuated throughout the study period. This was because it is necessary to consider the analysis of the number of components from a global perspective, since measuring the NC metric in isolation may be misleading. The disappearance of a component could mean an increase in connectivity (by joining this component with another) (Tiang et al., 2021) or the loss of the forest fragment that contains this component. The first option is owing to an increase in connectivity and the second to an increase in fragmentation (usually indicating a loss of connectivity), while the appearance of a new component may be owing to the loss of connectivity of a forest fragment or to the forestation of an area without connectivity (less fragmentation and greater global connectivity) (S5). For example, in our study, the ministry's reforestation plan has a similar number of components (NC) or areas without connectivity to those of dry forest in 2018 without reforestation (Table 17); however, it has more connectivity and more habitat areas because the ministry proposes to reforest areas that can form a component by not being connected to the existing native forest areas.

Patch to patch connectivity is declining and the links are becoming shorter, mainly owing to two factors: the change in land use that creates an increasingly hostile matrix, thus making connectivity difficult (Diniz et al., 2020), and the loss of native dry forest, which leads to a loss of habitat and patches, along with other types of forest that imply areas of high connectivity. It is important to bear in mind that an increase in the number of patches may favourably affect the number of links between patches, and the fact that even if they are small, they are very important for connectivity (Tulloch et al., 2016; Siquiera et al. 2021).

The change in land use towards a more hostile matrix leads to a loss of habitat and connectivity because links get shorter, thus isolating the remaining patches of the ecosystem, affecting biodiversity and reducing species abundance and richness (Sahraoui et al., 2017), which involves genetic and energy exchange (Ricotta et al., 2000). Our results show that other native forests mixed with dry forests play a fundamental role in connectivity, acting as "corridors" between remote patches or even nearby patches, and do not imply any displacement costs (Fig. 26 and Fig. 29).

The connectivity correlation metric (Ccor) examines the relationship between the degree of a node and the degrees of its neighbours (Rayfield et al., 2011). These could, perhaps, be priority conservation areas since they are highly connected regions with high forest cover. However, their highly interconnected nature also makes them vulnerable to the spread of disease (Minor and Urban, 2008). The authors showed that extrapolating the mean of patch metrics as a global connectivity metric is incorrect, since global connectivity decreases, whereas the average connectivity of the patches increases. As shown by our results, this is because small patches with low metric disappear, thus increasing the metric mean.

Graphab software has proved to be a powerful tool for connectivity studies, since it allows the analysis of connectivity from various points of view, from the macro (global connectivity) to the micro (patch connectivity), and passing through the intermediate (component connectivity), thus integrating different metrics (Foltête et al., 2012). Graphab can also be used to identify corridors between patches (Tiang et al., 2021), which would be the links generated by the software between the patches. These zones can be selected in order to protect, reforest, or even eliminate barriers, as occurs when overpasses are constructed over roads. Graphab software, therefore, makes it possible to not only analyse connectivity but also provide relevant information with which to improve it.

*4.4 Spatial evolution connectivity and fragmentation*

Upon comparing the images from 1990 and 2018 (Fig. 28), it will be noted that many hexagons classified as hotspots in 1990 had completely disappeared by 2018, which indicates where forest fragments may be isolated, with serious consequences for the ecosystem. The loss of connectivity has a negative effect on populations of species that inhabit these habitats, leading to serious risks of extinction (Tiang et al., 2021) and reducing the possibility of plant and animal populations exchanging genes and individuals (Herrera et al., 2017). The analysis used in this work makes it possible to identify areas with higher possibilities of becoming isolated and areas with many patches that are not connected, thus making it possible to identify the causes and take conservation or reforestation measures. This analysis also allows us to identify areas with high or low connectivity, which can also be obtained with the metrics by component. But, as will be noted in the number of component (NC) metric, this can increase or decrease with greater or less connectivity indistinctly, and its size is not homogeneous. This does not occur with this analysis, since the hexagons analysed are always the same size and always contain forest fragments (hexagons without forest fragments are not considered in the analysis). Nevertheless, there are very large components (NC) in which the majority of the area does not have forest surface.

*4.5 Reforestation plan.*

Graph theory has already proved its usefulness as regards increasing connectivity from an ecological restoration perspective (Sahraoui et al., 2017). In this study, we show that Graphab software has been efficient as regards selecting areas to reforest with the eventual goal of increasing connectivity. It has been demonstrated that the reforestation of small areas allows areas without connectivity to be eliminated, thus connecting the entire habitat. When all habitats are connected to form a single very large component, this indicates that no patches are isolated, thus creating a more connected habitat network (Herrera et al., 2017; Saura and Pascual-Hortal, 2007). This type of approach is already being carried out in several countries, as is the case of Nature Network (Ecologische Hoofdstructuur; EHS) in the Netherlands in order to improve connectivity between existing and new nature reserves that have yet to be created. The network helps prevent the extinction of plants and animals in isolated areas and the loss of value of natural areas (Donald and Evans, 2006).

The Ecuadorian Ministry of the Environment's reforestation programme proposes the reforestation of more surface area than our alternative. However, it has been shown that the number of components or areas without connection is greater than in our proposal, which shows that small patches or steppingstones are fundamental in connectivity studies (Herrera et al., 2017). According to this theory, and by using Graphab, areas in which connectivity has been lost owing to the loss of habitat or for which increased travel costs were selected and small patches (100x100 metres) were located could be connected, and it would, therefore, be possible to connect the entire ecosystem of Western Ecuador. This approach was even an improvement over the previous situation (1990) in which 85 components existed, while only 8 components remained in our plans, the majority of which were located on islands. In highly fragmented landscapes, scattered trees, small patches and corridors are recognised as important facilitators of movement (Tiang et al. 2021) and steppingstones help to connect distant or isolated patches (Baum et al. 2004). In summary, connectivity is shown to be a priority parameter when reforesting, thus prioritizing the connectivity of the entire study area rather than a particular reforested area.

**5. Conclusion**

Connectivity is considered a key issue for the biodiversity conservation, maintenance, stability and integrity of natural ecosystems, and is a fundamental attribute to take into account when conserving ecosystems (Hilty et al., 2020). When making conservation decisions, it is necessary to analyse from a global perspective and not only at the patch level, since each analysis and metric provides different relevant information (Saura and Pascual-Hortal, 2007). Corridors and stepping stones, which were very efficient as regards reducing the effects of fragmentation, are particularly important (Donald and Evans, 2006; Baum et al., 2004). Small patches are fundamental for the global connectivity of the ecosystem (Tulloch et al., 2016; Tiang et al., 2021) and for biodiversity (Wintle et al., 2019; Fahrig, 2020).

According to our results, the Ecuadorian dry forests are affected by fragmentation, deforestation and the loss of connectivity. Connectivity has, as regards some parameters, been reduced by more than 50% in 28 years. Our connectivity analysis shows that the other types of forest are fundamental to connectivity, since they are used as corridors or natural steppingstones, and although small patches have less connectivity per patch than large ones, they are also fundamental to global connectivity. Many areas are at high risk of disappearance, and it is, therefore, essential to take reforestation measures, with connectivity being a fundamental parameter to consider, along with prioritizing connectivity at the regional scale rather than reforesting a particular area.

The use of connectivity as a key factor, and the application of the Graphad software has proved important when taking protection, conservation and reforestation measures. This information allows the identification of important patches or areas in the ecosystem's connectivity, areas prone to disappearance and the loss of connectivity related to reforestation plans. This approach, therefore, enables a better choice of areas that will increase connectivity with the least amount of cost or effort by choosing areas in which to place steppingstones, thus eliminating areas without connectivity.

# Discusión general

El objetivo general de esta tesis ha sido estudiar el grado de protección del bosque seco de Ecuador, y la evolución de su deforestación, fragmentación y conectividad. Para ello se trabajó con herramientas de análisis de usos del suelo en un entorno de sistemas de información geográfica, se calculó el área cubierta por bosque seco costero ecuatoriano y esa información se analizó incorporando la capa de las figuras de protección de Ecuador. Los resultados principales indican que el bosque seco está poco protegido, que algunas de las figuras de protección que presenta son poco efectivas, y que tiene una alta tasa de deforestación que conllevan a un aumento de la fragmentación y a una pérdida de conectividad.

**Estado de protección y conservación.**

Los resultados de esta tesis muestran que el bosque seco presenta peor estado de conservación y grado de protección que el bosque siempreverde de la costa, ya que sólo el 12,9% del remanente de bosque seco está protegido, frente al 27.9% de bosque siempreverde. Además, un 14,72% de la superficie de las áreas denominadas "Patrimonio de Áreas Naturales del Estado" (PANE) son destinadas a proteger el bosque seco, en comparación con el 36,8% destinado a proteger el bosque verde. En el caso de los bosques protectores, el bosque seco sólo representa el 11,30% de su superficie total, y sin embargo, el bosque verde representa un 30,91%. En definitiva, hay un claro sesgo hacia favorecer la protección de los bosques siempreverdes, estando el bosque seco muy infrarrepresentado dentro de las áreas protegidas.

Al comparar el grado de fragmentación, conectividad, amenaza, vulnerabilidad y fragilidad, el bosque seco presenta peores valores en términos generales. La región costa es la región del país con menos protección (Lessmann et al., 2014; Parque Nacional Galápagos 2021) y según los resultados de este tesis, el bosque seco, a pesar de su peor estado de conservación, presenta menor protección que el bosque verde en la región costera. Esto podría deberse a diversos factores, como son el número de especies (diversidad alfa) que habitan los bosques secos (Sierra et al., 2002), el menor valor paisajístico o a la percepción de que los ecosistemas secos suelen ser ecosistemas con menor valor en términos de biodiversidad (Siyum, 2020). Sin embargo, los bosques secos son esenciales para mantener la diversidad gamma a nivel regional, ya que la mayoría de las especies de plantas que lo componen difícilmente son observables en bosques húmedos, albergando altos niveles de especies endémicas y otras con distribuciones restringidas (Sierra et al., 2002; León-Yánez et al., 2011). Por otro lado, la distribución del bosque seco está mayoritariamente en la costa (Ministerio del Ambiente del Ecuador, 2013), existiendo muy pocos remantes de este bosque en la región Andina. Sin embargo, la presencia del bosque siempreverde es más extensa tanto en Ecuador como en Sudamérica, lo que hace aún más crítica su situación, ya que su desaparición en la costa ecuatoriana llevaría a la desaparición del ecosistema en Ecuador.

Otro factor preocupante es la alta tasa de deforestación que presenta el bosque seco, que junto a que las pocas figuras de protección que tiene y a la poca efectividad de los Bosques Protectores para evitar la deforestación, ha llevado al bosque seco ecuatoriano a encontrarse actualmente en peligro crítico de extinción (Ferrer-Paris et al., 2018). Por todas estas razones se necesita un plan urgente de protección con medidas efectivas y la concienciación de la población que habita estos bosques para evitar la desaparición o el colapso del ecosistema.

### Deforestación y fragmentación del bosque seco Ecuatorial

El bosque seco costero de Ecuador ha sufrido una tasa de deforestación de -1,12% entre 1990 y 2018, perdiendo 2.631,91 km$^2$, es decir, un 27,04% de la superficie existente en 1990 (Fig. 30). Esta deforestación ha llevado a un aumento de la fragmentación, provocando que el índice de fragmentación reticular (RFI) haya aumentado un 11,61%, sufriendo una disminución del tamaño, del perímetro y del número de teselas (Fig. 31).

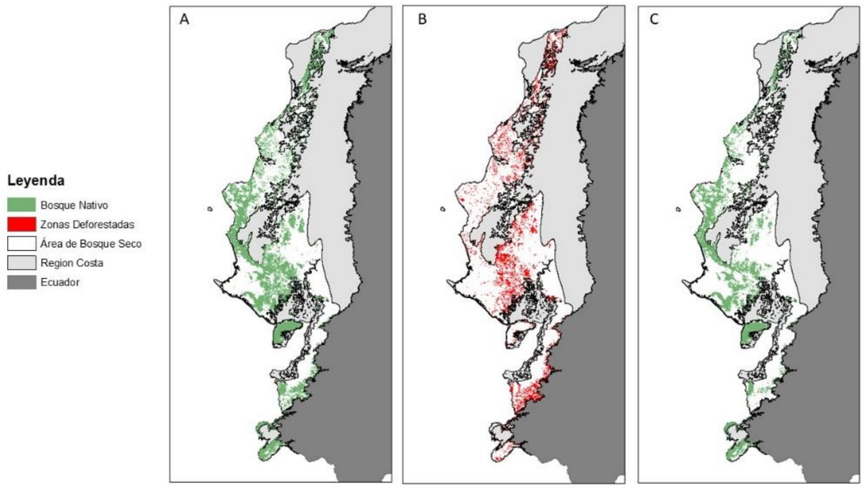


**Figure 30:** A) Bosque seco nativo en 1990. B) Deforestación entre 1990 y 2018. C) Bosque seco nativo en 2018. Fuente: elaboración propia a partir de los datos del Ministerio del Ambiente.

La deforestación, fragmentación y pérdida de bosque se producen principalmente en las zonas de Manabí, el Guayas y en el centro-sur de la costa, estando posiblemente correlacionada con el aumento de la población y de la superficie agraria (Instituto Nacional de Estadística y Censos, 2019a), siendo estas zonas las de mayor riesgo de desaparición de bosque seco.

La fragmentación aparece también al interior de las áreas naturales, presentando las zonas incluidas en la figura de Bosque Protector los valores más elevados de fragmentación, posteriormente las zonas sin protección y por ultimo las zonas protegidas con la categoría de PANE (Fig. 31). Por lo tanto, no todas las medidas de protección que se toman están siendo efectivas, particularmente la figura de Bosque Protector, donde ha ocurrido un aumento de fragmentación incluso por encima de las zonas no protegidas.

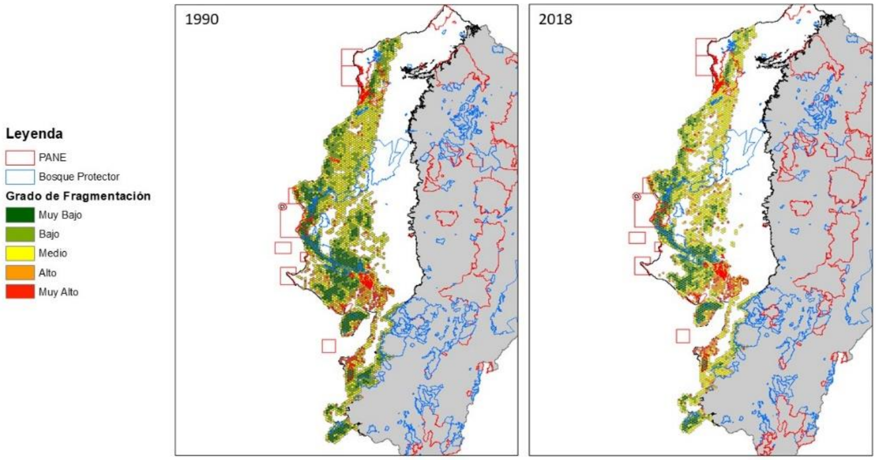


**Figura 31:** Evolución de la fragmentación del bosque seco entre 1990 y 2018 con las áreas protegidas incluidas en el PANE y Bosque Protector.

La mayoría del terreno deforestado se ha dedicado a agricultura (87%) (Fig. 32), y un 7% se ha transformado en matorrales. La agricultura es una de las principales amenazas de los bosques en los trópicos (Laurance et al., 2014), y este aumento de la agricultura viene dado por el fuerte aumento poblacional que ha sufrido Ecuador, ya que su población ha crecido un 550% en 71 años (Villacís Byron, 2012; Instituto Nacional de Estadística y Censos, 2021). El cambio de uso de suelo a matorrales puede haberse producido por un proceso de "sabanización" (Fig. 33), el cual consiste en que los bosques secos han degenerado en sabanas, matorrales o pastizales debido a la alta presión del ganado y al sobrepastoreo, lo que podría ser la causa de la desaparición de los pequeños fragmentos de bosque seco, limitando el crecimiento y extensión del bosque (Trejo y Dirzo, 2000; Sales et al., 2020). En otros casos el proceso viene provocado porque algunos campos de cultivos se abandonan, siendo colonizados por formación de matorral.

**Figure 32:** Deforestación del bosque seco para establecer cultivos de maíz. Es frecuente observar cómo pequeñas manchas de bosque se deforestan para establecer cultivos, lo que contribuye a aumentar la fragmentación, aunque la superficie deforestada no sea muy extensa. Autor: Carlos Rivas

**Figure 33:** En la parte inferior de la imagen se puede observar una zona de bosque seco que ha sufrido un proceso de “sabanización”, donde el estrato arbóreo ha sido reemplazado por estrato arbustivo de bajo porte. Autor: Carlos Rivas.

En la zona cubierta por bosque seco, porcentualmente el uso de suelo que más ha crecido entre 1990 y 2018 ha sido el correspondiente a las zonas antrópicas, aumentando un 350% en ese periodo, pasando del 0,98% de la superficie a un 3,43% en el área de estudio. Este aumento viene asociado al aumento poblacional mencionado en el párrafo anterior, y actúa como una causa de deforestación del bosque seco (Fig. 34).

Dentro de los bosques secos, el bosque semideciduo es el que más ha sufrido la fragmentación (IRF más alta y mayor aumento), lo que se puede deber a que los ecosistemas más húmedos se ven como más ricos en recursos y las áreas secas (como los bosques deciduos) como más pobres en recursos (Siyum, 2020) y menos aptos para la agricultura. Esto hace que aumente la presión sobre el bosque semideciduo, ya que sus árboles conservan sus hojas y su verdor durante más meses (Ministerio del Ambiente del Ecuador, 2013). Esta presión sobre el bosque semideciduo ha provocado una reducción del tamaño de las teselas, que presentan un valor medio muy bajo (0,58 $km^2$), aunque son precisamente este tipo de teselas las que han demostrado ser muy importantes en ecosistemas altamente fragmentados (Tulloch et al., 2016).

**Figura 34:** Es habitual que debido al aumento poblacional los habitantes deforesten el bosque seco para establecerse. Autor: Carlos Rivas.

**Un nuevo método de cálculo de fragmentación de una tesela.**

Debido a la gran cantidad de métricas a escala de teselas que sea han propuesto, y que ninguna por sí misma indica el estado de fragmentación real (Hargis et al., 1998; Jackson y Fahrig, 2013; Fahrig, 2019), en esta tesis se ha propuesto un índice de fragmentación de teselas (PFI), que agrupa varias de ellas para crear una nueva fórmula de cálculo. La fragmentación es parte de un amplio debate sobre si su efecto es perjudicial o si es beneficioso para la biodiversidad, así como la forma en que se mide, si como un patrón o como un proceso (Arasa-Gisbert et al., 2021). Este debate ha llevado a una constante investigación en este campo, donde existen múltiples parámetros para medir la fragmentación, aunque estas se pueden agrupar principal en 4 tipos de métricas (Chen et al., 2014). Además, se han identificado cuatro parámetros que se han correlacionado como aquellos que presentan mayores efectos para la biodiversidad, y que están correlacionados con el efecto o parámetro de área, la forma, el perímetro y el aislamiento (Fig. 35) (Rogan y Lacher, 2018).

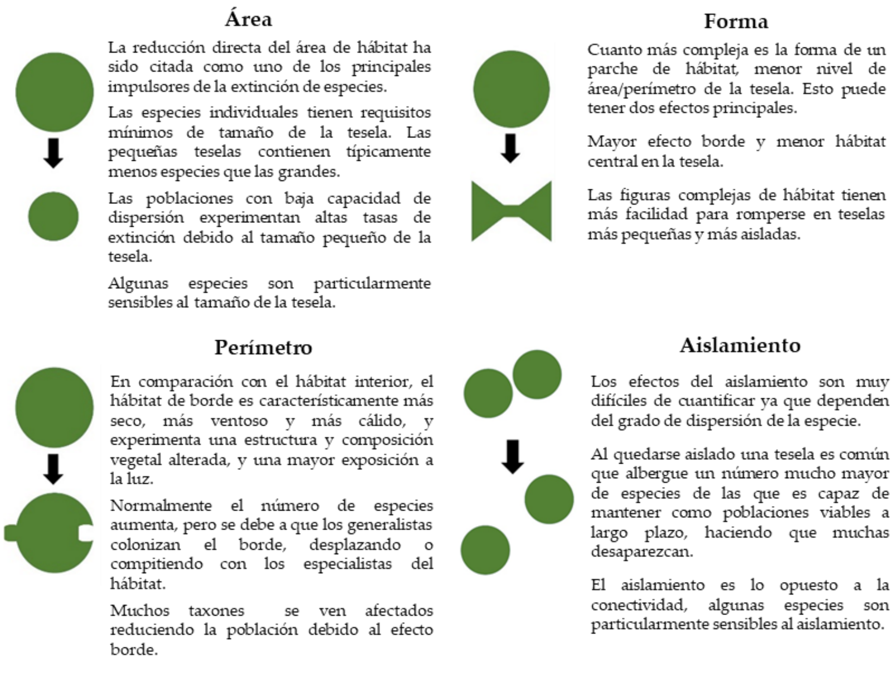


**Figura 35:** Representación de los principales efectos de la fragmentación (elaboración propia) y descripción de sus principales efectos sobre la biodiversidad. Adaptado de Rogan y Lacher (2018).

Uno de los problemas del cálculo de la fragmentación es que esta se mide como si todos los ecosistemas crecieran homogéneamente, es decir, sin condicionantes que limiten su crecimiento o hagan que se mezclen con otros ecosistemas. Por otro lado, otro de los problemas asociados al cálculo de fragmentación es que los investigadores han usado la zonificación para calcularlo, estas zonificaciones suelen ser ráster o hexágonos, los cuales también están condicionados por la orografía y por el tamaño de píxel. Para evitar estos problemas, el índice propuesto incorpora el parámetro área de influencia (Fig. 36), que es el área donde la tesela puede crecer o bien el terreno que ocuparía si no hubiera sufrido deforestación.

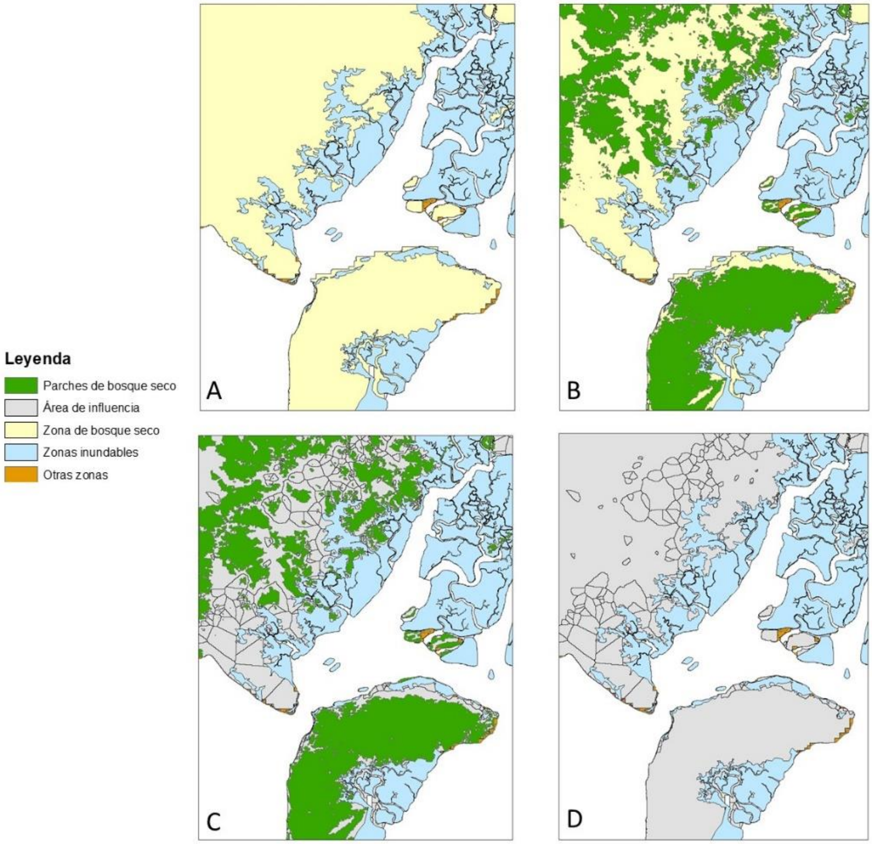


**Figura 36:** Zona de la costa ecuatoriana entre la desembocadura del río Guayas y la isla Puná, zona de mezcla de diversos ecosistemas como bosque seco, manglares (zonas inundables) y otros ecosistemas (otras zonas). A) Mezcla de ecosistemas. B) Mezcla de ecosistemas y los parches de bosque seco. C) Parches de bosque seco con su área de influencia. D) Área de influencia de cada parche se adapta al terreno donde puede crecer dicho parche (zona de bosque seco).

El PFI recoge los principales parámetros que afectan a la biodiversidad, lo que permite la identificación de la fragmentación de una tesela dándole un valor entre 0 y 1, siendo 1 el mayor grado de fragmentación teórico (nunca se alcanza el valor máximo de 1 al significar la desaparición del parche y por tanto no tener área ni forma). El índice se ha mostrado eficiente también en el monitoreo de la fragmentación durante un periodo de tiempo, e incluso puede ser usado para el cálculo de la fragmentación por áreas, pudiendo hacer análisis de patrones espaciales, donde se ha mostrado eficiente en la identificación de parches con riesgo de desaparición.

**Conectividad del bosque seco Ecuatorial.**

Uno de los efectos negativos de la fragmentación y de la deforestación es la perdida de conectividad, la cual es fundamental para para la conservación, el mantenimiento, la estabilidad y la integridad de la biodiversidad de los ecosistemas naturales, y es un atributo fundamental a tener en cuenta a la hora de conservar los ecosistemas (Hilty et al., 2020).

En los bosques secos de Ecuador se ha observado una pérdida de conectividad global superior al 50% en algunos de los parámetros durante el periodo comprendido entre 1990 y 2018. Esta pérdida de conectividad funcional se debe principalmente a la pérdida de hábitats y a su cambio hacia una matriz menos permeable. El bosque seco se está deforestando y fragmentando, lo que a su vez ha provocado que el número de teselas se haya reducido desde 6.908 en 1990 a 5.357 en 2018. Sin embargo, cuando se usa el área de influencia se observa una perdida mayor, de 3.451 teselas. Esta diferencia significa que las teselas más grandes se están dividiendo en teselas más pequeñas, mientras que las teselas pequeñas están desapareciendo, lo cual ya se observó al estudiar las métricas de fragmentación. Esta disminución del número de teselas, y el cambio de los usos de suelo hacia una matriz más hostil hace que las conexiones entre ellas sean menores y se reduzca la distancia media entre ellas.

Las teselas pequeñas (Fig. 37) juegan un papel fundamental en la conectividad (Diniz et al., 2021; Herrera et al., 2017; Siqueira et al., 2021), y en el mantenimiento de la biodiversidad (Tulloch et al., 2016; Fahrig, 2020), pudiendo ser la suma de muchas teselas pequeñas tan importantes como pocas teselas más grandes (Tulloch et al., 2016), lo que hace muy recomendable su protección (Fahrig, 2020). Las teselas pequeñas también han mostrado tener una gran importancia a la hora de aumentar la conectividad mediante acciones de restauración ecológica, ya que pueden actuar de teselas de paso para conectar zonas que antes no tenían conectividad. Por lo tanto, estas teselas pequeñas tienen que ser consideradas en una estrategia regional más amplia para aumentar la conectividad, lo cual es fundamental para asegurar la preservación de los bosques en estos ambientes altamente fragmentados (Fig. 37).

**Figura 37:** Las teselas o parches pequeños de bosque entre cultivos en la Región Costa de Ecuador son fundamentales para mantener la conectividad. Autor: José Guerrero.

La conectividad también se puede estudiar zonalmente, tanto por componentes o por teselas. Estos estudios permiten analizar una determinada zona y estudiar mejor las causas del aumento o pérdida de conectividad, así como determinar zonas prioritarias, por ejemplo, para acciones de reforestación en áreas que están desapareciendo y en las cuales se quiere aumentar la conectividad, o bien para conservar las funciones ecológicas propias de la conectividad.

Otro de los rasgos importantes que aporta la conectividad es la conectividad por teselas, que permite identificar teselas con una alta o baja conectividad, y que pueden ser útiles a la hora tomar decisiones como pueden ser las relacionadas con su conservación (Rayfield et al., 2011). Dado que la conectividad está relacionada con múltiples variables ecológicas, estas teselas también pueden usarse en el caso de especies invasoras o parásitos (Minor y Urban, 2008).

La conectividad, como ya se indicó, debe ser tenida en cuenta en las acciones de restauración, dado que muchos de los aspectos biológicos de la biodiversidad dependen de la conectividad (Kool et al., 2013), por lo cual mantener la conectividad del ecosistema es fundamental para mantener su funcionalidad (Hilty et al., 2020). Bajo esta premisa, el estudio de la conectividad en el diseño de políticas de restauración o reforestación tiene mucha importancia, debiendo priorizarse en la elección de las áreas prioritarias de restauración, el efecto que estas acciones tienen sobre conectividad del ecosistema, que el criterio relacionado con la cantidad de hábitat previamente deforestada. En otras palabras, cuando los recursos para reforestar son limitados, es preferible reforestar pequeñas superficies que aumenten mucho la conectividad del paisaje, en lugar de intentar aumentar la superficie reforestada sin que esto contribuya a un aumento significativo de la conectividad global.

**Posibles mejoras y futuros estudios:**

En términos generales, esta tesis muestra una visión completa del estado y evolución del boque seco ecuatoriano (Fig. 38) aportando información muy valiosa para su comprensión, la toma de medidas y el posible aumento de su protección.

El estudio sobre la conservación, la deforestación, la fragmentación y la conectividad del bosque seco se podría mejorar de diversas formas. La primera, es la fiabilidad de las capas de información geográfica utilizadas en los estudios de cambio del bosque seco. Por ejemplo, la capa de uso de suelo empleada en esta tesis presenta un índice Kappa de 0,7, por lo cual, aunque la información es veraz, es mejorable. Esto supone que parte de la información geográfica que se utiliza para estudiar los cambios de los ecosistemas puede hacerse a partir de información o de datos que pueden presentar inconsistencias o no tener en cuenta determinados parámetros. La progresiva mejora de las fuentes de datos numéricos y cartográficos, y su integración con trabajo de campo, puede contribuir a mejorar sustancialmente la calidad analítica de los estudios de cambio de usos en sistemas forestales. Por otro lado, el uso de teselas en los estudios de fragmentación, aunque tiene ventajas para identificar patrones espaciales puede no ser exacto en determinadas circunstancias. En el análisis de conectividad, los parámetros de permeabilidad y desplazamiento utilizados en este trabajo son orientativos, dado que son parámetros poco estudiados en muchos tipos de bosque, por lo que se pueden desarrollar estudios específicos para formaciones tropicales, así como para las especies de fauna asociadas.

A partir de las métricas de deforestación, fragmentación y conectividad, y aunque la agricultura y el aumento poblacional parecen ser los principales factores de deforestación, se podrían hacer estudios más avanzados para identificar otros agentes relacionados con estos procesos, como pueden ser factores de tipo climático, topográfico, o antrópicos que aumentan la probabilidad de deforestación mediante herramientas estadísticas más potentes. Finalmente, sería interesante comprobar la importancia de los pequeños remanentes de bosque para la conectividad funcional mediante trabajo de campo in situ seleccionando algunos grupos taxonómicos (e.g. mamíferos).

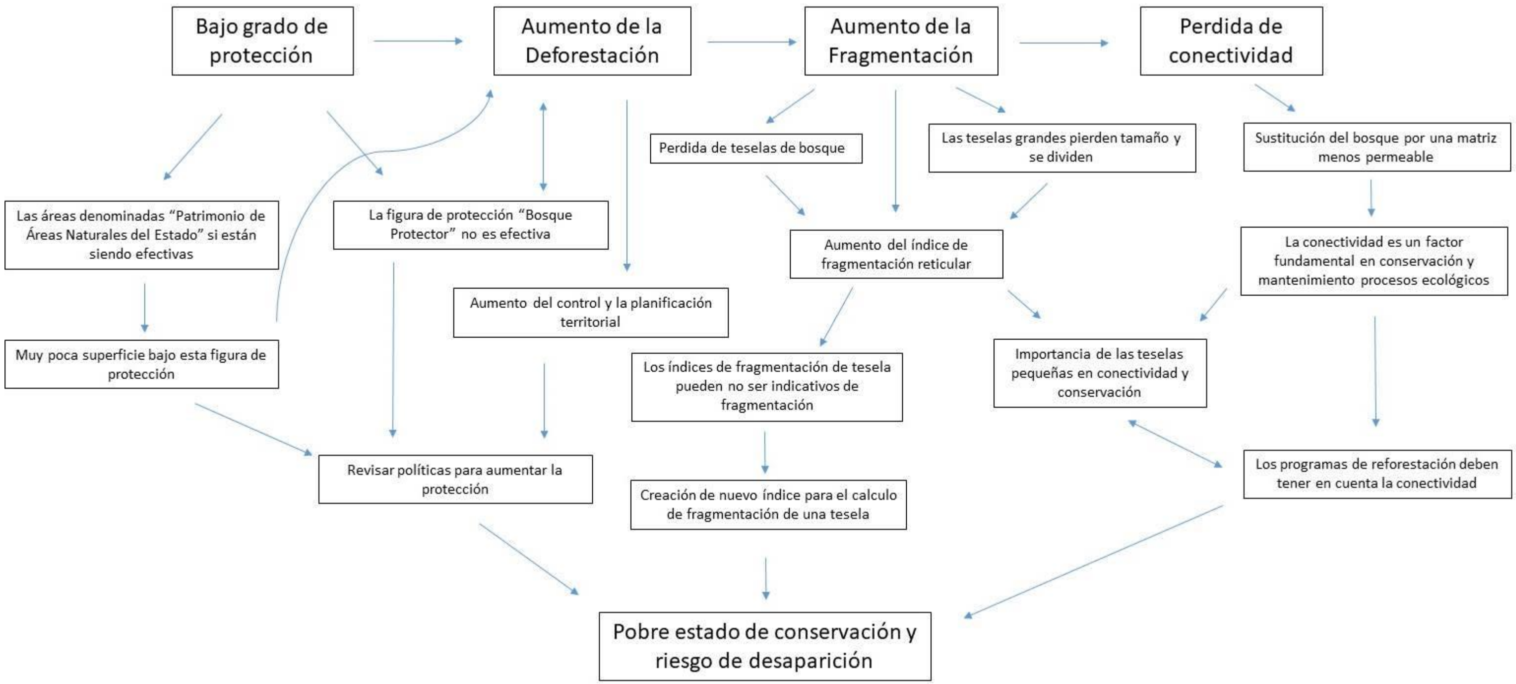


**Figura 38:** Esquema de la estructuración, conexión y aportaciones de los diferentes capítulos de esta tesis doctoral.

## Conclusiones

1. El bosque seco presenta un peor estado de conservación que el bosque siempreverde en la región de la Costa de Ecuador, con peores niveles de conectividad, fragilidad y amenaza.
2. El bosque seco presenta menor grado de protección que el bosque siempreverde de la costa ecuatoriana, donde sólo el 12,9% del bosque seco remanente está protegido, frente al 27,9% del bosque siempreverde.
3. Dado que los bosques siempreverdes han recibido más atención por parte de investigadores y administradores de recursos naturales, los esfuerzos de conservación ahora deberían estar más enfocados en la preservación de los ecosistemas secos, protegiéndolos como parte de una conservación regional más compleja.
4. El bosque seco presenta una alta tasa de deforestación en los últimos 28 años, perdiendo un 27,04% de la superficie remanente de 1990.
5. La fragmentación ha aumentado desde 1990 y el número de teselas ha disminuido como resultado de la reducción del área de bosque, aumentando así el borde y disgregando el bosque en fragmentos aislados, y en consecuencia haciéndolo más expuesto a cambios antrópicos y naturales.
6. Existen muchas zonas de bosque seco con un alto grado de fragmentación y en riesgo de desaparición, donde es urgente tomar medidas para evitar la pérdida de las teselas de bosque que todavía se conservan.
7. El bosque seco está poco representado en las áreas naturales de Ecuador, además la figura de Bosque Protector no está resultando efectiva para prevenir la deforestación y la fragmentación, frente a las áreas denominadas "Patrimonio de Áreas Naturales del Estado" que sí están resultado efectivas.
8. Dentro de los distintos tipos de bosque seco, el bosque semideciduo es el más afectado por la fragmentación, presentando un tamaño medio de tesela muy bajo.
9. El bosque seco presenta un índice de fragmentación por teselas muy alto, habiendo aumentado su valor considerablemente desde 1990, lo que representa un gran riesgo de desaparición de un gran número de teselas en la actualidad.
10. El bosque seco está perdiendo conectividad tanto a nivel global como zonal, que se produce por la pérdida de hábitats, la perdida de teselas y el cambio de usos de suelos hacia una matriz más hostil con menos permeabilidad.

11. Los remanentes de bosque son fundamentales en la conectividad, siendo utilizados como corredores o zonas de paso, y aunque las teselas pequeñas tienen menos conectividad que las grandes, también son fundamentales en la conectividad global.

## Conclusions

1. The dry forest presents a worse state of conservation than the evergreen forest in the Coastal region of Ecuador, with worse levels of connectivity, fragility and threat.
2. The dry forest presents a lower degree of protection than the evergreen forest of the Ecuadorian coast, where only 12.9% of the remaining dry forest is protected compared to 27.9% of the green forest.
3. As evergreen forests have received more attention from researchers and natural resource managers, conservation efforts should now be more focused on preserving the remaining dry ecosystems, protecting them as part of a more complex regional conservation.
4. The dry forest presents a high rate of deforestation in the last 28 years, losing 27.04% of the surface, remaining from 1990.
5. Fragmentation has increased since 1990 and the number of patches has decreased as a result of the reduction of the forest area, thus increasing the edge and disintegrating the forest into isolated fragments, and consequently making it more exposed to anthropic and natural changes.
6. There are many areas of dry forest with a high degree of fragmentation and at risk of disappearance where it is urgent to take measures to avoid the loss of the forest patches that are still preserved.
7. The dry forest is underrepresented in the natural areas of Ecuador, in addition the figure of the Protective Forest is not proving effective to prevent deforestation and fragmentation, compared to the areas called "Heritage of Natural Areas of the State" that are the result effective.
8. Among the different types of dry forest, semi-deciduous forest is the most affected by fragmentation, presenting a very low average patch size.
9. The dry forest presents a very high index of fragmentation by patch, having increased in value considerably since 1990, which represents a great risk of disappearance of a large number of tiles today.
10. The dry forest is losing connectivity both at a global and zonal level, which is produced by the loss of habitats, the loss of patches and the change of land uses towards a more hostile matrix with less permeability.

11. Forest remnants are fundamental in connectivity, being used as corridors or passageways, and although small patches have less connectivity than large ones, they are also fundamental in global connectivity.

## Agradecimientos

Durante estos tres años y medio han sido muchas personas las que han estado a mi lado, y sin ellas esta tesis no hubiera sido posible.

En primer lugar, a mi madre y mi padre, un pilar fundamental en mi vida, guiando y ayudando siempre, sin ellos no sería lo que soy.

A mis hermanos Antonio y Marga, que aunque estemos separados no estamos alejados, y al resto de mi familia.

A mis amigos, los de aquí y los de allá o los de allí y los de acá.

A Ecuador que tan bien me ha acogido y a los Jarre, más que amigos parte de mi familia.

A mi tutor Rafael Navarro, por guiarme y ayudarme, detrás de tantos mails también conocí a una persona magnífica.

A mi otro tutor Pepe, que más que mi tutor es un amigo, muchas gracias por tu paciencia y por tu ayuda.

Y como no, a la persona que más me ha ayudado y más me ha sufrido estos años, Lilian mi esposa. Muchas gracias por todo.

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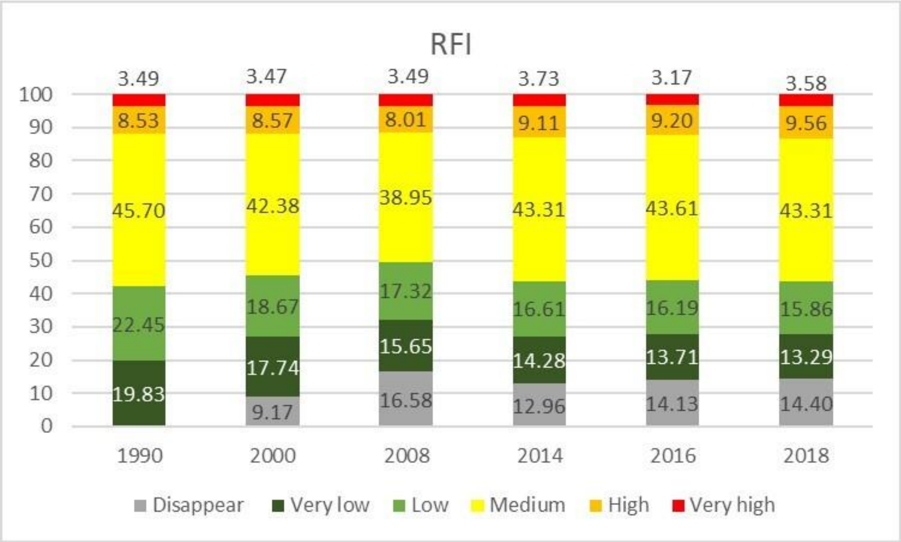
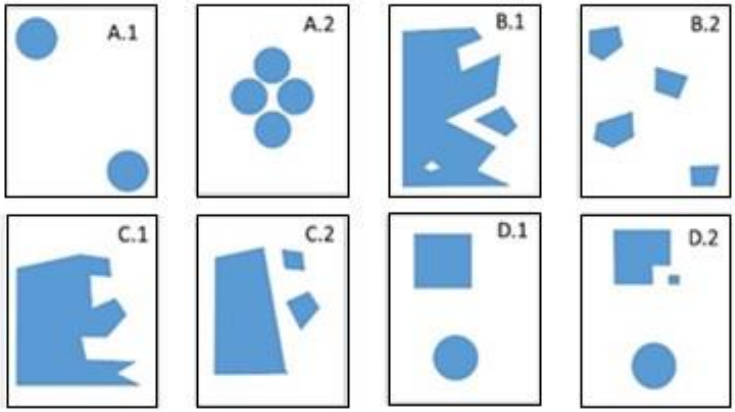
## Material suplementario.




**Supplementary material 1:** Percentage of RFI per tile between the years 1990 and 2018 for the following values: disappeared, very high, high, medium, low and very high.




**Supplementary material 2:** Possible states of fragmentation.

**Supplementary material 3:** Description of connectivity metrics analysed according to Clauzel et al. (2019).

| Flux (F) | Formula | Meaning |
|---|---|---|
| Global level | $S\#\mathrm{F} = \sum_{i=1}^{n}\sum_{\substack{j=1 \\ J\neq i}}^{n} a_j^{\beta}\ e^{-ad_{ij}}$ | For de entire graph: sum of potential dispersion from all patches |
| Local Level | $F_i = \sum_{\substack{j=1 \\ j\neq 1}}^{n} a_j^{\beta}\ e^{-ad_{ij}}$ | For the focal patch i : sum of capacity of patches other than i and weighted according to their minimum distance to the focal patch through the graph. This sum is an indicator of the potential dispersion from the patch i or, conversely to the patch i. |
| Reference | Urban and Keitt (2001); Saura and Torné, (2009); Foltête et al., (2012) | |
| Equivalent Probability (EC) | Formula | Meaning |
| Global level Component level | $EC = \sqrt{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i\ a_j\ e^{-ad_{ij}}}$ | Square root of the sum of products of capacity of all pairs of patches weigted by their interaction probability |
| Reference | Saura et at. (2011) | |
| Probability of connectivity (PC) | Formula | Meaning |
| Global level Component level | $PC = \frac{1}{A^2}\sum_{i=1}^{n}\sum_{j=1}^{n} a_i\ a_j\ e^{-ad_{ij}}$ | Sum of products of capacity of all pairs of patches weighted by their interaction probability, divided by the square of the area of the study zone. This ratio is the equivalent to the probability that two points randomly placed in the study area are connected. |

| | | |
|---|---|---|
| Reference | Saura and Pascual-Hortal (2007) | |
| Number of Components (NC) | Formula | Meaning |
| Global level | NC = nc | Number of components of the graph. |
| Reference | Urban and Keitt (2001) | |
| Integral Index of Connectivity (IIC) | Formula | Meaning |
| Global level | $ICC = \frac{1}{A^2}\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{a_i\, a_j}{1+nl_{ij}}$ | For the entire graph: product of patch capacities divided by the number of links between them, the sum is divided by the square of the area of the study zone. IIC is built like the PC index but using the inverse of a topological distance rather than a negative exponential function of the distance based on the link impedance |
| Reference | Pascual-Hortal and Saura (2006) | |
| Graph Diameter (GD) | Formula | Meaning |
| Global level | GD=$ma_{ij}xd_{ij}$ | Greatest distance between two patches of the graph. |
| Reference | Urban and Keitt (2001) | |
| Harary Index (H) | Formula | Meaning |
| Global level | $H = \frac{1}{2}\sum_{i=1}^{n}\sum_{j=1}^{n}\frac{1}{nl_{ij}}$ | Sum of the inverse of the number of links between all pairs of patches. |
| Reference | Ricotta et al. (2000) | |
| Connectivity Correlation (Ccor) | Formula | Meaning |

| | | |
|---|---|---|
| Local Level | $CCor_i = \frac{\lvert N_i \rvert^2}{\sum j \in Ni^{\lvert N_j \rvert}}$ | Ratio between the degree of the node i and the degree of its neighboring patches j |
| Reference | Minor and Urban (2008) | |
| Mean Size of the Components (MSC) | Formula | Meaning |
| Global level | $MSC = \frac{1}{nc} \sum_{k=1}^{nc} ac_k$ | Mean of the component capacities. |
| Size of the Largest Component (SLC) | SLC = max$\{ac_k\}$ | Largest capacity of components. |
| Class Coincidence Probability (CCP) | Formula | Meaning |
| Global level | $CCP = \sum_{k=1}^{nc} \left(\frac{ac_k}{\sum_l ac_l}\right)^2$ | Probability that two points randomly placed on the graph belong to the same component. |
| Reference | Pascual-Hortal and Saura (2006) | |
| Expected Cluster Size (ECS) | Formula | Meaning |
| Global | $ECS = \frac{1}{\sum_k ac_k} \sum_{k=1}^{nc} ac_k^2$ | Expected size of a component |
| Reference | O`Brien et al. (2006) | |
| | Formula | Meaning |
| Component level | $H = \sum_{i=1}^{n} \sum_{j=1}^{n} a_j^B e^{-ad_{ij}}$ | Sum of potential dispersions from all patches. |
| Reference | Urban and Keitt, (2001) Saura and Torné, (2009) Foltête et al., (2012) | |

Where:

N: number of patches.

nc: Number of componenets.

$n_k$: Number of pactches in de componente k

Ni: All parches close to the pacht i.

$a_i$: Capacity of the patch i (generally the surface area)

$ac_k$: Capacity of the component k (sum of the capacity of the patches composing k)

A: Area of the study zone

$d_{ij}$: Distance between the patches i and j (generally the least-cost distance between them).

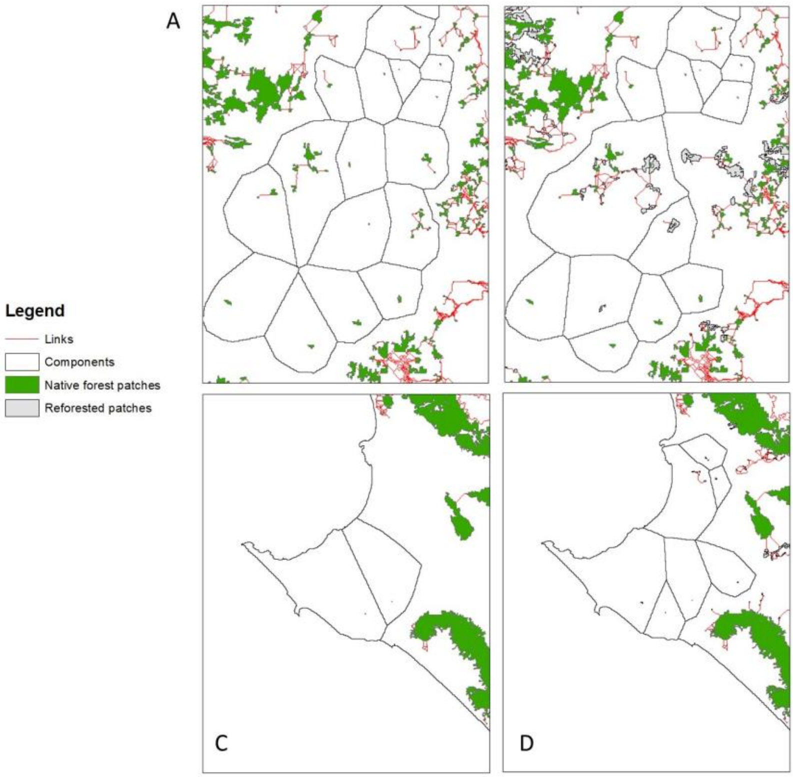
$e^{-\alpha dij}$: Probability of movement between the patches i and j

$\alpha$= Brake on movement distance

β= Exponent to weight more or less capacity

**Supplementary material 4:** Number of links distance in cost and meters. In the years 1990, 2000, 2008 and 2018 at distances 0.5, 5 and 10.

| | | | **Cost Distance** | | **Cost Distance (meters)** | |
|---|---|---|---|---|---|---|
| Distance | Year | Nº-links | Mean | S.E. | Mean | S.E. |
| 0,5 | 1990 | 6061 | 1,35 | 0,03 | 10964,61 | 1502,37 |
| | 2000 | 5805 | 1,12 | 0,03 | 7310,65 | 170,93 |
| | 2008 | 5389 | 1,07 | 0,03 | 6688,18 | 135,17 |
| | 2018 | 5284 | 1,07 | 0,03 | 5229,42 | 104,99 |
| 5 | 1990 | 17696 | 14,44 | 0,10 | 25954,10 | 1094,68 |
| | 2000 | 11865 | 10,99 | 0,12 | 7907,07 | 122,78 |
| | 2008 | 11394 | 11,69 | 0,13 | 9674,54 | 138,05 |
| | 2018 | 12053 | 12,20 | 0,13 | 6471,43 | 94,43 |
| 10 | 1990 | 21597 | 24,92 | 0,18 | 28592,95 | 930,26 |
| | 2000 | 14049 | 20,35 | 0,22 | 9880,40 | 140,16 |
| | 2008 | 13211 | 20,02 | 0,22 | 10293,42 | 131,13 |
| | 2018 | 14723 | 22,92 | 0,22 | 7953,78 | 101,49 |




**Supplementary material 5:** Effects of reforestation on the components. Figures A and C correspond to 2018 and Figures B and D correspond to 2018 with reforestations. In figures A and B it is observed how reforestation eliminates components and in figures C and D how reforestation creates new components.