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## Abstract

Mangroves are one of the most productive ecosystems in the world, sustaining millions of coastal livelihoods. However, their area of occurrence has been greatly reduced over the last century. In this study, we identify potential drivers of land use and land cover change adjacent to mangroves in the Pacific shorelines of Colombia, Panama and Costa Rica. We also evaluate the effectiveness of protected areas at halting mangrove deforestation between 2000 and 2012. Across all countries, agriculture was the most dominant land use type adjacent to mangroves, inside and outside protected areas. Results show that a combined total of 564 ha were lost, representing an average loss rate of only 0.02% per year. 75% of the total mangrove loss occurred in locations outside protected areas, with only 138 ha cleared from inside protected areas. Results suggest current conservation policies for mangrove protection in the study countries are effective at reducing deforestation and set a positive example for regions where mangroves are in decline.



**Key words:** agriculture, coastal development, aquaculture, land-cover change, land-use change, wetland.


## 1. Introduction

It is estimated that by 2050, global crop production must double to meet the demands of a rising global population (Tilman et al. 2011). Despite suggestions to prevent the increase of cultivated area, the global pattern of increasing agricultural field sizes is often driven by government incentives, demand for biofuels, and technology (White & Roy 2015). Worldwide rates of urban land expansion are higher than, or equal to, urban population growth rates (Seto et al. 2011). It is therefore expected that Land Use and Land Cover Change (LULCC) will increase as global population grows and developing countries become more affluent.

As LULCC intensifies, the effects of arable and urban land expansion may have significant and potentially irreversible consequences on ecosystem function and integrity (Foley et al. 2005). For instance, land conversion that removes primary forest has been shown to greatly reduce species diversity (Gibson et al. 2011). In the tropics, LULCC is associated with agricultural products for food, feed, and fuel (Gibbs et al. 2010; Blanco et al. 2012). Human reliance on natural environments is high in these regions and more than half of the new agricultural land created between 1980 and 2000 was via deforestation (Gibbs et al. 2010).

Mangrove forests are restricted to the interface between land and sea in tropical and subtropical latitudes. They are highly productive, provide a vast array of ecosystem services (Hogarth 2007), and diversify and sustain livelihoods for millions of people (UNEP 2014). Despite these widely appreciated values, mangrove cover is rapidly declining in different regions (Valiela et al. 2001; Alongi 2008; Richards & Friess 2015).

Estimates of global mangrove loss vary across regions and between methods used (Alongi 2002; Giri et al. 2011; López Angarita et al. 2016). The development of optical remote sensing technology has allowed for a better estimation of mangrove coverage, and for the exploration of LULCC dynamics (Manson et al. 2001; Dahdouh-Guebas et al. 2004). Recently, development of new radar technology sensitive to forest spatial structure has allowed for accurate estimates of mangrove deforestation rates (Lucas et al. 2007; Simard et al. 2008; White & Roy 2015; Hamilton 2013; Thomas et al. 2017). However to date there is little information on the proximate drivers of LULCC in mangrove forests or replacement land uses (Tilman et al. 2011; Richards & Friess 2015).

The Eastern Tropical Pacific (ETP) biogeographical region spans the continental shelf and oceanic islands of Southern Baja California to northern Perú (Briggs 1974), and supports a range of rich fisheries and exhibits many endemic species (Zapata & Robertson 2006; Fiedler & Talley 2006; Hogarth 2007). In terms of mangrove protection in the region, 58% of mangroves that occur on the Pacific coast of Costa Rica are inside protected areas, compared to 51% in Panama and 28% in Colombia (López Angarita et al. 2016).

Mangrove cover in the ETP has followed global trends of decline, with its greatest loss occurring between the 1960s and 1990s (Valiela et al. 2001; López Angarita et al. 2016). Since then, countries in the ETP have strengthened their conservation policies for mangroves, via creation of protected areas and laws regulating mangrove use (Lacerda et al. 1993; ANAMARAP 2013; López Angarita et al. 2016). To date there has been little or no assessment of the effectiveness of this protection.

In this study, we aim to identify the potential drivers of mangrove decline on the Pacific coasts of Costa Rica, Panama, and Colombia (Fig. 1) by mapping anthropogenic activities of LULCC in mangroves and performing analyses by country to compare trends within the region. Additionally, we determine the effectiveness of mangrove conservation policies by calculating rates of mangrove deforestation inside and outside protected areas, between 2000 and 2012.

## 2. Methods

### 2.1 Mangrove forest loss

To calculate the rate of mangrove deforestation we used the Global Forest Change dataset created by Hansen et al. (2013), which provides an index of annual deforestation between 2000 and 2015 per pixel (pixel size of 0.09 ha). These data are available up until 2015, but our study used the data between 2000 and 2012 only, to align with available land use data. We projected the Global Forest Change dataset for each country using UTM 18N/17N projected from WGS84. Mangrove areas were identified by overlaying the political limits of the studied countries with the global distribution of mangroves (Mangrove Forests of the World) in 2000 provided by Giri et al. (2011). Offshore islands were not included in our study. In a small section of the Colombian Pacific coast, we found a projection error causing misalignment of mangroves with the coastline in the Giri et al. (2011) global dataset, so we used Google satellite imagery and the mangrove distribution dataset for Colombia (IDEAM et al. 2007) to correct the error by manually fitting mangrove area polygons to the coastline. This resulted in a data layer of mangrove deforestation by year for the region of interest. This layer was used to calculate the percentage of mangroves deforested in the region for each country using number of pixels to estimate area. We obtained the rate of deforestation per year by dividing the percentage lost by the 12-years sampled (2000 not included and 2012 included). We used the same input layers (Global Forest Change and Mangrove Forests of the World) that Hamilton and Casey (2016) used, with a different methodological approximation, in their Global Database of Continuous Mangrove Forest Cover for the 21st Century, so we could compare our results with their deforestation rates.

## 2.2 Potential drivers of LULCC in mangrove areas

Ten different datasets of land cover with a resolution ≤ 30 $m^2$ were used to map the distribution of potential drivers of LULCC across the three countries (Table A1). We grouped potential drivers into three major classes: aquaculture, agriculture (includes cattle farms, oil palm plantations, and crops such as rice and fruits), and coastal development. Coastal development included towns and infrastructure such as ports and agricultural processing plants. Infrastructure was not analyzed as a separate class due to the few records associated with it. In this manuscript we used the term "potential drivers" to define land use types with potential to negatively impact mangroves as shown elsewhere in the literature (e.g. Hamilton 2013, Richards & Friess 2015, Thomas et al. 2017). However, it is important to clarify that in the studied period, these current land uses might not have been directly responsible for the loss of mangroves in our study area.

To quantify the spatial distribution of potential drivers of LULCC adjacent to mangroves, we used an overlaid 1 $km^2$ grid to divide the study area into sample units. A 1 $km^2$ grid was selected to simplify interpretation of results at scale, as our model was designed to be a tool for managers. Grid squares were ground-truthed in all countries between 2013 and 2015 to calibrate the interpretation of land use in the datasets. We chose areas to ground-truth based on the significant presence of anthropogenic activities close to mangroves. Of 9,812 1 $km^2$ grid cells placed over mangroves of the studied region, 401 were ground-truthed (see ground truthing section in Appendices and Table A2). We aimed to ground-truth areas that were highly representative of the mangrove-land use landscape in each country, but for logistical reasons, it was not possible to visit the south of Colombia. Therefore, our interpretations might be biased for this area. When errors were present in the land use classification of the datasets, we used Google Earth images calibrated with ground-truthing to re-classify the polygons.

To display the spatial patterns of potential drivers of LULCC adjacent to mangroves, we developed a cumulative model, where grid cells were scored based on the presence (1) or absence (0) of aquaculture, agriculture, and coastal development throughout the region. Scores were summed per cell to return a possible value between 0 and 3. The proportion of cells belonging to the different values of cumulative scores was calculated. Due to the complexity of measuring the cascading effects that potential drivers of LULCC have in mangroves on a regional scale, the same weighting was applied to all potential drivers. Cells were given a color scale according to the total score. The extent of different land use types and the impact score results were compared between countries using Chi-square tests.

**2.3 Protected areas**

We compared the extent of mangrove deforestation inside and outside protected areas between 2000 and 2012 by mapping the boundaries of protected areas present on the Pacific coast of the countries studied according to government datasets (Table A1, Table A3). We used the global forest change dataset (Hansen et al. 2013) to estimate figures of mangrove deforestation within protected areas. Protected areas established after the year 2000 were analyzed separately to accurately assess how deforestation had occurred inside and outside during the study period. Proportion of mangrove loss inside and outside protected areas was compared between countries using a Chi-square test. Finally, we compared the distribution of potential drivers of LULCC inside and outside protected areas by estimating the proportion of cells for each land use type. Cumulative score of mangroves was also compared inside and outside protected areas. Cells divided by protected area boundaries were classified as inside protected areas. Given the diverse types and protection levels of protected areas in the region, a comparative inside/outside approach was used. Geospatial analyses and calculations were performed in ArcGIS 10.3.1 and statistical tests in JMP version 13.

## 3. Results

### 3.1 Mangrove Forest Loss

The total area of mangroves on the Pacific coast of each country showed that Colombia has the largest area, followed by Panama then Costa Rica (Table 1). Over the reporting period, 564 hectares or 0.18% of the total mangrove area were lost in all countries combined (Table 1). In Costa Rica by 2012, 0.32% of mangroves present in 2000 had been deforested, with figures of 0.21% for Panama and 0.11% for Colombia.

Across all countries studied, the average annual deforestation rate was 0.02%. Temporal trends of mangrove deforestation showed that deforestation peaked in Panama and Costa Rica in 2008. An increasing trend of forest loss was observed in Colombia, whereas in Costa Rica deforestation has decreased with time (Fig. 2).

### 3.2 Potential drivers of LULCC in mangrove areas

According to the model, around 60% of cells across the three countries combined had no adjacent potential drivers of LULCC, whereas in 40% of cells, one or more land use types were present (Fig. 3). The proportion of cells with potential drivers was significantly different for three countries ($\chi^2(6, N = 9812) = 1132.36$, $p<0.0001$), as well as the extent of different land use types, including mangroves ($\chi^2(4, N = 4058) = 712.45$, $p<0.0001$). In Colombia, 73% of cells with mangroves had no potential drivers, whereas in ~26% of cells, one or more land use types were present (Table A4, Fig. A1). In this country, the most common land use was agriculture, present in 26% of cells within the grid, followed by coastal development (1.6%), while aquaculture did not occur (Fig. 4). In Panama, potential drivers were present in 53% of the cells with mangroves (Table A4). Agriculture was the most dominant land use, present in 30% of the cells, followed by coastal development (19%). 6% of cells were adjacent to aquaculture ponds (Fig. 4). In Costa Rica, 60% of mangrove cells had no proximate land use. Agriculture was the most common land use (28%), while aquaculture and coastal development had an equal representation of 9%.

### 3.3 Protected areas

Of the 31 protected areas mapped on the Pacific coast of Panama, 17 contained mangroves; in Colombia 6 of 9 contained mangroves; and for Costa Rica, 23 of 53 (Table A3). While figures for deforestation inside and outside protected areas varied between the three countries, significant differences between mangrove loss inside and outside were supported statistically for all countries ($\chi^2(2, N = 564) = 53.06$, $p<0.001$), with loss inside protected areas lower than outside in all cases (Fig. 5). Across all three countries 75% of deforestation occurred outside protected areas.

Ninety two percent of cells with mangroves inside protected areas in Colombia had no adjacent land use (Table A5). In Panama, agriculture was present in 50% of mangrove cells inside protected areas. In Costa Rica, 68% of mangrove cells inside protected areas had no proximate land use, 23% were surrounded by agriculture, and 8% by coastal development (Table A5).

## 4. Discussion

Knowledge of recent trends of mangrove deforestation is important in evaluating the effectiveness of current conservation policies. In a widely cited paper, Duke et al. (2007) raised concerns about the high rate of mangrove loss and estimated that the world could be without functional mangroves within 100 years. Fortunately, this scenario was based on extrapolated rates of mangrove deforestation from 1980s and 1990s and does not seem feasible now, as current research has shown that these trends appear not to have continued into the 21$^{st}$ century. This is certainly true in the ETP, where our calculated average annual loss rate of 0.02% between 2000 and 2012, confirm findings by Hamilton & Casey (2016), who report similar deforestation rates for the study countries (including the Caribbean coasts) (Table 1). Moreover, it is likely that our estimates are similar to country level estimates because the Pacific coast represent the majority of mangrove forest coverage for all study countries. Hamilton & Casey (2016) found a global rate of mangrove deforestation of 0.16%, with highest levels of deforestation in Southeast Asia, particularly in Indonesia with an annual rate of 0.26% - 0.66% per year. Our results support the low estimated deforestation rates in ETP countries, suggesting these low rates might be compensating for higher losses in other regions and thus, counterbalancing global mangrove loss rates.

In the ETP, annual loss rates calculated prior to 2000 were higher than 1% for the three countries included in this study, due to inclusion of figures of historic deforestation, when most mangroves were lost (Valiela et al. 2001; López Angarita et al. 2016). Recent trends of mangrove deforestation are consistent with historical trends, with Panama displaying the largest losses and Costa Rica the lowest (López Angarita et al. 2016). Despite Costa Rica having declared all mangroves no-take areas in 1998 (Valiela et al. 2001; RAMSAR 2015), it showed the highest annual rate of loss in this study. Temporal patterns of deforestation illustrate that Costa Rica is the only country showing a declining trend in recent years, while Panama and Colombia exhibit gradual increases in deforestation rates.

The rates of deforestation in the study countries are low compared to post-2000 deforestation in other regions and forest types (Valiela et al. 2001; Potapov et al. 2012; Nepstad et al. 2014; Richards & Friess 2015), which suggests that mangrove protection is effective. In contrast, Richards & Friess (2015) estimated that between 2000 and 2010, mangroves in South East Asia were being lost at an average rate of 0.18% per year. For the same time period, in the Democratic Republic of the Congo, average annual gross forest loss was 0.23% of forest area, across all forest types (Potapov et al. 2012). However, other studies have also reported significant declines in deforestation rates for other forests: In the Brazilian Amazon, forest loss declined by 70% between 2005 – 2013, passing from a ten year average of 19,500 $km^2$ per year, to 5843 $km^2$ (Nepstad et al. 2014). National deforestation rates, across all forest types, decreased after the year 2000 in Costa Rica (FAO 2013), and Colombia (Cabrera et al. 2011). Perhaps due to its isolation, the Pacific coast region of Colombia has had the least amount of forest loss nationwide (Cabrera et al. 2011). Despite overall deforestation in Panama having decreased compared to the 1990s, figures remain quite high for the region, with an annual rate of 0.41% between 2000 and 2008 (Mariscal 2012).

The cumulative model presented here offers perspective on the threats affecting mangroves at national scales, determined by proximity of potentially damaging activities adjacent to mangroves. Our analysis found that agriculture is now consistently the most dominant potential driver of LULCC adjacent to mangroves outside and inside protected areas. Ground-truthing showed that rice, watermelon, melon, sugar cane, and oil palm are the main crops grown, and that cattle farming also occurs. The intensity and extent of agriculture adjacent to mangroves varied among the countries examined, with small-scale agriculture prevalent on the Pacific coast of Colombia, in contrast to more productive agro-economic regions in Panama and Costa Rica (Pinto & Yee 2011). In Panama, rice and beef are the most commercially important agricultural commodities, and they are produced in rotation on the same land (Trejos et al. 2008). In Costa Rica, melon and oil palm have the highest yield per hectare, are planted in high-density monocultures, and receive large inputs of chemical pesticides and fertilizers (Bach 2007). Agriculture, particularly rice, has been shown to be an increasingly important driver of mangrove loss in other regions also (Richards & Friess 2015; Thomas et al. 2017).

Coastal development was a frequent potential driver of LULCC in Panama. In this country, mangroves adjacent to urban zones are commonly converted to areas of development for tourism and urban expansion (Benfield et al. 2005). On the contrary in Colombia, most of the Pacific coast population is scattered in small villages only accessible by boat (García 2010). This isolation from the rest of the country translates into good coverage of natural rainforest and mangroves (Sánchez-Paez et al. 1997). However, this lack of infrastructure and accessibility fosters poverty and drives a high dependence on natural resources (Leal 2000; Blanco et al. 2011).

Aquaculture, in particular extensive shrimp aquaculture, is widely claimed to be the most important driver of mangrove loss worldwide (Páez-Osuna 2001; Alongi 2002; Giri et al. 2008; 2011). Yet, in the three countries studied, it was far less important than agriculture, and not even observed as a land use in the Colombian Pacific. The low importance seen in this region may be explained by large scale abandonment of ponds (Bolanos 2012) following the outbreak of “white spot virus” in Central America, that affected both wild and cultured shrimp (Nunan et al. 2001).

Data used in this study to quantify forest loss were derived from Landsat images with a resolution of 30x30m. It is possible that the spatial resolution used underestimates mangrove deforestation by not detecting losses at smaller scales. Deforestation is likely to happen at the interface between forests and other land use types (Etter et al. 2006), which makes it hard to detect in satellite images (Heumann 2011; Thompson et al. 2013). For example, in the Gulf of Montijo, Panama, it was reported that the area of rice crops adjacent to mangroves has increased gradually (ANAM 2004) but the figures have not been quantified. Additionally, humid tropical regions such as the ETP are particularly challenging to map given the consistent cloud cover that affects the clarity of satellite images (Gibbs et al. 2010; Heumann 2011). The ETP also shows large transitions between mangroves and other forests types, in some instances related to river discharge diversions (Restrepo & Canterra 2013; Parra & Restrepo 2014) where deforestation is likely to become cryptic. Therefore, there is underlying potential for underestimation of the integrity of mangroves, as satellite images at the scale used in this study do not allow for the identification of small-scale forest clearing and the slow rate of habitat degradation at forest fringes and mangrove transitions. This bias may be even more significant for Colombia, where ground-truthing was limited, as cryptic ecological degradation due to selective logging, and clear cutting of basin mangroves is known to occur (Blanco et al. 2012). Nevertheless, our approach uses robust data of forest loss (Hansen et al. 2013) that has been applied to quantify forest cover change in many other regions worldwide (Potapov et al. 2012; Hansen et al. 2013; Richards & Friess 2015; Hamilton & Casey 2016). It is important to highlight however, that primary data sources, radar, high-resolution aerial images, and field ecological assessments are highly needed to improve the estimation of deforestation rates in the challenging environment of the ETP (Blanco-

Libreros & Estrada-Urrea 2015). As such, rates of loss are likely to be higher than the estimates presented here for the above reasons.

Our analysis showed that most mangrove deforestation (75%) occurred outside protected areas. In Colombia, this equated to ~8 Ha, whereas the figures for Costa Rica and Panama were 49 Ha and 81 Ha respectively. Other studies of tropical forests have also shown that the presence of protected areas significantly reduces deforestation inside them (Bruner 2001; Naughton-Treves et al. 2005; Andam et al. 2008; Gaveau et al. 2009; Miteva et al. 2015; Spracklen et al. 2015), and this has been shown specifically for mangrove forests (Miteva et al. 2015). Despite protected areas in this region being often undermanaged (López Angarita et al. 2014) and under increasing stress from human activities (Chape et al. 2005), our findings provide reassuring evidence that despite the diverse management approaches represented in this analysis (Table A3), protection has had an overall positive effect in reducing mangrove deforestation. All study countries recognize in law that any activity intended to exploit or modify mangroves requires prior government evaluation and permission (García 2010; Salas et al. 2012; ANAMARAP 2013; López Angarita et al. 2016).

In the ETP, communities have been reported to have increasing participation in the management of mangroves (Kaufmann 2012; Kothari et al. 2015; Vieira et al. 2016). Our results provide evidence that these initiatives combined with government input, are effective at reducing mangrove loss, and set a positive example for other regions where this ecosystem is being degraded. Our results are relevant to conservation and policy making in the region as they highlight the relative successes of formal protection of mangroves in the ETP. We recommend participatory land use planning at the community level to empower local stakeholders in mangrove protection.

## Supplementary material

Name and source of layers used for spatial analysis (Table A1); detail of ground-truthed area (Table A2); list of protected areas included in this study (Table A3), cumulative model results (Table A4); cumulative model results inside and outside protected areas (Table A5); frequency distributions of cumulative model results (Figure A1).

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# Tables

Table 1. Figures for mangrove deforestation between 2000 and 2012 on the Pacific coasts of Costa Rica, Panama and Colombia. For comparison purposes, mangrove deforestation rates found by Hamilton and Casey (2016) using the Global Database of Continuous Mangrove Forest Cover for the 21st Century (CMGFC-21), are also shown.

| Country | Mangrove area in the Pacific coast (Ha) | Mangrove area lost since 2000 (Ha) | % of total area lost 2000-2012 | Annual loss rate (% of total area per year) | National annual loss rate (CMGFC-21) | % of total area lost 2000-2012 (CMGFC-21) |
|---|---|---|---|---|---|---|
| Costa Rica | 37,266.5 | 120.4 | 0.32 | 0.03 | 0.029 | 0.35 |
| Panama | 135,955.8 | 287.7 | 0.21 | 0.02 | 0.025 | 0.29 |

| | | | | | | |
|---|---|---|---|---|---|---|
| Colombia | 141,271.6 | 156.0 | 0.11 | 0.01 | 0.011 | 0.137 |
| Total | 314,493.8 | 564.1 | 0.18 | 0.02 | NA | NA |

## Figure legends

Figure 1. Geographical extent of the study (red line), on the Pacific coasts of Costa Rica, Panama, and Colombia (shaded green).

Figure 2. Temporal trends in the deforestation of mangroves between 2000 and 2012 on the Pacific coasts of Panama, Colombia, and Costa Rica. The lower right panel shows cumulative forest loss for all countries.

Figure 3. Map of a cumulative model of potential drivers of land use and land cover change in mangroves on the Pacific coasts of Costa Rica, Panama and Colombia. Color grid represents 1 $km^2$ cells where the analysis was performed. Green cells represent mangroves without adjacent potential drivers of land use and land cover change, blue cells represent mangroves adjacent to one potential driver, yellow cells are mangroves adjacent to two land use types, and red cells are mangroves adjacent to three land use types. For visualization purposes only, red insets provide a magnified view of the selected area.

Figure 4. Distribution of agriculture, aquaculture, and coastal development next to mangroves on the Pacific coasts of Colombia, Panama, and Costa Rica. Percentages are calculated from a 1 $km^2$ grid placed over the mangroves of the Pacific coast of each country.

Figure 5. Total mangrove deforestation in hectares between 2000 and 2012 (black bars), highlighting deforestation inside (grey bars) and outside (white bars) protected areas on the Pacific coasts of Colombia, Costa Rica, and Panama.