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288

289 Abstract

290

Mangroves are one of the most productive ecosystems in the world, sustaining 291 millions of coastal livelihoods. However, their area of occurrence has been greatly 292 reduced over the last century. In this study, we identify potential drivers of land use 293 294 and land cover change adjacent to mangroves in the Pacific shorelines of Colombia, 295 Panama and Costa Rica. We also evaluate the effectiveness of protected areas at 296 halting mangrove deforestation between 2000 and 2012. Across all countries, 297 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, 298 representing an average loss rate of only 0.02% per year. 75% of the total mangrove 299 loss occurred in locations outside protected areas, with only 138 ha cleared from 300 301 inside protected areas. Results suggest current conservation policies for mangrove 302 protection in the study countries are effective at reducing deforestation and set a 303 positive example for regions where mangroves are in decline. 304 Key words: agriculture, coastal development, aquaculture, land-cover change, land-305 306 use change, wetland. 307

308 **1. Introduction**

309

310 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.

318

319 As LULCC intensifies, the effects of arable and urban land expansion may have significant and potentially irreversible consequences on ecosystem function and 320 integrity (Foley et al. 2005). For instance, land conversion that removes primary 321 forest has been shown to greatly reduce species diversity (Gibson et al. 2011). In the 322 323 tropics, LULCC is associated with agricultural products for food, feed, and fuel (Gibbs 324 et al. 2010; Blanco et al. 2012). Human reliance on natural environments is high in 325 these regions and more than half of the new agricultural land created between 1980 326 and 2000 was via deforestation (Gibbs et al. 2010).

327

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).

334

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).

345

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).

353

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.

360

361 In this study, we aim to identify the potential drivers of mangrove decline on the

362 Pacific coasts of Costa Rica, Panama, and Colombia (Fig. 1) by mapping anthropogenic

- 363 activities of LULCC in mangroves and performing analyses by country to compare
- 364 trends within the region. Additionally, we determine the effectiveness of mangrove

365 conservation policies by calculating rates of mangrove deforestation inside and
366 outside protected areas, between 2000 and 2012.

367

368 **2. Methods**

369 2.1 Mangrove forest loss

To calculate the rate of mangrove deforestation we used the Global Forest Change 370 dataset created by Hansen et al. (2013), which provides an index of annual 371 deforestation between 2000 and 2015 per pixel (pixel size of 0.09 ha). These data are 372 373 available up until 2015, but our study used the data between 2000 and 2012 only, to 374 align with available land use data. We projected the Global Forest Change dataset for 375 each country using UTM 18N/17N projected from WGS84. Mangrove areas were 376 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 377 et al. (2011). Offshore islands were not included in our study. In a small section of the 378 379 Colombian Pacific coast, we found a projection error causing misalignment of 380 mangroves with the coastline in the Giri et al. (2011) global dataset, so we used 381 Google satellite imagery and the mangrove distribution dataset for Colombia (IDEAM 382 et al. 2007) to correct the error by manually fitting mangrove area polygons to the 383 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 384 385 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 12-386 387 years sampled (2000 not included and 2012 included). We used the same input layers 388 (Global Forest Change and Mangrove Forests of the World) that Hamilton and Casey 389 (2016) used, with a different methodological approximation, in their Global Database 390 of Continuous Mangrove Forest Cover for the 21st Century, so we could compare our 391 results with their deforestation rates.

392

393 **2.2 Potential drivers of LULCC in mangrove areas**

394 Ten different datasets of land cover with a resolution $\leq 30 \text{ m}^2$ were used to map the distribution of potential drivers of LULCC across the three countries (Table A1). We 395 grouped potential drivers into three major classes: aquaculture, agriculture (includes 396 397 cattle farms, oil palm plantations, and crops such as rice and fruits), and coastal development. Coastal development included towns and infrastructure such as ports 398 399 and agricultural processing plants. Infrastructure was not analyzed as a separate 400 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 401 402 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 403 404 studied period, these current land uses might not have been directly responsible for the loss of mangroves in our study area. 405

406

To quantify the spatial distribution of potential drivers of LULCC adjacent to 407 mangroves, we used an overlaid 1 km² grid to divide the study area into sample units. 408 409 A 1 km² grid was selected to simplify interpretation of results at scale, as our model 410 was designed to be a tool for managers. Grid squares were ground-truthed in all 411 countries between 2013 and 2015 to calibrate the interpretation of land use in the 412 datasets. We chose areas to ground-truth based on the significant presence of 413 anthropogenic activities close to mangroves. Of 9,812 1 km² grid cells placed over 414 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 415 highly representative of the mangrove-land use landscape in each country, but for 416 417 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 418

419 use classification of the datasets, we used Google Earth images calibrated with420 ground-truthing to re-classify the polygons.

421

422 To display the spatial patterns of potential drivers of LULCC adjacent to mangroves, 423 we developed a cumulative model, where grid cells were scored based on the 424 presence (1) or absence (0) of aquaculture, agriculture, and coastal development 425 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 426 cumulative scores was calculated. Due to the complexity of measuring the cascading 427 428 effects that potential drivers of LULCC have in mangroves on a regional scale, the 429 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 430 431 score results were compared between countries using Chi-square tests.

432

433 2.3 Protected areas

We compared the extent of mangrove deforestation inside and outside protected 434 areas between 2000 and 2012 by mapping the boundaries of protected areas present 435 on the Pacific coast of the countries studied according to government datasets (Table 436 A1, Table A3). We used the global forest change dataset (Hansen et al. 2013) to 437 438 estimate figures of mangrove deforestation within protected areas. Protected areas 439 established after the year 2000 were analyzed separately to accurately assess how 440 deforestation had occurred inside and outside during the study period. Proportion of 441 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 442 LULCC inside and outside protected areas by estimating the proportion of cells for 443 each land use type. Cumulative score of mangroves was also compared inside and 444 445 outside protected areas. Cells divided by protected area boundaries were classified as

446 inside protected areas. Given the diverse types and protection levels of protected

447 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

449 version 13.

450

451 **3. Results**

452

453 3.1 Mangrove Forest Loss

454 The total area of mangroves on the Pacific coast of each country showed that

455 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

457 all countries combined (Table 1). In Costa Rica by 2012, 0.32% of mangroves present

458 in 2000 had been deforested, with figures of 0.21% for Panama and 0.11% for

459 Colombia.

460

461 Across all countries studied, the average annual deforestation rate was 0.02%.

462 Temporal trends of mangrove deforestation showed that deforestation peaked in

463 Panama and Costa Rica in 2008. An increasing trend of forest loss was observed in

464 Colombia, whereas in Costa Rica deforestation has decreased with time (Fig. 2).

465

466 **3.2 Potential drivers of LULCC in mangrove areas**

467 According to the model, around 60% of cells across the three countries combined had

468 no adjacent potential drivers of LULCC, whereas in 40% of cells, one or more land use

469 types were present (Fig. 3). The proportion of cells with potential drivers was

470 significantly different for three countries ($\chi^2(6, N = 9812) = 1132.36, p < 0.0001$), as

471 well as the extent of different land use types, including mangroves ($\chi^2(4, N = 4058) =$

472 712.45, *p*<0.0001). In Colombia, 73% of cells with mangroves had no potential

473 drivers, whereas in \sim 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 474 475 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% 476 477 of the cells with mangroves (Table A4). Agriculture was the most dominant land use, 478 present in 30% of the cells, followed by coastal development (19%). 6% of cells were 479 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 480 481 aquaculture and coastal development had an equal representation of 9%.

482

483 3.3 Protected areas

484 Of the 31 protected areas mapped on the Pacific coast of Panama, 17 contained

485 mangroves; in Colombia 6 of 9 contained mangroves; and for Costa Rica, 23 of 53

486 (Table A3). While figures for deforestation inside and outside protected areas varied

487 between the three countries, significant differences between mangrove loss inside

and outside were supported statistically for all countries ($\chi^2(2, N = 564) = 53.06$,

489 p<0.001), with loss inside protected areas lower than outside in all cases (Fig. 5).

490 Across all three countries 75% of deforestation occurred outside protected areas.

491

492 Ninety two percent of cells with mangroves inside protected areas in Colombia had

493 no adjacent land use (Table A5). In Panama, agriculture was present in 50% of

494 mangrove cells inside protected areas. In Costa Rica, 68% of mangrove cells inside

495 protected areas had no proximate land use, 23% were surrounded by agriculture, and

496 8% by coastal development (Table A5).

497

498 **4. Discussion**

500 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. 501 502 (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 503 scenario was based on extrapolated rates of mangrove deforestation from 1980s and 504 505 1990s and does not seem feasible now, as current research has shown that these 506 trends appear not to have continued into the 21st century. This is certainly true in the ETP, where our calculated average annual loss rate of 0.02% between 2000 and 2012. 507 confirm findings by Hamilton & Casey (2016), who report similar deforestation rates 508 for the study countries (including the Caribbean coasts) (Table 1). Moreover, it is 509 510 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. 511 512 Hamilton & Casey (2016) found a global rate of mangrove deforestation of 0.16%, 513 with highest levels of deforestation in Southeast Asia, particularly in Indonesia with 514 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 515 compensating for higher losses in other regions and thus, counterbalancing global 516 mangrove loss rates. 517

518

519 In the ETP, annual loss rates calculated prior to 2000 were higher than 1% for the 520 three countries included in this study, due to inclusion of figures of historic 521 deforestation, when most mangroves were lost (Valiela et al. 2001; López Angarita et 522 al. 2016). Recent trends of mangrove deforestation are consistent with historical trends, with Panama displaying the largest losses and Costa Rica the lowest (López 523 Angarita et al. 2016). Despite Costa Rica having declared all mangroves no-take areas 524 in 1998 (Valiela et al. 2001; RAMSAR 2015), it showed the highest annual rate of loss 525 in this study. Temporal patterns of deforestation illustrate that Costa Rica is the only 526

527 country showing a declining trend in recent years, while Panama and Colombia528 exhibit gradual increases in deforestation rates.

529

The rates of deforestation in the study countries are low compared to post-2000 530 deforestation in other regions and forest types (Valiela et al. 2001; Potapov et al. 531 532 2012; Nepstad et al. 2014; Richards & Friess 2015), which suggests that mangrove protection is effective. In contrast, Richards & Friess (2015) estimated that between 533 2000 and 2010, mangroves in South East Asia were being lost at an average rate of 534 535 0.18% per year. For the same time period, in the Democratic Republic of the Congo, 536 average annual gross forest loss was 0.23% of forest area, across all forest types 537 (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 538 539 by 70% between 2005 – 2013, passing from a ten year average of 19,500 km² per year, to 5843 km² (Nepstad et al. 2014). National deforestation rates, across all forest 540 541 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 542 has had the least amount of forest loss nationwide (Cabrera et al. 2011). Despite 543 544 overall deforestation in Panama having decreased compared to the 1990s, figures 545 remain quite high for the region, with an annual rate of 0.41% between 2000 and 546 2008 (Mariscal 2012).

547

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

554 also occurs. The intensity and extent of agriculture adjacent to mangroves varied among the countries examined, with small-scale agriculture prevalent on the Pacific 555 556 coast of Colombia, in contrast to more productive agro-economic regions in Panama 557 and Costa Rica (Pinto & Yee 2011). In Panama, rice and beef are the most 558 commercially important agricultural commodities, and they are produced in rotation 559 on the same land (Trejos et al. 2008). In Costa Rica, melon and oil palm have the 560 highest yield per hectare, are planted in high-density monocultures, and receive large inputs of chemical pesticides and fertilizers (Bach 2007). Agriculture, particularly 561 562 rice, has been shown to be an increasingly important driver of mangrove loss in other regions also (Richards & Friess 2015; Thomas et al. 2017). 563

564

Coastal development was a frequent potential driver of LULCC in Panama. In this 565 566 country, mangroves adjacent to urban zones are commonly converted to areas of 567 development for tourism and urban expansion (Benfield et al. 2005). On the contrary 568 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 569 into good coverage of natural rainforest and mangroves (Sánchez-Paez et al. 1997). 570 However, this lack of infrastructure and accessibility fosters poverty and drives a 571 high dependence on natural resources (Leal 2000; Blanco et al. 2011). 572

573

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 582 583 a resolution of 30x30m. It is possible that the spatial resolution used underestimates mangrove deforestation by not detecting losses at smaller scales. Deforestation is 584 likely to happen at the interface between forests and other land use types (Etter et al. 585 586 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 587 588 of rice crops adjacent to mangroves has increased gradually (ANAM 2004) but the 589 figures have not been quantified. Additionally, humid tropical regions such as the ETP 590 are particularly challenging to map given the consistent cloud cover that affects the 591 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 592 593 related to river discharge diversions (Restrepo & Canterra 2013; Parra & Restrepo 594 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 595 scale used in this study do not allow for the identification of small-scale forest 596 clearing and the slow rate of habitat degradation at forest fringes and mangrove 597 598 transitions. This bias may be even more significant for Colombia, where ground-599 truthing was limited, as cryptic ecological degradation due to selective logging, and 600 clear cutting of basin mangroves is known to occur (Blanco et al. 2012). Nevertheless, 601 our approach uses robust data of forest loss (Hansen et al. 2013) that has been 602 applied to quantify forest cover change in many other regions worldwide (Potapov et 603 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 604 aerial images, and field ecological assessments are highly needed to improve the 605 606 estimation of deforestation rates in the challenging environment of the ETP (Blanco-

607 Libreros & Estrada-Urrea 2015). As such, rates of loss are likely to be higher than the608 estimates presented here for the above reasons.

609

Our analysis showed that most mangrove deforestation (75%) occurred outside 610 protected areas. In Colombia, this equated to \sim 8 Ha, whereas the figures for Costa 611 612 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 613 deforestation inside them (Bruner 2001; Naughton-Treves et al. 2005; Andam et al. 614 2008; Gaveau et al. 2009; Miteva et al. 2015; Spracklen et al. 2015), and this has been 615 shown specifically for mangrove forests (Miteva et al. 2015). Despite protected areas 616 617 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 618 619 reassuring evidence that despite the diverse management approaches represented in this analysis (Table A3), protection has had an overall positive effect in reducing 620 621 mangrove deforestation. All study countries recognize in law that any activity intended to exploit or modify mangroves requires prior government evaluation and 622 permission (García 2010; Salas et al. 2012; ANAMARAP 2013; López Angarita et al. 623 2016). 624 625

626 In the ETP, communities have been reported to have increasing participation in the 627 management of mangroves (Kaufmann 2012; Kothari et al. 2015; Vieira et al. 2016). 628 Our results provide evidence that these initiatives combined with government input, 629 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 630 policy making in the region as they highlight the relative successes of formal 631 protection of mangroves in the ETP. We recommend participatory land use planning 632 at the community level to empower local stakeholders in mangrove protection. 633

634

635 Supplementary material

Name and source of layers used for spatial analysis (Table A1); detail of groundtruthed 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).

641

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840 Tables

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

Country	Mangrove area	Mangrove area	% of total	Annual loss rate	National annual	% of total area
	in the Pacific	lost since 2000	area lost	(% of total area	loss rate	lost 2000-2012
	coast (Ha)	(Ha)	2000-2012	per year)	(CMGFC-21)	(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

846 **Figure legends**

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

Figure 2. Temporal trends in the deforestation of mangroves between 2000 and 2012

850 on the Pacific coasts of Panama, Colombia, and Costa Rica. The lower right panel

- 851 shows cumulative forest loss for all countries.
- Figure 3. Map of a cumulative model of potential drivers of land use and land cover

853 change in mangroves on the Pacific coasts of Costa Rica, Panama and Colombia. Color

grid represents 1 km² cells where the analysis was performed. Green cells represent

855 mangroves without adjacent potential drivers of land use and land cover change, blue

856 cells represent mangroves adjacent to one potential driver, yellow cells are

857 mangroves adjacent to two land use types, and red cells are mangroves adjacent to

858 three land use types. For visualization purposes only, red insets provide a magnified

859 view of the selected area.

860 Figure 4. Distribution of agriculture, aquaculture, and coastal development next to

861 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

863 each country.

Figure 5. Total mangrove deforestation in hectares between 2000 and 2012 (black

865 bars), highlighting deforestation inside (grey bars) and outside (white bars)

866 protected areas on the Pacific coasts of Colombia, Costa Rica, and Panama.