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286 **Manuscript title:** Land use patterns and influences of protected areas on mangroves
287 of the Eastern Tropical Pacific

288

289 **Abstract**

290

291 Mangroves are one of the most productive ecosystems in the world, sustaining
292 millions of coastal livelihoods. However, their area of occurrence has been greatly
293 reduced over the last century. In this study, we identify potential drivers of land use
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
298 outside protected areas. Results show that a combined total of 564 ha were lost,
299 representing an average loss rate of only 0.02% per year. 75% of the total mangrove
300 loss occurred in locations outside protected areas, with only 138 ha cleared from
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

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

307

308 **1. Introduction**

309

310 It is estimated that by 2050, global crop production must double to meet the
311 demands of a rising global population (Tilman et al. 2011). Despite suggestions to

312 prevent the increase of cultivated area, the global pattern of increasing agricultural
313 field sizes is often driven by government incentives, demand for biofuels, and
314 technology (White & Roy 2015). Worldwide rates of urban land expansion are higher
315 than, or equal to, urban population growth rates (Seto et al. 2011). It is therefore
316 expected that Land Use and Land Cover Change (LULCC) will increase as global
317 population grows and developing countries become more affluent.

318

319 As LULCC intensifies, the effects of arable and urban land expansion may have
320 significant and potentially irreversible consequences on ecosystem function and
321 integrity (Foley et al. 2005). For instance, land conversion that removes primary
322 forest has been shown to greatly reduce species diversity (Gibson et al. 2011). In the
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

328 Mangrove forests are restricted to the interface between land and sea in tropical and
329 subtropical latitudes. They are highly productive, provide a vast array of ecosystem
330 services (Hogarth 2007), and diversify and sustain livelihoods for millions of people
331 (UNEP 2014). Despite these widely appreciated values, mangrove cover is rapidly
332 declining in different regions (Valiela et al. 2001; Alongi 2008; Richards & Friess
333 2015).

334

335 Estimates of global mangrove loss vary across regions and between methods used
336 (Alongi 2002; Giri et al. 2011; López Angarita et al. 2016). The development of optical
337 remote sensing technology has allowed for a better estimation of mangrove coverage,
338 and for the exploration of LULCC dynamics (Manson et al. 2001; Dahdouh-Guebas et

339 al. 2004). Recently, development of new radar technology sensitive to forest spatial
340 structure has allowed for accurate estimates of mangrove deforestation rates (Lucas
341 et al. 2007; Simard et al. 2008; White & Roy 2015; Hamilton 2013; Thomas et al.
342 2017). However to date there is little information on the proximate drivers of LULCC
343 in mangrove forests or replacement land uses (Tilman et al. 2011; Richards & Friess
344 2015).

345

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

353

354 Mangrove cover in the ETP has followed global trends of decline, with its greatest loss
355 occurring between the 1960s and 1990s (Valiela et al. 2001; López Angarita et al.
356 2016). Since then, countries in the ETP have strengthened their conservation policies
357 for mangroves, via creation of protected areas and laws regulating mangrove use
358 (Lacerda et al. 1993; ANAMARAP 2013; López Angarita et al. 2016). To date there has
359 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**

370 To calculate the rate of mangrove deforestation we used the Global Forest Change
371 dataset created by Hansen et al. (2013), which provides an index of annual
372 deforestation between 2000 and 2015 per pixel (pixel size of 0.09 ha). These data are
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
377 distribution of mangroves (Mangrove Forests of the World) in 2000 provided by Giri
378 et al. (2011). Offshore islands were not included in our study. In a small section of the
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
384 region of interest. This layer was used to calculate the percentage of mangroves
385 deforested in the region for each country using number of pixels to estimate area. We
386 obtained the rate of deforestation per year by dividing the percentage lost by the 12-
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
395 distribution of potential drivers of LULCC across the three countries (Table A1). We
396 grouped potential drivers into three major classes: aquaculture, agriculture (includes
397 cattle farms, oil palm plantations, and crops such as rice and fruits), and coastal
398 development. Coastal development included towns and infrastructure such as ports
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
401 “potential drivers” to define land use types with potential to negatively impact
402 mangroves as shown elsewhere in the literature (e.g. Hamilton 2013, Richards &
403 Friess 2015, Thomas et al. 2017). However, it is important to clarify that in the
404 studied period, these current land uses might not have been directly responsible for
405 the loss of mangroves in our study area.

406

407 To quantify the spatial distribution of potential drivers of LULCC adjacent to
408 mangroves, we used an overlaid 1 km^2 grid to divide the study area into sample units.
409 A 1 km^2 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 km^2 grid cells placed over
414 mangroves of the studied region, 401 were ground-truthed (see ground truthing
415 section in Appendices and Table A2). We aimed to ground-truth areas that were
416 highly representative of the mangrove-land use landscape in each country, but for
417 logistical reasons, it was not possible to visit the south of Colombia. Therefore, our
418 interpretations might be biased for this area. When errors were present in the land

419 use classification of the datasets, we used Google Earth images calibrated with
420 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
426 between 0 and 3. The proportion of cells belonging to the different values of
427 cumulative scores was calculated. Due to the complexity of measuring the cascading
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
430 according to the total score. The extent of different land use types and the impact
431 score results were compared between countries using Chi-square tests.

432

433 **2.3 Protected areas**

434 We compared the extent of mangrove deforestation inside and outside protected
435 areas between 2000 and 2012 by mapping the boundaries of protected areas present
436 on the Pacific coast of the countries studied according to government datasets (Table
437 A1, Table A3). We used the global forest change dataset (Hansen et al. 2013) to
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
442 using a Chi-square test. Finally, we compared the distribution of potential drivers of
443 LULCC inside and outside protected areas by estimating the proportion of cells for
444 each land use type. Cumulative score of mangroves was also compared inside and
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
448 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
456 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 ~26% of cells, one or more land use types were present (Table
474 A4, Fig. A1). In this country, the most common land use was agriculture, present in
475 26% of cells within the grid, followed by coastal development (1.6%), while
476 aquaculture did not occur (Fig. 4). In Panama, potential drivers were present in 53%
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
480 proximate land use. Agriculture was the most common land use (28%), while
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
488 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**

499

500 Knowledge of recent trends of mangrove deforestation is important in evaluating the
501 effectiveness of current conservation policies. In a widely cited paper, Duke et al.
502 (2007) raised concerns about the high rate of mangrove loss and estimated that the
503 world could be without functional mangroves within 100 years. Fortunately, this
504 scenario was based on extrapolated rates of mangrove deforestation from 1980s and
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
507 ETP, where our calculated average annual loss rate of 0.02% between 2000 and 2012,
508 confirm findings by Hamilton & Casey (2016), who report similar deforestation rates
509 for the study countries (including the Caribbean coasts) (Table 1). Moreover, it is
510 likely that our estimates are similar to country level estimates because the Pacific
511 coast represent the majority of mangrove forest coverage for all study countries.
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
515 deforestation rates in ETP countries, suggesting these low rates might be
516 compensating for higher losses in other regions and thus, counterbalancing global
517 mangrove loss rates.

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
523 trends, with Panama displaying the largest losses and Costa Rica the lowest (López
524 Angarita et al. 2016). Despite Costa Rica having declared all mangroves no-take areas
525 in 1998 (Valiela et al. 2001; RAMSAR 2015), it showed the highest annual rate of loss
526 in this study. Temporal patterns of deforestation illustrate that Costa Rica is the only

527 country showing a declining trend in recent years, while Panama and Colombia

528 exhibit gradual increases in deforestation rates.

529

530 The rates of deforestation in the study countries are low compared to post-2000

531 deforestation in other regions and forest types (Valiela et al. 2001; Potapov et al.

532 2012; Nepstad et al. 2014; Richards & Friess 2015), which suggests that mangrove

533 protection is effective. In contrast, Richards & Friess (2015) estimated that between

534 2000 and 2010, mangroves in South East Asia were being lost at an average rate of

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

538 in deforestation rates for other forests: In the Brazilian Amazon, forest loss declined

539 by 70% between 2005 – 2013, passing from a ten year average of 19,500 km² per

540 year, to 5843 km² (Nepstad et al. 2014). National deforestation rates, across all forest

541 types, decreased after the year 2000 in Costa Rica (FAO 2013), and Colombia

542 (Cabrera et al. 2011). Perhaps due to its isolation, the Pacific coast region of Colombia

543 has had the least amount of forest loss nationwide (Cabrera et al. 2011). Despite

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

548 The cumulative model presented here offers perspective on the threats affecting

549 mangroves at national scales, determined by proximity of potentially damaging

550 activities adjacent to mangroves. Our analysis found that agriculture is now

551 consistently the most dominant potential driver of LULCC adjacent to mangroves

552 outside and inside protected areas. Ground-truthing showed that rice, watermelon,

553 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
555 among the countries examined, with small-scale agriculture prevalent on the Pacific
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
561 inputs of chemical pesticides and fertilizers (Bach 2007). Agriculture, particularly
562 rice, has been shown to be an increasingly important driver of mangrove loss in other
563 regions also (Richards & Friess 2015; Thomas et al. 2017).

564

565 Coastal development was a frequent potential driver of LULCC in Panama. In this
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
569 accessible by boat (García 2010). This isolation from the rest of the country translates
570 into good coverage of natural rainforest and mangroves (Sánchez-Paez et al. 1997).
571 However, this lack of infrastructure and accessibility fosters poverty and drives a
572 high dependence on natural resources (Leal 2000; Blanco et al. 2011).

573

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

581

582 Data used in this study to quantify forest loss were derived from Landsat images with
583 a resolution of 30x30m. It is possible that the spatial resolution used underestimates
584 mangrove deforestation by not detecting losses at smaller scales. Deforestation is
585 likely to happen at the interface between forests and other land use types (Etter et al.
586 2006), which makes it hard to detect in satellite images (Heumann 2011; Thompson
587 et al. 2013). For example, in the Gulf of Montijo, Panama, it was reported that the area
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
592 large transitions between mangroves and other forests types, in some instances
593 related to river discharge diversions (Restrepo & Canterra 2013; Parra & Restrepo
594 2014) where deforestation is likely to become cryptic. Therefore, there is underlying
595 potential for underestimation of the integrity of mangroves, as satellite images at the
596 scale used in this study do not allow for the identification of small-scale forest
597 clearing and the slow rate of habitat degradation at forest fringes and mangrove
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
604 important to highlight however, that primary data sources, radar, high-resolution
605 aerial images, and field ecological assessments are highly needed to improve the
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 the
608 estimates presented here for the above reasons.

609

610 Our analysis showed that most mangrove deforestation (75%) occurred outside
611 protected areas. In Colombia, this equated to ~8 Ha, whereas the figures for Costa
612 Rica and Panama were 49 Ha and 81 Ha respectively. Other studies of tropical forests
613 have also shown that the presence of protected areas significantly reduces
614 deforestation inside them (Bruner 2001; Naughton-Treves et al. 2005; Andam et al.
615 2008; Gaveau et al. 2009; Miteva et al. 2015; Spracklen et al. 2015), and this has been
616 shown specifically for mangrove forests (Miteva et al. 2015). Despite protected areas
617 in this region being often undermanaged (López Angarita et al. 2014) and under
618 increasing stress from human activities (Chape et al. 2005), our findings provide
619 reassuring evidence that despite the diverse management approaches represented in
620 this analysis (Table A3), protection has had an overall positive effect in reducing
621 mangrove deforestation. All study countries recognize in law that any activity
622 intended to exploit or modify mangroves requires prior government evaluation and
623 permission (García 2010; Salas et al. 2012; ANAMARAP 2013; López Angarita et al.
624 2016).

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
630 where this ecosystem is being degraded. Our results are relevant to conservation and
631 policy making in the region as they highlight the relative successes of formal
632 protection of mangroves in the ETP. We recommend participatory land use planning
633 at the community level to empower local stakeholders in mangrove protection.

634

635 **Supplementary material**

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

641

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839

840 **Tables**

841 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
843 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 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

845

846 **Figure legends**

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

849 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.

852 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
854 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
862 are calculated from a 1 km² grid placed over the mangroves of the Pacific coast of
863 each country.

864 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.