

ASSESSING THE ROLE OF COMMERCIAL AQUACULTURE IN DISPLACING MANGROVE FOREST

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ABSTRACT

To fill a gap in the marine science literature, I calculated the amount of mangrove deforestation in tropical estuaries that is attributable to commercial aquaculture. The eight countries analyzed were Indonesia, Brazil, India, Bangladesh, China, Thailand, Vietnam, and Ecuador. Together these countries contain approximately 36% of the world's remaining mangrove forest. One precommercial aquaculture remote sensing survey and one current remote sensing survey were undertaken in the major mangrove holding estuaries in each nation. The time period of the analysis varied by country based on the first arrival of large-scale commercial aquaculture, with the majority of initial surveys occurring in the early to mid-1970s and all the current surveys occurring post-2004. The surveys classified the land cover of 110,557 randomly located estuarine point locations and examined the land-cover change over time for each of these discreet locations. Among the largest and most representative account of global mangrove to aquaculture conversion, this study revealed that mangrove forests have lost 51.9% of their aerial extent during the analysis period. Of the mangrove loss during this period, I estimate that commercial aquaculture accounted for 28% of total mangrove loss across all nations resulting in approximately 544,000 ha of mangrove forest converted to aquaculture. There were significant differences in mangrove loss and mangrove to aquaculture conversion from nation to nation and even within nations.

Mangrove forests provide numerous marine based services to the local communities that reside in and around tropical estuaries and provide equally important marine functions of global importance. On the macroscale, the role of mangrove forests in supporting wild catch, particularly finfish, is well documented (Costanza et al. 1997, Koenig et al. 2007, Lewis and Gilmore 2007, Naylor 2011). Mangrove forests are also often a primary habitat for other important seafood species such as crab, shrimp, clam, sea snail, and eel populations. During the last decade, the importance of mangroves in mitigating global climate change has come to the forefront with the realization that mangrove forests play an important role in carbon sequestration and the mitigation of atmospheric carbon (Ong 2002, Siikamäki et al. 2012). For example, it is estimated that 11% of the terrestrial origin carbon in the ocean and 15% of the total carbon accumulating in marine sediments are exported from mangrove forests (Jennerjahn and Ittekkot 2002) despite a coverage of <0.1% of the Earth's surface (Giri et al. 2011).

In addition to the global level, mangrove forests also provide goods and services at the regional scale. For example, mangroves provide renewable timber resources including firewood, charcoal, and timber for coastal residents' construction of boats and homes (Macnae 1968, Blanchard and Prado 1995, Armitage 2002). Aside from timber and seafood, mangrove forests are often exploited for wild honey, edible plants (Warne 2007), medicinal plants, and sugar derived from the nypa palm (Armitage 2002). For all of these reasons mangrove forests have been called an "entrepreneur's

dream," as they utilize seawater and other renewable energy sources to produce raw materials, which are then used by traditional communities (Tomlinson 1986) and the wider world.

Despite their ecologic and environmental importance, mangrove forests are under threat from numerous anthropogenic actions including clear cutting, land-use change (Blasco et al. 2001), and climate change induced sea-level rise (Gilman et al. 2008). It is often argued that mangrove forests suffer one of the highest rates of decline of any natural environment (Spalding et al. 1997, 2010, Valiela et al. 2001, Barbier and Cox 2004, Siikamäki et al. 2012), and that mangrove conversion into aquaculture ponds is one of the primary reasons for such high rates of deforestation (Aksornkoae et al. 1993, Spalding et al. 1997, Barbier and Cox 2004). Indeed, in certain countries in Latin America it appears that the expansion of commercial aquaculture can almost wholly explain the post-1970 deforestation of mangrove forests (Hamilton and Stankwitz 2012).

Counter arguments question the global level of mangrove to commercial aquaculture transition and even when conversion occurs, question the assumed economic loss incurred during the transition. These arguments often point to the fact that only approximately 10% of worldwide mangrove deforestation can be attributed to aquaculture (Menasveta 1997, Boyd and Clay 1998, Diana 2009) and even when mangrove to commercial aquaculture conversion occurs the annual return from commercial aquaculture may substantially outweigh the economic loss to fisheries. For example, when examining five coastal zones in the Gulf of Thailand, aquaculture economic returns were calculated to be approximately 10 times the magnitude of economic losses incurred by fisheries after mangrove to aquaculture conversion (Barbier 2003). Indeed, it is stated the entire relationship between declines in offshore fisheries and mangrove loss is dogma as studies that equate mangrove loss to fisheries loss are plagued by problems of autocorrelation, both spatially and temporally (Blaber 2007). That is, substantial overfishing occurred in the same locations and over the same time period as mangrove deforestation and hence the relationship between mangrove deforestation and fisheries loss is difficult to separate from commercial fisheries exploitation.

Unfortunately, much of the information on mangrove to shrimp aquaculture conversion is based either on country- or on regionally-specific case studies or is often anecdotal in nature, as calculating the actual conversion rate of mangrove to commercial aquaculture requires knowledge of a historic land cover that no longer exists. One major limitation of the mostly regional case studies in the literature is that the area analyzed usually appears to be selected with a priori knowledge of mangrove loss or mangrove to commercial aquaculture conversion. For example, when conducting a case study of mangrove loss a researcher is unlikely to select a region of limited or no mangrove loss, the same applies to studies of mangrove to commercial aquaculture conversion. For these reasons, regional case studies cannot be extrapolated into meaningful international, or even national, mangrove to commercial aquaculture conversion figures as they are often unrepresentative of the whole and based on small regional samples where conversion is already known to occur.

To accurately estimate global mangrove to aquaculture conversion levels, an international assessment of mangrove forests that have been converted to commercial aquaculture is needed, with representative samples of estuaries beyond those known a priori to have mangrove loss and a mangrove to commercial aquaculture transition.

The analysis must cover the major period of commercial aquaculture expansion from approximately 1970 to present and include enough nations and estuaries to be considered representative. Numerous analyses exist of the changes to mangrove forests over time but these were mostly conducted at the national (Conchedda et al. 2011, Hamilton 2011) or subnational (Terchunian et al. 1986, Satyanarayana et al. 2011) level. Although these studies provide important information on national and subnational trends, these cannot shed light on international patterns of mangrove displacement by aquaculture due to their small sample sizes and region- or nation-specific findings. Additionally, the numerous worldwide snapshots of current mangrove cover often report varying and conflicting levels of deforestation (Saenger et al. 1983, Groombridge 1992, Aksornkoae et al. 1993, Fisher and Spalding 1993, Aizpuru et al. 2000, FAO 2007, Giri et al. 2011) and they rarely attempt to calculate mangrove loss due to commercial aquaculture expansion.

The most robust spatiotemporal analyses of mangrove holdings are found in the two World Mangrove atlases (Spalding et al. 1997, 2010). These atlases undertake a meticulous accounting of mangrove levels internationally, by country, and over multiple time-frames. Unfortunately, the atlases also lack consistent and systematic accounting across countries of the land cover that is displacing mangrove forest at the estuarine and national level. The FAO keeps statistics on aquaculture expansion by monitoring commercial aquaculture output (FAO 2006, 2009, 2010b), but these reports do not systematically account for the land-cover transitions occurring due to aquaculture expansion. Giri et al. (2008) undertook a remotely sensed multi-nation analysis of mangrove displacement by aquaculture but only in areas around the Bay of Bengal, those directly impacted by the 2004 Indian Ocean earthquake and tsunami.

Ecuador and Thailand are used here as examples to provide insights into the difficulty of ascertaining reliable national levels of mangrove to aquaculture conversion from the academic literature and other sources. Within Ecuador, all of the remote sensing-based mangrove deforestation analyses either do not report mangrove to shrimp conversion numbers or are lacking spatial coverage. For example, Terchunian et al. (1986) examined aquaculture expansion and mangrove losses in one 1:25,000 quadrangle around Machala between 1966 and 1982. They documented mangrove losses and aquaculture expansion but did not report mangrove to aquaculture conversion levels. Additionally their analysis covers <1% of all Ecuadorian mangrove regions. Hamilton (2011) conducted an analysis in all of the northern Ecuadorian estuaries but was missing at least 60% of Ecuador's forests in the south, and despite reporting mangrove losses and aquaculture expansion, did not report actual conversion levels. Outside of the academic literature the Ecuadorian government produced a report on the loss of mangroves and expansion of aquaculture for all regions, but their report does not describe their full methodology nor does it report conversion levels (Centro de Levantamientos Integrados de Recursos Naturales por Sensores Remotos 2007).

Thailand differs from Ecuador in that many differing mangrove to aquaculture conversion levels are reported in the literature, but they have differing spatiotemporal resolutions producing a remarkable range of results. For example, the reported range of mangrove to aquaculture conversion values varies from 12% to 25% between 1961 and 1993 (Dierberg and Kiattisimkul 1996) to 64% between 1961 and 1986 (Aksornkoae et al. 1993). Aside from these two extremes, mangrove to aquaculture

conversion rates of 50%, 43%, 38%, 25%, and 21% are also reported for varying time periods and locations within Thailand (Dierberg and Kiattisimkul 1996). This pattern of mangrove loss studies omitting conversion details or being reported in an inconsistent manner is a common issue across nations.

The present study undertakes a remotely-sensed international assessment of mangrove forests that have been converted to commercial aquaculture. The mangrove to aquaculture conversion question is often at the crux of the coastal zone land-cover land-use change debate within the tropics, yet few if any analyses of this nature exist to assist researchers studying the implications of this important transition. This study addresses this need by providing an eight-country analysis covering the major mangrove holding regions of each nation and taken in conjunction with other regional and national analyses, provides not only a calculation of mangrove loss, but also an estimate of the amount of mangrove loss that can be attributed to the post-1970 expansion of commercial aquaculture.

MATERIALS AND METHODS

SITE SELECTION.—Eight countries were selected from the top 50 mangrove holding countries that also produce shrimp aquaculture (Table 1). The primary considerations when selecting nations and regions was the availability of remotely-sensed data for the required time periods, documented substantial historic mangrove holdings, and a modern-day commercial shrimp aquaculture industry; as shrimp aquaculture is the predominant form of aquaculture occurring in tropical estuarine mangrove forests. The countries that met these criteria are Indonesia, Brazil, Bangladesh, India, Thailand, Vietnam, Ecuador, and China. All of these countries have, or had, substantial mangrove coverage and contain the 1st, 3rd, 9th, 10th, 18th, 24th, 25th, and 48th ranked mangrove holdings as of 2000 (Spalding et al. 2010). All of these countries have a substantial aquaculture industry (FAO 2009, FAO 2010a). These eight countries combined constitute 36% of the world’s mangrove forests (Spalding et al. 2010) and 83% of the world’s aquaculture production (FAO 2009). These countries also account for 86% (FAO 2009) of the world’s farmed shrimp supply, which is the form of aquaculture most commonly associated with mangrove deforestation.

Within each country, the largest estuaries with mangrove forests were selected for analysis via a process of literature review and utilization of mangrove atlases. This resulted in 9 estuaries selected for Indonesia, 5 in Bangladesh, 6 in India, 6 in Thailand, 9 in Vietnam, 6 in southern China (Fig. 1A), 6 in Ecuador (Fig. 1B), and 9 in Brazil (Fig. 1C). Topographic maps, imagery, and paper maps were utilized to estimate the maximum possible extent of each mangrove forested estuary. In the present study, the term *estuary* includes deltas,

Table 1. Mangrove forest holding estimates by country, by the thousands of hectares, sorted by year with holdings rank (in parentheses). Only countries that also have shrimp aquaculture were included. Data taken or adapted from *Spalding et al. (2010) or †Giri et al. (2011). NR = not reported.

Country	1980*	1990*	2000*	2000†	2005*
Indonesia	4,200 (1)	3,500 (1)	3,150 (1)	3,113 (1)	2,900 (1)
Brazil	1,050 (4)	1,015 (3)	1,000 (3)	963 (3)	1,000 (3)
Bangladesh	428 (12)	460 (11)	476 (9)	437 (9)	476 (9)
India	507 (10)	467 (10)	448 (10)	368 (11)	448 (10)
Thailand	280 (18)	250 (18)	244 (18)	NR	240 (18)
Vietnam	269 (22)	214 (22)	158 (24)	NR	157 (24)
Ecuador	203 (26)	163 (26)	150 (26)	NR	151 (25)
China	34 (47)	28 (46)	23 (48)	NR	22 (48)

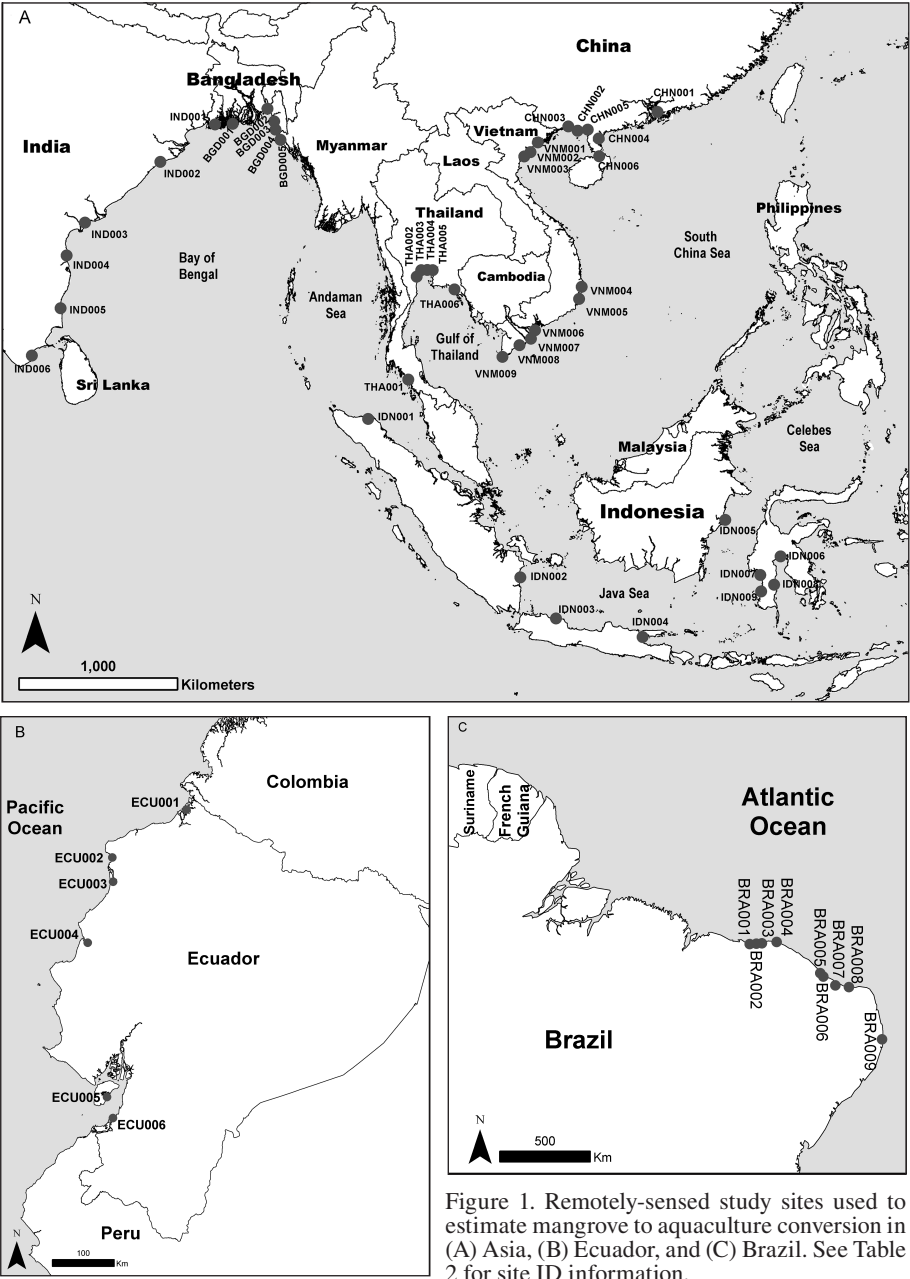


Figure 1. Remotely-sensed study sites used to estimate mangrove to aquaculture conversion in (A) Asia, (B) Ecuador, and (C) Brazil. See Table 2 for site ID information.

coastal lagoons, embayments, and other coastal regions with the required freshwater and saltwater inputs to support mangrove forests. This resulted in 11,055,700 ha of potential estuary analyzed across all nations. Within each estuary, random coordinate points were generated corresponding to the geographic envelope of each estuary at a density of one point per square kilometer and input into a geographic information system (GIS). This resulted in a total of 110,557 points analyzed across all estuaries with a breakdown by nation indicated below (Table 2).

Table 2. Study sites, major cities, point information, and coordinates of all estuaries analyzed.

Estuary ID	Study site	Reference city	Latitude	Longitude	Total points
BGD001	Ganges Delta	Bagerhat	22.283	89.478	10,000
BGD002	Feni River Estuary	Begamganj	22.739	91.541	599
BGD003	Matamuhuri River Estuary	Lama	21.752	91.942	1,403
BGD004	Bakkhali River Estuary	Cox's Bazar	21.457	92.022	198
BGD005	Naf River Estuary	Teknaf	20.934	92.268	195
BRA001	Delta do Parnaíba	Barra Grande	-2.973	-41.348	468
BRA002	Ilha dos Remédios	Barroquinha	-2.937	-41.091	39
BRA003	Rio Coreau	Camocim	-2.965	-40.810	103
BRA004	Rio Acaraú	Acaraú	-2.860	-40.052	130
BRA005	Rio Piranji	Fortim	-4.412	-37.937	79
BRA006	Rio Jaguaribe	Fortim	-4.717	-37.792	280
BRA007	Grossos Estuary	Grossos	-5.029	-37.190	309
BRA008	Macau Galinhos	Porto do Mangue	-5.133	-36.567	595
BRA009	Itamaracá	Caaporã	-7.669	-34.869	309
CHN001	Zhujjiang River Estuary	Shenzhen	22.921	113.167	9,782
CHN002	Fangchen/Zhenzhugang	Fangchenggang	21.714	108.566	1,814
CHN003	Beihai	Beihai	21.526	109.220	1,310
CHN004	Tieshangang	Shankouzheng	21.612	109.647	1,112
CHN005	Leizhou	Leizhou	20.962	110.386	4,183
CHN006	Hainan	Haikou	19.352	109.719	8,051
ECU001	Cayapas-Mataje	San Lorenzo	1.287	-78.920	510
EDU002	Muisne	Muisne	0.584	-79.995	67
ECU003	Cojimies	Pedernales	0.291	-79.965	274
ECU004	Chone	Bahía de Caráquez	-0.656	-80.317	89
ECU005	Rio Hondo	Isla Puná	-2.926	-80.124	313
ECU006	Grande	Machala	-3.288	-79.956	500
IND001	Sunderbans	Kolkata	21.993	88.598	10,964
IND002	Orissa	Satpada	19.715	85.409	2,546
IND003	Godavari	Bhimavaram	16.238	81.151	9,103
IND004	Nellore	Nellore	14.363	80.065	956
IND005	Vellar	Parangipettai	11.340	79.781	379
IND006	Tuticorin	Tuticorin	8.706	78.134	173
IDN001	Jampuaye River Estuary	Lhokseumawe	5.144	97.256	3,051
IDN002	Mesuji River Estuary	Bandar Lampung	-3.874	105.883	3,646
IDN003	Tarum River Estuary	Jakarta	-6.294	107.781	2,739
IDN004	Porong/Mas River Estuary	Surabaya	-7.238	112.745	2,464
IDN005	Mahakam Delta	Samarinda	-0.638	117.484	3,419
IDN006	Kalaena River Estuary	Malili	-2.712	120.644	1,346
IDN007	Sadang River Estuary	Pinrang	-3.765	119.530	698
IDN008	Kerra River Estuary	Watampone	-4.318	120.344	2,130
IDN009	Pangkajene River Estuary	Makassar	-4.715	119.553	2,248
THA001	Trang	Kantang	7.322	99.529	536
THA002	Mae Klong	Samut Songkhram	13.219	99.991	1,204
THA003	Ta Chin	Samut Sakhon	14.076	100.250	5,680
THA004	Chao Phraya	Bangkok	13.585	100.588	602
THA005	Bang Pakong	Chachoengsao	13.825	101.015	4,371
THA006	Chanthaburi	Laem Sing	12.462	102.147	685

Table 2. Continued.

Estuary ID	Study site	Reference city	Latitude	Longitude	Total points
VNM001	Vinh Ha Long Estuary	Ha Long	20.833	106.901	882
VNM002	Red River Delta	Thai Binh	20.254	106.459	362
VNM003	Day River Estuary	Long Châu	20.013	106.129	306
VNM004	Van Phong Bay	Van Ninh	12.602	109.342	633
VNM005	Cam Ranh Bay	Cam Ranh	11.939	109.200	291
VNM006	Tien-Ham Luong Estuary	My Tho	10.151	106.683	591
VNM007	Co Chien Estuary	Trà Vinh	9.763	106.495	1,041
VNM008	Bassac Hau Estuary	Bac Lieu	9.306	105.803	1,608
VNM009	Ca Mau Peninsula	Cà Mau	8.856	105.054	3,191

HISTORIC POINT CLASSIFICATION.—For each country analyzed, the period of early semi-intensive shrimp aquaculture was derived from Food and Agriculture Organization (FAO) aquaculture statistics (Fig. 2). Once this date was ascertained, the Landsat archive at US Geological Survey Global Visualization Viewer (GLOVIS) was queried for a Landsat scene with suitable atmospheric conditions immediately before the advent of widespread commercial aquaculture uptake. As artisanal aquaculture has existed in parts of Asia for many years before the advent of commercial aquaculture (Naylor et al. 1998), some limited small-scale aquaculture may have existed in the remote-sensing imagery before the FAO statistics indicate large-scale commercial production. This longitudinal image review process was repeated for all 56 estuaries and resulted in 90 Landsat scenes required for analysis. The wide range of historical scene dates is due to the differing time of arrival of commercial aquaculture in each estuary even within the same country (Table 3). This wide range of advent dates resulted in historical land use being derived from Landsat 1 (multispectral scanner systems, MSS), 2 (MSS), 4 (MSS/TM), and 5 (TM). It should be noted that Landsat 1 and Landsat 2 instruments have differing spatial and radiometric resolutions than Landsat 4 and Landsat 5. Within Ecuador, all but one of the estuaries (Cayapas-Mataje) had commercial aquaculture clearly visible in the earliest landsat scenes so aerial photography from 1968 or 1:25,000 topographic maps from the late 1960s and early 1970s were utilized as the basis for the historical land cover. For the region ECU006 (Fig. 1B), commercial aquaculture was clearly visible on the earliest maps and remote sensing data available so some initial survey points in Ecuador were listed as commercial aquaculture.

Once all Landsat scenes were geometrically adjusted, atmospherically corrected, and spatially aligned, a trained analyst was assigned to code each randomly generated point within each estuary. Each historic point was classified by the analyst as mangrove, aquaculture (very rare in historic scenes), other aquatic (mostly surface water), other terrestrial (land), or unable to classify. Each sample point was classified in a double blind system and points with a conflict were sent to a third analyst to arbitrate the appropriate classification. Additionally, the IsoData algorithm proved useful in deriving automatic clustering of mangrove forest and was utilized in all estuaries to assist the coder and help resolve classification conflicts. Only 30 IsoData clusters were generated as the classification only occurred in the predefined and delineated estuary environments with limited amounts of land-cover type. Clusters were generated at the 99% confidence level. Ancillary data such

Table 3. Historic and modern periods utilized in point classification within mangrove forests in major estuaries. “Historic” refers to pre- and “Current” to post-arrival of commercial aquaculture.

	Ecuador	Bangladesh	Brazil	China	India	Indonesia	Thailand	Vietnam
Historic	1966–1986	1972–1989	1987–1989	1973–1999	1972–1989	1972–1989	1973–1979	1975–1989
Current	2005–2006	2005–2010	2005–2009	2005–2007	2004–2009	2005–2010	2004–2006	2006–2009

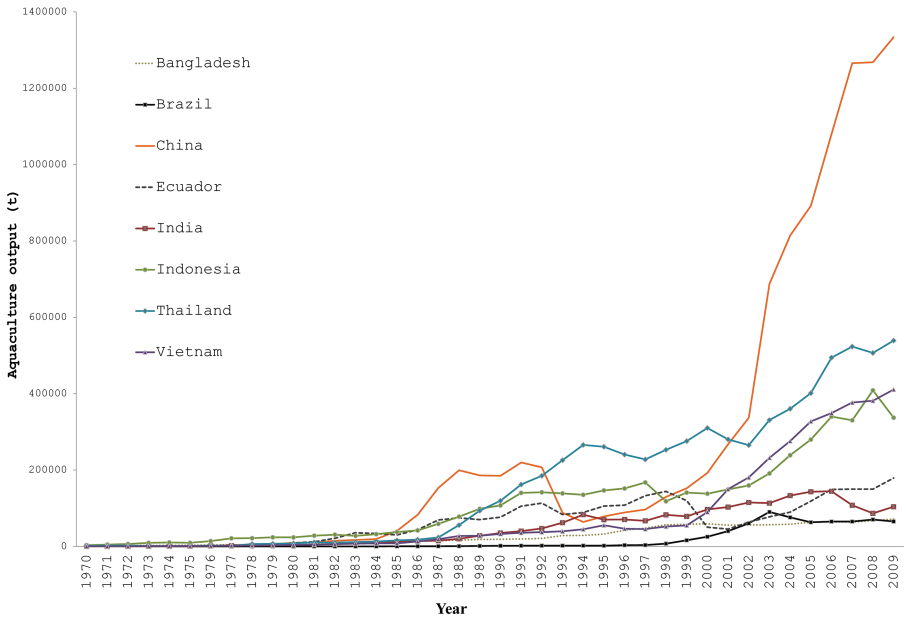


Figure 2. Commercial shrimp and prawn output in metric tons (t) by country by year (FAO 2009).

as paper maps, aerial photography, and ground-based photography were utilized as needed to assist in historic classification of the clusters and points.

MODERN POINT CLASSIFICATION.—The current classification system (image range 2004–2010) employed the same methods as the historic classification discussed above but with three significant differences. Firstly, IsoData and other unsupervised or supervised classification systems had only poor to moderate success identifying aquaculture land cover. This is due to the shrimp aquaculture process in which newly constructed shrimp farms exchange water on a daily basis with the estuary (Gautier et al. 2001, Stram et al. 2005). Particularly at the start of a shrimp grow-out cycle, when estuary waters flood the shrimp farms, the spectral signature of the shrimp pond is indistinguishable from the water in the estuary, making automated techniques based on spectral signature ineffective. This is likely the reason for limited international scale analyses of this type. For this reason, almost all of the aquaculture ponds were manually delineated using the same double-blind system as above but without automated assistance. Secondly, the range and availability of ancillary data is far greater in quantity, easier to access, and of higher quality for the modern period than the historic period. Thus, the classification success rate is likely higher. For example, we now have widespread availability of aerial or satellite photography from vendors such as Google, GeoEye, and Microsoft available to assist with classification. Additionally, georeferenced ground-based photography is available from numerous vendors and obtaining modern maps or atlases is far easier than obtaining historic ancillary or supporting sources.

The final difference in the historic vs modern analysis is when issues of cloud cover, poor atmospheric conditions, or other data quality issues occur in modern satellite scenes, an alternate is usually available. For example, for most of the current period, if Landsat TM had cloud cover we could find an alternate scene in close time proximity (a few months earlier or later) from another Landsat TM pass or utilize Landsat 7 or Advanced Spaceborne Thermal Emission and Reflection Radiometry (ASTER). Again, this leads to a higher confidence in classification as well as higher rates of classification when compared to

pre-aquaculture time slices. The modern classification of points utilized Landsat 5 (TM), Landsat 7 (Enhanced Thematic Mapper, ETM), and occasionally ASTER data, or some combination of all three. Again, due to poor atmospheric conditions or unsuitable imagery, the modern land-use classification involves a date range (Table 3), although the date range is far more compressed than for the historic point classification.

The ease of obtaining ancillary data and the increase in remote sensing systems available with improved spectral, spatial, temporal, and radiometric resolutions for the latter survey as opposed to the earlier survey could introduce some bias in the results, but this bias would most likely result in an overestimation of mangrove growth. For example, classification of mangrove forests during the current survey could occur in areas where mangroves existed in the earlier survey but could not be classified due to data deficiencies. In the earlier survey, such mangroves were likely coded as other aquatic, other terrestrial, or unclassified, but could now be depicted as mangrove. Errors of classification from mangrove to aquaculture, though limited, are possible due to differing spatial resolutions employed during the survey process. For example, if a survey point fell near the edge of a mangrove forest it could be classified as mangrove during the coarser earlier survey but classified as another land cover in the later survey of higher resolution. This error, although possible, is equally likely to cause an error in the opposite direction as the mangrove to aquaculture direction. For example, the earlier survey is equally likely to classify an area as terrestrial or aquatic due to being near the edge of one of these environments, but in the later more resolute survey, it could now be distinguished as mangrove. Little or no commercial aquaculture exists in the original surveys due to using the FAO dates of the inception of commercial aquaculture and examining all GLOVIS scenes for the initial period of large-scale aquaculture arrival in each estuary. Therefore, the potential for error originating from misclassification of aquaculture in the earlier survey is extremely limited, aside from ECU 006, where commercial aquaculture clearly exists in the initial survey.

RESULTS

MANGROVE LOSS.—Of the 110,557 analyzed points across all estuaries in all countries, 19,409 of these points were historically mangrove forest. Thus, 17.6% of all estuarine points were classified as mangrove forests before aquaculture arrived. This is not surprising as the analysis design targeted areas that currently, or historically, contained large and documented mangrove forests. Less than 1% of points could not be determined indicating that this 17.6% number has a limited margin of error.

Of these historic mangrove points, 48.1% remained mangrove forest as of the latest survey. This implies that across these eight countries analyzed, mangrove forests lost 51.9% of their aerial extent during the post-1970 period of commercial aquaculture expansion. While this sheds significant light on the plight of mangroves forests in the modern period, such losses may be even more substantial if not for mangrove regeneration and new growth. This methodology utilized in this analysis not only allows for mangrove loss and mangrove conversion to aquaculture to be reported but also allows for estimation of new mangrove growth. The percent of new mangrove was calculated to be 13%. In other words, some mangrove forest now exists in areas that were not mangrove in the past, such as other terrestrial or, more likely, aquatic environments. This new mangrove growth likely offsets some of the deforestation caused by aquaculture. Indeed, due to the relatively rapid growth rate of mangroves this growth could be in former shrimp ponds that once had mangrove but were converted to aquaculture earlier in the analysis period. Although mangroves lost 51.9% of their total area, a large variance exists between nations. For example, India has

85% of their forests remaining, Bangladesh has 79%, Brazil has 69%, and Ecuador has 57%, conversely China has only 9% of its mangrove remaining, whereas Indonesia, Thailand, and Vietnam have 39%, 42%, and 31% remaining, respectively (Table 4).

The total percentage of initial mangrove now classified as commercial aquaculture is calculated at 28% across all estuaries and regions with current terrestrial usage accounting for 19.5% of the original mangrove area reported in the first survey (Table 4). That is, 28% of these former mangrove forests are now coded as commercial aquaculture and 19.5% are coded some other terrestrial land cover. The "other terrestrial" conversion is likely urban encroachment or conversion to agriculture uses through the long established process of mangrove reclamation (Ellison and Farnsworth 1996). This urban encroachment pattern appears most vividly in China and Thailand, with 7% and 14%, respectively, more mangrove lost to terrestrial activity than aquaculture (Table 4). China and Thailand are the only nations analyzed where terrestrial expansion and not commercial aquaculture appears to be the primary driver of mangrove deforestation.

Although the combined eight-country rate of mangrove to aquaculture conversion is 28%, and aquaculture appears to have displaced 54% of all mangrove that have been lost, substantial regional variations occur and these differences suggest that direct extrapolation of these findings to estimate a global pattern of mangrove displacement by aquaculture is problematic. For example, within Ecuador, Indonesia, and Vietnam, 40% or more of the former mangrove forest is now aquaculture (Table 4). These patterns are in stark contrast to Bangladesh, India, Brazil, and Thailand, where 19% or less of first survey mangrove forest is now utilized for commercial aquaculture. Although the level of mangrove deforestation may be lower in Bangladesh, India, Brazil, and Thailand, in all of these nations aquaculture accounts for a substantial portion of deforested mangrove even when the deforestation level is limited (Table 4). This suggests that aquaculture is responsible for a significant portion of mangrove deforestation, not only when mass deforestation is occurring such as in Ecuador but also when limited deforestation occurs such as in India (Table 4).

Some caveats and error rates should be noted in interpreting the results of this analysis. Due to the time span of the analysis and the growth rate of mangrove forest, it is unlikely, though not impossible, that some of this forest could have been deforested and then regrown during the analysis period. Indeed, this could even occur in forests converted to aquaculture. Ecuador was oversampled at least once a decade to test for this scenario and <1% of points indicated this occurred. Although a field quality procedure is not viable for any of the historic land-cover classifications, ancillary data is utilized to verify findings including historic topographic maps and aerial photography as well as multiple sensor readings when possible.

DISCUSSION

Within the eight countries and 56 estuaries analyzed, 51.9% of historic mangrove coverage was lost. Of the mangroves lost, 28% were displaced by commercial aquaculture, representing 54% of all mangroves that have been displaced. This analysis contributes to the increasing body of research that equates aquaculture expansion to mangrove forest decline, although not necessarily at the high rates reported in regional or country level analyses (Falherty and Karnjanakesorn 1995, Dieberg and Kiattisimkul 1996, Binh et al. 2005, Centro De Levantamientos

Table 4. Mangrove loss by land-cover type. "A" is the percent of historic mangrove forest listed as current land-cover type. For example, 57% of past mangrove in Ecuador is currently mangrove and 40% of what was mangrove is now aquaculture. "B" is the percent of mangrove loss that can be explained by each land-cover type. For example, of the mangroves lost in Ecuador, aquaculture accounts for 93% of that loss. "C" is the estimated land cover change from historic mangrove to current land-cover, estimated in hectares. In Ecuador, for example, 34,500 ha of land that was mangrove is now aquaculture.

	Ecuador			Bangladesh			Brazil			China			India			Indonesia			Thailand			Vietnam		
	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
Current cover	(%)	(%)	(ha)	(%)	(%)	(ha)	(%)	(%)	(ha)	(%)	(%)	(ha)	(%)	(%)	(ha)	(%)	(%)	(ha)	(%)	(%)	(ha)	(%)	(%)	(ha)
Undetermined	0	0	0	0	0	0	<1	0	100	<1	0	100	0	0	0	<1	0	1,500	0	0	0	0	0	0
Aquatic	3	7	2,600	6	29	27,600	5	16	2,300	4	4	19,600	5	33	16,000	2	3	5,700	6	10	3,100	5	7	6,100
Terrestrial	<1	0	400	8	38	34,700	13	42	6,500	47	52	251,100	6	40	18,000	11	18	38,700	33	57	15,700	11	16	13,700
Aquaculture	40	93	34,500	7	33	30,600	13	42	6,500	40	44	212,700	4	27	12,600	48	79	174,500	19	33	9,000	53	77	64,100
Mangrove	57	NA	49,500	79	NA	347,600	69	NA	34,900	9	NA	46,600	85	NA	255,300	39	NA	140,700	42	NA	20,100	31	NA	38,200

Integrados De Recursos Naturales Por Sensores Remotos 2007, Satapathy et al. 2007, Giri et al. 2008, Hamilton and Stankwitz 2012). One reason for the present lower conversion rates is that few regional mangrove to aquaculture analyses are likely to be conducted in locations with little or no conversion, as there would be no conversion to report. Due to the random point allocation utilized in the present study, regions with little conversion were as likely to have been sampled as those with high rates of conversion generating a more complete estimate of conversion rates than those based on regional case studies.

All of the nations analyzed, aside from China, have previous mangrove to aquaculture analyses to which results of the present study can be compared (Fig. 3). For example, a major study of mangrove to aquaculture conversion in Vietnam reports a 75% mangrove loss between 1968 and 2003 in and around the Ca Mau peninsula with aquaculture responsible for 40% of these losses (Bihn et al. 2005). This equates to a 30% mangrove to aquaculture conversion level. The Binh et al. (2005) 75% mangrove loss calculation corresponds well with the 69% loss reported herein, but their 30% mangrove to aquaculture conversion figure is below the 53% estimated in the present analysis (Fig. 3). This difference is likely due primarily to the different aerial extent of the study sites, as the Binh et al. (2005) analysis area constitutes a small subset of the region depicted in these analyses. Additionally, the present study continued six years beyond that of Binh et al. (2005).

There are other estimates of mangrove loss and mangrove conversion in Asia; following the 2004 Indian Ocean earthquake and Tsunami, Giri et al. (2008) undertook a robust remote sensing driven spatiotemporal analysis from 1977 to 2005 of mangrove change in and around the Bay of Bengal including parts of India, Bangladesh, Thailand, and Indonesia. Their findings indicate a 41% ($\pm 3\%$; 16,816 ha) mangrove to aquaculture conversion rate for their study sites in Thailand, a 63% ($\pm 4\%$; 20,965 ha) conversion rate for their study site in Indonesia, a 22% ($\pm 1\%$; 7554 ha) conversion rate for their study site in India, and an 11% ($\pm 1\%$) conversion rate for their study sites in Bangladesh (Fig. 3). All of these mangrove to commercial aquaculture conversion values are similar to results reported herein: Thailand (41% vs 33%), Indonesia (63% vs 79%), and India (22% vs 27%). Although the Giri et al. (2008) analysis contains a spatial subset of the present analysis and has differing temporal resolutions, their analysis methods are similar to those used herein and allow for direct comparison. Only Bangladesh (11% vs 33%) appears to require further explanation into the differences reported and this is likely due to the different aerial extent and temporal resolution of the two studies.

Within China, estimates of mangrove to aquaculture conversion are lacking. Li and Lee (1997) utilized a review of earlier, mostly Chinese literature to estimate mangrove loss but not conversion and much of the literature they reviewed was not survey or remote-sensing based. Comparative analyses outside of Asia include the mangrove to aquaculture conversion findings of Hamilton and Stankwitz (2011) conducted in all Ecuadorian estuaries, which agree with estimates herein, and the similar conversion level reported by Guimarães et al. (2010) in their study on the Brazilian coast (Fig. 3).

Consideration of all countries together lowers the mangrove to aquaculture conversion estimates for Thailand, Indonesia, India, and Bangladesh compared to the previous remote-sensing estimates. For Ecuador and Brazil, the present analysis did not change previous estimates and in only Vietnam do the present

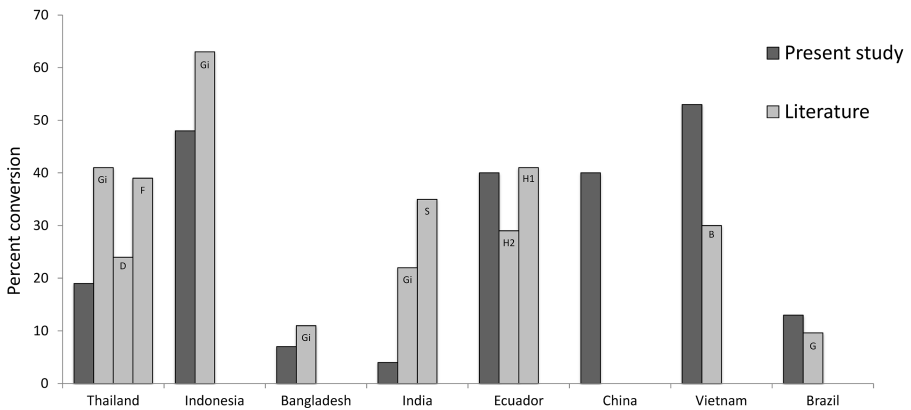


Figure 3. Comparisons of conversion of mangrove to commercial aquaculture rates for the present study and previous studies for each of the study locations. Gi from Giri et al. (2008), D from Dierberg and Kiattisimkul (1996), B from Binh et al. (2005), G from Guimaraes et al. (2010), S from Satapathy et al. (2007), F from Flaherty and Kamjanakesom (1995), and H from Hamilton and Stankwitz (2012). When other findings report a range the mean is utilized. Hamilton and Stankwitz (2012) report two results (H1) including and (H2) excluding the national parks so both are reported.

estimates of mangrove to commercial aquaculture conversion increase by a substantial amount.

Protected areas appear to play an important role in determining conversion rates. For example, four of the five study sites in Bangladesh are either wholly or partly in the Sundarbans world heritage and Ramsar site. This protected area status almost entirely explains why Bangladesh (and India to some degree) has maintained the majority of their mangrove forests. For example, within Bangladesh, outside of these protected areas, deforestation and conversion rates are still relatively high. Within the heavily converted Ecuadorian estuaries, protected status also appears to play a mitigating role. The second largest estuary in Ecuador is another Ramsar site (Mangrove Ecological Reserve Cayapas-Mataje) and this site is responsible for almost all of the unconverted remaining mangrove forest in the country. If not for this single site in Ecuador, mangrove loss and aquaculture conversion would be substantially higher.

While this land-cover change analysis provides international, national, and estuarine specific measures of mangrove displacement by aquaculture, it also gives insight into some of the processes that may underlie the differences noted. For example, within Thailand, and to a more limited extent in Vietnam, a pattern of some mangroves remaining inside of the aquaculture ponds was noted. This is in direct contrast to many other estuaries in the world where mangroves appear to be clear cut to make way for aquaculture. This practice may account for the high levels of aquaculture output noted in Thailand but the lower levels of reported mangrove deforestation.

The interplay between mangrove deforestation and aquaculture expansion differs from nation to nation and within each nation, making it more complex than a linear relationship between the amount of aquaculture produced and the level of mangrove deforestation within each country. For example, both Nigeria and Australia have substantial remaining mangrove holdings, but neither has a large

aquaculture industry located within their estuaries, so conversion from mangrove to shrimp aquaculture is likely close to zero, although loss for other reasons likely occurs. This may cause an overestimation of conversion in the present study as countries that do not produce shrimp via aquaculture were excluded. However, these results also indicate that part of the reason substantial mangrove forests remain in these non-aquaculture producing countries is that aquaculture is not present. Conversely, this analysis likely simultaneously underestimated mangrove deforestation to some degree, as one of the criteria for inclusion in the analysis was that some mangrove forest must be present in an estuary, hence estuaries entirely denuded of mangrove were likely excluded from the analysis. Further analysis is needed to ascertain if this large sample of mangrove to aquaculture conversion is consistent with other mangrove holding countries and regions.

ACKNOWLEDGMENTS

Funding for this research was provided by internal funding from the College of William and Mary, the University of West Florida, and the University of Southern Mississippi. In-kind support was provided by the Nature Conservancy. Thank you to S Byrd for directing the land-cover change analysis, assisting in the creation of figures, tables, charts, and for much needed editorial assistance. In addition to S Byrd, many students at College of William and Mary contributed to the time-consuming and systematic point coding required to obtain these results. Particular thanks to K Branch, M Yuan, S Young, A Grant, E Caverly, and V Chung for undertaking this task. Thank you to J Holden for assisting with the batch IsoData runs required. As always, appreciation is given to the USGS for the Landsat program and NASA and the government of Japan for providing researchers free access to ASTER data. Without data provided free of charge by the FAO Fisheries and Aquaculture department of the UN, analyses such as this would not be possible. Finally, it is important to acknowledge the work of other authors (Giri, Spalding, and others) who have spent considerable time and effort documenting the aerial extent of mangroves at the global and national levels. Without leaning on such resources an analysis such as this would be too time consuming to undertake.

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DATE SUBMITTED: 6 September, 2012.

DATE ACCEPTED: 20 February, 2013.

AVAILABLE ONLINE: 28 February, 2013.

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