Cascading Globalization and Local Response: Indian Fishers' Response to Export Market Liberalization

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Abstract

Scholars have long debated whether trade liberalization has positive or negative effects on resource use and ecosystems. This study examines the conditions under which resource use increases or decreases in response to reduced trade barriers, specifically after the 2008 World Trade Organization decision that led the United States to reduce anti-dumping duties on Indian shrimp. At the district level in South India, fishing fleet expansion was correlated with access to global market information via mobile phones. Model simulations indicate that increased mobile phone saturation could expand fishing effort sufficiently to deplete multiple marine species groups, while other species benefit from the loss of predators. However, scenario analysis suggests that regulatory interventions could mitigate these ecosystem pressures while still permitting fishers to benefit from increased access to global market information.

Keywords

globalization and environment, trade and environment, coastal socioecological systems, coupled human and natural systems, coastal fisheries, information technology

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Fishing communities around the world confront crucial dilemmas in the face of globalization. Trade liberalization may provide new opportunities for previously isolated villages if they can gain access to global markets. However, without careful management, such market entry may also interfere with long-term resource sustainability and poverty reduction. This article addresses this dilemma at the subnational level by studying fishing behavior and ecosystem shifts after trade liberalization, as well as possible interventions that moderate these shifts. We examine how fishers have changed their fishing behavior (i.e., as measured by fleet size at the district level), in response to a discrete episode of market liberalization.

Export markets were opened by a 2008 World Trade Organization (WTO) decision forcing the United States to reduce anti-dumping duties imposed on Indian shrimp. Exporters benefited from these declining market barriers, but little research has examined how global market signals reach primary, local resource users and affect their harvest decisions. In this article, we explore fleet size changes and then project how such shifts translate into ecosystem effects under different resource management scenarios.

There is considerable debate over the environmental effects of trade liberalization (Esty, 2001). Some scholars argue that liberalized trade has detrimental effects on the environment due to increased production (Daly, 1993), which leads to greater pressure on resources (Brewer, Cinner, Fisher, Green, & Wilson, 2012; Shrivastava & Kothari, 2012), limits on domestic regulation (Porter, 1999), and constraints on international environmental rules (Axelrod, 2011; Stilwell & Tuerk, 1999). Others suggest that increased trade, by promoting stringent regulation (Prakash & Potoski, 2006; Vogel, 1995) and raising living standards (Bhagwati, 1993) that in turn promote livelihood diversification (Cinner & Bodin, 2010), reduces pressure on natural resources and increases sustainability (de Soysa & Neumayer, 2005).

As with other resource users, fishers' response to trade liberalization is still in question. Increased market access can potentially result in higher prices for fish products (Schmitt & Kramer, 2010). Fishers may respond to price increases by expanding fishing fleets (Barkin & DeSombre, 2013; Bort, 1987), assuming there are no institutional constraints on such expansion, with the expectation to garner greater profits. In this scenario, further ecosystem degradation might be expected to follow. Alternatively, greater certainty of reaching subsistence income levels may lead to less fishing or more cautious resource use. This scenario could occur if market participants either focus on maintaining their livelihoods for the long run or engage in alternative behaviors other than fishing.

However, external disturbances such as global market shifts do not have a homogenous effect on fishing behavior and coastal ecosystems. Rather, as we demonstrate, fishers' responses to global market shifts are not uniform across locations, indicating a need to identify the factors that allow trade liberalization to trigger increased effort and resulting ecological responses. Although some previous studies have suggested uniformly positive or negative effects of trade liberalization, we explore the *conditions under which such market perturbations lead to robust or vulnerable marine socioecological systems (SESs)*. We hypothesize that global market shifts have a greater impact on fishing fleets when fishers have greater access—via mobile phones—to market information. We assess how fishers and fish stocks respond to market liberalization, finding support for this information access hypothesis using district-level data from India's southernmost states, Kerala and Tamil Nadu (TN). Furthermore, we find that this relationship is particularly strong in the mechanized fishing sector, whereas mobile phone access is correlated with a shift away from nonmotorized artisanal fishing. This pattern suggests that information is necessary to reap short-term benefits of globalization, which are most clearly observed through increased capital expenditure. We then simulate the ecosystem impacts of such fleet size changes under different management scenarios. These simulations demonstrate how communities may intervene to limit overexploitation of marine resources in the context of market liberalization.

We proceed by reviewing recent literature on SES sustainability in the face of market disturbances. We then develop and test the information access hypothesis, using our statistical results to project substantive ecosystem impacts of market liberalization and demonstrate how resource managers can mitigate these impacts. Finally, we conclude by addressing the implications of our findings.

Theory

Coastal SES

SESs are complex linked systems in which social and ecological components mutually affect each other. To understand how fish stocks and coastal communities respond to disturbances such as trade liberalization, coastal ecological and social systems must be considered together as interrelated parts of a coupled system (Perry, Barange, & Ommer, 2010; Thébaud et al., 2014). We recognize the long history of coastal ecosystem collapse due to overfishing and natural variation (Jackson et al., 2001) and do not suggest that coastal SESs remain in stable states prior to external disturbance. Rather, this model provides a starting point for examining systemic impacts of such disturbances independent of the prior state.

In the coastal SES, fishing effort (i.e., fleet size or fishing hours) affects available stocks (A, in Figure 1) because targeted species decline when individuals are removed at rates greater than replacement (Walters, 2001). In addition to targeted species, bycatch also affects fish stocks. Shrimping has particularly large impacts on marine ecosystems through benthic substrate trawl damage (Watson, Revenga, & Kura, 2006) and bycatch (Venkataraman & Melkani, 2007). Worldwide, shrimp fishermen discard an estimated 15 million tons of

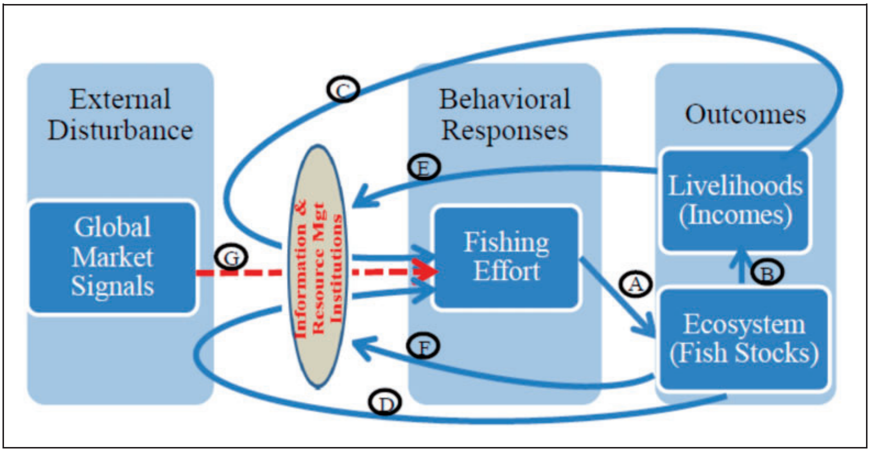


Figure 1. Sample coastal socioecological system schematic, showing how interventions moderate the impact of external disturbances such as global market shifts.

bycatch yearly, whereas all other fishermen combined discard roughly 5 million tons per year (Venkataraman & Melkani, 2007). Indian shrimp trawlers discard up to 90% of their catch (Lobo, 2007; Venkataraman & Melkani, 2007). Therefore, shrimp trawler fleet expansion could have substantial effects on other marine species. Declining fish stocks then directly impact livelihoods, in pre- and post-harvest sectors as well as fishing households (B, in Figure 1; Cinner, Daw, & McClanahan, 2009; International Collective in Support of Fishworkers [ICSF], 2005).

Livelihood outcomes and fish abundance may then affect vessel investment in subsequent periods (C and D, in Figure 1), as people regularly evaluate whether it is worthwhile to harvest fish—both in terms of the fleet size required to obtain the necessary catch and whether selling this catch will meet financial needs. Alternatively, fishers may consider whether alternative professions offer better livelihood security (Coulthard, 2008). If they continue fishing, adaptation may involve adopting different harvest methods or targeting species that are not (yet) depleted. These strategies may lead to further ecosystem degradation even if livelihoods are protected in the short term (Essington, Beaudreau, & Wiedenmann, 2006; Pauly, Christensen, Dalsgaard, Froese, & Torres, 1998).

Overuse of natural resources, including overfishing, is a proximate driver of ecological change (Brewer, Cinner, Green, & Pressey, 2012). Others note, however, that commons management solutions provide incentives for cooperation to avoid disastrous consequences (Berkes, 2006; Ostrom, 1990). Indeed, in coastal systems, fisheries managers try to balance social and ecological goals through rules and regulations (i.e., institutions), many of which aim to limit fishing effort (e.g., vessel limitations, temporal restrictions, etc.) and positively impact fish stocks and livelihoods in the long term. These institutions exist at local, national, or international levels and often provide incentives to reduce overfishing. As a result, feedback from stocks and livelihoods to future effort (i.e., fleet size; C and D, in Figure 1) is constrained by institutional arrangements in place for that particular area.

Adaptive governance represents feedback from current ecological conditions and livelihoods to management decisions at the same scale (E and F, in Figure 1). When managers realize that available management options do not achieve desired outcomes, they may try to adjust rules to maximize certain goals (Olsson, Folke, & Berkes, 2004). We address one situation in which such information may be used to improve management interventions.

Adaptive governance may address internal perturbations to the SES. However, as external perturbations—such as market shifts—are introduced, they may still alter relationships within the SES (Schoon & Cox, 2012). A wide range of interacting external disturbances may exist, including regional credit arrangements (Crona, Nyström, Folke, & Jiddawi, 2010), national-level policies such as fishing subsidies (Pauly, 2002), global market shifts (Young et al., 2006), and natural disturbances such as disasters (Hughes, Bellwood, Folke, Steneck, & Wilson, 2005) or climate change (Allison et al., 2009). These external disturbances (G, in Figure 1) raise the specter of linkages across social and ecological systems operating at multiple levels. Community-level SESs are adjacent to other communities and may be nested within regional, national, and international systems as well.

Market Perturbations as Systemic Disturbances

As communities begin to rely on other locales for production inputs and consumer markets, resource flows increase, thereby disturbing communities' traditional uses of local natural resources (A to F, in Figure 1; Gadgil & Guha, 1993). As a result, the *system to be governed* then extends beyond traditional governance system boundaries, particularly in the context of fishing villages that previously governed themselves as closed social systems (Bavinck & Salagrama, 2008), even though related ecological systems were always connected at larger scales. Thus, as the scale of social activities expands, some scholars suggest that the system faces increased pressure (Young et al., 2006), including expanded global seafood demand (Smith et al., 2010). Fishers often respond to increased demand by expanding effort through market entry and vessel investment (Iwasaki & Shaw, 2009), though market processes are moderated by barriers to international trade (Salim & Biradar, 2009).

Trade liberalization provides one potential disturbance by increasing incentives for producers, including fishers, to further exploit resources for which demand has increased (Brewer, Cinner, Fisher, et al., 2012; Rock, 1996; Shrivastava & Kothari, 2012). These consequences of market liberalization may be intensified because decision makers often subordinate fisheries management decisions to other policy objectives such as economic development (Graff Zivin & Damon, 2012).

However, other scholars suggest that disturbances may instead provide an opportunity for increased resilience, opening the possibility of adaptation by finding more efficient and sustainable resource exploitation methods (Young et al., 2006). In particular, some find that market liberalization may promote stringent regulation to meet wealthy consumer preferences (Garcia-Johnson, 2000; Prakash & Potoski, 2006; Vogel, 1995; Zeng & Eastin, 2012). In addition, increased competition may lead to more efficient production and therefore increased resource sustainability (de Soysa & Neumayer, 2005). Regulation and competition may limit or balance against the ecological impacts of increased fishing.

Moving beyond this simple dichotomy, some scholars find that fish trade has more ambiguous impacts on economic development (Béné, Lawton, & Allison, 2010), and therefore fishing decisions. As Folke (2006) notes, the charge for scholars is to identify *what conditions* facilitate successful adaptation to disturbance. Indeed, others show that response to global market signals *varies* based on local institutions and socioeconomic factors (Garrett & Lange, 1996; Rudra & Jensen, 2011), providing an opportunity to analyze specific conditions that promote SES sustainability in the face of disturbance (Perry et al., 2010). Nonetheless, most vulnerability studies address only national-level variation (e.g., Allison et al. 2009), despite within-country behavioral variation (Bavinck, 1998; Castello, McGrath, Arantes, & Almeida, 2013; Coulthard, 2008). In the next section, we examine information access as a possible cause of this variation.

Explaining Fishing Response Variation

When trade liberalization increases market access, exporters expect higher prices due to increased demand. In turn, exporters may be willing to pay producers (i.e., fishers) more for each unit due to increased competition for available fish (Schmitt & Kramer, 2010). While fishers do not export their product directly, they negotiate prices for their catch, often with middlemen who sell on to processors and/or exporters. Therefore, when fishers are aware of increased export prices, they may be able to negotiate higher sale prices, possibly incentivizing increased fishing.

This sequence of responses to trade liberalization relies on assumptions of efficient markets, including fishers' access to market information. However, information is often scarce and unevenly distributed in rural areas, particularly in developing countries. When fishers are unaware of price shifts, they do not have an opportunity to gain greater prices for their products. As a result, resource harvesters with information access will likely obtain higher prices than those without (Stigler, 1961), meaning that we should see greater behavioral responses (i.e., fleet expansion) in the presence of market information. Therefore, we anticipate that trade liberalization will increase resource use only when fishers are aware of price changes.

Mobile phones are one tool for bridging information gaps. Recent studies show that mobile phones play an important role in marine governance coordination (Hoefnagel, de Vos, & Buisman, 2013). Overå (2006) finds that mobile phone usage also decreases transaction costs and increases trust throughout agricultural supply chains by increasing the frequency of interactions. They may also make agricultural and fishing work easier by supporting extension work, providing weather information, and facilitating emergency rescue (Mittal, Gandhi, & Tripathi, 2010). More important, for our purposes, mobile phones may allow producers to engage in price arbitrage by providing information that allows them to negotiate among multiple buyers (Aker & Mbiti, 2010). Such a technique is particularly important for perishable goods where information must be transmitted quickly before the harvest spoils (Muto & Yamano, 2009).

Similarly, mobile phones should allow fishers to acquire market information and negotiate accordingly with exporters and their agents, particularly in regions such as South India that are characterized by monopsony (Sathyapalan, Srinivasan, & Scholtens, 2008). As a result, when mobile phones are available, scholars observe greater competition and price convergence in Indian coastal fisheries (Jensen, 2007). Under these conditions, price increases may influence fisher incomes and therefore fishing fleet investments. We therefore hypothesize that trade liberalization increases fleet size more when fishers have greater access to information through mobile phones. Fleet size increases can be observed both in terms of overall number of vessels, as well as the shift from artisanal boats to mechanized vessels that can harvest larger quantities of marine resources per vessel. If this hypothesis is accurate, then we should observe a greater impact of trade liberalization on fleet size in areas with greater mobile saturation. Conversely, if mobile phones lead to better fisheries management and reduced wastage as Jensen (2007) and Abraham (2007) suggest, then mobile phone access may reduce fleet size in the presence of market access if fishers wish to achieve a certain level of income rather than maximizing profit. In the next section, we test the explanatory value of mobile phone saturation on fleet sizes.

Research Methods—Analyzing Indian Fisheries in the Face of Market Liberalization

Study Period and Location

To test the information access hypothesis, we examine fleet size changes in light of a particular trade liberalization episode. We focus on the presence and removal of protectionist measures placed against Indian shrimp by the United States. Fisheries are a key economic development sector in India (Thorpe, Reid, van Anrooy, & Brugere, 2005). Shrimping dominates the Indian seafood export market such that shrimp alone compose over 60% of India's total marine fish exports by value (Marine Products Export Development Authority [MPEDA], 2014). While shrimp are relatively abundant in Indian coastal waters, shrimping may impact broader marine ecosystems as discussed earlier.

After becoming India's main shrimp export market, the United States accused India and others of dumping shrimp on the U.S. market at less than fair market price. In response, the United States instituted anti-dumping duties on shrimp imports from India in 2004. India successfully petitioned the World Trade Organization Dispute Settlement Body (WTO DSB) to overturn these duties in 2008. The United States complied with the ruling and decreased duties on Indian shrimp imports from 10.17% to 0.79% over a 1-year period (Punnathara, 2009b). As a consequence, the market for Indian shrimp experienced a discrete episode of market access liberalization in late 2008 and early 2009, allowing us to examine impacts by comparing fishing fleet size before and after this event.

The 2004 anti-dumping duties had a major impact on Indian shrimp exports to the United States. Following a steady rise from 1996 to 2003, the period of anti-dumping duties (2004–2008) demonstrates a continuous decline in shrimp imports from India, despite an almost continuous—though shallow—increase in total U.S. shrimp imports from other countries (e.g., Ecuador, Indonesia, and Malaysia) over that period (Figure 2). In monetary terms, Indian shrimp exports to the United States declined from $409 million in 2003 (before duties were

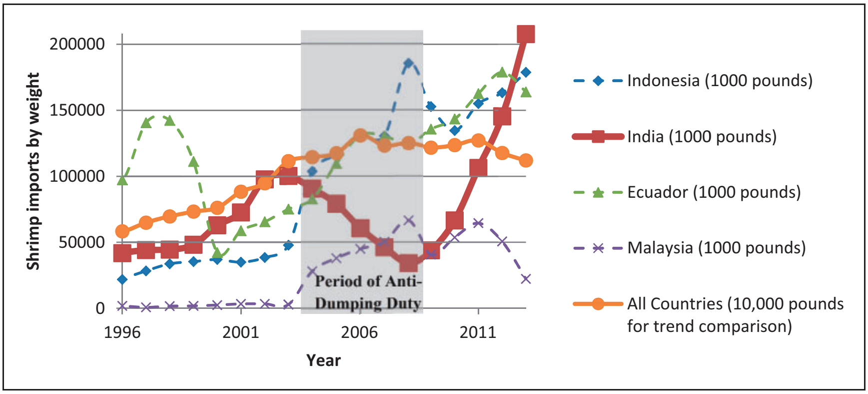


Figure 2. U.S. shrimp imports from selected countries and overall, 1996–2013.
Source. U.S. Department of Agriculture, Economic Research Service (Retrieved from http://www.ers.usda.gov/data-products/aquaculture-data.aspx).

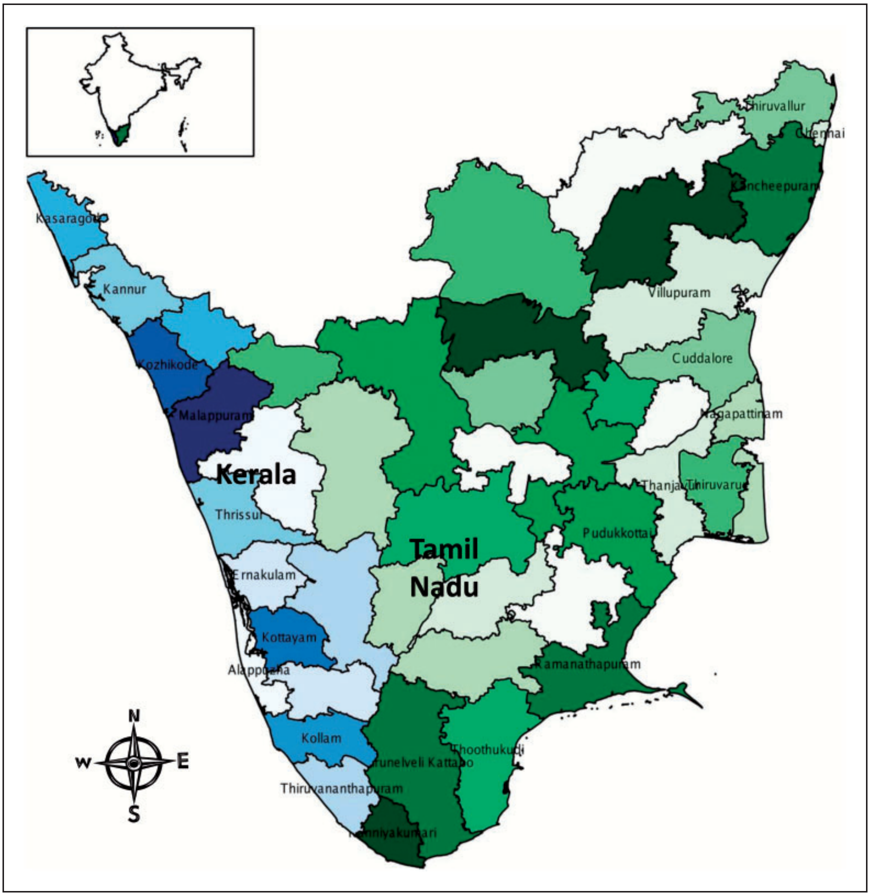
enacted) to $142 million in 2008, the year in which WTO DSB found U.S. duties unacceptable. Despite a slight 2010 rise in duties, 2010–2011 data indicate a return to the trend of increasing shrimp imports from India to the United States, to $452 million (Salim & Narayanakumar, 2012). U.S. imports have long provided the largest market for Indian shrimp. Indeed, though exports to the United States declined steeply while the duties were in place, the United States remained the top purchaser of Indian shrimp (by volume) until 2007–2008 when Japanese imports surpassed the U.S. market for 3 years. Anecdotal news reports suggest that the increased U.S. exports in 2010–2011 and beyond were appreciated by the industry, stating that "the saving grace for [export to] the US market remains the successive slashing of the anti-dumping duties, which has enabled the Indian shrimp exporters to hold their own in the fiercely competitive US markets" (Punnathara, 2009a).

Although overall Indian seafood exports continued to grow during the 2004–2008 period, the rate of growth slowed considerably, sharply expanding again after the 2009–2010 fiscal year (MPEDA, 2014). As such, demand for Indian shrimp (the largest portion of India's seafood exports) appears to have shifted along with the changing U.S. duties.[1] Our goal is to determine how such demand shifts affect fleet size and therefore ecosystem responses, and what factors condition this effect. Domestic demand for shrimp in India remains relatively low, with imports less than 0.005% of exports on average between 2000 and 2011 (India Department of Commerce Export-Import Data Bank). As a result, fishers do not compete locally with foreign producers, meaning that prices should primarily rise with expanded access to United States and other markets.

Data

This study responds to calls for research on the relationships between market, fishing, and ecosystem dynamics. We therefore analyze district-level fleet size for each vessel type, and fishery-independent biomass data collected by India's Central Marine Fisheries Research Institute (CMFRI). We first compare 2007–2011 fleet size changes across districts and vessel types (nonmotorized, motorized, mechanized) in India's southernmost states, Kerala (9 districts) and TN (12 districts—CMFRI combines data from adjacent districts Thiruvarur and Thanjavur; see Map 1). These states are major shrimp producers, accounting for 12% (Kerala) and 8% (TN) of India's crustacean landings in 2004–2011. Together, there are 21 districts, with up to three vessel types (nonmotorized artisanal boats, motorized artisanal boats, and vessels with mechanized fishing gear) in each district. Therefore, we have a total of 63 possible observations. However, some are dropped in each analysis due to missing covariate data or lack of a particular vessel type in certain districts (Table 1).

Our dependent variable is the change in district-level fleet size by vessel type over the period 2007 (i.e., the last full year before U.S. duties were reduced) to



Map 1. Coastal districts of Tamil Nadu (green) and Kerala (blue), India.

2011 (by which time vessel investments would have fully responded to the duty reduction).[2] Fleet size is measured on an annual basis as the number of active fishing units of each type in that year. We reached similar results as those below when replacing fleet size with the number of fishing hours by vessel type in each district.

Once we specify the model estimates, we use CLARIFY (Tomz, Wittenberg, & King, 2001) to simulate how mobile phone access moderates responses to global market shifts (G, in Figure 1). CLARIFY uses Monte Carlo simulation to predict how fleet size and composition will change when one explanatory variable shifts values and all others are held at their mean. This simulation does not add additional findings but helps to interpret the magnitude of mobile phones' effects on fishing effort. Furthermore, having established this

Table 1. Summary Statistics.

Variable	Source	# of Observations	*M*	*SD*	Range	Expected influence on fleet size
DV: Fleet size change (2007–2011)	CMFRI	56	0.29	1.81	−1, 10.89	N/A
IV: Mobiles per family	Marine Fisheries Census (MFC)	63	0.44	0.20	0.17, 0.92	+
Preservation infrastructure facilities per village	MFC	63	1.04	1.12	0, 4.10	+
Educational institutions per family	MFC	63	0.0061	0.0047	0.0013, 0.021	+
CPUE change (2007–2011)	CMFRI	51	0.57	2.11	−1, 11.10	+

Note. CMFRI = Central Marine Fisheries Research Institute; CPUE = catch per unit effort.

likely level of fleet size change, we can then use Ecopath with Ecosim (EwE) to simulate how such increased fishing will impact fish stocks (A, in Figure 1) under different management scenarios (Christensen & Walters, 2004).

In response to trade liberalization, shifts in fleet size vary by district and vessel type in both Kerala and TN (Figure 3). For instance, while the Kanyakumari district (TN) motorized fishing fleet follows the pattern one might expect based on overall export trends, nonmotorized fishing in Kanyakumari declined over the same period, and the mechanized fleet declined but eventually began to recover (Figure 3(a)). In contrast, Alappuzha district (Kerala) demonstrates rising motorized and nonmotorized fleets while the duties were in place, followed by declines after markets opened. This variation demonstrates that response to global market shifts is not uniform across districts or vessel types. The remainder of this article analyzes the causes of this variation.

Our key independent variable is information access. This factor is measured at the district level by the mean number of mobile phones per family in coastal areas (i.e., mobile saturation). This measure is drawn from CMFRI's 2010 Marine Fisheries Census (CMFRI, 2010), a survey of all Indian marine fishing villages.[3] Seventy-one percent of respondents were active fishermen, with another 25% working in allied professions such as boat construction and fish vending. There is no reason to believe that fishermen's mobile phone use was any different than others within their communities. Furthermore, Indian mobile phone spectrum allocation (and therefore geographic location of cell towers)

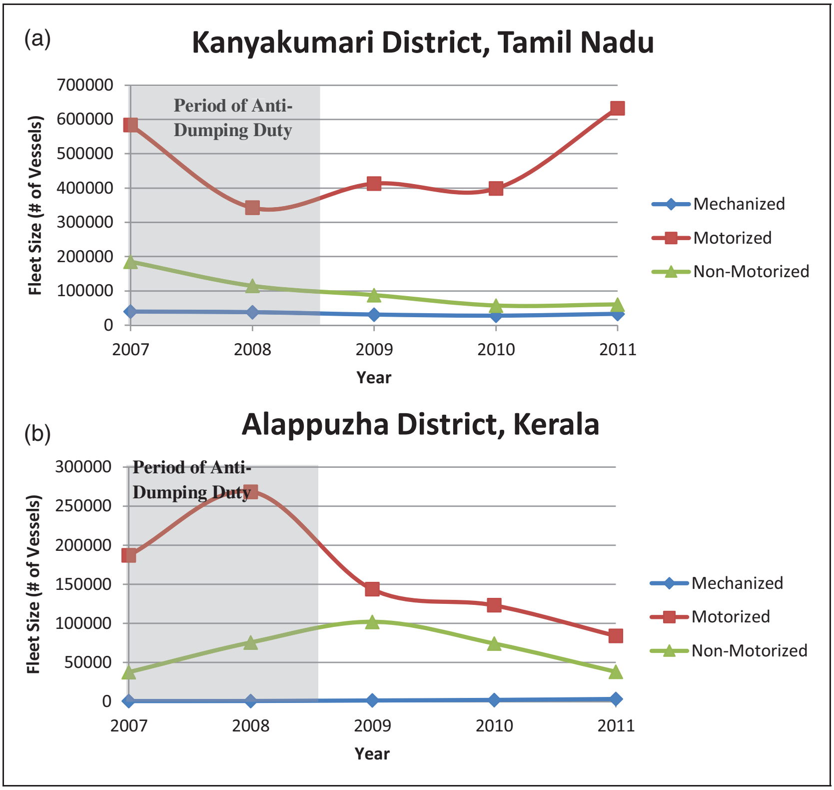


Figure 3. Fleet size in (a) Kanyakumari (TN) and (b) Alappuzha (Kerala) districts, 2007–2011.
Source. CMFRI.

was driven by political connections,[4] while individual phone use is an inexpensive capital investment (Doron & Jeffrey, 2013) or occasionally driven by extension organization donations (MSSRF gets award for fishermen mobile app, 2014). Therefore, mobile phone saturation is not dependent on community wealth or other potential explanatory variables.[5]

If the information access hypothesis is accurate, then the change in fleet size should be greater in districts with greater mobile phone saturation.[6] In addition, these districts should demonstrate greater shifts away from artisanal (especially nonmotorized) fishing toward more capital-intensive mechanized vessels to capture the newfound benefits of market access. This shift may occur through three distinct behavioral responses: (a) investment in mechanized vessels, if artisanal fishermen have gained funding via collective savings, subsidies, development aid, or loans; (b) previously artisanal fishermen working as crew members on mechanized vessels; or (c) increasing competition causes artisanal fishermen to exit the profession while outsiders invest in mechanized vessels. This study does not distinguish between these possible responses, nor can we differentiate based on the type of market information (e.g., price information, buyer interest, etc.) fishers derive from mobile phones.

To ensure that these outcomes were not driven by other factors, we control for other district-level characteristics. First, we assess availability of fish preservation infrastructure (ice factories, cold storages, and freezing plants) because these facilities should facilitate long-distance trade necessary to participate in export markets. Farmers who sell nonperishable crops (e.g., maize) seem to start with an advantage in global markets (Muto & Yamano, 2009). Therefore, districts with preservation facilities may already benefit from market shifts due to their ability to wait for preferred market conditions. As such, we expect that preservation facilities allow fishers to take advantage of improved market access and are therefore correlated with increased fleet size over this time period.

Second, we control for the number of educational institutions per family as a proxy for fishers' opportunities to adapt by changing livelihoods. Such opportunities should allow fishers to exit the profession when market access (and therefore price) is low, with some returning as market prospects improve. Therefore, we anticipate that fleet size will increase more over this period for those with greater access to education. Both infrastructure and educational measures are available from the 2010 Marine Fisheries Census.

Third, we control for changing catch per unit effort (CPUE) over this time period. CPUE serves as a proxy for ecosystem health because it provides a measure of the fish available in a given area. We expect that—all else equal—a declining fishery leads to less fleet investment, while greater fish abundance leads more people to enter the profession (Cinner et al., 2009), leading us to anticipate a positive correlation between exogenously driven CPUE change and changes in fleet size. This measure allows us to control for the impact of ecosystem health, as well as its effect on livelihoods, on subsequent fishing decisions (B, C, and D, in Figure 1). Unfortunately, CPUE data are not available for all observations, so we also present models without this measure for comparison.

Fourth, we control for state by using a dichotomous indicator to separate districts in TN (1) from those in Kerala (0). The state may impact fleet investment for geographic or political reasons due to differences in East and West Coast fisheries as well as state-level fishing regulations. We have no a priori expectation regarding the influence of state characteristics on fleet size shifts.

Finally, we control for craft type to assess whether fleet changes vary depending on the vessel type, which also usually correlates with wealth (e.g., nonmotorized craft require lower investment than mechanized trawlers). Kurien (1998) explains that small-scale fishers respond differently to globalization than industrial fleets do. Similarly, De Lopez (2002) notes that behavioral responses to market liberalization often vary by wealth, and Cinner et al. (2009) show that the poor have more difficulty mobilizing to shift professions so they may be stuck in the sector despite shifting market conditions. Shrivastava and Kothari (2012) further suggest that poorer communities cannot access the economic benefits of export market access in India. If the wealth explanation is accurate, then nonmotorized fishers will have the weakest—yet still positive—response to global market changes, while mechanized fishers will adapt to these conditions most directly. Alternatively, we may see different responses based on vessel type because increased investment may shift fishing from nonmotorized toward mechanized vessels. In this case, all fishermen within a district should respond to market changes, leading to a decline in nonmotorized fishing and a simultaneous rise in mechanized fishing, with little net response in the motorized sector as nonmotorized fishermen invest in motors and motorized fishers seek employment on new mechanized vessels. In addition to the full model, we also therefore partitioned the model to see if market responses differ depending on vessel type.

Table 1 includes summary statistics for all variables used in this study. Variance inflation factors (VIF) were calculated for each variable in each regression model to ensure that collinearity did not affect results. No VIF exceeded 3.5, well below the standard level of concern when a VIF exceeds 10. Nonetheless, preservation infrastructure and state account for the greatest level of collinearity among independent variables, with Kerala districts likely to have more infrastructure available. Therefore, all models were also examined when dropping these two variables independently. Direction, significance, and magnitude of results do not change appreciably, other than increased significance for the remaining one of these two variables.

To assess the information access hypothesis, we conducted multilevel and ordinary least squares (OLS) regressions. Table 2, which includes observations from all three vessel types, uses gllamm software with robust standard errors and adaptive quadrature in Stata 10 (Rabe-Hesketh, Skrondal, & Pickles, 2005). This multilevel model is necessary because multiple vessel types are present within each district, with some of our key explanatory variables measured at the district level (e.g., mobile phone saturation) and not varying by vessel type (Moulton, 1986). As a result, this analysis considers variables at two levels (i.e., vessel type and district). Table 3 includes separate OLS models for each vessel type.

Results

Results are presented in Table 2 (multilevel model, all observations) and Table 3 (OLS models differentiated by vessel type). Despite the small number of observations, mobile phone saturation has a positive effect on district-level fleet size changes during the period when U.S. import duties were removed. This effect is not statistically significant for the full multilevel Model 1 ($b \sim 1.64$, $p \sim .200$).

Table 2. Factors Predicting District-Level Fleet Size Change, 2007–2011 (multilevel models using gllamm with robust standard errors in STATA 10).

Variable	(1) Full model	(2) Without CPUE control variable	(3) Removed observations lacking CPUE data
Mobiles per family	1.64	2.37**	1.59*
	(1.3)	(1.1)	(1.20)
Preservation infrastructure facilities per village	0.21	0.47**	0.21
	(0.13)	(0.21)	(0.14)
Educational institutions per family	−35.2	−110.7**	−33.8
	(33.9)	(52.3)	(32.6)
CPUE change (2007–2011)	−0.016		
	(0.05)		
Motorized sector	0.65**	0.57**	0.65**
	(0.26)	(0.25)	(0.25)
Mechanized sector	0.67*	1.22*	0.68*
	(0.37)	(0.69)	(0.38)
Tamil Nadu	−0.14	−0.55	−0.13
	(0.32)	(0.48)	(0.32)
Constant	−0.97**	−0.82*	−0.97*
	(0.49)	(0.43)	(0.51)
Variance at Level 1 (Residual)	0.15	0.21	0.15
Variance at Level 2 (District)	1.00	1.00	1.00
Total variance	0.15	0.21	0.15
Observations (*N*)	50	56	50
Number of districts	21	21	21
Log-likelihood	−72.1	−105.3	−72.2

Note. SE in parentheses. CPUE = catch per unit effort.
$^{*}p < .1$.
$^{**}p < .05$.
$^{***}p < .01$.

However, the results indicate an increase in motorized ($b \sim 0.65$, $p \sim .012$) and mechanized ($b \sim 0.67$, $p \sim .069$) vessels relative to nonmotorized vessels over the period in question. This result suggests that there may be a shift from nonmotorized fishing to mechanized effort in these conditions, and Table 3 further explores conditions correlated with this relationship. Model 2, which does not control for CPUE, demonstrates similar results, with a larger coefficient and

Table 3. Factors Predicting District-Level Fleet Size Change, 2007–2011, by Vessel Type (OLS models).

Variable	(4) Nonmotorized	(5) Nonmotorized, without CPUE	(6) Motorized	(7) Motorized, without CPUE	(8) Mechanized	(9) Mechanized, without CPUE
Mobiles per family	−0.70	−1.58**	−0.42	−0.036	4.92*	8.32**
	(0.83)	(0.73)	(1.2)	(0.97)	(2.3)	(3.6)
Preservation infrastructure facilities per village	−0.047	0.068	0.26	0.23	0.47	1.09
	(0.19)	(0.16)	(0.29)	(0.25)	(0.60)	(0.95)
Educational institutions per family	5.97	12.5	17.7	19.8	−133.1	−336.2*
	(39.3)	(30.6)	(50.9)	(47.2)	(113.3)	(165.3)
CPUE change (2007–2011)	−0.023		0.047		−0.54	
	(0.07)		(0.08)		(0.90)	
Tamil Nadu	−0.72	−0.46	0.75	0.62	−0.14	−2.12
	(0.47)	(0.37)	(0.71)	(0.59)	(1.6)	(2.41)
Constant	0.25	0.41	−0.38	−0.42	−1.30	−0.74
	(0.52)	(0.48)	(0.78)	(0.73)	(2.0)	(3.3)
Power analysis for mobiles per family ($\alpha < .1$)	.27	.77	.122	.101	.8	.83
Power analysis for mobiles per family ($\alpha < .05$)	.16	.65	.063	.051	.67	.72
Observations (*N*)	15	19	20	21	15	16
R^2	.61	.56	.10	.08	.42	.47
Adjusted R^2	.40	.43	−.22	−.15	.10	.28

Note. *SE* in parentheses. OLS = ordinary least squares; CPUE = catch per unit effort.
*$p < .1$.
**$p < .05$.
***$p < .01$.

greater significance for the explanatory variable of interest, mobile phone saturation ($b \sim 2.37$, $p \sim .026$). Removing observations without CPUE data (Model 3) produces results more similar to Model 1, with a smaller coefficient and lower significance for the mobile phone saturation variable than in Model 2 ($b \sim 1.59$, $p \sim .084$). The similarity between Models 1 and 3 suggests that Model 1's limitations likely result from skipped observations rather than any substantive limitation of the CPUE control variable. Therefore, we report results both with and without this control variable for the vessel-specific models in Table 3.

To explore whether information access effects vary by vessel type, we partition the model. Table 3 presents separate OLS models for each vessel type. Due to the small number of observations for each vessel type (maximum of 21 districts), we also conducted post hoc power analyses to determine how much these regressions can tell us about the influence of mobile phones on fleet size. Nonmotorized vessel users *reduced* their active fleets as mobile saturation increased, though the result is not statistically significant when all controls are included in Model 4 ($b \sim -0.70$, $p \sim .42$). However, when additional observations are included by dropping the CPUE variable, the mobile phone coefficient is statistically significant while remaining negative ($b \sim -1.58$, $p \sim .048$). In contrast, mechanized fleet size *increased* significantly with mobile saturation ($b \sim 4.92$, $p \sim .066$; and $b \sim 8.32$, $p \sim .041$ without the CPUE control), while motorized fishing demonstrates a statistically insignificant decline with mobile phone saturation ($b \sim -0.42$, $p \sim .74$). This result is consistent with the expectation that nonmotorized fishing would decline and mechanized fishing would increase in the face of knowledge about market access, suggesting a decision to invest in more capital-intensive vessel types with knowledge of market changes. Together, these results suggest that information access may trigger a shift from nonmotorized to mechanized fishing as Kurien (1998) anticipates. Figure 4 compares the substantive relationship between district-level mobile phone saturation and fleet size changes in the nonmotorized and mechanized sectors, using Models 5 and 9.

Using CLARIFY software, we simulated the substantive impact of particular mobile saturation levels on fleet size changes by vessel type. These simulated fleet changes can then be used to simulate impacts of fishing changes on fish biomass. In the mechanized sector (Model 9), which has the greatest impact on fisheries, if every district had the lowest observed level of district mobile phone saturation among fisherfolk (~ 17%), the model predicts a 136% *decline* in fleet size over this period despite the strengthened export market. In contrast, if every district reached the highest level of mobile saturation (~92%) with other factors at mean levels, the simulation predicts a 478% increase in mechanized fishing fleets over this period. Nonmotorized fleets (Model 5) experience the opposite shifts under these conditions, with fleet size 4% higher under the low information scenario and 111% lower with high mobile saturation.

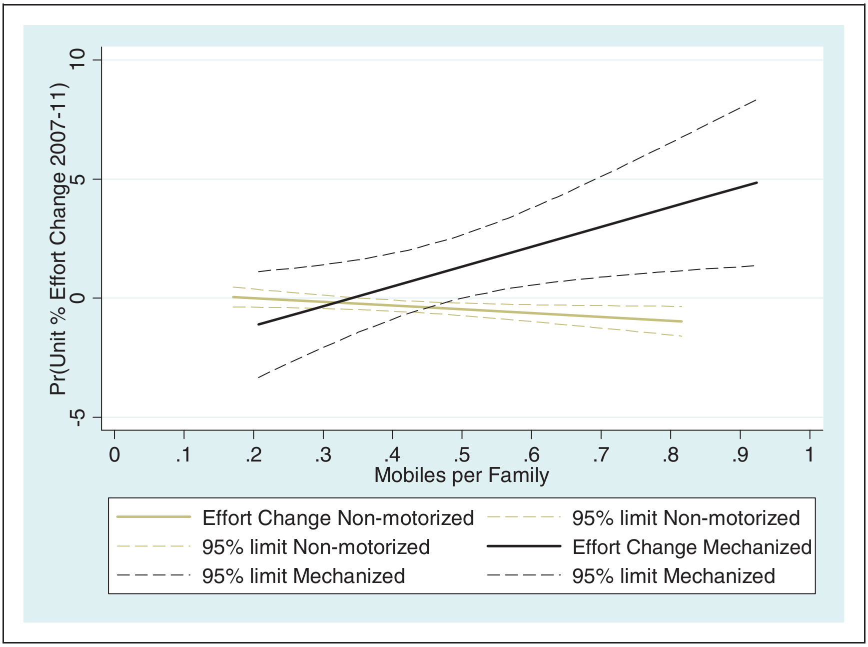


Figure 4. Predicted change in fleet size (2007–2011) for nonmotorized and mechanized vessels, based on district-level mobile phone saturation.

To understand how such fleet changes actually affect shrimp and other species, we used EwE to simulate how the Arabian Sea ecosystem near India's West Coast would respond to altered shrimping behavior. We used EwE, with a model developed using fishery-independent biomass data for the Arabian Sea (Mohamed, Zacharia, Muthiah, Abdurahiman, & Nayak, 2008) and used CLARIFY to examine the Kerala-specific (i.e., Arabian Sea coast) impacts of increased (92%) mobile phone saturation. In this scenario, nonmotorized fleet size declines 89% (i.e., to 11% of the previous fleet size), while the mechanized fleet expands 591% (i.e., 6.91 × previous levels). We then used EwE to explore how the expanded fleet would affect other marine life, both through bycatch and food web effects. First, we simulated what would happen if the fleet simply expanded to catch additional shrimp without other policy changes. In this model, changes take effect in Year 10. Most of the biomass pools stabilize within 10 years after increased mechanized fishing and decreased nonmotorized fishing (i.e., Year 20 overall), without other behavior changes (column 1 in Table 4, Figure 5(a)). The model suggests that this expanded shrimping fleet will result in substantial declines, including near depletion of benthic pelagic species, medium benthic carnivores and benthic omnivores, and greater than

Table 4. Simulated Ecosystem Effects After 10 Years of Increased Kerala Shrimp Fleets: Fishers Increase Mechanized Single-Day Fleet to Increase Shrimp Effort by ~591%, While the Nonmotorized Fleet Declines by 89%.

	No additional behavior changes	Fishermen able to target shrimp without bycatch
1. Marine mammals	0.56753993	0.873505592
2. Sharks	0.113387041	0.990224063
3. Skates and rays	0.339167267	1.173226833
4. Large pelagics	0.590129077	1.001246452
5. Tunas	1.099603295	0.792211235
6. Cephalopods	0.886235535	0.89329654
7. Large benthopelagics	0.035418235	0.946228921
8. Large benthic carnivores	0.529301822	0.987645507
9. Medium benthic carnivores	7.95E-08	0.959467053
10. Small benthic carnivores	0.851374388	1.033620119
11. Small benthopelagics	1.25430429	0.955374599
12. Mackerel	1.197243929	1.058695316
13. Clupeids	1.307017088	1.047351003
14. Anchovies	1.020685554	1.041407347
15. Crab and lobsters	1.237426877	1.02022624
16. Shrimps	0.77938658	0.800370038
17. Benthic ominvores	1.23E-19	1.002559423
18. Heterotrophic benthos	1.032538652	1.00316298
19. Meiobenthos	1.006228089	1.001912951
20. Micronekton	1.018155336	0.990528882
21. Largezoo	0.99256742	1.011546135
22. Microzooplankton	0.999395549	0.994382083
23. Phytoplankton	0.9982481	1.001868248
24. Detritus	0.99870187	1.000449538

40% declines among a number of other groups. These declines suggest that market liberalization and vessel type shifts will indeed trigger negative ecosystem responses when fishers have access to market information. In the age of information availability, such increased fishery exploitation may harm smaller fishermen and natural resources more generally (Doron & Jeffrey, 2013; Shrivastava & Kothari, 2012). At the same time, some species groups, including economically important clupeids and mackerel, may actually increase under these conditions as their predators decline.

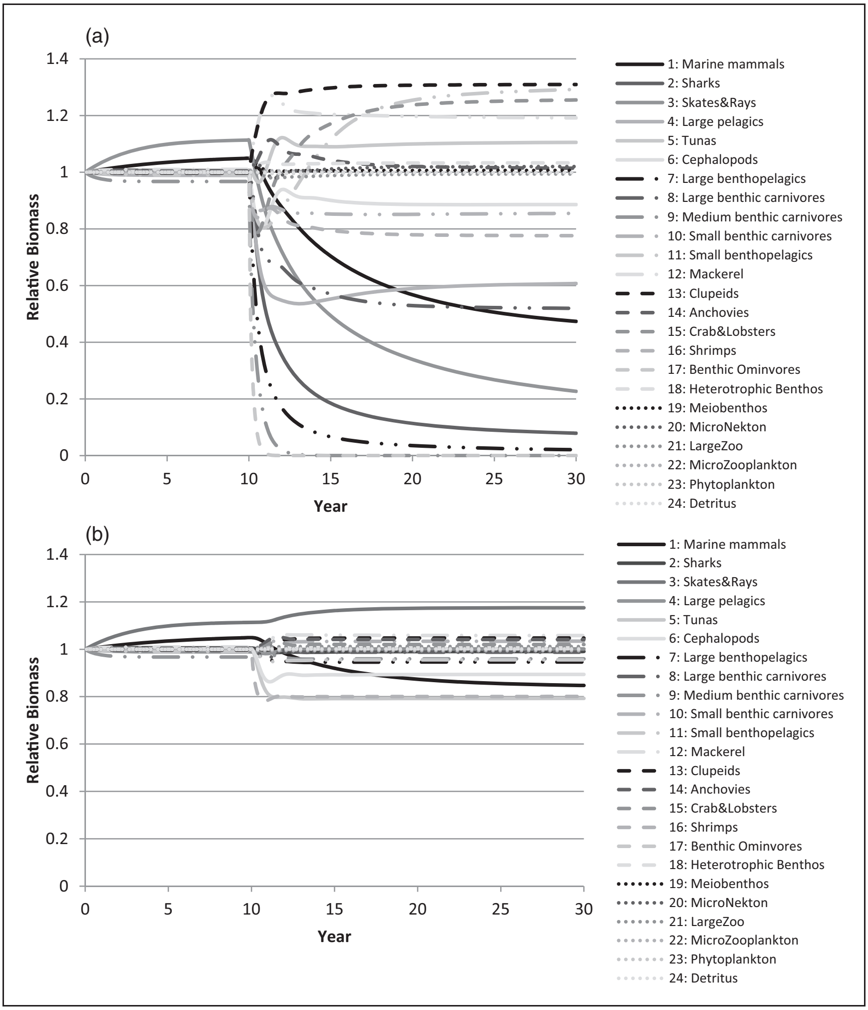


Figure 5. Simulated ecosystem effects of increased Kerala shrimp fleets: Fishers increase mechanized single-day fleet to increase shrimp effort by ~591%, while the nonmotorized fleet declines by 89% (a), with the ability to target shrimp (b).

We also simulate the same increased shrimp catch under conditions where shrimp can be specifically targeted by the fleet (i.e., when bycatch does not deplete other marine species). By using “more target oriented” gear, this scenario would allow successful fishermen to continue harvesting shrimp while limiting harm to other fishery resources (Kurien, 1998, p. 29). In this simulation, negative effects can be moderated by regulatory measures designed to limit ecosystem damage. As the second column of Table 4 and Figure 5(b) demonstrate, under these conditions, no species group declines more than 20% from its previous biomass. In addition, clupeids, mackerel, and other beneficiaries of predator removal remain advantaged—though by a smaller margin—in this scenario. Therefore, when fishermen can target the lucrative shrimp species for harvest without bycatch, we project that increased shrimp catch has little impact on other species while still allowing fishers to take advantage of shrimp market changes. As a result, trade liberalization could retain its positive income benefits while still yielding more benign ecosystem impacts, if bycatch can be limited through management efforts such as gear restrictions or bycatch reduction devices.

Discussion and Conclusion

Our results suggest that fishers respond to economic opportunities when they are aware of market changes. We demonstrate that the impacts of globalization are moderated by vessel type and access to information rather than uniformly increasing or decreasing resource use in the face of market liberalization. Districts with greater access to market information via mobile phones are more likely to increase fishing and shift from artisanal to mechanized fishing in response to globalization. Although our findings are consistent across multiple model specifications, they are limited by a lack of temporal variation in data collected for our key explanatory variable. Future studies could be strengthened by additional time series data collection.

Nonetheless, this outcome suggests that those who are endowed with technological resources are best positioned to gain from globalization in the short term. As Jensen (2007) and Abraham (2007) have similarly demonstrated, mobile phone use improves fishers' access to market information. As they both note, such information may lead to price convergence and reduced wastage. However, while reduced wastage may increase the amount of fish sold and thereby enhance fishers' livelihoods, we have no reason to believe that increased sales would lead fishermen to catch less if they are profit maximizing actors who wish to sell the additional fish rather than leaving them in the sea. Indeed, Jensen (2007, p. 914) demonstrates that mobile phones increased the quantity sold rather than decreasing the quantity caught. As a result, we have no reason to believe that reduced waste would balance out the ecosystem impacts of increased fishing because the waste reduction does not lead to reduced catch.

While resource users may obtain short-term benefits from fleet expansion, increased resource use may have detrimental long-term development effects if it negatively affects resource sustainability and leads to overcapitalization of fishing vessels or gear. Indeed, increased fishing capacity may lead fishers to put forth additional effort to recoup their investment (Kurien, 1998). In this case, expanded mechanized fleets could have substantial negative effects on the ecosystem beyond only increased shrimp exploitation, with the potential for significant declines in multiple species groups.

Nonetheless, local institutions may also condition the impact of external market shifts on natural resources (Rudra & Jensen, 2011). Li (2005), for instance, finds that trade expansion would lead to *dirty* economic growth in Thailand unless environmental measures were simultaneously pursued. Similarly, while trade liberalization may yield overexploitation in open-access fisheries, restricted access may lead to positive welfare effects of trade liberalization at the national level (Nielsen, 2009). Similarly, we find that when increased vessel investments are also matched with increased regulation, negative consequences may be mitigated. In particular, we demonstrate that bycatch reduction would increase the sustainability of many species groups in Indian coastal fisheries without damaging those groups (e.g., clupeids and mackerel) that may benefit from removal of predators. However, this type of regulation comes at a cost. Bhatta (2002) notes that "poor fisherm[e]n of India cannot afford to harvest shrimp by adopting turtle excluding devices" (p. 18). However, this option may become possible when offset by increased incomes or with government or foreign assistance. Our results therefore suggest that institutional and technological interventions can also moderate the negative impacts of increased production while still maintaining the short-term developmental benefits of information access and market liberalization. As such, information access coupled with regulation could provide a means to manage the impact of globalization on fisheries, increasing the social benefits of increased catch while limiting negative ecological and livelihood effects.

While our analysis supports the information access hypothesis, future research should also examine the regulatory measures that are already in place and under what conditions they have most successfully limited the negative repercussions of trade liberalization while maintaining short-term livelihood benefits. Other research demonstrates that fisheries rules are most effective at constraining adaptation strategies when they rely on a combination of state regulation and local norms (Novak & Axelrod, in press). Therefore, subsequent studies should consider what combination of policies minimizes negative ecological effects while maximizing fisherfolk livelihoods.

A third area for future research would be exploring the relationship between economic and ecological changes. Our results show a lack of response to ecological conditions such as CPUE change, despite fishers frequently stating that their fishing decisions are driven by fish availability. Thus, future studies should consider the market conditions under which fishers respond to fish stock abundance.

Finally, we have suggested three possible behavioral responses that would influence the shift from artisanal to mechanized fishing: (a) previously artisanal fishers investing in mechanized vessels, (b) previously artisanal fishers working as crew members on mechanized vessels, or (c) artisanal fishers exiting the profession. The available data do not allow us to differentiate between these explanations, but future studies should further address the underlying causal process. Qualitative analysis would complement this study by confirming how fishermen actually make decisions to increase or decrease effort in the face of global market changes.

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Notes

1. As one reviewer noted, exports may also be sensitive to exchange rate shifts, with the Rupee's value (relative to the U.S. dollar) much lower in late 2011 than in late 2007. However, despite exchange rate shifts, shrimp exports to the United States more closely track the emergence and removal of anti-dumping duties over this time period (Figure 2). We cannot control for more fine-grained exchange rate shifts because other data are not available on the same temporal scale.
2. We obtain similar results using 2007–2010 changes, though with lower significance levels for the nonmotorized sector, and substantively larger effects of mobile phone

saturation. Examining 2008–2011 changes results in the same signs and similar magnitude coefficients, but lower significance levels, likely because some fleet investments began as soon as the United States agreed to begin cutting duties in mid-2008.

3. The 2005 Marine Fisheries Census did not differentiate mobile phones from other *electronic gadgets*. As a result, data on mobile phone saturation are unfortunately limited to one observation per district during this period.
4. Political influence is often strongest in inland parts of these districts (Subramanian, 2009), limiting concerns about political influence as an omitted variable that could impact both mobile phone saturation and ability to invest in larger vessels.
5. Ideally, as one reviewer suggested, we would employ a difference-in-differences (DD) model to rule out omitted variable bias. Unfortunately, because we cannot observe temporal variation of mobile phone saturation, we cannot use the district fixed effects necessary for DD. Statistical remedies for this concern are not amenable to small sample sizes.
6. One reviewer asked whether fishing rates may have responded to mobile phone access, regardless of market changes. However, despite increasing rates of mobile phone adoption in previous years, fishing nonetheless declined in some districts during that time period. As such, mobile phones, by themselves, cannot be responsible for fishing increases. Instead, mobile phones must have an impact in conjunction with market shifts or some other information that fishermen acquire through these devices.

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