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Outcomes of gear and closure subsidies in artisanal coral reef fisheries

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Abstract

The outcomes of subsidizing natural and fishing capital were studied in nearshore coral reef fisheries initiated by the devolution of governance and management from national to county governments. One county promoted a net subsidy program of preferred mesh sizes with a distinct purchase and distribution date, while the other supported the ongoing maintenance of fisheries closures. This provoked a BACI design where standard fisheries statistics were measured 3 years prior to and 2.5 years after the net subsidy, or the life expectancy of these nets. Only ~50% of the purchased nets were utilized, indicating a replacement rather than addition of capital and low need. For example, net fishing effort did not change and there was an overall demographic change in gear use away from traps toward lower cost speargun. Net subsidized fisheries displayed a 9% increase in the mean length of captured fish but also a decline in catch-per-unit effort (CPUE) and personal incomes and no change in prices, yields, and per area incomes. In contrast, the fisheries surrounding closures displayed increased fishing effort and a 4% decline in fish lengths but increased CPUE, yield, incomes, and per area revenues. The net cost of maintaining closures was less than the gear subsidy purchase. Significant time × treatment interactions in all indicators support the conclusion that gear subsidies, apart from larger fish sizes, worsened the fisheries, while closures improved it. Increased recruitment rather than growth of fish appeared to be the mechanism for improvement.

KEYWORDS

community closures, East Africa, gear management, marine reserves, political ecology, spillover, western Indian Ocean

1 | INTRODUCTION

Achieving sustainable fishing by 2030 is one of the prime activities of the United Nations Sustainable Development Goals (United Nations, 2015). Yet, there are many ways to measure sustainability and manage fisheries (Halpern et al., 2012; McClanahan, 2018). For example, the stability of profits, yields, human employment, poverty levels, and ecological health all measure aspects of sustainability (Bell & Morse, 2012). Moreover, the complexity of fisheries and the veracity of sustainability evaluations are undermined by a variety of metrics and the poorly controlled nature of most large-scale fisheries management studies (Pauly, Hilborn, & Branch, 2013). Experimental management by plan or serendipity, on the other hand, provides a more rigorous way to evaluate management outcomes.

Many present-day fisheries fail to produce net incomes without various subsidies (Sala et al., 2018; Watson et al., 2013). Subsidies can include reducing the cost of capture but also by protecting stocks (Sumaila, Lam, Le Manach, Swartz, & Pauly, 2016). Subsidies have been described as (a) good subsidies that invest in protecting natural assets to optimize human welfare or as (b) capital subsidies that disinvest in natural capital and maintain fishing effort above maximum economic yields (Sumaila et al., 2010). Small-scale fisheries are estimated to receive 16% of global subsidies and 48% of them are considered beneficial, while 52% are either capital enhancing or ambiguous (Schuhbauer, Chuenpagdee, Cheung, Greer, & Sumaila, 2017). In contrast, only 28% of subsidies for large-scale fisheries are beneficial. A disparity in subsidies between small-nearshore and large-offshore or export versus non-export fisheries can lower prices and give off-shore and export fisheries a market advantage over small and unsubsidized local consumption fisheries. This can prompt governments to compensate by subsidizing nearshore fisheries but with consequences the may depend on the state of the stocks.

Small-scale fisheries produce high yields and support many of the world's poorest and most natural-resource dependent people (Schuhbauer & Sumaila, 2016; Teh, Teh, & Sumaila, 2013). Small-scale yields often come from diverse and ecological sensitive ecosystems such as coral reefs (Karr et al., 2015). For example, capture and storage technologies are commonly subsidized and lead to overexploited coral reef fisheries (Cinner et al., 2016). Fisheries closures can also be driven by tourism-based markets and undermined by low expenditures and weak governance (Cinner et al., 2018; Gill et al., 2017). Therefore, closures may also need government subsidies but the decision to spend on fisheries capital or the operational cost of closures is complex (Halpern, Gaines, & Warner, 2004; Nickols et al., 2019).

Fisheries closures can influence nearby fisheries but the net benefits are often difficult to empirically evaluate (Gaines, White, Carr, & Palumbi, 2010). Fisheries closure models often indicate larval and adult spillover can increase CPUE in growth overfished fisheries (Nowlis & Roberts, 1999). Recruitment overfished fisheries are more difficult to evaluate but some population models suggest that they can have long-terms benefits on yields and incomes (Nickols et al., 2019; Rodwell, Barbier, Roberts, & McClanahan, 2002; White & Kendall, 2007). Similarly, management of fishing gear is associated with increased natural resources and catch rates (McClanahan, 2010; McClanahan, Graham, MacNeil, & Cinner, 2015). Studies of fishing gear manipulations find competitive interactions that can challenge efforts to increase yields (McClanahan & Kosgei, 2018). These findings provoke the need to evaluate management choices in well-designed experimental contexts (Halpern et al., 2004).

Here, we studied a coral reef fishery where political and management power was devolved from the nation to counties (Cinner et al., 2012). Devolution can create divergence among management choices among neighboring governance jurisdiction. Thus, providing an opportunity to evaluate local decisions in a scientific way, especially when it is done in the context of ongoing monitoring of resources. This study evaluates two neighboring coastal counties that followed different fisheries management paths (Figure 1). One county, Kilifi, maintained national and community governed no-take fisheries closures. The second county, Kwale, instituted a subsidy of gear to lower the cost of net fishing. To test the impacts, we monitored changes in fisheries metrics 3 years before and 2.5 years after the purchase of nets.

2 | METHODS

2.1 | Historical background

Governance of the Kenyan coast and these two counties has undergone a number of political changes associated with the effort to resolve local versus national governance issues (Bienen, 2015). Informal assessments of the fisheries status in Kwale during the early 1990s indicated a need for fisheries restrictions. Subsequently, nationally protected areas were proposed and legally gazetted in Kwale in 1994 but conflicts over governance authority led to nonenforcement (McClanahan, 2007). Subsequently, the national fisheries service implemented gear restrictions in 2001 by eliminating small-mesh drag nets within tourist locations. This led to some recovery in catch rates but an eventual leveling (McClanahan, 2010). The increased power of the county governments was established by Kenya's 2010 constitutional change and implemented in 2013 after the 2012 elections. Subsequently, Kwale county implemented a net purchase and distribution program where landing sites received nets until September 2014.

Kilifi county contained two successful no-take management systems prior to and after the constitutional and governance changes. One was a ~6-km^2 no-take closure and a marine reserve (multiple use) managed since 1991 by the national park service—Kenya Wildlife Services. This national park was created in a heavily fished tourism area, underwent large recovery in fish biomass but, after political conflicts until 2012, experienced declining visitation and revenue. Kenya's first community managed closure, Kuruwitu Community Conservation and Welfare Association, a 0.3 km^2 no-take established in 2007 was an early effort to support local control of fisheries resources. Despite some early resistance, this closure has been stable and experienced a modest recovery of fish biomass (Cinner &

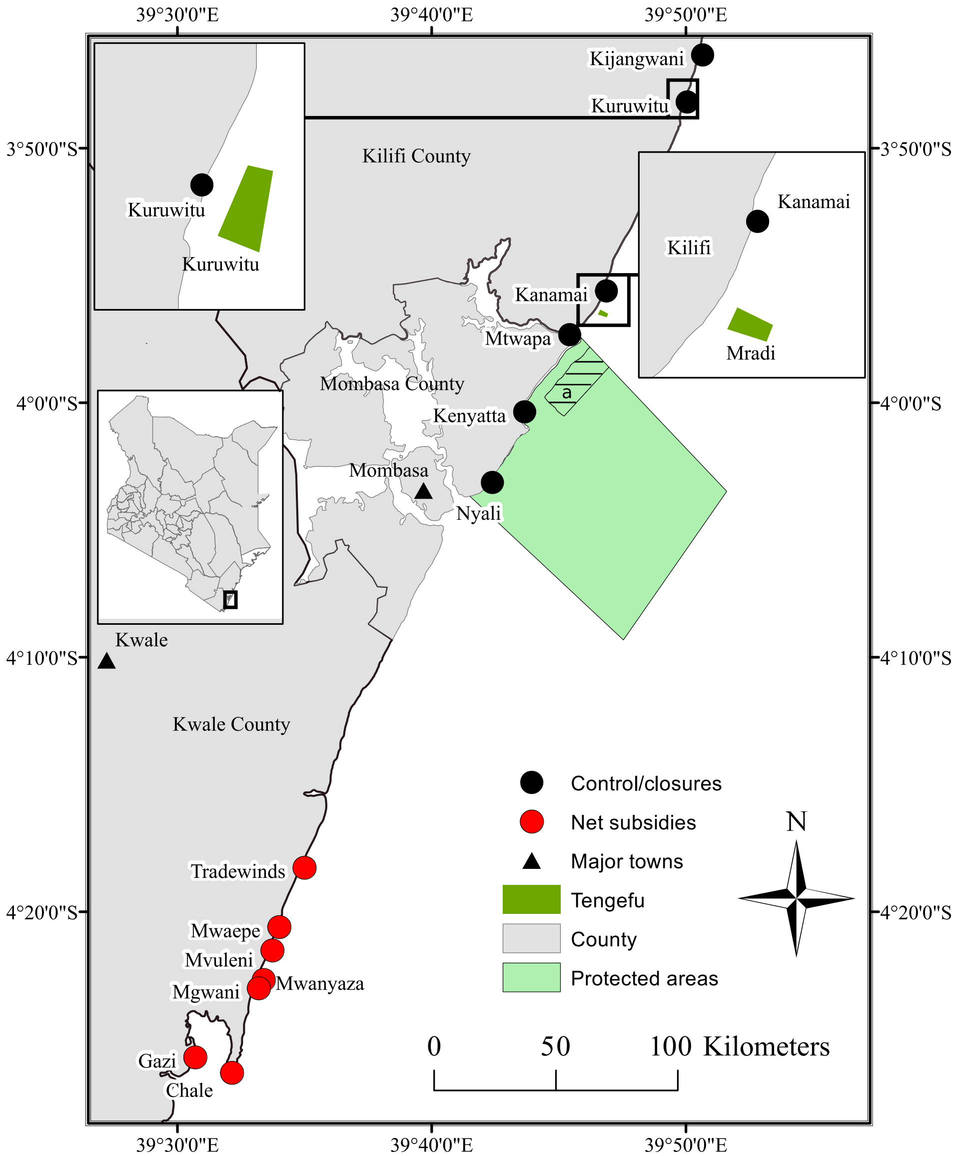

FIGURE 1 Map of studies control/closures and net subsidies sites in Kilifi and Kwale counties. Mombasa marine protected area has a no-take zone (*a) within an area-based management system. *Tengefus*' are community managed closures

McClanahan, 2015). Additionally, there were a few additional small community closures namely Bureni and Mradi but their management was weak and their status suggests minor increases of fish biomass (McClanahan, Muthiga, & Abunge, 2016). Prior to and during this period, we made regular measurement on fishing gear and catches among a number of the landing sites of both counties. Details of these coral reefs and study sites have been presented in a number of papers (Maina et al., 2015; McClanahan & Kosgei, 2018).

2.2 | Experimental design and site selection

We evaluated fish landing from 16 possible monitored sites in both Kwale and Kilifi counties that had regular catch measurements over the January 2010 to May 2017 time period. We monitored Kwale county sites that included the following seven landings sites, Tradewinds, Mwaepe, Mvuleni, Mwanyaza, Mgwani, Chale, and Gazi (Supplementary Information). In order to achieve a balanced design with appropriate "controls" with a comparable before and after impact for Kwale county, we evaluated Kilifi for similarities with Kwale sites based on the inputs of effort and gear diversity. Based on the Ward and ANOVA similarity methods (JMP statistics, Sall, Lehman, & Creighton, 2001), we identified six comparable landing sites in Kilifi, namely Kijangwani, Kuruwitu, Kanamai, Mtwapa, Kenyatta, and Nyali. All of these sites were < 10 km from one of the closures (Figure 1). We estimated the fishing grounds of our study sites in Kwale County to cover an area of 26.8 km^2 and 36.1 km^2 in Kilifi county (Supplementary Information). These represent 22.6% and 20.1% percent of the total fishing grounds of these counties (Supplementary Information).

The before and after component aspect of the study was based on a survey to determine the time when nets were received and the duration of net use before they were abandoned. We established the beginning date of September 2014. We found that nets last for 2.8 ± 0.4 years and therefore set the after period October 2014 to May 2017. Therefore, we base our evaluations and conclusions on a before and after and "control" (Kilifi) and impact (Kwale), or BACI design.

2.3 | Field data collection

We used two methodologies to sample fish catch. The first resulted in calculations of catch per unit effort (CPUE, kg/fisher/day), yields (kg/km^2/day), and income (Kenya shillings/fisher/day). The second method evaluated the body lengths of the catch. We used experienced fisheries data collectors employed by our program for both methods. We established the areas of the fishing grounds based on conversations and observations of usage at each landing site, whereas fishing effort and incomes were based on regular monthly sampling over time. In the first method, we visited landing sites 2 to 5 times per month, and we weighed and priced fish by the dominant taxa or groupings that fishers used to sell them. These groupings were goatfish, parrotfish, octopus, scavengers (Haemulidae, Lethrinidae, Lutjanidae), and a mixed group that included a diversity of coral reef fishes that had low market value. These fish are consumed locally between fisher families and consumers living in nearby villages. Some fish are transported to nearest city in Mombasa county that lies between these two countries. Octopus catches were not included in evaluations as their effort responds more to international prices (Wamukota & McClanahan, 2017). These factors and decisions reduced possible complexities of the market dynamics that could potentially influence fishing effort and profits.

We weighted landed fish to the nearest 0.5 kg and the number of boats, fishers, and the gear used for each recorded catch measurement. We analyzed the common demersal fish and invertebrates and not the pelagic and offshore catches, as they did not come from the habitat and specific gears being studied. We estimated the numbers of fishers for CPUE, as it was more stable and reliable than boats and gear metrics of effort that was less predictable (McClanahan & Kosgei, 2018). We estimated the number of nets distributed and used in net subsidies sites by asking our trained data enumerator and the county clerk (Supplementary Information). In the second method, we visited landing sites one to four times per month, and randomly sampled a subset of the fish from different gears and identified individuals to the genus or species and measured their total lengths to the nearest 0.1 cm. We collected price data for each catch group during sampling visits calculated averages to get monthly prices. Then, we estimated monthly and yearly prices and incomes of fishers based on their daily catch rates pooled per month.

In estimating cost and benefits, Kenya Wildlife Service (KWS) and Kuruwitu Conservation and Welfare Association (KCWA) provided us with the annual revenues and costs data for Mombasa park and Kuruwitu *tengefu*, respectively. We obtained KWS revenues from entrance fees, but expenditures were more difficult to estimate because of the national nature of the institution and accounting system. Nevertheless, we asked the head Warden to estimate annual expenditure for Mombasa park based on their 5-year budget plan. Additionally, we obtained the cost of net purchases from the County government of Kwale.

2.4 | Statistical analyses

We tested the raw daily and pooled monthly length, CPUE, yield, and income data for normality with the Kolmogorov-Sminorv tests of normality. Normality tests failed and, therefore, we employed Kruskal-Wallis nonparametric analyses tests for differences by dominant fishing gears before and after periods in controls/closure and net subsidy sites. We undertook Kruskal-Wallis statistical analyses with JMP Software (version 12.0) (Sall et al., 2001). Monthly pooled CPUE, yield, income, and raw fish length passed normality test after Box-Cox transformations. This allowed two-way ANOVA tests of significance of treatment, time, and period for CPUE, yield, income, and fish lengths. After Bonferroni correction, we used post-hoc Dunn's tests analyses between pre and post periods and gear type to test for statistical differences (*R* package version 4.2; https://CRAN.R-project.org/package=PMCMR) and *R* Package reshape v 0.8.7. We displayed trends of effort, CPUE, yield, and income with R package version 2.2.1; https://CRAN.R-project.org/package=ggplot2). We plotted changes in CPUE, yield, and income per total and individual gear effort in *R* package ggplot2 v 2.2.1; https://CRAN.R-project.org/package=ggplot2). We adjusted prices and incomes for the general level of inflation in Kenya based on the national consumer price index from the 2013 World Economic Outlook and 2010 as the base year (IMF, 2013).

3 | RESULTS

3.1 | Gear distributions

A total of 620 nets were procured by the Kwale County government to subsidise fishing in 14 Beach Management Units (BMUs) (Supplementary Information). Purchased nets were then distributed to the county fisheries offices by September

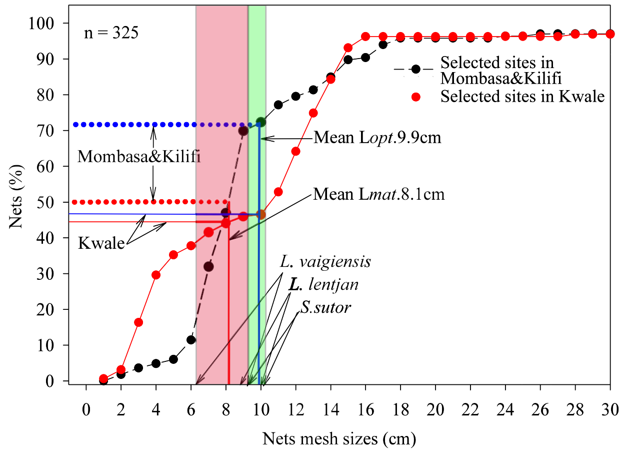

FIGURE 2 Cumulative frequency of mesh sizes in the study sites of Kwale (net subsidy) and Kilifi (closures) counties in relationship to estimated optimal mesh sizes of the dominant 3 taxa in the catch. Red band represents length at maturity while green band represents size at optimal yield ranges

2014 and subsequently to BMU landing sites of which a subset was monitored by Wildlife Conservation Society (WCS) employees. County Fisheries offices recorded 320 nets and WCS landing site enumerators recorded 85 nets in use. The seven landing sites closely monitored here in four BMUs showed 106 nets recorded by the County Fisheries office and 55 nets by WCS enumerators.

Of the total nets procured by the Department of Agriculture, Livestock and Fisheries Kwale County office, 300 nets could not be accounted for at the County Director of Fisheries office. Two hundred and thirty-five nets could not be accounted for as per WCS enumerators. Moreover, 51 nets were not accounted for between Fisheries offices and WCS enumerator for the same landing sites. Net mesh sizes at the end of the study indicate that control site meshes were variable and estimated at 9.88 ± 6.53 cm (*SD*) and not statistically different from the net-subsidy sites of 9.65 ± 7.11 cm but their frequency distribution patterns differed (Figure 2).

3.2 | Changes in fishing effort, lengths, yields, and incomes

Changes in fish catch statistics over the study period showed high variability for each gear type (Figure 3). Fish prices had less monthly variability but fluctuated yearly with major inflation-adjusted depreciations in early 2011, followed by a rise, and another depreciation in mid 2015. In general, the inflation-adjusted prices depreciated after 2013 with the highest prices recorded in the experimental sites in early 2013 and in controls in 2014. Thus, the net subsidy was associated with stable prices and did not appreciably reduce fish prices as expected.

Effort: The total fishing effort at the net subsidy sites was high at ~6.3 fishers/km^2/day and constant across time and the subsidization event (Table 1, Figure 4, Supporting Information). The effort was ~25% lower at ~5.0 fishers/km^2/day in control sites but increased by ~12% after 2014. Two-way ANOVA results indicated that effort was significantly influenced by the treatment, time, and their interaction. At the individual landing sites, there were no changes in effort in the net subsidy sites but some increases in the control sites, namely Kanamai and Kijangwani. Spear fishers increased significantly in both treatments during the postperiod by 12% in net subsidy and 20% in closures sites. The number of trap fishers declined significantly by 51% in net subsidy sites and resulted in a significant time × treatment interaction.

Fish lengths: Standard fish lengths of all species combined significantly increased by 9% in net subsidy and declined by ~4% in closure sites resulting in a significant treatment × time interaction term. There was high between-site variability with a number of closure sites displaying significant increase in length, including Kanamai, Kuruwitu, and Kijangwani but lengths significantly declined by ~11% in Mtwapa—a fishing area reliant on small-meshed seine nets. In net subsidies sites, fish lengths generally increased except in Tradewinds. Two-way ANOVA analyses by gear showed all the before and after and interaction terms were significant—largely due to increases in lengths in the net subsidy sites relative to declines or lack of changes in control sites.

Catch per Unit Effort (CPUE): Mean CPUE decreased significantly in net subsidies sites by 9% but increased by 15% in closures sites during the postperiod. The interactions between time and treatment was significant and reflected the decrease from 3.9 to 3.6 kg/man/day in net subsidy sites and increase from 3.3 to 3.9 kg/man/day in closure sites (Table 1). These overall results were consistent with changes in CPUE at site levels. In net subsidy sites, there were minor and nonsignificant changes with the exception of Mwaepe, which showed the largest 23% decline. In closure sites, there were significant increases in CPUE in a number of closure sites including Kenyatta, Kanamai, Kuruwitu, and Nyali. Evaluations of CPUE by gear showed no changes in handlines in both treatments but that nets and traps declined in the net subsidy sites, while closures showed large increases of 39% and 53%, respectively. CPUE of spears increased in net subsidy but did not change in the closure sites. Thus, interaction terms were strongly significant for nets, traps, and also spears.

Yields: The per area yields in net subsidy declined insignificantly by 11%, while it increased significantly by ~29% (15.4 to 21.8 kg/km^2/day) in closure sites across the subsidy period. In individual subsidy sites, changes were uncommon

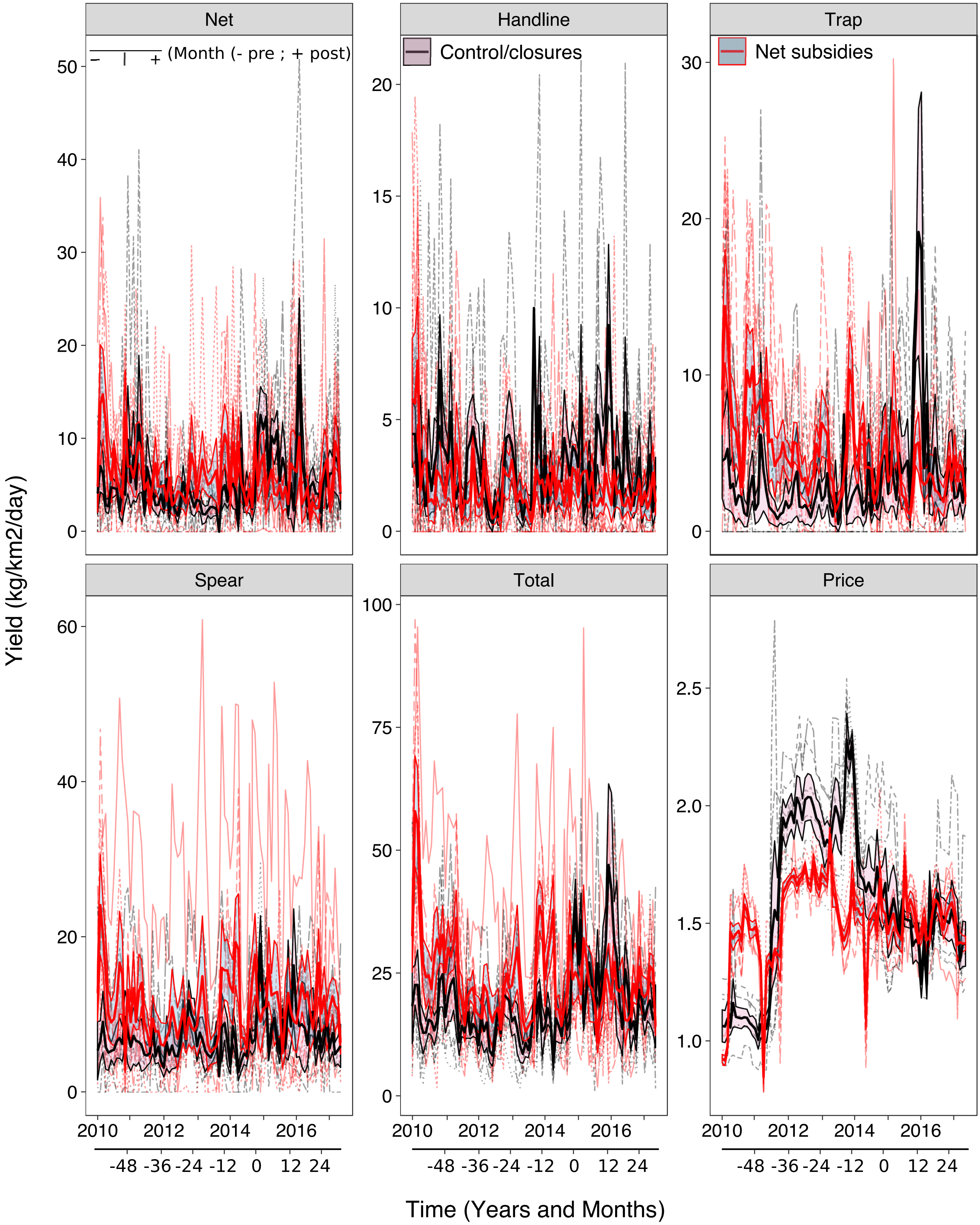

FIGURE 3 Line scatter frequency trends, means, and *SE* ribbons for yields (kg/km^2/day) during the study periods for dominant fishing gears and prices in $US in control/closures and net subsidies treatments. Second *x*-axis represents time in months; *− (negative) months is pretime period, *+ (positive) is posttime period, and *0 is the time zero when the nets were subsidized

TABLE 1 Summary of fisheries catch statistics in net and closure subsidy sites

Treatment	Time period	Fishers[a]	Fish lengths	CPUE	Yield	Income/fisher	Revenue
Net subsidies	Pre	6.27 ± 0.11	18.62 ± 0.10	3.92 ± 0.08	24.83 ± 0.82	5.85 ± 0.12	35.85 ± 1.08
	Post	6.33 ± 0.14	20.41 ± 0.12	3.60 ± 0.09	22.06 ± 0.84	5.46 ± 0.15	32.67 ± 1.16
	ChiSquare	2.98	176.88	6.79	1.78	4.97	1.17
	p value	NS	<.0001	0.009	NS	0.03	NS
Control/closures	Pre	4.77 ± 0.08	21.81 ± 0.21	3.32 ± 0.08	15.38 ± 0.52	5.39 ± 0.15	25.01 ± 0.92
	Post	5.41 ± 0.12	20.96 ± 0.18	3.92 ± 0.12	21.80 ± 0.96	5.95 ± 0.20	32.99 ± 1.52
	ChiSquare	29.32	24.11	21.56	38.18	7.53	24.75
	p value	<.0001	<.0001	<.0001	<.0001	0.006	<.0001
Two-way ANOVA	Treatment[control]	−8.52; <.0001	10.60; <.0001	−1.89; NS	−6.24; <.0001	−0.75; NS	−5.33; <.0001
	Period[post]	3.65; 0.0003	3.95; <.0001	1.51; NS	3.11; 0.002	0.97; NS	2.81; 0.005
	Treatment[control] *period[post]	2.38; 0.02	−11.07; <.0001	5.50; <.0001	5.57; <.0001	3.49; <0.0005	4.73; <.0001

[a]Estimates of total number of fishers (fishers/km^2), fish lengths (cm), catch per unit effort = CPUE (kg/fisher/day), yield (kg/km^2/day), income per fisher ($US/fisher/day), and revenue ($US/km^2/day) with the ±*SE*s and Kruskal-Wallis tests of significance for sites in control/closures and net subsidies per gear type. Two-way ANOVA summaries (*t* ratio; Prob>|*t*|) evaluating the effect between treatments, time and, treatment × time interaction for the dominant fishing gears presented.

with the exception of a 17% decline in Chale. In closure sites, yields generally increased significantly with highest increase in Kijangwani by 50%. Evaluations of yields by gear showed a 57% yield decline in traps in net subsidy sites but increases in nets, traps, and spears in closure sites. Except for handlines, treatment × time interactions were significantly positive for each gear and all gears combined.

Plots of change in yields as a function of fishing effort indicate a number of consistent differences between net subsidy and closure sites (Figure 5). For example, only three of the seven subsidy sites had yields slightly above the zero-change threshold, while five of the six closures sites exhibited net increases in yields by nets over the subsidy period. Spear yields were quite variable. Most handline yields did not change and the two most positive changes were in closure sites. None of the net subsidy sites but all trap and total yields in closures sites were above the zero-change threshold.

Fisher Income: The mean inflation adjusted prices before and after were 1.50 (± 0.24 *SD*) and 1.52 (± 0.15 *SD*) $/kg in the net subsidy and 1.62 (± 0.46 *SD*) and 1.52 (± 0.23 *SD*) $/kg in the closure sites. Fisher's daily income declined significantly in the net subsidy sites by 7%, while it increased significantly in closures sites from 5.39 to 5.95 $USD/day, resulting in significantly negative interaction terms. In net subsidy sites, there were significant declines by 22% and 20% in Chale and Mwape and a 15% increase in the Mgwani landing sites. Evaluations of individual closures sites indicated no significant changes in income between time periods. Evaluations of income by gear in net subsidy sites found incomes declined for nets and traps by ~15% but increased for spears by 10%. In closure site, incomes increased for nets and traps by 36% and 49%, respectively. Therefore, interactions between treatment and time were significant for nets, handlines, traps, and all gears combined.

Per area revenue: In general, the per areas revenue declined non-significantly by 9% in net subsidy sites, while it increased significantly in closures sites by 24% (25.0 to 33.0 $USD/km^2/day), producing a significant interaction term. In net subsidy sites, there were few significant changes except for a decline in in the wealthiest landing site, Chale, from 66.2 to 52.2 $USD/km^2/day. Revenues increased in all of the closure sites with the exception of Mtwapa. Net subsidy sites showed no changes in nets and handlines but per area decline in trap revenue by 52% and a 6% increase in spear revenue. Evaluations of revenues by gear in closures showed significant increases in traps, nets, and spears by 40%, 37%, and 13%. Interaction terms were significant for nets and traps and all gear combined.

Cost–benefit analysis of closures and fishing: The average annual revenues for the Mombasa Park in Kilifi County included an estimated annual revenue of $US 154,398 with an estimated expenditure of $US 160,000 between 2013 and 2017 (Supplementary Information). Yearly revenues and expenditure for Kuruwitu Conservation area were estimated at $US 1,155 and $US 1,092, respectively. The estimated total expense for procuring nets in Kwale country was $US 131,400.

4 | DISCUSSION

Changes in governance and fisheries metrics indicate high variability and complexity of interactions within artisanal

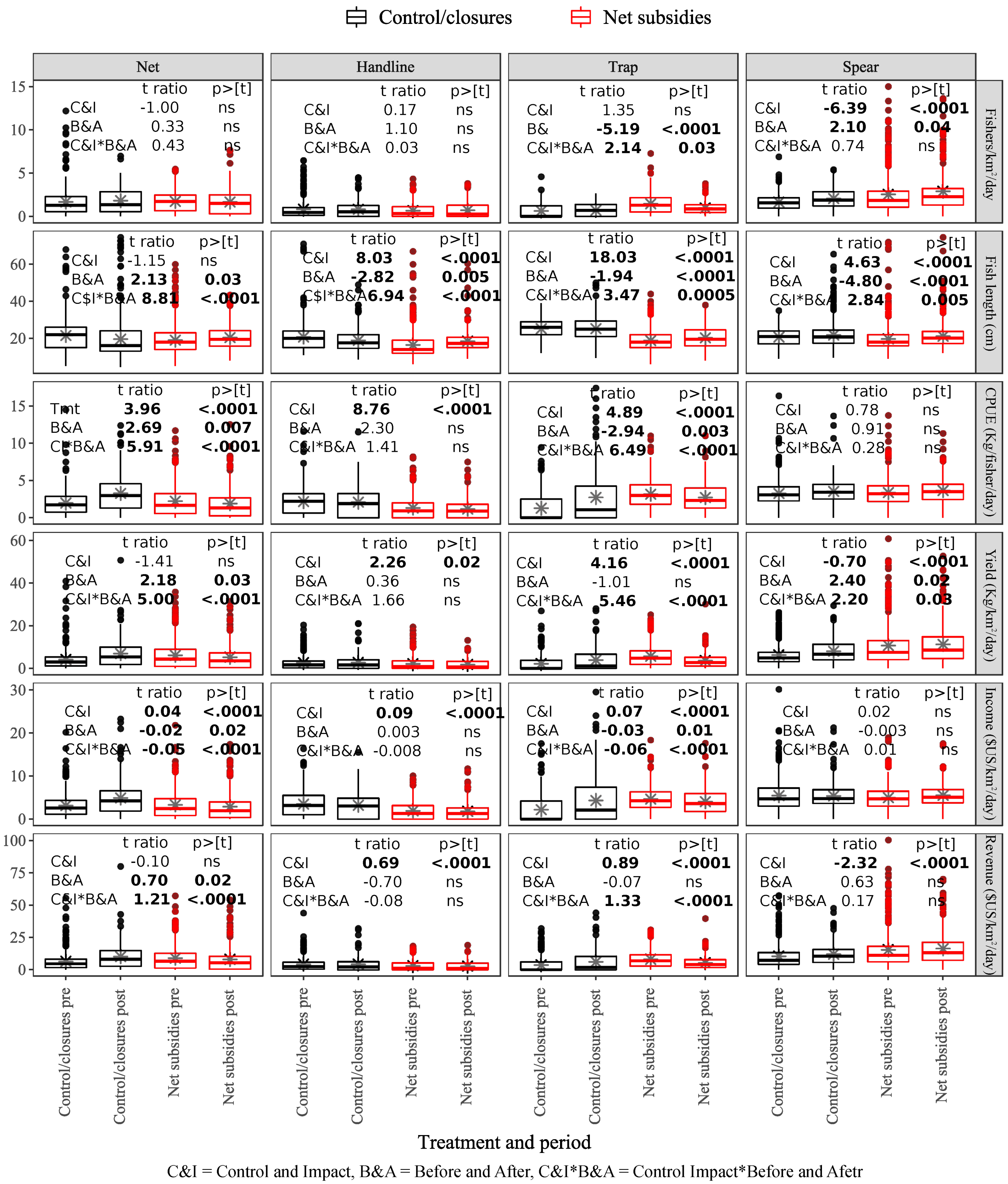

FIGURE 4 Box plots of the average mean and median of fishers per square kilometer, catch per fisher per day (CPUE), yield per day, and income per day in the control/closures and net subsidies sites for the period before and after the net subsidies for the dominant fishing gears. Letters above plots indicates significance, where box plots sharing the same letters are not statistically different from each other (NS = no statistical significance between period and treatments)

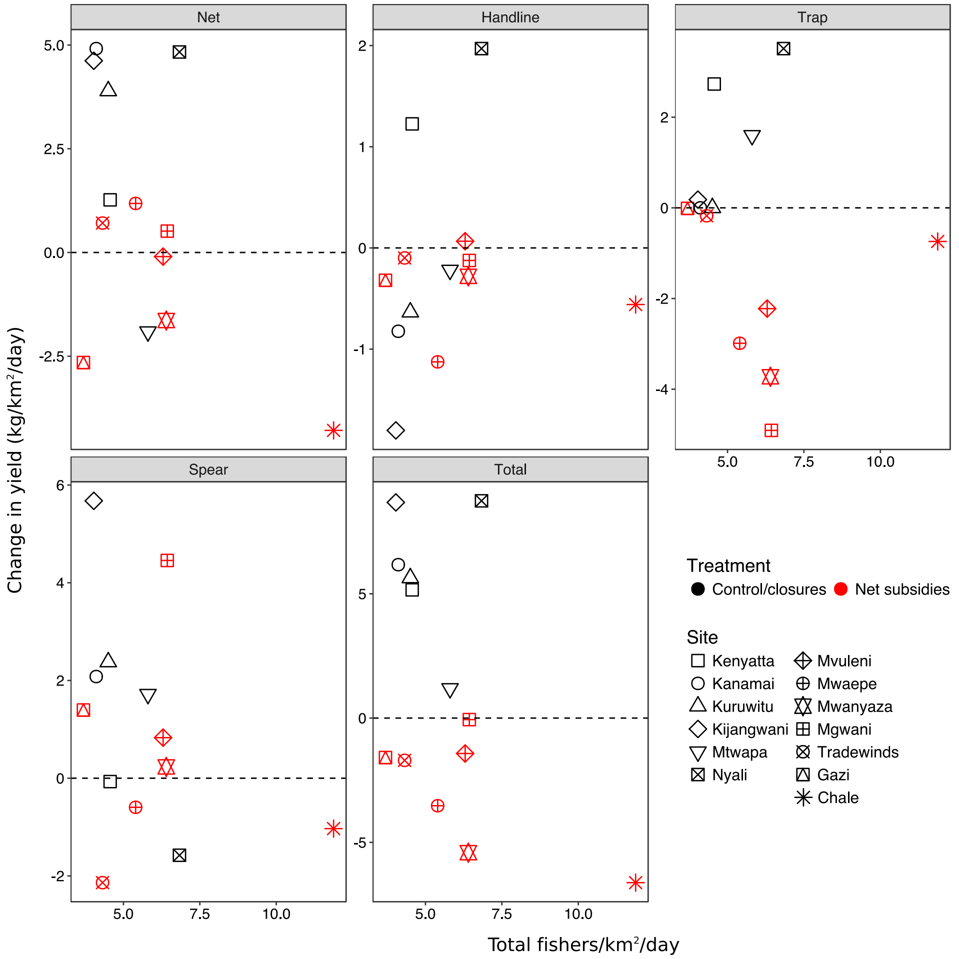

FIGURE 5 Changes (difference between pre and post) in yields per day against the total effort per square kilometer for individual sites in the control/closures and net subsidies sites for the four dominant fishing gears and the total of all gears pooled

fisheries. The complexity occurs in terms of site variability, the variable responses among gears, and outcomes of closures versus uptake of subsidies and reducing fishing costs. The net subsidy alone did not produce a large number of changes; rather, modest uptake and broad-scale demographic change in gear use along with site-specific declines in yield statistics. Uptake of free nets was modest and did not cause an increase in net fishing, declining prices, and increasing income as might be predicted. Rather, it was associated with declining trap use and modest increases in spear fishers.

Given the stability of effort and patchy declines in many metrics, it seems reasonable to assume that cost–benefits of net fishing could not improve in this market. Therefore, reducing costs created an incentive to replace rather than add nets and possibly remain in a fishery becoming dominated by spearguns. Many nets were not absorbed by fishers and net numbers depreciated along the transfer chain from procurement to usage. After the subsidy, the net income of fishers declined 15%, which was probably similar to the reduced costs of purchasing their own nets. Because Kenyan fisheries have low stocks and are exploited beyond maximum sustained yields (MSY), fishers are expected to replace old rather than adopt a new or add gears (McClanahan, 2018; Samoilys, Osuka, Maina, & Obura, 2017; Tuda & Wolff, 2015). Consequently, the motivation for subsidizing nets may have included social considerations other than improving fisheries incomes and yields.

An increase in fish lengths in the net subsidy county was among the few changes in their yield metrics. This contrasts with closure sites where there were small declines in lengths in some sites but an increase in CPUE and yields. Consequently, it may be that gift nets replaced some of the smallest mesh nets and therefore increased the mean length of capture, thus increasing natural capital. Thus, gift nets are an example of an ambiguous subsidy because they played some role in building natural capital rather than just lowering fishing costs. Nevertheless, the inverse relationship between lengths and yields in the two counties suggests that low recruitment rather than capturing sub-optimal size fish or growth overfishing accounts for the declining yields. Thus, it is likely that closures enhanced the recruitment of fish and reduced recruitment overfishing. In contrast, net management did not fully end growth overfishing despite increasing fish lengths. Among the dominant catch species, growth overfishing is evident and rabbitfishes and emperors show signs of recruitment overfishing (Hicks &

McClanahan, 2012). Overall, the findings here suggest that fish recruitment is more limiting than size-at-capture and that, for the purpose of recovering yields, closures are a greater need than increasing capture size. Net replacement of smaller by larger mesh sizes may be a form of natural capital subsidy but only if recruitment is not limiting the fishery.

One of the weaknesses of this study is the lack of systematically collected mesh size information prior to the subsidy program. A study back-calculating mesh sizes from length at capture suggests that prior to 2008 mean mesh sizes in both counties were smaller at 6.3 cm compared with the ~9–10 cm found in 2017 (Hicks & McClanahan, 2012). Consequently, larger recommended mesh sizes appear to have been adopted since 2008 resulting in an increase in length-at-capture (McClanahan, T. unpublished data). Yet, this increase has not compensated for recruitment overfishing—likely due to the failure to establish closures in Kwale county (McClanahan, 2007). The study does, however, indicate high variability in mesh sizes, indicating patchy responses to recommendations for increased mesh sizes in both counties.

A number of studies indicate competition among gears that should produce changes when gears are promoted, eliminated, or modified (Mangi & Roberts, 2007; McClanahan, 2010; McClanahan & Kosgei, 2018). Competitive effects were, however less evident because fish lengths increased among all studied gears, mesh sizes were variable, and fishing effort was stable in the subsidy sites. In contrast, a decline in trap fishing effort, their CPUE, and income was associated with the increase in spear fishing effort, CPUE, and income in the net subsidy county. The simplest explanation is that trap fishers began spear fishing. However, there are age and capital investment differences in spear and trap fishers—spear fishing often being undertaken by younger men having low entry and trap fishing undertaken by older men with higher entry costs (Mangi, Roberts, & Rodwell, 2007). Therefore, it is unlikely that the same people switched their gear use. Moreover, spear fishing effort, yields, and per area revenue increased in the closure county but without an increase in fish lengths, CPUE, and incomes. Thus, it is more likely that these patterns of gear use represent an overall demographic age and capital investment toward younger fishers with lower entry costs. Net subsidies may, therefore, be seen as a means to subsidize older fishers being slowly excluded by younger and low-capital fishers.

The lack of strict paired controls may be seen as one of the weaknesses of this study. In fact, there was no active government-induced management change in our closure sites during this time; but, rather management established prior to the net subsidy in Kwale county (McClanahan, 2010). Closure impacts are expected to have long-term responses often associated with the slow process of fish population recovery (McClanahan, Graham, Calnan, & MacNeil, 2007). Thus, the control site had the benefit of being able to evaluate a closure management subsidy. This turned out to be useful in that it allowed us to evaluate the different outcomes for management recommendations for recruitment versus growth overfishing—closure impacts versus increased capture size where effort did not change greatly. The result was a consistent set of significant interaction terms in the fisheries statistics that produced evidence for recruitment limitation. Except for fish lengths, all catch metrics indicated that subsidizing closures improved fisheries indicators relative to lowering the costs of nets and increasing capture size. Particularly notable were the 24% and 29% increases in yields and revenue in Kilifi. This contrasts with the increased costs and the lack of yield changes in Kwale. Even during this period when tourism had waned, the net economic costs of managing the closures were considerably less than net purchases. The results support the value of closures in enhancing recruitment where fisheries stocks have been reduced (Nickols et al., 2019; White & Kendall, 2007). Given that Kenyan reefs are nearshore, highly accessible, and broadly and heavily fished, reducing recruitment limitations via refuge from fishing is a management priority.

A global survey of coral reef fish stocks found that natural capital disincentives have created a number of underperforming or "dark spot" fisheries (Cinner et al., 2016). Despite evidence for the lack of benefits of reducing fishing costs, this activity remains a popular approach to managing poor and offshore fisheries (Sala et al., 2018; Schuhbauer et al., 2017). In fact, even environmental-based organizations have used food security and pro-poverty arguments to request greater redistribution of subsidies to poor small-scale fisheries (Paolo, III, & Fonseca-Marti, 2005). The benefits of fisheries subsidies will, however, depend on their focus, status, and the factors limiting stocks. Frequently management decisions are made without knowing status, often underestimating yields, overestimating stock production potential, or basing estimates on resilient and high production taxa (Kaunda-Arara et al., 2016; Pauly & Zeller, 2016; Szuwalski, Burgess, Costello, & Gaines, 2017).

Combining unrealistic projections of production, the political expediency of reducing some user's costs, and arguments for increasing overall food security should greatly influence political decision-making. Together, they can create incentives for policies that persist in the face of market failures. Moreover, we found evidence for the depreciation of the gear subsidy prior to the end user, which suggests rent-seeking and transactional profiteering. While governance devolution was expected to increase accountability, economic failures can be difficult to eliminate when pro-poverty and elite capture strategies combine—regardless of the scale of governance (Béné et al., 2009). Perhaps, a solution would be to better tie development projects to evidence-based outcomes—a persistent problem created by perennially insufficient funding to undertake the needed post-project evaluations (Banerjee & Duflo, 2011). Moreover, there are persistent development theory debates, such as expedient governance needs versus market failures that are seldom resolved by specific cost–benefit projects (Acemoglu & Robinson, 2013). Nevertheless, we found that the cheaper alternative of subsidizing closures worked most effectively in reducing the recruitment limitations impeding Kenya's fishery. The benefits of closures are likely to have social costs and accrue at longer time scales than appreciated by short-term interests and projects (McClanahan & Abunge, 2016). Nevertheless, closures appear critical for supporting widely accepted fisheries sustainability goals and are our policy recommendation for improving Kenya's fisheries.

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CONFLICT OF INTEREST

The authors declare no potential conflict of interest.

AUTHOR CONTRIBUTIONS

TRM conceived the study, supervised the data collection and analysis, and wrote the first draft of the manuscript; JKK collected the field data and performed the analysis of the data. All authors contributed to revision and preparation of the final version.

DATA AVAILABILITY STATEMENT

The fisheries catch are available upon a formal request to the authors.

ETHICS STATEMENT

The research data collection did not require ethics approval.

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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