Title

Understanding Cooperation and Coping to Enhance Small-Scale Fisheries Management

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Publication Date

2024

Peer reviewed|Thesis/dissertation

University of California
Santa Barbara

Understanding Cooperation and Coping to Enhance Small-Scale Fisheries Management

A dissertation submitted in partial satisfaction
of the requirements for the degree

Doctor of Philosophy
in
Environmental Science and Management

by

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March 2024

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To my parents, who taught me by their example, the values of perseverance, patience, curiosity, and a deep love for one's craft, which were crucial for completing this degree.

Acknowledgements

I am profoundly thankful for the financial support provided by the National Agency of Research and Development of Chile through their program ANID/Becas Chile/Doctorado72180436, the Latin American Fisheries Fellowship, and the Bren School of Environmental Science & Management. This dissertation would not have been feasible without the invaluable guidance and encouragement of my committee members: Steve Gaines, Robert Heilmayr, Kelsey Jack, and Stefan Gelcich. I consider myself incredibly lucky to have been guided by such a group of perceptive, kind, and bright scholars throughout this journey. My gratitude also extends to my friends in Santa Barbara, whose friendship marked this overseas experience with nature, laughter, personal growth, and unwavering support. The friends I made during this period have greatly expanded my perspectives and profoundly enriched my life. I am equally thankful to my family and friends in Chile; despite the distance, their constant support and encouragement have always been a source of strength, especially in challenging times. Lastly, I am deeply grateful to all the fishers, Chilean government officials, and researchers who played a pivotal role in this research. Their readiness to collaborate, share their knowledge, and set an inspiring example of caring for our coastal resources has fueled my desire to make a meaningful contribution to their management, aiming for a sustainable and equitable future.

Curriculum Vitæ

María Ignacia Rivera-Hechem

Education

2024	Ph.D. in Environmental Science and Management (Expected), University of California, Santa Barbara.
2021	Ph.D. in Social Complexity Sciences, Universidad del Desarrollo, Santiago, Chile.
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Publications

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- Project Investigating the Effect of Thermoelectric Plants on Marine Fisheries' Production Along the Coast of Chile of High-school Student: Gautam Nair, 2022, Extension Programs, UCSB.
- Master Group Thesis Evaluating Drivers for Tracking Technology Adoption in Small-scale Fisheries of Students: Thomas Butera, Gage Clawson, Corinna Hong, Barbara Rose Hoover, and Juan Silva, 2019, Bren School of Environmental Science & Management, UCSB.

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- Becas Chile. International Student's Masters Scholarship.
- Latin American Fisheries Fellowship. Masters Fellowship

Abstract

Understanding Cooperation and Coping to Enhance Small-Scale Fisheries Management

by

María Ignacia Rivera-Hechem

Small-scale fisheries (SSFs) are critical to global food security, livelihoods, and cultural heritage. As one of the oldest forms of wild harvesting, SSFs can support sustainable practices through the self-organization of fishers. Yet, global market integration and environmental change reshape fishers' incentives, influencing behaviors like cooperation and coping. Understanding and quantifying these behaviors is key to fostering the sustainability and resilience of SSFs. I employ social science methods to investigate cooperation and coping in SSFs quantitatively. In my first chapter, I apply experimental economics to investigate the impact of game experiment designs on measurements of cooperation levels among fisher groups in real-world settings. My second chapter presents a nationwide evaluation of a fisheries co-management policy in Chile, implemented more than two decades ago, to encourage cooperative and sustainable practices among fishing communities. I assess the survival of co-management projects as a measure of success and study its variability across social and ecological conditions. In my third chapter, I study coping responses to fisheries closures triggered by harmful algal blooms in Southern Chile. Using econometric methods, I analyze fishers' mobility across resources and space, and the influence of market dynamics and management regulations. My research seeks to contribute to more informed and effective fisheries management that considers the complex interplay of incentives, behaviors, and policy outcomes in SSFs.

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Chapter 1

Introduction

Feeding a growing population within rapidly changing and deteriorating ecosystems presents a significant challenge to humanity. The diversity that small-scale fisheries (SSFs) bring through their products, distribution channels, and production methods enhances the resilience of our food systems and can be pivotal in meeting this challenge (Short et al., 2021; Viana et al., 2023). Contributing to at least 40% of the global fish catch and about 66% of the fish consumed by humans, SSFs support the livelihoods of approximately 500 million people (FAO et al., 2023). Moreover, SSFs hold profound cultural value, especially for indigenous communities, embodying one of humanity's oldest forms of wild harvesting.

Despite their ancient origins and localized operations, many SSFs are now integral to global food and social systems, operating within interconnected markets and geopolitical institutions (Arthur et al., 2022). Moreover, the ecosystems that support these fisheries face continuous and rapid changes due to anthropogenic pressures and climate change. This evolving scenario presents a new set of incentives for fishers, potentially diverging from those underpinning historically sustainable practices.

Understanding these incentives and their impact on fisher behavior is essential for developing effective fisheries management strategies (Andrews et al., 2021). The goal extends beyond merely establishing restrictive policies and regulations; it involves harnessing the inherent management abilities and stewardship values of small-scale fishers (Short et al., 2021). This strategy is particularly relevant in addressing the challenges of limited government capacity to devise, implement, and enforce regulations across a diverse and extensive range of fishing activities.

Cooperation and coping are two critical types of behavior within SSFs, particularly in the face of overexploitation and abrupt environmental changes, which represent constant threats to fishers' livelihoods and subsistence. Cooperation involves individuals forgoing personal gains for collective benefits, crucial for sustaining shared fish stocks (Ostrom, 1990). This includes activities such as rule-making and enforcement, limiting overextraction, and adhering to regulations. Coping, on the other hand, entails utilizing available skills, resources, and opportunities to navigate adverse conditions and preserve essential functions when facing short-term challenges like sudden environmental shocks (Field et al., 2012). Therefore, accurately measuring these behaviors, understanding their variations, and recognizing their broader effects are essential steps in enhancing the sustainability and resilience of SSFs.

My dissertation investigates cooperation and coping within SSFs, aiming to uncover the drivers and outcomes of these behaviors to inform fisheries management. I approach SSFs as complex social-ecological systems, where outcomes emerge from the non-linear interplay of variables across different subsystems (Ostrom, 2009). The challenge of quantitatively capturing these intricate relationships is significant, yet can be met by integrating insights and methodologies from various disciplines (Biggs et al., 2021).

Game theory offers a robust framework for studying the incentives that drive cooperation in managing natural resources (Sethi & Somanathan, 1996). Meanwhile, experimental economics provides critical tools for testing these theoretical insights using methods like game experiments designed to quantify cooperation (Ostrom, 2006). My first chapter explores how the design of these experiments affects our ability to measure differences in the levels of cooperation among fisher groups in real-world settings. I demonstrate that variations in game experiment design can significantly influence the accuracy with which we understand cooperative behaviors in the field.

Common-pool resources theory has been crucial in delineating the conditions conducive to cooperation, highlighting that well-defined boundaries of the resource system and its user group enhance cooperative incentives (Ostrom, 1990). This principle has shaped national policies, such as the Chilean Territorial User Rights for Fishing (TURF). This initiative seeks to promote sustainable fishing practices by allocating exclusive access to coastal resources to designated fishing communities. In the second chapter of my dissertation, I conduct a comprehensive nationwide evaluation of this pioneering program, examining the durability of these TURF projects across diverse social and ecological conditions.

To effectively analyze coping behaviors triggered by abrupt environmental change, it is crucial to differentiate between behavioral changes directly caused by such shocks and those attributable to inherent behavioral variations. Econometric techniques are particularly suited for this task. In the third chapter, I apply these methods to investigate how small-scale fishers coped with the immediate closure of fisheries following a harmful algal bloom induced by extreme weather conditions. Additionally, I examine the role of market dynamics and management strategies in influencing these coping responses and their subsequent effects across actors and ecosystems.

These three projects underscore the importance of interdisciplinary research in unraveling the dynamics of coping and cooperation within SSFs. They offer crucial insights into refining methodologies for measuring these behaviors and their variability and suggest an intricate interplay of material and non-material incentives shaped by social, institutional, and ecological factors. This work highlights the need for policies and regulations designed to motivate fishers to act as managers and stewards of the resources, providing them with autonomy and support to overcome the challenges imposed by environmental change, market pressures, and economic needs.

Chapter 2

Effects of increased ecological validity on the external validity of game experiments to measure cooperation in natural resource use

Abstract

Understanding cooperation in common-pool resources is crucial for guiding sustainable development. Game experiments are increasingly being used for this, but to be useful, their results must reflect outcomes in the field or be externally valid. To improve external validity, researchers are increasing the ecological validity of their games, which is the extent to which the game resembles the decision in the field. However, the effectiveness of this approach has not been tested. We assessed how external validity is affected by two features that increase ecological validity in common-pool resource games: a contextual frame and a payoff structure that allows peer enforcement. We do this by comparing the outcomes of games with varying designs to the differences in cooperation displayed by fishing communities in the management of common-pool resources in Chile. We found that the contextual frame improved external validity, likely by changing norms and expectations, whereas peer enforcement did not.

Introduction

Common pool resources (CPRs), such as forests and fisheries, provide millions of livelihoods worldwide (Tambe, 2022). Owing to their shared and rival nature, CPRs are prone to overexploitation under open access (Gordon, 1954). However, multiple cases demonstrate that people can cooperate to sacrifice individual gains from over-extraction for the collective benefits of sustainable use (Ostrom, 1990). Understanding why and when CPR users cooperate is crucial for conservation science and practice. However, measuring cooperative decisions in the field can involve biases inherent in sensitive behaviors (Gavin et al., 2010). Additionally, research on isolating the determinants of cooperation in the field is often ethically and logistically constrained.

Game experiments developed in experimental economics recreate CPR incentives under controlled conditions (Ostrom, 2006). These experiments use game theory to design participants' payoffs, information, and actions, allowing researchers to measure cooperation and identify its determinants (Viceisza, 2016). Game experiments have become popular to study and inform CPR management (Lindahl et al., 2021). For this, the extent to which decisions in games reflect behaviors in the field, known as external validity, is a critical consideration (Viceisza, 2016). However, evidence of the external validity of game experiments is mixed (Galizzi & Navarro-Martinez, 2018; Naar, 2020).

These mixed results may arise from methodological differences. Studies that examine the external validity of game experiments in CPR settings vary in ecological validity - the extent to which the game resembles the decision in the field (**Table A.1**). Because increasing ecological validity is expected to improve external validity (Levitt & List, 2007; Roe & Just, 2009), researchers are incorporating more field elements into games (Janssen et al., 2010; Rommel & Anggraini, 2018). Although this approach is widely used, we lack assessment of the extent to which increased ecological validity leads to improved external validity. Experimental economics recommends keeping games clean to ensure experimental control, parsimony, and generalizability (Roe & Just, 2009; Smith, 1976). Identifying field elements that enhance external validity can prevent unnecessarily compromising the advantages of an experimental approach.

To address these gaps, we conducted an experiment with fishers in Chile who operate under territorial users' rights for fishing (TURFs), where cooperation is crucial for sustainable use (Gelcich et al., 2010). We compared the measurements of cooperation from games with varying levels of ecological validity to the differences in cooperation exhibited by these communities in real-life TURF use (Gelcich et al., 2017; Marín et al., 2012). We tested the effects of two features known to increase ecological validity in common-pool resource settings: a contextual frame and a peer-enforcement mechanism. Results show that a contextual frame was necessary to achieve external validity, while a peer-enforcement mechanism did not affect it. By applying insights from experimental economics and psychology, we explored the underlying mechanisms improving external validity. Our conclusions are critical for the design of future game experiments aimed at informing conservation policy and practice.

Hypotheses

Several game features can be adjusted to increase ecological validity, including the language used to frame the game (Levitt & List, 2007; Loewenstein, 1999; Viceisza, 2016). Experimental economics typically uses abstract language to increase control and generalizability by avoiding subjective valuation (Henrich et al., 2001; Smith, 1976). Nonetheless, behaviors in the field and the lab are context-dependent (Goerg & Walkowitz, 2010;

Rivera-Hechem et al., 2021; Röttgers, 2016; Schill et al., 2019). Contextual frames are argued to activate in the game the norms, expectations, and other heuristics that subjects learn in the field (Bouma & Ansink, 2013; Levitt & List, 2007; Rivera-Hechem et al., 2021). Familiar contexts can also aid task comprehension via memory cues (Alekseev et al., 2017; Ferraro & Vossler, 2010). Activating field heuristics in the game and the correct understanding of payoffs are important for achieving external validity. We hypothesize that using a contextual frame that evokes the field CPR will improve the external validity of CPR game results (**H1**).

Making the payoff structure more ecologically is thought to enhance external validity (Levitt & List, 2007). The payoff structure determines how the game decisions translate into payoffs. Experimental economics usually favors a simple payoff structure for the sake of control and parsimony (Smith, 1976). The simplest CPR payoff structure simulates the social dilemma caused by overextraction's negative externality (Dawes, 1980). Game theory predicts that this game will result in the tragedy of the commons. In the field, CPR payoffs vary due to factors, such as peer enforcement against overextraction. Peer enforcement is widespread and essential for sustaining CPRs in the field and the lab (Fehr & Schurtenberger, 2018; Ostrom, 1990, 2006). If punishment is sufficiently likely and costly, peer enforcement can alter the payoff structure. Peer enforcement can also signal norms of cooperation, making overextraction less attractive to norm followers (Fehr & Schurtenberger, 2018). Given the influence of payoffs and norms on CPR use in the field, we hypothesize that including a peer-enforcement mechanism will improve the external validity of CPR game results (**H2**).

Frames that evoke moral contexts can enhance peer enforcement and, as a result, cooperation compared to abstract frames (Mieth et al., 2021). If peer enforcement depends on the context, using a contextual frame and a peer-enforcement mechanism may be more effective at improving external validity than using them individually. Furthermore, if increased ecological validity leads to improved external validity, incorporating two field elements into the game could enhance external validity more than using them separately. Therefore, we hypothesize that combining a contextual frame and a peer-enforcement mechanism will have a synergistic effect on the external validity of CPR game results (**H3**).

Methods

Experimental design and implementation

We recruited members from two communities that displayed relatively high cooperation in TURF use (HC, n=60) and three communities that displayed relatively low cooperation (LC, n=60). Communities were classified as HC or LC based on an index reflecting cooperation in TURF use, using prior field measurements (**Table A.2**). Participants played a repeated static CPR game in groups of five with other community members. In each of 20 rounds, participants individually and anonymously decided the number of resource units to extract above their individual quota (i.e., overextraction) from 0 to 50. The payoff structure simulated a CPR because each over-extracted unit resulted in a loss of half a unit for other group members (see **Appendix** for a detailed description of the CPR game).

We randomly assigned half of the fishers from each community to play the game under an abstract frame. In this frame, subjects were told that they were extracting coins from a shared pool. The remaining participants played the game framed as the extraction of loco (*Concholepas concholepas*). This fishery is a prime example of TURFs in Chile, as it is exclusive to TURFs and evokes the TURF context (Gelcich et al., 2010). Game instructions were almost identical between frames (see **Appendix** for game instructions). The CPR game consisted of two stages that differed in the presence of a peer-enforcement mechanism. During the first ten rounds (unenforced), participants cannot engage in peer enforcement. In the last ten rounds (enforced), we introduced a peer-enforcement mechanism to simulate peer enforcement in the field.

Our experiment had four designs to assess external validity at varying ecological validity levels (**Table 2.1**). The abstract-unenforced (AU) design had the lowest ecological validity, while the contextual-unenforced (CU) and abstract-enforced (AE) designs had intermediate levels, incorporating context and peer-enforcement mechanisms from the field, respectively. The contextual-enforced (CE) design was the most ecologically valid, incorporating two elements from the field.

Table 2.1: Experimental design.

	Game Design	
	Abstract	**Contextual**
Community type	**Unenforced and Enforced**	**Unenforced and Enforced**
Highly cooperative (HC)	30 fishers (6 groups)	30 fishers (6 groups)
Less cooperative (LC)	30 fishers (5 groups)	30 fishers (5 groups)*

[*] In one of the experimental sessions, groups were randomly reallocated in each round. Because subjects were unaware of the reallocation, behaviors should not differ from those expected in fixed groups. Subjects in this session potentially interacted with all the other nine subjects in the session. To obtain independent observations, we computed the mean group compliance across all 10 subjects participating in this session.

To measure cooperation, we averaged the compliance percentage with the individual quota across group members per round (see **Appendix** for the outcome variable rationale). We considered a larger difference in compliance between HC and LC in the game as indicative of higher external validity given that HC exhibits higher cooperation than LC in TURF use in the field.

Analyses

The compliance percentage for LC (Y_{LC}) and HC (Y_{HC}) in AU, CU, AE, and CE can be represented as linear combinations of the estimator in **Equation** (2.1), as shown by **Equations** (2.2), (2.3), (2.4), (2.5), (2.6), (2.7), (2.8), (2.9).

$$Y_{i,t} = \alpha + \beta_1 HC_i + \beta_2 C_i + \beta_3 C_i \times HC_i + \beta_4 E_t + \beta_5 HC_i \times E_t + \beta_6 C_i \times E_t + \beta_7 HC_i \times C_i \times E_t + \epsilon_{i,t} \tag{2.1}$$

Where $Y_{i,t} \in \{0, ..., 100\}$ represents the mean compliance percentage of group i in round t, HC_i is equal to one if group i belongs to HC and zero otherwise, C_i is equal to one if group i is playing the game under the contextual frame and zero otherwise, and E_t is equal to one if round t is enforced ($t > 10$) and zero otherwise.

$$Y_{LC,AU} = \alpha \tag{2.2}$$

$$Y_{HC,AU} = \alpha + \beta_1 \tag{2.3}$$

$$Y_{LC,CU} = \alpha + \beta_2 \tag{2.4}$$

$$Y_{HC,CU} = \alpha + \beta_1 + \beta_2 + \beta_3 \tag{2.5}$$

$$Y_{LC,AE} = \alpha + \beta_4 \tag{2.6}$$

$$Y_{HC,AE} = \alpha + \beta_1 + \beta_4 + \beta_5 \tag{2.7}$$

$$Y_{LC,CE} = \alpha + \beta_2 + \beta_4 + \beta_6 \tag{2.8}$$

$$Y_{HC,CE} = \alpha + \beta_1 + \beta_2 + \beta_3 + \beta_4 + \beta_5 + \beta_6 + \beta_7 \tag{2.9}$$

According to **H1**, differences in compliance between HC and LC will be larger in CU than in AU or $Y_{HC,CU} - Y_{LC,CU} > Y_{HC,AU} - Y_{LC,AU}$. Rewriting this condition using Equations (2.2), (2.3), (2.4), (2.5) leads to $\beta_3 > 0$. Similarly, **H2**, predicts $Y_{HC,AE} - Y_{LC,AE} > Y_{HC,AU} - Y_{LC,AU}$, which is equivalent to $\beta_5 > 0$ when replaced with **Equations** (2.2), (2.3), (2.6), (2.7). Finally, **H3** indicates that $Y_{HC,CE} - Y_{LC,CE} > Y_{HC,CU} - Y_{LC,CU} + Y_{HC,AE} - Y_{LC,AE}$, which is equivalent to $\beta_7 > \beta_1$ when replaced with Equations **Equations** (2.4), (2.5), (2.6), (2.7), (2.8), (2.9). We tested the null hypotheses $H_{1,0} : \beta_3 = 0$, $H_{2,0} : \beta_5 = 0$, and $H_{3,0} : \beta_7 = \beta_1$ by running an OLS regression of **Equations** (2.1) and two additional specifications. One controls for rounds as a continuous variable, and the other includes a fixed effect for the per-stage round. We ran all the regressions with robust standard errors using `sandwich` (R-4.0.2) and used `estimatr` to test $H_{3,0}$.

Results

Hypotheses

The contextual frame marginally increased compliance for HC and LC (**Figure 2.1**, Coefficient Contextual = 11.43 [-0.1, 23.0], p < 0.10, **Table 2.2**). This increase in compliance was 14% higher for HC than for LC, supporting **H1** (HC x Contextual = 14.43[0.3, 28.6], p < 0.05). The contextual frame was required for external validity, as compliance did not differ between community types under the abstract frame (HC =

0.83[-8.1, 9.8], $p > 0.10$). These results are robust to different specifications (**Table 2.2**).

We found no support for **H2**. Adding a peer-enforcement mechanism to the game did not have a significantly different effect on compliance for HC relative to LC (**Figure 2.1**, **Table 2.2**, HC x Enforced =-6.75[-19.1, 5.6], p>0.10). Our results did not support H3 ($\beta_7 - \beta_1$ = -5.55 [-21.6, 10.5], p>0.10). The difference in compliance between HC and LC in CE was not greater than the differences observed in AE and CU combined (**Figure 2.1**).

Table 2.2: OLS regressions of the mean group percent of compliance, weighted by the number of players in a group, with robust standard errors in parentheses.

	Dependent variable:		
	Mean group percent of compliance		
	No round controls	Rounds (1-20)	Fixed effects rounds (1-10)
HC	0.8 (4.6)	0.8 (4.5)	0.8 (4.5)
Contextual	11.4* (5.9)	11.4* (5.9)	11.4* (5.9)
HC x Contextual	14.4** (7.2)	14.4** (7.1)	14.4** (7.2)
Enforced	−3.5 (5.3)	8.1 (6.4)	−3.5 (5.3)
HC x Enforced	−6.7 (6.3)	−6.7 (6.2)	−6.7 (6.3)
Contextual x Enforced	12.1 (7.8)	12.1 (7.8)	12.1 (7.8)
HC x Contextual x Enforced	6.4 (9.4)	6.4 (9.3)	6.4 (9.4)
Round (1 to 20)		−1.2*** (0.4)	
Constant	41.2*** (3.8)	47.6*** (4.5)	46.9*** (5.3)
AIC	4246.5	4239.4	4253.5

Note: *p<0.1; **p<0.05; ***p<0.01

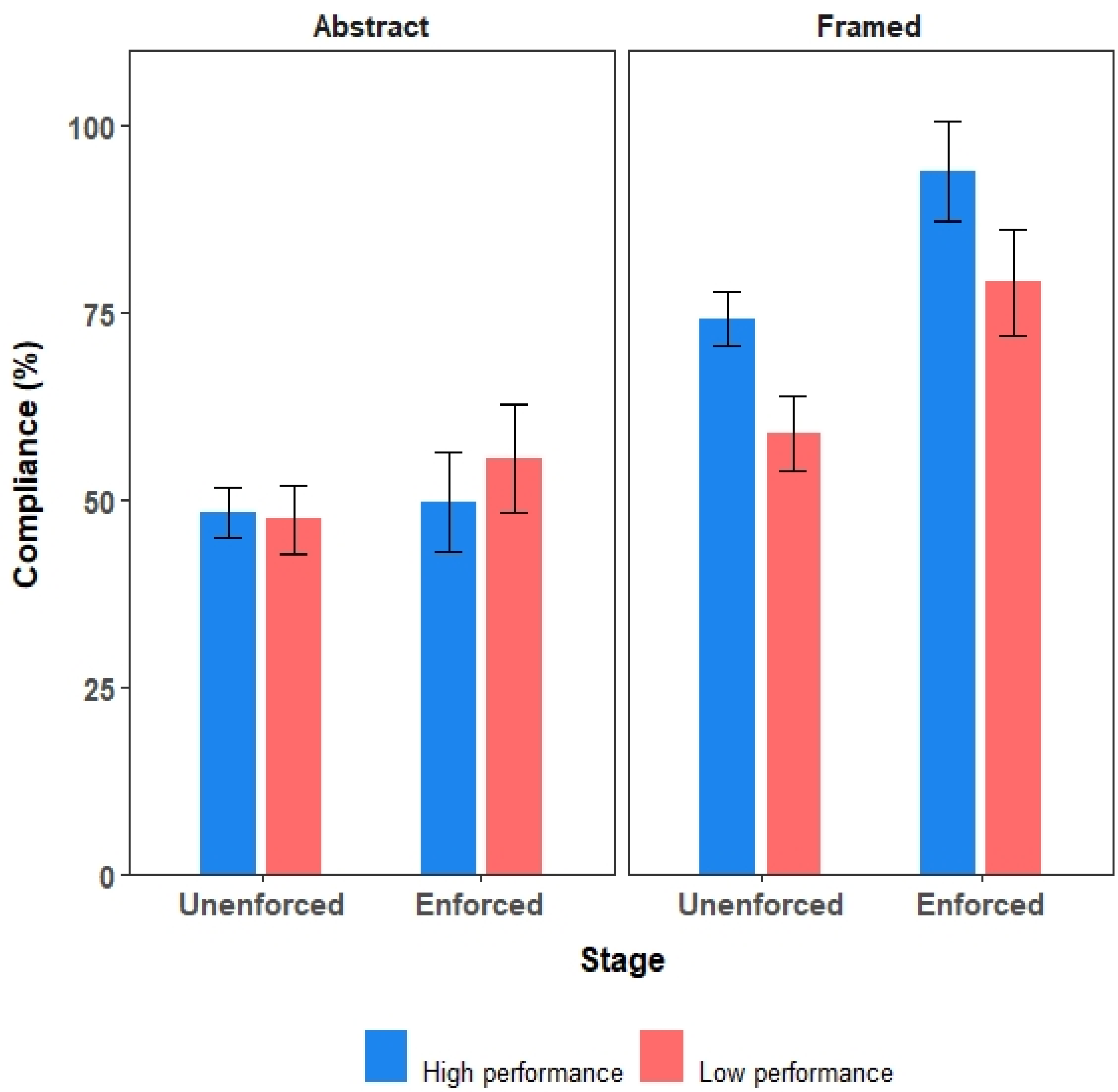

Figure 2.1: Group mean percentage of compliance in a common-pool resource game under an abstract (left panel) and a contextual frame (right panel) displayed by members of fishing communities that exhibit relatively high (blue) and low (red) cooperation in the use of territorial users rights for fishing in an unenforced and an enforced stage. Error bars represent standard errors. Bar plot based on linear combinations of the most parsimonious model (Column 2 in Table 2.2).

Exploring mechanisms underlying improved external validity via contextual frames

Contextual frames can improve understanding of the payoffs enhancing rational behavior (Alekseev et al., 2017). In our game, the rational decision in the unenforced stage involved extracting all units (see **Appendix** for details). If the contextual frame improved payoffs comprehension, compliance should have decreased relative to the abstract frame. Instead, the contextual frame slightly increased compliance for HC and LC (**Table 2.2**). This suggests that the contextual frame did not operate by enhancing the understanding of the payoffs. Furthermore, it is unclear why the contextual frame cues would affect task comprehension differently for HC than LC given that both groups are familiar with TURFs.

Contextual frames can also affect expectations about others' cooperation (Ellingsen et al., 2012; Fehr & Schurtenberger, 2018). Compliance decisions reflect expectations as people tend to match their cooperation with what they expect from others (Fehr & Schurtenberger, 2018). In the absence of information on others' behavior in the first round, participants likely relied on field-drawn expectations. Thus, to explore this effect, we conducted pairwise comparisons of first-round compliance using the Wilcoxon test with multiple hypotheses testing correction.

HC displayed significantly higher first-round compliance in the contextual compared to the abstract frame (**Figure 2.2**, Median difference= 20, IQR= 50–0, p-corrected <0.01), suggesting that HC members had more optimistic expectations of others' compliance in the contextual frame than the abstract frame. This was not the case for LC members (p-corrected $= 0.75$). The framing effect on expectations was relevant for external validity as first-round compliance differed significantly between HC and LC in the contextual

(Median difference= 10, IQR= 60–0, p-corrected < 0.05), but not in the abstract frame (p-corrected $= 0.75$). We explored the activation of social norms through framing as a final mechanism (Lesorogol, 2007). Punishing was unlikely for rational players in our game (see **Appendix** for details). Therefore, we interpreted punishment as a signal of moral disapproval stemming from norm violation (Fehr & Schurtenberger, 2018). To assess the strength of cooperative norms, we regressed the group probability of punishment per round on the type of frame and community, and their interaction. The contextual frame activated stronger cooperative norms than the abstract frame for both HC and LC (**Figure 2.3**, **Table 2.3**, Contextual =0.13[-0.01, 0.3], $p<0.1$), but significantly more so for HC (HC x Contextual =0.30[0.1,0.5],$p<0.01$). These results are held under specifications that control for the mean observed over-extraction and rounds as fixed and continuous effects (**Table 2.3**). Peer enforcement had no significant effect on compliance for HC or LC under either frame (**Table 2.3**), but it sustained compliance over rounds for HC in the contextual frame (**Figure 2.2**). To confirm this, we conducted pairwise comparisons of the group compliance slope over the rounds between the unenforced and enforced stages for each frame-community type using the Wilcoxon test. The only statistically significant positive change in slopes between the enforced and unenforced stages was displayed by HC under the contextual frame (Median difference= 2.98, IQR= 5.24–0.63, p-corrected < 0.05).

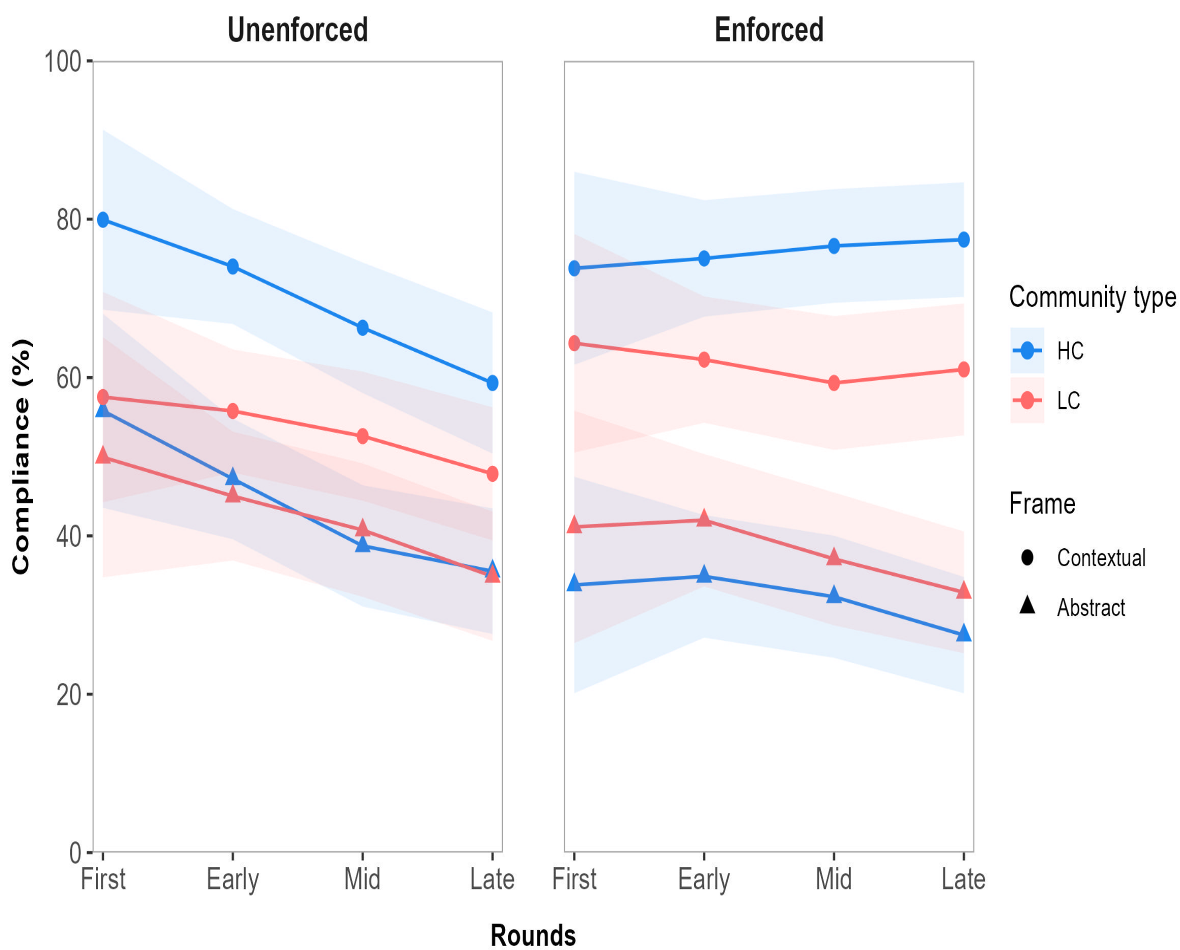

Figure 2.2: Evolution of the mean individual compliance percent in the unenforced (left panel) and enforced stage (right panel) of a common-pool resource game displayed by the members of fishing communities that exhibit relatively high (blue) and low (red) cooperation in the use of territorial users rights for fishing under a contextual (circles) and abstract (triangles) frame. Shaded regions represents 95 percent confidence intervals (n = 30).

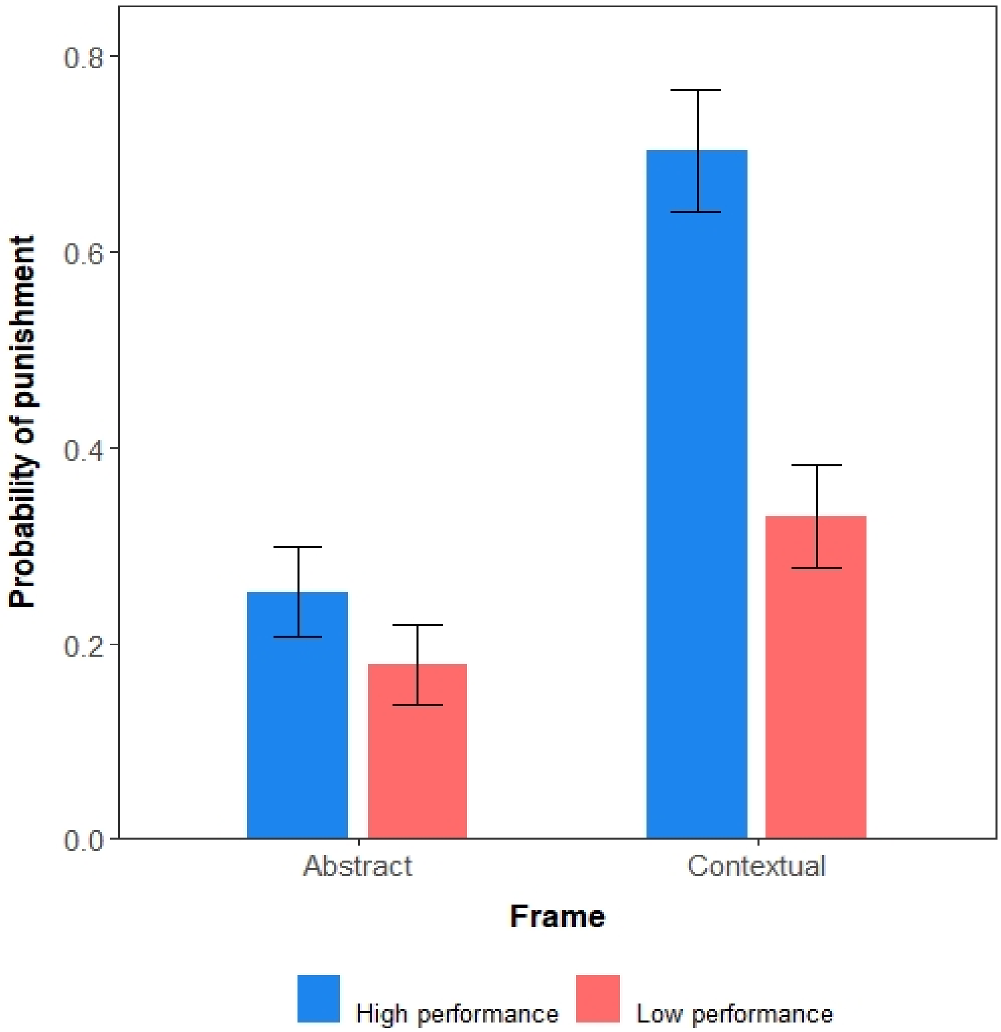

Figure 2.3: Group mean probability of punishment in a common-pool resource game under an abstract (left) and a contextual frame (right) displayed by members of fishing communities that exhibit relatively high (blue) and low (red) cooperation in the use of territorial users' rights for fishing in an unenforced and an enforced stage. Error bars represent standard errors. Bar plot built based on the most parsimonious model (Column 1 in Table 2.3).

Table 2.3: OLS regressions of the group probability of punishing, with robust standard errors in parentheses.

	Dependent variable:			
	Probability of punishment			
	No controls	No controls II	Continous rounds	Fixed effects rounds
HP	0.1 (0.1)	0.1 (0.1)	0.1 (0.1)	0.1 (0.1)
Contextual	0.1^{*} (0.1)	0.2^{**} (0.1)	0.2^{**} (0.1)	0.1^{*} (0.1)
HC x Contextual	0.3^{***} (0.1)	0.3^{***} (0.1)	0.3^{***} (0.1)	0.3^{***} (0.1)
Group overextr.		0.002 (0.002)	0.002 (0.002)	0.001 (0.002)
Round (11 to 20)			−0.004 (0.01)	
Constant	0.2^{***} (0.04)	0.1 (0.1)	0.2 (0.2)	0.1 (0.1)
AIC	180.5	181.8	183.6	193.4
Note:				*p<0.1; **p<0.05; ***p<0.01

Discussion

Game experiments are a promising tool for studying and promoting sustainable CPR use and conservation (Lindahl et al., 2021; Ostrom, 2006). Researchers are enhancing the ecological validity of games to improve their external validity. To test the empirical support for this approach, we assessed the effect of increasing the ecological validity of the game's frame and payoff structure on the external validity of results. We found that the external validity of CPR games depended on the ecological validity of the task. Specifically, framing the game within the field's context was necessary to achieve external validity. However, adding a peer-enforcement mechanism to better simulate the field payoffs did not improve external validity. Our results indicate that the contextual frame increases external validity by activating, in the game, the context-specific expectations and norms that guide field behaviors.

Our results contribute to ongoing debates on whether game experiment should be contextualized (Alekseev et al., 2017). Based on our observations, research using CPR games to measure outcomes in the field may benefit from contextual frames. This aligns with common practices among sustainability scholars (Cardenas, 2003; Gelcich et al., 2013). Using a contextual frame in our game likely activated field heuristics, revealing the differences in cooperation between HC and LC in TURF use. Norms and expectations are crucial in determining CPR outcomes in the field (Cinner et al., 2019; Ostrom, 1990, 1998). Our results confirm previous findings regarding the context-specificity of these heuristics (Bouma & Ansink, 2013; Krupka & Weber, 2013; Rivera-Hechem et al., 2021). Additionally, our study provide empirical support to previous calls to consider context specificity in the design of CPR games for achieving external validity (Röttgers, 2016; Torres-Guevara & Schlüter, 2016).

Not all increases in ecological validity resulted in improved external validity. Peer enforcement has proven a key determinant of CPR outcomes in the field, including in Chilean TURFs (Crona et al., 2017; Ostrom, 1990). Despite significant differences in the probability of punishment between HC and LC under the contextual frame, there was no improvement in external validity compared with using the contextual frame alone. Nonetheless, peer enforcement is just one of the many factors that can influence payoffs in CPR settings. Other elements, such as temporal and spatial resource dynamics, affect cooperation in CPR games and can influence incentives in the field, making them potentially important for external validity (Janssen et al., 2010; Rommel & Anggraini, 2018). We found that norms and expectations were crucial for the external validity of CPR games. Future research could use methods designed to identify these heuristics to further explore their role in external validity (Ellingsen et al., 2012; Krupka & Weber, 2013).

Designing game experiments involves trade-offs between ecological and internal validity (Roe & Just, 2009; Viceisza, 2016). Abstract frames are often preferred in cross-group studies to increase comparability and avoid cultural differences in wording (Henrich et al., 2001); but, they may not provide a complete understanding when comparing groups. Relying solely on the abstract frame would have led us to conclude equal cooperation between HC and LC, whereas using the contextual frame revealed differences in cooperation in the TURF context. While contextual frames may increase ecological validity, they are argued to compromise internal validity by introducing subjective valuation outside the experimenter's control (Smith, 1976). We found that abstract frames can also compromise internal validity. Our attempt to recreate the incentives of TURF use using an abstract frame seemed to have resulted in interpretations outside the TURF context, based on the different outcomes observed between frames. This shows that participants may interpret the game differently than intended under abstract frames (Engel & Rand, 2014; Loewenstein, 1999). By setting the same context for all participants and the experimenter, contextual frames can provide control while improving external validity.

Future CPR game experiments should carefully evaluate the use of contextual frames to ensure improved external validity, while avoiding potential demand effects (Alekseev et al., 2017). Using reference frames that elicit cooperative and noncooperative environments is a promising method to measure outcomes across groups. These frames can establish upper and lower bounds of cooperation making them more comparable across groups (Goerg & Walkowitz, 2010). Debriefing participants at the end of the experimental session could also help account for differences in game interpretations (Viceisza, 2016). To increase the robustness of conclusions, it is also recommended to triangulate game experiment results with other field observations (Anderies et al., 2011).

Understanding and measuring cooperation in CPR use in the field is crucial to inform conservation policies and practice. Game experiments are a valuable tool for this purpose, but they should be rigorously designed to achieve external validity. Our research demonstrated that contextualizing the game within the CPR of interest in the field is critical for achieving external validity. Our findings highlight the need for greater attention and rigor in the design of game experiments, to balance internal and external validity, particularly when used to measure environmental outcomes in the field.

Chapter 3

Evaluating marine area-based co-management across varying conditions for collective action using survival analysis

Abstract

In the evolving landscape of the blue economy and ambitious conservation objectives, area-based co-management stands out as a strategy for preserving local communities' access to marine resources. Accurate policy evaluation is key to effective implementation, yet traditional metrics, despite being data-intensive, often fail to capture the full spectrum of benefits to users participating in these initiatives. This study employs survival analysis to evaluate the Chilean Territorial User Rights for Fishing (TURF) system, a prominent example of area-based co-management in fisheries. Analyzing the longevity of 750 TURF projects over two decades, we used survival probability as a measure of success. The results indicate that 75% of projects persist beyond 15 years and that termination risk decreases over time with TURF projects starting with lower resource abundance tend to have lower survival rates. Guided by the social-ecological systems framework, we grouped TURF projects into clusters based on their conditions for collective action and assessed survival variations across these groups. We observed that lower initial resource abundance is linked to lower survival, but this relationship is moderated in settings characterized by higher resource productivity, well-established user groups, significant resource dependency, close proximity to regional markets, and lower surveillance costs. Our study showcases the value of survival analysis in evaluating area-based co-management and empirically testing social-ecological systems theory, offering a broader perspective compared to traditional performance metrics. This approach reduces bias towards long-standing cases and underscores the importance of accounting for initial conditions and resource dynamics in accurately assessing the impacts of area-based co-management and its drivers.

Introduction

The ocean faces increasing pressure from a variety of activities, including fisheries, energy generation, aquaculture, shipping, and tourism, leading to challenges such as resource depletion and habitat degradation (Jouffray et al., 2020; Paolo et al., 2024). Amid efforts to balance economic development (i.e., the blue economy) with environmental conservation, it is crucial to safeguard local communities' access to coastal resources essential for their livelihoods and diets (Bennett et al., 2019; Cisneros-Montemayor et al., 2021; Winther et al., 2020). In this context, area-based co-management (ABCM), a strategy that entails granting localized rights over marine resources and engaging community members in decision-making and management, shows promise (d'Armengol et al., 2018; Gurney et al., 2021; Reimer et al., 2021). One such policy, implemented around the world, is Territorial User Rights for Fishing (TURFs) (Castilla & Defeo, 2001; Gelcich et al., 2019; Lubchenco et al., 2016). However, the success of TURFs, and ABCM more broadly, varies greatly due to complex interactions between ecological and social factors, such as resource availability, community engagement, and regulatory frameworks (Cinner et al., 2012; Gelcich et al., 2017; Gutiérrez et al., 2011; Nguyen Thi Quynh et al., 2017; Ostrom, 2009). This variability underscores the need for comprehensive evaluations of ABCM's performance to identify the drivers of successful management, which are critical for developing coastal policies that promote sustainable and equitable use of ocean resources.

In fisheries, quantitative evaluations of ABCM typically track variability in a wide range of social and ecological indicators over space or time (Aaron MacNeil & Cinner, 2013; Cinner et al., 2012; Gutiérrez et al., 2011). These assessments are valuable for measuring specific dimensions of performance. For example, abundance and catch are often used for assessing sustainable fishing practices, a central aim of ABCM policies (Rivera et al., 2017). Performance can also be measured through user perceptions, which have revealed incentives beyond fishing profitability, such as enhanced territorial empowerment (Franco-Meléndez, Cubillos, et al., 2021; Gelcich et al., 2017). To encompass the institutional, economic, social, behavioral, and ecological dimensions of performance, researchers have expanded the number of assessed outcomes (Anderson et al., 2015; Arias & Stotz, 2020). However, this approach requires extensive data collection and may still not comprehensively capture value to users. Moreover, accurate performance evaluation often demands data-rich settings, especially when looking to identify factors driving success (Cinner et al., 2019; Fidler et al., 2022; Villaseñor-Derbez et al., 2019). Such reliance on extensive data tends to bias results toward longer-established experiences, potentially overlooking insights from newer or discontinued ABCM initiatives (Gelcich et al., 2019).

The longevity or 'survival' of ABCM projects is a crucial indicator of their value to users, reflecting the net benefits that motivate continued investment. Utilizing survival as a performance metric offers several advantages. It assesses the overall value of ABCM, remaining neutral regarding the specific benefits to different fishing communities. Although survival may not be an explicit policy objective, its significance is considerable, as the long-term social benefits of sustainable practices often accumulate over time. Using survival as a success measure broadens the scope of analysis, enabling the inclusion of enduring, terminated, and short-term projects, thereby shedding light on the dynamics and challenges of ABCM maintenance. Persistent community commitment to area-based management indicates strong collective action, making survival an essential metric for assessing the factors that facilitate or hinder successful management as suggested by social-ecological systems (SES) theory (Ostrom, 2009). Despite the potential insights from studying the survival of ABCM and its variation across socio-ecological contexts, comprehensive and systematic studies on the long-term viability of ABCM projects are still lacking.

In this study, we utilize survival analysis to evaluate the Chilean TURF policy, a pioneering initiative in ABCM (Gelcich et al., 2010). We analyzed the survival probability of 750 TURF-Fishing Community (TURF-FC) combinations established over 23 years of the program, observing how fishing dynamics vary between long-standing and terminated short-term TURFs. Our findings revealed an association between lower initial resource abundances and decreased TURF-FC survival. To explore survival variation, we clustered TURF-FCs based on their conditions for collective action as suggested in the SES framework and assessed survival probabilities across these clusters. Three distinct clusters emerged, two of which had significantly lower initial abundances. One of these low-abundance clusters, characterized by factors like higher productivity, larger and more established user groups, greater resource dependence, market proximity, and lower surveillance costs, demonstrated survival rates similar to the cluster having the highest initial abundances. This indicates that certain conditions relevant for collective action can offset the challenges of TURF maintenance under lower initial resource abundance.

Our study demonstrates the utility of using survival analysis in evaluating ABCM policies, providing a perspective that complements traditional assessments that cover specific dimensions of value to users. By incorporating data from long-standing, terminated, and recent projects, survival analysis offers a comprehensive evaluation of the value of area-based management to fishing communities across diverse social-ecological contexts. This methodological approach enables more holistic policy evaluations and yields insights less skewed toward long-established cases. Our analysis revealed a significant association between initial ecological conditions and the success of ABCM projects, but indicated that factors conducive to self-organization can mitigate ecological constraints.

Background: The Chilean TURF system

Established in 1991, Chile's TURF system allocates exclusive exploitation rights to a Fishing community (FC) for the sustainable use of benthic resources within specific coastal zones. Any FC can propose a management project for a designated TURF area, which must include a baseline assessment of the resources and a management plan. Approved FCs enter into binding usage agreements, obligating them to submit annual reports conducted by certified consultants. These reports assess the abundance and biomass of resources and are used to negotiate quotas with the government. FCs are tasked with quota management, enforcement, and anti-poaching efforts. Non-compliance, such as failing to submit annual reports, can lead to agreement revocation. In such cases, the TURF becomes available for new proposals from either the same or different FCs (Decreto 355 from the Ministerio de Economia).

Substantial research has been conducted on the outcomes of TURFs in Chile (Gelcich et al., 2019). Initial studies predominantly focused on ecological aspects, comparing factors like abundance, biomass, and species richness in TURFs and open-access sites, as well as their temporal changes (Gelcich et al., 2008, 2012). Subsequently, research expanded to incorporate social science methodologies, evaluating outcomes and drivers such as cooperation and social cohesion. This shift allowed for a deeper understanding of behavioral and governance aspects of TURF performance (Crona et al., 2017; Marín et al., 2012; Rivera-Hechem et al., 2021; Rosas et al., 2014).

Recent literature includes comprehensive evaluations using multiple indicators, spanning economic, ecological, social, equity, and cultural dimensions (Arias & Stotz, 2020; Franco-Meléndez, Cubillos, et al., 2021; Franco-Meléndez, Tam, et al., 2021; Gelcich et al., 2017; Outeiro et al., 2015; Zúñiga et al., 2008). Economic indicators typically encompass catch, prices, and income, while social indicators consider factors like education and housing. Institutional aspects are assessed through variables such as compliance, participation, and trust. Data collection methods commonly involve questionnaires or surveys with fishers, underwater surveys, and secondary data from government agencies or TURF reports (Arias & Stotz, 2020).

The scope of TURF studies in Chile has broadened over time. Initially, research primarily consisted of case studies focusing on a single or a small number of TURF projects, predominantly along the central Chilean coast and the northern region of Coquimbo (Gelcich et al., 2019). Recently, the focus has shifted to more extensive regional or larger assessments (Arias & Stotz, 2020; Cerda & Stotz, 2022; Franco-Meléndez, Cubillos, et al., 2021; Franco-Meléndez, Tam, et al., 2021). Nation-wide studies, however, remain limited. These larger-scale evaluations include diverse aspects such as fishers' perceptions of TURF management's challenges and benefits, poaching levels, trend in catch per unit area, and the effects of public enforcement and upwelling on resource abundance and catch (Anguita et al., 2020; Beckensteiner et al., 2020; Gelcich et al., 2017; Quezada & Chan, 2023; Romero et al., 2022).

TURFs primarily contribute to food provision and enhanced local stewardship (Gelcich et al., 2019), yet their effectiveness varies significantly, even within the same region (Arias & Stotz, 2020; Franco-Meléndez, Cubillos, et al., 2021). This variability has been attributed to factors like leadership quality, social capital, enforcement intensity, resource population dynamics, and ocean productivity (Aburto et al., 2013; Anguita et al., 2020; Crona et al., 2017; Gelcich et al., 2012; Marín et al., 2012; Pérez-Matus et al., 2017). Poaching consistently emerges as a significant challenge, impeding FCs from maintaining TURFs and realizing their full benefits, underscoring the critical role of government enforcement in mitigating this issue (Davis et al., 2017; Gelcich et al., 2017; Oyanedel et al., 2018;

Quezada & Chan, 2023; Romero et al., 2022). This research examines the costs and benefits to FC of TURF maintenance. Nonetheless, there is no systematic assessment of TURF survival and how it varies across social-ecological conditions.

Methods

Modelling survival probability of TURF-FCs

Our primary objective was to quantify the 'probability of survival' of a TURF under the management of a specific FC, specifically the likelihood of a TURF-FC reaching a certain age. This involved modeling the time span from the initiation of a TURF-FC to its termination. Survival analysis, an ideal methodology for this type of time-to-event data, was employed. These techniques account for data censoring, which occurs when the termination age of some TURF-FCs is not observable because they remain operational at the end of the study period (Clark et al., 2003). Survival analysis also effectively handles the skewness and strictly positive nature of time as a variable (Clark et al., 2003). Specifically, we computed the probability of a TURF-FC surviving until age t_j using the Kaplan-Meier estimator presented in **Equation** (3.1).

$$S(t_j) = S(t_{j-1}) \times \left(1 - \frac{d_j}{n_j}\right) \tag{3.1}$$

Where $S(t_j)$ represents the survival probability of TURF-FCs at age t_j, $S(t_{j-1})$ is the survival probability for the age right before t_j, d_j is the number of TURF-FC abandoned at age t_j and n_j represents the number of TURF-FC at risk or that were active just before age t_j.

To calculate the Kaplan-Meier estimator, we need to determine the age reached by each

TURF-FC and whether termination is observed within the study period. While no public database comprehensively records the ownership history of each TURF, this information can be reconstructed from publicly available government resolutions. However, official termination of use agreements often occurs some time after FCs have ceased active management of their TURFs. Hence, we used the pattern of required report submissions as an indicator of the active management period by FCs. Data from these reports, compiled by the Institute of Fisheries Development in Chile, spans from 1998 to 2019 and is available upon request. For the years 2020 to 2021, data was provided by the Undersecretary of Fisheries and Aquaculture, who has automated the digitization of these reports since 2019.

We consolidated available datasets to compile a comprehensive record of all approved TURF reports from 1998 to 2021, conducted within approved management projects. Since report entries are linked to TURFs but not explicitly to Fishing Communities (FCs), we determined the start of each TURF-FC combination as the year of the first submitted report for that TURF. We then calculated the age of the TURF-FC from this first report to the last, including a three-year grace period, as long as there were no report submission gaps exceeding three years. For cases with longer gaps, we consulted government resolutions to categorize each TURF-FC into one of three scenarios: (a) those that remained operational despite delayed reports or submission exemptions; (b) those that were terminated and subsequently resumed by the same FC; and (c) those that were terminated and later managed by a different FC.

For TURF-FCs in case (a), age was computed as the interval from the first to the last report, plus a three-year grace period (**Figure 3.1**). In case (b) age was calculated from the first report to the last report before the gap, extended by three years, plus the time from the first to the last report post-gap, also extended by three years. For case (c), where TURF ownership changed, the first TURF-FC's age spanned from the first report to the last report before the gap, with an additional three-year extension. The subsequent TUR-FC's age was calculated from their first to last report after the gap, including the three-year grace period. No instances of a TURF changing ownership more than once were observed. The final year for all cases was capped at 2021.

We cross-checked the resulting database with government resolutions to resolve timing inconsistencies and accurately link each TURF with the corresponding FC over time. We classified a TURF-FC as 'terminated' if its last technical report or management activity was conducted before 2018; otherwise, it was considered 'censored'. Of the 697 TURFs with at least one approved report, 240 exhibited gaps in report submission. Specifically, 162 TURF-FCs remained active despite reporting gaps (case a), 25 experienced temporary abandonment before being resumed by the same FC (case b), and 53 were relinquished by one FC and later adopted by another (case c), as determined from the history of government resolutions. This process resulted in a dataset encompassing 750 unique TURF-FCs. We used the `survival` R package to calculate Kaplan-Meier estimators, including confidence intervals, and the `bshazard` package for non-parametric hazard curve smoothing.

Fishing dynamics in long-term versus short-term TURF-FCs

To explore challenges in maintaining TURFs, we compared fishing dynamics between long-term and short-term TURF-FCs using aggregated data on exploitable abundance, catch, quotas, and prices from TURF reports and the National Service of Fisheries and Aquaculture (SERNAPESCA). All prices were adjusted for inflation according to Chilean consumer price indices. Our analysis of fishing dynamics focused on "loco" (*Concholepas concholepas*), the most documented, valuable, and prevalent resource across TURFs. We

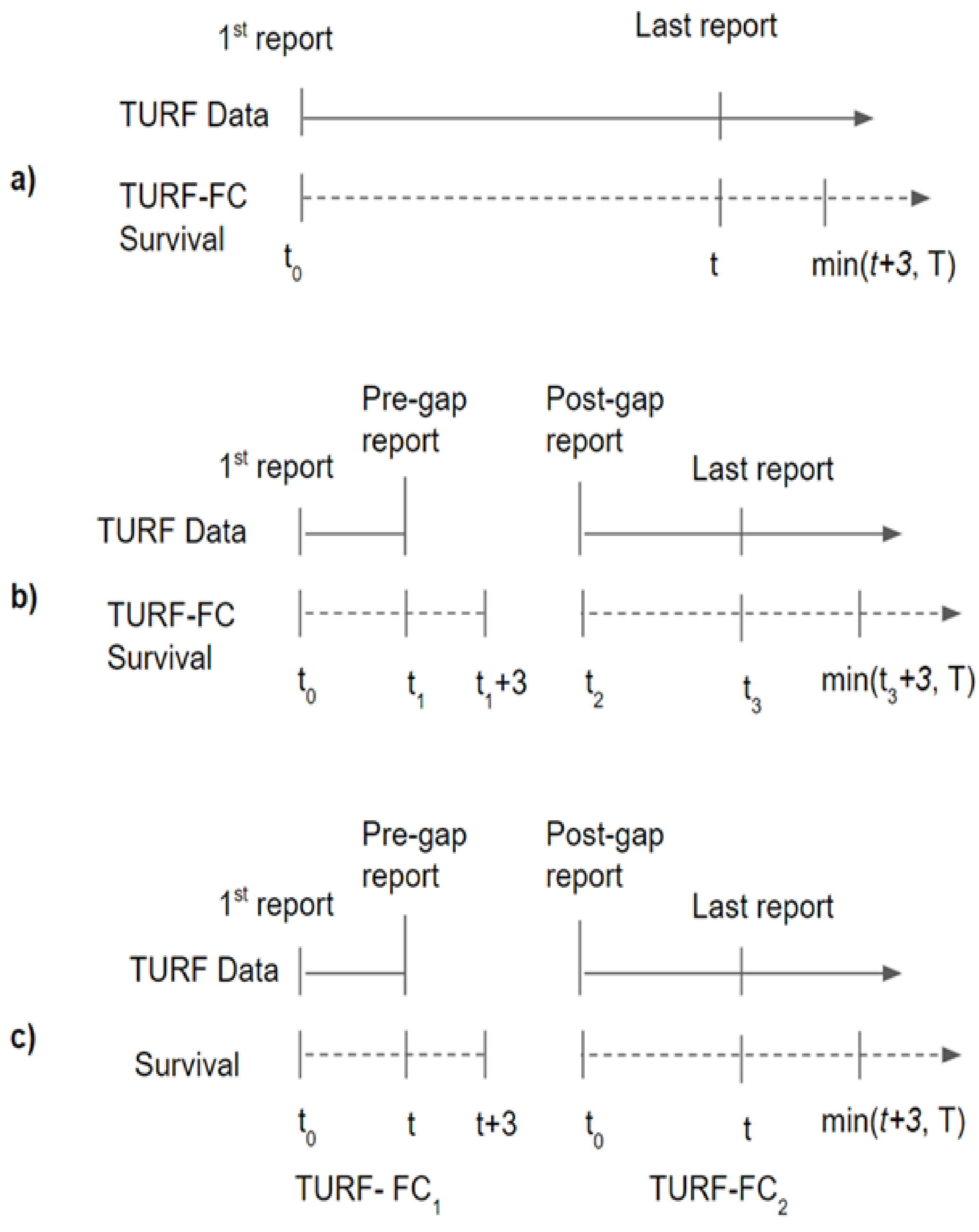

Figure 3.1: Survival time computation for TURF-FCs with gaps in technical reports longer than three years. Cases are categorized into (a) TURFs that remained under the same FC's management with data gaps due to paperwork delays or exemptions, (b) TURFs that were initially abandoned but later resumed by the same FC, and (c) TURFs that were abandoned and subsequently allocated to a new FC. T represents the year 2021.

classified TURF-FCs into long-term (surviving beyond 15 years), short-term (terminated before 15 years), and censored categories (active for less than 15 years and not yet terminated). Censored observations were excluded to specifically assess differences between long-term and short-term TURF-FCs. To account for outliers we winsorized values of the outcome variable between the 5^{th} and 95^{th} percentiles. We applied mixed models to examine trends in exploitable abundance, quota, catch, and revenue across the age of TURF-FCs, treating each variable independently. Age, long-term category, and their interaction were modeled as fixed effects, with random intercepts for each TURF-FC to accommodate baseline variability (**Equation** (3.2)).

$$Y_{i,j} = \beta_0 + \beta_1 \text{Age}_j + \beta_2 \text{Long}_i + \beta_3 (\text{Age}_j \times \text{Long}_i) + u_{0i} + \epsilon_{ij} \tag{3.2}$$

Where $Y_{i,j}$ is the outcome variable (either exploitable abundance, quota, catch, or revenue) for TURF-FC i at age j, β_0 is the intercept of the model, capturing the average level of the outcome variable for short-term TURF-FCs at age zero, β_1 represents the average change of the outcome over age for short-term TURF-FCs, β_2 is the difference in the outcome variable between long-term and short-term TURF-FCs when age is zero, β_3 captures how the effect of age on the outcome variable differs between long-term and short-term TURF-FCs, u_{0i} is the random intercept for each TURF-FC, and ϵ_{ij} is the error term.

Clustering based on conditions for collective action

To examine variations in TURF-FC survival across different conditions for collective action, we first characterized each TURF-FC using indicators. We then performed cluster analysis to group TURF-FCs into distinct clusters based on these indicators. For each cluster, we computed Kaplan-Meier survival estimate and then conducted a log-rank test to determine if survival patterns significantly differed between clusters. Additionally, a Cox proportional hazards model was used to assess differences in hazard rates across clusters, with clusters treated as categorical variables.

We initially focused on ten variables from the SES framework identified as key in shaping user self-organization for resource management (Ostrom, 2009). These include the size and productivity of the resource system, resource unit mobility, collective-choice rules, user numbers, leadership, norms, knowledge of the SES, and resource importance. For Chile's TURF system, we selected corresponding indicators (detailed in **Table 3.1**). Leadership and predictability were excluded due to the lack of suitable indicators at our analysis scale. Considering the benthic nature of TURF resources and the consistent governance policies across FCs, we assumed minimal variation in resource unit mobility and collective-choice rules. We also included indicators for market proximity and surveillance costs, both factors known to impact marine co-management outcomes (Cinner, Maire, et al., 2018; Cinner et al., 2022; Davis et al., 2015; Davis et al., 2017; Gelcich et al., 2017). Additionally, initial exploitable abundance was incorporated, identified as a potential survival predictor in our trend analysis. All variables were scaled between 0 and 1, log-transformed to avoid long-tailed distributions, and checked for high correlation against other variables. Missing data were filled with mean values (see **Figures B.1**, **B.2**, and, **B.3** for variables distribution and correlation).

In SES like our TURF-FCs, characterizing relationships between variables is often challenging due to their simultaneous, non-linear interactions. Non-parametric, data-driven approaches can facilitate capturing this complexity (Epstein et al., 2021; Fidler et al., 2022; Gutiérrez et al., 2011; Rocha et al., 2020; Rocha & Daume, 2021). Depending on the analysis goals, suitable methods may include machine learning algorithms, such as random forests, or multivariate ordination techniques, like cluster analysis. For our study, we utilized cluster analysis to categorize TURF-FCs into distinct groups. This method allowed us to maximize homogeneity within each cluster and heterogeneity between clusters, based on a multidimensional space of indicators relevant to collective action in resource management.

We followed the protocol presented in Rocha et al. (2020) to identify the most suitable clustering method and the optimal number of clusters for our dataset. This protocol entails assessing the internal and stability validation of various clustering techniques and evaluating 30 different clustering performance indices to determine the best number of clusters. Using the `clValid` R package, we tested the internal validity and stability of three commonly used clustering methods: hierarchical, k-means, and Partitioning Around Medoids (PAM) (Brock et al., 2008). Additionally, we utilized the `NbClust` package to find the optimal cluster number (Charrad et al., 2014). Following this protocol, k-means emerged as the most appropriate clustering algorithm for our data (**Figure B.4**). According to Charrad et al. (2014), two main decision rules guide the selection of the optimal number of clusters: the majority rule and indices with superior simulation performance. The majority rule suggested eight as the optimal number of clusters for our data set, with three as a close second (**Figure B.5**). However, when considering indices known for their strong simulation performance, specifically the CH, Duda, Cindex, and Beale indices, all pointed to three clusters as the most suitable choice for our analysis. Considering these results and the aim of our study, we opted for three clusters. To check clustering tendency we implemented the Hopkins' statistic which test the spatial randomness of the data (Lawson & Jurs, 1990).

Table 3.1: Indicators used to characterize TURF-FC based on conditions for collective action.

Indicator	**Description**	**Variable Rationale**
Initial exploitable abundance	Mean scaled initial exploitable abundance in the TURF-FC, computed after winsorizing and normalizing across sample, within species using TURF report data.	Initial abundance was found to be associated with the survival rate.
TURF size	Surface of the TURF obtained from IDE, 2021.	Large system sizes elevate costs of monitoring and learning, while small sizes reduce resource flows (Ostrom, 2009).
Upwelling index	The complement of the mean seasonal amplitude of sea surface temperature (SAM-STT) of the TURF. Reproduced from Anguita et al., 2020.	Resources that regenerate too slowly or rapidly do not justify managing for the future. Upwelling increases productivity in benthic ecosystems in Chile (Anguita et al., 2020; Perez-Matus et al., 2017)
Number of users	Mean number of fishers registered as part of the FC or the multiple FCs, averaged from 2013 and 2023 reported by SERNAPESCA.	Large groups of users face coordination challenges but offer greater monitoring and management capacity (Ostrom, 2009)
Years FC	Number of years that have elapsed since the FC was established, as of 2021 reported by SERNAPESCA.	Norms and knowledge of socio-ecological dynamics are more likely to exist in established groups facilitating collective action (Ostrom, 2009).
Mean poverty	Mean income poverty rate in the TURF municipality, averaged from 2011, 2013, 2015, 2017, and 2020 data from the Social Development Ministry of Chile.	Collective action is cost-effective when resource dependence is high, where greater poverty often indicates stronger reliance on resources (Ostrom, 2009; Fedele et al., 2021).
Distance from nearest market	Euclidean distance from the TURF centroid to the centroid of the nearest city. Computed using IDE, 2021 data.	Market incentives can crowd out intrinsic motivations for collective action and encourage poaching, yet also enhance collective action benefits through greater, steady revenues (Cinner et al., 2021; Epstein et al., 2021)
Distance from nearest fishing cove	Euclidean distance from the TURF centroid to the centroid of the nearest fishing cove. Computed using IDE, 2021 data.	Remote TURFs face higher monitoring challenges, raising the costs of deterring poaching (Davis et al., 2014; 2015).

Results

Survival probability of Chilean TURF projects

Among the 750 TURF-FCs analyzed, 168 were terminated during the study period (Figure 3.2 A). A rapid surge in TURF adoption by FCs was observed within the first ten years of the program, with the number of active TURF-FCs plateauing near 500 between 2009 and 2013. A second uptake in the number of active TURF-FCs was observed starting in 2014. There were no periods showing a significant decline in active TURF-FCs. Geographically, TURF-FCs are unevenly distributed along the Chilean coast (Figure 3.2 B). The Los Lagos region in southern Chile has the highest concentration of TURF-FCs, followed by Coquimbo in the north, with Aysen and Biobio, also in the south, closely behind. The ratio of active to initiated TURF-FCs varies regionally, with Coquimbo displaying the highest ratio of active to initiated TURF-FCs.

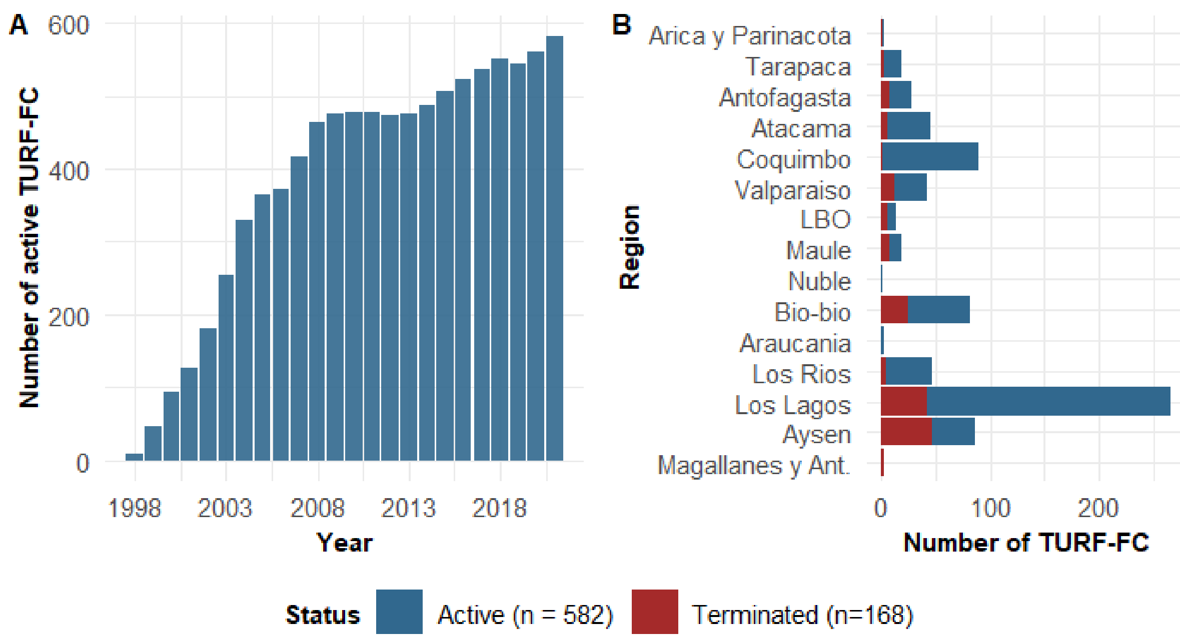

Figure 3.2: Number of active TURF-FCs per year (A) and number of active and terminated TURF-FC per region in Chile (B) organized from North to South (top to bottom).

The survival function for the Chilean TURF program, depicted in **Figure 3.3 A**, illustrates the probability that a TURF-FC will last beyond a specific age. For example, there is approximately a 75% chance that a TURF-FC will survive for more than 15 years. Censored observations, indicated by ticks on the curve, represent TURF-FCs that were still active by the study's end, spanning all ages. No significant differences were found in the survival curves across different TURF-FC cohorts (**Figure B.6**).

The hazard function, shown in **Figure 3.3 B**, displays the probability of a TURF-FC's termination for each age, given its survival up to that point. The highest estimated termination hazard, at 5.6%, occurs between the third and fourth years, aligning with TURF-FCs that had their management projects approved and were deemed active during the year their baseline technical study was conducted, including the subsequent three-year grace period, but failed to submit further reports. Another hazard peak, at 3.2%, arises between the seventh and eighth years. Overall, the termination risk decreases as TURF-FCs age.

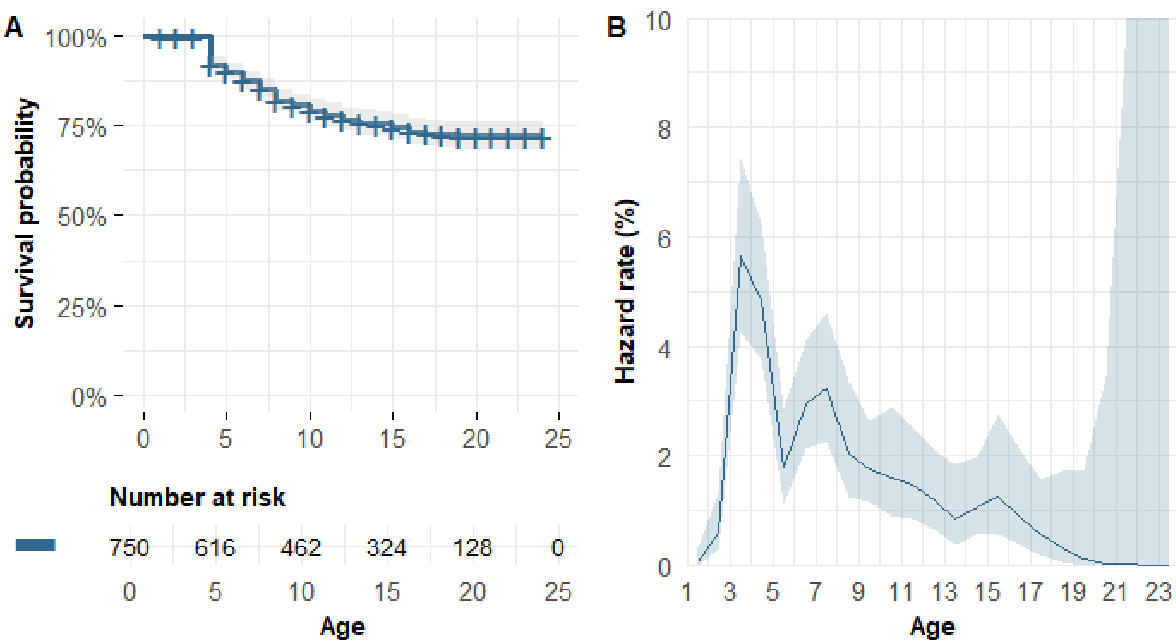

Figure 3.3: Survival curve estimated employing the Kaplan-Meier estimator across all TURF-FCs with the risk table below (A) and hazard curve obtained using non-parametric smoothing of the termination rate at a given age (B).

Fishing dynamics in long-term versus short-term TURF-FCs

Results from our mixed-effects model indicate initial condition differences between long-term and short-term TURF-FCs, without significant differences in their trends as they age. Specifically, long-term TURF-FCs start with significantly higher exploitable abundances (Long-term = 106.7, 95% CI = [56.67, 156.76], $p < 0.001$, Column 1, **Table 3.2**) and quotas (Long-term = 10.61, 95% CI = [6.57, 14.66], $p < 0.001$, Column 2, **Table 3.2**) compared to short-term TURF-FCs. Initial catch does not significantly differ between long-term and short-term TURF-FCs and both groups show a significant decline in catch with age (Age = -3.4, 95% CI = [-6.80, -0.09], $p < 0.001$, Column 3, **Table 3.2**). This decrease is more pronounced in short-term TURF-FCs, though not significantly different from the decline observed in long-term TURF-FCs (Long-term × Age = 2.54, 95% CI = [-0.83, 5.90], $p = 0.14$, Column 3, **Table 3.2**). There are no significant differences in initial revenue from loco between short-term and long-term TURF-FCs, with no observed significant trends in revenue for either group (Column 4 in **Table 3.2**). The fixed effects, including age and survival category, account for 1.7% of the variance in exploitable abundance and catch, 2.5% in quota, and only 0.5% in revenues. Incorporating random effects to account for variability across TURF-FCs allows our model to explain 62% of the variance in exploitable abundance, quota, and catch, and about 40% of the variance in revenues, underscoring the substantial variability among TURF-FCs.

Table 3.2: Mixed-effects models to evaluate differences in initial levels and trends of Loco exploitable abundance, quota, catch, and revenue in the long-term and short-term TURF-FCs.

	Dependent variable:			
	Abundance (Thousand units)	Quota (Thousand units)	Catch (Thousand units)	Revenue (Million CLP)
Age	5.63	0.08	-3.44^{**}	-0.63
	(4.02)	(0.31)	(1.71)	(0.53)
Long-Term	106.71^{***}	10.61^{***}	13.57	1.59
	(25.52)	(2.06)	(8.62)	(2.34)
Long-Term x Age	-3.94	-0.06	2.54	0.66
	(4.06)	(0.32)	(1.71)	(0.53)
Constant	85.67^{***}	5.84^{***}	30.52^{***}	6.45^{***}
	(21.96)	(1.78)	(8.21)	(2.24)
Observations	3,233	2,714	2,593	2,360

Note: *p<0.1; **p<0.05; ***p<0.01

Survival probability across TURF-FCs with distinct conditions for collective action

The Hopkins statistic, which assesses the clustering tendency of our dataset on collective-action indicators, was 0.75, indicating a strong clustering tendency. Cluster 1 is distinguished by having the highest initial exploitable abundance and the largest TURF sizes compared to Clusters 2 and 3, as confirmed by Tukey post-hoc comparisons ($p < 0.01$) (**Figure 3.4 A**). It also exhibits higher levels of upwelling and user numbers than Cluster 2 ($p < 0.01$), but similar to Cluster 3. Additionally, Cluster 1 shows intermediate levels of FC age, poverty, and distance to the nearest city and cove, differing significantly from Clusters 2 and 3 in these metrics ($p < 0.01$ to $p < 0.05$).

Cluster 2 has initial exploitable abundance and TURF sizes significantly lower than Cluster 1 but similar to Cluster 3 ($p < 0.01$). It experiences the lowest upwelling ($p < 0.01$), has the smallest and youngest FCs, and displays the lowest poverty levels, with its TURFs being the farthest from large cities and fishing settlements ($p < 0.01$ to $p < 0.05$). Conversely, Cluster 3, with intermediate levels of upwelling, includes the most established FCs ($p < 0.05$), highest poverty levels ($p < 0.01$), and TURFs closest to fishing coves and cities ($p < 0.01$). Detailed statistical comparisons are available in **Table B.1**.

There is a spatial pattern in the distribution of clusters (**Figure 3.4 B**). Cluster 1 is uniformly distributed along the Chilean coast, while Cluster 2 is predominantly found in central Chile. In contrast, Cluster 3 is mainly located in the northernmost and southernmost regions of the country. The survival curves for the three clusters, depicted in **Figure 3.5 A**, show that the survival probability of Cluster 2 is significantly lower than those of Clusters 1 and 3 (log-rank test, p-value < 0.01). Cox model comparisons reveal that TURF-FCs in Cluster 2 are 4.25 times more likely to terminate at any given age

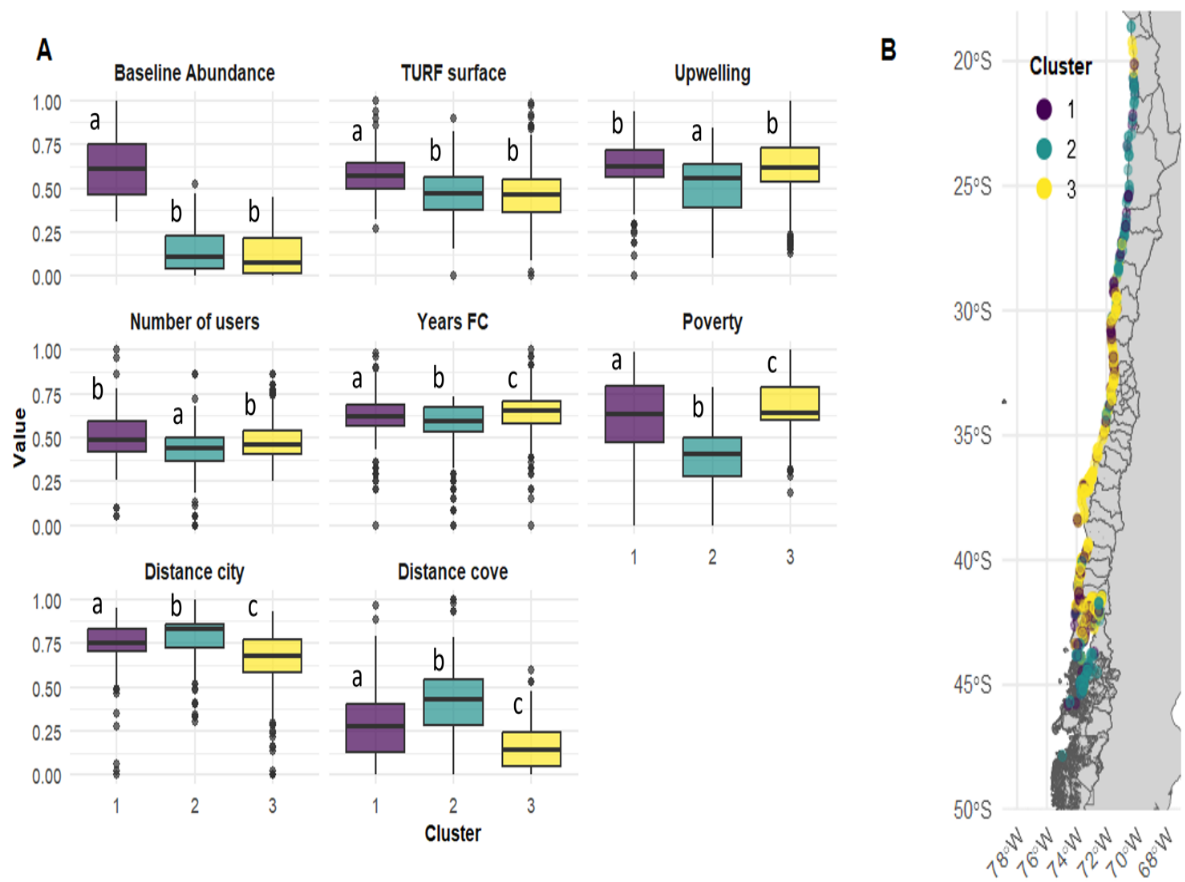

Figure 3.4: Boxplots showing the distribution, per cluster, of scaled indicators used to characterize conditions for collective action across TURF-FCs (A) and the geographic distribution of TURF-FCs colored based on their assigned cluster (B). Labels on boxplots show significant differences between clusters according to ANOVA and post-hoc Tukey tests.

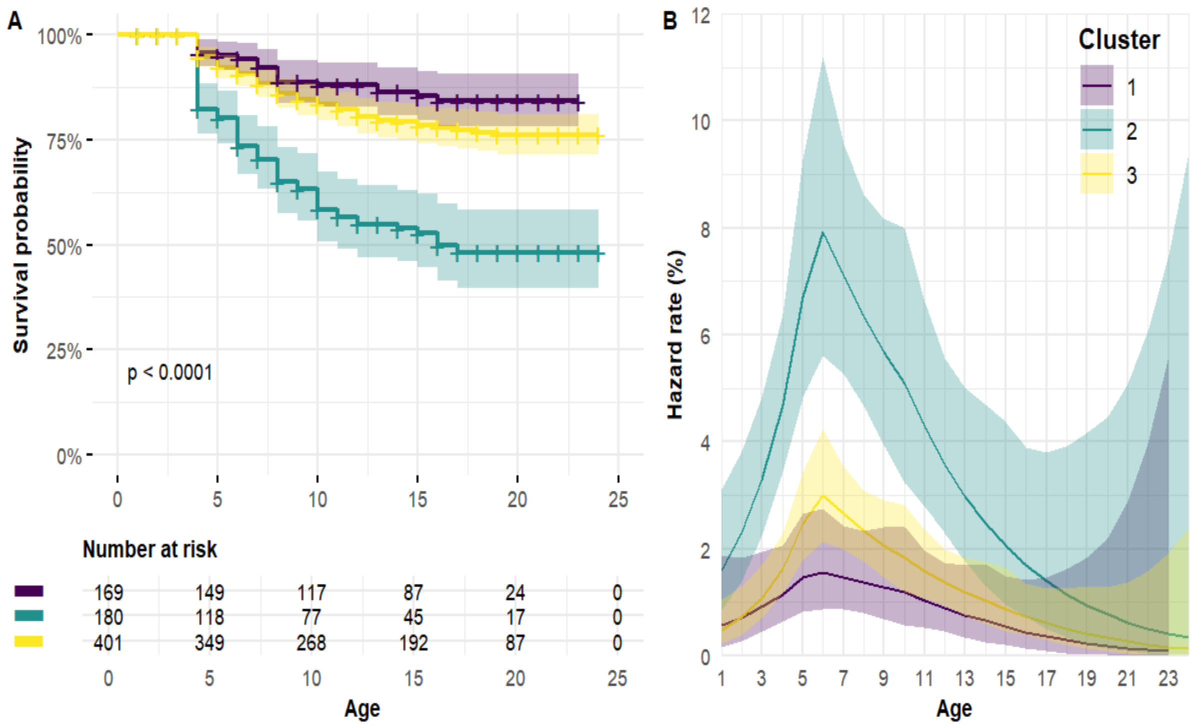

Figure 3.5: Pannel (A) shows the survival curve estimated employing the Kaplan-Meier estimator for each cluster differentiated by color and the corresponding risk table below. Pannel (B) displays the hazard curve per cluster obtained using non-parametric smoothing of the termination rate at a given age.

compared to those in Cluster 1 (95% CI = [2.62, 6.87], $p < 0.001$), as shown in **Figure 3.5 B**. However, the termination risk for Cluster 3 compared to Cluster 1 is not significantly different (95% CI = [0.96, 2.47]). Our model satisfies the proportional hazards assumption ($X^2(2) = 0.75$, $p = 0.69$), indicating consistent hazard ratios across ages between clusters.

Discussion

Evaluating ABCM is crucial in addressing the challenges posed by the contested and declining health of marine territories. Through survival analysis of 750 Chilean TURF projects spanning 23 years, we have demonstrated the efficacy of survival probability as an indicator of ABCM success. Notably, our findings indicate a 75% probability of FCs successfully sustaining their TURFs beyond 15 years, suggesting the benefits of TURF management generally surpass the costs to fishers. Survival analysis revealed critical periods of termination risk, particularly in the early years, a risk further intensified in areas with low initial resource abundance. Yet, a more nuanced analysis reveals that other conditions conducive to collective action can mitigate these risks. This aligns with the SES framework, suggesting variables such as natural productivity, long-established user groups, high resource dependency, and low surveillance costs as pivotal for long-term TURF sustainability, even in low-abundance scenarios. Our study not only affirms the value of ABCM for FC but also illustrates how SES principles can empirically evaluate and enhance ABCM outcomes.

Our analysis of the Chilean TURF system indicates that FCs generally benefit from and are committed to sustaining their TURFs over time, offering a positive evaluation of the policy. This finding aligns with holistic evaluations of the TURF program that utilize multiple indicators (Gelcich et al., 2019) but contrasts with studies focusing on catch declines, which may suggest less favorable outcomes (Anguita et al., 2020; Beckensteiner et al., 2020; Cerda & Stotz, 2022). These discrepancies underscore that FC motivations for maintaining TURFs extend beyond direct economic gains (Gelcich et al., 2017). Moreover, interpreting catch data as a performance indicator is complicated by the challenge of accurately modeling stock productivity, especially within the data constraints of the Chilean TURF system.

Our investigation into fishing dynamics replicated reported declines in Loco catch. Importantly, we observed stable revenue and abundance levels, suggesting that catch decreases do not necessarily indicate unsustainable practices. Sustainable use may manifest as varying catch trends, influenced by stock productivity, initial conditions, and the balance between fishing costs and revenues (Norman-López & Pascoe, 2011). This highlights the complexity of defining and measuring ABCM success. While assessments focusing on specific outcomes like sustainable fishing practices and economic viability are key for identifying ABCM's direct impacts on FC and society, comprehensive evaluations like those offered by survival analysis are essential for capturing patterns in the overall benefits and challenges of ABCM, remaining neutral about the specific advantages to different user groups.

The relationship between low initial abundances and reduced survival rates in TURF projects, coupled with increased termination risks during the early stages, indicates that delayed investment returns might be a key factor behind TURF failures. TURF management entails initial expenses, including baseline studies, fees, and surveillance measures (Jarvis & Wilen, 2016). Although sustainable fishing practices are expected to eventually boost and stabilize profits, the urgent need for quick returns on these investments poses significant challenges, particularly in small-scale fisheries (Salas et al., 2007).

Low initial abundances in TURF projects can result from inherent low productivity or historical overextraction. When productivity is naturally low, management and exclusion efforts may not significantly increase returns on investment. In contrast, for areas with previously overexploited but inherently productive stocks, these efforts can lead to substantial improvements in returns, thereby supporting TURF survival. Our findings align with this distinction, showing that TURF-FCs with initially low abundances yet higher productivity and reduced surveillance costs, which likely mitigate poaching, are more likely to survive. This underscores the importance of understanding the dynamics of resource productivity and poaching pressures to accurately assess TURF profitability and sustainability. While proxies like upwelling indices and distance to fishing coves provide some insight, particularly in benthic fisheries affected by upwelling, a more precise characterization of stock productivity and connectivity is crucial for fully evaluating the long-term viability and economic rationale of TURF initiatives.

The concept of TURFs as investment extends beyond mere material costs and benefits. The notion of TURFs as investments transcends mere material costs and benefits. Shared norms among users, for instance, can reduce the social costs of coordination for resource management, increasing the likelihood that FC with such norms will sustain their TURFs over time. The SES framework helps navigate both material and non-material incentives for collective action in natural resource use contexts (Ostrom, 2009). By grouping TURF-FCs into clusters and analyzing differences in survival rates, we adopted a flexible and insightful approach to explore how does the viability of TURF projects over time varies across social and ecological conditions. Our findings suggest that collective action factors can effectively counterbalance the challenges for TURF maintenance imposed by low initial resource abundances.

collective action is theorized to exhibit a curvilinear relationship with the size and productivity of resource systems. Ostrom (2009) suggests that the costs associated with delineating boundaries, monitoring, and acquiring ecological knowledge can become prohibitively high in very large systems, whereas very small systems might not yield sufficient returns from product flows to justify these efforts. Our analysis revealed no clear relationship between the sizes of TURFs and their survival rates. Two clusters with similarly small TURF surfaces demonstrated contrasting survival rates, but the highest survival rate was observed in the cluster with the largest TURFs' surfaces.

Regarding productivity, Ostrom (2009) posits that while minimal resources may not produce adequate returns to justify management costs, a certain degree of resource scarcity is necessary to motivate user management. The cluster with the least productive TURF-

DISCUSSION

FCs exhibited the lowest survival rate, which aligns with the ascending phase of the theorized curvilinear relationship. Our study may not have captured the descending phase, likely because TURFs within the Chilean program rarely reach sizes or productivity levels that render management unnecessary or impractical. Moreover, competition between FC within the TURF system could shift this relationship from curvilinear to linear, as fishers may prioritize securing highly productive sites to ensure access before other FC does so.

User group size is often highlighted as a relevant factor influencing the outcomes of SES. Nonetheless, the relationship between group size and collective action is theorized to be context-dependent (Ostrom, 2009). While coordination costs and social uncertainty generally rise with group size, larger groups may enhance monitoring and management capabilities in complex management contexts. Our analysis suggests that clusters with higher survival rates often comprise larger and more established user groups. This implies that, within the intricate management environment of TURFs, larger groups possess superior management capabilities. Furthermore, the longevity of these groups likely mitigates the challenges associated with size, such as social uncertainty, by fostering the development of shared norms and trust among users.

Our findings indicate that TURF survival is more likely in areas with higher poverty levels, aligning with the notion that a greater reliance on resources for livelihoods justifies the costs of organizing and maintaining self-governing systems (Ostrom, 2009). This result adds to empirical evidence that documents a positive correlation between resource dependence and participation in resource management (Ann Zanetell & Knuth, 2004; Lise, 2000). The significance of this relationship is accentuated in contexts where self-organization leads to the exclusion of outsiders, such as in TURFs, but may not be as pronounced in other forms of customary tenure that restrict use to group members

(Cinner et al., 2007), underscoring the nuanced incentives in ABCM.

Our results challenge the notion that proximity to large markets necessarily undermines collective action, revealing that TURF-FCs near large city markets exhibit higher survival rates. This suggests that market access might actually support enhanced management and TURF maintenance through increased commercial opportunities. This finding is consistent with evidence that effective self-organization and management correlate with market access in various settings, particularly in the presence of strong leadership (Elsler et al., 2022; Epstein et al., 2021; Kaganzi et al., 2009; Rustagi et al., 2010). Notably, TURF-FCs near major markets exhibiting high survival rates also presented long-established user groups, highlighting the potential role of robust organizational structures.

The context of Chilean TURFs may diverge from that in more traditional tenure management systems, where intrinsic motivations and social norms, often underpinned by taboos and religious beliefs, are central to sustaining collective action and could be more vulnerable to the disruptive effects of market incentives (Cinner et al., 2007, 2012). Our findings highlight the complex impact of market access on self-governance and indicate that the success of ABCM, especially under strong market influences, relies on a blend of self-organization capabilities and the specific informal institutional context.

Our results indicate that TURF-FCs located farthest from fishing coves have the lowest survival rates, underscoring poaching as a significant challenge for TURF sustainability in Chile (Gelcich et al., 2017; Oyanedel et al., 2018; Romero et al., 2022). This emphasizes the need for government support in enhancing patrolling and enforcing anti-poaching measures to ensure that fishing communities can fully benefit from their TURFs (Davis et al., 2017; Quezada & Chan, 2023).

Previous research on the Chilean TURF system often emphasizes landings as a key outcome or uses them to identify active TURFs for sampling. Given the likelihood of misreporting in officially reported catches, especially for locos (Oyanedel et al., 2018), reliance on such data could introduce biases. However, our study did not prioritize catch data as the main outcome, reducing concerns about misreporting effects. Instead, we defined TURF activity based on report submissions, a crucial maintenance requirement and an indicator of FCs' commitment to their TURFs. Our analysis of active and terminated TURFs under this criterion uncovered significant patterns, such as an increase in TURF activity around 2015 linked to new aquaculture-friendly regulations (D.S. N 96-2015) and higher survival rates in the frequently studied regions of Coquimbo and central Chile (Gelcich et al., 2019). These observations highlight a potential bias in the literature towards long-standing projects and underscore the value of incorporating insights from both terminated and ongoing projects. Our methodology reduces such biases and broadens the sample size compared to prior studies, offering a richer understanding of the variability in ABCM outcomes across different social and ecological contexts.

While expanding the analysis scale, we sacrifice the ability to characterize each TURF and FC's unique social and ecological attributes. Leadership is a critical factor influencing ABCM outcomes, including in the Chilean TURF system (Crona et al., 2017; Epstein et al., 2021). However, our study lacked a suitable measure for leadership, leaving its impact unexplored or confounded. This underscores the need for multi-scale studies that capture broad patterns and delve into specific dynamics, while remaining vigilant of potential biases

The SES framework proved useful in navigating the complexity of incentives faced by FC to sustain their TURF and explaining patterns of TURFs' survival rates. Our findings generally align with prior applications of the SES framework to ABCM, but also revealed nuances related to how ABCM success is defined and assessed in the literature. Users willingness to maintain a TURF over time is a reflection of users deriving net benefits from their collective action efforts. Thus, variability in survival metrics can better explore the social and ecological conditions that shape the costs and benefits of collective action in ABCM compared to variables often used in such studies, such as catch, income, abundance, or biomass. However, focusing solely on survival does not capture the specific contributions of ABCM to fishing communities and broader societal goals. It is crucial to recognize that users' self-organization may not always aim at sustainability (Cinner et al., 2012). Therefore, while survival analysis is valuable for gauging the overall benefits of ABCM and exploring the drivers of collective action, it should be complemented with evaluations of sustainable fishing practices, social outcomes, and users' perceptions to fully understand the impact of ABCM on sustainable resource use.

Our results underscore the value of ABCM to fishers, identifies low initial resource abundance as a challenge to users to sustain ABCM experiences that can be mitigated under conditions that favor collective action, and provides empirical support to the SES framework as a tool to assess variability in ABCM outcomes.

Chapter 4

Markets and management mediate coping responses to weather-induced livelihood shocks and cascading effects in socio-ecological systems

Abstract

Natural resource users' ability to adjust their harvesting across space, resources, and time is crucial for mitigating the impacts of weather-induced livelihood shocks. These coping responses unfold within complex and interconnected social-ecological systems, highlighting the importance of understanding how management practices and market dynamics mediate them and their cascading effects. However, robust empirical evidence on the effectiveness of these responses and the conditions that shape them, particularly in Global South contexts, remains scarce. This study examined the response of boats in southern Chile to livelihood shocks caused by a massive harmful algal bloom in 2016, which significantly impacted clam fisheries. Using landing data, we analyzed how clam-reliant (n = 470) and non-clam-reliant boats (n = 143) adjusted their landings during various stages of the event, characterized by spatial-clam closures and a strike that halted all fishing activities in the region. To isolate the impacts of these shocks from seasonal variations in fisheries, we compared the evolution of landings within the affected season, before and after the onset of the shocks, to similar weeks in previous, unaffected seasons. Employing equivalent approaches, we quantified changes in landings, the number of boats, and local fishery prices, and qualitatively analyzed the influence of management practices and market dynamics on these changes. Our findings reveal that operating boats demonstrated significant mobility across space and resources, with early increases in clam prices incentivizing spatial mobility, including for boats initially unaffected by the closure of their fishing grounds. The existence of a regional urchin quota also motivated non-clam-reliant boats to increase their landings to avoid losing quota shares. This study empirically underscores the critical role of markets and management in shaping coping responses and illustrates their ripple effects across actors and ecosystems.

Introduction

Extreme weather events, such as heatwaves, wildfires, and floods, pose increasing risks to people and ecosystems, especially in regions where livelihoods and subsistence depend on the harvesting of natural resources (Kendon et al., 2019; Mora et al., 2018; Oliver et al., 2018). Strengthening the resilience of these systems against extreme weather is crucial for supporting poverty alleviation and food security objectives. A key component of this resilience is harvesters' ability to smooth income through diversified labor activities (Chuang, 2019; Mulungu & Kilimani, 2023). When shocks temporarily disrupt access to specific resources and areas, individuals may adapt by reallocating harvesting efforts across space, resources, and time within their primary sector (Cinner et al., 2009). These coping responses unfold within complex and interconnected social-ecological systems (SES), where human behaviors are influenced by multiple factors and can trigger cascading effects (Folke, 2006). Understanding these coping responses through an SES perspective is vital for guiding effective adaptation and transformation strategies, thereby preventing poverty traps, conflicts, and ecosystem degradation in the face of abrupt changes.

Small-scale fisheries provide a compelling arena to study these coping responses from a SES perspective. Exhibiting a diverse range of target species and fishing grounds, these fisheries afford opportunities for income smoothing within the sector when confronted with spatially and temporally concentrated extreme-weather-induced livelihood shocks. The complexity of small-scale fisheries is further marked by dynamic markets and overlapping regulations (Short et al., 2021). These characteristics offer valuable insights into how various factors, such as markets and management, interact to facilitate or hinder coping responses and influence their cascading effects. The insights garnered from studying small-scale fishing communities are particularly significant, given their frontline position in the face of environmental and climate change.

There has been significant interest in understanding how individuals and communities respond to environmental change in fisheries. Researchers have utilized surveys on historical or hypothetical scenarios (Cinner et al., 2009; Gianelli et al., 2021; Papaioannou et al., 2021; Salgueiro-Otero et al., 2022), syntheses of case studies (Green et al., 2021; Ilosvay et al., 2022), analysis of observational data (Gonzalez-Mon et al., 2021), and the development of analytical frameworks (Cinner, Adger, et al., 2018). These approaches often highlight that the displacement of effort to alternative fishing grounds and resources can mitigate the impacts of environmental change. This supports the concept that a diversified fishing portfolio acts as a risk buffer, stabilizing income fluctuations in fisheries (Anderson et al., 2017; Finkbeiner, 2015; Kasperski & Holland, 2013; Sethi, 2010). However, current research predominantly focuses on adaptive responses to gradual changes, which differs from coping with sudden shocks (Ojea et al., 2023).

Only a few studies provide robust empirical evidence that mobility across space and resources is pivotal for coping with actual extreme-weather-induced livelihood shocks and regime shifts (Cline et al., 2017; Fisher et al., 2021; Jardine et al., 2020; Liu et al., 2023). Nonetheless, this evidence predominantly comes from case studies in the Global North, where management capacity, governmental support, and market stability may differ from those in Global South contexts. Indeed, the benefits of livelihood diversification in small-scale fisheries within low- and middle-income countries have been called into question (Roscher, Allison, et al., 2022). Therefore, it remains unclear whether spatial and resource mobility are effective coping strategies for weather-induced-livelihood shocks in contexts differing from those in the Global North.

Effort reallocation across space, resources, or time can be constrained or facilitated by combinations of management regulations and market dynamics (Papaioannou et al., 2021). A recent simulation study showed that management can be pivotal in averting maladaptation and optimizing economic and ecological outcomes during abrupt changes (Beckensteiner et al., 2023). Likewise, market dynamics shape the opportunities available to fishers. For example, spatial mobility may become less viable when the economic costs associated with shifting fishing grounds or resources outweigh the potential revenue gains. Market dynamics can also ripple, creating incentives for fishers that are not directly affected by the shock to adjust their behaviors. Thus, identifying mechanisms via which market and management shape coping responses and documenting their ripple effects is crucial to consider the broader consequences and drivers of coping when designing strategies to deal with environmental change. Nonetheless, current analyses rely on aggregated diversity indices that limit our understanding of how effort displacement is influenced by fishery-specific market and management conditions (Cline et al., 2017; Fisher et al., 2021; Jardine et al., 2020; Liu et al., 2023).

In this study, we use landing data from multiple fisheries to investigate the responses of small-scale fishing boats to the livelihood shocks induced by a massive harmful algal bloom (HAB) event that occurred in Los Lagos region of Chile during March to September 2016. Triggered by anomalous warm weather, this event involved spatial-resource-specific closures and a region-wide strike that suspended fishing activities for nearly three weeks. Given the significant impact of the closures on the clam fishery, our primary focus is on boats historically reliant on clam harvesting. To capture ripple effects, we also analyze changes in the landings of boats targeting two alternative resources available during the HAB event: urchins and *Luga* seaweed. To dissect the effects of the different shocks and explore the roles of management regulations and market dynamics, we divided the HAB event timeline into five distinct stages, each differing in the spatial extent of closures and strike occurrence.

Our main analysis evaluates changes in individual boats' weekly landings of clams, urchin, and Luga across different fishing areas. The primary analytical challenge in studying coping responses to shocks using landing data is isolating the effects of the shocks from the seasonal variations inherent in fishing landings. To overcome this challenge, we leveraged the weekly dynamics of the livelihood shocks and used unaffected previous fishing seasons to account for seasonal variations in fisheries. Specifically, we compared the evolution of landings between the weeks right before and after the onset of the shock within the affected season to the evolution observed between corresponding weeks within unaffected seasons. Employing a methodologically consistent approach, we also evaluated market dynamics by measuring changes in total landings, the number of boats, and prices in local clam fisheries and qualitatively assessed how these changes elucidate spatial mobility responses among clam fishers. Additionally, we quantified changes in the urchin fishery relative to past seasons to explore the potential role of the regional quota in mediating responses among both clam-reliant and non-clam-reliant boats. This case study offers valuable insights by providing evidence of coping responses in the Global South and underscoring the significance of management regulations and market dynamics in shaping coping responses and their ripple effects in social-ecological systems.

Case study

The Los Lagos region is the most prolific region for benthic fisheries in Chile and is home to the largest number of registered small-scale fishers in the country (IFOP, 2022). It also serves as a major center for aquaculture, particularly for salmon and shellfish farming (Niklitschek et al., 2013; SUBPESCA, 2003). In the period spanning 2015 to 2016, the region experienced the convergence of an El Niño event and the positive phase of the Southern Annular Mode, creating ideal conditions for a massive HAB (Leon-Munoz et al., 2018). During February 2016, the proliferation of the microalga *Pseudochattonella verruculosa* resulted in the mass mortality of farmed salmon, significantly impacting the salmon industry (Mardones et al., 2021; Trainer et al., 2020). In the following weeks, another microalgal bloom occurred, this time involving *Alexandrium Catenella*, which produces biotoxins that accumulate in filter-feeding organisms like bivalves, rendering them toxic for human consumption.

To mitigate health risks, the Chilean government implemented emergency closures prohibiting the harvesting and commercialization of filter-feeding organisms from affected fishing grounds. These closures began in the south of the region and expanded northward as new spots with high toxin concentrations were identified. The unpredictability of HABs makes it unlikely that fishers could have anticipated this phenomenon and the spatial or temporal distribution of the closures, especially given the unprecedented scales of the event in 2016 (Trainer et al., 2020; Ugarte et al., 2022). Although the precise economic toll on small-scale fisheries has not been quantified, the loss of income within coastal communities incited social unrest, culminating in a three-week strike at the beginning of May 2016 (Ugarte et al., 2022; Valdebenito-Allendes, 2018). Protesters demanded financial assistance to cope with the livelihood impacts and advocated for stricter regulations on the salmon industry, which is suspected of contributing to the HAB through nutrient discharges (Armijo et al., 2020).

Clams were among the resources affected by the closures during the HAB event, along with other filter-feeding organisms and specific seaweeds. The clam fishery holds great regional importance, both in terms of volume and profits, representing approximately 80% of the national clam landings (Molinet et al., 2011a). This fishery operates year-round and is primarily regulated by minimum size restrictions. Clams are both sold domestically and exported for direct human consumption and processing. This fishery is exclusively carried out by small-scale fishers, typically through diving. To land specific resources, fishers require resource-specific permits, many of which were obtained when these fisheries were open to new applicants in the past. These fishers commonly hold permits for various other resources that require similar fishing skills, including bivalves, snails, urchins, octopus, and a local seaweed known as *Luga* (Molinet et al., 2011a). Fishers are permitted to catch and land their resources anywhere within the region for which they are registered.

The clam fishery is highly seasonal with higher landings between September and December and a sharp drop in landings due to end-of-the-year holidays. While total landings and profits for clams did not show a noticeable reduction during the season hit by the HAB event when compared to past seasons, weekly dynamics displayed major declines that correlated with the timing of closures and the strike (**Figure 4.1**).

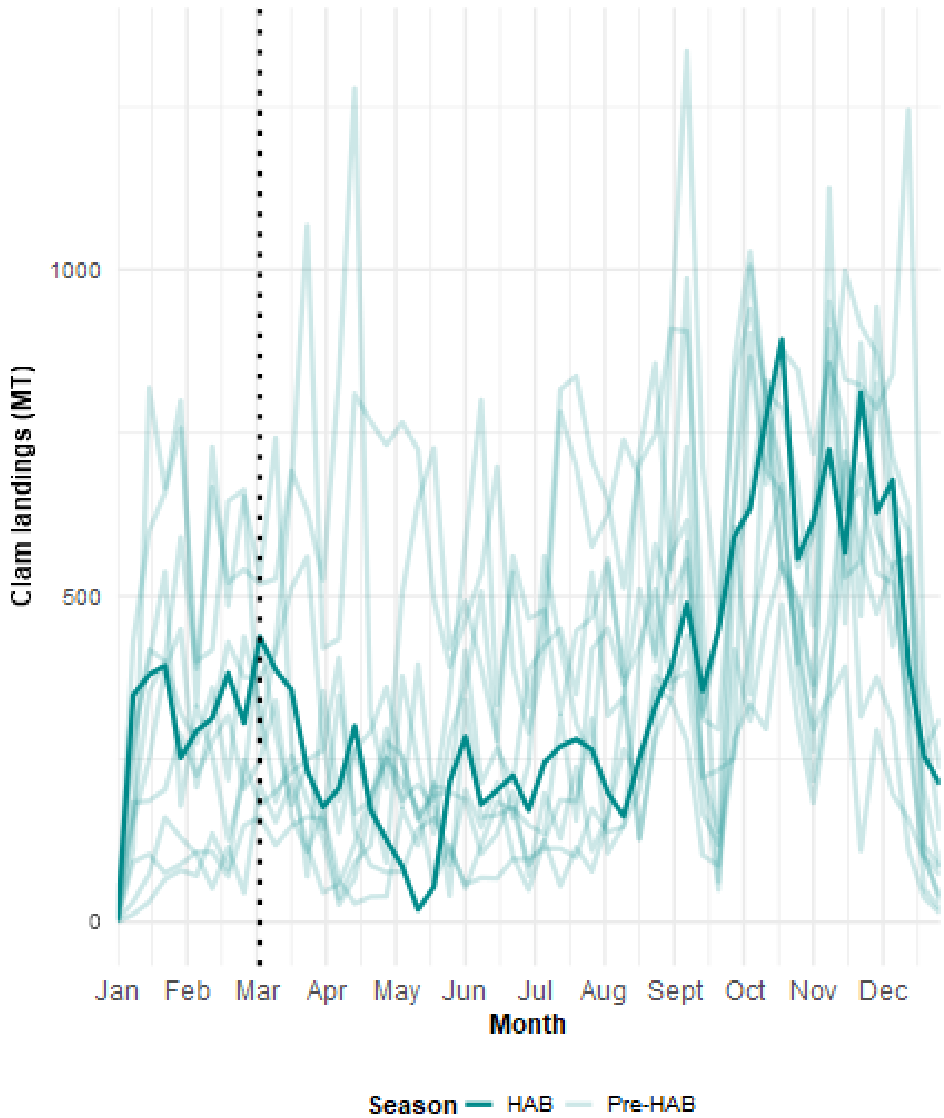

Figure 4.1: Clam landings in MT in the Los Lagos region in the season affected by the HAB event (dark cyan) and previous ten seasons since September 2005 (light cyan). The dotted line shows the week when the first closure due to HABs was issued.

Methods

Data

To characterize changes in landings, we used landing data compiled by the Chilean National Service of Fisheries and Aquaculture (SERNAPESCA), which comes from reports submitted by fishers as mandated by law. This database provides detailed information on the dates of fishing trips, the quantity of metric tons (MT) landed, the type of resource, a vessel identification number, and the port of landing. Access to these data is available upon request to SERNAPESCA. We used data from the period between 2012 to 2016 for analyses at the boat level since changes to data collection procedures were implemented in 2012 to facilitate self-reporting. For analyses at the aggregated fishery level, we employed data dating back to 2005, coinciding with the start of a major management plan in the region (XI REGIONES, 2005).

To determine the geographical and temporal scope of the closures, we digitally mapped the spatial information provided in government-issued resolutions. These resolutions listed the species and fishing grounds subject to harvesting restrictions from the date of issuance until the closure was rescinded through a subsequent resolution. These official resolutions are accessible on the BCN website. Data on weekly prices were provided by the institute of Fisheries Development of Chile (IFOP). Through on-site observers, IFOP collects price data informed by fishers and buyers after transactions made in the main ports of the region. We converted prices to 2013 USD using Chilean consumer price indices and USD conversion rates informed by the Chilean Central bank.

Spatial and temporal scales of analysis

To effectively capture coping responses to dynamic livelihood shocks, it is crucial to establish appropriate scales of analysis. Spatial data for the closures is defined within the ocean, while spatial data for landings is associated with terrestrial ports. At the vessel level, there is no specific information that identifies the fishing ground from which a particular landing originated. Additionally, vessels registered in the Los Lagos region are granted the flexibility to fish in any fishing ground and to choose any port within the region for their landings. Therefore, the association of landings with a specific fishing ground is not straightforward. To overcome this challenge, we used the polygons designed by Molinet et al. (2011b) for the spatial analysis of benthic fisheries within the region depicted in **Figure 4.2**. They represent the primary fishing areas that contribute to landings at the main ports in the region and were crafted with input from fishing trips, surveys, and expert insights.

In practice, we assumed that landings reported in a port located within a specific polygon were sourced from within that polygon's boundaries. It is worth noting that some fishers from the Los Lagos region are authorized to fish for clams and other resources in the adjacent Aysen region. Despite this, their landings are still reported in the Los Lagos region (XI REGIONES, 2005). We aggregated the six polygons into three major fishing areas based on the timing at which they were hit by the closures: South, Inner, and North, respectively indicated in blue, red, and green in **Figure 4.2**. Landings occurring within the gray-shaded polygon labeled `NA` were excluded from our analysis because clam landings are minimal and this polygon was only affected by closures in its southernmost area, where no clam landings were reported since 2012. As closures presented weekly variations, we used weeks as our primary time unit (**Figure 4.3**). We defined the HAB

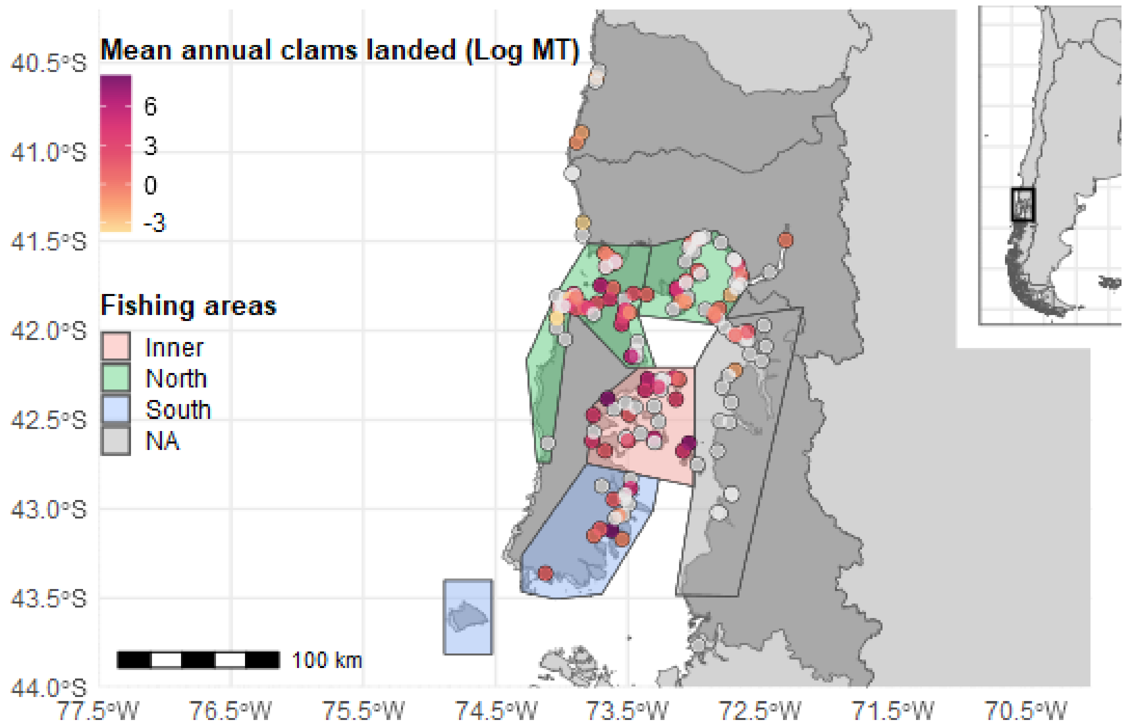

Figure 4.2: Map of the Los Lagos region (depicted in dark gray) displaying polygons previously outlined by Molinet et al., 2011, denoting the primary fishing grounds supplying landings to the main ports within the region (depicted as circles). The color of each circle corresponds to the logarithmic mean annual clam landings (MT) between 2012 and the HAB event. Ports without clam landings are depicted in white. The color shading of the polygons represents the consolidation of these regions into one of the three fishing areas based on the timing of the closures

event as starting in the week when the first closure was issued, specifically on March 3rd, 2016, and analyzed responses until the end of the season of the fishery under analysis. For the clam fishery, which is fished year-round, we considered seasons that went from September 16th of one year to September 15th of the next. This choice positions the start of the HAB event in the middle of the time window, which is convenient for analysis purposes.

Acknowledging that the different livelihood shocks can elicit distinct responses, we segmented the timeline of the HAB event into five stages based on the primary shocks that occurred during specific weeks. Stage I covers the period from week 0 to week 5 following the initial closure. During this stage, closures were primarily concentrated in the South fishing area, while most of the Inner area and the entire North area remained open. Stage II extends from week 6 to 8 after the initial closure. During these three weeks, the South and Inner fishing areas briefly reopened, while the North area experienced a rapid increase in closures, affecting a substantial portion of its fishing area.

Weeks 9, 10, and 11 since the first closure are Stage III, which is marked by the strike, with the entire fishing sector stopping operations in support of affected fishers. Between week 12 and 14 after the first closure, which we refer to as Stage IV, approximately three-quarters of each of the three major fishing areas remained closed. The last stage we examined is Stage V, spanning from week 15 after the initial closure until the end of the season. During this period, all fishing areas gradually started to reopen.

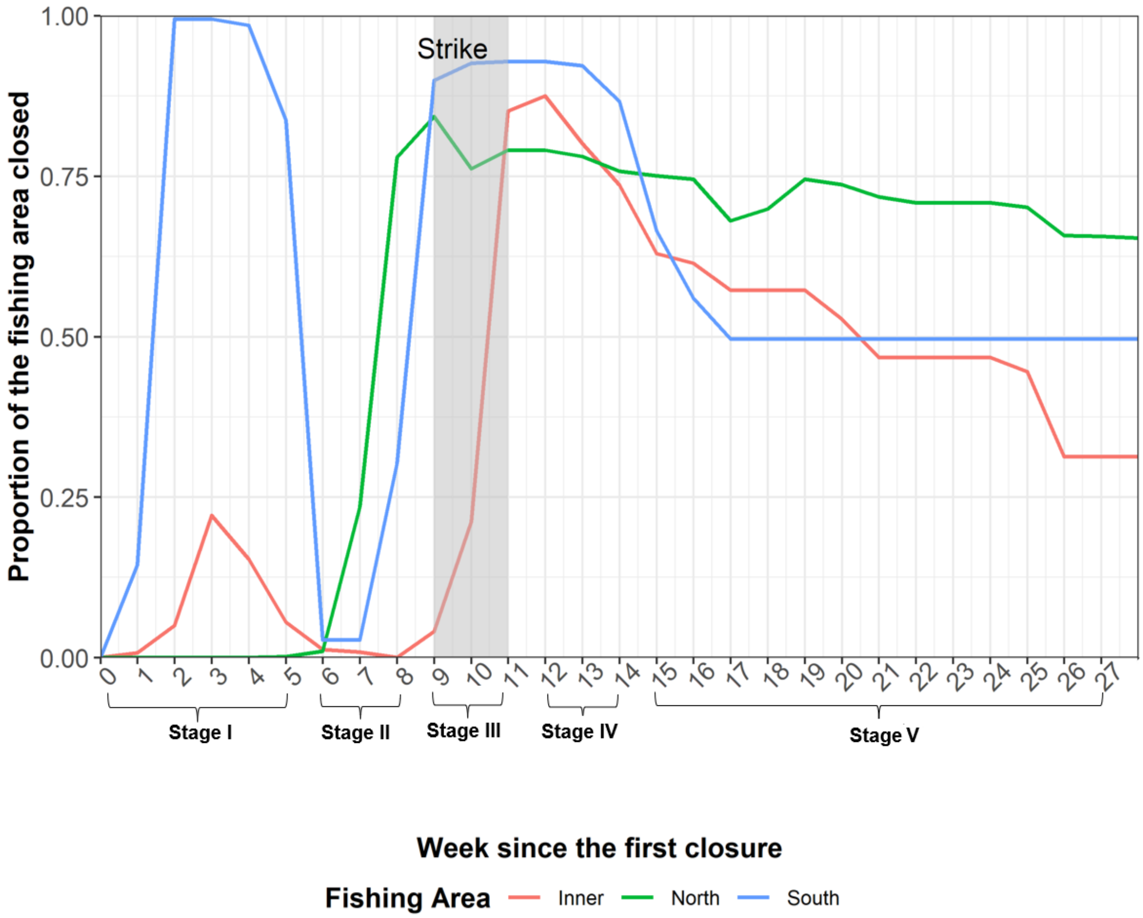

Figure 4.3: Timeline of the HAB event, showing the portion of each fishing area closed in each week since the closures started, the timing of the strike, and the weeks considered within each stage.

Outcome variables and samples

Our primary objective was to evaluate three specific types of responses exhibited by fishing boats during the various stages of the HAB event: spatial mobility, temporal effort displacement, and shifts to alternative resources. Spatial mobility involves changes in the locations where boats typically land clams. Temporal effort displacement is reflected by a rise in landings before or after a shock. Shifts to alternative resources are marked by increased landings of resources offering viable substitutes for clams.

METHODS

We specifically focused on urchin and *Luga*, as these resources were most frequently targeted by clam-fishing boats and remained unaffected by the closures, according to fisheries participation networks built using data from the pre-HAB period, defined between September 16^{th}, 2012, to March 3^{rd}, 2016 (**Figure C.1**). Although octopus emerged as another potential alternative, its landings were significantly lower than those of clam, urchin, and *Luga* (**Figures C.2, C.3, C.4**).

Since clam-reliant boats may have responded differently to each stage of the HAB event, based on whether their primary fishing area was being affected, we created three distinct samples of clam-reliant boats: those predominantly reliant on the 1) South, 2) Inner, and 3) North fishing areas. To construct these samples, we calculated the proportion of each boat's total clam landings from each of the three fishing areas over the pre-HAB period. We classified boats as either South-, Inner-, or North-clam reliant based on the area where they had the highest proportion of clam landings (**Figure C.5**).

Of the 654 boats that were classified into one of the samples, we only analyzed changes in landings for those that reported landings of any type of resource at least once during the HAB event. This filter was done to improve balance when comparing landings before and during the HAB event and resulted in 168 South-clam-reliant, 70 Inner-clam-reliant, and 232 North-clam-reliant boats. In the Results, we analyzed the differences between clam-reliant boats that continued fishing during the HAB event and those that fished in the pre-HAB seasons but not during the HAB event.

We also defined a sample of boats that might have experienced indirect impacts due to the shifts of clam-reliant boats to alternative resources during the HAB event. This sample consisted of boats that had reported landings of urchins, *Luga*, or both, yet did not report any pre-HAB landings of clams or any other resource directly affected by the closures. Of these, we also only considered boats that continued landing any type of resource during the HAB event to form the sample of non-clam reliant boats (n = 142).

For South-, Inner-, and North-clam reliant boats outcome variables were weekly individual clam landings in the South, Inner, and North areas to identify spatial mobility. To capture resource mobility for these boats, outcome variables were weekly individual landings of urchin and *Luga*. Since urchin is managed under a regional quota, we considered urchin landings in absolute terms and also as a percent of the total landings for a given season. For non-clam reliant boats, outcome variables included weekly individual landings of urchin and *Luga*.

Estimation of changes in boats' individual landings

To assess the mean responses of individual boats to the HAB event, we compare each boat's landing differences before and after the HAB's onset within the affected season (first difference), to differences in landings between equivalent weeks across the three preceding seasons, which we will refer as pre-HAB seasons' (second difference). This method estimates the HAB's impact while accounting for seasonal variability unrelated to the event. Our approach mirrors a difference-in-difference estimation, utilizing historical behavior in place of control units, a useful strategy to analyze weekly-scale impacts in the absence of adequate control units (Brodeur et al., 2021; Saavedra, 2023).

To provide an unbiased estimate of the HAB's impacts on individual landings, our method assumes that, in the absence of HAB-related shocks, boats' weekly landings during the HAB season would have followed similar trends as those observed in the pre-HAB seasons. This assumption, known as the parallel trend assumption in difference-in-differences analyses, is crucial for the validity of our approach (Angrist & Pischke, 2009). While this assumption cannot be empirically confirmed, we assessed its plausibility by graphically comparing average individual landings during the HAB season with those in the pre-

HAB seasons in the weeks leading up to the closures (**Figures C.2**, **C.3**, **C.4**, and **C.6**). We further examined the assumption's validity through dynamic difference-in-difference analyses detailed in the **Appendix**.

We combined difference-in-difference with fixed effects to account for unobserved characteristics of boats, week of the season, and seasons and to mitigate potential sources of bias. Our objective was to evaluate the impact of each HAB stage on the average weekly individual landings across various resources and fishing areas. To achieve this, we used the estimator presented in **Equation** (4.1).

We applied this estimator to our three clam-reliant boat samples, analyzing outcomes that included weekly clam landings in the South, Inner, and North fishing areas, weekly urchin landings (both absolute and as a percentage of the quota), and *Luga* landings. Using the same estimator, we examined weekly urchin and *Luga* landings for non-clam-reliant boats and total weekly landings for all four boat samples. Missing boat landings were filled with zeroes to maintain a balanced panel for specific resources and areas each week.

$$Y_{i,t,j} = \beta_0 + \beta_1 SI_{t,j} + \beta_2 SII_{t,j} + \beta_3 SIII_{t,j} + \beta_4 SIV_{t,j} + \beta_5 SV_{t,j} + \gamma_{i,t} + \theta_j + \epsilon_{i,t,j} \quad (4.1)$$

Where $Y_{i,t,j}$ represents the quantity of the resource, measured in MT, landed by boat i in week of the season t and season j; $SI_{t,j}$, $SII_{t,j}$, $Strike_{t,j}$, $SIII_{t,j}$, and $SIV_{t,j}$ are dummy variables that indicate whether week of the season t and season j correspond to each of the studied stages in the HAB season; $\gamma_{i,t}$ is a boat-week fixed effect that controls for boat i's characteristics that are fixed for a given week t across seasons such as fishing decisions based on weather conditions that vary seasonally or holidays; and θ_j controls for shocks that are common to all boats and weeks within a season.

We also conducted an estimation using the estimator in **Equation** (4.2). This analysis aimed to evaluate the overall impact of the entire HAB event on the average individual landings of each resource of interest for each of our four samples of boats.

$$Y_{i,t,j} = \beta_0 + \beta_1 HAB_{t,j} + \gamma_{i,t} + \theta_j + \epsilon_{i,t,j} \tag{4.2}$$

Where $Y_{i,t,j}$ represents the quantity of the resource, measured in MT, landed by boat i in week of the season t and season j; $HAB_{t,j}$ is a dummy variable that takes the value of one when week t is either the week when closures started or any week after that, and when season j corresponds to the HAB season; $\gamma_{i,t}$ represents a boat-week fixed effect that controls for characteristics specific to boat i that remain constant within a given week t across different seasons. It accounts for factors like a boat's fishing decisions based on weather conditions, which may vary seasonally, or holidays that impact fishing patterns; and θ_j is a season fixed effect that accounts for shocks or factors that affect all boats and weeks within a particular season j in the same way.

To ensure comparability across different seasons, we defined specific time windows for each resource, considering relevant management regulations. For *Luga*, a seasonal closure occurs in July and August, safeguarding crucial phases of their life cycle. While this closure consistently falls in these months every year, it may not start and end in the exact same week. Therefore, we considered landings occurring from the first landing to the end of Stage II in the HAB season and the equivalent calendar weeks in the pre-HAB seasons. Although the *Luga* season coincides with Stage III, Stage IV, and Stage V, landings across all analyzed seasons are typically too low for accurate estimation (**Figures C.2**, **C.3**, **C.4**, and **C.6**). Consequently, for *Luga*, we removed observations after Stage II for estimation.

On top of a seasonal closure, urchins are managed through a quota that is assigned annually to the entire region. Once the quota is reached, the fishery closes. To capture the constraint and the potential competition that the quota places on effort displacement towards urchins, we compared corresponding weeks across different seasons (i.e., the first week in a pre-HAB season to the first week in HAB season), accounting for variations in their timing rather than relying on exact calendar weeks. The number of weeks for this analysis is given by the longest season. For details on total landings, length, dates, and average prices for each of the seasons considered in the analyses (see **Tables C.1, C.2, C.3** for clam, *Luga*, and urchin, respectively).

Assesing changes in local clam and urchin fisheries during the HAB event

When assessing the impact of the HAB on total landings, the number of active boats, and prices within the clam fisheries, we cannot apply the same difference-in-difference analysis used at the boat level. At the individual level, we compared each boat's landing differences before and after the HAB event within the affected seasons against those in the unaffected seasons, providing multiple comparison units to estimate the HAB's average impact through the mean of these differences. However, at the local fishery level, we only have one affected unit, the local fishery during the HAB season, and a handful of unaffected units, the fisheries in the pre-HAB seasons. With such a small set of units, estimating the HAB's impact by comparing outcome evolutions before and after the HAB's onset in the affected season to that of pre-HAB seasons, could result in estimates heavily influenced by idiosyncratic shocks specific to each pre-HAB season.

A more effective approach in settings like this is to employ synthetic control analysis (SCA) (Abadie & Gardeazabal, 2003). This technique models the counterfactual evolution of the fishery's outcomes in the absence of the HAB event by combining data from multiple pre-HAB seasons. We term this modeled trajectory as the 'synthetic season.' The synthetic season is constructed by assigning weights to each pre-HAB season based on its similarity to the weekly trajectory observed in the HAB season prior to the event's onset. These weights are then used to estimate what the weekly outcomes during the HAB season might have looked like if the HAB had not occurred. The impact of the HAB is reflected by the differences in trajectories between the HAB and the synthetic season.

Our objective was to construct valid synthetic seasons for weekly total clam landings, the number of active boats in each fishing area, and regional clam prices during the HAB season following the start of closures. To build the synthetic season we considered 11 pre-HAB seasons, spanning from September 16^{th} of one year to September 15^{th} of the next, commencing in 2005 and concluding in 2015. The HAB season spans from September 16^{th} to September 15^{th}, 2016. In each season there are 53 weeks. As the first closure occurred in week 25 during the HAB season, we designated the initial 24 weeks as the weeks pre-HAB period, with the post-treatment period commencing from week 25 onward.

For a valid SCA, unaffected units must remain unimpacted by similar interventions or idiosyncratic shocks and share characteristics with the affected group (Abadie, 2021). Traditional SCA constructs the synthetic control using unaffected control units distinct from the affected group. Our approach is unconventional because we use data from equivalent weeks in past seasons to construct the 'synthetic season.' This method eliminates the threat of interference, as these past seasons occurred before the HAB event and thus were not influenced by it. Anticipation of the HAB event is also unlikely, considering the event's onset at the first closure was unexpected and predicting the timing and location of HABs is nearly impossible (Trainer et al., 2020; Ugarte et al., 2022). However, our method introduces the additional assumption that there are no other factors affecting the seasonality of the fishery in the HAB season other than the HAB event beyond those observed in past behavior. We provide a visual comparison of the trends of total landings, number of boats, and prices for the clam fishery in different seasons in **Figure C.11**.

Our research setting offered a substantial number of pre-event periods, which can contribute to bias reduction when estimating the synthetic control (Abadie, 2021). This feature also facilitated the assessment of the synthetic season's credibility. To do this, we backdated the week of the first closure by 5 and 10 weeks and visually examined the synthetic season's ability to track the outcome variable throughout the remaining pre-event period (**Figures C.12, C.13, C.14**). Additionally, we conducted robustness checks using a leave-one-out test to evaluate the sensitivity of the results to changes in the composition of the set of unaffected units (**Figures C.15, C.16, C.17**).

We employed the R package `scpi` to compute synthetic controls for the outcomes in the local clam fisheries during the HAB event (Cattaneo et al., 2022). We used the standard constrain of weights to be non-negative and sum up to one to avoid extrapolation (Abadie et al., 2010) (see **Tables C.4, C.5, C.6** for assigned weights). To account for uncertainty, we incorporated prediction intervals under random potential outcomes (Cattaneo et al., 2022). These prediction intervals considered sub-Gaussian out-of-sample prediction errors and in-sample uncertainty that was resampled 500 times. As all our variables represent non-negative quantities, we established a lower bound of zero for the prediction intervals. To quantify the impacts of the HAB event, we calculated the difference between the observed outcome and the synthetic control for a specific period. We considered these impacts statistically significant whenever the observed value fell outside the 90% prediction intervals in this analysis.

The urchin fishery initiated operations approximately seven weeks prior to its first closure.

This short time frame only allows for seven pre-treatment periods, and given the low variability within these periods, constructing a valid synthetic control is unlikely. Due to this limitation, we opted to use graphical representations and descriptive statistics to contrast the HAB season with pre-HAB season.

Results

Who continues fishing?

Assessing the impact of the HAB event on the number of active clam-reliant boats is challenging due to the irregular patterns of landings observed for individual boats within and across seasons. Nevertheless, we found suggestive evidence that some boats chose to exit the fishing sector for the remainder of the season in response to the HAB event. Specifically, when examining changes in the number of active clam-reliant boats between the pre-closure weeks and the post-closure weeks, we observed an increase in all pre-HAB seasons but a drop during the HAB season (**Table 4.1**). This drop was not observed among boats that relied on non-clam-reliant boats (**Table 4.2**).Out of the 654 boats that had reported clam landings in the South, Inner, or North fishing areas at least once during any of the pre-HAB seasons or in the pre-closure weeks of the HAB season, only 470 boats continued to report landings during the HAB event. In **Figure 4.4**, we illustrated the average differences in characteristics and fishing behavior between the clam-reliant boats that remained active during the HAB event and those that did not.

Clam-reliant boats that continued fishing during the HAB event were larger, and had greater storage and power capacity. They were often sheltered and more recently registered. Historically, these boats not only landed higher volumes but also had a more diverse catch, with higher proportions of urchin and *Luga*. Compared to inactive boats during the HAB event, they fished a wider variety of resources and operated across more areas. In the subsequent section, we analyze the average responses of these 470 boats, categorized into South-, Inner-, and North-clam-reliant boats, throughout the various stages of the HAB event.

Table 4.1: Change in the number of active clam-reliant boats between pre and post closure weeks.

Season	Boats active pre-closure	Boats active post-closure	Percent change
HAB	479	470	-1.9
Pre-HAB 1	318	391	18.7
Pre-HAB 2	334	419	20.3
Pre-HAB 3	419	458	8.5

Table 4.2: Change in the number of active non-clam-reliant boats between pre and post closure weeks.

Season	Boats active pre-closure	Boats active post-closure	Percent change
HAB	123	142	13.4
Pre-HAB 1	69	85	18.8
Pre-HAB 2	76	89	14.6
Pre-HAB 3	95	113	15.9

Figure 4.4: Standardized mean difference of boat characteristics and fishing behavior between boats that continued reporting landings during the HAB event and those that did not.

Average responses of clam-reliant boats that remained active during the HAB event

The estimated impacts of each stage and the Total HAB event on the landings of South-clam-reliant, Inner-clam-reliant, and North-clam-reliant boats are illustrated in **Figure 4.5** (for detailed estimates of changes in weekly landings, see **Tables C.7**, **C.8**, and **C.9**). We began by examining the responses of South-clam-reliant boats to the different HAB event stages (**Figure 4.5a**). In Stage I, the South fishing area remained mostly closed, resulting in reduced clam landings by South-clam-reliant boats in this area (weekly coefficient = -0.81, SE = 0.08, p <0.05). During this stage, these boats primarily shifted

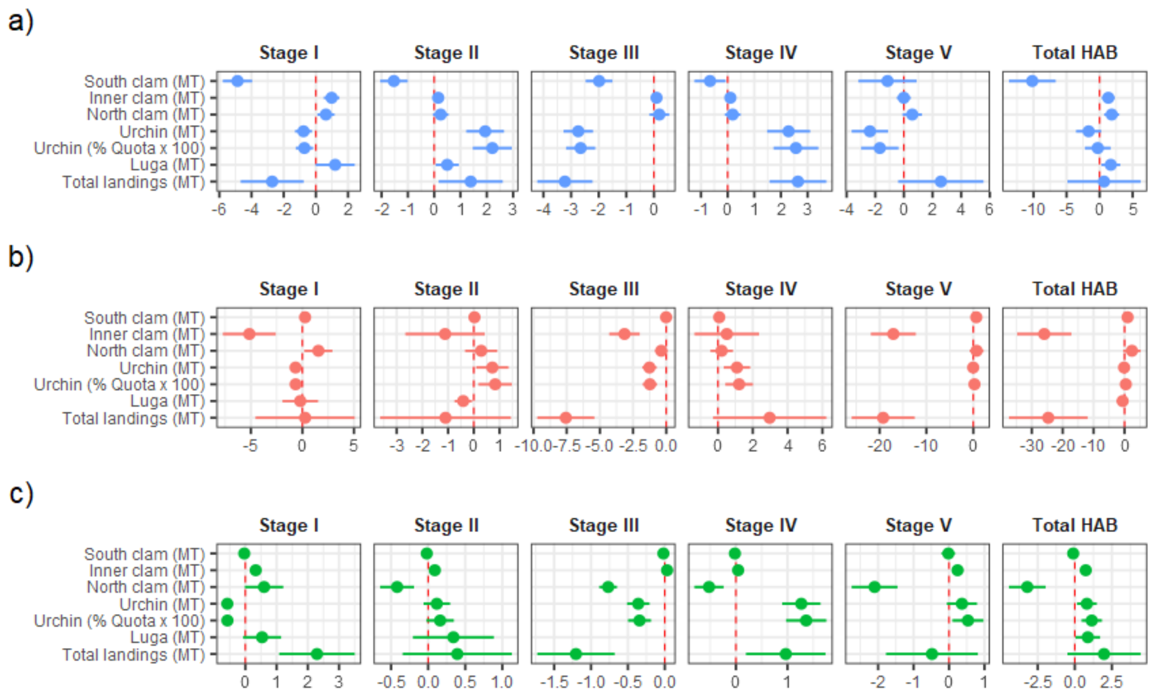

Figure 4.5: Difference in difference estimation of the effect of each stage and the Total HAB event on individual weekly landings of clam landed in the South, Inner, and North fishing areas, and weekly landings of luga and urchin (represented as MT and as percent of the regional quota) for (a) south-clam-reliant boats (in blue), (b) inner-clam-reliant boats (in red), and (c) north-clam-reliant boats (in green).

their clam landings to the North (weekly coefficient = 0.10, SE = 0.04, $p < 0.05$) and Inner areas (weekly coefficient = 0.16, SE = 0.04, $p < 0.05$), which remained largely open. South-clam reliant boats exhibited a marginally significant increase in *Luga* landings (weekly coefficient = 0.20, SE = 0.10, $p < 0.10$), while urchin landings decreased (weekly coefficient = -0.13, SE = 0.04, $p < 0.05$) in Stage I. In Stage II, despite a brief reopening of the South, these boats still exhibited reductions in their clam landings in the area (weekly coefficient = -0.51, SE = 0.09, $p < 0.05$), though less severe than in Stage I, suggesting partial recovery. Additionally, during Stage II, these boats significantly increased their *Luga* (weekly coefficient = 0.16, SE = 0.07, $p < 0.05$) and urchin landings (weekly coefficient = 0.65, SE = 0.12, $p < 0.05$).

For South-clam-reliant boats, clam landings in the Inner and North areas are typically minimal, which likely accounts for the lack of significant changes during Stage III, marked by the strike. However, the strike did result in decreased clam landings in the South (weekly coefficient = -0.66, SE = 0.08, $p < 0.05$) and a drop in urchin landings (weekly coefficient = -0.91, SE = 0.09, $p < 0.05$). In Stage IV, these boats increased their urchin landings (weekly coefficient = 0.76, SE = 0.14, $p < 0.05$), compensating for strike-induced losses. Additionally, they began to recover their South clam landings (weekly coefficient = -0.22, SE = 0.10, $p < 0.05$), suggesting diminishing HAB event impacts compared to earlier stages. By Stage V, the partial reopening of the South further spurred this recovery, rendering the impacts on South clam landings statistically insignificant. However, during this stage, urchin landings by South-clam-reliant boats were negatively impacted (weekly coefficient = -0.16, SE = 0.04, $p < 0.05$).

For South-clam-reliant boats, the HAB event significantly reduced clam landings in the South (weekly coefficient = -0.35, SE = 0.06, $p < 0.05$). In response, these boats reallocated their clam landings to the Inner (weekly coefficient = 0.05, SE = 0.02, $p < 0.05$) and Northern areas (weekly coefficient = 0.06, SE = 0.02, $p < 0.05$) and, when possible, augmented their *Luga* landings (weekly coefficient = 0.19, SE = 0.08, $p < 0.05$). While there were shifts toward urchin landings during Stages II and IV, these were offset in other stages, resulting in no net change in urchin landings or urchin quota use throughout the HAB event. Overall, the coping strategies of South-clam-reliant boats effectively counteracted the impacts of the HAB event, leading to no significant drop in their total landings during the defined HAB period.

Despite only a small portion of the Inner area being closed during Stage I, Inner-clam-reliant boats experienced a significant drop in clam landings there (weekly coefficient = -0.86, SE = 0.22, $p < 0.05$ and **Figure 4.5b**). This decline was partly offset by increased clam landings in the North (weekly coefficient = 0.26, SE = 0.11, p <0.05). Similar to South-clam-reliant boats, the Inner-clam-reliant boats also reduced their urchin landings in Stage I (weekly coefficient = -0.11, SE = 0.03, p <0.05). In Stage II, a brief reopening of the Inner area led to recovery of clam landings in this area with lower decreases than those observed in Stage I (weekly coefficient = -0.37, SE = 0.26, p <0.05). These boats also increased their urchin landings (weekly coefficient = 0.24, SE = 0.11, p <0.05) while decreasing *Luga* harvests (weekly coefficient = -0.14, SE = 0.06, p <0.05) in Stage II. In Stage IV, Inner-clam-reliant boats ramped up urchin landings (weekly coefficient = 0.36, SE = 0.13, p <0.05), partly compensating for strike-induced reductions in Stage III (weekly coefficient = -0.42, SE = 0.09, p <0.05). The Inner area began to reopen in Stage IV, with clam landings there by Inner-clam-reliant boats showing no significant impacts. Despite further reopening in Stage V, Inner-clam-reliant boats faced a significant decline in total landings (weekly coefficient = -1.37, SE = 0.24, p <0.05), largely driven by reduced clams landed in the Inner area (weekly coefficient = -1.22, SE = 0.17, p <0.05).

Overall, Inner-clam-reliant boats increased clam landings in the North during Inner area closures and increased urchin landings in Stages II and IV. Yet, these coping strategies were not effective in mitigating the impacts of the HAB event (weekly coefficient = -0.85, SE = 0.22, p <0.05). Their weekly landings diminished by 25 MT per boat, predominantly due to fewer clam landed in the Inner area (weekly coefficient = -0.90, SE = 0.15, p <0.05).

North-clam-reliant boats (**Figure 4.5c**) exhibited significant increases in their clam landings in the Inner (weekly coefficient = 0.06, SE = 0.01, p <0.05) and a marginally significant increase in the North areas (weekly coefficient = 0.10, SE = 0.05, p <0.10) in Stage I. Like the other boats, North-clam reliant boats also reported reduced urchin landings during Stage I (weekly coefficient = -0.10, SE = 0.01, p <0.05). The northern expansion of closures in Stage II, resulted in the significant decline of clam landings in the North (weekly coefficient = -0.14, SE = 0.04, $p < 0.05$). Nonetheless, these boats continued exhibiting increased clam landings in the Inner area (weekly coefficient = 0.03, SE = 0.01, $p < 0.05$), which remained accessible throughout Stage II. The strike in Stage III, caused notable drops in clams landed in the North (weekly coefficient = -0.26, SE = 0.02, $p < 0.05$), urchin (weekly coefficient = -0.12, SE = 0.03, $p < 0.05$), and total landings (weekly coefficient = -0.40, SE = 0.09, $p < 0.05$) for North-clam-reliant boats as did for the other boats samples. Despite continued reduced clam landings in the North in Stage IV (weekly coefficient = -0.17, SE = 0.05, $p < 0.05$), urchin landings of these boats more than compensated for earlier losses during the strike (weekly coefficient = 0.42, SE = 0.06, $p < 0.05$). Increases in weekly landings of urchins were still observed in Stage V, although landings were marginally significant and lower than those observed during Stage IV (weekly coefficient = 0.02, SE = 0.01, $p < 0.10$). During stage V North-clam-reliant boats also showed small increased clam landings in the Inner area (weekly coefficient = 0.02, SE = 0.01, $p < 0.05$).

Throughout the HAB event, North-clam-reliant boats exhibited increased clam landings in the Inner (weekly coefficient = 0.03, SE = 0.01, $p < 0.05$) area and enhanced urchin harvesting (weekly coefficient = 0.03, SE = 0.01, $p < 0.05$). They also recorded overall significant increases in *Luga* landings (weekly coefficient = 0.10, SE = 0.05, $p < 0.05$). Notably, North-clam-reliant boats reacted to closures happening in the South during Stage I. The overall responses of North-clam-reliant boats offset the negative impacts of the HAB event on clam landings in the North, with no significant impacts on total landings registered over the defined HAB period in terms of landed volumes for this sample of boats.

Overall, we observed spatial and resource mobility to cope with the closures, coupled with delayed increased effort, to mitigate losses from the strike. For both South- and North-clam-reliant boats, *Luga* served as a compensatory resource during its season, while urchin played a similar role specifically for North-clam-reliant boats. In Section 4.3, we explore whether market dynamics might have facilitated the spatial shifts of affected clam-reliant boats and accounted for harvesting intensification by initially unaffected clam-reliant boats in the early weeks of the HAB event. Section 4.4 examines the influence of clam-fishers' resource mobility on other boats targeting *Luga* and urchin, considering the potential role of management regulations.

Spatial mobility and mediation by market dynamics

Given our focus on how market dynamics mediate spatial mobility, and since this response was mainly observed during Stage I, our anlaysis of results centers on changes during this stage. The closures affecting most of the South during Stage I led to immediate and significant reductions in clam landings in this area (**Figure 4.6a**). By comparing these landings to a synthetic control based on pre-HAB seasons, we determined a decrease of 760 MT in South clam landings during Stage I. This decline was partly offset by a 378 MT increase in the North, particularly noticeable in the initial weeks (**Figure 4.6e**). Meanwhile, the Inner area saw a non-significant rise of 74 MT in clam landings (**Figure 4.6c**). There was a marked reduction in the number of boats landing clams in the South in Stage I, with weekly averages dropping from around 40 boats to just 6 (**Figure 4.6b**). In contrast, both the Inner and North areas saw an uptick in the mean weekly number of boats landing clam of about 10 boats (**Figure 4.6d,f**), although this was only statistically significant in the Inner area during the early weeks of the HAB event. This pattern corroborates our findings from section 4.2, highlighting the shift of South-clam-reliant boats to the North and Inner areas in Stage I.

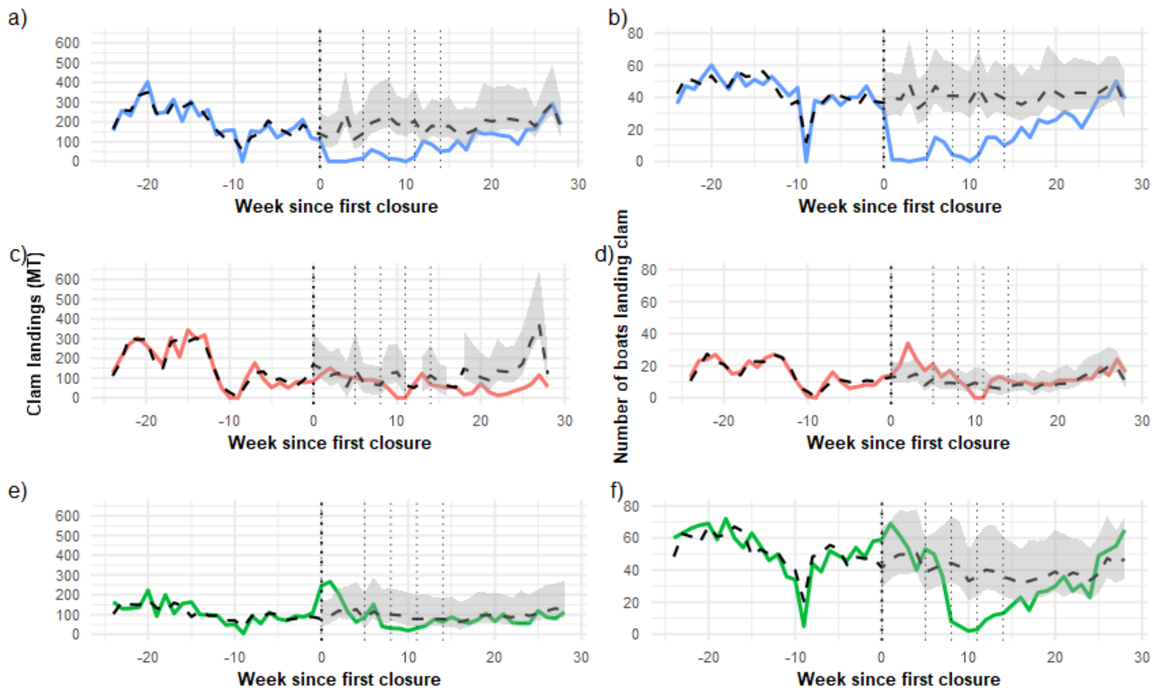

Figure 4.6: Observed weekly trajectories of clam landings (in MT) and the number of boats landing clams in the South (a,b), Inner (c,d), and North (e,f) fishing areas compared to the synthetic control trajectories predicted using pre-HAB seasons since 2005 (dashed lines). The shaded area around the synthetic control indicates 90 percent prediction intervals.

Shifting to different fishing areas can increase travel costs, requiring compensatory revenue to remain profitable. During Stage I, there was a marked rise in the regional clam price, becoming significant by the third week after the first closure was issued (**Figure 4.7**). The weekly clam price was, on average, \$112.6 USD per ton above the synthetic control's prediction during Stage I. This price surge likely drove the mobility of South-clam-reliant boats towards fishing areas that remained opened and spurred North-clam-reliant boats to increase their landings in Stage I. Our results suggest a ripple effect: reduced clam supply by impacted boats caused price increases, promoting mobility, and intensifying harvesting by boats whose fishing grounds had not yet been affected by the closures.

Our SCA also identified a significant decline in prices following the peak in Stage I (**Figure 4.7**). This decrease likely stems from consumers becoming increasingly aware of the HAB's expansion, leading to a subsequent reduction in demand. The recorded zero values during the strike weeks arise because price data are derived from transactions, and thus no data were recorded in those weeks when commercial activities were halted.

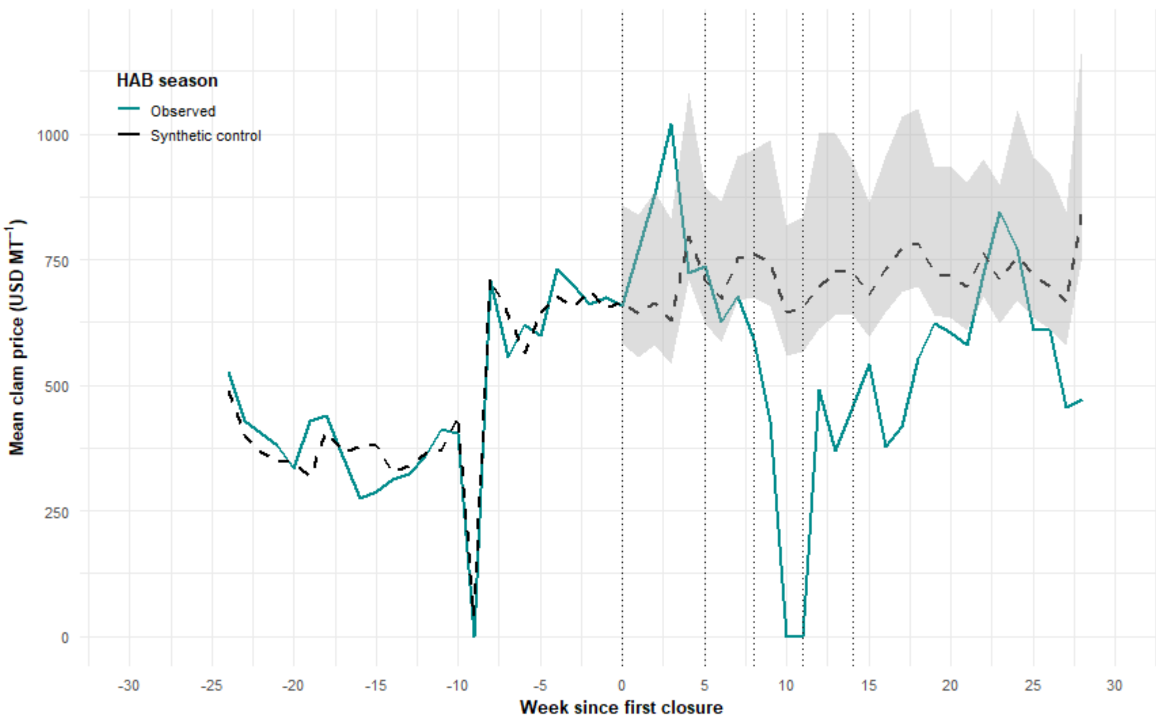

Figure 4.7: Observed weekly trajectory of mean clam price (USD/MT) across the analyzed fishing areas compared to the synthetic control trajectories predicted using pre-HAB seasons since 2005 (dashed lines). The shaded area around the synthetic control indicates 90 percent prediction intervals.

Cascading effects on urchin harvesting and mediation by a regional quota

To explore the potential ripple effects from clam-reliant boats' responses on other boats, we examined landing shifts in boats that historically relied on *Luga* and urchin, but not on clam or other HAB-affected resources. During the early stages of the HAB event, these non-clam-reliant boats significantly increased their urchin landings (**Figure 4.8**). However, considering the event as a whole, we observed no impact on total urchin landings for this group, as changes in different stages effectively neutralized each other (for detailed estimates of changes in weekly landings see **Table C.10**).

In particular, non-clam-reliant boats started the urchin season with a notable rise in urchin landings, averaging an increase of 1.4 MT (SE = 0.10) per boat in Stage I. This trend persisted in Stage II with continued growth in urchin landings. This trend persisted into Stage II. While these boats reduced their landings during the strike in Stage III, they rebounded in Stage IV, registering an average increase in urchin landings of 3.4 MT (SE = 0.12) per boat, which surpassed the 2.3 MT (SE = 0.12) of landings lost to the strike. The proactive increase in urchin landings can be attributed to the existence of a regional quota, which likely prompted non-clam fishers to act preemptively and prevent significant quota losses to clam fishers shifting to urchin. However, by Stage V, non-clam-reliant boats decreased their urchin landings by approximately -4.2 MT (SE = 0.05), likely due to hitting the quota cap in the 25^{th} week following the initial closure.

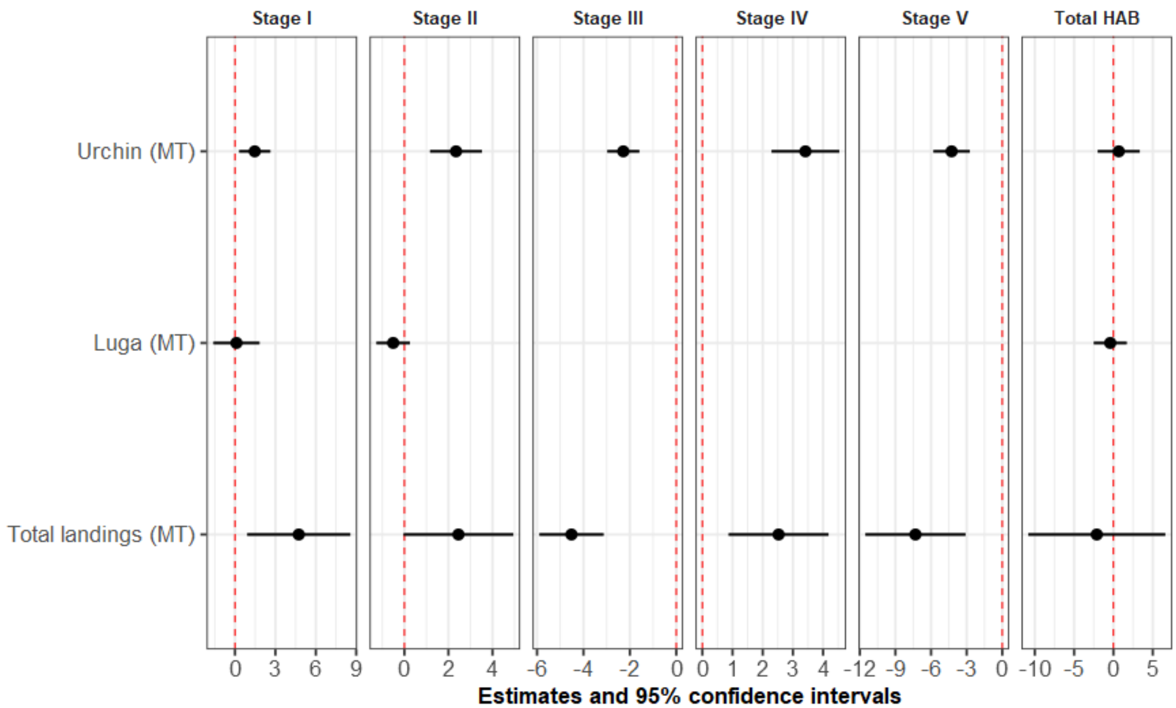

Figure 4.8: Difference in difference estimation of the effect of each stage and the Total HAB event on individual weekly landings of luga and urchin (represented as MT and as percent of the regional quota) for non-clam-reliant boats.

The overall increased harvesting of urchin during the HAB season shortened it by 2.2 weeks relative to the pre-HAB seasons' average. The cumulative urchin quota consumption presented unusual dynamics during the HAB season (**Figure 4.9a**). Nonetheless, the total use of the quota fell within values observed in pre-HAB seasons for each group, suggesting no major distributional changes across these two groups relative to pre-HAB seasons. During the HAB season, participation in the urchin fishery was higher than in pre-HAB seasons, especially in late stages for clam-reliant boats (**Figure 4.9b**), and in early stages for non-clam-reliant boats (**Figure 4.9b**). We did not observe variation in urchin prices that could explain changes in the urchin landing behavior in the HAB season relative to pre-HAB seasons (**Figure C.18**).

In contrast to urchin landings, *Luga* landings, which were not managed under a quota system, exhibited no significant changes in landing patterns throughout the HAB season for non-clam-reliant boats (**Table C.10**).

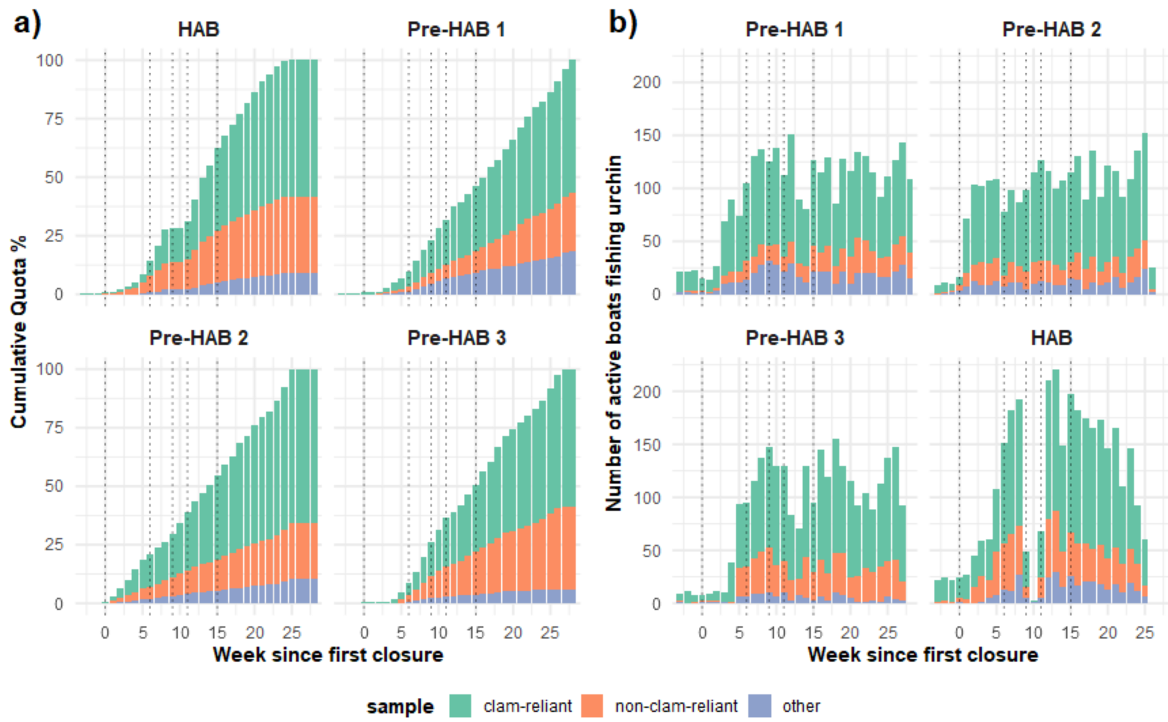

Figure 4.9: Weekly trends in urchin fisheries of (a) cumulative percent of the quota consumed by each sample and (b) number of boats fishing urchin belonging to each sample for each of the three pre-HAB and HAB seasons.

Discussion

Understanding coping responses to extreme weather events within socio-ecological systems is critical for advancing development and food security in an era of escalating environmental change. Drawing from a case study on small-scale fisheries in the Global

South, we documented how fishers reallocated their effort across space, resources, and time to counter the livelihood shocks caused by a massive HAB event. By incorporating insights from market and management conditions, we could elucidate not only the direct coping responses of those affected by the shock but also reactions from actors that were indirectly affected. Our results underscore the pivotal role of market dynamics and management practices in facilitating or hindering coping responses and how their effects can ripple across stakeholders and ecosystems.

We showed that boats affected by the closure of their fishing grounds exhibited mobility to areas that remained open. This shift was likely facilitated by a surge in clam prices, possibly attributable to regional clam shortages. This price surge also appears to have motivated clam-reliant boats whose fishing grounds were not closed to boost their clam landings. Although we observed a clear shift in effort towards *Luga* and urchin, these changes were constrained by management rules: a seasonal closure for *Luga* and a regional quota for urchin. Our data suggest that non-clam-reliant boats proactively increased their urchin landings earlier in the harvest season, aiming to prevent significant quota losses to clam-reliant boats that adopted urchin as a coping resource. Furthermore, we found that boats in our sample increased landings either before or after the strike to counteract the associated losses.

Spatial mobility and changes in resource targeting are frequently viewed as coping strategies for harvesters when their primary harvesting areas and resources become restricted. Prior case studies have identified spatial and resource mobility as coping mechanisms in response to actual weather-induced livelihood shocks, particularly in fisheries in California and Alaska (Cline et al., 2017; Fisher et al., 2021; Jardine et al., 2020; Liu et al., 2023). Our study brings new empirical evidence from observational data in the context of the Global South which complements existing results from approaches based on self-reporting in this part of the world (Cinner et al., 2009). Furthermore, we evidenced an often-neglected coping strategy: the reallocation of effort across time. Reallocation of effort over time was observed as both a direct effect of the strike and closures, and as an indirect effect in response to shifts in effort over time by other fishers. Such responses have been documented in cases where access is restricted due to management constraints (McDermott et al., 2019). Recognizing the reallocation of effort over time as a coping strategy is vital, as its timing can impact crucial productivity cycles of the harvested resource.

Consistent with prior studies, and supporting the notion that assets and flexibility enhance adaptive capacity, we found that boats active during the HAB event were generally larger, had greater power and storage capacity, and had a more diverse catch history (Cinner, Adger, et al., 2018; Jardine et al., 2020; Liu et al., 2023). This compared to boats that recorded landings in the analyzed pre-HAB period but ceased operations during the HAB event. It is important to note that our dataset inherently leans towards larger boats. This bias stems from the fact that smaller fishing operations in Chile are more likely to go unreported (Donlan et al., 2020). Conversely, boats measuring 12m or longer are legally mandated to land their catch under the supervision of enforcement authorities. This, combined with the patchy reporting of landings over time by many boats, limited our ability to accurately estimate the number of boats that exited the fisheries due to the HAB event. Furthermore, the Los Lagos region houses a large community of shore gleaners, who, despite being affected by the HAB event, were not factored into our study. Nonetheless, anecdotal evidence and the occurrence of the strike suggest that a significant segment of fishers opted to halt their activities (Mascareño et al., 2018; Ugarte et al., 2022).

Although we characterized the strike as a shock in our analysis, it can also be viewed as a coping mechanism. Fishers who could not adapt by switching areas or resources saw the strike as a means to access financial aid. In support of impacted fishers, several productive sectors joined the mobilization, widely disrupting the economy of the region (Mascareño et al., 2018; Ugarte et al., 2022). This underscores the importance of timely governmental aid to mitigate the cascading effects of extreme-weather-induced livelihood shocks that may include disruptions across various supply sectors. Obstacles for fishers to undertake coping responses within the fishing sector might encompass regulatory challenges, such as the lack of specific resource permits, limited market avenues for selling catch, or the lack of particular gear or expertise. Governmental support in this regard, can be useful but needs to be tailored to the needs and capacities of specific communities.

While our study primarily concentrated on ex-post coping responses within the intensive margin — where fishers intensified labor in their sector — coping can also manifest in the extensive margin, outside the fishing activity. This could explain the behavior of Inner-clam-reliant boats, which did not fully recover their clam or total landings during our study period. Small-scale harvesters frequently maintain multiple livelihoods (Roscher, Eriksson, et al., 2022). In the Los Lagos region, many fishers cultivate land and own livestock. These alternative livelihoods likely provided some relief during the adverse effects of the HAB event in 2016. Additionally, many fishers in the region work as divers for aquaculture centers (Outeiro & Villasante, 2013), which are also vulnerable to the impacts of HAB events (Mardones et al., 2021; Trainer et al., 2020). Such interconnected vulnerabilities highlight the importance of considering multi-sectoral risks associated with extreme weather events, especially when addressing the resilience of socio-ecological systems (Thiault et al., 2019).

Extreme weather events can affect markets by influencing prices through supply and demand. We observed unusual price dynamics for clam in the Los Lagos region during the HAB season. Given the timing of these fluctuations, it is likely that the spike in price at the onset of the HAB event resulted from a clam shortage, which occurred as closures were primarily concentrated in the South. The subsequent price drop was likely caused by a decrease in demand as the HAB event expanded throughout the region. This drop in demand is commonly observed in HAB events as buyers and consumers become aware of the potential risks of consuming toxic products (Mao & Jardine, 2020; Trainer, 2020). The duration of low demand during a HAB event is influenced by factors such as consumer misinformation, trust between sellers and buyers, and transparency of monitoring programs (Mao & Jardine, 2020). Thus, HAB impacts on livelihoods can vary based on the types of commercialization channels established by different harvesters.

The early increase in clam prices might have played a pivotal role in facilitating mobility as a coping strategy during the early stages of the HAB event. By making longer trips profitable, boats facing closed fishing grounds had a financial incentive to relocate. These findings emphasize the instrumental role of markets in determining coping opportunities. The initial spike in price also seems to have driven increased harvesting efforts by clam-fishers outside the affected areas, showcasing how market dynamics connect responses across spatial scales. In this region, small-scale fishers often select their target resources based on profitability. As such, the sudden surge in clam prices likely explains why clam-reliant boats did not immediately switch to harvesting urchin as the season began and stayed focused on targeting clams.

While previous studies have established the influence of management rules on available coping opportunities, our research goes further by examining cascading responses due to the interaction between management rules and extreme-weather- induced livelihood shocks. We found suggestive evidence that the presence of a quota system triggered anticipatory increases in effort by non-clam-fishers who remained unaffected by the closures at the beginning of the HAB event. These findings resonate with concerns raised by fishers during field conversations and reported in local newspapers (Alarcón, 2016). Anticipating these indirect responses is important to avoid conflicts among users and cascading degradation across biological populations and ecosystems.

Our results highlight an interesting challenge for resources management posed by extreme-weather-events-induced livelihood shocks. While resource use restrictions such as quotas, seasons, and permits are helpful in sustaining healthy populations, they can reduce the flexibility to cope and adapt in the face of environmental change (Anderson et al., 2017; Holland et al., 2017; Kasperski & Holland, 2013; Schaap et al., 2021). We found that flexibility in terms of allowing for reallocation of effort across space, resources, and time were useful to mitigate the impacts of extreme-weather-induced livelihood shocks. On the other hand, is likely that the quota in the urchin fishery prevented overexpoitation motivated by economic need which could have generated another lagged indirect effect by reducing the sustainability of the urchin fishery. In the literature addressing livelihood shocks induced by environmental change, a significant methodological challenge is distinguishing the impacts of the shock from other sources of variability. Effective assessments of the impacts of and responses to environmental shocks should employ counterfactuals, contrasting observed outcomes with those that would have occurred in the absence of the shock (Trainer, 2020). Constructing reliable counterfactuals generally involves data from affected and unaffected units, both pre and post-shock. This concept parallels the Before-After-Control-Impact (BACI) designs frequently used in ecology. The HAB event's weekly dynamics enabled us to utilize previous fishing seasons as our control units, since other boats or areas could not act as reliable controls. However, when data on control units that differ in dimensions other than time are available, they should be consider for more accurate impact or response estimations. In situations where such units are absent, as in our study, methods like ours offer an effective way to account for inherent seasonality without the need for complex modeling assumptions

Selecting appropriate spatial and temporal scales is pivotal when analyzing the impacts of and responses to extreme-weather-induced livelihood shocks. In our study, examining impacts at broader annual and regional scales could have masked many of the observed nuanced responses and impacts. Failing to capture livelihood disruptions at finer weekly or monthly intervals might overlook significant implications for household economies. Furthermore, by not differentiating spatially, we risk misrepresenting impact distributions, as some regions might compensate for downturns in others. For example, during the early stages of the HAB event, increased landings from north-clam-reliant boats balanced declines from their southern and inner-clam counterparts. Therefore, we should carefully consider relevant scales of analysis to effectively target aid and gauge vulnerability across different harvesting groups.

Our analysis is constrained by data limitations. Absent specific data on individual fishing grounds, boats were broadly categorized based on landing patterns, which might misrepresent their primary fishing areas. Additionally, our spatial data limitations required us to represent the impact of closures in different fishing areas as different binary stages of the HAB event. Using only the percentage of an area closed as our metric could have been misleading, since a small closed area might have represented the most vital fishing grounds for certain boats. This may explain why Inner-clam-reliant boats experienced significant landing declines in Stage I, even with most of their fishing area remained open. Similarly, in stage V, despite a considerable reopening of all fishing areas, their landings did not rebound. Such nuances highlight the potential for underestimation of responses in our analysis. However, given that spatial tracking data are often limited to larger vessels in developed contexts, our study offers a valuable contribution to avoid biasing the literature on coping responses in extreme-weather-induced livelihood shocks based on data and technology access.

The challenge of anticipating the impacts of climate change and extreme weather on socio-ecological systems lies in the uncertainty of human behavior. Gaining empirical insights into how harvesters cope with shocks induced by extreme weather can significantly reduce this uncertainty. Our study highlights the crucial role of analyzing coping responses within the interconnectedness of socio-ecological systems. By recognizing how market dynamics and resource management regulations influence these responses, we can better understand the strategies available to users and their potential cascading effects on stakeholders and ecosystems. Such knowledge is essential for enhancing the resilience of communities dependent on natural resources and for preventing maladaptive outcomes and conflicts.

Appendix A

Appendix for Chapter 2

Table A.1: Summary of previous studies testing the external validity of game experiments in common-pool resources.

Reference	Subjects	Field_outcome	Game	Frame	Enforced	Externally_valid
Basurto et al., 2016	Fishers	Opinions on social norms, marine protected areas, local authorities and social organizations	Public Good	Abstract	No	Yes
Basurto et al., 2016	Fishers	Opinions on social norms, marine protected areas, local authorities and social organizations	Modified joy-and-destruction	Abstract	No	Yes
Bluffstone et al., 2020	Loggers	Per hectare carbon, trees and seedlings to capture forest quality	Public Good	Abstract	No	No
Carpenter & Seki, 2011	Fishers	Fishing productivity as catch per trip	Public Good	Abstract	Yes	Yes
Fehr & Leibbrandt, 2011	Fishers	Hole sizes of shrimp traps with larger holes considered more sustainable as they allow juveniles to scape	Public Good	Abstract	No	Yes
Gelcich et al., 2013	Fishers	Resource management performance under territorial rights considering self-reported enforcement and compliance, third-party assessments, and ecological measurements	Common-pool resource	Contextual	No	Yes
Gurven & Winking, 2008	Forager-Farmers	Participation in beer provisioning and consumption, and time spent in social visitation	Third-party punishment	Abstract	Yes	No
Hopfensitz & Miguel-Florensa, 2017	Farmers	Self-reported side-selling coffee in free market	Public Good	Abstract	No	Mixed evidence
Lamba & Mace, 2011	Foragers-Farmers	Amount of salt taken from a common pool	Public Good	Abstract	No	Mixed evidence
Rustagi et al., 2010	Loggers	Potential crop trees per hectare	Public Good	Abstract	No	Yes
Torres-Guevara & Schlüter, 2016	Fishers	Fishing impact based on fishing spots and gears	Public Good	Abstract	No	No

* This summary is limited to studies that investigated the external validity of game experiments using outcomes in the field that captured common-pool resource problems.

Table A.2: Summary of the design features and findings of previous studies testing the external validity of game experiments in common-pool resources settings.

Dimension	**Variable**	**Survey__question__or__source**	**Scale**
Social Integration	Pride in TURF	In our community we are proud of our TURF	4 = Highly agree to 1 = Highly disagree
	Self-assessed compliance	Our community members observe the management agreement to the letter	4 = Highly agree to 1 = Highly disagree
Management capacity	Trend in annual allowed catch	SUBPESCA statistics	1 = Increased, -1 = Decreased, 0 = No significant trend
	Assessed as successful community by government officials	TURFs considered a successful model in its region	1 = Yes, 0 = No
	Current performance of TURF assessed by government officials	Which of these words better describe the overall performance of the TURFs	5 = Success, 4 = Stability, 3 = Improvement, 2 = Stagnation, 1 = Failure
	Future projection of TURF assessed by government officials	Assign a score to each community reflecting its future projections with respect to their TURF performance?	5 = Good performance to 1 = Bad performance

* Based on the social integration and management capacity dimensions of the co-management performance index developed by Marín et al. (2012).

† We estimated a cooperation index as the average score of the included variables using data provided by Marín et al. (2012) and Gelcich et al. (2013). To define the cutoff at which a community would be considered as displaying high (HC) or low cooperation (LC), we calculated the median value in the whole sample by Marín et al. (2012), which is representative of the region of interest. All communities with indices above the median were considered HC and those below the median were considered LC.

The Common Pool Resource Game

At the beginning of each round $t \in \{1, \ldots, 20\}$, each player was endowed with 100 units of the resource, which were assumed to be used completely. This endowment represented the individual quota that fishers agree upon when extracting resources from their TURFs. Then, simultaneously, each player $i \in \{1, \ldots, 5\}$ had to privately decide the $x_{i,t} \in \{0, \ldots, 50\}$ number of units to extract above their individual endowment (i.e., overextraction). A negative externality was associated with this overextraction to mimic the cost imposed on other users in the field. For each unit that a subject decided to overextract, each other member of their group $j \in \{1, \ldots, 5\} \neq i$ lost half a unit. Thus, $x_{i,j}$ can be translated into a compliance percent with $x_{i,j} = 50$ and $x_{i,j} = 0$ representing 0% and 100% compliance, respectively. The unitary price of a unit was 10 CLP. The individual payoff $\pi_{i,t}$ per round was given by **Equation** (A.1).

$$\pi_{i,t} = (100 + x_{i,t} - \frac{1}{2} \sum_{j \in S_{-i}} x_{j,t}) \times 10 \tag{A.1}$$

In round 11, a peer-enforcement mechanism was introduced and remained operative until the end of the game. This change divided the game into two stages. The first ten rounds constituted the unenforced stage. The last ten rounds were part of the enforced stage. In the field, fishers may observe noncompliance and decide whether to report it to the community board. The board often sanctions offenders by seizing their catch or implementing other material costs. This situation was recreated in the enforced stage, increasing the ecological validity of the game relative to the unenforced stage.

In the game, the peer-enforcement mechanism operated as follows. At the end of each round in the enforced stage, two participants per group were randomly selected as inspectors. After each fisher had decided on the number of units to extract, the inspectors saw the harvest of another randomly selected group member. If the inspected fisher had over extracted, the inspector could punish the offender by paying a fixed cost. Fishers who were punished lost all their harvest for that round. To recreate the payment a fisher would earn for patrolling their TURFs in the field, we added 250 CLP to a participant's account each time they were appointed as an inspector. Inspectors had to pay 250 CLP to report in the game. This is to recreate the fact that reporting a peer in the field usually involves material and nonmaterial costs.

The game is a linear CPR with complete information that is repeated a predetermined number of times. Therefore, the appropriate concept of equilibrium for the game is subgame perfection. To determine the subgame perfect equilibrium, we use a backward induction argument.

The final round of the game is, in practice, a one-shot CPR game. Hence, the dominant strategy for each player is to overextract the maximum amount of the resource, which is 50 units. This is because each unit yields a profit of 10 CLP for the player, at no cost. However, if players anticipate that free riders will be punished (and as a result, lose all their gain), they may choose not to overextract. Since punishing a free rider has a cost of 250 CLP and punishment has no effect on their future behavior (because there is no next round after the final one), inspectors will refrain from punishing. Players will anticipate this and will overextract 50 units.

The outcome of the second-to-last round is identical: no one will pay attention to what happens in the final round, because (as previously shown) the behavior of players in the final round is fixed – all will overextract 50 units and the inspectors will not punish. By backward induction, we conclude that the players' behavior will be the same in all rounds.

In each round of the first stage, each player obtains a payoff of 50 CLP (this can be verified by replacing $x_{it} = x_{jt} = 50$ in **Equation** (A.1)). This is less than the payoff they would obtain if they all abstained from overextracting, in which case their payoff would be 100 CLP (replace $x_{it} = x_{jt} = 0$ in **Equation** (A.1)).

In the second stage, the expected payoff for each player is 150 CLP. This payoff includes the gains of the game (which are 50 CLP, because they free ride) and the expected value of the windfall of 250 CLP given to each inspector, which they retain because they do not punish free riders (the expected value is given by $\frac{2}{5} \times 250$ CLP $= 100$, because two of the five players are made inspectors). This payoff is also less than the one they would obtain if they cooperated, in which case the payoff would be 200 CLP. In this case, the inspectors would also retain their windfall, since there are no free riders to punish.

An important consideration is the minimum probability of punishment necessary to induce self-interested subjects to comply with their quotas. In the scenario where a free rider is not punished, his gain would be equal to his quota (100 units) plus the number of units he has over-extracted, minus any negative externality caused by other subjects. If he is punished, his profit would be zero. This means that the expected payoff for a free rider is highest when he over-extracts the maximum possible number of units (50 units) and all other subjects comply with the quota. It's worth noting that this calculation is independent of the probability that the free rider will be punished.

Suppose that a free rider who is inspected has a probability p of being punished. Given that the probability of being inspected is 0.4, the unconditional probability that a free rider will not be punished is $1 - 0.4p$. As a result, his expected payoff is:

$$E(\pi_{it}) = (1 - 0.4p) \times (100 + 50 - 0) \times 10 = 1500 - 600p.$$

On the other hand, a cooperator's expected payoff is:

$$E(\pi_{it}) = (100 + 0 - 0) \times 10 = 1000.$$

Therefore, a subject who is solely interested in maximizing his expected payoff will cooperate if and only if $1500 - 600p < 1000$. In other words, punishment will only be a deterrent if $p > 0.83$. Note that this analysis assumes that inspectors do not play their optimal strategy, which is to never punish.

Suppose that the probability of punishment is insufficient for deterrence (i.e., $p < 0.83$). In that case, a subject who cooperates does so for reasons other than his own material interest. His behavior indicates that he has some intrinsic motivation to cooperate, such as internalizing a social norm.

Game Instructions (implemented in spanish)

The words in brackets differed between frames. Those to the left of the "/" were used under the contextualized frame and those to the right, in the abstract frame.

"Welcome and thank you for being here. This research is part of a project carried on jointly by Pontificia Universidad Católica de Chile and the Research Center in Social Complexity from the Universidad Del Desarrollo. Your association's board and national fisheries authorities, such as the National Service of Fisheries (SERNAPESCA) and the Undersecretary of Fisheries and Aquaculture (SUBPESCA), are not involved in this study. The game will last around an hour. By participating you could earn up to $32,500 CLP each. Once the session ends you will receive your payoffs individually and privately.

Now we will read the instructions aloud. If you have any questions, please rise your hand in silence and we will answer them aloud. Let us start. You will play the game via computer. Do not worry if you have never used a computer before because we will only be using the numeric keypad, which works very similar to the numeric keypad in a cellphone or a calculator. Before starting the game, the ten participants in the room will be randomly assembled into two groups of five fishers each. All the interactions with your partners will be anonymous via the computer. You will never know who the other members of your group were, neither during the game nor after it.

The game recreates a situation in which you are [harvesting loco/extracting coins from a common pool] and have to decide individually how many [locos/coins] to [harvest/extract]. The game is divided into 20 rounds, which represent [fishing trips/visits to the pool of coins]. Each one of you has an individual [quota/endowment] of 100 [locos/coins] per round. The value of each [loco/coin] is $10 CLP. The computer will always assume that you will [harvest/extract] all the [locos/coins] from your individual [quota/endowment]. In addition, you will have the chance to [harvest/extract] up to 50 more [loco/coins] above your [quota/endowment] in each round.

[Harvesting/extracting] more [locos/coins] than the [quota/endowment] brings more economic benefits to you, but it produces economic harm to the other members in your group. This is because for every two [locos/coins] you [overharvest/overextract] the other members in your group will lose one [loco/coin] each from their individual [catch/account]. [This mimics the damage that overharvesting generates over the marine ecosystem reducing everyone's' productivity] (only under the contextual frame).

Now, we will show you the screens that you will see in the computer during the game. In each round the computer will ask the same question: How many [locos/coins] above your individual [quota/endowment] you want to [harvest/extract] (from 0 up to 50)? If you want to [comply with your quota/take no more coins above your endowment], the answer must be zero. Your answers will be recorded anonymously in the computer. Neither the researchers, nor the other participants will know how many [locos/coins] you [harvested/extracted] during the game. Your identity will never be revealed. To remain anonymous, it is very important to be quiet during the game and not make any comments. It is not allowed to speak during the game, if you need any help, please raise your hand in silence and a facilitator will assist you. Once every group's member has entered their responses, you will see a summary screen, summarizing the results of the round. As you can see, it will tell you:

- The number [locos/coins] that you [harvested/extracted] in that round.
- The number of [locos/coins] [harvested/extracted], on average, by the rest of the members in your group in that round.
- The number of [locos/coins] that you lost due to the [overharvesting/overextraction] of the other members of your group in that round.
- The number of [locos/coins] that you ended up with.
- The amount of money in CLP that you earned in that round.

Once you have read the screen press the red button to continue. Once everyone has read their screens, the next round will automatically begin. Summarizing:

- You have an individual quota of 100 [locos/coins] in each round.
- You have the chance of [harvest/extract] beyond your [quota/endowment] up to 50 additional [loco/coins] in each round.
- For each two additional [loco/coin] you [overharvest/overextract] the rest of the members in your group will lose a [loco/coin] from their [catch/account] in that round.
- We will pay you $10 CLP for each [loco/coin] at the end of each round.

- Your earnings will be accumulated during the 20 rounds of the game and will be paid privately at the end of the session.

Before starting you will play three trial rounds just to practice. These rounds are not for real money. Please rise your hand in silence if you have any question and wait until a facilitator can assist you. Once the trial rounds are completed, the real game will start, and you will be playing for real money.

Please remember that communication during the game is not allowed!

Starting from round 11, a new rule will be implemented. After everyone has decided how many [locos/coins] they want to [overharvest/overextract], the computer will randomly match two players in each group. One person in each of the couples formed by the computer, will be assigned as the inspector and will be allowed to observe their partner's [catch/account] for that round without knowing their identity. Since the groups have five players, two persons in each group will be inspectors, two will be inspected, and one person will remain inactive. The computer will randomly assign the roles in each round.

If you are randomly chosen as an inspector, you will see a screen that will show you your partner's decision. It is like you could see how many [locos/coins] the other [fisher/player] have in their [boat/account] in that round. If your partner has exceeded their [quota/endowment], you will have the chance of [reporting them to the community's board this is just a recreation given that the community's board is not really involved in this study/making them lose all what they have accumulated in their private account in that round]. If you are selected as an inspector, $250 CLP will be added to your account. You can use them to [report your partner/ make your partner lose what they have accumulated in their private account for that round] if they have exceeded their [quota/endowment]. If you do not spend the $250 CLP, they will be accumulated in your account. [If the community's board is informed about a quota violation, it will punish the offender by seizing all their catch for that round]. In the game, the computer will [play the role of the community's board/implement the seize of coins]. Since the game is anonymous, no one will really know who exceeded their individual [quota/endowment].

If you are being inspected in a given round, the screen will let you know that your [catch/account] is being inspected by other player, and you should wait in silence for the next screen. If you have been selected to remain inactive during this stage, it means you do not inspect anyone's [catch/account] nor will someone inspect yours. The screen will ask you to wait in silence for the next screen.

After inspectors have decided whether to [report/make other player lose their coins], everyone will see a summary screen. As you can see, it will tell you:

- The number of [locos/coins] that you [harvested/extracted] in that round.
- The number of [locos/coins] [harvested/extracted], on average, by the rest of the members in your group in that round.
- The number of [locos/coins] that you lost because of the [overharvesting/overextraction] of the other members of your group in that round.
- Whether you have [been reported to the community's board and your catch has been seized in that round/lost the coins in your private account because an inspector decided it].
- The number of [locos/coins] that you ended up with in that round.
- The amount of money in CLP that you earned in that round.

Once everyone had read their results, the next round will start. Summarizing:

- You have an individual [quota/endowments] of 100 [locos/coins] in each round.

- You have the chance of [harvest/extract] beyond your [quota/endowment] up to 50 additional [locos/coins] in each round.
- For every two additional [locos/coins] you [overharvest/overextract], the rest of the members in your group will lose a [loco/coin] from their [catch/account] in that round.
- After everyone has entered their decisions, the computer will randomly assign two players as inspectors, two as inspected, and one will remain inactive in each round. Each inspector will see the [catch/account] of an inspected [fisher/player].
- If the inspector sees that the inspected fisher has exceed their individual [quota/endowment], they can decide whether to [report them to the community's board/ make them lose what they have accumulated in their private account for that round]. The inspector will receive $250 CLP in their account that can be spent in [reporting a quota violation/making an inspected fisher lose their extraction for a given round if they have extracted coins above their individual endowment]. If the inspector does not use the $250 CLP for this, they will be accumulated in her or his account.

- The [computer will play the role of the community's board and/computer] will seize all the [catch/coins] of a reported offender, including their individual [quota/endowment].
- We will pay you $10 CLP for each unit of [loco/coin] at the end of each round.
- Your earnings will be accumulated during the 20 rounds of the game and will be paid privately at the end of the session.

Before starting you will play three trial rounds just to practice. These rounds are not for real money. Please rise your hand in silence if you have any question and wait until a monitor can assist you. Once the trial rounds are completed, the real game will start, and you will be playing for real money. Please remember that communication during the game is not allowed!

Rationale for the outcome variable

Compliance requires sacrificing individual gains to prevent harmful externalities, thus, measuring cooperation. We used the mean group compliance percent in each round as the dependent variable. We averaged compliance percent across subjects in the same group for two reasons. First, the categorization into HC and LC is done at the community rather than at the individual level. The game groups are the smallest unit at which we expect norms and expectations from the fishing community to affect the outcome. Thus, we treated player groups as replicates of each community type. Secondly, averaging decisions in the group provides independent observations since individual decisions of subjects playing in the same group are likely correlated. In one of the experimental sessions in an LC under the contextual frame, groups were randomly reallocated in each round. Because subjects were unaware of the reallocation, behaviors should not differ from those expected in fixed groups. Nonetheless, subjects in these sessions potentially interacted with all the other nine subjects in the session. Therefore, we computed the mean compliance percent across all 10 subjects in this session to obtain independent observations. We added weights to the OLS regression based on the number of players aggregated in each observation to account for the imbalance.

Appendix B

Appendix for Chapter 3

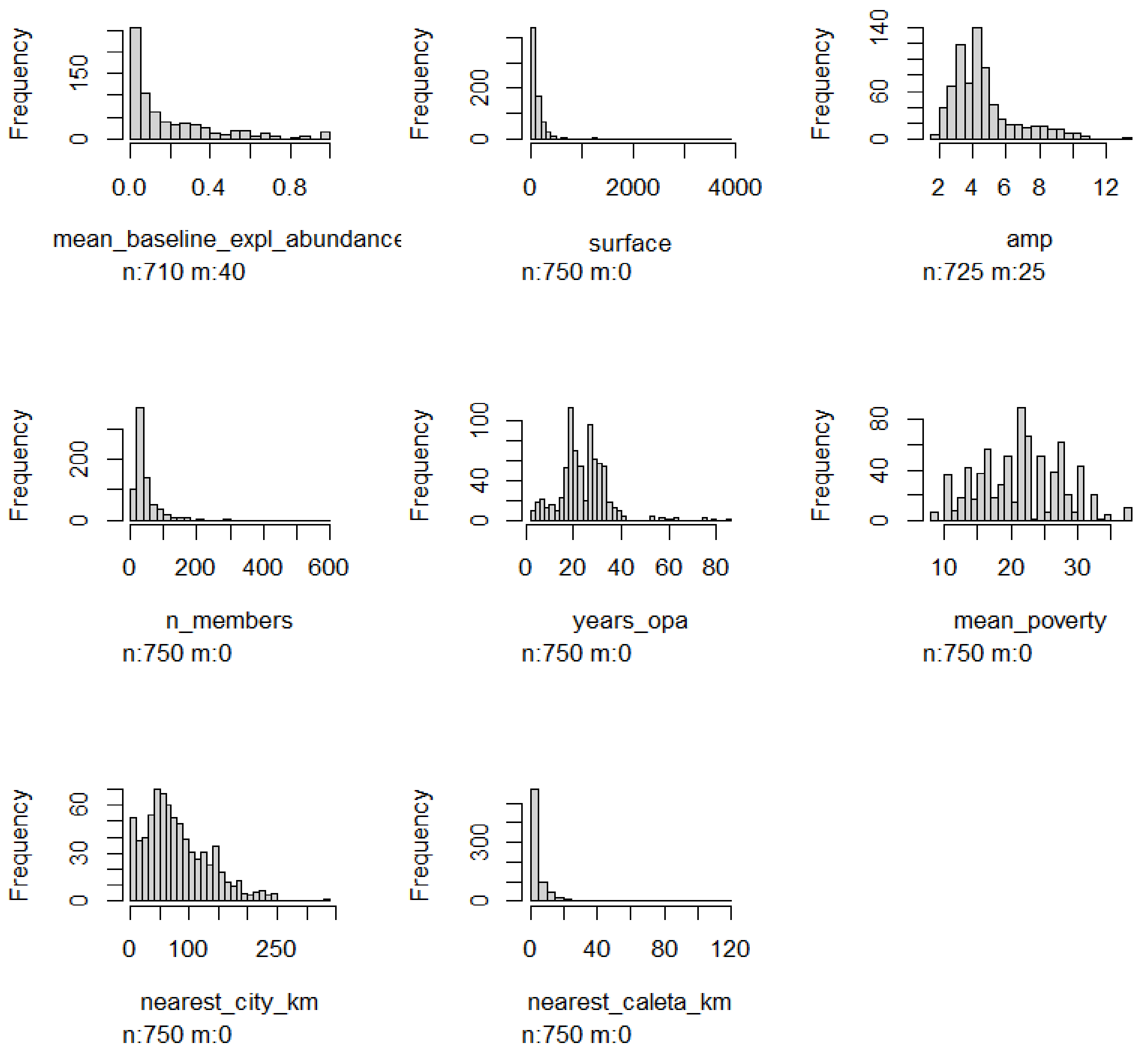

Figure B.1: Histograms of the distribution of values of the indicators used to characterize conditions for collective action before being scaled and transformed. The number of missing observations is indicated by ”m”.

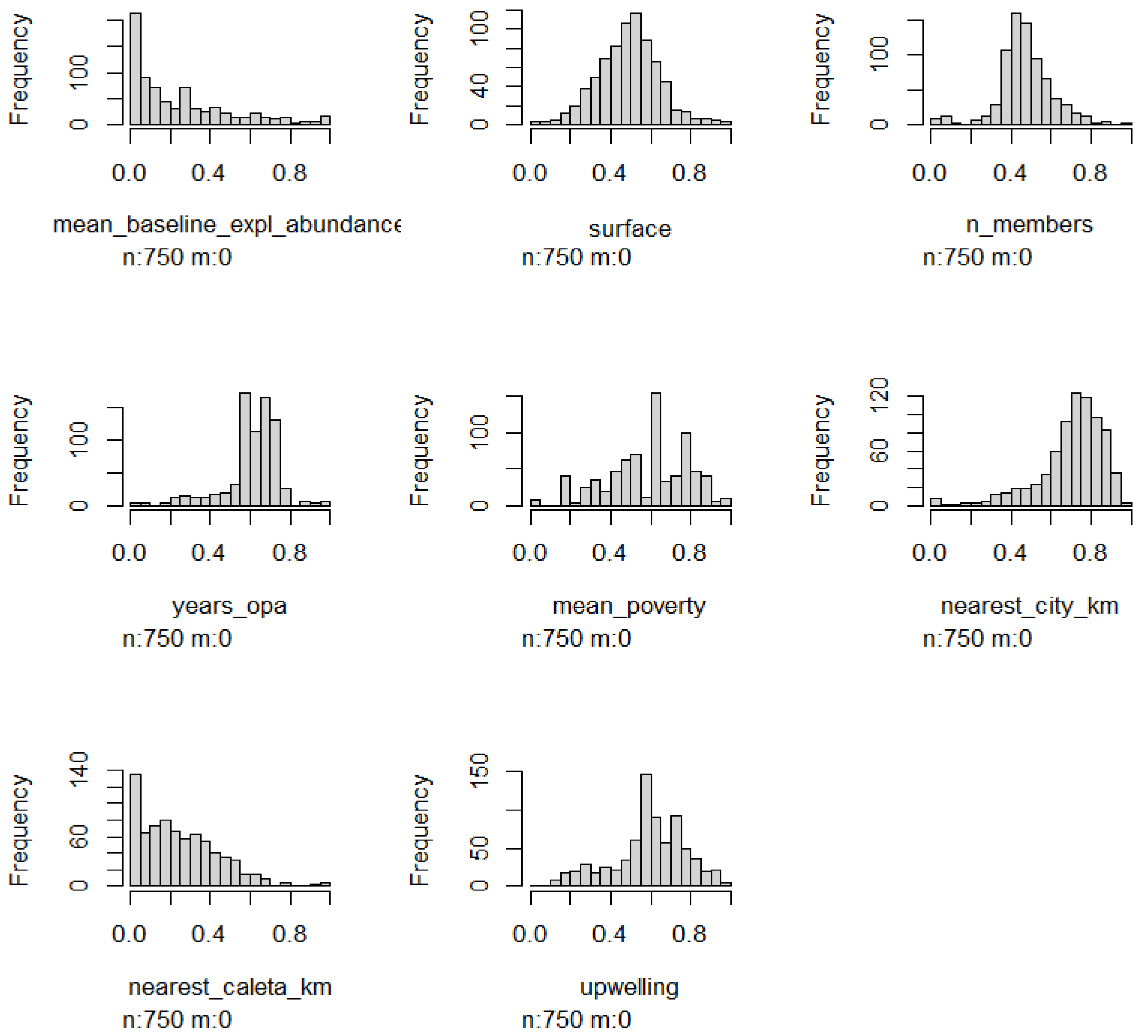

Figure B.2: Histograms of the distribution of values of the indicators used to characterize conditions for collective action after being scaled and transformed. The number of missing observations is indicated by "m".

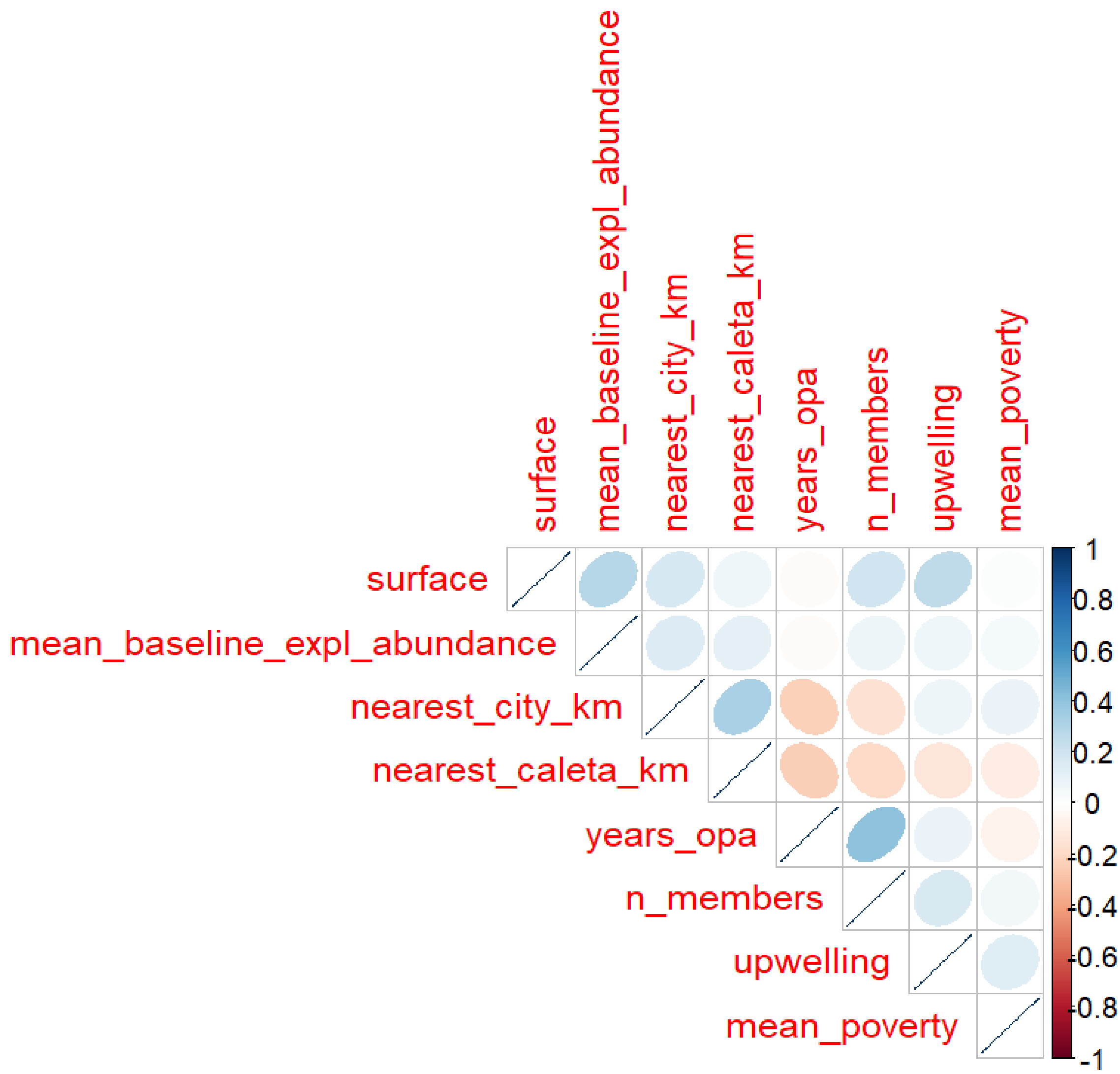

Figure B.3: Correlation plot for indicators used to cluster TURF-FCs.

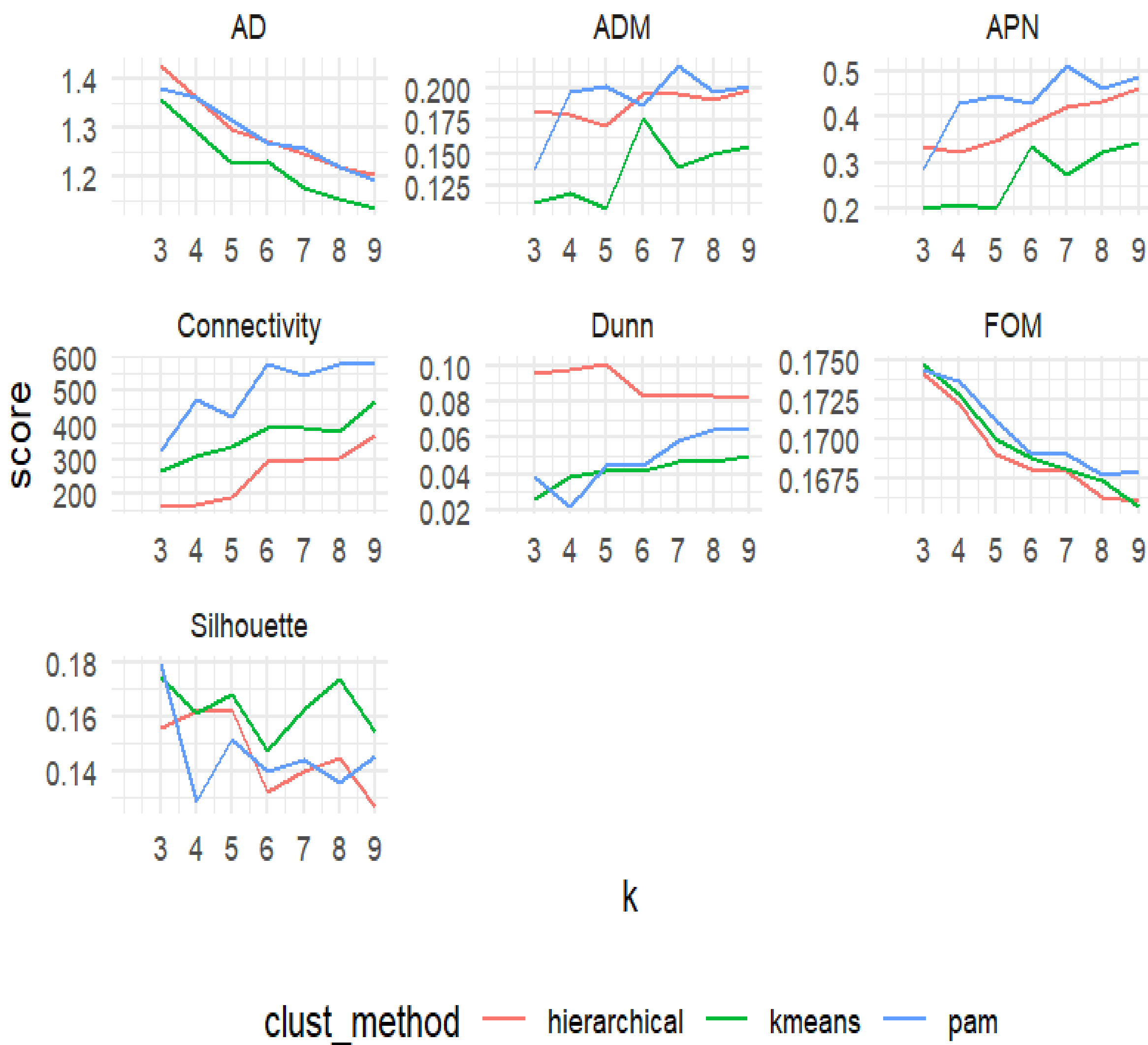

Figure B.4: Metrics for internal validation and stability obtained using three widely-used clustering algorithms-K-means, Hierarchical, and Partitioning Around Medoids (PAM)-across various cluster counts (k) using the 'clValid' package.

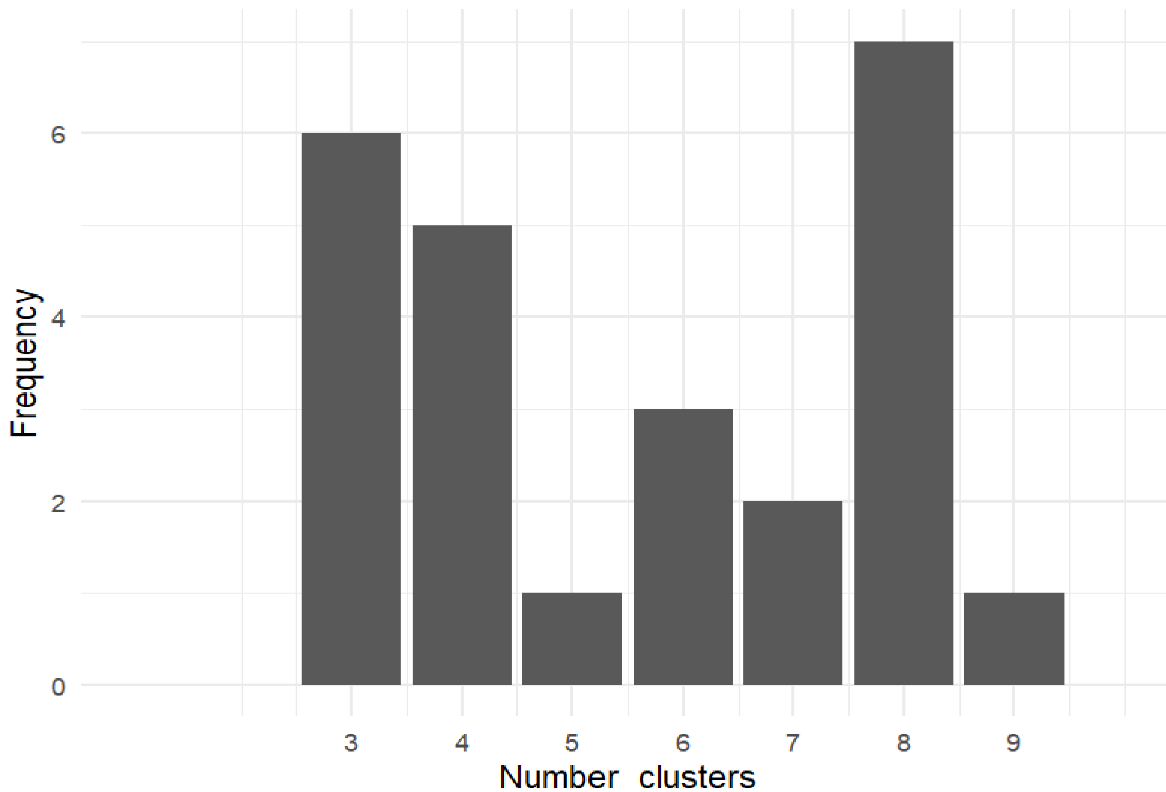

Figure B.5: Frequency with which each of 25 indices recommended each cluster count as the optimal. Computed using the ‘NbClust‘ package in R for the k-means clustering algorithm.

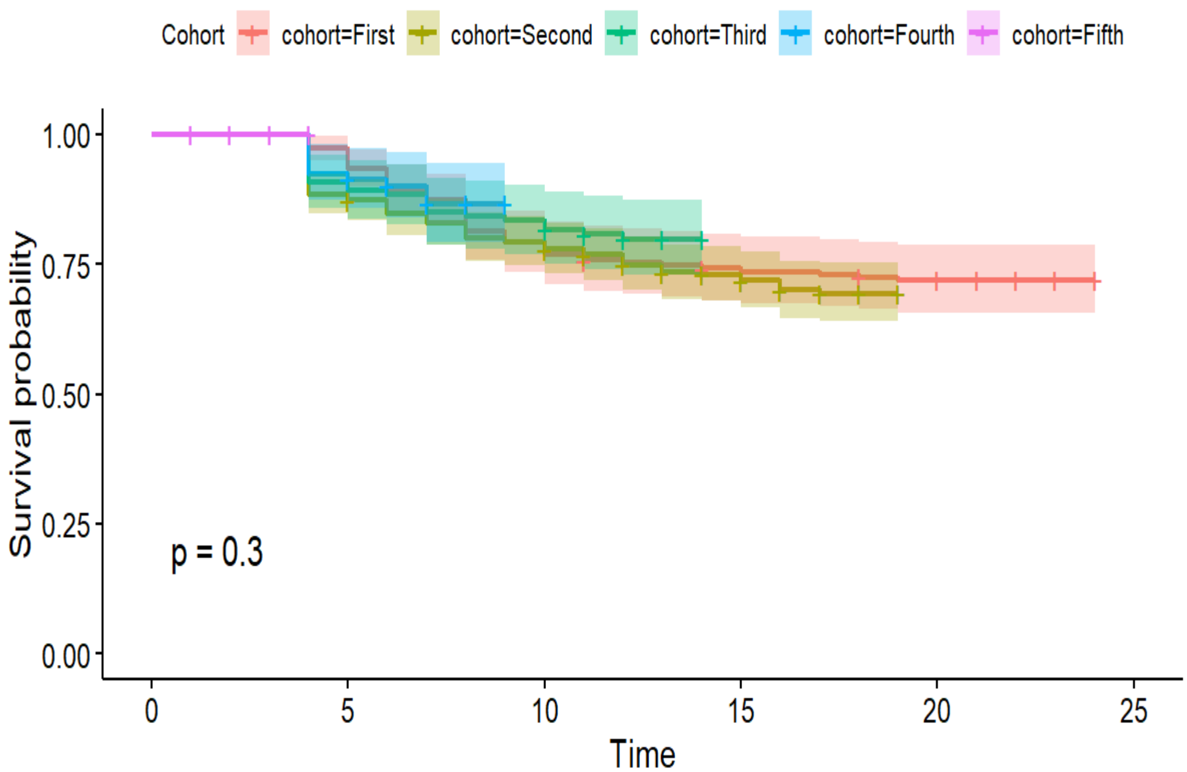

Figure B.6: Survival curves for each 5-year cohort of TURF-FCs.

Table B.1: The difference in observed means between clusters and the lower and upper endpoints of the interval for each indicator. The p-value is adjusted for multiple comparisons.

Feature	Clusters compared	Difference	Lower	Upper	Adj. p-value
Base_expl_abundance	2-1	-0.49	-0.52	-0.45	0.00
Base_expl_abundance	3-1	-0.51	-0.54	-0.48	0.00
Base_expl_abundance	3-2	-0.02	-0.05	0.00	0.12
surface	2-1	-0.11	-0.15	-0.07	0.00
surface	3-1	-0.11	-0.14	-0.08	0.00
surface	3-2	0.00	-0.03	0.03	0.97
upwelling	2-1	-0.11	-0.16	-0.07	0.00
upwelling	3-1	-0.01	-0.05	0.03	0.71
upwelling	3-2	0.10	0.06	0.14	0.00
n_members	2-1	-0.10	-0.13	-0.07	0.00
n_members	3-1	-0.02	-0.05	0.01	0.17
n_members	3-2	0.08	0.05	0.10	0.00
years_opa	2-1	-0.05	-0.08	-0.01	0.00
years_opa	3-1	0.04	0.01	0.06	0.01
years_opa	3-2	0.08	0.05	0.11	0.00
mean_poverty	2-1	-0.19	-0.24	-0.15	0.00
mean_poverty	3-1	0.06	0.02	0.10	0.00
mean_poverty	3-2	0.25	0.22	0.29	0.00
nearest_city_km	2-1	0.04	0.00	0.08	0.04
nearest_city_km	3-1	-0.09	-0.13	-0.06	0.00
nearest_city_km	3-2	-0.13	-0.17	-0.10	0.00
nearest_caleta_km	2-1	0.14	0.10	0.18	0.00
nearest_caleta_km	3-1	-0.12	-0.15	-0.09	0.00
nearest_caleta_km	3-2	-0.26	-0.29	-0.22	0.00

Appendix C

Appendix for Chapter 4

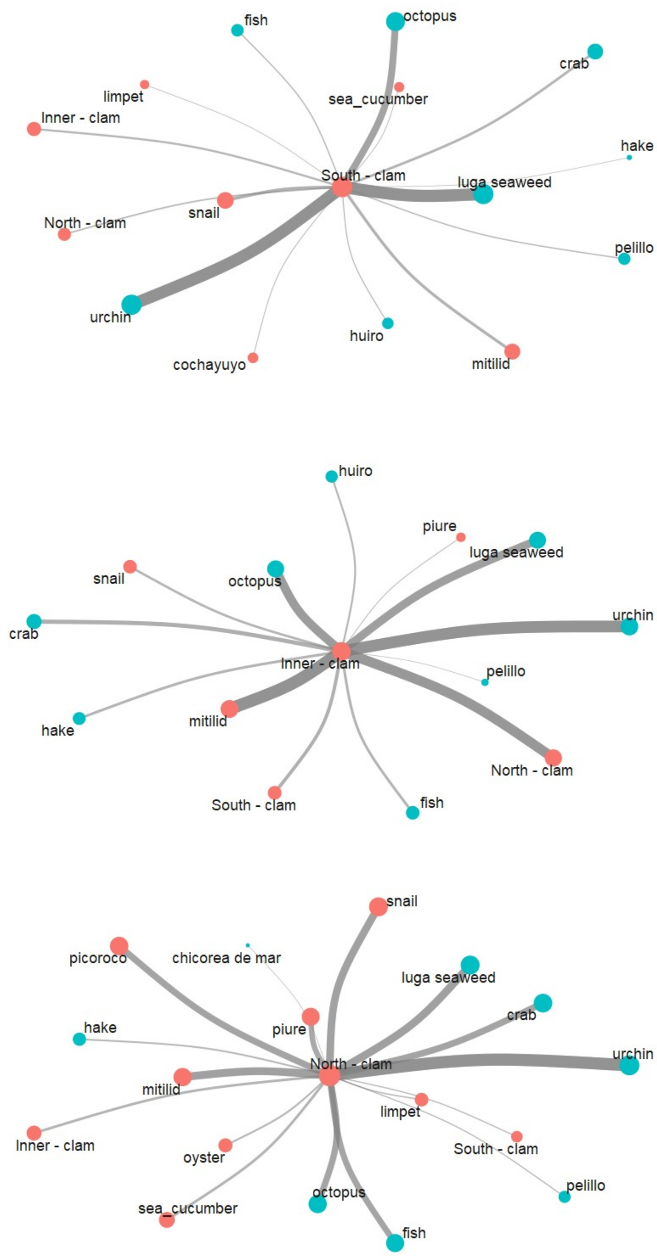

Figure C.1: Co-occurrence of resources in the pre-HAB landings of South- (n = 168), Inner- (n=70), and North-clam-reliant (n=232) boats. The color of the nodes displays whether a resource was closed during HAB (red) or open (blue) and the width of the edges and node sizes, represent the number of boats that landed a given pair of resources.

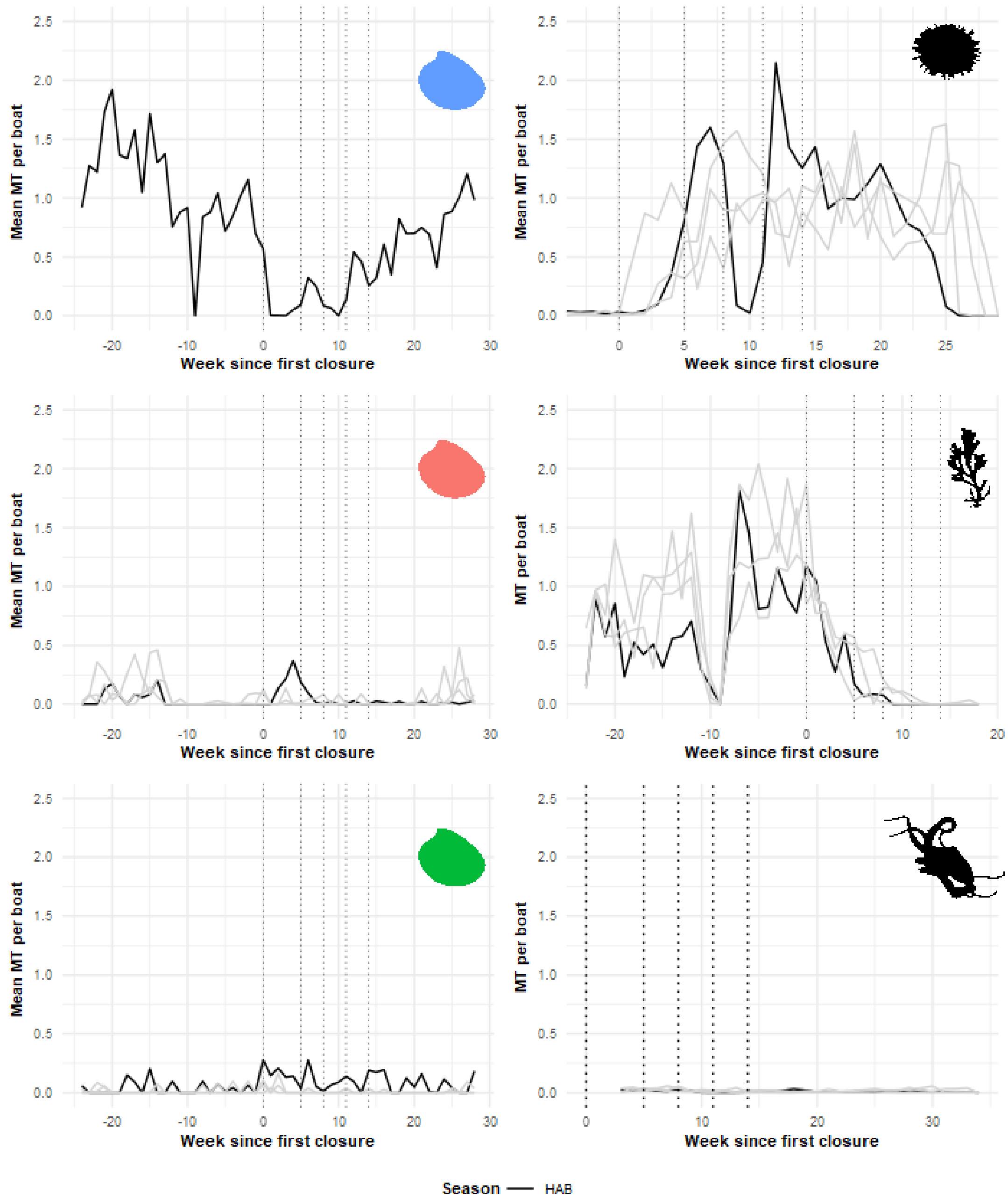

Figure C.2: Mean individual weekly landings by South-clam reliant boats of clams landed in the South (blue clam), Inner (red clam), and North areas (green clam), urchin, and Luga seaweed in pre-HAB seasons (gray lines) and the HAB season (black line).

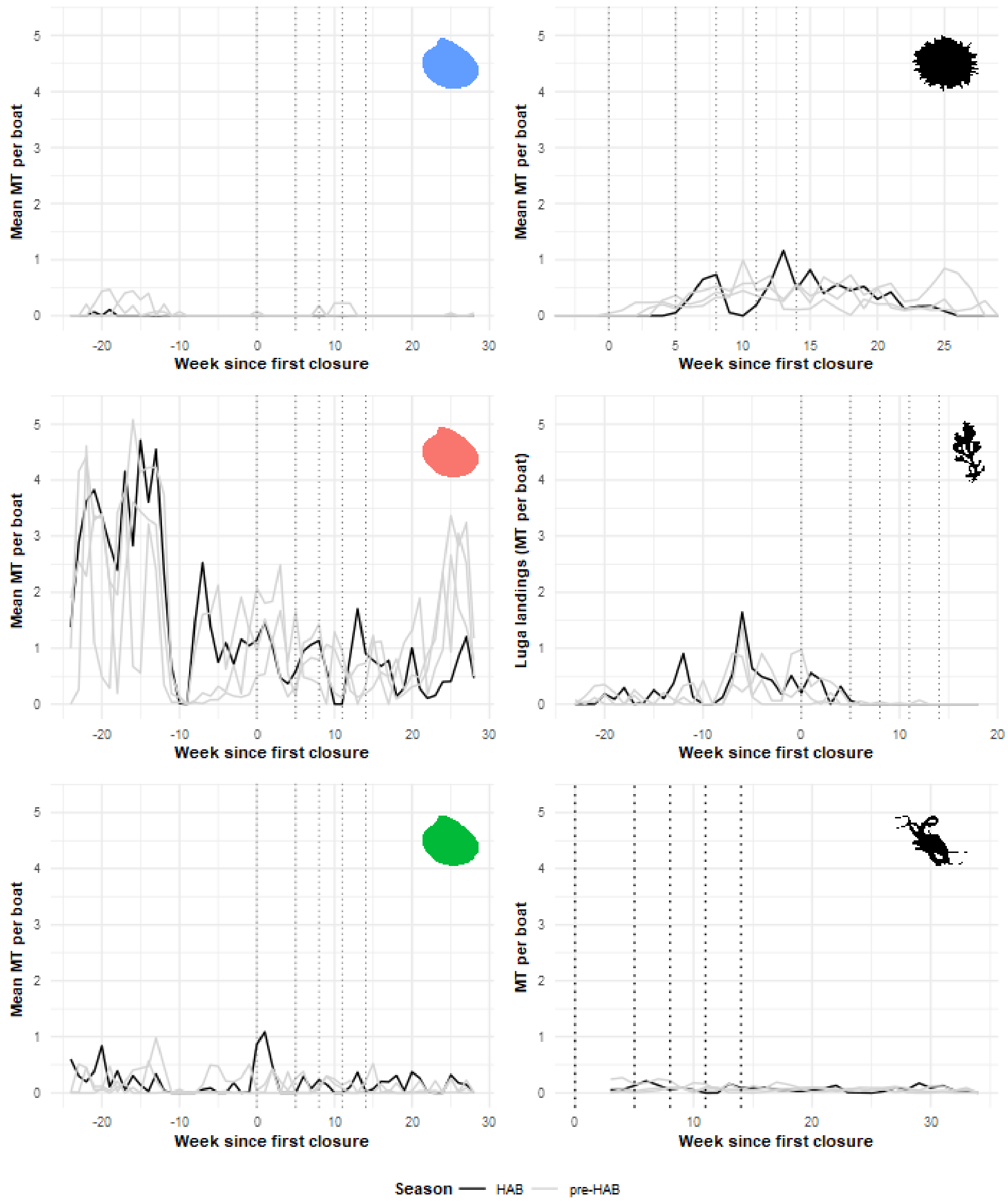

Figure C.3: Mean individual weekly landings by Inner-clam reliant boats of clams landed in the South (blue clam), Inner (red clam), and North area (green clam), urchin, and Luga seaweed in pre-HAB seasons (gray lines) and the HAB season (black line)..

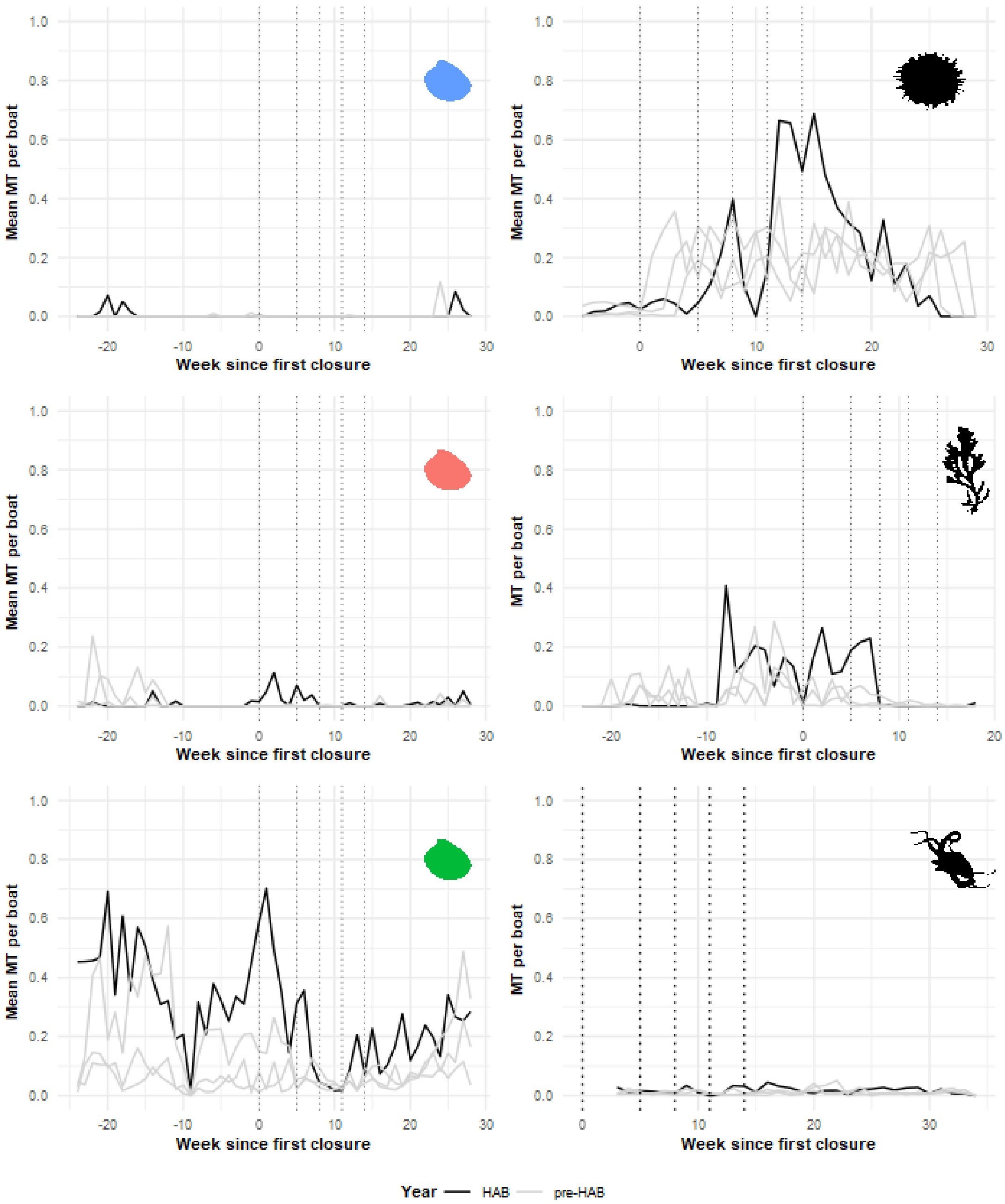

Figure C.4: Mean individual weekly landings by North-clam reliant boats of clams landed in the South (blue clam), Inner (red clam), and North area (green clam), urchin, and Luga seaweed in pre-HAB seasons (gray lines) and the HAB season (black line).

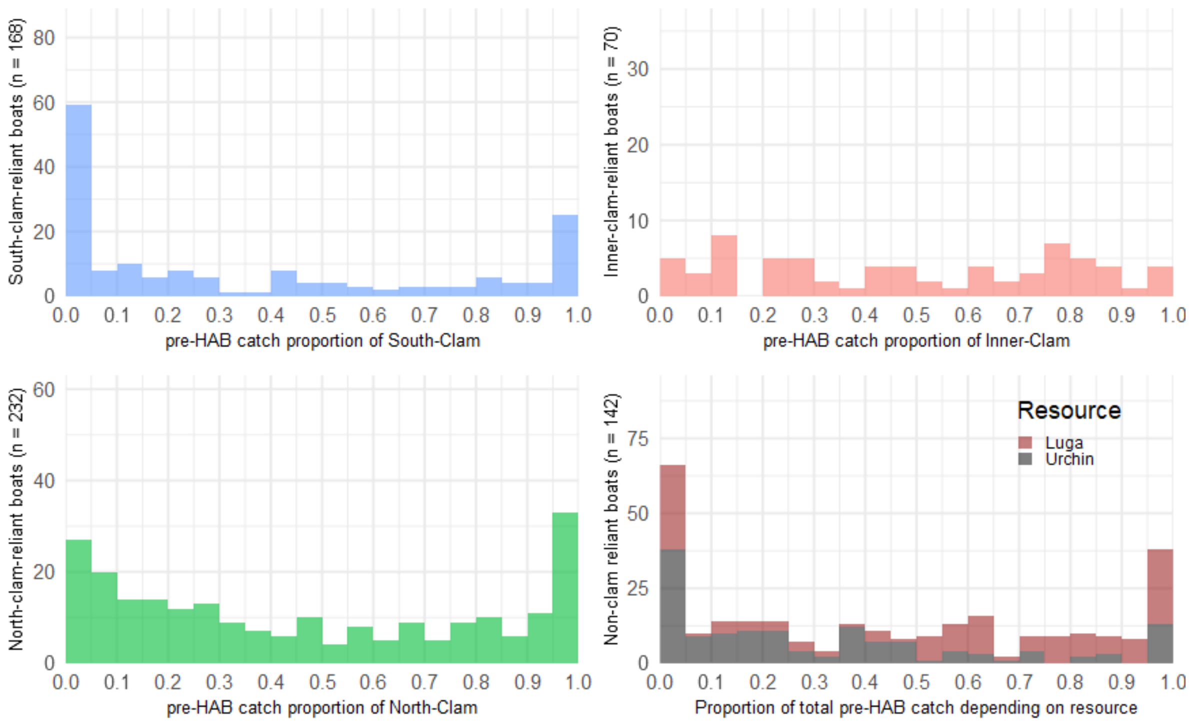

Figure C.5: Proportion of pre-HAB landings corresponding to clam landed in the primary area for samples of clam-reliant boats and proportion of pre-HAB landings corresponding to urchin and luga for the non-clam-reliant boats sample.

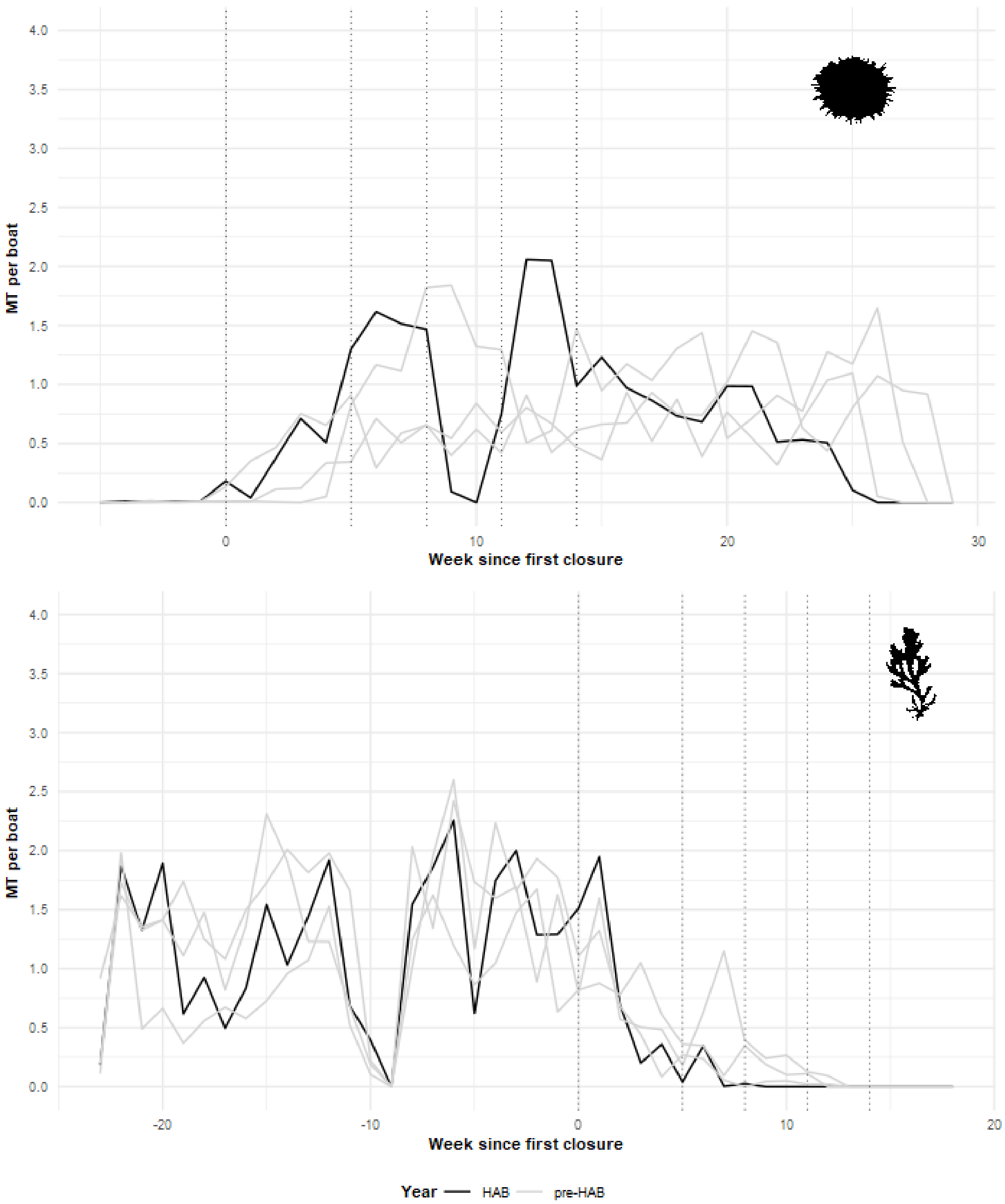

Figure C.6: Mean individual weekly landings by non-clam reliant boats of urchin, and Luga seaweed in pre-HAB seasons (gray lines) and the HAB season (black line).

Dynamic difference-in-difference analyses to check support for parallel trend assumption

We conducted dynamic difference-in-difference analyses, sometimes referred to as event studies, to assess whether there were parallel trends between landings during the HAB season and the pre-HAB seasons in the weeks preceding the closures (i.e., the pre-treatment period). This type of analysis involves comparing the differences in outcomes between a control group and a treated group for each time period, using the difference in a reference period as the baseline (Huntington-Klein, 2022). The week immediately before the closures was as our reference week. The estimation of the effect of the HAB season in each week comes from Equation (C.1).

$$Y_{i,t,j} = \alpha + \sum_{\substack{k=-K \\ k\neq 0}}^{K} \beta_k W_{t,k} \times \mathrm{HAB}_i + \gamma_{i,t} + \theta_j + \epsilon_{i,t,j} \tag{C.1}$$

Where $Y_{i,t,j}$ is the resource landed in MT by boat i in week t, and season j. The variable k indicates the week relative to the reference week. Specifically $k < 0$ represents weeks preceding the first closure, $k > 0$ denotes the week of the first closure and subsequent weeks, and $k = 0$ denotes the week immediately before the first closure. This week is excluded from the summation and is encapsulated within the intercept α.The term $W_{t,k}$ is a binary indicator, equal to one when week t corresponds to being k weeks away from the reference week, and zero otherwise. Coefficients β_k capture the effect of the HAB season in each week compared to the effect in the week prior to the first closure.HAB_i is an indicator variable that is equal to one if the observation originates from the HAB season and zero if it is from a pre-HAB season. Fixed effects for boat i and week t are denoted by $\gamma_{i,t}$, and $theta_j$ represents a fixed effect for each season j.

Support for the parallel trend assumption is seen when β_k coefficients for pre treatment periods ($k < 0$) are statistically insignificant. This indicates that the differences in landings between the pre-HAB seasons and the HAB season remained consistent until the closures began. However, when analyzing multiple pre-treatment periods, as in our case, significant coefficients may occur by chance (Huntington-Klein, 2022).

We ran a separate dynamic difference-in-difference analysis for each sample (i.e., south-clam-reliant, inner-clam-reliant, north-clam-reliant, and non-clam reliant boats) and outcome variables (i.e., landings of clam in the South, Inner, and North fishing areas, *Luga*, urchin, and total landings). Results are presented graphically in C.7, C.8, C.9, and C.10 for South-clam-reliant, Inner-clam-reliant, North-clam-reliant and non-clam-reliant boats.

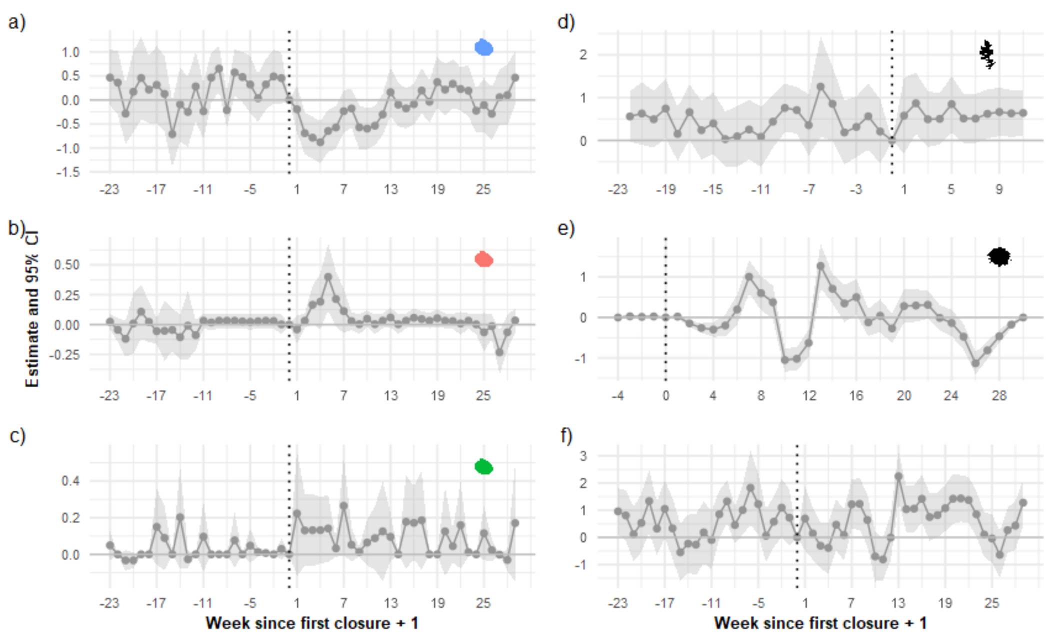

Figure C.7: Dynamic difference-in-difference for landings of clam in the South (a), Inner (b), and North (c) areas, luga (d), urchin (e), and total (f) by South-clam-reliant boats. Reference week is equal to zero and corresponds to the week right before the first closure.

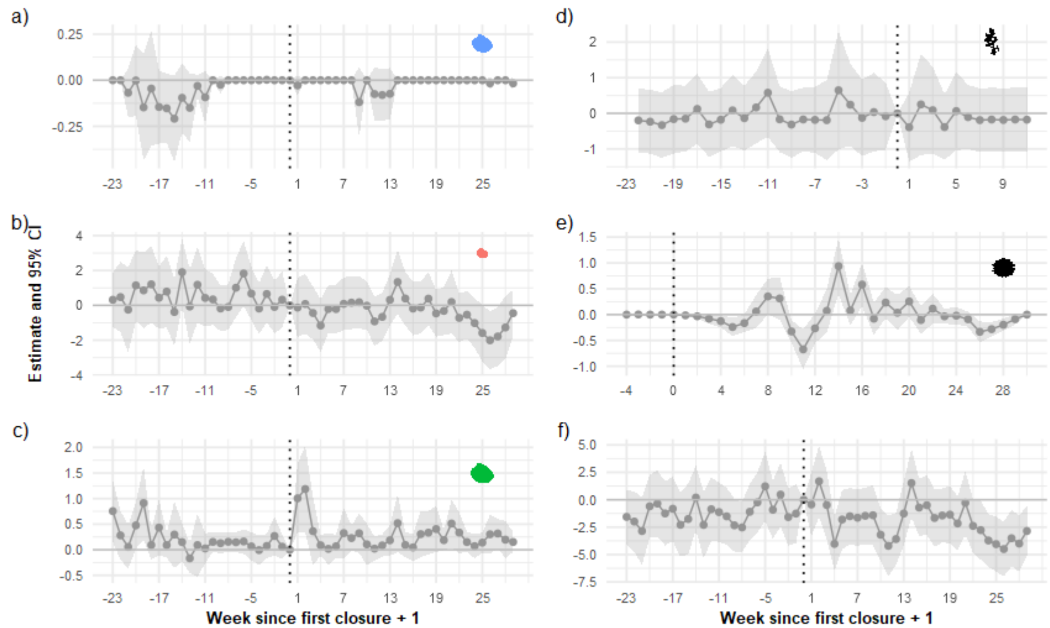

Figure C.8: Dynamic difference-in-difference for landings of clam in the South (a), Inner (b), and North (c) areas, luga (d), urchin (e), and total (f) by Inner-clam-reliant boats. Reference week is equal to zero and corresponds to the week right before the first closure.

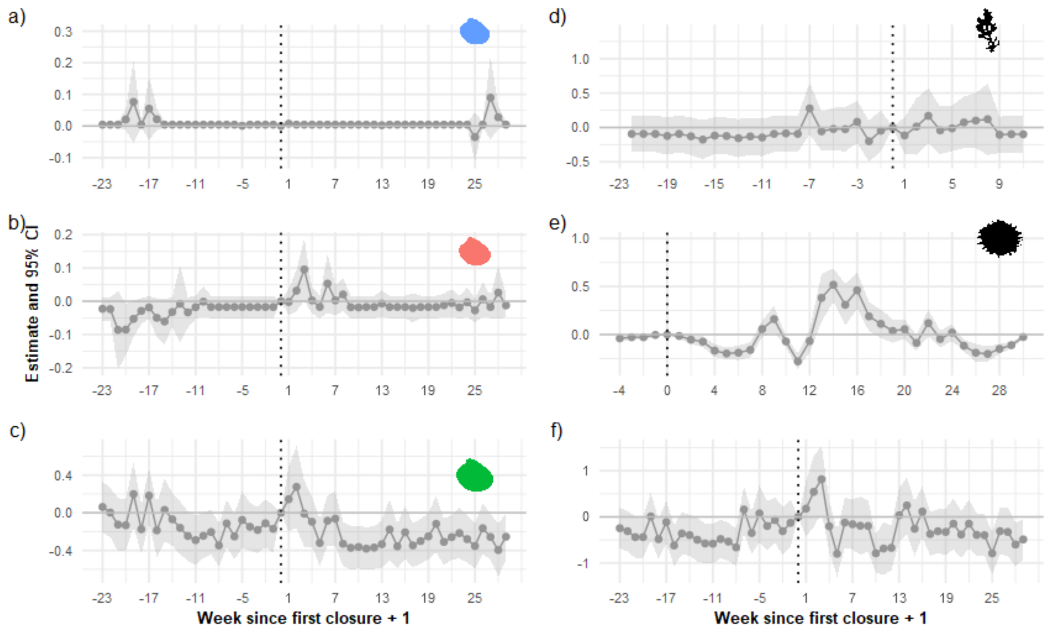

Figure C.9: Dynamic difference-in-difference for landings of clam in the South (a), Inner (b), and North (c) areas, luga (d), urchin (e), and total (f) by Nurth-clam-reliant boats. Reference week is equal to zero and corresponds to the week right before the first closure.

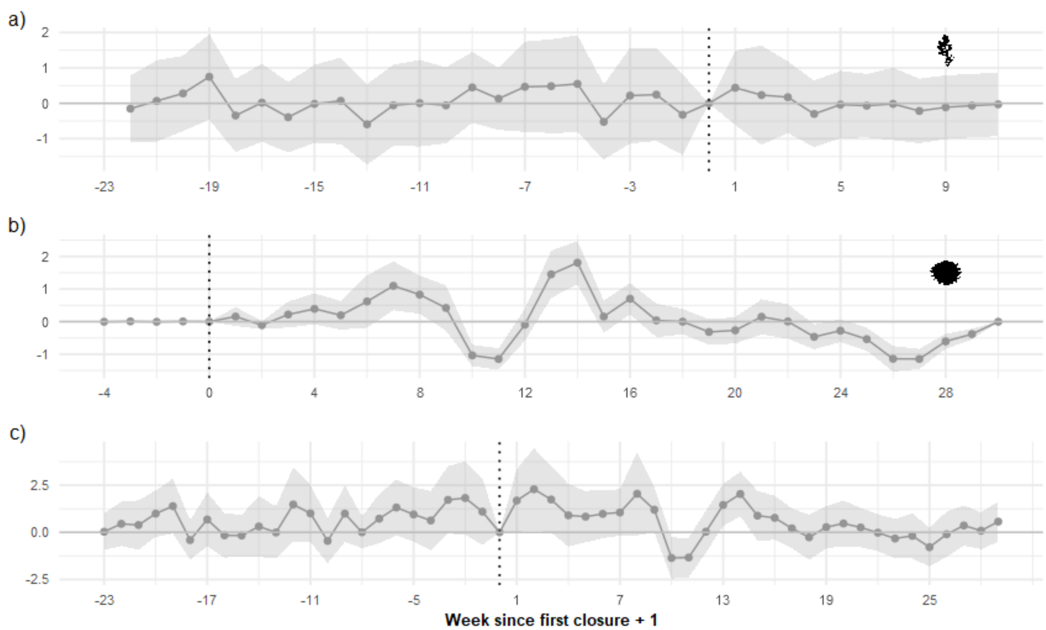

Figure C.10: Dynamic difference-in-difference for landings of luga (a), urchin (b), and total (c) by non-clam-reliant boats. Reference week is equal to zero and corresponds to the week right before the first closure.

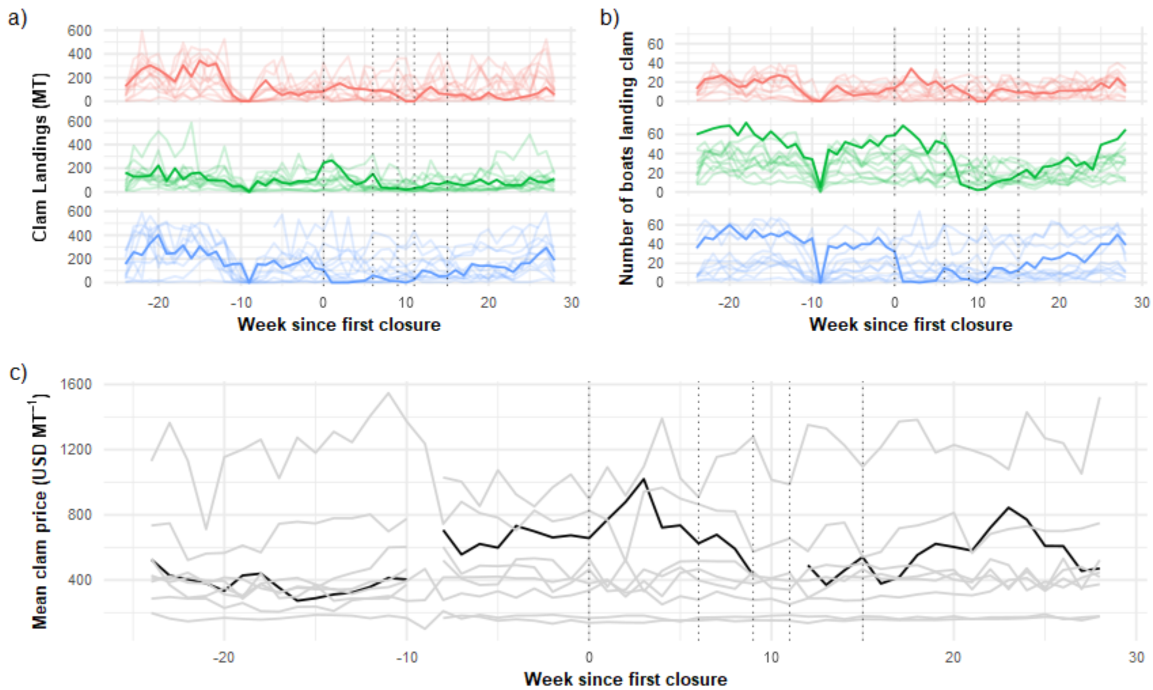

Figure C.11: Weekly trends in clam fisheries in pre-HAB seasons (in gray) and HAB seasons (in black) of (a) total landings of clam, (b) number of boats landing clams in teh south (blue), inner (red), and north (green) fishing areas. Pannel (c) shows the synthetic control constructed for clam prices for the HAB season (dark blue) and the observed price (black) during the HAB season. The shaded area represent around the synthetic control indicates 90 percent prediction intervals.

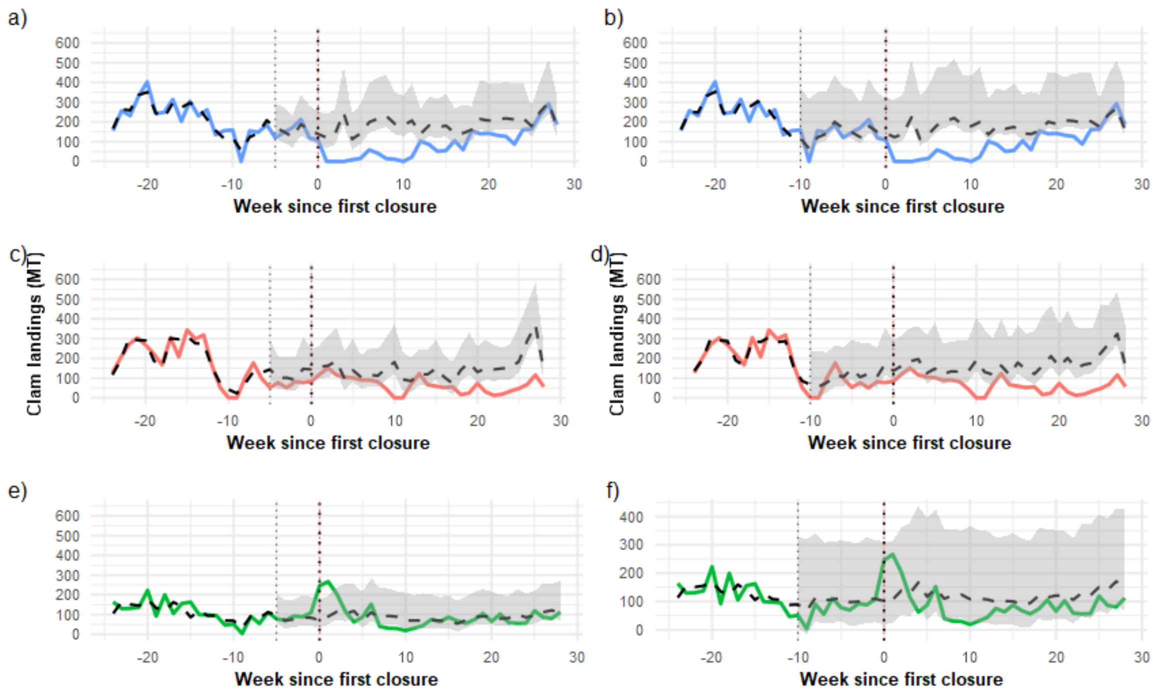

Figure C.12: In-time placebo test for the weekly landings of clam (MT) in each fishing area. The blue solid line represents the observed landings in the South (blue), Inner (red), and North (green) fishing areas. The dashed line represent the synthetic control generated with the treatment backdated 5 (a,c,e) and 10 (b,d,f) periods. The shaded gray area represents 90 percent prediction intervals.

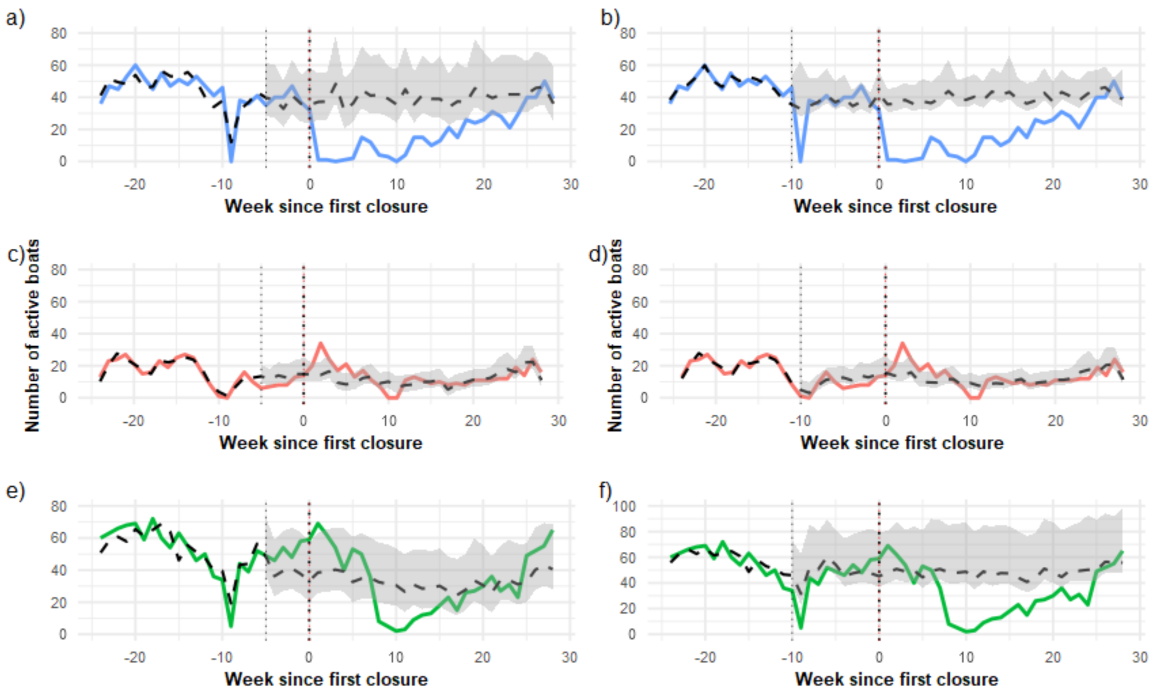

Figure C.13: In-time placebo test for the weekly number of boats landing clams in each fishing area. The blue solid line represents the observed number of boats in the South (blue), Inner (red), and North (green) fishing areas. The dashed line represent the synthetic control generated with the treatment backdated 5 (a,c,e) and 10 (b,d,f) periods. The shaded gray area represents 90 percent prediction intervals.

Figure C.14: In-time placebo test for weekly clam price in the Los Lagos region (USD/MT). The colored solid line represents the observed mean weekly price. The dashed line represent the synthetic control generated with the treatment backdated 5 (a) and 10 (b) periods. The shaded gray area represents 90 percent prediction intervals.

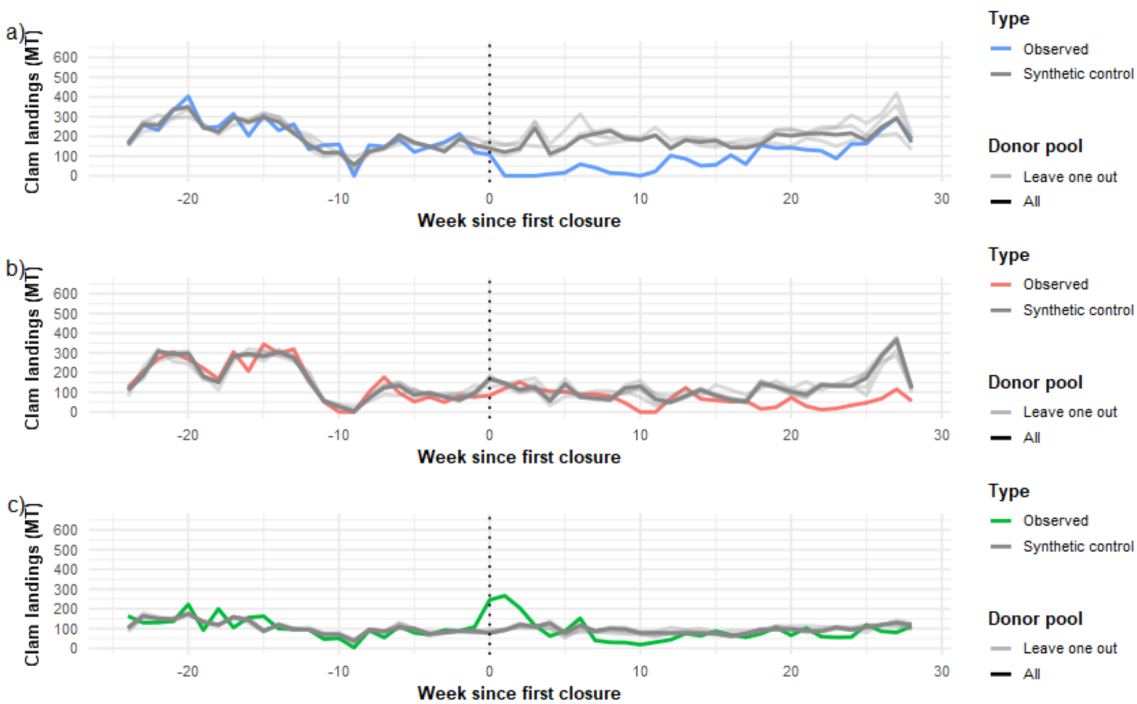

Figure C.15: Leave-one-out robustness check for clam landings (MT). Light gray lines represent synthetic controls for clam landings in the South (a), Inner (b), and North (c) fishing areas generated by iteratively leaving one active donor out of the donor pool. The dark gray line represents the synthetic control built with all the donor units. The colored line represents the total weekly landings of clam observed in the South (blue), Inner (red), and North (green) fihsing areas.

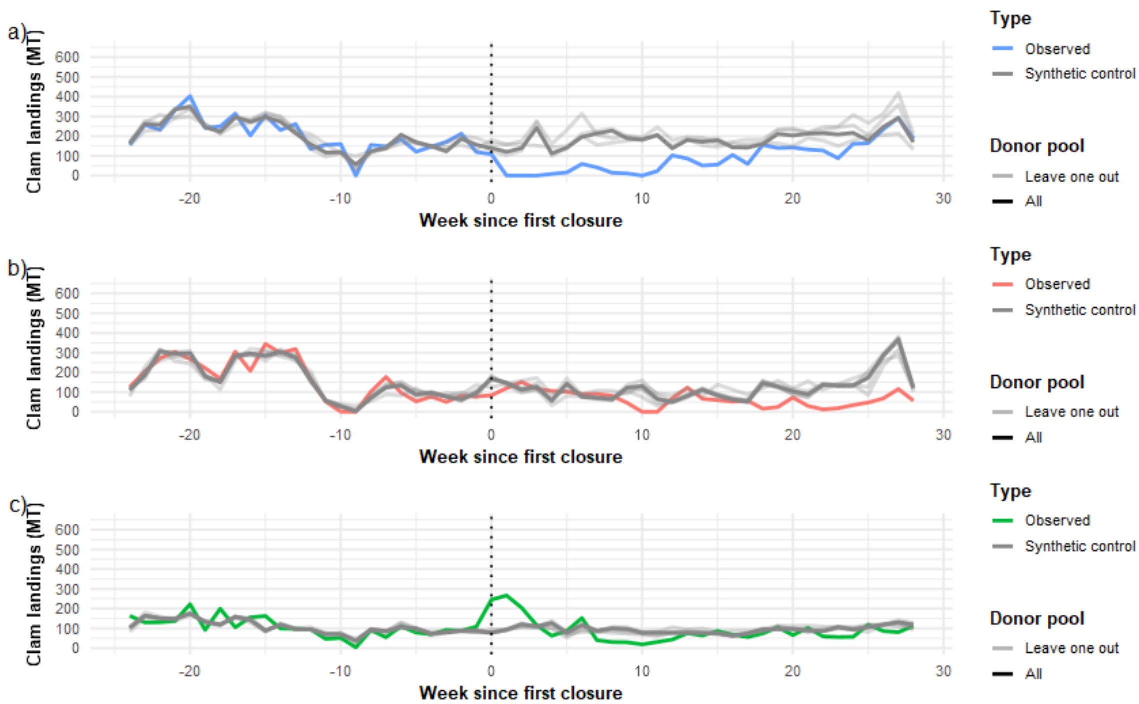

Figure C.16: Leave-one-out robustness check for number of boats landing clams. Light gray lines represent synthetic controls for the number of boats in the South (a), Inner (b), and North (c) fishing areas generated by iteratively leaving one active donor out of the donor pool. The dark gray line represents the synthetic control built with all the donor units. The colored line represents the observed number of boats landing clams in the South (blue), Inner (red), and North (green) fihsing areas in MT.

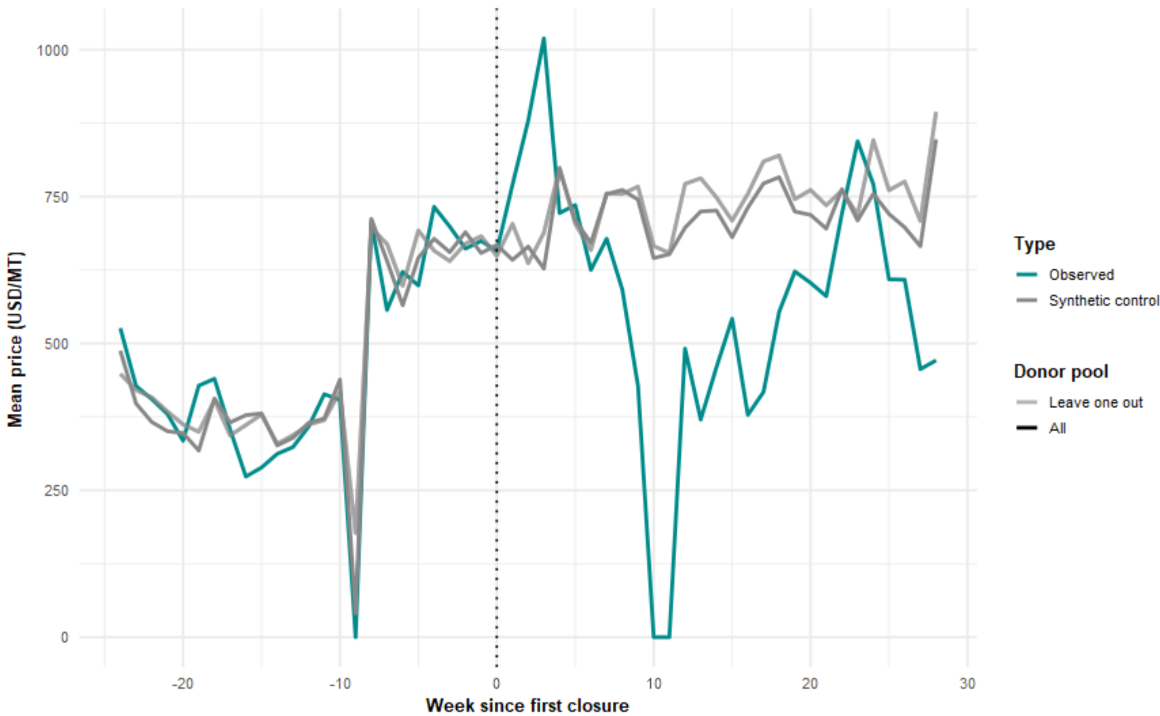

Figure C.17: Leave-one-out robustness check for mean weekly regional clam price (USD/MT). Light gray lines represent synthetic controls for the mean clam price per week in the los Lagos Region, generated by iteratively leaving one active donor out of the donor pool. The dark gray line represents the synthetic control built with all the donor units. The colored line represents weekly mean clam price observed.

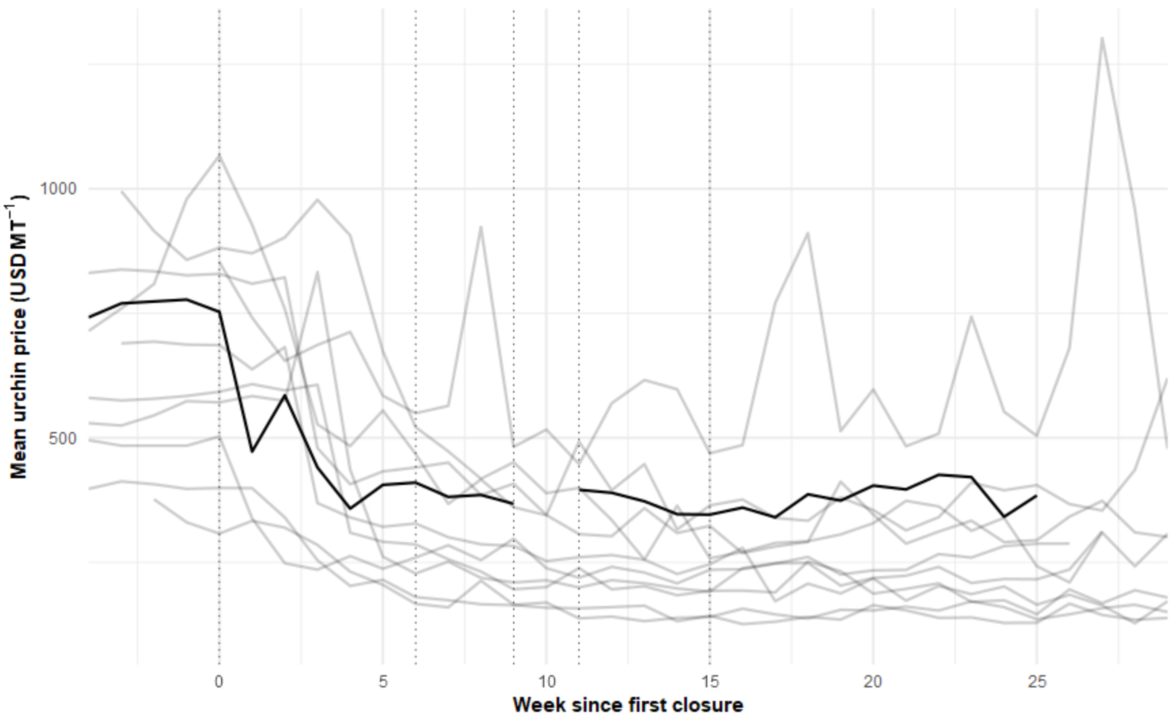

Figure C.18: Weekly trends in mean weekly urchin price (USD/MT) in pre-HAB seasons (in gray) and HAB seasons (in black).

Table C.1: Summary clam seasons

Area	Start date	End date	Landings (MT)	Boats number	Mean price (USD/MT)
Inner	2005-09-18	2006-09-14	2935.340	37	350.9124
Inner	2006-09-21	2007-09-12	3051.925	57	189.1819
Inner	2007-09-21	2008-09-15	2179.300	36	175.0365
Inner	2008-09-16	2009-09-15	6893.331	69	-211.0597
Inner	2009-09-22	2010-09-15	11102.223	77	263.2210
Inner	2010-09-20	2011-09-14	7288.347	71	329.0794
Inner	2011-09-19	2012-09-11	8618.510	69	1104.3863
Inner	2012-09-19	2013-09-15	5709.835	71	1204.4569
Inner	2013-09-16	2014-09-14	6539.777	74	564.3860
Inner	2014-09-17	2015-09-15	7509.560	85	445.9999
Inner	2015-09-19	2016-09-15	5873.994	121	556.0075
North	2005-09-16	2006-09-15	8385.362	215	432.3721
North	2006-09-16	2007-09-15	7970.429	211	299.6799
North	2007-09-16	2008-09-15	3840.852	153	223.6596
North	2008-09-16	2009-09-15	4729.271	175	-302.6336
North	2009-09-16	2010-09-15	5059.639	233	150.9842
North	2010-09-16	2011-09-15	6612.413	246	377.8631
North	2011-09-16	2012-09-15	2165.126	175	782.6246
North	2012-09-16	2013-09-15	1511.488	171	913.0235
North	2013-09-16	2014-09-15	1277.553	168	577.8821
North	2014-09-16	2015-09-15	3717.200	241	421.6066
North	2015-09-16	2016-09-15	5171.533	310	536.0477
South	2005-09-21	2006-09-14	617.945	31	271.4397
South	2006-09-21	2007-09-14	2142.895	89	187.4695
South	2007-09-17	2008-09-14	13783.382	102	123.7325
South	2008-09-16	2009-09-15	10818.118	149	-259.2601
South	2009-09-16	2010-09-15	15681.118	212	114.5613
South	2010-09-17	2011-09-15	16857.316	202	267.4498
South	2011-09-16	2012-09-13	10829.369	158	562.7989
South	2012-09-17	2013-09-14	5969.287	87	733.4755
South	2013-09-19	2014-09-15	4578.818	97	455.0719
South	2014-09-16	2015-09-15	7783.606	152	325.2918
South	2015-09-16	2016-09-15	7459.715	193	434.6573

Table C.2: Summary Luga seasons

Start date	End date	Weeks number	Landings (MT)	Boats number	Mean price (USD/MT)
2012-09-20	2013-07-29	46	15427.63	237	610.3876
2013-09-06	2014-06-06	41	13767.08	302	400.3948
2014-09-14	2015-07-01	43	11336.56	306	268.4948
2015-09-28	2016-07-03	42	10028.61	313	291.2237

Table C.3: Summary urchin seasons

Start date	End date	Weeks number	Landings (MT)	Boats number	Mean price (USD/MT)
2005-01-02	2005-11-16	45 weeks	9494.73	445	221.75
2006-01-06	2006-11-01	43 weeks	8618.80	352	357.76
2007-01-02	2007-10-27	43 weeks	9908.25	360	156.76
2008-01-02	2008-10-02	39 weeks	9464.63	277	154.33
2009-01-28	2009-11-01	40 weeks	9756.69	364	-350.71
2010-01-13	2010-10-15	39 weeks	10452.40	386	340.97
2011-01-03	2011-10-27	42 weeks	9907.34	358	224.02
2012-02-04	2012-10-10	36 weeks	9676.25	354	642.45
2013-01-15	2013-09-13	34 weeks	10534.04	390	407.97
2014-01-23	2014-08-27	31 weeks	9781.56	366	254.78
2015-01-14	2015-08-25	32 weeks	10249.02	362	240.10
2016-01-28	2016-08-22	30 weeks	9520.65	477	397.07

Table C.4: Weights used to built synthetic control for clam landings in the South, Inner and North fishing areas

Season	Weight	Outcome
1	0.290	Landings in the South
10	0.000	Landings in the South
2	0.000	Landings in the South
3	0.000	Landings in the South
4	0.000	Landings in the South
5	0.302	Landings in the South
6	0.000	Landings in the South
7	0.000	Landings in the South
8	0.408	Landings in the South
9	0.000	Landings in the South
1	0.000	Landings in the Inner
10	0.143	Landings in the Inner
2	0.021	Landings in the Inner
3	0.000	Landings in the Inner
4	0.000	Landings in the Inner
5	0.064	Landings in the Inner
6	0.000	Landings in the Inner
7	0.264	Landings in the Inner
8	0.156	Landings in the Inner
9	0.353	Landings in the Inner
1	0.000	Landings in the North
10	0.000	Landings in the North
2	0.092	Landings in the North
3	0.067	Landings in the North
4	0.081	Landings in the North
5	0.285	Landings in the North
6	0.000	Landings in the North
7	0.154	Landings in the North
8	0.000	Landings in the North
9	0.320	Landings in the North

Table C.5: Weights used to built synthetic control for number of boats landing clams in the South, Inner and North fishing areas

Season	Weight	Outcome
1	0.095	Boats in the South
10	0.000	Boats in the South
2	0.000	Boats in the South
3	0.000	Boats in the South
4	0.000	Boats in the South
5	0.364	Boats in the South
6	0.196	Boats in the South
7	0.000	Boats in the South
8	0.345	Boats in the South
9	0.000	Boats in the South
1	0.157	Boats in the Inner
10	0.289	Boats in the Inner
2	0.000	Boats in the Inner
3	0.010	Boats in the Inner
4	0.000	Boats in the Inner
5	0.000	Boats in the Inner
6	0.069	Boats in the Inner
7	0.053	Boats in the Inner
8	0.205	Boats in the Inner
9	0.218	Boats in the Inner
1	0.000	Boats in the North
10	0.000	Boats in the North
2	0.396	Boats in the North
3	0.000	Boats in the North
4	0.000	Boats in the North
5	0.190	Boats in the North
6	0.069	Boats in the North
7	0.345	Boats in the North
8	0.000	Boats in the North
9	0.000	Boats in the North

Table C.6: Weights used to built synthetic control for clam regional prices

Season	Weight	Outcome
1	0.076	Clam price
10	0.562	Clam price
2	0.005	Clam price
3	0.000	Clam price
4	0.000	Clam price
5	0.000	Clam price
6	0.000	Clam price
7	0.357	Clam price
8	0.000	Clam price
9	0.000	Clam price

Table C.7: Coefficients from difference-in-difference estimation of the effect of the HAB event and its stages on the weekly landings of South-clam-reliant boats.

Outcome	Term	Estimate	Std.Error	P.value
South Clam (MT)	Stage I	-0.81	0.08	**
South Clam (MT)	Stage II	-0.51	0.09	**
South Clam (MT)	Stage III	-0.66	0.08	**
South Clam (MT)	Stage IV	-0.22	0.10	**
South Clam (MT)	Stage V	-0.08	0.07	0.27
Inner Clam (MT)	Stage I	0.16	0.04	**
Inner Clam (MT)	Stage II	0.05	0.03	0.13
Inner Clam (MT)	Stage III	0.03	0.02	0.07
Inner Clam (MT)	Stage IV	0.03	0.02	0.07
Inner Clam (MT)	Stage V	0.00	0.02	0.98
North Clam (MT)	Stage I	0.10	0.04	**
North Clam (MT)	Stage II	0.08	0.05	0.1
North Clam (MT)	Stage III	0.06	0.06	0.28
North Clam (MT)	Stage IV	0.06	0.05	0.21
North Clam (MT)	Stage V	0.04	0.02	0.09
Urchin (MT)	Stage I	-0.13	0.04	**
Urchin (MT)	Stage II	0.65	0.12	**
Urchin (MT)	Stage III	-0.91	0.09	**
Urchin (MT)	Stage IV	0.76	0.14	**
Urchin (MT)	Stage V	-0.16	0.04	**
Urchin (% Quota x 100)	Stage I	-0.12	0.05	**
Urchin (% Quota x 100)	Stage II	0.74	0.13	**
Urchin (% Quota x 100)	Stage III	-0.88	0.09	**
Urchin (% Quota x 100)	Stage IV	0.85	0.14	**
Urchin (% Quota x 100)	Stage V	-0.11	0.04	**
Luga (MT)	Stage I	0.20	0.10	0.05
Luga (MT)	Stage II	0.16	0.07	**
Total landings (MT)	Stage I	-0.45	0.17	**
Total landings (MT)	Stage II	0.46	0.21	**
Total landings (MT)	Stage III	-1.07	0.17	**
Total landings (MT)	Stage IV	0.88	0.18	**
Total landings (MT)	Stage V	0.18	0.11	0.09
South Clam (MT)	Total HAB	-0.35	0.06	**
Inner Clam (MT)	Total HAB	0.05	0.02	**
North Clam (MT)	Total HAB	0.06	0.02	**
Urchin (MT)	Total HAB	-0.06	0.03	0.09
Urchin (% Quota x 100)	Total HAB	-0.01	0.03	0.78
Luga (MT)	Total HAB	0.19	0.08	**
Total landings (MT)	Total HAB	0.02	0.10	0.81

Table C.8: Coefficients from difference-in-difference estimation of the effect of the HAB event and its stages on the weekly landings of Inner-clam-reliant boats.

Outcome	Term	Estimate	Std.Error	P.value
South Clam (MT)	Stage I	0.04	0.02	**
South Clam (MT)	Stage II	0.01	0.04	0.81
South Clam (MT)	Stage III	0.00	0.04	0.93
South Clam (MT)	Stage IV	0.02	0.03	0.4
South Clam (MT)	Stage V	0.05	0.01	**
Inner Clam (MT)	Stage I	-0.86	0.22	**
Inner Clam (MT)	Stage II	-0.37	0.26	0.16
Inner Clam (MT)	Stage III	-1.05	0.19	**
Inner Clam (MT)	Stage IV	0.17	0.31	0.59
Inner Clam (MT)	Stage V	-1.22	0.17	**
North Clam (MT)	Stage I	0.26	0.11	**
North Clam (MT)	Stage II	0.10	0.11	0.36
North Clam (MT)	Stage III	-0.13	0.08	0.1
North Clam (MT)	Stage IV	0.07	0.11	0.52
North Clam (MT)	Stage V	0.05	0.05	0.33
Urchin (MT)	Stage I	-0.11	0.03	**
Urchin (MT)	Stage II	0.24	0.11	**
Urchin (MT)	Stage III	-0.42	0.09	**
Urchin (MT)	Stage IV	0.36	0.13	**
Urchin (MT)	Stage V	0.00	0.03	0.96
Urchin (% Quota x 100)	Stage I	-0.11	0.03	**
Urchin (% Quota x 100)	Stage II	0.28	0.11	**
Urchin (% Quota x 100)	Stage III	-0.41	0.09	**
Urchin (% Quota x 100)	Stage IV	0.40	0.13	**
Urchin (% Quota x 100)	Stage V	0.02	0.03	0.62
Luga (MT)	Stage I	-0.03	0.15	0.82
Luga (MT)	Stage II	-0.14	0.06	**
Total landings (MT)	Stage I	0.04	0.41	0.91
Total landings (MT)	Stage II	-0.37	0.43	0.4
Total landings (MT)	Stage III	-2.52	0.36	**
Total landings (MT)	Stage IV	0.99	0.55	0.07
Total landings (MT)	Stage V	-1.37	0.24	**
South Clam (MT)	Total HAB	0.03	0.02	**
Inner Clam (MT)	Total HAB	-0.90	0.15	**
North Clam (MT)	Total HAB	0.08	0.05	0.09
Urchin (MT)	Total HAB	0.00	0.03	0.85
Urchin (% Quota x 100)	Total HAB	0.01	0.03	0.61
Luga (MT)	Total HAB	-0.07	0.11	0.52
Total landings (MT)	Total HAB	-0.85	0.22	**

Table C.9: Coefficients from difference-in-difference estimation of the effect of the HAB event and its stages on the weekly landings of North-clam-reliant boats.

Outcome	Term	Estimate	Std.Error	P.value
South Clam (MT)	Stage I	-0.01	0.00	0.12
South Clam (MT)	Stage II	-0.01	0.00	0.09
South Clam (MT)	Stage III	-0.01	0.00	0.09
South Clam (MT)	Stage IV	-0.01	0.00	0.07
South Clam (MT)	Stage V	0.00	0.01	0.85
Inner Clam (MT)	Stage I	0.06	0.01	**
Inner Clam (MT)	Stage II	0.03	0.01	**
Inner Clam (MT)	Stage III	0.01	0.00	**
Inner Clam (MT)	Stage IV	0.01	0.01	**
Inner Clam (MT)	Stage V	0.02	0.01	**
North Clam (MT)	Stage I	0.10	0.05	0.05
North Clam (MT)	Stage II	-0.14	0.04	**
North Clam (MT)	Stage III	-0.26	0.02	**
North Clam (MT)	Stage IV	-0.17	0.05	**
North Clam (MT)	Stage V	-0.15	0.02	**
Urchin (MT)	Stage I	-0.10	0.01	**
Urchin (MT)	Stage II	0.04	0.03	0.2
Urchin (MT)	Stage III	-0.12	0.03	**
Urchin (MT)	Stage IV	0.42	0.06	**
Urchin (MT)	Stage V	0.02	0.01	0.09
Urchin (% Quota x 100)	Stage I	-0.10	0.01	**
Urchin (% Quota x 100)	Stage II	0.05	0.03	0.08
Urchin (% Quota x 100)	Stage III	-0.11	0.03	**
Urchin (% Quota x 100)	Stage IV	0.45	0.07	**
Urchin (% Quota x 100)	Stage V	0.04	0.01	**
Luga (MT)	Stage I	0.09	0.05	0.08
Luga (MT)	Stage II	0.11	0.09	0.22
Total landings (MT)	Stage I	0.38	0.10	**
Total landings (MT)	Stage II	0.13	0.13	0.29
Total landings (MT)	Stage III	-0.40	0.09	**
Total landings (MT)	Stage IV	0.32	0.13	**
Total landings (MT)	Stage V	-0.03	0.05	0.47
South Clam (MT)	Total HAB	0.00	0.00	0.41
Inner Clam (MT)	Total HAB	0.03	0.01	**
North Clam (MT)	Total HAB	-0.11	0.02	**
Urchin (MT)	Total HAB	0.03	0.01	**
Urchin (% Quota x 100)	Total HAB	0.04	0.01	**
Luga (MT)	Total HAB	0.10	0.05	**
Total landings (MT)	Total HAB	0.07	0.04	0.12

Table C.10: Coefficients from difference-in-difference estimation of the effect of the HAB event and its stages on the weekly landings of non-clam-reliant boats.

Outcome	Term	Estimate	Std.Error	P.value
South Clam (MT)	Stage I	0.00	0.00	0.13
South Clam (MT)	Stage II	0.00	0.00	0.11
South Clam (MT)	Stage III	0.00	0.00	0.11
South Clam (MT)	Stage IV	0.00	0.00	0.11
South Clam (MT)	Stage V	0.04	0.01	**
Inner Clam (MT)	Stage I	0.05	0.02	**
Inner Clam (MT)	Stage II	0.00	0.00	0.07
Inner Clam (MT)	Stage III	0.00	0.00	0.07
Inner Clam (MT)	Stage IV	0.00	0.00	0.07
Inner Clam (MT)	Stage V	0.00	0.00	0.05
North Clam (MT)	Stage I	0.02	0.01	**
North Clam (MT)	Stage II	0.00	0.00	0.32
North Clam (MT)	Stage III	0.00	0.00	**
North Clam (MT)	Stage IV	0.00	0.00	**
North Clam (MT)	Stage V	0.01	0.00	**
Urchin (MT)	Stage I	0.24	0.10	**
Urchin (MT)	Stage II	0.78	0.20	**
Urchin (MT)	Stage III	-0.76	0.12	**
Urchin (MT)	Stage IV	1.14	0.19	**
Urchin (MT)	Stage V	-0.28	0.05	**
Urchin (% Quota x 100)	Stage I	0.27	0.10	**
Urchin (% Quota x 100)	Stage II	0.89	0.21	**
Urchin (% Quota x 100)	Stage III	-0.73	0.12	**
Urchin (% Quota x 100)	Stage IV	1.26	0.20	**
Urchin (% Quota x 100)	Stage V	-0.23	0.05	**
Luga (MT)	Stage I	0.02	0.15	0.9
Luga (MT)	Stage II	-0.17	0.13	0.19
Total landings (MT)	Stage I	0.79	0.33	**
Total landings (MT)	Stage II	0.82	0.42	0.05
Total landings (MT)	Stage III	-1.51	0.24	**
Total landings (MT)	Stage IV	0.84	0.28	**
Total landings (MT)	Stage V	-0.52	0.15	**
South Clam (MT)	Total HAB	0.02	0.01	**
Inner Clam (MT)	Total HAB	0.01	0.00	**
North Clam (MT)	Total HAB	0.01	0.00	**
Urchin (MT)	Total HAB	0.02	0.05	0.62
Urchin (% Quota x 100)	Total HAB	0.08	0.05	0.08
Luga (MT)	Total HAB	-0.04	0.12	0.71
Total landings (MT)	Total HAB	-0.07	0.15	0.63

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