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Governance Arrangements That Fit Social-Ecological Context Are Associated With Fishery Sustainability

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ABSTRACT

In many parts of the world, natural resources are primarily managed by governance arrangements at the local level. Those arrangements range from collective, such as cooperatives and associations, to individualistic, such as patrons and owner-operators. A key question is how such arrangements influence resource use and associated social and ecological outcomes. We hypothesize that sustainable resource use is not associated with a specific governance arrangement, but instead that every type of arrangement has a certain set of social and ecological conditions under which sustainable resource use is feasible. This hypothesis was empirically investigated using fisheries landings data from Mexico. We found that sustainable resource use can be achieved through multiple governance arrangements provided that they are matched to appropriate social-ecological contexts. Our findings help identify which governance arrangements are best suited to particular contexts and provide the initial building blocks for developing a theory of governance contextual fit in social-ecological fishing systems.

1 | Introduction

Governance of shared natural resources, such as fisheries, is in many parts of the world dependent on local arrangements (Frawley et al. 2024; Ostrom 2005; Wilson 2006). Such localised governance arrangements are considered by many practitioners and scholars to be better suited to address the dynamics inherent in local social-ecological systems to which they belong, as opposed to one-size-fits-all prescriptions (Leslie et al. 2015; Ostrom 2007; Young, Webster, et al. 2018). Given their importance and relevance, more emphasis should be placed on understanding how diverse governance arrangements interact with the context-specific social and ecological conditions to foster positive outcomes (Aguión et al. 2025; Frawley et al. 2024; Gladstone et al. 2025).

Gaining such insights is particularly relevant for small-scale fisheries (SSF) as they are a critical provider of global food security, poverty alleviation and biodiversity conservation worldwide (Basurto et al. 2025; Short et al. 2021), They account for approximately 90% (60 million) of commercial fishers and 40% of global fish catches (Basurto et al. 2025). In addition, the livelihoods of an estimated 492M are at least partially dependent on SSF (Basurto et al. 2025). While we have considerable understanding of the types of governance interventions that foster sustainable resource use in industrial fisheries (e.g., Costello et al. 2008), knowledge of similar relationships in SSF—and many other natural resource management contexts characterised by local control and/or collective action—is limited. Empirical evidence suggests that co-management arrangements in which community-based organisations actively engage with government entities in resource management promote resource sustainability (Cucuzza et al. 2021; Gutiérrez et al. 2011). However, in many parts of the world, official (de jure) government entities in charge of fisheries management are either fully absent or financially and administratively constrained to carry out their mandated activities (Hilborn et al. 2020; Mora et al. 2009), thus limiting their ability to effectively engage in management.

From this perspective, most SSF governance and management functions primarily depend on resource users themselves (Townsend and Shotton 2008). Because of this widespread and largely inadvertent devolution of management and governance responsibilities, greater focus should be on governance arrangements that fishers use to self-organise their fishing activities, rather than a particular management tool (e.g., fishing quotas) or a governance regime (e.g., co-management). We define governance arrangements as a set of formal and informal rules and norms (i.e., institutions) that structure the relations of power and authority within a group (Kooiman 1999). Understanding the relationships between various types of governance arrangements and social-ecological conditions in achieving positive outcomes of SSF is important to inform the design of fisheries management systems that sustain healthy fisheries and ecosystems upon which fishers' livelihoods depend. This approach will help illuminate factors that contribute to effective collective action and community-based natural resource management more broadly (Poteete et al. 2010).

Local governance arrangements in SSF can be broadly characterised as either collective or individualistic relationships (Figure 1). The principal difference between these two general types is in their nature of decision-making, although there is considerable variation in terms of how this is manifested at the level of an individual fishing entity.

Collective arrangements come in many forms, from cooperatives and guilds to fishery management areas and associations (García-Lorenzo et al. 2019; Gelcich et al. 2010; Ovando et al. 2013; Valdés-Pizzini 1990). While collectives can vary substantially in their structure and function, this governance model can be broadly described as a form of joint ownership of the modes of production (e.g., permit, boat, gear, market access) and democratic decision-making for the purposes of achieving certain benefits for its members (ICA 1995). In contrast, individualistic arrangements are characterised by an unequal relationship between an owner of modes of production and one or more fishers who are hired as labor (Pelras 2000). Such arrangements are characterised by centralised decision-making and can range from a small operation, such as owner-operator, to a much larger enterprise, such as patron-client (or patron). While owner-operator generally represents an owner of a single boat who actively fishes (Schadeberg et al. 2021), patrons are largely in charge of a fishing fleet consisting of two or more boats and are rarely active fishers themselves (González-Mon et al. 2019). Instead, they are focused on managing their enterprise in terms of recruitment of fishers and commercialization of catch. Decisions about how to organise fishing activities at the local level are shaped by a range of factors, including state and local regulations, access to capital, histories of cooperation and the characteristics of communities, among others (Basurto, Bennett, et al. 2013; Jentoft and Finstad 2018; McCay et al. 2014). For example, Basurto, Bennett, et al. (2013) found that reducing the costs associated with commercialization was one of the principal reasons that led fishers to collectively organise in some of the remote locations in Baja California Sur, Mexico.

Empirical evidence for the relationships between governance arrangements and sustainable resource use in the field of marine fisheries is sparse but work to date suggests that collective arrangements like cooperatives are more supportive of sustainability than individualistic arrangements, such as patron-client relationships (Bennett and Basurto 2018; Frawley et al. 2019) (Figure 1, hypothesis A), although the opposite situation cannot be fully discounted (Figure 1, hypothesis B). An alternative argument posits that sustainable resource use is a product of the social-ecological context within which these governance arrangements are embedded, rather than the governance arrangement itself, as well as the practices that they employ (Basurto, Bennett, et al. 2013; Leslie et al. 2015) (4, 24) (Figure 1, hypothesis C). In other words, every governance arrangement has a certain set of conditions under which the likelihood of sustainable resource use is greatest given the broader set of conditions within which such arrangements are developed. These conditions, or variables, range from social and economic to ecological and oceanographic (Table 1). This relationship between a positive outcome and a particular set of attributes of the focal system that matter for a specific governance arrangement is considered as social-ecological systems fit (Bodin 2017; Epstein et al. 2015; Young 2002).

From this perspective, a key unanswered question is whether and how these different governance arrangements influence

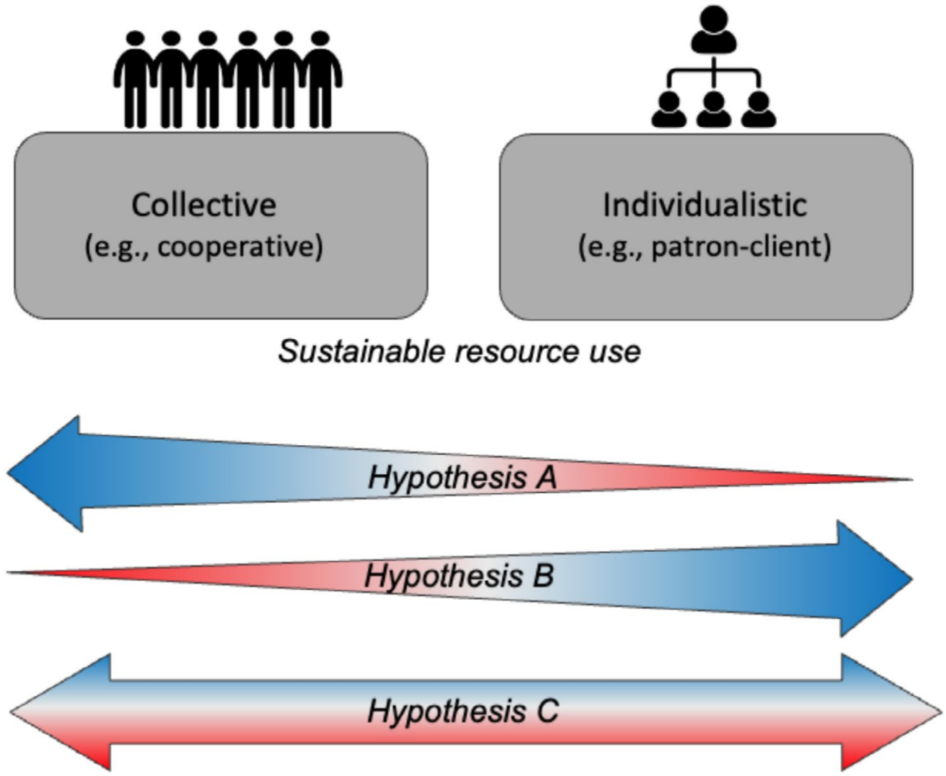

FIGURE 1 | Possible ordering of principal governance arrangements of SSF based on sustainable resource use potential. Greater potential is indicated by the arrows' blue colour and increasing width. Hypothesis A indicates that collective arrangements have the greatest potential in achieving sustainable resource use while hypothesis B indicates the opposite. Hypothesis C, suggests that sustainable resource use is not dependent on a particular governance arrangement but rather on the social-ecological context it inhabits, and as such, is equally possible under all governance arrangement types.

sustainable resource use given the social-ecological conditions in which they are embedded. In this study, we focus on the two most commonly reported governance arrangement types in small-scale fisheries: cooperatives, representing collective arrangements and patrons, representing individualistic arrangements (e.g., Deacon 2012; Ferrol-Schulte et al. 2014; Ovando et al. 2013; Pelras 2000). Drawing on a case study within the Baja California region in Mexico, we examine the hypotheses (Figure 1) through the following research questions:

(Q1) What is the relationship between these two governance arrangements and sustainable resource use?

(Q2) What contextual factors are associated with sustainable resource use?

To examine the relationship between governance arrangements and sustainable resource use, we focus on SSF of benthic and reef-associated taxa from Baja California Sur (BCS), Mexico (Figure 2). This approach was intentional and grounded in theoretical and empirical insights from common-pool resource theory, which suggests that sustainable resource use is more likely to occur for species with limited ranges (Basurto, Gelcich, and Ostrom 2013; Shin et al. 2020). This is due to the ease of monitoring, enforcement, and better matching governance arrangements to resource dynamics (Fleischman et al. 2014; Schlager et al. 1994). We investigate the possible effects of different governance, social and ecological attributes by identifying and parameterizing a series of variables that prior theoretical and empirical work has demonstrated are associated with and/or contribute to the sustainability of locally managed natural resources (Table 1). Our findings contribute the initial building blocks for a theory of governance contextual fit (Epstein et al. 2015; Wilson 2006).

2 | Methods

To address the two research questions, we divided data analysis into seven consecutive stages (Figure S1). The focal point of our analysis was the development of a 'catch sustainability index', which reflected the overall status of landed stocks for individual governance arrangements. As such, it is an indicator of the overall fishery portfolio status. This index enables us to evaluate sustainable resource use at the level of individual governance arrangements by integrating information on the biomass of harvested populations and fishing mortality using standard fisheries science approaches (Hilborn et al. 2020). We define sustainable resource use in the context of fisheries management based on maximum sustainable yield (MSY), where stocks are not overfished or experiencing overfishing (Hilborn et al. 2020). MSY is the greatest catch level that can be continuously harvested (i.e., without depletion) from a targeted stock (Jennings et al. 2001) and is generally considered a useful measure of sustainability within the field of fisheries management. The 'catch sustainability index' served as the outcome variable in a model that examined how the two governance

TABLE 1 | Description of variables used in this study. For additional information on the variables see Tables S8 and S9.

Variable name	Level of observation	Variable role	Variable type	Variable description
Catch sustainability index	Governance arrangement	Response	Categorical, ordinal	Overall status of the fishing portfolio based on weighted average (by volume) of individual taxa's B/B_{msy} and F/F_{msy} values.
Marginalisation	Community	Predictor	Continuous	Level of community's marginalisation. Marginalisation index is composed of 9 indicators that cumulatively measure community's deficiencies based on four broad themes: access to education, housing conditions, monetary income, and population size.
Isolation	Community	Predictor	Continuous	Level of community's geographic isolation. Calculated based on the distance and quality of the roads that connect the community to the closest administrative center (i.e., municipality capital).
Immigration rate	Community	Predictor	Continuous	Proportion of people within a community that were not born in BCS.
Total fishers	Community	Predictor	Continuous	Total number of fishers within a community.
Total fishers with permits	Community	Predictor	Continuous	Proportion of fishers with fishing licences, either as individuals or as cooperative members, to the total number of fishers within a community.
Fishing area size	Community	Predictor	Continuous	Area of the fishing grounds associated with a particular community (in m^2). Size of the fishing area determined using a 35 km radius from all the landing sites associated with a particular community (based on geographic proximity).
Environmental productivity (a)	Community	Predictor	Continuous	Mean concentration of chlorophyll-a (mg per m^3) of the fishing grounds for a given year. A layer of monthly mean values of Chl-a associated with each pixel was overlaid over the fishing polygons and the zonal mean associated with each polygon was calculated.
Environmental variability (b)	Community	Predictor	Continuous	Intra-annual variation in sea surface temperature of the fishing grounds. A layer of variance values of SST associated with each pixel was overlaid over the fishing polygons and the zonal mean associated with each polygon was calculated.
Years fishing	Governance arrangement	Predictor	Continuous	Number of years that the governance arrangement reported landings.
Migration for fishing	Governance arrangement	Predictor	Continuous	Average monthly migrations to fishing sites (in km) weighted by volume.
Number of taxa caught	Governance arrangement	Predictor	Continuous	Number of taxa caught at the family level for a given year.
Trophic level of catch	Governance arrangement	Predictor	Continuous	Average trophic level of yearly catches weighted by volume.
Revenue	Governance arrangement	Predictor	Continuous	Average monthly revenue from fishing for a given year.
Concessions	Governance arrangement	Predictor	Categorical, binary	Presence/Absence of fishing concession(s). This variable only applies to the collectives as none of the patrons reported having concessions.

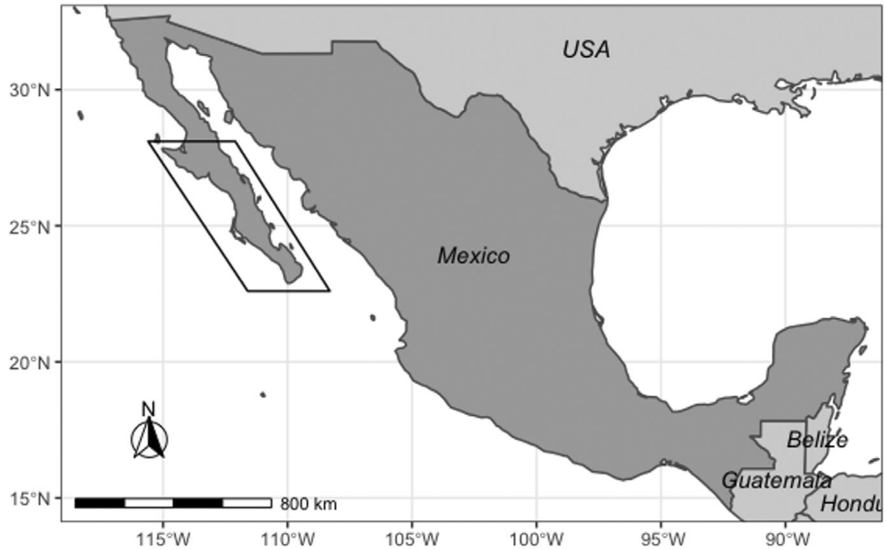

FIGURE 2 | Location of the focal social-ecological systems indicated by the marked polygon.

arrangements (cooperatives and patrons) and related variables (Table 1) influenced sustainability outcomes.

2.1 | Governance and Social-Ecological Setting

The state of Baja California Sur (BCS), Mexico, encompassing the Gulf of California to the east and Pacific Ocean to the west, is one of the world's biodiversity hotspots (Roberts et al. 2002; Spalding et al. 2007) (Figure 2). It is also one of the most productive marine ecosystems in the world (Lluch-Cota et al. 2007). The small-scale fishing sector is vital for coastal communities in this region, particularly in rural areas. The importance of the fishing sector is evident from the national landings where BCS consistently ranks among the top states in terms of the revenue generated by its SSF fleet (CONAPESCA 2015). The most economically important species include abalone, lobster, and certain finfish destined for regional tourism centers or to export markets, such as China and the U.S. (CONAPESCA 2015). The SSF fleet consists of ~3000 fibreglass boats, 6–8 m in length, equipped with outboard motors (Leslie et al. 2015). This fleet represents 99% of BCS' total registered boats (Cortés Ortiz et al. 2006).

Lawful participation in a commercial fishery is based on a fishing permit or concession granted by the CONAPESCA (Comisión Nacional de Acuacultura y Pesca, that is, the National Aquaculture and Fisheries Commission), the federal government entity in charge of fisheries management (Leslie et al. 2015). Fishers may operate either as individuals (e.g., under patron-client or owner-operator arrangements) or as collectives (e.g., cooperatives or businesses) with a valid permit and/or a concession. However, within our study area, only collectives were granted concessions and no individuals received them from the federal government.

In BCS, we focus on two main types of governance arrangements: patron-clients and cooperatives (Figure S2). Patron-clients are entities that hold permits for two or more boats and rely on hired labor for fishing. These fishers do not own permits and are known as independent or 'free' fishers (*pescadores libres*). Cooperatives are legally recognised organisations composed of at least five members (as defined by law) and possess one or more fishing permits, regardless of fleet size. We compiled data on permits and landings using the official databases provided by CONAPESCA.

Patron-clients and cooperatives dominate the BCS small-scale fisheries sector, accounting for 99% of the total landings by volume and value between 2005 and 2016 (Figures S3 and S4). Owner-operators, defined as those holding permits for a single boat, account for the rest. Given their limited impact on landings and stock status, owner-operators were excluded from analysis.

Federal authorities can influence fishers' choice of selecting governance arrangements through laws, executive decrees or subsidy programs; yet authorities' limited capacity at the local level, caused by financial and personnel constraints, responsibilities for fisheries management are effectively devolved to local fishing entities (Cinti et al. 2010). Consequently, as in many other parts of the world, the effectiveness of fisheries governance in BCS largely hinges on these local governance arrangements.

The importance of the SSF for the local economy as well as the diversity of SSF governance arrangements makes this region an ideal setting to test the hypotheses articulated above. Furthermore, through our previous work in the region and ongoing engagement, we have developed detailed, place-based case studies of social-ecological systems dynamics and feedbacks (e.g., Basurto, Bennett, et al. 2013; Frawley et al. 2019; González-Mon et al. 2019; Leslie et al. 2015; Nenadovic and Epstein 2016). The current study is an attempt to increase the spatial extent of our previous work by integrating knowledge generated from these case-specific findings.

2.2 | Study's Analytical Stages

We divided data analysis into seven consecutive stages, where output from the preceding stage served as input for the one that followed (Figure S1). In order to evaluate sustainable resource use, we identified the main fished species in the study region, identified stocks, calculated stock-specific landings, estimated stock-specific biological reference points, calculated the catch sustainability index of each stock and developed, implemented and evaluated an analytical model to test our hypotheses.

2.2.1 | Stage I: Identification of Main Fished Species

We used the official landings data from CONAPESCA. The data include individual trip records, separately listed for each of the species caught, indicating date reported, volume landed, price per kg, total value of catch, duration of a fishing trip in days, number of boats used and landing site, among others, from 2005 through 2016.

We curated the database following two steps. First, since the landings records included only common names and fishing localities for fished resources, we identified the most likely scientific names by relying on the official identification key of the commercial species in the Mexican Pacific (Ramírez-Rodríguez 2013). We identified all fished species to the lowest possible taxonomic level. For example, identification ranged from the species (e.g., 'huachinango' or Pacific red snapper to *Lutjanus peru*) and the genus levels (e.g., 'almeja chocolata' or chocolate clam to *Megapitaria* spp.) to the level of family (e.g., 'pargo' or snapper to Lutjanidae) and even higher ones (e.g., 'tiburon' or shark to *Selachimorpha*). We excluded unidentified species (e.g., 'otras' or other and 'no asignada' or not assigned).

Second, we identified any landing record at the level of each individual governance arrangement for which landed volume, unit price, number of boats or fishing days were greater than three standard deviations of the respective mean of all records for a given taxa and year. We replaced identified outliers with the corresponding mean value (volume/boats/days). If the mean for a particular taxon could not be calculated this way, we expanded the year range to include the two adjacent years (year ±1), thus increasing the possibility of landings for a particular taxon. In this way, we tried to minimise the underreporting of catch as much as possible given that underreporting contributes to biased estimates of biological reference points calculated in stage IV. We removed any remaining outliers from the analysis. Lastly, we adjusted unit prices for inflation using the consumer price index to present them as 2016 MXN equivalents. In summary, in the period between 2005 and 2016, fishers in the region landed 137 taxa, a total volume of 603,838 metric tons and a total revenue of MXN 9902 million (USD 530 million).

Upon completion of the taxonomic identification and outlier cleaning, we extracted the taxa, which, combined, contributed to 90% of the total landings in both volume and revenue within the 2005–2016 period (Table S1). This resulted in further evaluation of 38 taxa with a total volume of 566,768 metric tons and a total revenue of MXN 9227 million (USD 494 million). For the taxa that we could not identify to the species level, we reviewed literature to check if any of the specific species within a given taxon was identified as a principal fishery target and in that way increase the taxonomic resolution (Table S2). Through this process, we were able to narrow down the number of focal species for some taxa. For example, although the taxon *Callinectes spp* consists of three commercially important species (*C. arcuatus*, *C. bellicosus* and *C. toxotes*) (OOF 2010), recent studies indicate that approx. 98% of the catch consists of *C. bellicosus* (Cisneros-Mata et al. 2019).

We used FishBase (www.fishbase.se) and SeaLifeBase (www.sealifebase.ca) to obtain information on trophic level, vulnerability, reproduction rate and mobility for all selected taxa (Table S3). In cases where information on specific taxa was not available, we used average values from related taxa within the same family that occur within the focal region. We also reviewed official fisheries documents and publications to identify all fishery management tools in place for each of the focal taxa.

The focus of our study was on taxa that have a limited adult range of movement (i.e., resource mobility). This approach is premised on the idea that a low level of resource mobility is one of the critical conditions that promote sustainable resource use (Fleischman et al. 2014; Schlager et al. 1994). From this perspective, taxa with low levels of mobility, such as lobster and abalone, are much more likely to be subject to spatial management, such as area-based permits or territorial use rights (TURF) and exclusively accessed by the SSF sector compared to highly mobile taxa such as mackerel or squid. Through this process, we excluded all pelagic taxa ($N=9$) (Table S1, column 'Taxon mobility'). The most dominant taxa in this group was *Dosidicus gigas*, which represented 35.5% in volume and 6.9% in value of the total landings.

Furthermore, we excluded any additional taxa that were not exclusively fished by the small-scale fishing sector ($N=2$) (Table S1, column 'Fishery type'), since such catch records were incomplete and could not be solely attributed to SSF. Lastly, we excluded an additional five taxa from the assessment for which the larval model (see Stage II) could not be adequately specified, mainly because those taxa were a compilation of multiple species that substantially differed in their life histories (Table S1, column 'Inclusion in analysis'). We proceeded with the assessment of 22 taxa, which represented 43.8% in volume and 69.9% in value of the total landings.

2.2.2 | Stage II: Identification of Distinct Biological Stocks

The goal of this step was to identify biological stocks associated with the assessed taxa ($N=22$). For each of them, we incorporated information about ecological connectivity (the demographic linking of local populations mediated by the dispersal of individuals as larvae, juveniles or adults) in a spatially explicit metapopulation context that identified the role of each population as a larval source or sink. We focused on larval dispersal driven by ocean currents because it is the main mechanism acting at regional geographic scales (Palumbi 2004). For each taxon, we reviewed published literature to define the spawning season and pelagic larval duration (PLD), defined as the time spent as planktonic larvae (Table S4). We then ran three-dimensional oceanographic models assuming passive larvae and calculated connectivity matrices describing the probability of larval export and import between local populations. We used the baroclinic numerical model HAMSOM developed by Backhaus (1985) and adapted to the Gulf of California (Marinone 2003), which has been validated based on empirical genetic data for multiple species of fish and invertebrates (Munguia-Vega et al. 2018). For the Pacific coast of the Baja California Peninsula, we implemented the AGRIF version of the Regional Ocean Modelling System (ROMS) (Parés-Sierra et al. 2018). Each model used spatial units of analysis—59 polygons for the Gulf of California and 119 for the Pacific (Figure S5)—defined from the coast and up to a 200 m isobath line that includes the main habitats where small-scale fishing takes place around BCS and included the adjacent states of Baja California, Sonora and the northern part of Sinaloa to account for boundary effects.

The connectivity matrices were analysed as spatial networks that were directed (i.e., explicitly considering directionality, from the larval source towards the larval sink) and weighted by the probabilities observed with the oceanographic model. Here, nodes represented the spatial units of analysis connected by links that represented the predicted probabilities of larval export/import to/from each site, based on the combination of spawning season and PLD for each taxon (Figures S6 and S7). Then we used a heuristic method based on modularity optimization (Blondel et al. 2008) to describe the structure of each larval dispersal network and identify clusters that show strong connections within nodes in the same cluster but sparser links between nodes from distinct clusters. Through this process we identified and delineated 414 distinct clusters for the 22 taxa (Table S5). We used the identified clusters as proxies for distinct biological stocks to generate the necessary inputs for calculating relative biomass and relative exploitation rate (stage IV), which, combined, gives us a measure of sustainable resource use (stage V).

2.2.3 | Stage III: Calculation of Stock-Specific Landings

Using the CONAPESCA landings database, we extracted all unique landing sites for the states of Baja California Sur (BCS), Baja California (BCN), Sonora (SON) and Sinaloa (SIN). Although our focus was on BCS, we included the additional states as some stocks were shared among the different states. We then georeferenced each site by matching it to the published database of fishing site locations for the region (Ramírez-Rodríguez et al. 2006). If two or more landing sites occurred within a 10 km radius, we merged them into a randomly selected site within the cluster to substantially reduce computational requirements. We excluded all the landing sites associated with inland fisheries ($N=37$). This generated 290 unique landing sites across the four states.

Based on data collected through surveys and interviews with BCS fishing sector participants over the last 10 years (e.g., Basurto, Bennett, et al. 2013; Finkbeiner 2015; Frawley et al. 2019; González-Mon et al. 2019; Nenadovic and Epstein 2016), we found that the average distance of the longest recorded fishing trips in our dataset is 35 km. We used this distance to delineate the fishing ground area (as a radius) for each landing site (Figure S8). We then overlaid fishing areas on top of 414 stock polygons identified in the previous stage and calculated the percentage overlap of every fishing area within each stock polygon. We used the percentage overlap to calculate the amount of landings associated with a particular stock at each landing site. For example, if a landing site overlaps 70% with stock A and 30% with stock B of a taxa X, the total landings of taxa X at that landing site were divided among the two stocks based on their respective percentage overlaps: 70% of the landings were assigned to stock A and 30% to stock B. Since the stock delineation approach (stage II) did not consider each taxon's geographic range, we used landings data to account for it. We excluded all stocks for which no landings were reported ($N=241$), which left us with 173 stocks for further evaluation (Table S5). In the last step, we calculated total landings (by volume) per stock per year, which we adjusted for illegal, unregulated and unreported (IUU) catch based on taxa-specific correction factors for Mexican fisheries (Cisneros-Montemayor et al. 2013).

2.2.4 | Stage IV: Estimation of Stock-Specific Biological Reference Points

We used a data-poor stock assessment model (catch-based maximum sustainable yield or CMSY) to estimate biological reference points, including biomass (B), exploitation rate (F), the maximum sustainable yield (MSY), relative biomass size (B/B_{msy}) and relative exploitation rate (F/F_{msy}). We primarily focused on the relative biomass (B/B_{msy}) and relative exploitation rate (F/F_{msy}) parameters, given that they are commonly used indicators for characterisation of the status of diverse fisheries around the world (Costello et al. 2008; Hilborn et al. 2020). These two fishery indicators are often combined in a single plot (i.e., Kobe plot) to differentiate among four potential conditions of a fishery:

1. Poor: fishery is both overfished ($B/B_{msy} < 1$) and experiencing overfishing ($F/F_{msy} > 1$),
2. Declining: fishery is experiencing overfishing ($F/F_{msy} > 1$) but is not overfished ($B/B_{msy} > 1$),
3. Recovering: fishery is overfished ($B/B_{msy} < 1$) but is not experiencing overfishing ($F/F_{msy} < 1$), and
4. Healthy: fishery is neither overfished ($B/B_{msy} > 1$) nor experiencing overfishing ($F/F_{msy} < 1$).

CMSY is a Monte Carlo MSY estimation model that was developed by Froese and colleagues (Froese et al. 2017). Giron-Nava and colleagues (Giron-Nava et al. 2019) subsequently modified this model for the Mexican SSF context. To conduct the estimation process, the CMSY model requires a catch time series with a suggested minimum of 10 years, indicators of species' resilience (derived from FishBase and SeaLifeBase) and biomass priors from the initiation of the time series (B_0) (Froese et al. 2017).

To model the initiation year (2005) and to assign informed biomass priors (B_0), we relied on official evaluations of relevant taxa (INAPESCA 2004) (Table S6). Given that the evaluations were qualitative in nature, we used the following conversion metrics: for taxa whose populations were considered already overexploited, we assigned B_0 a range of possible values from 0.01 to 0.4; for populations that were considered to be sustainably exploited, we assigned B_0 a range of possible values from 0.2 to 0.6; and for populations that were considered to have potential for increased fishing pressure, we assigned B_0 a range of possible values from 0.5–0.9 (Froese et al. 2017).

We estimated biological reference points for 173 stocks (Figure S9); however, for 36 of them the results were inconclusive, mainly because these stocks did not have a minimum of 10 years' worth of landings data. We therefore removed those stocks from further analysis, resulting in the exclusion of two additional taxa (*Chione spp* and *Panopea globosa*). The model estimated biological reference points for each stock and for every year that the stock was caught. Overall, we successfully evaluated 123 spatially explicit stocks from 20 taxa (Table S7). Lastly, we excluded stocks that were not fished in BCS ($N = 25$). The remaining 98 stocks comprised 43% of the full BCS landings dataset (593,638 metric tons) in terms of volume and 56% (9819 millions in 2MXN (2016)) in terms of value.

It is important to recognise that the CMSY approach relies exclusively on fisheries-dependent data and as such is prone to produce biased estimates, particularly when applied too broadly and without regard for the particular social-ecological context or in the case of fisheries with a limited catch series (Free et al. 2020; Ovando et al. 2022). However, given that we excluded stocks with sparse catch data (< 10 years of landings) and those for which preliminary analyses indicated we did not have reliable priors of population parameters (resilience, *r* or *k*—see Table 1 in (Froese et al. 2017) for definitions), we believe that the resulting assessments provided reliable estimates of stock status.

2.2.5 | Stage V: Calculation of a 'Catch Sustainability Index'

To evaluate sustainable resource use at the level of individual governance arrangement, we developed the 'catch sustainability index,' which is based on the estimations of B/B_{msy} and F/F_{msy} obtained from the previous stage. This index represents the outcome variable in our final model (stage VI).

The first step in the calculation of a 'catch sustainability index' is to compute a 'catch weighted average' (CWA), based on individual stock landings by volume, of B/B_{msy} and F/F_{msy}, separately, for all stocks that a particular governance arrangement fished within a given year:

$$\text{Catch weighted average (CWA)} = \frac{\sum_{i=1}^{n} w_i X_i}{\sum_{i=1}^{n} w_i}$$

where X_i indicates the status of either B/B_{msy} or F/F_{msy} for an individual stock i, w_i equals catch (kg) of stock i; n represents the number of stocks in a fishing portfolio.

In the second step, we converted every computed CWA into a binary variable, based on the threshold value of 1: not overfished or overfished categories for B/B_{msy}; and experiencing overfishing or not experiencing overfishing categories for F/F_{msy}. In the last step, we combined the two binary indicators to produce an ordinal 4-point 'catch sustainability index', which resembles general categorizations of stock conditions (e.g., Hilborn et al. 2020):

- overfished and experiencing overfishing = poor (coded as 1)
- not overfished and experiencing overfishing = declining (coded as 2)
- overfished and not experiencing overfishing = recovering (coded as 3)
- not overfished and not experiencing overfishing = healthy (coded as 4)

In this way, the 'catch sustainability index' is an indicator of the overall fishery portfolio status of a particular governance arrangement, based on both B/B_{msy} and F/F_{msy} for a given year, and as such serves as a proxy of sustainable resource use. To help determine the robustness of our main model, we also calculated a 3-point 'catch sustainability index' in which we merged 'declining' and 'recovering' categories into an 'intermediate' one (stage VII).

2.2.6 | Stage VI: Analytical Model for Assessing Catch Sustainability

We used the catch sustainability index as the dependent variable in our final model. We conducted a variable selection process for the final model based on our previous work in the region that focused on studying the small-scale fishing sector in relation to various governance, economic, social and ecological conditions (e.g., Basurto, Bennett, et al. 2013; Leslie et al. 2015; McCay et al. 2014; Nenadovic and Epstein 2016) (Table 1, Table S8). The unit of observation varies across the selected variables: some were measured at the individual governance arrangement level and others at the community level within which a governance arrangement resides (Table 1, Table S8). The full dataset contained 482 individual governance arrangements nested within 52 communities.

We conducted all analyses in RStudio (version 1.4.1717). To compare the difference of 'catch sustainability index' values for the two governance arrangement types as well as to evaluate the association between predictor variables and the 'catch sustainability index' we developed a multilevel model, given that we had repeated measures at the level of each governance arrangement (catch over time) and nesting of individual governance arrangements within their respective residence communities. To address Q1 we implemented a single multilevel model, with governance arrangement type as a predictor variable. To address Q2 we conducted two multilevel models, separately for each of the two governance arrangement types—cooperative and patron-client. We calculated a correlation matrix (stats package, version 4.1.1) to identify and exclude all predictor variables that exhibited strong or very strong levels of correlation (±0.70) prior to the final analysis (Table S8) (Schober et al. 2018). The final set of variables was identical for both models, with the exception of 'concessions,' which applied only to cooperatives, as none of the patrons were granted concessions in our area of study.

We log-transformed a number of the predictor variables to better approximate normality and, furthermore, standardised all predictor variables to have a mean of zero and a standard deviation of one (i.e., z-score) (Table S9). Unit of observation in our dataset was a governance arrangement, with the 'catch sustainability index' reported for each year that it fished.

We implemented a cumulative ordered probit model using *brms* (version 2.16.3), in which we weighted each observation based on the average fishing effort. We calculated fishing effort from days at sea and the number of boats associated with each fishing trip. We specified the model as follows:

$$\Pr\left(Y = k \mid \left\{\tau_k\right\}, \mu_{ij}\right) = \Phi\left(\tau_k - \mu_{ij}\right) - \Phi\left(\tau_{k-1} - \mu_{ij}\right)$$

$$\mu_{ij} = \sum_1^n \beta_n x_n + u_i + v_j$$

$$u_i \sim N\left(0, \sigma_u\right)$$

$$v_j \sim N\left(0, \sigma_v\right)$$

$$\tau_1 \sim N(-0.674, 1)$$

$$\tau_2 \sim N(0, 1)$$

$$\tau_3 \sim N(0.674, 1)$$

$$\beta_n \sim N(0, 1)$$

$$\sigma_u \sim \text{Exponential}(1)$$

$$\sigma_v \sim \text{Exponential}(1)$$

where Y represents the observed ordinal variable and Φ the cumulative standard normal distribution function of a latent continuous variable $\underset{-}{Y}$ from which Y is derived (60, 61). τ_k indicates individual thresholds, k, which divide $\underset{-}{Y}$ and the corresponding Φ into $k+1$ ordered categories of Y (in our case 4: poor < declining < recovering < healthy). We calculated the three thresholds, τ_1, τ_2, and τ_3, assuming that our 4-point ordinal variable is evenly distributed, thus resulting in each having the same probability mass [87] (Figure S9). μ_{ij} represents additive effect of the full set of predictor variables, x_n (Table 1) and their coefficients (β_n) within a multilevel model while u_i and v_j indicate random intercepts for the fishing entities (i.e., repeated observations across years) and the communities (i.e., fishing entities nested within communities), respectively. Lastly, σ_u and σ_v indicate level-1 and level-2 standard deviations.

Since all of our parameters were standardised (i.e., scale-free) to ensure a smoother estimation process, we assigned weakly informative priors to all of them, relying on the conventions established in the literature (Gelman 2006). We ran two separate models, one for cooperatives and one for patrons, using three Markov chains with 2000 warm-up and 4000 iterations, which satisfied the minimum number specified by the Raftery-Lewis test to achieve accuracy of ±0.005 and probability of 0.95.

2.2.7 | Stage VII: Evaluation of the Model for Assessing Catch Sustainability

Prior to inspecting the results from either model, we conducted a range of model evaluation and posterior probability checks to assess the overall model fits (Depaoli and Van De Schoot 2017). First, we assessed Markov chain convergence using a graphical representation of time-series plots of the chains (Figure S11), Gelman-Rubin, and Geweke convergence diagnostics (Table S10). Visual inspection suggested chains were well mixed, which was indicated by a random scatter around a mean value (Figure S11). Gelman-Rubin test evaluates convergence among multiple Markov chains by comparing the estimated between-chains and within-chain variances for each model parameter, known as the potential scale reduction factor (PSRF or $\widehat{R}$) (Gelman and Rubin 1992). For chains that are at equilibrium, the PSRF will be 1 or very close to 1. Results for both models suggested that the chains converged (Table S10). Geweke test compares the posterior mean of the first 10% of a Markov chain to the last 50% and calculates z-score for each pair (Geweke 1992). Cutoff of |1.96| is then used to assess convergence. We found five parameters from the cooperatives model and four from the patrons model out of bounds, albeit for only one of the three chains (Table S10). To ensure that local convergence was not an issue, we ran the two models with the number of iterations doubled (8000). Both visual inspection and Gelman-Rubin test indicated that local convergence was not an issue in either of the two models (results not shown but available upon request).

Second, we visually evaluated posterior marginal distributions of the model parameters and concluded they had a sufficient amount of information (Figure S12) (Depaoli and Van De Schoot 2017). Third, we visually inspected autocorrelation plots of the model parameters to evaluate the amount of autocorrelation present in the chains. We could not identify any substantial issues in the autocorrelation plots and proceeded without thinning the sample (Figure S13). Fourth, we compared the kernel density estimate of the observed outcome to the density estimates for 50 simulated outcomes drawn from the posterior predictive distribution to determine the level of overlap between the two distributions. We found that both of our models slightly underestimate one of the two intermediate categories (i.e., recovering) of our ordinal response variable (Figure S14). Fifth, we compared the mean from the observed dataset and a histogram of the mean computed from 4000 draws from the posterior predictive distribution for both of the models and concluded that it was adequately captured by the posterior predictive distributions (Figure S15). As a sixth step, we checked for multicollinearity in the two models by calculating the Variance Inflation Factor (VIF), which measures the amount of variance of a predictor variable that is influenced by its correlation with the other predictor variables. No variable had a VIF higher than 10 and therefore, we did not remove any additional variables from the models (results not shown but available upon request) (Hair et al. 2010).

Lastly, we performed sensitivity analysis to evaluate robustness of the posterior parameter estimates from our two models to four alternative model specifications. In alternative model (AM) 1, we substituted the residual variance prior Exponential(1) with half-Cauchy (0, 1) (Gelman 2006). In AM2, we substituted weakly informative priors with uninformative ones for β_n and σ. In AM3, we used the two default models but with the doubled number of iterations (8000). In AM4, we implemented the alternative specification of the response variable by reducing the 'catch sustainability index' from four to three levels. We achieved this by reclassifying 'declining' and 'recovering' conditions as 'mixed'. This reclassification resulted in new intercept priors $\tau_1 \sim N(-0.43, 1)$ and $\tau_2 \sim N(0.43, 1)$, each with the same probability mass under the assumption that our 3-point ordinal variable is evenly distributed (Kurz 2021). Overall, the results we obtained from our baseline model seem robust to alternative model specifications we tested (Table S11; Figures S16 and S17). None of the predictor variables changed directionality across the four alternative models. However, three predictor variables changed the significance effect. For the cooperatives, variable 'migration for fishing' was significant in the default and AM1 models and nonsignificant in the other three. Variable 'number of taxa caught' was non-significant in the default and AM1-3 models and significant in the AM4. For the patrons, variable 'environmental variability (b)' was significant in the default and AM1-3 models and nonsignificant in the AM4.

3 | Results

To address Q1, we conducted a Bayesian multilevel model to compare the sustainability of resource use as assessed by the catch sustainability index between the two governance arrangements, cooperatives and patron-client relationships, controlling for other relevant variables (Table 1). We found no substantial difference in the catch sustainability index across the two governance arrangements (posterior estimate [PE] −0.12, with a 95% credible interval (CI) of −0.31 to 0.07) (Figure 3, Figure S18, Table S12).

We addressed Q2 using multilevel models to identify governance, social and ecological variables (Table 1) that are associated with sustainable resource use. The multilevel modelling was conducted separately for the two governance arrangement types as detailed in Stages VI and VII of the Methods.

Modelling results revealed important commonalities and differences among the two governance types (Figures 4 and 5, Tables S13 and S14). Variables with the greatest effect on both governance arrangement types included 'trophic level of catch' and 'revenue'. A one unit increase in the trophic level (which was, in turn, highly correlated with measures of vulnerability and reproductive output) was associated with an increase in 'catch sustainability index' of 1.11 (95% CI: 1.08–1.15) for cooperatives and 1.08 (95% CI: 1.02–1.14) for patrons. In contrast, a one unit increase in revenue was associated with a decrease in 'catch sustainability index' of 0.57 (95% CI: −0.60 to −0.53) for cooperatives and 0.46 (95% CI: −0.52 to −0.40) for patrons. 'Migration for fishing' was another variable that had the same directionality of effect for both governance types, albeit a weak one. A one unit increase in 'migration for fishing' was associated with a decrease in 'catch sustainability index' of 0.04 (95% CI: −0.07 to −0.01) and 0.09 (95% CI: −0.14 to −0.03) for cooperatives and patrons respectively.

The remaining five variables had differential effects on catch outcomes between the two governance arrangements (Figures 4 and 5). Three variables, 'fishing concessions', 'years of fishing' and 'environmental variability', were exclusively and significantly related to the 'catch sustainability index' of cooperatives, while 'marginalization' and 'environmental productivity' were exclusively related to patrons. Furthermore, 'marginalisation' was the only variable that had a negative association with the 'catch sustainability index'. In the case of variables related to cooperatives, the largest effect size was of 'fishing concessions' (PE 1.11; 95% CI: 0.64–1.58), followed by 'environmental variability' (PE 0.40; 95% CI: 0.37–0.43), and then by 'years of fishing' (PE 0.18; 95% CI: 0.11–0.26). In the case of variables related to patrons, 'marginalisation' had a larger absolute effect size (PE −0.55; 95% CI: −0.91 to −0.19) than 'environmental productivity' (PE 0.12; 95% CI: 0.03–0.22).

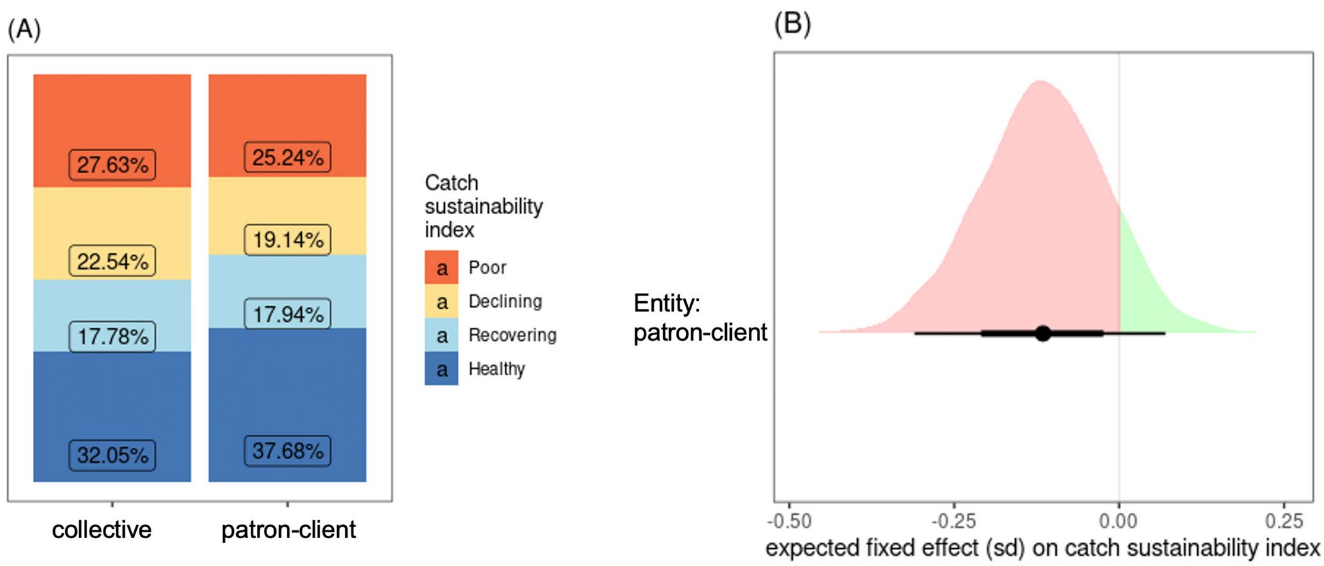

FIGURE 3 | Comparison of the two principal governance arrangements based on the catch sustainability index. (A) Percentage of stocks found in each of the four categories. (B) Model result comparing patron-client to collective governance arrangement. Colours differentiate negative (red) from positive (green) relationships to the catch sustainability index. The dot represents the posterior median, thick lines 50% and thin lines 95% credible intervals. Full model results are available in Supporting Information (Figure S18, Table S12).

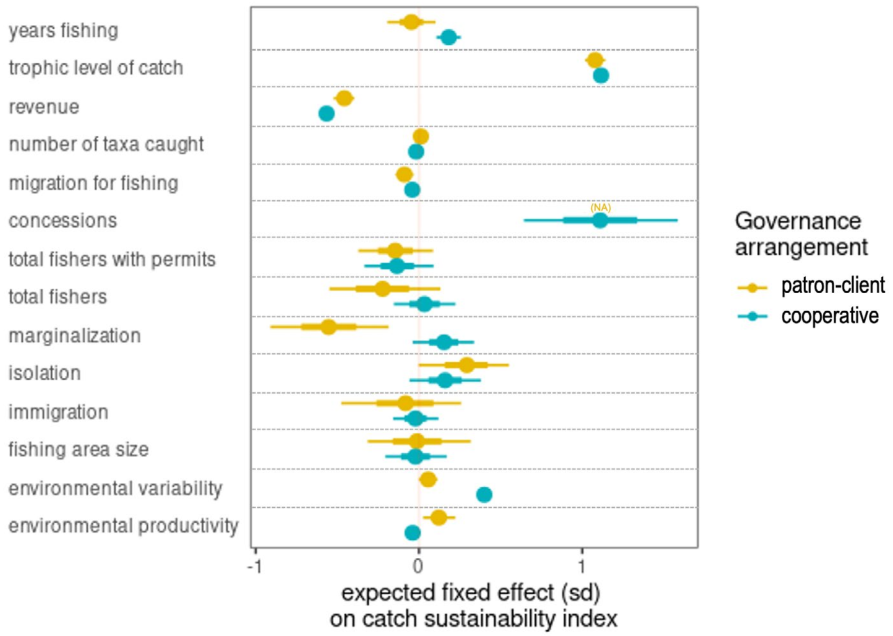

FIGURE 4 | Estimates for the models' fixed effects of 'catch sustainability index' for the cooperative and patron-client governance arrangements. The dots represent the posterior median, thick lines 50% and thin lines 95% credible intervals. Horizontal dotted lines separate variables for improved readability. Full model results can be found in Tables S13 and 14 and variable descriptions in Table 1. All variables were tested in both models, except for the 'concessions' which was only included in the model for the cooperative governance arrangement (see Methods for details).

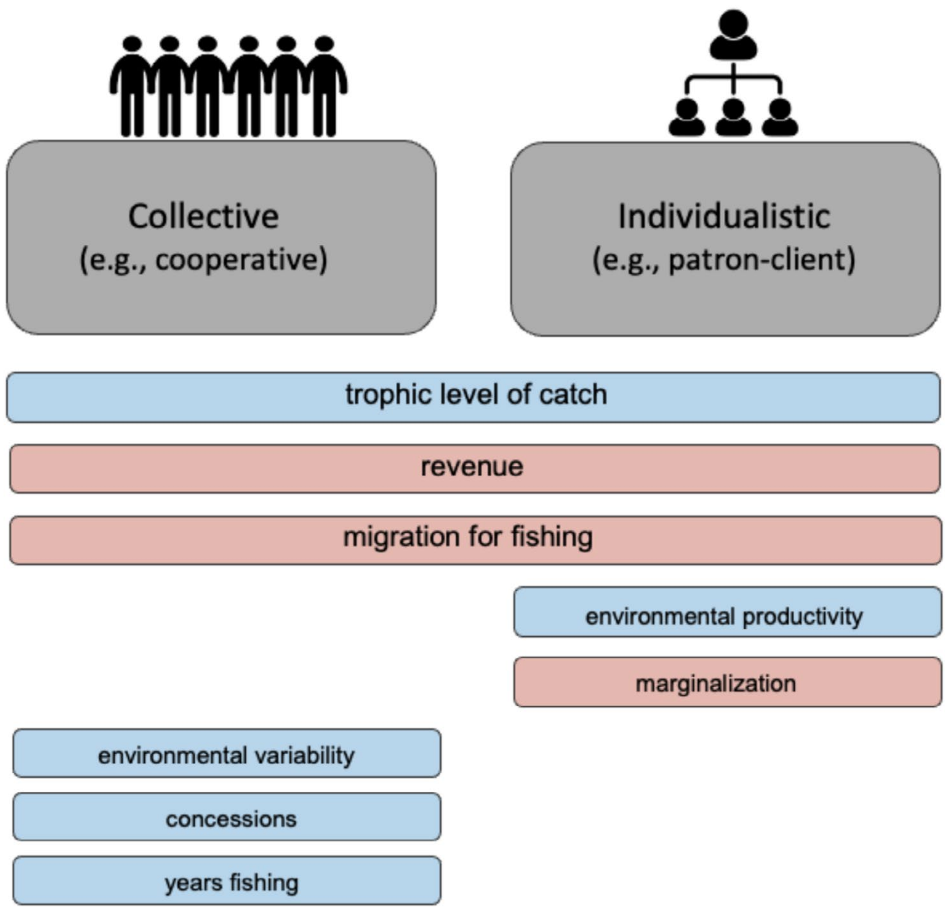

FIGURE 5 | Summary of the significant findings extracted from Figure 4. Variables shaded in blue indicate a positive association with the 'catch sustainability index' of a governance arrangement under which they are positioned while variables shaded in red indicate a negative one. Relative effect sizes of individual variables can be seen in Figure 4, Tables S13 and S14.

Plots of marginal effects illustrate how the probabilities of an arrangement's catch being poor, declining, recovering or healthy, x' vary with different values of each significant predictor variable (Figure 6A,B). 'Trophic level of catch', 'revenue' and 'migration for fishing' (Figure 5) had a consistent effect across both governance arrangements and showed similar patterns across the four categories of the index. For example, the probability of 'catch sustainability index' being in the 'poor' category decreases from more than 75% for the stocks in the lowest trophic level to < 5% for the stocks in the highest trophic level whereas the probability of being in a 'healthy' category increases from < 3% for the stocks in the lowest trophic level to more than 60% for the stocks in the highest trophic level (Figure 6A). A similar trend is observed under the patron governance arrangement (Figure 6B).

Among the variables specific to the cooperative governance arrangement (Figure 6A), an increase in 'years fishing' reduces the probability of the 'catch sustainability index' being in the 'poor' category from 60% to about 25%, while increasing the probability of being in the 'healthy' category from 2% to some 10%. Similarly, the presence of a 'concession' reduces the probability of the 'catch sustainability index' being in the 'poor' category from approximately 30%–5% and raises the probability of being in the 'healthy' category from 10% to 42%.

As for the patron-client arrangements, key variables include 'marginalisation' and 'environmental productivity' (Figure 6B). As marginalisation increases, the probability of the 'catch sustainability index' being in the 'poor' category increases substantially, rising to over 75% at the highest marginalisation levels, while the probability of being in the 'healthy' category declines sharply, approaching zero at intermediate to high marginalisation levels. In the case of 'environmental productivity', the increase in this variable is associated with the decrease in the probability of the 'catch sustainability index' being in the 'poor' category from some 35% to 20%, while the probability of being in the 'healthy' category increases from < 10% to 20%. Some of the estimates associated with the 'poor' category, especially in the case of 'migration for fishing' and 'environmental productivity', exhibit a high degree of uncertainty, as indicated by wider credible intervals.

4 | Discussion

Our findings suggest that for the SSF of the BCS region there is no single best governance arrangement in terms of achieving sustainable resource use as measured by the 'catch sustainability index' (Figure 3). This suggests that neither of the two most dominant arrangements within this region is, in and of itself, more prone to sustainable or unsustainable resource use of benthic and reef-associated species. Instead, each of the two dominant governance arrangements that we evaluated, cooperatives and patrons, is associated with sustainable resource use under certain social and environmental conditions (Figures 4 and 5). Overall, we find that the performance of cooperatives, in comparison with patron arrangements, is superior in contexts characterised by greater environmental variability and the presence of fishing concessions. Patrons, conversely, perform better in contexts with higher environmental productivity and lower community marginalisation. Note that our analyses identify associations among variables, not causal relationships. Although the latter is an important area of future work, our findings allow us to start outlining the initial building blocks for a theory of governance contextual fit in social-ecological fisheries systems.

Our findings support the hypothesis of social-ecological systems fit (Epstein et al. 2015; Young 2002) (hypothesis C, Figure 1), which refers to a context where positive outcomes emerge from the coherence of scale and/or dynamics among ecological, social, and governance processes (Wilson 2006). This is consistent with other environmental governance results related to property regimes (Cole and Ostrom 2012; Ojanen et al. 2017) and management arrangements (Acheson 2006). For example, Cole and Ostrom (2012) summarised the effects of different property rights regimes (government, private and communal) on forest resources and found that none were associated with a higher probability of achieving sustainable resource use. Instead, it was the fit between the property rights regimes and local social and ecological conditions that mattered for achieving a positive outcome (i.e., sustainable timber harvest) for forest management. However, our findings also revealed that the social-ecological systems fit is not characterised by entirely idiosyncratic variable configurations, given that some variables had the same effect across the two dominant governance arrangements in this study.

4.1 | Generalised Variables for a Theory of Governance Fit for Social-Ecological Fishing Systems

The presence of variables that had the same effect on sustainable resource use across the two governance arrangements (Figure 5), such as 'trophic level of catch', 'revenue' and 'migration for fishing', suggests the existence of underlying attributes of social-ecological systems that need to be considered and addressed regardless of the governance arrangements in place. This is particularly relevant in terms of the 'trophic level of catch' and 'revenue'. Although the positive association between trophic level and sustainable resource use contradicts the general notion of 'fishing down food webs' (Pauly et al. 1998), it is consistent with a 'based on availability' scenario (Branch et al. 2010). Under this scenario, fisheries first target the most accessible taxa with the highest economic returns before proceeding to harvest the other ones. According to Sethi et al. (2010), the taxa with the highest economic returns in terms of value and revenue are principally benthic and, as such, are targeted before the other less economically valuable alternatives. Furthermore, benthic taxa are more susceptible to environmental stressors such as heat waves and hypoxia, factors that can additionally impact their status (Carrier-Belleau et al. 2021; Watson and Neo 2021). The SSF of BCS share many of these factors considering that lower trophic level taxa are primarily benthic, of higher value and affected by climate change (Smith et al. 2022). They are also found in poor condition (Figure S19). However, as previously stated, it is important to recognise that our analysis excluded pelagic species, a group that includes many high trophic level taxa (e.g., sharks and rays), many of which are overexploited (INAPESCA 2010; Saldaña-Ruiz et al. 2019).

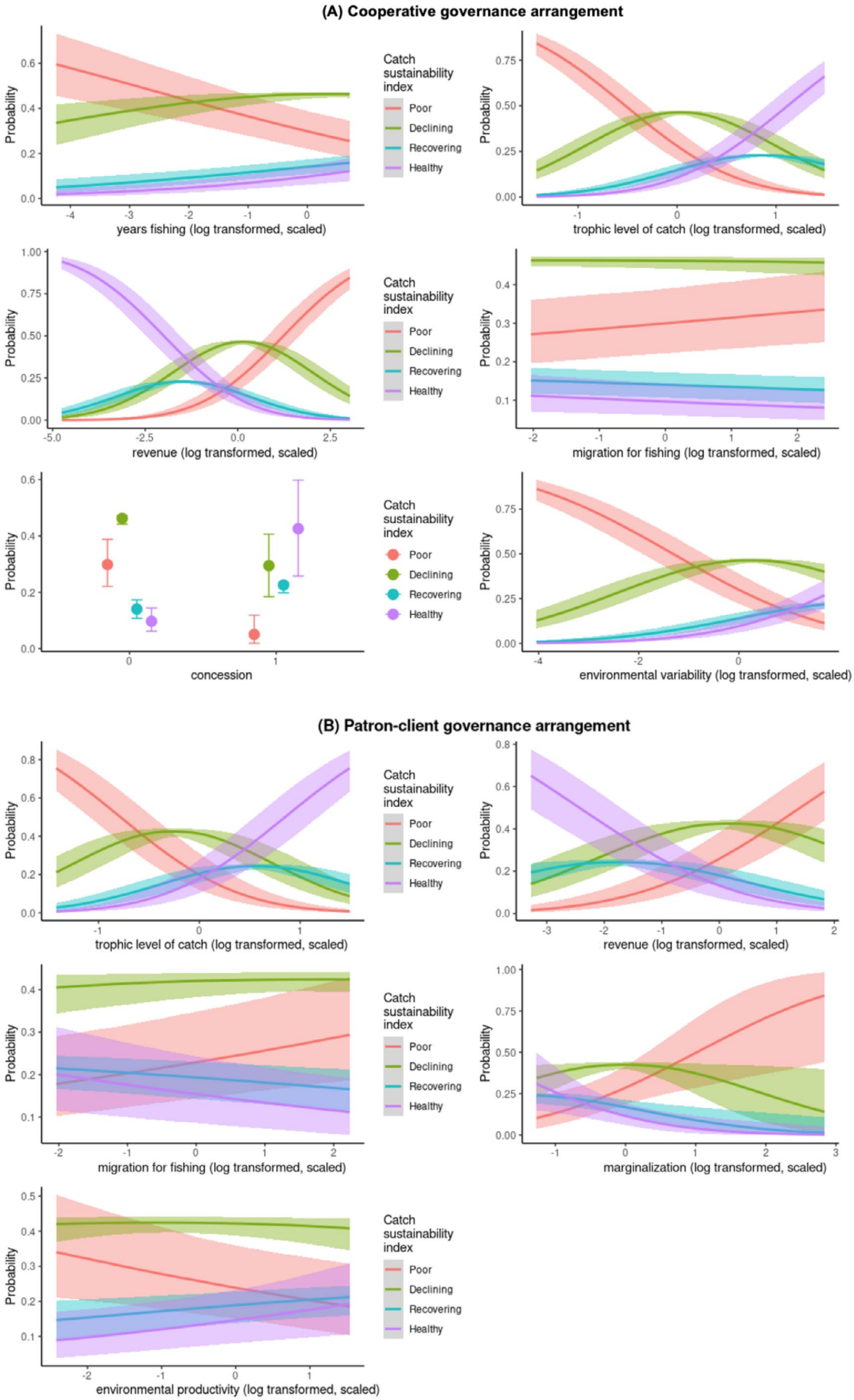

FIGURE 6 | Legend on next page.

FIGURE 6 | Marginal effects for the significant variables for (A) cooperative and (B) patron-client governance arrangements models based on Figure 4. *Y*-axis indicates the probability of each of the four categories of the catch sustainability index for a particular value of the predictor variable (*x*-axis). Shaded regions represent 95% credible intervals.

'Migration for fishing' is another variable with a consistent effect across the two governance arrangements. Although 'migration for fishing' can be a form of adaptive capacity aimed at ensuring stable income in the face of uncertainty (Gonzalez-Mon et al. 2021; Sievanen 2014), its negative association with the 'catch sustainability index' can be interpreted as an indicator of local stock depletion, prompting fishers to expand their operations beyond their immediate fishing grounds. Such patterns of sequential spatio-temporal overexploitation, or 'roving banditry' (Berkes et al. 2006), have been demonstrated for a number of high-value benthic taxa at the global level (Anderson et al. 2011; Berkes et al. 2006), as well as in more localised contexts (Eriksson et al. 2012; Fahrenbruch 2018). In BCS, fishing-related migration is considered an adaptive strategy aimed at minimising uncertainty associated with the effects of climate change on local fished stocks. However, this practice is mainly associated with targeting high-value species that move seasonally along the coast (Sievanen 2014). Fishers intercept these species by using temporary fishing camps established principally for the purposes of such seasonal trips. If migration remains unconstrained, it can result in broader ecological and social problems by increasing pressure on marine resources and challenging local governance arrangements (Cudney-Bueno and Basurto 2009; Eriksson et al. 2012).

4.2 | Contextual Variables That Enable Governance Social-Ecological Fit

Aside from the variables that produced consistent effects across the two governance arrangements, our findings emphasise the contextual nature of social-ecological relationships related to a particular form of governance. Variables 'years of fishing', 'fishing concessions' and 'environmental variability' were exclusively associated with the cooperative governance arrangement and were positively associated with the 'catch sustainability index' (Figure 5). The significance of age of a cooperative might suggest that resource conservation is more likely to occur in older cooperatives. A number of case studies from the Northern Pacific region of Baja California indicated that some of the oldest cooperatives actively reduce fishing effort in the presence of environmental uncertainty (Aburto-Oropeza et al. 2016; Finkbeiner et al. 2018). The positive relationship between 'environmental variability' and the 'catch sustainability index' in our study might also be a result of a precautionary approach to fisheries management, through which cooperative governance arrangements are willing to forego short-term gains for long-term income stability when faced with high environmental uncertainty. Lastly, the presence of secure property rights has been shown to promote sustainable resource use across a diverse range of ecosystems and taxa (Costello et al. 2008). In our case, these were in the form of fishing concessions (an example of territorial use rights in fisheries—TURF). Although results from the literature show mixed results in terms of their performance (Franco-Meléndez et al. 2021; Nguyen Thi Quynh et al. 2017), our findings suggest that the presence of concessions is associated with a greater probability of healthy and recovering stock when compared to their absence (Figure 6A). Considering that benthic high value taxa are found to be in the worst condition (Figures S19 and S20), expanding this management tool to include more taxa and/or cover a greater number of stocks might further improve the sustainable use of such resources. Furthermore, since concessions in our study were exclusively granted to cooperatives, it remains an open question how the absence of access to this fisheries management tool may influence the fishing practices and strategies of patron-client arrangements. Understanding this disparity could provide additional insights into how the two governance arrangements respond to secure property rights in the context of sustainable resource use.

In terms of patron-client governance arrangements, the social-ecological system fit was manifested in the form of 'marginalisation' and 'environmental productivity'. 'Marginalisation', a multifaceted measure of a local community's level of development, exhibited a negative association with the 'catch sustainability index' (Figure 5). Considering that marginalisation takes into account education levels, income status, and household access to basic services, such as electricity, running water and sewage (de la Vega Estrada et al. 2010), our findings likely indicate the effect of asymmetric relationships of power in fisheries governance. In the presence of often dire living conditions and the absence of economic alternatives, fishers from marginalised communities often lack bargaining power and, as such, are vulnerable to exploitative practices by patrons (Ferrol-Schulte et al. 2014; Nunan et al. 2020). Furthermore, lack of internal monitoring and opportunistic behaviour aimed at maximising catch are dominant characteristics of patron-client relationships, as evidenced from other parts of Mexico (Basurto et al. 2020). However, patrons have also been recognised for acting as a source of physical, financial, and even social capital to their clients, thus decreasing their vulnerability, at least in the short term (Ferrol-Schulte et al. 2014; Miñarro et al. 2016). Given that previous studies showed that patrons often prioritise income generation by engaging in exploitative practices (Johnson 2010; Miñarro et al. 2016), the positive relationship between 'environmental productivity' and sustainable resource use that we observed might simply be the result of productive ecosystems' capacity to buffer negative effects associated with such fishing practices.

4.3 | Strengthening Governance Through Better Fit

Beyond the theoretical implications, insights from our study could help inform policy design by fisherfolk, practitioners, and government sectors from the local to national levels to constrain actions associated with negative outcomes and promote those associated with positive ones. For example, emphasis on improving value-added practices from the existing catch has the potential to improve both fishers' wellbeing and resource conservation (Micheli et al. 2014; Venugopal et al. 1995). Such practices have been shown to reduce fishing pressure by ensuring greater returns per unit of catch (Hidayat et al. 2020). However, promotion of value-added practices is not only dependent on adequate infrastructure availability; it also is contingent upon strengthening management and administrative procedures of established governance arrangements to enable successful implementation and performance of such practices (Peñalva Rosales and Solís Tepexpa 2020). Furthermore, although 'migration for fishing' is often cited as a diversification strategy aimed at strengthening adaptive capacity (Gonzalez-Mon et al. 2021), we find that it is associated with less sustainable fisheries. To incentivise a shift away from migration, a greater emphasis should be placed on the provision of secure property rights, such as fishing concessions, in a way that fosters coordination of activities among users of neighbouring concessions (Aceves-Bueno et al. 2023). This would likely require granting concessions for a broader suite of taxa (Table S15) and expanding access to concessions to other forms of governance arrangements besides cooperatives. Initiatives like these may further strengthen individual and community identities by building on a sense of place through explicit recognition of the role fisheries play in the social and cultural fabric of a community (van Putten et al. 2018).

Targeted promotion of collective forms of fishing, such as cooperatives, in marginalised communities would enable creation of alternative forms of engagement with patron-client relationships, particularly in those with existing traditions of working together (Basurto, Bennett, et al. 2013). However, such actions may require sustained support from outside groups in order to overcome often multiple administrative and financial challenges faced by nascent cooperatives (Nenadovic et al. 2018). Furthermore, a place-based focus should not ignore the role that climate change plays in shifts in marine species (Young, Fuller, et al. 2018), particularly considering that both biomass and catch are expected to continue to decrease due to the increasing occurrence of high temperature extremes (Cheung et al. 2021). Given that the increase in environmental variability is positively associated with the increase in sustainable resource use for cooperatives, it becomes especially important to understand how such entities adapt their activities to deal with uncertainty. Further analyses should also go beyond the cooperative and patron-client dichotomy that we analysed in this study and address in greater detail the asymmetric relationships of power embedded across different governance arrangements.

4.4 | Study Limitations

It is important to recognize that our findings are based on evaluating sustainable use of marine species with limited home ranges, specifically benthic and reef-associated taxa. Furthermore, our analysis excluded the industrial fishing sector and the interactions that exist between the two, which is a common reality in many fishery settings throughout the world (Rousseau et al. 2024).

Because of these limitations, our conclusions are specific to SSF contexts and may not hold for more heterogeneous fisheries. Targeting of pelagic or highly mobile species tends to exacerbate the costs and constrain the benefits of conservation efforts, which often results in the classic tragedy of the commons (Epstein et al. 2014). Given the much larger spatial extent of pelagic taxa, it is plausible that in such settings the specific governance arrangement might play a more decisive role in shaping out the resource sustainability outcomes than the particular social-ecological fit.

Regarding the nature of the landings data, we acknowledge that the self-reporting nature of the CONAPESCA landings could have led to an underestimation of actual catch volumes, as has been observed in other fisheries (McCormick et al. 2013). This would inevitably bias the stock parameter estimates and, ultimately, the sustainable catch indices. However, the dataset remains the most comprehensive source of small-scale fisheries information in Mexico and has been used in a number of studies over the years, with recognition of the potential shortcomings of this approach (e.g., Aburto-Oropeza et al. 2008; Erisman et al. 2011; Ferrer et al. 2022; Finkbeiner 2015; Giron-Nava et al. 2019). Some of these studies evaluated its validity. For example, Giron-Nava et al. (2019) demonstrated close alignment between estimates derived from the CONAPESCA dataset and those generated through assessments of fishery-independent data conducted by the Mexican National Fisheries Institute (INAPESCA).

Lastly, our study was exclusively focused on the effects of individual variables. The synergistic effects of more than one variable on social-ecological outcomes, however, deserve further exploration. Thus, while we caution that these limitations must be considered in the interpretation and application of results to other contexts, we maintain that our findings are robust within the scope of the taxa, governance arrangement types, and data constraints considered.

5 | Conclusions

This study expands our understanding of the contextual SES factors that influence the relationships among different governance arrangements and sustainable resource use. Our findings highlight that sustainable resource use can be achieved through multiple governance arrangements provided that they are matched to appropriate social-ecological contexts. By identifying variables that facilitate social-ecological fit, we can avoid governance panaceas and instead focus on strengthening management strategies locally and regionally, at geographic scales where we observe variation in governance arrangements. Such regionalization of fisheries management to social-ecological context requires support from a broad suite of actors, including those in the production and non-governmental sectors and is consistent with the best practices in fisheries management, including ecosystem-based approaches to fisheries management. The variables that we identified as contributing to social-ecological fit in this region could serve as a foundation for similar assessments in other contexts and geographies, thereby supporting a more robust understanding and a more generalizable theory of governance social-ecological fit and its role in the sustainability of fisheries and other natural resources.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

Data related to the principal analysis is included in the Supporting Information. All additional data is available upon request from the main author.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Data S1:** faf70060-sup-0001-supinfo.docx.