Extreme events and coupled socio-ecological systems

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ABSTRACT

Rare, but potentially impactful, extreme events in socio-ecological systems (SES) can trigger significant consequences. The scarcity of theoretical frameworks for such events in SES is due to data limitations and difficulty in parameterizing coupled SES models. We explore the effect of extreme events on coupled socio-ecological systems using two stylized case studies: harvesting of old-growth forests and coral reef fisheries. We found that extreme events alter the long-term and transient dynamics of the systems. We identify counter-intuitive situations where the degradation of forests or coral habitat can prevent extinction through social dynamics feedback. Management outcomes show maximum variability at intermediate disturbance frequencies, complicating predictions of ecological recovery. We also found that initial conditions significantly influence system responses to shocks. Our work lays a foundation for future studies on extreme events in socio-ecological dynamics. Future work could explore more detailed models rooted in the literature, especially related to the modeling of the social dynamics.

1. Introduction

Socio-environmental systems (SES) consist of linked human communities and natural ecosystems embedded within broader social and economic systems, governance regimes, and environmental contexts (Ostrom, 2009). Because of linkages between human and environmental subsystems, SES are potentially vulnerable to unexpected and severe changes in both socioeconomic and environmental conditions. A central question in socio-environmental systems research is predicting how SESs will respond to these shocks, and understanding what characteristics of a particular system, and of a particular shock, determine the system's ability to resist and recover effectively (Schoon and Cox, 2012).

There is no standard definition for an extreme event, but such definitions typically include a notion of how a particular metric falls within the tails of its historic statistical distribution (Aoki et al., 2022). Here, we refer to shocks, extreme events, and disturbances interchangeably, but related terms are also often used in the literature, e.g., black-swan events, catastrophes, or perturbations (Anderson et al., 2017; Aven, 2013; Lande, 1993). There are several different properties that characterize any given shock event, including duration, magnitude, and frequency. For example, Patrick et al. (2022) examined tropical cyclones of differing intensities to assess how they affected various taxa. In addition, the properties of shocks can also change over time. A extreme event can also occur within certain sub-systems of a larger system, i.e. shocks that may be directed at either the social component of the system, the environmental component of the system, or both (Gephart et al., 2016, 2017). Shocks to the environmental attributes of the system, such as a disease or hypoxia event, may affect the availability of given harvested resource, while shocks to the social attributes of the system, such as the COVID-19 pandemic or an economic recession, may affect the magnitude or frequency of harvesting activities (Hughes et al., 2018; Gephart et al., 2017; White et al., 2021b). Collectively, the loss of sources of income and sustenance from harvesting creates important challenges for local communities that may compromise their livelihoods and well-being (Andrews et al., 2021; Colburn et al., 2016). Conversely, an ecological shock, such as a marine heatwave, might affect an ecological system directly, but affect harvesting only indirectly (Cheung and Frölicher, 2020).

Ecological systems are regularly exposed to anthropogenic and natural shocks, such as oil spills, hurricanes, extreme temperatures, and economic recessions (Folke et al., 2005; Bender et al., 2010; McCrea-Strub et al., 2011; Hughes et al., 2018; Smee et al., 2020). Systems with direct harvesting of natural resources, such as forestry or fisheries systems, couple a social system, including harvesters, to the environment via harvesting and ecosystem services (Fig. 1). Inherent linkages and feedbacks in these coupled SES can cause complex dynamics, including oscillations (Oraby et al., 2014; Bauch et al., 2016; Thampi et al., 2018). Each of these interactions plays out across several spatial scales
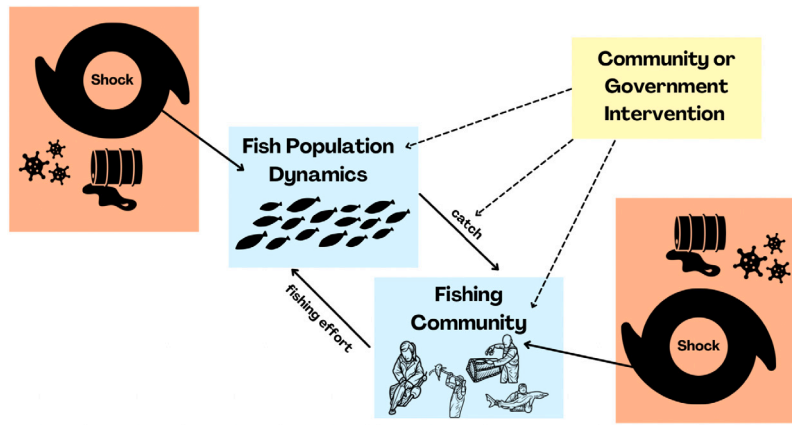

Fig. 1. Conceptual figure illustrating linkages between the ecological (e.g., fish population) and social (e.g., fishers) systems and disturbances.

in the context of an existing, and evolving, legal, social, and economic contexts that can modify human behaviors and consequently the SES as a whole (Refulio-Coronado et al., 2021).

Coral reef fisheries are an ideal case study for understanding shocks and socio-ecological systems more generally. Coral reef fisheries are known for their tight coupling between fishers, reef fish, and the habitat-forming benthic community (Cinner et al., 2013; Gurney et al., 2019; Darling et al., 2019). Coral reefs are also exposed to numerous shock events, including extreme temperatures, hurricanes, oil spills, destructive fishing practices (Fabina et al., 2015; Norstro et al., 2015; Lamb et al., 2016; Hughes et al., 2018). Early theoretical work on modeling coral reefs highlighted the impact of shock events (Mumby et al., 2007). This work highlighted the importance of alternative stable states, and consequently hysteresis, in determining system dynamics after a system shock (Mumby et al., 2007). Blackwood et al. (2012) built on this work by explicitly including parrotfish population dynamics and fishing. Parrotfish are an important grazer of algae, which then prevents the overgrowth of macroalgae on coral reefs. Blackwood et al. (2012) also added the "social" component to these models indirectly by including a term for fishing of parrotfish and found that reducing fishing pressure could allow coral to recover after coral bleaching. More directly, Thampi et al. (2018) focused specifically on how social dynamics affected coral reef systems. They found that the coupled system often acted in unanticipated ways compared to the ecological system alone. This included long boom-bust cycles in coral cover given changes in human opinion. Despite all of these efforts, there has been limited work on the interaction between large shock events and socio-ecological dynamics.

A key aspect in considering how socio-ecological systems might respond to shocks is the role of feedbacks. Related work on generalized socio-ecological modeling has shown that stochasticity can induce longer mean times to extinction; this is in contrast to most ecological literature (Jnawali et al., 2022). However, this work has focused on a non-mechanistic sources of small noise that affect the whole system. Empirical work on how systems may respond to shocks is more mixed. For example, policy responses to disaster impacts are less affected by disaster frequency and magnitude (Nohrstedt et al., 2022). Past work has also shown that conservation organizations may receive more land donations during more difficult economic conditions (Larson et al., 2014). Other work suggests that financial crises can weaken environmental protection (Gaveau et al., 2009; Lekakis and Kousis, 2013). In coral reef systems, past work has suggested that simple income incentives are unlikely to induce behavioral change (Cinner, 2014). Instead, a more holistic understanding of why individuals may alter their behavioral is needed.

In this paper, we examine the effect of different extreme events on socio-ecological systems. As a case study, we build on previously-developed socio-ecological models of old growth forests and Caribbean coral reef fisheries (Fig. 1). These models include explicitly couple together ecological dynamics and human opinions on harvesting and conservation. We use these models to address four specific questions: (1) how does a shock in one part of the socio-ecological system propagate through the rest of the system, (2) how do disturbance characteristics (magnitude, duration) affect the overall system dynamics, (3) how do the initial conditions affect the influence of disturbances on these systems, and (4) how sensitive are the dynamics to changes in the parameter values.

2. Methods

2.1. Old growth forests

We use a stylized model of forest-harvester dynamics developed previously (Bauch et al., 2016). The model explicitly links forest (F) population dynamics with the fraction of the population engaged in conservation (x) as opposed to harvesting.

$$\frac{dF}{dt} = RF(1 - F) - \frac{h(1 - x)F}{F + s} - dF \tag{1}$$

$$\frac{dx}{dt} = kx(1 - x)\left[\phi(2x - 1) + \frac{1}{F + c} - w\right] \tag{2}$$

The forest equation is governed by simple logistic growth, including R as the intrinsic growth rate, h as the harvesting efficiency, and s controlling the shape of the function between harvest rate and the forest population size. As the fraction of non-conservationists ($1 - x$) increases, harvesting also increases and the forest decreases. The fraction of conservationists (x) is the replicator equation (Nowak, 2006) where k is the interaction rate between people, ϕ is the social norm strength, c is the perceived value of the forest, and w is the cost of conservation.

The combined model has seven equilibria compared three for the environmental sub-model and two for the social sub-system. Model dynamics include stable and unstable equilibrium points and limit cycles. The cyclic behavior can occur over long time scales with periods of high and low conservation effort responding to the environment. Depending on the parameter, the model includes transcritical, Hopf, and fold bifurcations. Model outcomes include scenarios where the forest collapses or cases where support for conservation is high enough for forest persistence.

Unlike previous work (Bauch et al., 2016), we also include a term $d(t)$ that explicitly accounts for increased forest mortality as a result of extreme events. In each year of the model, we use a Bernoulli distribution (a coin flip) with two possible outcomes:

$$d(t) = \begin{cases} 0, & \text{if } X = 0 \\ d_M, & \text{if } X = 1 \end{cases} \tag{3}$$

with $X \sim Bern(d_P)$ where d_P is the probability of a disturbance event. Here d_M is the magnitude of the disturbance. Thus, if a disturbance occurs, there is a additional mortality term in the model for one year in the model.

Bauch et al. (2016) fit the above model, without shocks, to old growth forest cover time series in the Pacific Northwest United States and opinion on conservation opinion in the state of Oregon (Bauch et al., 2016). In their supplementary material, they describe this model fitting procedure.

2.1.1. Old growth forest analysis

All model simulations were solved with the deSolve (Soetaert et al., 2010) package in R (R Core Team, 2021). We specifically studied how an ecological shock, with an increase of forest mortality, affected model dynamics. We investigate these types of dynamics, and social shocks, more fully in the coral reef model.

2.2. Caribbean coral reef fisheries

We use the modeling framework developed by Mumby and Hastings 2007, Blackwood et al. 2012, and Thampi et al. 2018. This series of papers went from a coral-macroalgae-turf model to including parrotfish and human behavior. The coupled parrotfish-fisher components of the model are very similar to the old growth forest model above. However, in the coral reef fishery system, humans respond to both coral cover and parrotfish abundance. Thampi et al. (2018) used the following continuous-time model detailing benthic cover of macroalgae (M), coral (C), and turf algae (T) along with parrotfish (P) abundance, and the amount of conservation effort (x) within the community.

$$\frac{dM}{dt} = aMC - \frac{PM}{M+T} + \gamma MT \tag{4}$$

$$\frac{dC}{dt} = rTC - dC - aMC \tag{5}$$

$$\frac{dT}{dt} = \frac{PM}{M+T} - \gamma MT - rTC + dC \tag{6}$$

$$\frac{dP}{dt} = sP(1 - \frac{P}{1-0.5C}) - \sigma P(1-x) \tag{7}$$

$$\frac{dx}{dt} = \kappa x(1-x)(-1 + J(1-C) - \sigma P(1-x) + \phi(2x-1)) \tag{8}$$

In this model, coral, macroalgae, and turf algae compete for benthic cover and the sum of the three groups adds to one. Macroalgae grows over coral at rate a, over turf at rate γ, but is consumed by parrotfish according to the term $\frac{PM}{M+T}$. Corals can overgrow turf algae at rate r, but decrease from natural mortality d and competition a with macroalgae. Turf algae recolonizing area where macroalgae is consumed by parrotfish and where coral is lost naturally, but decreases with competition from macroalgae and coral.

Parrotfish dynamics are governed by logistic growth and fishing effort. The parrotfish population has an intrinsic growth rate of s and a carrying capacity that depends on the amount of coral cover: $1 - 0.5C$. Blackwood et al. 2012 added fishing to this model as a static term at rate σ. Thampi et al. (2018) followed up this work by allowing the fishing term to depend on the number of fishers, or overall fishing pressure, $(1 - x)$ in the population.

We use a very simple model for the social system. The model is the replicator equation where individuals in the population interact with one another and the environmental. The human population here is inclusive of a whole community that is able to make management decisions. The community could be a local group of fishers with certain fishing rights, a village with a locally-managed marine area, or a nation. The fraction of the population engaged in conservation (x) is determined by the interaction rate (κ) between people, the sensitivity of individuals to coral cover (J), the amount of potential parrotfish catch ($\sigma P(1-x)$), and the injunctive social norms term, ϕ. The injunctive

Table 1
Coral reef model parameters with definitions and default parameters. See Thampi et al. 2018 for additional details.

Parameter	Definition	Value	Units
a	Rate of macroalgal overgrowth over corals	0.1	year^{-1}
γ	Rate macroalgal growth over ungrazed algal turfs	0.8	year^{-1}
r	Rate of coral growth over grazed algal turf	1	year^{-1}
d	Coral mortality rate	0.44	year^{-1}
s	Growth rate of parrotfish	0.49	year^{-1}
σ	Maximum parrotfish mortality rate	0.5	year^{-1}
κ	Term to represent human interactions	1.014	year^{-1}
J	Sensitivity of humans to current density of coral cover	1.68	
ϕ	The adjusted strength of injunctive social norms	0.2	
d_p	Probability of a disturbance event	Varies	
m	Magnitude of the disturbance	Varies	

social norms term describes the social pressure that may sway community opinion. Thampi et al. (2018) describes the full derivation of this model, especially the social equation.

The full coupled coral reef model includes a rich set of behavior depending on parameter values. There are cases where corals can be preserved without or with human intervention. There are several scenarios that can also lead to complete coral loss and dominance by macroalgae. In addition, there are smaller regions of parameter space that can lead to bi- or tri-stable behavior and limit cycles similar to the forestry model. For example, high levels of social norm strength can lead to situations of low or high coral cover depending on the initial system state. Another key dynamic is the ratio between parrotfish growth rates and the maximum fishing rate. The social learning rate did not have a large impact on model dynamics compared to other parameter. Sensitivity to coral cover was important, but mostly at higher levels of fishing pressure.

Similar to our approach in the old growth forest model, we include shocks in the above modeling framework through temporary changes in parameter values. For example, as an ecological disturbance, we increased coral mortality (d) during the model simulation. We can control the amount increase in mortality and the frequency of such an increase. Formally, we allow coral mortality to now be a function of time: $d(t)$. Similarly, we defined $j(t)$ as the sensitivity of humans to current density of coral cover at time t. In the original model, these parameters were constant over time. In each year of the model, we use a Bernoulli distribution (a coin flip) with two possible outcomes:

$$d(t) = \begin{cases} d_{\text{baseline}}, & \text{if } X = 0 \\ m_d d_{\text{baseline}}, & \text{if } X = 1 \end{cases} \tag{9}$$

with $X \sim Bern(d_P)$ where d_P is the probability of a disturbance event. Here m is the magnitude of the disturbance. Thus, if a disturbance occurs, there is a additional mortality in the model for one year in the model. We use the same approach for social disturbances. Past work has indicated that social shocks (e.g., financial crisis) can weaken environmental protection (Gaveau et al., 2009; Lekakis and Kousis, 2013).

We specified our model for an idealized coral reef fishery in the Caribbean (Table 1). Past work (Mumby et al., 2007; Blackwood et al., 2012; Thampi et al., 2018) describes the literature and time series data used to parameterize the model.

2.2.1. Coral reef analysis

All model simulations were solved with the deSolve (Soetaert et al., 2010) package in R (R Core Team, 2021). We specifically studied two types of shocks: ecological and social. The ecological shock was represented by an increase in coral mortality, d. This increase in coral mortality could represent coral bleaching or destruction from a hurricane. The increase in the mortality rate is different than simply shifting the system state. For the social shock, we studied decreases
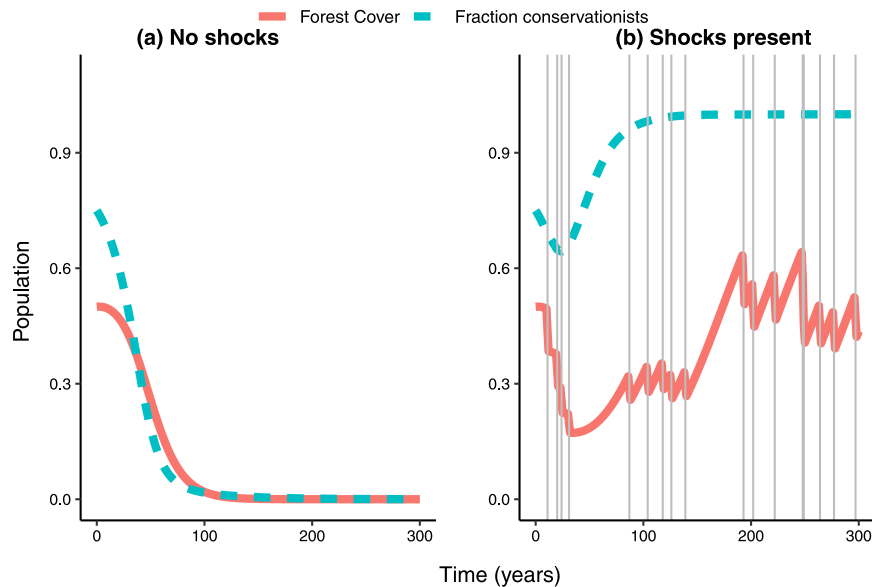
Fig. 2. Old growth forest harvesting model outputs with default parameters ($R = 0.03$, $s = 0.8$, $k = 0.17$, $w = 1.3$, $c = 0.6$, $d = 0.5$, h = 0.075) for a system (a) without or (b) with a moderate level of shocks.

in sensitivity to coral cover, J. This type of shock could represent an economic recession or an extreme event that takes focus away from the dynamics of the reef (see Fig. 1).

3. Results

3.1. Old growth forest harvesting

Without shock events or variability, the original model from Bauch et al. (2016) reaches a simple equilibrium point with high forest cover and high conservation opinion (Fig. 2). However, the dynamics depended heavily on social and harvesting parameters and there were situations with forest persistence or extinction. We found that if there are shocks, in the form of increased forest mortality, the decrease in forest cover can spur an increase in conservation. This feedback then prevents forest extinction and maintains a high fraction of conservation effort in the community (Fig. 2). The dynamic is actually in line with Bauch et al. (2016) as they noted the possibility of limit cycles on long timescales. With shocks, the model was also able to spur conservation support on shorter time scales. As we increased the probability of shock events, we observed increases in the conservation effort (Fig. 3). However, the forest cover peaked at intermediate levels of shock frequencies (Fig. 3). Thus, even at high levels of conservation support, forests could not be maintained in the presence of frequent shock events.

3.2. Caribbean coral reef fisheries

The coupled coral-fisheries model had a rich set of possible behavior. Even without shocks, there were regions of parameter space with coral persistence, macroalgae dominance, or limit cycles. The original model was sensitive to even small changes in the fishing or the parrotfish growth rate (Thampi et al., 2018). In line with our expectations, the mean coral cover through simulations was lower as shock events increased in magnitude or frequency (Fig. 5). For our default parameter values, with no disturbances, coral drops below 1%, our threshold for extinction (Figs. 4, 5). We see that the presence of shocks, in the form of increased coral mortality, can prevent corals from going extinct (Figs. 4, 5). This happens as a result of an increase in conservation following a shock event. Cyclic behavior was possible in the original model, but over longer timescales and for narrow ranges of parameter space. If instead a shock occurs on the social side, in the form
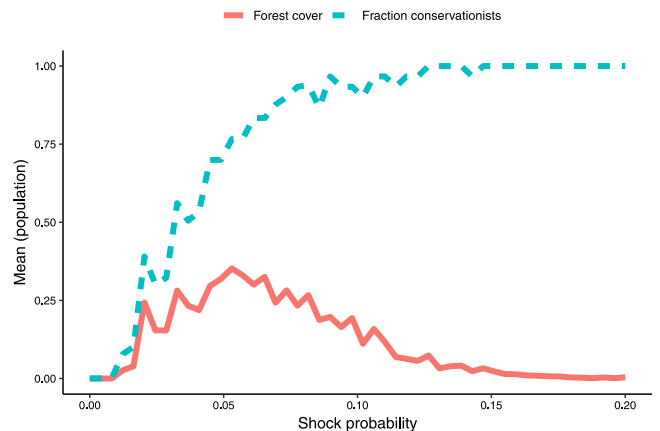

Fig. 3. Mean forest cover and the fraction of conservationists for different probabilities of shocks occurring using default parameters from Fig. 2 across 100 trials for each shock probability level.

of decreased sensitivity in coral cover, both coral cover and time to extinction decreases with higher shock magnitude or frequency. These results are also in line with Thampi et al. (2018) as coral sensitivity and social learning rate were typically less important than fishing pressure or parrotfish growth rates.

We also see that the system initial conditions interacts with a shock event to determine overall system dynamics (Fig. 6). With our default set of parameters, high initial coral cover actually led to cover dropping to zero after a shock event (Fig. 6). This is driven by increase in fishing pressure as a result of the high coral cover. Conversely, coral cover recovered after a shock if the conservation effort was high at the time of the shock event (Fig. 6).

We also see that the variability in both coral cover and time to extinction generally increased with higher shock frequencies and magnitude (Fig. 7). However, variability in coral peaked at an intermediate shock magnitude for direct shocks to coral mortality. Conversely, shocks to the social system led always led to an increase in coral cover variability (Fig. 7). The peak of persistence at immediate shock magnitudes appears to be the interplay of two factors. Without shocks, the corals go to extinct as overfishing leads to a decrease in parrotfish.
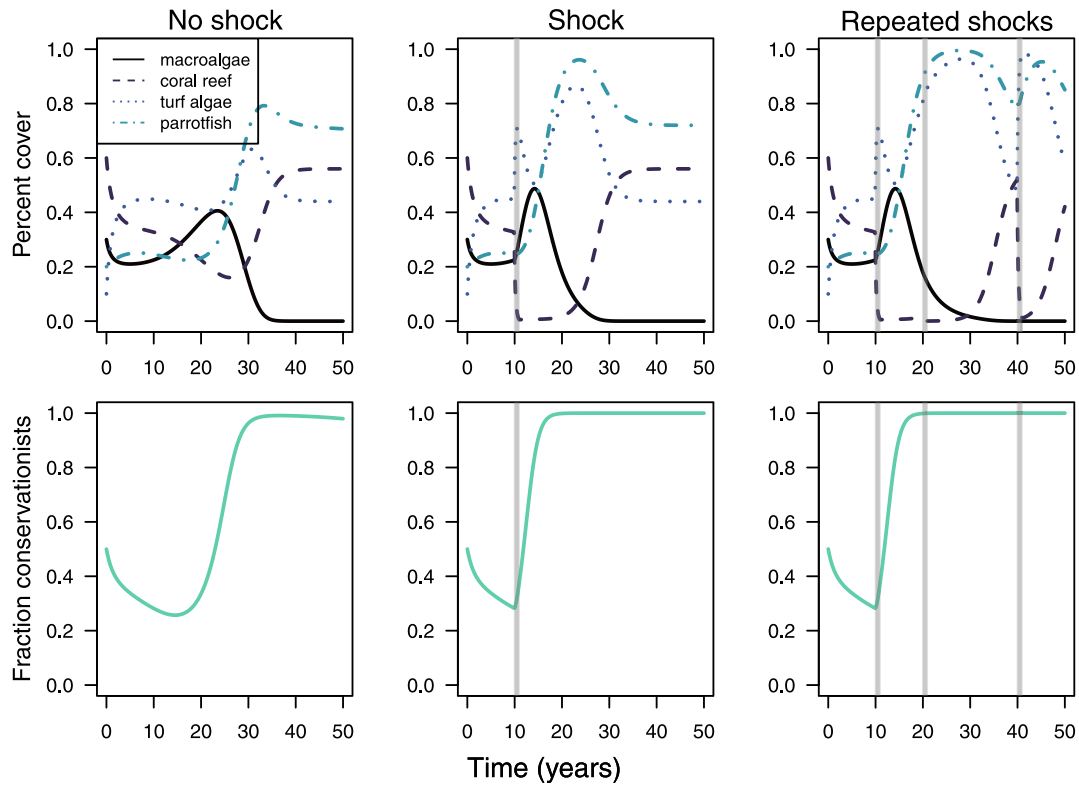
Fig. 4. Time series without (left) and with (right) shock events present. The upper panels represent the ecological system (macroalgae, coral, and turf cover which sums to one and parrotfish abundance) and the bottom panels represent the social system trajectory. Default parameter values: $a = 0.1$, $\gamma = 0.8$, $r = 1.0$, $d = 0.44$, $s = 0.49$, $\kappa = 1.014$, $j = 1.68$, $\sigma = 0.5$, $\phi = 0.2$.
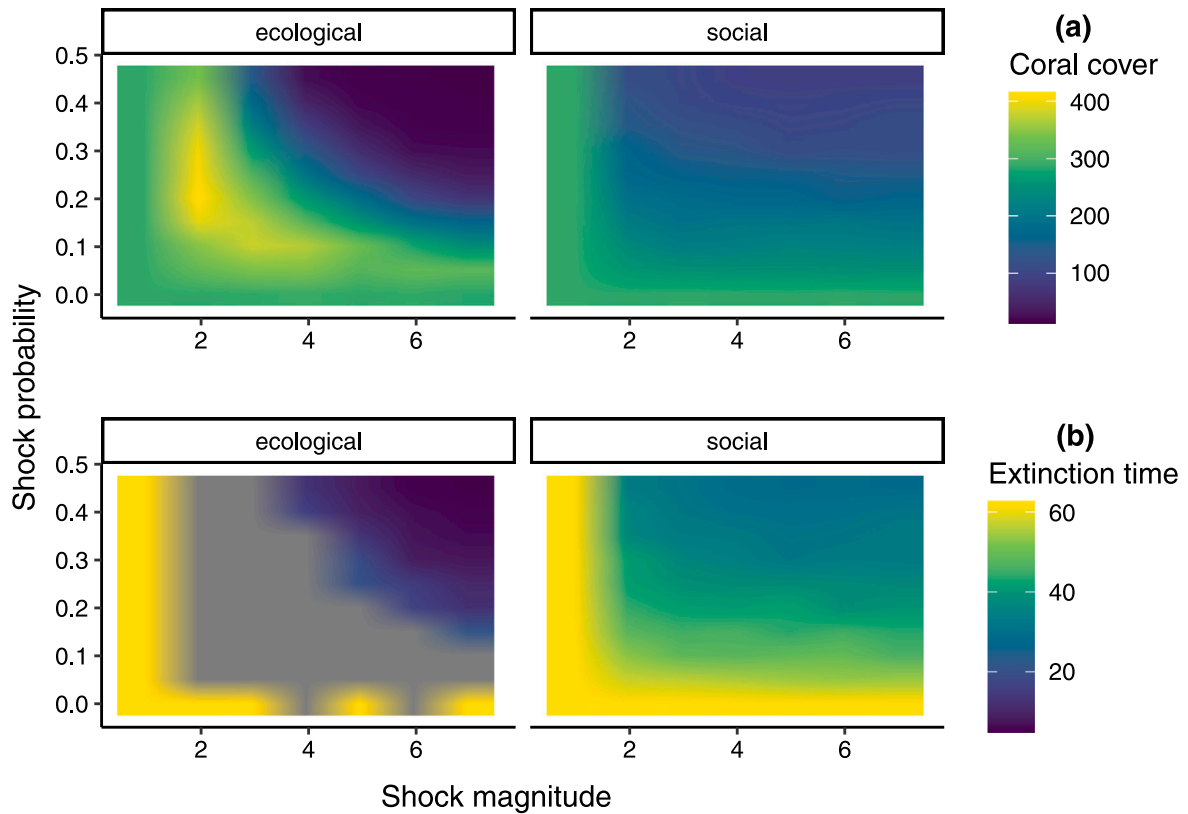

Fig. 5. The relationship between time to coral extinction (when the coral cover drops below a certain threshold) and shock magnitude and frequency for (left) an ecological shock, with an increase in coral mortality, and (right) a social shock, with a decrease in sensitivity to coral cover. The gray areas indicate regions where extinction never occurred.
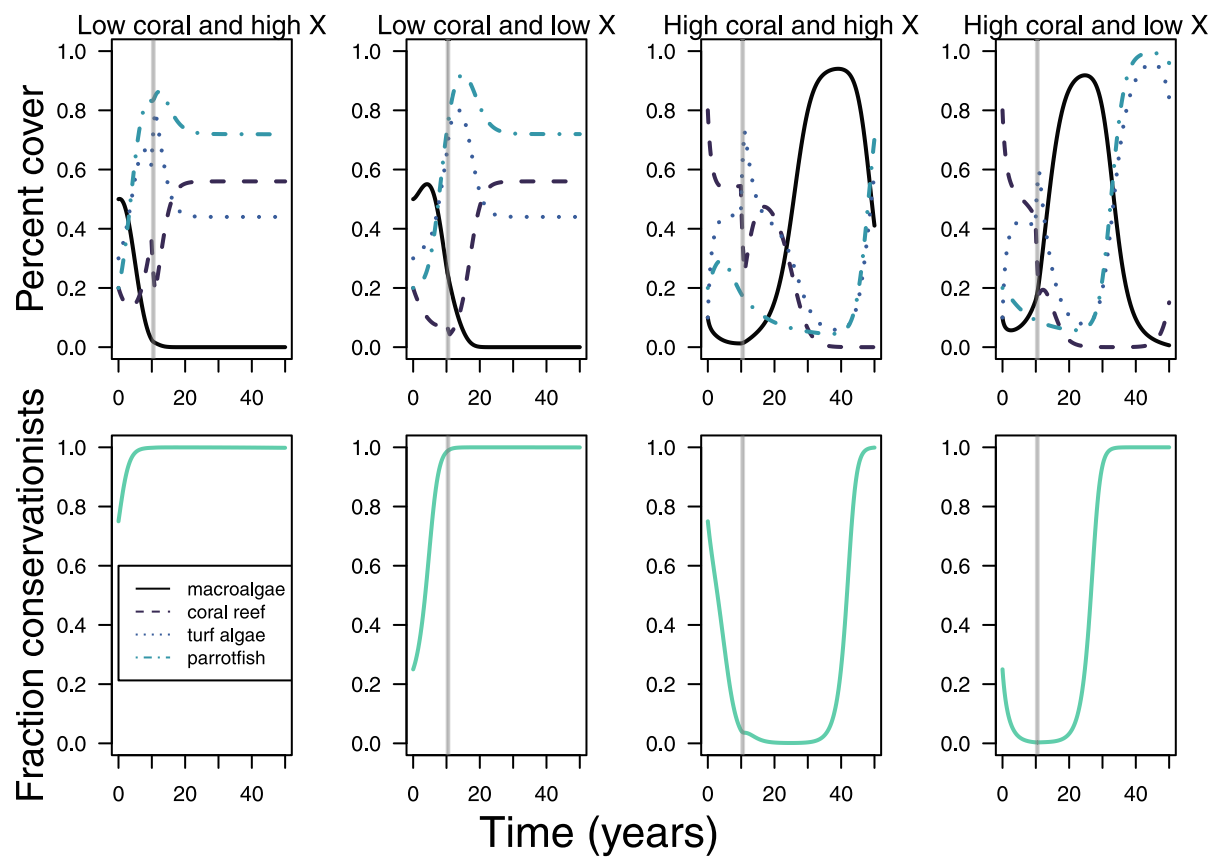
Fig. 6. The effect of different initial conditions, specifically levels of coral cover and the fraction of conservationists, on system dynamics in the presence of an ecological shock.
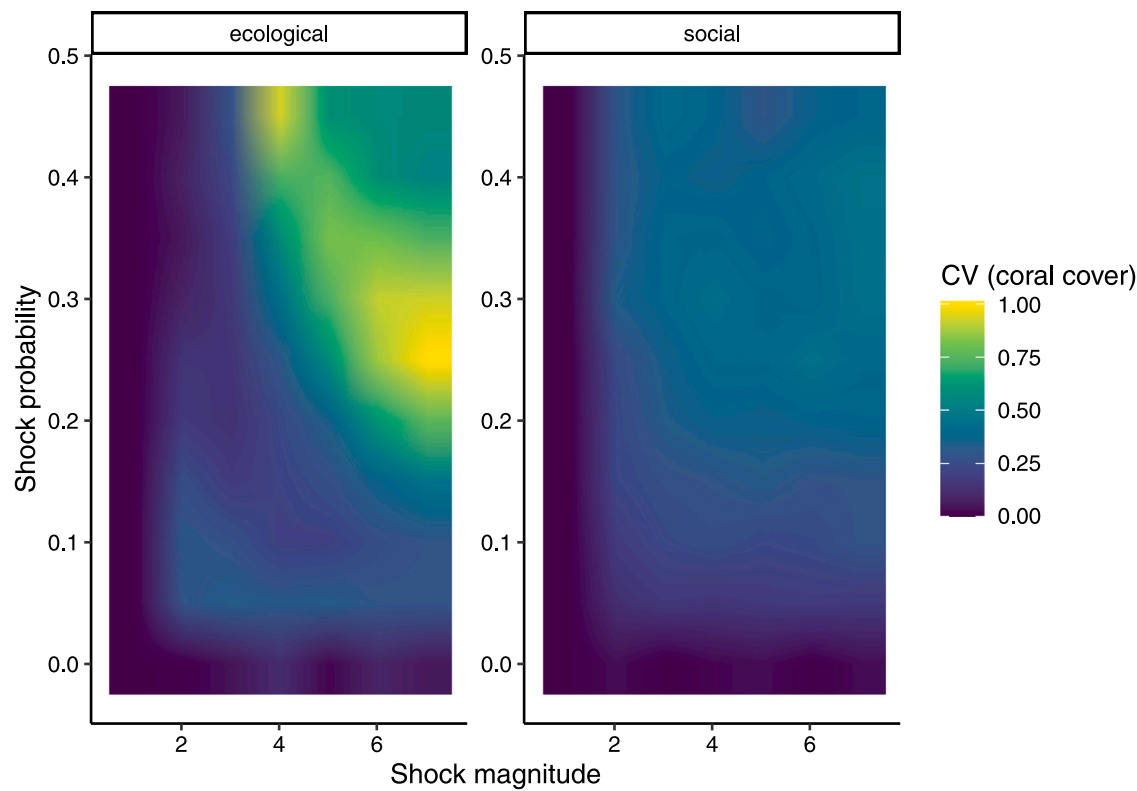
Fig. 7. Coefficient of variation in coral cover for different levels of Shock probability and magnitude for both (left) ecological and (right) social shocks.
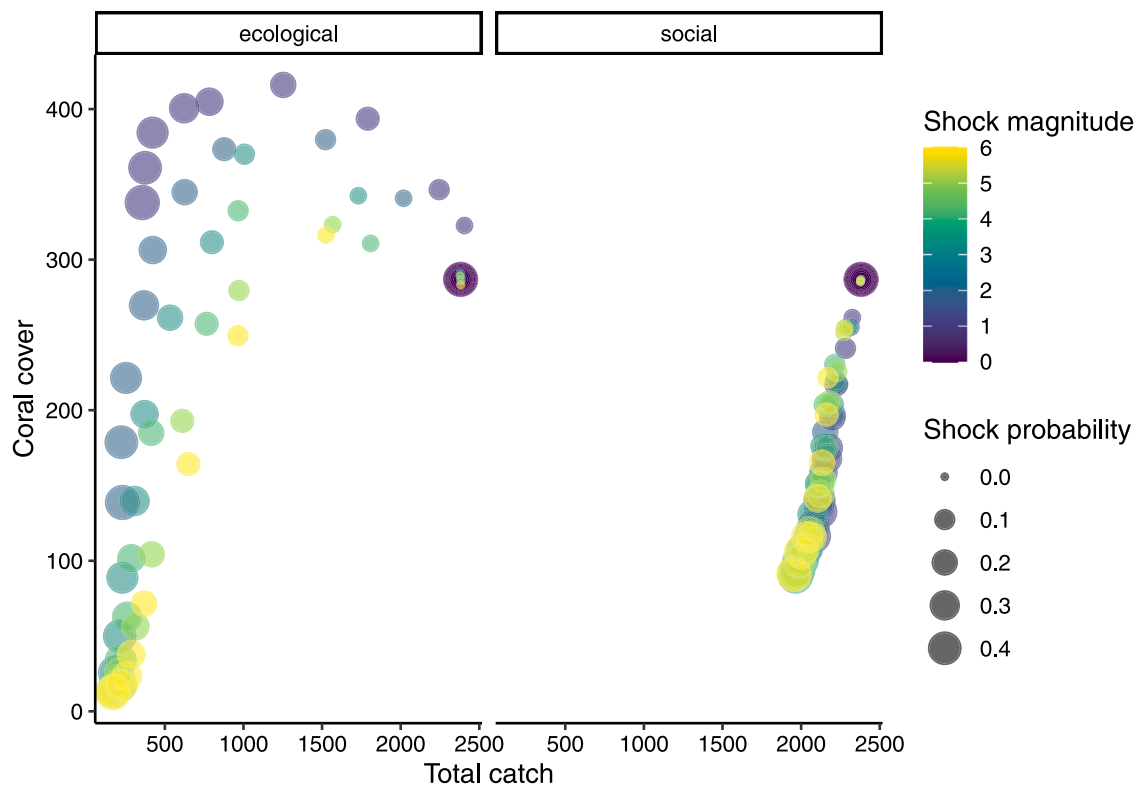
Fig. 8. Coral cover versus total fisheries catch for different types (ecological versus social) shocks and varying shock probabilities and magnitudes.

However, with some level of shocks, coral coverage decreases rapidly, spurring conservation action, which reduces fishing pressure. Frequent shock events do not allow time for recovery regardless of conservation action.

We also examined how disturbances alter potential tradeoffs between conservation and harvesting. Specifically, we compared different regimes of ecological and social shocks to see how each affected overall coral cover and total fisheries catch. In general, higher coral cover meant higher total catch and vice versa (Fig. 8). There was far less variation in both coral cover and catch in terms of the social shock as a short term decrease in sensitivity to coral loss (j) still allowed the coral, and ecological system more generally, to recover quickly (Fig. 8). In addition, there was generally higher coral cover and total catch for cases with lower shock magnitudes and frequencies. However, for the ecological shock (i.e., an increase in coral mortality) had a maximum coral cover at intermediate shock frequencies (Fig. 8) as this region of parameter space spurs conservation action but also allows sufficient time between shocks for recovery.

4. Discussion

We found that the interaction of extreme events and socio-ecological dynamics can produce counter-intuitive outcomes. Specifically, an extreme event that causes an increase in forest or coral mortality would normally cause the forest or coral to decrease with a long time to recover, if at all. However, when we coupled a simple social model with the ecological model, a large decrease in forest or coral cover led to an increase conservation efforts within the community. For the coral system, the increase in conservation effort led to a reduction in fishing pressure allowing the parrotfish population to increase and increase grazing on coral competitors (Fig. 5). Even with a relatively simple model of coral reef fisheries, we show that the system dynamics, especially transient dynamics, can be highly dependent on the current system state, the disturbance regime, and the timescales considered.

Our findings show that accounting for extreme events in understanding and managing socio-ecological systems is important in predicting how systems may resist or recover from such an event. We found that coral cover always decreased with higher disturbance frequencies and magnitudes (Fig. 5). This result is important as past work has indicated an increase in severe coral bleaching events (Hughes et al., 2018). However, we also found that coral persistence, or the time to coral extinction, was higher for intermediate disturbance frequencies and magnitudes (Fig. 5). The dynamics were affected strongly by both shock probability and magnitude. This result is in contrast to Fabina et al. (2015), whom used a coral reef ecosystem model, without social dynamics, and explored the consequences of shocks. They found that shock magnitude was more important than frequency for coral persistence.

We also found that the overall system dynamics, and how the system responded to a disturbance, was highly dependent on initial conditions (Fig. 6). Specifically, we found that coral cover only recovered in systems that had enough conservation support at the time of the disturbance (Fig. 6). In fact, situations with high initial coral cover went extinct because the conservation support eroded quickly and before the shock event (Fig. 6). The sensitivity to initial conditions is in line with past work on socio-ecological dynamics (Bauch et al., 2016; Thampi et al., 2018). Combined, these findings imply that while socio-ecological systems may be able to resist and recover from some extreme events, small changes in their frequency or magnitude may not allow the system to persist. In addition, the current system state interacts with the specific timing of a shock event.

We also found that the temporal variability in coral cover was highest for intermediate levels of disturbance (Fig. 7). We do not know of past empirical work that has showed this same pattern. However, past work has indicated the some fish communities had greatest variability during moderate El Nino events (Possamai et al., 2018). Accounting for variability is important for how we interpret and predict recoveries from extreme events and how we assess management actions (White et al., 2019; Hopf and Wilson White, 2023). For example, if restoration

efforts are in place to grow more corals, the line between success and failure may have a lot to do with the natural variability of the system. Thus, in systems with intermediate levels of disturbance, which is likely most systems, we will see high variance in outcomes of responses to disturbances and management. In addition, we see regions of parameter space at high levels of disturbance where no increase in coral cover or conservation could recover the system (Fig. 5).

We also examined potential trade offs between ecosystem health and harvesting (Fig. 8). Past work in fisheries has indicated there is often, but not always, a tradeoff between conservation and fisheries or if spillover is possible (Di Lorenzo et al., 2020; Chollett et al., 2017). However, other work suggests that optimal management strategies differ in cases when disturbances are included (White et al., 2021a; Aalto et al., 2019; Milne et al., 2022). Generally, we found that when coral cover was high, parrotfish catch was also high (Fig. 8). Both conservation and fisheries generally performed best in situations with low levels of extreme events, but coral cover was maximized at intermediate exposures to bleaching or hurricane events (Figs. 5,7,8). Without shocks, overfishing occurred which resulted in coral extinction. At high levels of shocks, coral was not able to recover on meaningful time scales, even with low fishing. The intermediate levels of shocks drove conservation action while also allowing sufficient recovery time. Thus, a core assumption in our models is a change in human behavior based on the condition of the environment. Specifically, human behavior to switch between more or less harvesting was based on the amount of coral cover and parrotfish abundance. Although individuals can be sensitive to coral health, the feedback in our model with sensitivity to the amount of parrotfish is more in line with empirical work (Diedrich, 2007). In general, more empirical work is needed to fully assess the incentives and motivations for why individuals and communities may alter their behavior (Cinner, 2014). For example, work in Belize has suggested that local perceptions of quality of life was more important for predicting shifts to tourism and conservation from fishing as opposed to coral reef health (Diedrich, 2007). Even within small communities, there is a lot of diversity within groups of fishers (Carter and Garaway, 2014). Combined, this past literature and our results highlight the need for better developed social models of behavioral change for understanding socio-ecological dynamics.

There are a number of important limitations of our study. Our pair of models are relatively simple with a only two or five equations representing the entire socio-ecological system. Future work could include additional species interactions, such as other predators or specific species of benthic cover taxa. We also only include static parameters, ignoring seasonal (White and Hastings, 2020) or long-term changes (García-Carreras and Reuman, 2013) in parameter values. In addition to this structural uncertainty, we also relied on parameter estimates from past work and a generalized systems (Mumby et al., 2007; Bauch et al., 2016; Thampi et al., 2018) that were not designed specific to our set of models. Currently, the social dynamics of the are determined by a single state variable measuring community buy-in, in terms of the fraction of individuals willing to stop harvesting, over time. However, this is extremely limiting given it assumes a binary response of individuals in terms of harvesting. This oversimplifies the real context of most situations. For example, many fisheries involve a small group of people that are actually involved in decision-making (Gurney et al., 2019; Baker-Médard et al., 2021). In addition, there are often gender, or other social stratification, in access to resources (Baker-Médard, 2017; Stacey et al., 2019). Our model includes shocks, but does not contain any other forms of stochasticity, e.g., demographic or environmental variation (Melbourne and Hastings, 2008). In addition, our state variables often reached very low levels of benthic cover where Allee effects could ultimately take over. Past work has shown that shocks can interact with Allee effects to alter management outcomes (White et al., 2020). Our model also only considers a single location with implicit spatial dynamics through benthic competition. We know from past work (White et al., 2020; Milne et al., 2022) that the interaction

between shocks and spatial ecology can be important in studying how systems are able to recover after a shock event. Specifically, shock events that are too frequent or affect an entire system simultaneously will lead to a population collapse. However, spatial patterning in the shock events can allow a system to persist via connectivity. Future work on other socio-ecological systems is also needed to explore the generality of our findings.

5. Conclusions

Shocks can reshape ecological and socio-economic systems, leading to unfavorable states. We found that in coupled socio-ecological systems, counter-intuitive responses may emerge due to internal feedbacks. For example, a temporary increase in forest or coral mortality resulted in more conservation effort, reducing harvesting pressure and promoting coral recovery. Predicting shock effects requires considering the specific ecological and social context preceding the event. Adequate conservation effort can facilitate coral reef recovery post-shock in some circumstances. However, the timing and magnitude of shocks (e.g., coral bleaching) significantly impacts system outcomes. In addition, we found high variability in coral cover for moderately disturbed systems. This variability suggests that management outcomes are likely to be unpredictable for most systems. Future work could incorporate more ecological and social nuances, tailor models to specific systems, and explore how socio-ecological dynamics respond to increased extreme events from climate change.

CRediT authorship contribution statement

Easton R. White: Writing – review & editing, Writing – original draft, Visualization, Software, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization. **Sophie Wulfing:** Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Easton R. White reports financial support was provided by National Science Foundation. If there are other authors they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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